

# **ICES COOPERATIVE RESEARCH REPORT**

**RAPPORT DES RECHERCHES COLLECTIVES**

**NO. 233**

## **REPORT OF THE ICES ADVISORY COMMITTEE ON THE MARINE ENVIRONMENT 1998**

Copenhagen, Denmark  
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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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Mr M. Nicholson <sup>2</sup>		Chairman, WGSAM
Dr J. Rice <sup>2</sup>		Chairman, WGECC

<sup>1</sup>Attended part-time.

<sup>2</sup>Invited participant, attended part-time.

All countries were represented by *Members* except Estonia and Portugal (represented by *Alternate Members*). Representation for Denmark was shared between the *Member* and the *Alternate Member*.

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

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 8–13 June 1998 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 and the Helsinki Commission. This report contains these responses. In addition to responses to direct requests, this report summarizes the deliberations of ACME on topics for which advice was not directly requested but for which the ACME felt that there was information that would be of 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 1998, the ACME continued work on the development of biological effects monitoring programmes. In particular, the design of effective sampling schemes for biological effects monitoring programmes has been considered, and guidelines have been provided for the design of 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 (Section 4.1).

In response to the OSPAR request for ICES to prepare monitoring guidelines for PAHs in sediments and biota, the ACME provided detailed guidelines for monitoring PAHs in sediments in 1997. In this report, guidelines have been prepared for monitoring PAHs in biota, mainly shellfish (Annex 1). However, to accommodate potential measurements in association with PAH-specific biological effects monitoring, the guidelines also include procedures to cover benthic fish.

The ACME reviewed new information on statistical considerations relative to monitoring programmes (Section 4.7). Statistical methods for designing and assessing monitoring programmes are discussed in Section 4.7.1, covering in particular the use of fuzzy set theory in regional assessments of data. In continuation of work begun in 1997, further consideration has been given to the development of harmonized statistical methods for the analysis of temporal trends in atmospheric and riverine inputs of nutrients and contaminants (Section 4.7.3), and to the robustness of methods of trend assessment of inputs data (Section 4.7.2). Finally, information is provided in Section 4.7.5 on variance components relevant to the use of seabird eggs in monitoring contaminants.

In the context of identifying which contaminants can be monitored on a routine basis with adequate interlaboratory comparability, Section 4.5 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.

The ACME reviewed and accepted guidelines for the determination of tributyltin (TBT) and its metabolites in sediments (Annex 2).

#### **Quality Assurance and Intercomparison Exercises**

The ACME reviewed the results of quality assurance-related activities conducted during the past year and has provided summaries of this work in Section 5.

For quality assurance work in relation to biological measurements in the Baltic Sea, ICES endorses the following basic principles: 1) regular ring-tests should be conducted on all biological core and main variables on a five-year basis, with mandatory participation by all laboratories that submit data to the database; 2) training courses on sampling, analytical procedures, and taxonomy should be mandatory; and 3) steps should be taken towards taxonomic certification for experts in national laboratories and private companies (Section 5.1). For the OSPAR area, ICES recommends the production of in-house laboratory quality assurance manuals as an important part of the strategy for the implementation of quality assurance programmes for biological measurements in relation to eutrophication effects (Section 5.2). The production of 'Standard Operating Procedures' for all relevant field and laboratory biological methods is an essential part of this manual production. Section 5.3 describes the aims and work programme of the newly EU-funded programme Biological Effects Quality Assurance in Monitoring Programmes (BEQUALM).

In terms of chemical measurements, Section 5.5 reports on progress that 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 5.6 contains information on certified reference materials that are available for use when monitoring organic contaminants in marine biota and sediments.

## **Overviews of Contaminants in the Marine Environment**

The ACME has considered updated information on mercury in the marine environment (Section 7.1); the detailed overview is contained in Annex 4. Based on this overview, the ACME noted that more information is needed about the inputs of mercury to the marine environment, including a better understanding of processes occurring in estuaries and coastal areas, so that flux estimates to the ocean can be further refined. The ACME also considered a brief review paper on endocrine disruptors in the marine environment (attached as Annex 6) and concluded that there are indications from an increasing number of studies that marine organisms from contaminated areas are exposed to a variety of contaminants with the potential to induce reproductive and non-reproductive disorders via changes in the endocrine system (Section 7.4).

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

### **Concentrations and Effects of Chlorobiphenyls in Marine Mammals**

Available data on concentrations of non-*ortho* and mono-*ortho* chlorobiphenyls have been compiled and presented in Annex 10, along with information relevant to the toxicity of these compounds to marine mammals. Biological effects on marine mammals, including on reproduction and early development, the immune and endocrine systems, and pathological effects are also described in this Annex. The brief summary of this material in Section 12.2 emphasizes the need for more experimental research to improve the knowledge of dose-response relationships for marine mammals exposed to specific compounds, as well as long-term studies on cause-effect relationships to link population-level effects to environmental contaminants.

### **Impact of Fishing Activities on the Age/Size Distribution and Spatial Distribution of Target Fish Populations**

Background information on the main fisheries in the five OSPAR regions is presented in Section 13.1; this information is relevant to the other OSPAR requests concerning fisheries issues. This section also provides the available information required for the five species selected by OSPAR to be covered in this request: cod, herring, sole, mackerel, and hake. For each species and fishing area in each OSPAR region in which it is fished, available information is provided on the overall fishery (pattern of numbers, biomass, landings, and fishing mortality), changes in size distribution and/or age composition, and changes in spatial distribution. The time frame for most of this information is at least twenty years, with trends for many stocks covering thirty years.

### **Incidental Mortality of Marine Mammals owing to Fishing Activities**

Estimated figures on by-catches of several species of marine mammals in different types of fishing gear have been compiled for the five OSPAR regions. A summary of these by-catch estimates is provided in Section 12.1, and the detailed material is contained in Annex 9. Although many of these estimates do not represent the complete by-catch, as data are not available for all fisheries, there is some cause for concern about certain populations of marine mammals impacted in several sea areas.

### **Quantities of Discards of Commercially Exploited Stocks of Fish and Shellfish**

Information available to ICES on discards from fisheries is presented in Section 13.2 for each of the five OSPAR regions. Although efforts were made to prepare as complete a compilation of data as possible, ICES is aware that the information is not complete, either owing to the lack of availability of data to ICES or because data were still preliminary at the time this compilation was being made. Nonetheless, the information presented here provides a picture of the main fisheries with significant discards and the species that are discarded in each of the five OSPAR regions.

### **Changes in Abundance of Individual Non-Target Fish Species owing to Fishing Activities**

Case studies of changes in the abundance of individual non-target fish species are presented in Section 13.3 for each of the five OSPAR regions. Many of the species studied showed considerable fluctuations in abundance over time, but it is rarely possible to relate changes in the abundance of particular species to changes in the fishing regime. However, in certain areas, such as the North Sea and the Irish Sea, declines in certain species of skates and rays appear to be related to fishing.

## **Data Handling**

The annual review of data handling activities relevant to OSPAR requirements by the ICES Environmental Data Centre is contained in Section 15.1 of this report. An important development in early 1998 was the implementation of a web-based inventory of the data held at the ICES Environmental Data Centre. This permits data originators, OSPAR bodies, and other interested persons to check the current holdings of data on contaminants in marine media, biological effects, and fish disease. Section 15.2 summarizes the work of the ICES Oceanographic Data Centre in handling nutrients data relevant to the OSPAR programmes.

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

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

Comments on the draft report, 'Contaminants in Baltic Sea Sediments', are provided in Section 4.3. Further follow-up of this work is proposed to include the convening of a workshop on Baltic Sea sediments to review the results of the Baseline Study of Contaminants in Baltic Sea Sediments, together with the results of more recent studies of Baltic sediments. The results of this workshop will be used to prepare the background for recommendations on the monitoring of sediments in the Baltic Sea, for transmission to HELCOM.

### **Information on topics of general interest**

#### **Fish Diseases**

The ACME has reviewed progress in statistical analyses of disease prevalence data for dab and flounder submitted by ICES Member Countries (Section 8.1). The results of the analyses for temporal trends in estimated disease prevalence in the North Sea and adjacent waters have been compiled in a report that is contained in Annex 8.

Information on the further development of studies of fish liver histopathology as part of biological effects monitoring programmes is presented in Section 8.2, and ICES Member Countries are encouraged to conduct such studies and submit the data to ICES.

#### **The M-74 Syndrome in Baltic Salmon**

An update on the progress in understanding the role of environmental factors in the aetiology of the M-74 syndrome that affects the Baltic stock of Atlantic salmon is contained in Section 8.3. There is a continued need for studies to investigate the occurrence of the M-74 syndrome and similar conditions in wild Atlantic salmon stocks and other fish species in areas other than the Baltic Sea.

#### **Combined Effects of Exposure of Marine Organisms to Contaminants**

The ACME has given initial consideration to the issue of the interactions between individual, and groups of, contaminants in terms of biological responses. An overview on this issue is contained in Section 11.2.

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

In Section 6.1, the ACME has provided information available on the quantities of marine sand and gravel extracted from the coastal regions in the ICES area. In addition, information is provided on the results of research projects to evaluate the effects of extractions in several of the areas where extraction has taken place. The importance of following the ICES Guidelines for Environmental Impact Assessment and the ICES Code of Practice on Commercial Extraction of Marine Sediments is stressed.

#### **Ecosystem Effects of Fishing Activities**

General recommendations deriving from recent studies of the effects of fishing activities on benthos are reported in Section 10.4. In addition, the ACME considered other aspects relevant to the mortality of benthos and degradation of their habitat, and recommended that research projects be supported to develop tools for ecosystem management in the ICES area. In



Section 13.4.2, the ACME reviewed an assessment of the consequences of changes in the populations of benthic-feeding fish in the North Sea during the past thirty years on the level of predation pressure exerted on the benthos.

The ACME has provided initial information concerning measures to evaluate ecosystem effects of fishing activities and the various problems associated with them, including the difficulties of combining data on fish catches using different gears and problems of different metrics and modelling approaches on fish assemblages (Section 13.4.1). Section 13.4.3 provides an initial discussion of reference points that includes ecosystem considerations in relation to the precautionary approach for fisheries management.

### **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 9.1). A number of such transfers have now been documented, and some of the introduced species are causing significant problems in certain areas.

Issues relevant to the transfer of organisms via ships' ballast water and sediments are reviewed in Section 9.2. 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.

### **Issues Related to Mariculture**

Section 14.1 provides some general information on the trends in marine fish production and describes several environmentally friendly initiatives to promote the sustainable development of mariculture. In Section 14.2, further information is provided on the escape of salmon from fish farms in several ICES countries; this is intended to supplement information published in the 1997 ACME report.

### **Discharge of Produced Water by the Offshore Oil and Gas Industry**

The ACME considered a brief paper that estimated the amounts and composition of produced water from the offshore oil and gas industry, and discussed possible impacts on the environment. The ACME expressed concern that there is insufficient scientific evidence to clearly evaluate the long-term sub-lethal effects of produced water in the wider receiving environment and recommended additional studies on specific types of effects (Section 7.6). The full paper is contained in Annex 7.

### **ICES Environmental Report**

On the basis of discussions over the past few years, ICES Working Groups have agreed to contribute to an ICES Environmental Report, which will be updated annually. When fully implemented, this Environmental Report is intended for publication on the ICES website (<http://www.ices.dk>). The first contributions, covering oceanographic conditions, zooplankton monitoring results, and harmful algal blooms, are contained in Section 6.2, with maps of areas of harmful algal blooms presented in Annex 3.

### **Global Programmes**

The ACME considered progress in the development of potential contributions of ICES to the Global Ocean Observing System (GOOS) in Section 16.1, and reviewed recent activities by ICES for the North Atlantic in relation to the Global Ocean Ecosystem Dynamics (GLOBEC) programme.

### Sources of Information Considered by the ACME at its 1998 Meeting

At its 1998 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
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
WGBEC	Working Group on Biological Effects of Contaminants
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
WGABD	ICES/IOC Working Group on Harmful Algal Bloom Dynamics
WGITMO*	Working Group on Introductions and Transfers of Marine Organisms
WGMMHA	Working Group on Marine Mammal Habitats
WGMMPD	Working Group on Marine Mammal Population Dynamics and Trophic Interactions
WGMS	Working Group on Marine Sediments in Relation to Pollution
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
WGZE	Working Group on Zooplankton Ecology

Reports of the following other activities were also considered:

ICES/HELCOM Benthos Taxonomic Workshop (1997)

ICES/HELCOM Workshop/Training Course on Phytoplankton (1997)

Third ICES/GLOBEC Backward Facing Workshop

ICES/GLOBEC North Atlantic Regional Coordination Group

Working Group on Cod and Climate Change

Workshop on Application of Environmental Data in Stock Assessment

Workshop on Prediction and Decadal Scale Ocean Climate Fluctuations

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\*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 pollution, as may be requested by ICES Member Countries, other bodies within ICES, and relevant regulatory Commissions. In handling these 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 Sections 2 and 3, where the individual work items from each Commission are listed and related to relevant sections of the main text.

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

A summary of the progress on the 1998 programme of work requested by the OSPAR Commission is given below, along with reference to the relevant sections and annexes of this report where more detailed information can be found. This summary is provided according to the format of the Work Programme, with the questions on the Work Programme shown in italics and a summary of the ICES advice below in normal print.

### SCIENTIFIC ADVICE

#### 1 MONITORING ACTIVITIES

##### *1.1 To develop monitoring guidelines for polyaromatic hydrocarbons (PAHs), including attention to:*

- a) detection limits in sediments and biota which are in line with relevant assessment criteria;*
- b) the number of replicate samples required per sampling area in order to characterize the sampling area, for both sediments and biota; taking into account general guidelines if established;*
- c) normalization techniques (e.g., grain size, total organic carbon).*

Detailed guidelines for monitoring PAHs in marine sediments were presented in Annex 1 of the 1997 ACME report. Section 4.2 of the present report provides further information on the progress in the development of overall guidelines for monitoring PAHs. In particular, guidelines for monitoring PAHs in biota have been prepared and are attached as Annex 1 to this report. However, the portions of this request concerning the number of replicate samples per area to characterize the sampling area and approaches to normalization of PAH concentrations in sediments have not been completed, owing to the requirement of considerable extra work to obtain scientific agreement on recommendations on these topics. This work will continue in ICES and the results will be reported in due course.

#### 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.*

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 5.2 of this report. Although there were more countries participating in the work of this group during 1998, full participation by representatives of all relevant laboratories contributing to the OSPAR monitoring programme has not been attained. ICES ACME recommends that all laboratories participating in this monitoring programme should participate in the work of SGQAE; they should also prepare in-house laboratory quality assurance manuals, an essential part of which is the preparation of descriptions of Standard Operating Procedures for field and laboratory biological methods.

#### 3 OTHER TOPICS

##### *3.1 To collect information and data on concentrations of non-ortho and mono-ortho CBs in marine mammals and on any relevant biological effects, and to prepare a report on the findings and potential implications.*

A brief summary of the outcome of the ICES work on this topic is contained in Section 12.2 of this report, while a full overview providing available information on concentrations of CBs in marine mammals, as well as biological effects of contaminants, is attached as Annex 9.

#### 4 FISHERIES

##### *4.1 To provide information on the impact of fishing activity on the growth and spatial distribution of the target fish population for commercially exploited stocks of fish and shellfish in the five OSPAR regions that are subject to regular assessment.*

As requested by OSPAR IMPACT, ICES has provided information on the impact of fishing activity on the growth and spatial distribution of stocks of cod, herring, sole, mackerel, and hake in the five OSPAR regions.

This information is contained in Section 13.1 of this report. It should be noted that a great attempt has been made to place information on these fish stocks in the relevant regions, however, certain fish species such as mackerel are highly migratory and move among two or even three OSPAR regions; thus, information from adjoining regions should also be consulted.

In order to present the response to this request in the clearest manner possible, an overall description of the fisheries is provided at the beginning of the treatment of each OSPAR region. This contextual material is equally relevant to the responses to the requests for items 4.3 and 4.4, below, which are contained in Sections 13.2 and 13.3, respectively, of this report.

- 4.2 *To provide information on incidental mortality of marine mammals owing to fishing activities on a species (and gear type) basis for each of the five OSPAR regions.*

Summary information, based on the most reliable data on by-catches of marine mammals in commercial fishing operations, is contained in Section 12.1 of this report. The full details on which this summary table is based are contained in Annex 8. This annex provides available information on estimated by-catches of various species of marine mammals according to fishing gear type for each of the five OSPAR regions. For several of these regions, however, very little material is available.

In order to obtain better information on by-catches of marine mammals in fishing operations, appropriate monitoring programmes need to be established.

- 4.3 *To provide information on quantities of discards by gear type for commercially exploited stocks of fish and shellfish in the maritime area subject to regular assessment.*

Section 13.2 of this report contains information on discards of target and non-target species of fish and shellfish in commercial fisheries in the five OSPAR regions. Although great efforts have been made to obtain as complete information as possible, not all information on discards has been made available for use by ICES. More complete information is expected to become available within the next few years as the results of on-going discard monitoring programmes are reported.

- 4.4 *To provide information on changes in abundance of individual species of non-target fish in the maritime area owing to fishing activities.*

Information available on changes in abundance of individual species of non-target fish owing to fishing activities has been provided for the five OSPAR regions in Section 13.3 of this report. As this topic can potentially cover a very large number of species, this information has been provided on the basis of case studies of species for which adequate temporal trend information is available.

## DATA HANDLING IN 1998

- 1 *To carry out data handling activities relating to:*
  - 1.1 *contaminant concentrations in biota and sediments;*
  - 1.2 *measurements of biological effects;*
  - 1.3 *the implementation of the Nutrient Monitoring Programme.*
- 2 *To continue to establish databanks for phyto-benthos, zoobenthos and phytoplankton species.*
- 3 *To provide technical analyses associated with temporal trend assessments at a meeting hosted by ICES at the ICES Secretariat.*

The ICES Secretariat Environmental Data Centre has handled all data submitted in 1997, covering monitoring activities in 1996. A major activity has been the support provided for the temporal trend assessment of data on contaminants in fish and shellfish, which was conducted by the OSPAR Ad Hoc Working Group on Monitoring in February 1998. Further information on this work is contained in Section 15.1 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 information is contained Section 15.2 of this report.

### 3 PROGRESS ON TASKS FOR THE HELSINKI COMMISSION

The present status of work on 1998 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).*

Section 11 of the 1997 ACME report contained detailed information in response to this request. As this evaluation should only be conducted every third year, the next information on this topic will be presented in the year 2000.

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 overall development of quality assurance procedures for biological measurements is summarized in Section 5.1 of this report. One important outcome of this year's work has been the review and adaptation of the general Guidelines for Quality Assurance of Chemical Measurements in the Baltic Sea to apply also to biological measurements. In terms of other relevant work, specific information on the outcome of the 1997 ICES/HELCOM Workshop/Training Course on Phytoplankton is contained in Section 5.1.1. In addition, information on the results of the ICES/HELCOM Benthos Taxonomic

Workshop is contained in Section 5.1.2. These Workshops stress the need for continued attention to the development of taxonomic expertise to support biological monitoring in the Baltic Sea.

In terms of quality assurance of chemical measurements, the ACME approved a minor revision to the general Guidelines for Quality Assurance of Chemical Measurements in the Baltic Sea, that were transmitted to HELCOM in 1997, as well as two Technical Annexes to these Guidelines: 1) a revised Annex H 'Technical notes on the determination of trace metals in sea water', that has been amended to include treatment of mercury, and 2) a new Annex J 'Technical notes on the determination of total mercury in marine biota by cold vapour atomic absorption spectrometry'. Additional technical annexes are under preparation. Information on progress in this work is contained in Section 5.5 of this report.

#### SPECIAL ISSUES

3. *To coordinate the finalization of the overall report of the Baseline Study of Contaminants in Sediments 1993 and prepare recommendations for sediment monitoring in the Baltic Sea.*

Due to circumstances beyond the control of ICES, the final report on the results of this Baseline Study has not yet been completed in a form acceptable for publication. Steps have been agreed by ACME for the completion of the report according to an agreed set of guidelines and comments. Nonetheless, the ACME has prepared some general comments on monitoring contaminants in Baltic Sea sediments; these comments are contained in Section 4.3 of this report.

4. *To develop a biological data reporting format.*

Initial work has begun on the preparation of a biological data reporting format and accompanying data entry program. It is intended that it will be ready for trial by relevant user groups in early 1999.

## 4 MONITORING TECHNIQUES AND GUIDELINES

### 4.1 Biological Effects Monitoring

#### *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 1998 reports of the Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM) and the Joint Meeting of the Working Group on Biological Effects of Contaminants and the Working Group on Statistical Aspects of Environmental Monitoring (JBSAEM), and ACME deliberations.

#### *Status/background information*

Biological effects monitoring has been, to date, somewhat *ad hoc* and comparisons between areas and over time have been difficult. Furthermore, interpreting the data produced in terms of an impact at the population or ecosystem level has not been attempted. The objective in bringing together WGBEC and WGSAEM was to address these problems and to produce a methodology for integrating the suite of measurements made at the individual animal level to an assessment at the population level which could be of use to environmental managers.

The ACME noted the discussions of WGBEC and WGSAEM and considered that progress has been made in three areas:

- 1) design of effective sampling schemes for a single biological effects variable;
- 2) design of effective sampling schemes for a suite of biological effects variables;
- 3) methods for integrating a suite of biological effects variables into a meaningful index of impact.

These issues are discussed in detail in Sections 4.1.1, 4.1.2, and 4.1.3, respectively.

The general principles for designing an effective sampling scheme are:

- 1) to quantify the objectives;
- 2) to identify the statistical population;
- 3) to identify and estimate the relevant components of variance;

- 4) to choose an appropriate sampling strategy and associated estimators/tests;

- 5) to identify the practical and resource constraints.

In order to demonstrate how these principles apply to a real problem, Section 4.1.1 exploits data on ethoxy-resorufin-*O*-deethylase (EROD) measurements to consider two potential objectives: comparing two sites and assessing temporal trends. In Section 4.1.2, this is extended to data on both EROD and DNA adducts in fish livers.

The ACME noted that as the complexity of the problem grows, the availability of data decreases. Hence, Section 4.1.3 presents a theoretical description of how methods for integrating biological effects variables might be addressed.

#### **4.1.1 Design of effective sampling schemes for a single biological effects variable**

This section considers how an effective sampling scheme for a single biological effects variable might be designed for two specific objectives:

- a) to detect a difference in the mean level of a biomarker between two sites;
- b) to detect a temporal trend in a biomarker.

Specifically, EROD measurements have been considered using plausible estimates of variability from various sources. However, the techniques below can be applied to other biomarker measurements, provided that they satisfy the relevant statistical assumptions.

#### **Comparison between two sites**

The problem is to determine the number of fish ( $n$ ) required to detect a given difference ( $\Delta$ ) in the mean EROD activity between two sites.

The framework is as follows:

- Sample fish at random from each site and measure the EROD activity in each fish.
- The log-EROD measurements are Normally distributed, with mean  $\mu$  and standard deviation  $\sigma$  at site 1, and mean  $\mu + \Delta$  and standard deviation  $\tau$  at site 2. Here, the difference in levels of  $\Delta$  on the log-scale means that, on the original scale, the mean level at site 2 is  $\delta = \exp(\Delta)$  times the mean level at site 1.

- We can take  $n_1$  and  $n_2$  fish from sites 1 and 2, respectively, with the constraint that  $n_1 = kn_2$  (where  $k$  is known beforehand).

The problem is then how to choose  $n_1$  and  $n_2$  to detect a difference  $\Delta$  (or equivalently  $\delta$ ) with a given statistical power  $(1 - \beta)$  at a given significance level ( $\alpha$ ) subject to the levels of variability in the data.

Beliaeff and Burgeot (1997) give the classical statistical background and the corresponding formulae that are needed to choose the sample sizes. They provide results for several combinations of the different parameters described above.

To illustrate, suppose that the standard deviations are  $\sigma = 1.17$  and  $\tau = 0.58$ . These correspond, respectively, to very high and high levels of variability observed in EROD measurements from the French biological effects field surveys described in Beliaeff and Burgeot (1997). Note that here variance estimates have been presented on the natural logarithmic scale, whereas in Beliaeff and Burgeot (1997) variances are reported on the base-10 logarithmic scale.

The following table shows the required sample sizes ( $n_1$  and  $n_2$ ) for various combinations of difference between sites ( $\delta$ ), statistical power  $(1 - \beta)$ , significance level ( $\alpha$ ), and ratio between sample sizes ( $k$ ).

$\delta$	$1 - \beta$	$\alpha$	$k$	$n_1$	$n_2$
2	0.8	0.2	1	18	18
3	0.8	0.2	1	8	8
5	0.8	0.2	1	5	5
10	0.8	0.2	1	3	3
3	0.9	0.2	1	10	10
3	0.8	0.05	1	12	12
5	0.8	0.2	2	4	8

The first four values show the effect of increasing  $\delta$  for fixed values of  $1 - \beta$ ,  $\alpha$ , and  $k$ . The next three values show some examples of varying all of them. The following general comments can be made:

- For demonstration, a value of  $\delta = 5$  has been chosen to represent a severe impact, whereas a lower value of  $\delta = 2$  might represent two sites in the same geographical region that have been subject to the same contamination scenario, but might have different localized responses. In a real application, the value of  $\delta$  would be based on biochemical and environmental knowledge.

- A minimum power of 0.8 (80 %) has been considered because, from an environmental point of view, one certainly wants to detect an impact when it exists; note that a minimum power of 80 % corresponds to a maximum probability of 0.2 of failing to detect the impact when it exists.
- The significance level  $\alpha$  has been taken to be greater than the mythical value of 0.05 to give greater power for the same sample size, although, of course, the risk of a type I error is four times greater.

Results for other variance estimates are given in Beliaeff and Burgeot (1997).

### Detection of a temporal trend

The second objective that can be reached is to detect a temporal trend.

To be specific, suppose we are interested in detecting a linear trend in log-EROD levels by an annual monitoring programme where we take  $F$  fish at the same site at the same time each year over  $T$  years. Nicholson *et al.* (1997) give statistical tools to choose  $F$  and  $T$  to detect a particular size of trend, again given the relevant variance estimates, the significance level, and the desired statistical power.

The variance of the annual mean log-EROD can be written:

$$\psi^2 = \sigma_y^2 + \frac{\sigma_w^2}{F}$$

where  $\sigma_w^2$  and  $\sigma_y^2$  are the within- and between-year variances in the log-EROD measurements, respectively. Note that here  $\sigma_w^2$  incorporates both within-year field variability and within-year analytical variability. Similarly,  $\sigma_y^2$  incorporates both between-year field variability and between-year analytical variability. Look-up tables are provided to relate the size of the trend detected with a particular power (at the 5 % level) to  $\psi$ .

To illustrate, plausible estimates of the two variance components are required. Estimates of the within-year variance were obtained from the French EROD surveys. Low, medium, and high levels of  $\sigma_w$  were 0.224, 0.316, and 0.500, respectively. No estimates of the between-year variance were available. Therefore, for demonstration, values corresponding to low, medium, and high levels of between-year variation in mercury levels in fish and mussels (Nicholson *et al.*, 1997) were taken. Further, low within-year and low between-year variances, etc., were assumed to occur together. Results for a zero level of between-year variance (providing a lower limit) are also presented.



The table below shows the number of years ( $T$ ) required to detect a slope  $|b|$  of 0.05 on the log-scale (corresponding to a change of 5 % per year on the original scale) for different standard deviations ( $\sigma_y$  and  $\sigma_w$ ) and different numbers of fish ( $F$ ), with 80 % power ( $\beta = 0.2$ ) and  $\alpha = 0.05$ .

$\sigma_y$	$\sigma_w$	$F$	$T$
0	0.224	5	9
		10	7
		100	5
0.082	0.224	5	10
		10	9
		100	8
0.255	0.316	5	16
		10	15
		100	15
0.516	0.500	5	24
		10	23
		100	23

When the between-year standard deviation is greater than about 0.1, at least ten years are required. Further increasing the number of fish sampled each year has little effect on the required number of years. In the unlikely case where there are no random between-year changes, increasing the number of fish per year has a much greater effect on the required number of years, and seven years are needed to detect the trend for a practical number of fish (ten).

#### 4.1.2 Design of effective sampling schemes for a suite of biological effects variables

Ways of designing monitoring programmes for a single biomarker are discussed above. Here, the more complicated situation where several biomarkers are of interest is considered. In particular, a sampling scheme for two variables (EROD and DNA adducts in liver) is discussed and the following questions need to be considered:

- In how many fish should EROD levels be measured?
- In how many fish should DNA adducts in liver be measured?
- How can the situation be addressed in which multiple measurements can be made on some fish, and only single measurements can be made on others?

Unfortunately, the solution depends on the objective of the specific monitoring programme, and there is no generic answer.

Here the types of things that need to be considered are demonstrated by way of a very simple stylized example.

Suppose that EROD levels and DNA adducts in liver can be measured, and that they are to be integrated in some index that measures the 'health of the environment'. Again, for simplicity, assume that this index is just a linear combination of the mean EROD level and the mean level of DNA adducts (on a suitable scale of measurement):

$$\bar{y} = \lambda \bar{y}_1 + (1 - \lambda) \bar{y}_2 \quad (1)$$

where  $\lambda \in [0, 1]$  gives the relative weight (importance) of the two biomarkers, as best defined by ecotoxicologists and statisticians and managers,  $\bar{y}_1$  is the mean EROD activity, and  $\bar{y}_2$  is the mean level of DNA adducts. Our goal is now to design a programme to estimate  $\bar{y}$  with a suitable level of precision.

It will be assumed that our sampling design consists of taking  $H$  hauls at random in the area of interest, and then taking  $F$  fish from each haul. In the first instance, it will be assumed that EROD levels and DNA adducts can then be measured on each fish.

Assuming that the two measurements (of EROD and DNA adducts) are stochastically independent, the variance of the index is given by

$$\text{Var}(\bar{y}) = \lambda^2 \left( \frac{\sigma_H^2}{H} + \frac{\sigma_F^2}{HF} \right) + (1 - \lambda)^2 \left( \frac{\omega_H^2}{H} + \frac{\omega_F^2}{HF} \right) \quad (2)$$

where  $H$  and  $F$  are, respectively, the number of hauls and the number of fish per haul, and

- $\sigma_H^2$  and  $\omega_H^2$  are the between-haul variances in EROD and DNA measurements;
- $\sigma_F^2$  and  $\omega_F^2$  are the between-fish variances in EROD and DNA measurements.

It is now necessary to choose  $H$  and  $F$  to make  $\text{Var}(\bar{y})$  suitably small.

One way of doing this objectively is to consider the cost associated with each combination of  $H$  and  $F$ . Here, a plausible cost function is:

$$C = C_0 + HC_H + HF(C_1 + C_2) \quad (3)$$

where  $C$  is the total cost,  $C_0$  is a fixed cost related to the use of a vessel,  $C_H$  is the cost per haul, taking into account salaries and equipment depreciation,  $C_1$  is the sample preparation and analytical costs for EROD, and  $C_2$  is the sample preparation and analytical costs for DNA-adduct measurements. The trade-off between variance and cost can now be explored.

For example, suppose the costs are  $C_0 = 25U$ ,  $C_H = 10U$ ,  $C_1 = 0.05U$ ,  $C_2 = 1U$ , where  $U$  is an arbitrary monetary unit. These costs have realistically been estimated in French francs ( $U = 1000FF$ ). Note the much greater cost of making DNA-adduct measurements compared to EROD measurements.

Further, suppose we have the following variance estimates:

$$\sigma_F^2 = 0.13,$$

$$\omega_F^2 = 0.52,$$

$$\sigma_H^2 = 0.13,$$

$$\omega_H^2 = 0.13.$$

The between-fish variances correspond to working with  $\log_e(\text{EROD})$  and  $\log_e(\text{DNA})$  and are based on French biomarker data. They reflect the much greater variability in DNA adducts between fish than is the case for EROD. There were no estimates of between-haul variability, so some values were estimated.

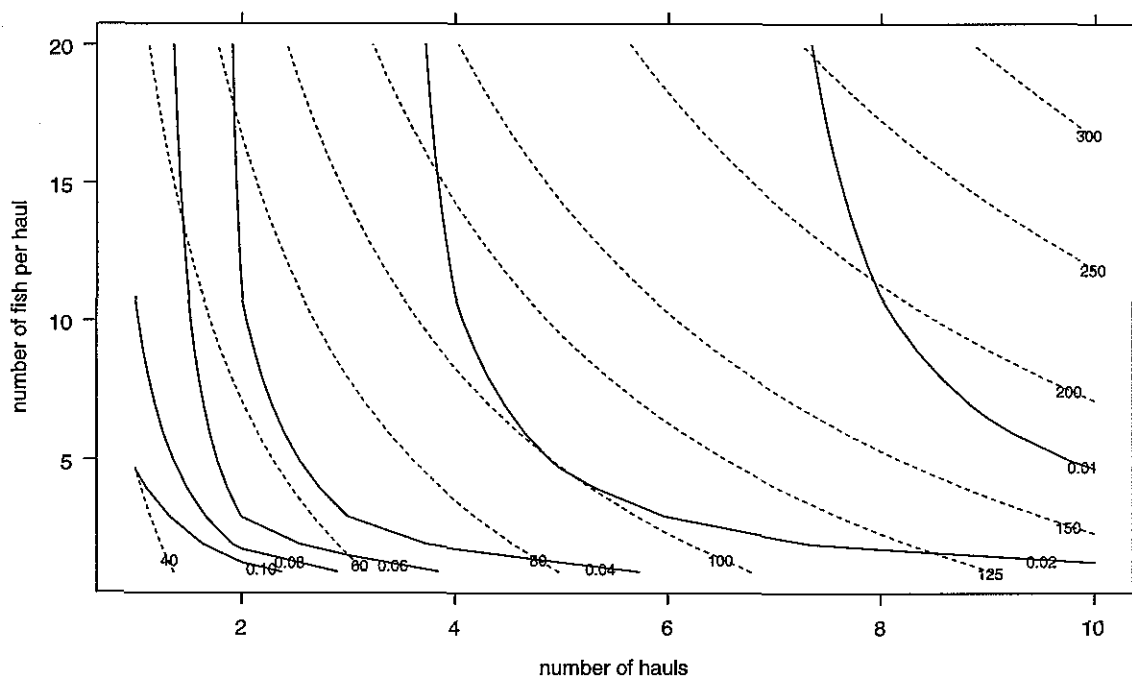
Figure 4.1.2.1 shows the trade-off between cost and variance when  $\lambda = 0.5$ , i.e., equal weight is given to EROD and DNA adducts. The dotted lines show the cost values, whereas the solid lines indicate the variance

values. For example, the dotted line for cost = 100U gives all the combinations of number of hauls and number of fish per haul in the range of the frame leading to this particular cost. The higher the number of hauls, the more precise the mean index estimate, but also the more costly the sampling design. Surprisingly, for this particular example, it is not appropriate to increase the number of fish above five when only two hauls are considered.

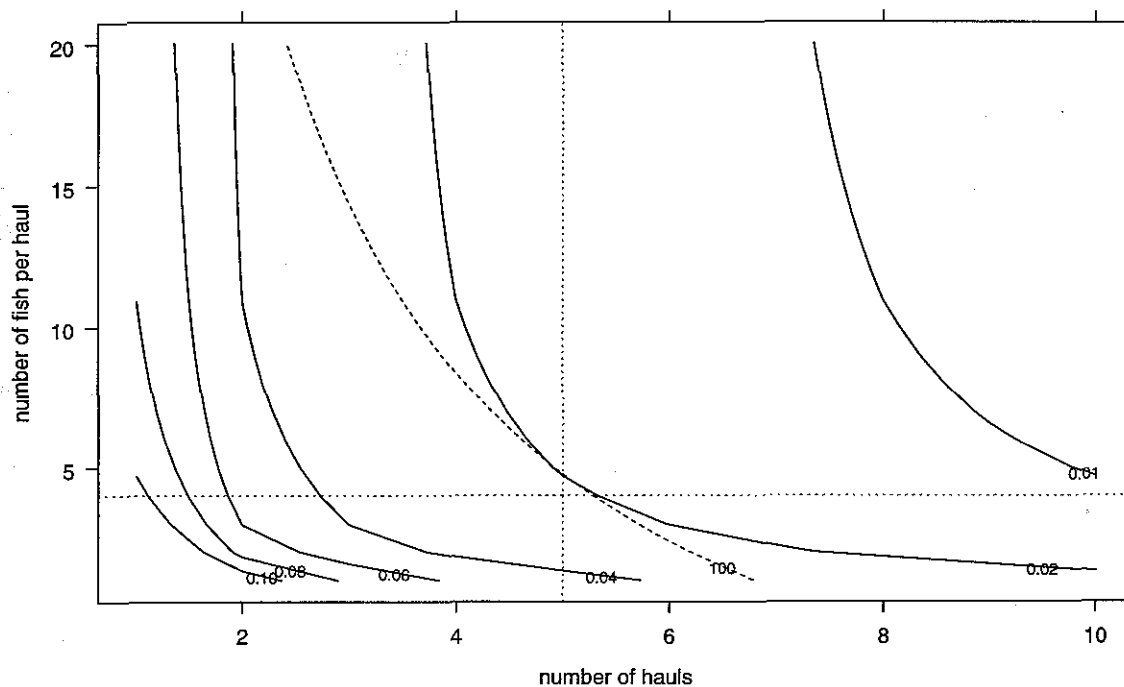
One way of using both cost and variance to find an optimal design is to minimize the variance for a stipulated cost. Suppose we have a total budget of 100U. Figure 4.1.2.2 shows the same plot as before with only the 100U cost contour. Any combination of  $H$  and  $F$  to the left of this contour will satisfy our cost constraints. This contour moves across variance isolines and we consider having an optimal sampling design for the coordinates of the intersection of this cost line with the minimum variance value. Of these values,  $H$  and  $F$  are chosen to give the minimum variance. They turn out to be  $H = 5$ ,  $F = 4$ .

The optimal design depends on the choice of index. Here, if the value of  $\lambda$  is changed, thus giving different weights to the different biomarkers, the optimal design will vary.

**Figure 4.1.2.1.** Contours of variance (solid lines) and cost (dotted lines) for combinations of numbers of hauls and numbers of fish per haul ( $\lambda = 0.5$ , i.e., equal importance is given to EROD and DNA adducts).



**Figure 4.1.2.2.** The same plot as in Figure 4.1.2.1 but focusing on solutions in the region of a cost of 100U.



The table below shows how the number of hauls ( $H$ ) and number of fish per haul ( $F$ ) at a cost of 100U change as the weight given to each biomarker ( $\lambda$ ) runs from 0 (all weight on DNA adducts) to 1 (all weight on EROD).

$\lambda$	$H$	$F$
0.0	4	8
0.1	4	8
0.2	4	8
0.3	4	8
0.4	4	8
0.5	5	4
0.6	5	4
0.7	5	4
0.8	5	4
0.9	5	4
1.0	6	2

Now suppose that EROD and DNA adducts cannot be measured on the same fish.  $F$  fish per haul might now be taken for EROD measurements, and  $G$  fish per haul for DNA adduct measurements. The variance of the index is now

$$\text{Var}(\bar{y}) = \lambda^2 \left( \frac{\sigma_H^2}{H} + \frac{\sigma_F^2}{HF} \right) + (1 - \lambda)^2 \left( \frac{\omega_H^2}{H} + \frac{\omega_G^2}{HG} \right). \quad (4)$$

The cost function now becomes

$$C = C_0 + HC_H + HF(C_1) + HG(C_2). \quad (5)$$

Figure 4.1.2.3 shows how cost and variance depend on  $H$ ,  $F$ , and  $G$  using the same costs and variances as before, and taking  $\lambda = 0.5$ . Again, the dotted lines give the cost values, whereas the solid lines indicate the variance values. Each panel in the figure corresponds to a different value of  $H$  running from 3 to 6.

Supposing again a maximum budget of 100U, the values of  $H$ ,  $F$ , and  $G$  that minimize the variance of our index can be found. They are  $H = 5$ ,  $F = 20$ ,  $G = 4$ .

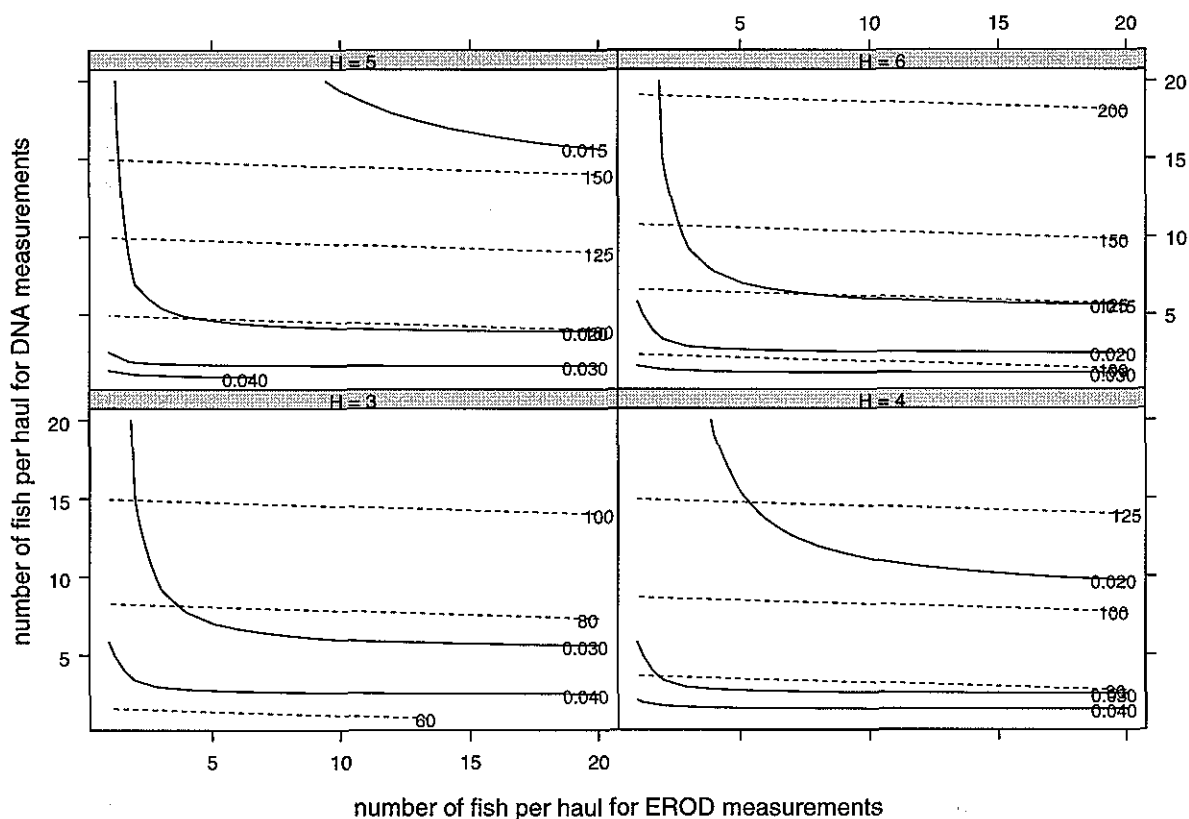
The following table shows how the optimal design with  $H$  hauls,  $F$  fish for EROD, and  $G$  fish for DNA adducts varies with  $\lambda$  (the weight given to EROD).

$\lambda$	$H$	$F$	$G$
0.0	5	0	5
0.1	4	15	8
0.2	4	15	8
0.3	4	15	8
0.4	5	20	4
0.5	5	20	4
0.6	5	20	4
0.7	6	10	2
0.8	6	10	2
0.9	6	30	1
1.0	7	14	0

## Comments

- This example is for demonstration only, but shows what could be done, and identifies what data are needed.
- Just because the minimum variance for a fixed cost has been found, this does not mean that that variance is good enough for the overall objective of the monitoring programme. A more sensible optimization criterion might be to achieve a target variance at minimum cost.
- A graphical presentation is useful for showing the sensitivity of the optimum solution to variation in the sample sizes  $H$  and  $F$ . It might also be informative to consider the effect of varying the assumed values of variances and costs.
- A very simple index has been used here. In a more general and realistic solution, the index might be much more complex, and involve measurements with different statistical distributions. Similarly, the biomarkers may not be independent, as assumed here.
- The index must be specified before an appropriate design can be chosen. In the above example, the weights might be chosen from some modelling exercise. Note how much the optimal design changes as  $\lambda$  varies.
- In practice, costs and variance estimates are needed; these might not always be available, and data generation might be necessary. Alternatively, costs and variance estimates from similar studies could be used.
- Practical constraints must be taken into account, and may over-ride purely statistical considerations.
- A sensible design will only emanate from strong collaboration between statisticians and ecotoxicologists.
- If the monitoring programme has multiple objectives, some compromises between them may be necessary. However, it is important not to compromise so much that none of the objectives are achieved. It is better to achieve little rather than nothing at all.

**Figure 4.1.2.3.** Variance and cost contours for different numbers of hauls, and different samples of fish for EROD and DNA adduct measurements.



#### 4.1.3 Methods for integrating a suite of biological effects variables into a meaningful index of impact

This section gives a theoretical description of possible approaches to integrating a number of biological effects-related variables into an effective management tool. It uses the following example of chemical and biomarker measurements relevant to polycyclic aromatic hydrocarbons (PAHs) and their effects:

- $x_1$  PAH concentrations in sediments,
- $x_2$  PAH metabolites in bile,
- $x_3$  EROD activity in liver,
- $x_4$  PAH-DNA adducts in liver,
- $x_5$  liver histopathology (pre-neoplastic lesions),

and a biological endpoint:

- $x_6$  liver neoplasm.

This series of chemical measurements and biomarkers is progressive in the sense of both the seriousness of the effect, and in the increasing time scale for it to become apparent.

The problem is how to use these measurements to provide some 'weight of evidence' that there is a problem, however this may be defined, e.g., effect, impact, or harm.

One possibility is that the weight of evidence can be quantified in some index, say  $y$ . The statistical problem is to estimate  $y$  from the vector of observed effects  $(x_1, x_2, x_3, x_4, x_5, x_6)$ .

An obvious way to do this would be to establish the relationship between  $y$  and  $(x_1, x_2, x_3, x_4, x_5, x_6)$  from a calibration data set, and then use this relationship to measure the weight of evidence in further samples. Hence, the procedure would be to:

- 1) collect data at a site across several stages of development of the 'problem';
- 2) subjectively assign values of  $y$  using some scoring system;
- 3) fit some function of  $(x_1, x_2, x_3, x_4, x_5, x_6)$  which best predicts  $y$ ;
- 4) use this function to predict  $y$  for some future observation of  $(x_1, x_2, x_3, x_4, x_5, x_6)$ .

If the second step is not possible, a simpler procedure might be to replace  $y$  by a categorical variable with levels such as Low, Medium, and High. If these could be assigned, then a simple discriminant analysis, sometimes in this case called a neural network, would allow future

observations of  $(x_1, x_2, x_3, x_4, x_5, x_6)$  to be allocated to one of these three states.

The difficulty with the weight of evidence approach is the subjective measurement of  $y$  in the calibration data set. An alternative is to let the biological endpoint,  $x_6$ , serve as the weight of evidence. The function of  $(x_1, x_2, x_3, x_4, x_5)$  that best predicts  $x_6$  could then be fitted. In the simple case,  $(x_1, x_2, x_3, x_4, x_5, x_6)$  would have a multivariate Normal distribution and the function is simply the linear regression of  $x_6$  on  $(x_1, x_2, x_3, x_4, x_5)$ . In practice, the relationships are unlikely to be linear and indeed may not even be monotonic.

Further, the biological endpoint  $x_6$  and/or one or more of the variables  $x_1, x_2, x_3, x_4, x_5$  may be a categorical or dichotomous variable. For example, suppose  $x_6$  is measured as presence/absence. We could then model the relationship between  $(x_1, x_2, x_3, x_4, x_5)$  and  $p$ , where  $p$  is the probability that the endpoint is present. Again, a calibration data set would be required and an appropriate relationship fitted for estimation of  $p$  from new data.

The ACME noted another possible approach for use with biological effects data. Suppose we wish to investigate the relationship between various quantities that describe biological effects, up to the occurrence of a severe endpoint such as disease or death. These relationships involve different time scales, as some biomarkers react within hours, while serious endpoint events will require years to develop. This means that the value of an endpoint variable  $y(t)$  may depend on values of a (suspected) explaining variable  $x$  observed at an earlier time  $t - \tau$ , possibly weighted by a function  $w(\tau)$ :

$$y(t) = \sum_{\tau \in D} w(\tau) x(t - \tau), \quad (1)$$

where  $D$  is a set of suitably chosen time lags. It can be extended to other than continuous target variables by embedding equation (1) as a linear predictor in a Generalized Linear Model (GLM):

$$y(t) = f(\sum_{\tau \in D} w(\tau) x(t - \tau)). \quad (2)$$

Furthermore, it can be extended to the case of more than one explanatory variable:

$$y(t) = f(\sum_j \sum_{\tau \in D} w_j(\tau) x_j(t - \tau)), \quad (3)$$

where  $w_j$  is the weight function for  $x_j$ . The weight functions  $w_j(\tau)$  in equation (3) can be evaluated only for those time lags, for which observations exist. Hence, only a finite number of values of  $w_j(\tau)$  can be estimated from data, even if  $w_j$  is thought of as a function defined for every real  $\tau$  in an interval  $[0, \tau_{\max}]$ . The set  $D$  contains a user-chosen set of time lags out of the set of all  $\tau$  values. The estimation itself can be done by standard methods for GLMs. However, in order to facilitate the

interpretation of the  $w_j(\tau)$ , it might be useful to calculate a smooth version  $s(w_j(\tau))$  of them. This seems particularly appropriate if the observed variable  $x_j$  indicates that a process has been initiated, the progress of which is important for  $y(t)$ .

Some precautions have to be observed when using such an approach. Residual values for  $y(t)$  should be inspected in order to obtain information about, for example,

- 1) inappropriate choices of  $D$  (too few elements, elements located at uninformative lag positions),
- 2) dependencies of  $w_j$  on either  $t$  or on values of  $x(t - \tau)$ .

The ACME considered that further development of these ideas is required. There is also a need for appropriate data sets to be collated to support the development of these models.

#### Need for further research or additional data

There is a need for further development and collation of existing data sets. The ACME endorsed the following work programme proposed by the joint meeting of WGBEC and WGBAEM:

#### Objectives

- 1) To develop integrated sampling designs for the entire range of biomarkers and biological effects endpoints relevant to the OSPAR Joint Assessment and Monitoring Programme (JAMP) and other marine monitoring programmes in the ICES area.
- 2) To design modelling tools to assist in the interpretation of environmental biological effects data that will integrate observations on PAHs and their effects in flatfish. The ultimate aim will be to develop generic tools that can be applied to other suites of biological and chemical endpoints (e.g., TBT, metals).

#### Problems to be tackled, for each relevant biological endpoint, under Objective 1

- 1.1 Identify and quantify analytical and environmental sources of variability.
- 1.2 Explore the types and magnitude of changes that can be detected given these levels of variability, particularly in relation to existing sampling guidelines.
- 1.3 Identify where item 1.2, above, indicates that existing ICES and other international guidelines should be modified.
- 1.4 Identify the size of change in chemical concentrations/burdens needed to produce a mean

ingful change in a given biomarker. Similarly, identify the size of change in a biomarker needed to produce a meaningful change in a given biological endpoint. (In practice, this might be rather difficult.)

- 1.5 Propose how to integrate the findings of items 1.1–1.4, above, into a practical holistic monitoring programme, with a corresponding sampling design, that would give maximum efficiency and economy of effort.

#### Problems to be tackled under Objective 2

- 2.1 Initial development of a suitable statistical model (based on discussions at the JBSAEM), using published laboratory data, of the interrelationships between PAH exposure, PAH-related biomarkers and liver neoplastic disease in fish.
- 2.2 Identification of the data sets required for model validation, including Vethaak's mesocosm studies with *Platichthys flesus* (Vethaak *et al.*, 1997), and Myers' field studies with various flatfish in Puget Sound (e.g., Myers *et al.*, 1991).
- 2.3 Data gathering (plus generation of any required data judged to be missing from existing data sets).
- 2.4 Model validation using laboratory, mesocosm and field data. This must take into account the fact that different data sets could refer to different species.

#### References

- Beliaeff, B., and Burgeot, T. 1997. Sampling design optimisation for EROD measurements in fish. *Marine Ecology Progress Series*, 153: 239–246.
- Myers, M.S., Landahl, J.T., Krahn, M.M., and McCain, B.B. 1991. Relationships between hepatic neoplasms and related lesions and exposure to toxic chemicals in marine fish from the U.S. West Coast. *Environmental Health Perspectives*, 90: 7–16.
- 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.
- Vethaak, A.D., Jol, J.G., Meijboom, A., Eggens, M.L., Rheinalt, T.A., Wester, P.W., Zande, T.v.d., Bergman, A., Dankers, N., Ariese, F., Baan, R.A., Everts, J.M., Opperhuizen, A., and Marquenie, J.M. 1997. Skin and liver diseases induced in flounder (*Platichthys flesus*) after long-term exposure to contaminated sediments in large-scale mesocosms. *Environmental Health Perspectives*, 104: 1218–1229.

## 4.2 Monitoring Guidelines for PAHs in Sediments and Biota

### *Request*

Item 1.1 of the 1998 Work Programme from the OSPAR Commission: to develop monitoring guidelines for polyaromatic hydrocarbons (PAHs), including attention to:

- a) detection limits in sediments and biota which are in line with relevant assessment criteria;
- b) the number of replicate samples required per sampling area in order to characterize the sampling area, for both sediments and biota; taking into account general guidelines if established;
- c) normalization techniques (e.g., grain size, total organic carbon).

### *Source of the information presented*

The 1998 reports of the Marine Chemistry Working Group (MCWG), the Working Group on Environmental Assessment and Monitoring Strategies (WGEAMS), and the Working Group on Marine Sediments in Relation to Pollution (WGMS), and ACME deliberations.

### *Status/background information*

In 1997, ICES provided the first part of the response to this OSPAR request (ICES, 1997, Annex 1), in the form of a Technical Annex to the ICES Guidelines for the Use of Sediments in Marine Monitoring (ICES, 1994, 1996) on analytical methods for the determination of polycyclic aromatic hydrocarbons (PAHs) in sediments. This Technical Annex was forwarded to OSPAR for proposed incorporation into the Guidelines for the Joint Assessment and Monitoring Programme (JAMP). Information and recommendations were also provided in Section 4.4.2 of the 1997 ACME report (ICES, 1997) on the choice of organisms for monitoring concentrations of PAHs and their trends, as well as for PAH-related integrated chemical and biological monitoring.

In 1998, this OSPAR request was considered again by three ICES Working Groups which were requested to deal with three outstanding issues, as covered below.

#### **1) Finalization of a Technical Annex on monitoring guidelines for PAHs in biota**

The ACME reviewed a draft Technical Annex on this topic, prepared by MCWG with comments from WGEAMS, and agreed that it met the general requirements and format of existing Technical Annexes to the JAMP Biota Monitoring Guidelines. Accordingly, the ACME concluded that this Technical Annex should be forwarded to OSPAR for proposed incorporation into

the JAMP Guidelines. This document is attached as Annex 1.

The ACME noted that Annex 1 provides guidelines not only for the analysis of PAHs in shellfish, specifically mussels, but also in benthic fish. It is recognized that fish rapidly metabolize PAHs, thereby limiting their usefulness for monitoring temporal or spatial trends of PAHs. However, fish were included because there may be requirements to analyse PAHs in fish muscle tissue for the purposes of food quality, and also because there may be a need to analyse various fish tissues in conjunction with PAH-specific biological effects monitoring. Furthermore, at offshore locations, the collection of appropriate shellfish samples may be problematic if populations are absent, sparse or scattered, and the collection of fish samples may be the only option.

The ACME also noted that MCWG had incorporated into these guidelines (see Section 2.2 of Annex 1) advice based on the results of monitoring following the oil spill from the 'Sea Empress' in Wales, UK in 1996, as the assessment of the shellfish monitoring data had revealed aspects of particular relevance to other PAH monitoring studies. Principal component analysis (PCA) of the PAH data for bivalve molluscs (cockles, mussels and oysters) had revealed the presence of a seasonal cycle in combustion-derived PAH concentrations unrelated to the effects of the oil spill. Concentrations of benzo[a]pyrene, for example, showed minimum levels in summer (August) and maximum levels in winter (March). This seemed to result from a combination of increased inputs during winter, and progressive storage of lipids within the animals during the autumn as they prepared for spawning in the spring. Because PAHs are lipophilic, they would be expected to be stored with the lipids (as the capacity of molluscs to metabolize PAHs is poor) and released with the gametes at spawning. Rapid growth during the post-spawning period would reinforce the decrease in concentrations, leading to the summer minimum. In order to make temporal and spatial comparisons in monitoring studies, therefore, it is of paramount importance that all samples are collected at the same time of year, preferably in the first two months of the year before the animals spawn. The study reported that at two sites in Milford Haven the concentrations of benzo[a]pyrene ranged from essentially zero to 20–50  $\mu\text{g kg}^{-1}$  wet weight—hardly a trivial difference.

#### **2) Number of replicate samples per area to characterize the sampling area when monitoring contaminants**

The ACME reaffirmed that a large impediment to the conduct of effective monitoring programmes has been the continued disagreement about how to design and analyse surveys. This is an issue that goes beyond monitoring PAHs, and applies generally. Indeed, the ACME recognized that further work is needed on the question of the number of replicate samples required per sampling area, for both sediments and biota, as well as on the

variance associated with the measurement of the spatial distribution of contaminants. To examine this broad question, the ACME will request that various tasks concerning this issue, related to the monitoring of both sediments and biota, be given to relevant ICES Working Groups for 1999, with assurance that the necessary interactions will take place between them.

### 3) Normalization techniques for sediments

The ACME noted that, for the normalization of concentrations of PAHs in sediments, the use of organic carbon concentrations is widely accepted and used for the assessment of sediment quality criteria. This approach is based on the assumption that sorption characteristics of organic matter are sufficiently represented by the organic carbon content. However, WGMS has reported that recent evidence indicates that organic carbon is not necessarily a homogeneous normalizer. Further information on this issue is contained in the discussion of the generic issue of normalization in Section 4.6, below.

#### *Recommendations*

ICES recommends that the Guidelines for the Determination of PAHs in Biota, contained in Annex 1 of this report, be forwarded to OSPAR for proposed incorporation into the JAMP Guidelines. It should be noted that these guidelines are of general applicability and can be employed for monitoring in other areas as well, e.g., in the Baltic Sea.

#### **References**

- ICES. 1994. Report of the ICES Advisory Committee on the Marine Environment, 1993. ICES Cooperative Research Report, 198: 45–57.
- ICES. 1996. Report of the ICES Advisory Committee on the Marine Environment, 1996. ICES Cooperative Research Report, 217: 100–104.
- ICES. 1997. Report of the ICES Advisory Committee on the Marine Environment, 1997. ICES Cooperative Research Report, 222: 21–24; 118–124.

### 4.3 Monitoring Contaminants in Baltic Sea Sediments

#### *Request*

Item 3 of the 1998 requests from the Helsinki Commission: to coordinate the finalization of the overall report of the Baseline Study of Contaminants in Sediments 1993 and prepare recommendations for sediment monitoring in the Baltic Sea.

#### *Source of the information presented*

The draft report on the results of the 1993 ICES/HELCOM Baseline Study of Contaminants in Baltic Sea Sediments, the 1998 reports of the Working Group on Environmental Assessment and Monitoring Strategies (WGEAMS) and the Working Group on Marine Sediments in Relation to Pollution (WGMS), and ACME deliberations.

#### *Status/background information*

The ACME reviewed the draft report on the results of the 1993 ICES/HELCOM Baseline Study of Contaminants in Baltic Sea Sediments and concluded that the data collected are unique and valuable and provide important insights into sedimentary processes in the Baltic Sea which must be used to guide the design of future monitoring programmes. The ACME concluded that the most significant results from the programme are concerned with factors which have been shown to most strongly influence the distribution of contaminants, particularly metallic contaminants, in the sediment cores.

The ACME is of the opinion that this Baseline Study has identified a large number of problems and uncertainties regarding the application of sediments for the monitoring of contaminants. These apply to:

- the selection of sampling sites;
- comparable techniques of sampling;
- the interpretation of results.

After examination of the sediment core data, the ACME agreed with the authors of the scientific chapters in the report on the results of the Baseline Study that Baltic basins are strongly influenced by hydrographic conditions (severe mixing events and changing redox potential); therefore, most of these stations are not suitable for monitoring based on core sampling. However, the conclusions from the scientific chapters are not accurately reflected in the Conclusions section (Chapter 2) of the current draft report, and the Recommendations in this draft are not a logical extension of the scientific conclusions. The ACME offers the following discussion of the interpretation of the results and of their significance for future monitoring activity.

The Baseline Study has clearly shown that sediments are extremely active chemically. Post-depositional changes have resulted in the mobilization of metals within the sediment and the formation of a range of authigenic mineral phases. A clear example is the presence of rather large amounts of manganese carbonate and the mobility of manganese and associated trace elements that this implies. In addition to post-depositional mobility, it is possible to identify clear effects on metal concentrations



arising from the hydrochemical conditions in the overlying water. The chemistry of the sediments underlying oxygenated water is very different from that underlying anoxic water. In some cores, the effects of oscillations between oxygenated and anoxic conditions can be identified.

The section on metals in sediments correctly concludes that it is not possible to describe the history of contaminant input to the Baltic Sea from the study of sediment cores, as has been carried out in this Baseline Study. The signal from changes in inputs is completely obscured by other factors, such as the hydrochemistry of the water and post-depositional mobility of metals.

The situation is less clear for organic contaminants, but the Baseline Study has demonstrated that the efficiency of transfer of organic contaminants from water to the sediments has varied systematically, possibly in relation to primary production levels. It is therefore unlikely that any simple and reliable interpretation of the core data in terms of inputs of organic contaminants will be possible.

These observations unavoidably raise the question of how sediment monitoring in the Baltic Sea should move forward. The current Introduction to the report includes a discussion of two alternative strategies that can be adopted for temporal trend studies. These involve the analysis of surface sediments at regular intervals, or the analysis of sediment cores. The Baseline Study has demonstrated that the interpretation of the analyses of sediment cores in terms of temporal (historical) patterns of contaminant inputs to the Baltic Sea is not possible. Therefore, it is necessary to consider the applicability of the analysis of surface sediments and of the objectives that such a programme could address.

Temporal trend studies can be concerned with historical conditions and also with the forward development of current conditions. Surface sediment studies address the latter. Benthic organisms live in contact with surface and near-surface sediments and, therefore, analyses of these sediments are directly relevant to the assessment of the quality of the environment in which organisms are living. The measurement of significant aspects of environmental quality is the main objective of monitoring programmes and provides information on which environmental management is based and from which investigations of environmental problems must start. The regular updating of descriptions of the quality of the sedimentary environment on the seabed is an important objective for marine monitoring.

As has been noted above, the Baseline Study has indicated that the distributions of contaminants in surface sediments appear to be influenced by a wide range of factors, including hydrochemistry, post-depositional mobility, primary productivity, and probably input rates. However, it is not possible at the moment to fully understand how these factors interact to produce the

concentrations that are observed in the sediments. Surface sediment monitoring offers an opportunity to increase our understanding of how these processes operate, and thereby to develop a predictive ability to forecast the likely future quality of seabed sediments or at least to be able to predict the likely effect that major events (such as flushing of bottom waters or raising of the oxic/anoxic boundary in the water column) would have on sediment chemistry. Future work in relation to increasing understanding of the influence of the various factors identified in the Study should build on the scientific conclusions of the Baseline Study, and should include:

'... a more extensive sampling of surficial sediments. The surface sediment sampling locations should be selected to investigate the effects of the various processes that are recognized as having influenced the profiles observed in the 1993 study, i.e., water column chemistry, sediment physico-chemical conditions, primary production, input rates of contaminants, etc. The new data should be combined with the results of other components of the HELCOM programme, such as hydrochemical monitoring, primary production monitoring, and pollution load (input) monitoring. In this way, it may be possible to understand the influence of these processes on sediment chemistry and to move towards a fuller understanding of the meaning of the sediment analyses. The mechanism used to organize the 1993 Baseline Study, through a Steering Group responsible for all aspects of the planning, execution, analyses, and reporting of the study, should be retained, as it has proved to be efficient.' (ICES, 1997)

The ACME noted that this Baseline Study was purely a 'chemistry monitoring exercise'. It was suggested that, in keeping with ICES advice on monitoring strategies, strong consideration should be given to integrating sediment chemical measurements with appropriate biological effects measurements.

A critical aspect of a monitoring programme based on surface sediment sampling will be the design of a sampling programme to address the dual, but compatible, objectives of regularly updating the description of the quality of the benthic environment and investigating the factors controlling that quality. In addition to a broad geographical spread of sampling points, it will be necessary to include stations along carefully selected critical gradients, for example, transects of stations up the sides of basins, crossing the boundary between oxic and anoxic waters, and transects in relation to major inputs.

The ACME emphasized again the importance of the scientific results obtained from this Baseline Study and the need for a good quality report to do justice to the data and the effort that has been expended by scientists in many countries. The results of this study should guide

future studies of Baltic sediments as well as the future policy of the Baltic countries towards the monitoring of sediments within the forthcoming Cooperative Monitoring in the Baltic Marine Environment (COMBINE) programme.

#### *Conclusions/Recommendations*

ICES ACME emphasized that it is essential that the Baseline Study data be made available for the scientific community on diskette with the final report. These data should also be stored at the ICES Environmental Data Centre.

The results of the Baseline Study of Contaminants in Baltic Sea Sediments have revealed a significant amount of uncertainty regarding Baltic Sea sediment monitoring. It is apparent that further discussion is needed among experts to ensure that sediment monitoring in the Baltic Sea is effective. In order to promote a fruitful interchange of knowledge and ideas among relevant experts, the ACME recommends that an ICES/HELCOM Workshop be held to review on a multidisciplinary basis the results of the Sediment Baseline Study as well as results from other Baltic Sea sediment research. The Workshop will be used to generate discussion regarding future Baltic Sea sediment research and monitoring, which should serve as a basis for the development of recommendations concerning sediment monitoring in the framework of HELCOM.

#### **Reference**

ICES. 1997. Report of the ICES Advisory Committee on the Marine Environment, 1997. ICES Cooperative Research Report, 222: 47-48.

#### **4.4 Arctic Monitoring and Assessment Programme: Future Monitoring Developments**

##### *Request*

There was a request from AMAP for cooperation with ICES on specific scientific topics.

##### *Source of the information presented*

The 1998 report of the Working Group on Oceanic Hydrography (WGOH) and ACME deliberations.

##### *Status/background information*

The ACME noted that the Arctic Monitoring and Assessment Programme (AMAP) is currently planning the content of its work programme for 1998-2003, that focuses on data and information acquisition needed for AMAP assessment purposes. In addition to the data and information needed for these assessments, AMAP should

produce long-term data for future assessment work to be implemented beyond the year 2003. It was noted that gaps in scientific knowledge needed to meet these objectives are expected to be filled through a closer collaboration with the relevant international scientific bodies.

The ACME was informed that, based on the Arctic countries Ministers' request, AMAP should now have obligatory core monitoring programmes. These core programmes are important to meet the basic AMAP objectives to continue time series of observations and other activities, which are essential to maintaining the circumpolar character of the Programme. However, it was also agreed that, in addition to the core programme, more flexible monitoring should take place. This includes discrete study units and the Arctic countries should determine which study units they will participate in. The following scientific areas will be included in the second phase of AMAP:

Persistent organic pollutants (POPs)	Levels, trends and effects; new chemicals of concern
Heavy metals	Special focus on mercury; levels, trends and effects
Organometals (TBT)	Special report on levels and effects
Radioactivity	Special focus on impacts on terrestrial ecosystems
Oil and PAHs	Special report on levels and effects
Climate change effects	Special focus on effects on Arctic ecosystems and human life <ul style="list-style-type: none"> <li>• a terrestrial programme</li> <li>• a marine programme</li> </ul>
UV-B effects	Special focus on effects on Arctic ecosystems and human life
Combined effects	Effects of several contaminants, climate change and UV
Human health	Effects of pollutants on life and lifestyle

For climate change, UV/ozone, and combined effects, the ACME was informed that AMAP is interested in developing cooperation with other organizations with relevant ongoing programmes. As ICES already has ongoing work related to climate change in the North Atlantic, the ACME felt that developing cooperative efforts with AMAP in this area could be useful. In this connection, the ACME noted that at least two ICES Working Groups are involved in climate-related problems: the Working Group on Oceanic Hydrography (WGOH) and the Working Group on Cod and Climate Change (WGCCC). WGOH, WGCCC, and possibly other Working Groups, may wish to contribute to a

discussion between ICES and AMAP to develop studies related to climate change and its effects on the ecosystem.

The ACME also noted the special interest of AMAP in the issue of combined effects. The monitoring of combined effects is one of the main challenges for AMAP in the forthcoming period. Accordingly, the ACME agreed that cooperation with AMAP in holding a special workshop on biological/combined effects to develop a strategy for the design and implementation of a combined effects monitoring programme would be a good step forward.

#### **4.5 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 for organizations coordinating international or regional monitoring programmes on contaminants in marine media.

##### *Source of the information presented*

The 1998 report of the Marine Chemistry Working Group (MCWG) and ACME deliberations.

##### *Status/background information*

The MCWG discussed the performance of laboratories which participated in the QUASIMEME 2 Laboratory Performance Scheme (LPS) exercises between June 1996 and December 1997 and decided that this could be used as an indicator of the ability of laboratories to perform routine monitoring. It must, however, be realized that the participating laboratories do not represent the ICES community as a whole, for which comparable material was not available. The QUASIMEME scheme included analyses of chlorobiphenyls (CBs), organochlorine pesticides (OCPs), and trace elements in biota; trace elements, CBs, OCPs, and polycyclic aromatic hydrocarbons (PAHs) in sediments; and trace elements and nutrients in sea water.

In the QUASIMEME 2 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. No indication of such targets has been specified within the guidelines of the regional monitoring programmes of the environmental regulatory commissions.

The MCWG used the information presented on nutrients, trace elements, and organic compounds in slightly different ways when evaluating the performance of the laboratories.

##### Nutrients

During the period mentioned above, a total of six intercomparison sample sets, 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). The intercomparison samples covered both low salinity (estuarine) water and oceanic water. For either group, up to 40–50 laboratories 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 (Table 4.5.1) shows the number of intercomparison rounds where the performance of the group as a whole was satisfactory; this number can be 6 at the most. It indicates that for nitrite, silicate and total oxidized nitrogen, there were only a few problems encountered, although some individual laboratories may have returned unsatisfactory results. Similarly, it indicates that for total phosphorus and total nitrogen, problems were more frequent, although some laboratories may have returned satisfactory results.

The laboratory performance with respect to the nutrients can be regarded as encouraging, as it must be realized that a situation where all laboratories show simultaneously good performance is unlikely to materialize.

**Table 4.5.1.** Summary assessment of laboratory group performance in QUASIMEME nutrient exercises, June 1996–December 1997.

Determinand	Medium	<sup>1</sup> Range of assigned values ( $\mu\text{mol l}^{-1}$ )	<sup>2</sup> Range of $\pm$ Target Bias (%)	<sup>3</sup> Range of between-lab CVs (%)	<sup>4</sup> Range for % No. obs. with $ Z  < 2$	<sup>5</sup> Satisfactory group performances/ total rounds
Ammonia	low salinity water	5.56–24.29	7.03–10.5	11–16	71–92	5/6
Ammonia	saline water	0.90–22.1	7.13–33.88	13–64	34–84	4/6
Nitrite	low salinity water	1.02–6.37	6.39–8.52	5–8	89–94	6/6
Nitrite	saline water	0.42–1.77	7.42–11.97	4–16	80–92	6/6
Phosphate	low salinity water	1.56–6.54	6.38–7.60	4–16	62–92	5/6
Phosphate	saline water	0.05–1.65	7.52–60.59	5–76	67–93	5/6
Silicate	low salinity water	4.83–21.86	6.46–7.15	7–10	78–92	6/6
Silicate	saline water	1.79–17.20	6.58–11.59	7–21	65–89	6/6
Total-N	low salinity water	24.73–65.05	6.38–7.01	8–22	76–88	5/6
Total-N	saline water	8.76–51.60	6.48–7.21	9–23	53–85	3/6
Total-P	low salinity water	1.56–6.62	6.38–7.61	6–21	48–89	4/6
Total-P	saline water	0.20–1.78	7.41–18.76	7–44	63–92	4/6
Total oxN	low salinity water	15.86–36.27	6.69–7.58	5–7	60–89	6/6
Total oxN	saline water	1.19–22.41	7.12–27.06	3–16	83–95	6/6

<sup>1</sup>Range of assigned values for six rounds of the QUASIMEME scheme. The determined assigned values are only indicative.

<sup>2</sup>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.

<sup>3</sup>Range of between-laboratory coefficients of variance (CVs) (%) over six rounds.

<sup>4</sup>Range of the number of laboratories achieving the set QUASIMEME standard of  $|Z| < 2$  (expressed as %).

<sup>5</sup>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  $\times$  2) > 0.

### Trace elements

The QUASIMEME data represent six concentration levels of trace elements in sediments and in tissues of biota (cod muscle, cod liver, mussels). Unfortunately, there were not enough data to evaluate the outcome of the intercomparison exercises on trace elements in sea water.

Table 4.5.2 gives the minimum trace element concentrations in sediments and biota for which the majority of the laboratories achieved  $|Z|$  scores < 2. Generally, most of the laboratories participating in the QUASIMEME LPS do not have problems analysing concentrations down to the values in the table and, therefore, are able to analyse these trace elements in sediments and biota on a routine basis. However, the QUASIMEME results demonstrate that only a limited number of laboratories can be expected to produce comparable data for sandy sediments containing small amounts of trace elements, as special experience is needed to analyse these very low concentrations. For trace element determinations in biota, the laboratory performance for some trace elements seems to be dependent on the tissue type.

### Organic Compounds

QUASIMEME results for exercises on analyses of organic contaminants in biota and sediments are listed in Table 4.5.3. The following points were noted:

- 1) A number of laboratories do not perform to the standard as defined by the QUASIMEME criteria ( $|Z|$  score < 2).
- 2) The performance for CBs in sediments was better than that for CBs in biota.
- 3) The performance for most OCPs was poor.
- 4) The tables were extracted from the QUASIMEME database and, while they should include all of the laboratories submitting monitoring data to ICES, additional laboratories were also included (this is because some laboratories subscribing to QUASIMEME are not involved in analyses for routine monitoring purposes and so do not submit data to ICES).

**Table 4.5.2.** Lowest concentrations of trace elements in sediments and biota which can be monitored on a routine basis by the majority of laboratories (outcome of QUASIMEME LPS, Rounds 6, 8, and 10).

Trace element	Sediments	Biota
Zn	75 mg kg <sup>-1</sup>	≤ 4.6 mg kg <sup>-1</sup>
Cd	340 µg kg <sup>-1</sup>	depends on the tissue: • for cod muscle 5.2 µg kg <sup>-1</sup> • for cod liver 12 µg kg <sup>-1</sup>
Pb	40 mg kg <sup>-1</sup>	problems for the majority of the labs, even at 1 mg kg <sup>-1</sup>
Cu	21 mg kg <sup>-1</sup>	≤ 0.3 mg kg <sup>-1</sup>
Cr	28 mg kg <sup>-1</sup>	problems for the majority of the labs, even at 2 mg kg <sup>-1</sup>
Ni	23 mg kg <sup>-1</sup>	depends on the tissue, for cod liver and cod muscle 0.1 mg kg <sup>-1</sup>
As	6 mg kg <sup>-1</sup>	≤ 1.3 mg kg <sup>-1</sup>
Hg	120 µg kg <sup>-1</sup>	≤ 8 µg kg <sup>-1</sup>
Al	no concentration dependence	
Mn	≤ 850 mg kg <sup>-1</sup>	
Fe	≤ 2.8 %	
Li	≤ 35 mg kg <sup>-1</sup> (5 labs only)	
Sc	≤ 7.6 mg kg <sup>-1</sup>	

'≤' means that only a less-than concentration can be given and not a minimum concentration which the majority of the laboratories is able to analyse; the minimum concentration could not be calculated from the results of the QUASIMEME LPS, Rounds 6, 8, and 10, as the concentrations of these analytes in the samples used were not low enough.

- 1) Different laboratories may have different analytical performance criteria depending on the purpose of their analyses (as more laboratories that are not involved in environmental commission-related monitoring programmes join QUASIMEME, an assessment of the overall performance of laboratories within QUASIMEME may no longer reflect the performance of laboratories engaged in routine monitoring programmes).
- 2) These are difficult analyses and a substantial investment in resources (time and money) is required to produce consistently good quality results.
- 3) The importance of including data from other pertinent sources, such as other intercalibration studies, was recognized, particularly from those schemes involving North American laboratories.
- 4) Information on polychlorinated dibenzodioxins and dibenzofurans (PCDDs/PCDDFs), tributyltin (TBT), toxaphene, non-ortho CBs, and PAHs in shellfish is not currently available. Some information for these parameters derived from the QUASIMEME programme will be available for MCWG in 1999.

#### *Recommendations*

ICES recommends that environmental regulatory commissions that coordinate marine monitoring programmes define targets for the analytical performance

required for laboratories involved in their monitoring programmes, so that information on monitoring capability can be presented with these targets as a reference.

## **4.6 Techniques for Sediment Monitoring**

### **4.6.1 Normalization of contaminant concentrations in sediments**

#### *Request*

This topic is part of continuing ICES work and is also of relevance to item 1.1 of the 1998 Work Programme from the OSPAR Commission.

#### *Source of the information presented*

The 1998 report of the Working Group on Marine Sediments in Relation to Pollution (WGMS) and ACME deliberations.

#### *Status/background information*

The ACME noted that WGMS proposes to revise Technical Annex 2 on Normalization Techniques for Studies on the Spatial Distribution of Contaminants

**Table 4.5.3.** Summary assessment of laboratory group performance in QUASIMEME organic contaminant exercises, June 1996–December 1997.

Determinand	<sup>1</sup> Range of assigned values	<sup>2</sup> Range of $\pm$ Target Bias (%)	<sup>3</sup> Range of between-lab CVs (%)	<sup>4</sup> Range for % No. obs. with $ Z  < 2$	<sup>5</sup> Satisfactory performances/total rounds
<b>CBs in biota</b>					
	$\mu\text{g kg}^{-1}$				
CB28	0.31–12.09	12.91–28.52	44–90	42–79	2/6
CB52	0.53–28.12	12.68–22.01	29–68	40–70	0/6
CB101	1.47–102.01	12.55–15.90	17–55	45–74	2/6
CB105	0.38–40.79	12.62–25.58	33–58	44–65	1/6
CB118	1.05–147.29	12.53–17.25	32–39	52–68	0/6
CB138	2.32–286.91	12.52–14.65	24–36	48–79	1/6
CB153	3.20–391.95	12.51–14.06	27–41	48–68	0/6
CB156	0.15–16.41	12.80–44.90	31–82	53–82	3/6
CB180	0.43–85.93	12.56–24.08	23–45	52–79	4/6
<b>OCPs in biota</b>					
	$\mu\text{g kg}^{-1}$				
HCB	0.07–12.93	12.89–82.43	30–68	48–63	0/3
<i>p,p'</i> -DDE	0.58–160.20	12.53–21.07	27–57	52–71	0/6
$\alpha$ -HCH	0.16–2.58	14.44–43.75	41–114	50–71	0/3
$\gamma$ -HCH	0.07–3.07	14.13–89.42	49–122	48–58	0/2
<i>p,p'</i> -DDD	0.25–46.28	12.61–32.23	22–82	46–75	2/6
<i>p,p'</i> -DDT	0.58–26.28	12.69–21.12	50–114	45	0/1
<i>o,p'</i> -DDT	0.09–17.71	12.78–68.06	103–161	38	0/1
<i>Trans</i> -Nonachlor	0.14–22.27	12.72–48.47	26–94	65–90	4/6
Dieldrin	0.53–48.67	12.60–22.00	47–67	19–67	0/6
<b>Lipids in biota</b>					
	%				
lipid, total	1.14–58.34	12.59–16.90	8–40	64–100	3/4
lipid, extracted	1.07–56.15	12.59–17.15	6–35	72–100	4/4
<b>CBs in sediments</b>					
	$\mu\text{g kg}^{-1}$				
CB28	0.15–2.05	14.94–45.83	30–52	39–94	4/6
CB52	0.19–1.24	15.54–38.54	28–73	54–81	4/6
CB101	0.50–1.98	15.02–22.50	17–47	60–86	2/6
CB105	0.14–1.42	16.01–47.34	25–112	61–100	4/6
CB118	0.36–2.75	14.32–26.43	23–58	63–78	3/6
CB138	1.03–3.73	13.84–17.35	25–43	52–77	3/6
CB153	0.94–3.75	13.83–17.81	22–44	52–83	2/6
CB156	0.10–0.47	23.04–62.50	28–72	47–78	4/5
CB180	0.59–2.89	14.23–20.97	35–75	42–75	1/6
<b>OCPs in sediments</b>					
	$\mu\text{g kg}^{-1}$				
HCB	0.12–1.19	16.71–55.86	24–55	70–89	2/4
<i>p,p'</i> -DDE	0.40–2.49	14.51–24.89	28–65	55–89	4/6
$\alpha$ -HCH	0.08–0.21	36.13–71.68	56–87	80	2/2
$\gamma$ -HCH	0.14–0.54	21.84–42.47	40–119	59–83	4/6
<i>p,p'</i> -DDD	0.49–11.52	12.93–22.70	35–60	40–63	0/6
<i>p,p'</i> -DDT	0.25–3.74	13.84–32.50	48–97	36–73	0/6
<i>o,p'</i> -DDT	0.08–0.44	23.91–76.78	41–122	67–89	2/3
<i>Trans</i> -Nonachlor	0.05–0.19	39.22–123.61	61–115	-	-
dieldrin	0.17–1.58	15.70–41.08	38–86	50–90	2/6
<b>PAHs in sediments</b>					
	$\text{mg kg}^{-1}$				
benz[a]anthracene	0.26–1.18	12.58–12.89	25–36	56–77	2/6
benzo[a]pyrene	0.18–1.17	12.59–13.06	22–37	56–77	2/6
benzo[a]fluoranthene	0.27–1.42	12.57–12.88	28–48	44–68	0/6
benzo[e]pyrene	0.18–1.34	12.54–12.78	16–43	41–88	2/6
benzo[ghi]perylene	0.18–1.22	12.91–15.21	26–51	56–73	1/6
chrysene	0.31–1.48	12.50–12.50	28–47	36–73	0/6
fluoranthene	0.81–2.46	12.54–12.62	17–32	60–82	5/6
indeno[123-cd]pyrene	0.18–1.16	12.93–15.23	28–46	43–70	1/6
phenanthrene	0.49–1.39	12.86–13.52	21–35	56–77	3/6
pyrene	0.55–2.16	12.55–12.68	17–30	68–77	3/6

<sup>1</sup>Range of assigned values for six rounds of the QUASIMEME scheme. The determined assigned values are only indicative.

<sup>2</sup>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.

<sup>3</sup>Range of between-laboratory coefficients of variance (CVs) (%) over six rounds.

<sup>4</sup>Range of the number of laboratories achieving the set QUASIMEME standard of  $|Z| < 2$  (expressed as %).

<sup>5</sup>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  $\times$  2) > 0.

contained in the Guidelines for the Use of Sediments in Marine Monitoring, that ACME adopted in 1993 (ICES, 1994), particularly with regard to its lack of consideration of organic contaminants and in the light of some comments received. Many new approaches have been extensively debated at WGMS meetings since this Technical Annex was first accepted.

Recent information and the following examples have shown that normalization procedures may not apply equally well to all contaminants at all sites. Of particular importance in this respect are trace metals that participate in diagenetic reactions. Care should be taken to normalize in cases where there is a lack of full understanding of the geochemical processes operating. These processes can create important natural enrichments at the sediment surface as a result of the superficial recycling of oxyhydroxides, or deeper in the sediment as a result of co-precipitation of the metals with sulphides (e.g., Gobeil *et al.*, 1997). These natural enrichment processes cannot be accounted for by normalization.

The geochemical normalizers aluminium and lithium behave conservatively as they are not affected by early diagenetic processes and the strong redox effects frequently observed in sediments. However, problems may occur in cases where the sediment is derived from glacial erosion of igneous rocks, with significant amounts of aluminium present in feldspar minerals contributing to the coarse fraction. In such cases, lithium may be preferable (Loring, 1991).

The normalization of concentrations of polycyclic aromatic hydrocarbons (PAHs) in sediments based on the widely accepted and used TOC (total organic carbon) cofactor (ICES, 1996, 1997) was also recently questioned by observations of anomalously high  $K_{oc}$  values measured in pore waters of harbour sediments, which indicate that highly sorptive organic matter can occur. ( $K_{oc}$  is the partition coefficient between the organic carbon concentration in the solid phase and that in the aqueous phase.) Excess  $K_{oc}$  values not supported by ambient natural organic matter would indicate that a decreasing fraction of the PAHs present in the sediment is available for partition with the aqueous phase. A likely explanation is that this anomalous behaviour is caused by the contribution of soot particles with an excessively high sorption capacity per unit of organic carbon. As a consequence, this observation clearly indicates that organic carbon is not necessarily a homogeneous normalizer.

The ACME agreed with WGMS that it may be more appropriate to use combinations of normalizers rather than a single normalizer, such that the advantages of both sieving and geochemical normalizer approaches can be combined. This may also include multiple regression of contaminant and normalizer concentrations in the whole sample and in (sieved) fine sediment fractions (Albrecht, 1993). Multiple regression methods may also use iron

and manganese concentrations (representing the oxyhydroxides) for the further investigation of anomalous results. Statistical regression models have also been developed to correct for the natural metal concentrations in the coarse fraction (Smedes, 1997). The latter approach is applicable with various normalizers, or combinations thereof, and for total as well as sieved fractions. A simple criterion for the applicability of an adequate normalizer is that after normalization of equally polluted sediment samples with very different grain-size distributions, the results should not differ significantly. However, sample sets to test normalization approaches for this criterion are only rarely available. An alternative approach is to take one sample and to produce subsamples with varying grain-size distributions (Smedes, 1997; Smedes *et al.*, 1997). Both the fine and the coarse subsamples are analysed for contaminants and potential normalizers. In that way, a high variability in the normalizer concentration, i.e., a worse case than ever would occur in nature, can be obtained which provides a sensitive test for the usefulness of potential normalizers. Moreover, the overall standard error for the normalization procedure should be calculated from the contribution of the natural variability and the analytical error in both the contaminant as well as the normalizer concentration determinations, which appears to be indispensable in a meaningful data interpretation (Smedes *et al.*, 1997).

#### *Need for further research or additional data*

A comprehensive amount of information on the advantages and disadvantages of normalization procedures has been gathered during recent years, and the ACME feels that there is a strong need to channel that information into a more organized form (e.g., a comprehensive review) that will be of use in the preparation of general or specific normalization guidelines.

#### **References**

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#### 4.6.2 Analysis of TBT in sediments

##### *Request*

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

##### *Source of the information presented*

The 1998 report of the Working Group on Marine Sediments in Relation to Pollution (WGMS) and ACME deliberations.

##### *Status/background information*

In response to a request from the Helsinki Commission in 1995 to provide advice on analytical methods and choice of matrices for the measurement of the presence of organotin compounds in the marine environment, the ACME recommended that sediments especially from coastal areas, including harbours and marinas, and in important shipping lanes, should be analysed for organotins, given that the highest concentrations in the marine environment are expected to be found in the sediments, because organotin compounds are accumulated there and are only very slowly degraded (ICES, 1995). In anaerobic sediments, for example, tributyltin (TBT), which is used in antifouling paints on ships, has a half life of several years. In 1996, the ACME continued its review of several aspects of the TBT issue and recommended the measurement of TBT and its metabolites dibutyltin (DBT) and monobutyltin (MBT), as well as triphenyltin (TPT), in sediments and biota (e.g., bivalves and snails) (ICES, 1996). TBT is metabolized in organisms to DBT and MBT. TBT is the most toxic of these three compounds. TPT is also added to antifouling paints in smaller amounts (in the order of 10 % of TBT) and additionally is used in some countries as a pesticide.

It was also concluded that several analytical methods are available which give good results when properly applied, in particular with respect to TBT, DBT, MBT, and TPT.

The ACME noted that QUASIMEME has in hand a development exercise for the analysis of organotins in standard solutions, water, and biota, and ACME encouraged QUASIMEME to develop a programme for organotins in sediments.

The ACME then reviewed guidelines for the determination of TBT, including its metabolites DBT and MBT, in sediments; these guidelines were prepared by WGMS for inclusion as a Technical Annex to the overall Guidelines for the Use of Sediments in Marine Monitoring (ICES, 1994). After deliberation, the ACME agreed that these guidelines should be included as a Technical Annex to the Sediment Guidelines and, accordingly, attached them as Annex 2 to this report.

In 1996, the ACME also considered whether measurements of triphenyltin (TPT) should be included in monitoring programmes (ICES, 1996). After deliberation, the ACME decided that it would be appropriate to include TPT and optionally its metabolites diphenyltin (DPT) and monophenyltin (MPT) in marine monitoring programmes.

##### *Need for further research or additional data*

The ACME felt that the inclusion of procedures for the measurement of TPT and its metabolites DPT and MPT to the Technical Annex on the determination of TBT would be appropriate.

##### *Recommendations*

ICES adopts the guidelines for the determination of TBT in sediments, contained in Annex 2 of this report, for inclusion as a Technical Annex to the Guidelines for the Use of Sediments in Marine Monitoring (ICES, 1994). Noting the probable interest of OSPAR and HELCOM in these guidelines, ICES agreed to forward this annex to the Commissions with the recommendation of their possible inclusion, respectively, in the JAMP Guidelines and/or the Baltic Monitoring Programme Manual.

##### **References**

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

ICES. 1995. Report of the ICES Advisory Committee on the Marine Environment, 1995. ICES Cooperative Research Report, 212: 30–31.



ICES. 1996. Report of the ICES Advisory Committee on the Marine Environment, 1996. ICES Cooperative Research Report, 217: 31–34.

#### 4.7 Statistical Aspects of Monitoring

##### 4.7.1 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 trend monitoring data.

###### *Source of the information presented*

The 1998 reports of the Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM) and the Working Group on Environmental Interactions of Mariculture (WGEIM), and ACME deliberations.

###### *Status/background information*

At its 1997 meeting, the ACME discussed fuzzy logic approaches to environmental management, and ways in which they might be applied. The ACME noted discussions by the WGEIM on the value of fuzzy logic in integrated coastal zone management. Several projects were described which aimed to facilitate identification of the interactions between various stakeholders in the coastal zone in order to assist in the development of management strategies which optimize the use of natural resources. The fuzzy logic approach was used to handle both uncertainty in data and inexact reasoning, and to incorporate them into a knowledge-based system. This approach could be used to anticipate future pressures for mariculture development and suggest how to optimize potential developments. The ACME encouraged further development of this approach (described in, e.g., Zadeh, 1965), which could have a wide area of application.

At its 1996 and 1997 meetings, the ACME also discussed how environmental target values could be incorporated into the assessment of monitoring data and the development of more effective monitoring objectives (ICES, 1996, 1997). It was noted that OSPAR had included the use of ecotoxicological assessment criteria (EACs) or, where they did not exist, background reference concentrations (BRCs) in its 1998 temporal trend assessments of data on contaminants in biota. The ACME considered the problem of summarizing the large number of results from the analyses of many contaminant data sets in regional monitoring programmes. One simple way is to construct fuzzy sets based on the probability that the estimated mean concentration in the final year is above the EAC/BRC. The following development of this idea is taken from the WGSAEM report.

#### Classification with Fuzzy Sets

In the context of temporal trend monitoring, consider a group of contaminant time series which we want to classify as either 'contaminated' or 'uncontaminated'. We want to do this for each time series, and for the group as a whole. There are two obstacles to doing this. The first is that any summary statistics constructed from the time series will be subject to sampling and environmental variability. The second is that there may be degrees of contamination, and classification as 'contaminated' or 'uncontaminated' may itself be uncertain. In this case, the sets are fuzzy.

One way of dealing with this uncertainty is to employ a score,  $P$ , between 0 and 100 % indicating the degree of membership of each set. In *Fuzzy Set* jargon,  $P$  is called the membership, and when, e.g.,

$$P_{\text{contaminated}} + P_{\text{uncontaminated}} = 100 \%$$

the memberships are normalized, and simply correspond to the percentage probability of being *Contaminated* or *Uncontaminated*. With normalized memberships, the notation can be simplified to  $P_{\text{contaminated}} = P$ .

#### Incorporating Fuzzy Set Theory into the Regional Assessments

This fuzzy approach can easily be incorporated into the current method for analysing contaminant trend monitoring data (Nicholson *et al.*, 1998) used in regional programmes such as OSPAR and HELCOM. Based on this method, Figure 4.7.1.1 shows the logarithms of the annual median concentrations of  $\gamma$ -HCH ( $\mu\text{g kg}^{-1}$ ) measured in the livers of cod from off the coast of the UK in the western North Sea. Superimposed are:

- 1) a fitted smoother using a seven-year span together with its 95 % point-wise confidence limits;
- 2) a reference line corresponding to an ecotoxicological assessment criterion (EAC) of  $100 \mu\text{g kg}^{-1}$ .

In their 1998 assessment of contaminants in biota, OSPAR extended the standard method of analysis by incorporating either EACs or BRCs. They constructed the ratio of the estimated value in the final year to the EAC or BRC to provide a simple dimensionless index of contamination. Since this ratio takes no account of the precision of the estimated value in the final year, they also constructed the ratio of the upper 95 % confidence limit of the estimated value in the final year to the EAC or BRC.

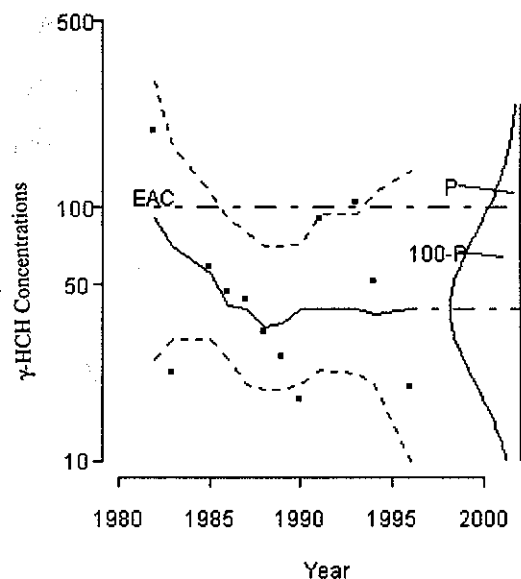
However, together with the fitted value in the final year, the EAC or BRC can be used to define a fuzzy set. Thus, Figure 4.7.1.1 also shows:

- 1) the projected estimate of the mean log concentration in the final year;

- 2) a probability density function indicating the precision of the estimate (corresponding to a Student's *t* distribution, cf. Fryer and Nicholson, 1993), from which *P* can be derived;
- 3) the percentage probability, *P*, that the fitted value in the final year is above the EAC.

Hence, if we define *Contaminated* to indicate that the true mean level in the final year is above the EAC (or BRC), *P* measures the degree of membership of an estimated mean to this set.

**Figure 4.7.1.1.** An application of fuzzy set theory to an analysis of data on  $\gamma$ -HCH in cod liver. The points show the log annual median concentrations of  $\gamma$ -HCH ( $\mu\text{g kg}^{-1}$ ), superimposed by a fitted smoother using a seven-year span with its 95 % point-wise confidence limits (dashed lines). The EAC is shown as a dot-dashed line. To the right of the graph, the lines cut a probability density function from which we derive *P*, which is the probability that at the end of the time series the true mean  $\gamma$ -HCH concentration will be above the EAC.



### Combining Fuzzy Set Memberships

To summarize the memberships from a group of time series, e.g., all data sets in some region, consider the memberships  $P_i$ ,  $i = 1, 2, \dots, n$ , where  $P_i$  is the percentage probability of being *Contaminated* measured in the *i*'th time series. The  $P_i$  are called partial memberships and there are various rules for combining them to measure the overall membership of the group. For example, we might use the arithmetic mean

$$P = \sum_{i=1}^n P_i / n. \quad (1)$$

This rule is symmetric, in that low and high memberships will balance each other. A somewhat pessimistic non-symmetric rule, focusing only on the worst case, is

$$P = \max(P_1, P_2, \dots, P_n). \quad (2)$$

Other rules, including the arithmetic mean, are compensatory, in the sense that low partial memberships can compensate for the high ones. The geometric mean

$$P = (P_1 \times P_2 \times \dots \times P_n)^{1/n} \quad (3)$$

is a simple example of an asymmetric compensatory rule. It is a special case of

$$P = \frac{(P_1^{w_1} \times P_2^{w_2} \times \dots \times P_n^{w_n})^\beta}{[100 - (100 - P_1)^{w_1} \times (100 - P_2)^{w_2} \times \dots \times (100 - P_n)^{w_n}]^\gamma} \quad (4)$$

where the weights  $w_i$  allow for different elements to be given different weight.

The geometric mean may overcompensate for a small number of high partial memberships. In fact, if any single partial membership is zero (indicating very low contamination), the group membership will also be zero, taking no account of any highly contaminated sites. A more environmentally protective rule would be the complementary geometric mean, given by

$$P = 100 - [(100 - P_1) \times (100 - P_2) \times \dots \times (100 - P_n)]^{1/n} \quad (5)$$

and obtained by setting

$$\beta = 0, w_1 = w_2 = \dots = 1/n \text{ and } \gamma = -1$$

in equation 4. With this rule, if any individual membership equals 100 (i.e., highly contaminated), then the group membership will also be 100. Thus, the group will be classified as highly contaminated if any one or more individual data sets are classified as highly contaminated. These rules and alternatives are discussed by Silvert (1997).

### Application to Data from the OSPAR Joint Monitoring Programme: Zinc in Blue Mussels

*P* was calculated using a BRC of  $15,000 \mu\text{g kg}^{-1}$  for 23 data sets on zinc measured in the blue mussel, *Mytilus edulis*. The partial memberships are given in Table 4.7.1.1 together with the regional memberships for six somewhat arbitrary geographical regions. The group memberships were calculated using the complementary geometric mean (equation 5). They give a simple and clear indication of the regional differences in levels of zinc relative to the BRC. All of the regions show a low membership of the set defined as *Contaminated*, except for the Hardanger Fjord, which is an area associated with zinc smelting.

**Table 4.7.1.1.** Summary of partial memberships ( $P_i$  = Probability of exceeding a BRC of 15,000  $\mu\text{g kg}^{-1}$  at the  $i$ 'th site) and regional memberships (complementary geometric mean) for zinc (N.B. 0.0<sup>+</sup> indicates  $0 < P < 0.1$ ).

Region	$P_i$							Regional Membership
English Channel	0.0 <sup>+</sup>	0.0 <sup>+</sup>	0.0 <sup>+</sup>	0.0 <sup>+</sup>	0.0 <sup>+</sup>	0.2	7.8	1.2
Southern North Sea	2.9	7.6						5.2
Kattegat	0.3	19.7						10.5
Oslo Fjord	0.0 <sup>+</sup>	4.6	12.2	13.1	20.4			10.3
Hardanger Fjord	19.8	36.0	37.3	54.5	92.6			60.0
Orkdals Fjord	0.0 <sup>+</sup>	0.3	0.5					0.3

#### *Need for further research or additional data*

The ACME considered that, applied in this simple way, summaries constructed from memberships of a fuzzy set may not offer anything more than the summary statistics employed by OSPAR based on the ratio of the estimated current mean contaminant level to the EAC or BRC. However, fuzzy logic offers more flexibility in the ways that memberships may be combined, e.g., to emphasize environmental protection.

Further, this fuzzy approach lends itself to the integration of both continuous and categorical variables. In this respect, fuzzy logic may provide a useful framework for combining non-contaminant information such as that used to indicate biological effects.

The ACME recommended that further work be done to explore alternative rules and the sensitivity of the complementary geometric mean to individual high partial memberships. Consideration should also be given to the usefulness of this approach to summarizing biological effects data.

#### **References**

- ICES. 1996. Report of the ICES Advisory Committee on the Marine Environment, 1996. ICES Cooperative Research Report, 217: 40–41.
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- Fryer, R.J., and Nicholson, M.D. 1993. Predicting future contaminant levels in relation to EQOs using temporal trend monitoring data. ICES CM 1993/E:18.
- Nicholson, M.D., Fryer, R.J., and Larsen, J.R. 1998. Temporal trend monitoring: Robust method for analysing contaminant trend monitoring data. ICES Techniques in Marine Environmental Sciences, No. 20.

Silvert, W. 1997. Ecological impact classification with fuzzy sets. *Ecological Modelling*, 96: 1–10.

Zadeh, L.T. 1965. Fuzzy sets. *Information and Control*, 8: 338–353.

#### **4.7.2 Robustness of methods of trend assessment**

##### *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 1998 report of the Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM) and ACME deliberations.

##### *Status/background information*

In 1997, the ICES/OSPAR Workshop on the Identification of Statistical Methods for Trend Detection was held to discuss the merits of different statistical methods and the specific difficulties of applying them to time series of contaminant inputs. In particular, the Workshop compared four statistical methods for trend detection in terms of their statistical power to detect three different patterns of change. The methods were:

- 1) linear regression;
- 2) non-parametric Mann-Kendall test;
- 3) mean squared successive difference test;
- 4) LOESS smoothing.

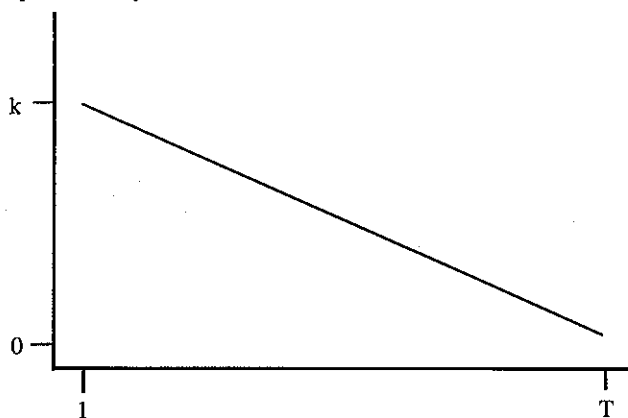
The methods were compared using three stylized scenarios describing changes in inputs (Figures 4.7.2.1.a, b,c):

- Scenario 1:** a linear decrease;
- Scenario 2:** a step decrease in the middle of an otherwise constant series;
- Scenario 3:** a linear increase for the first half of the series followed by an identical linear decrease.

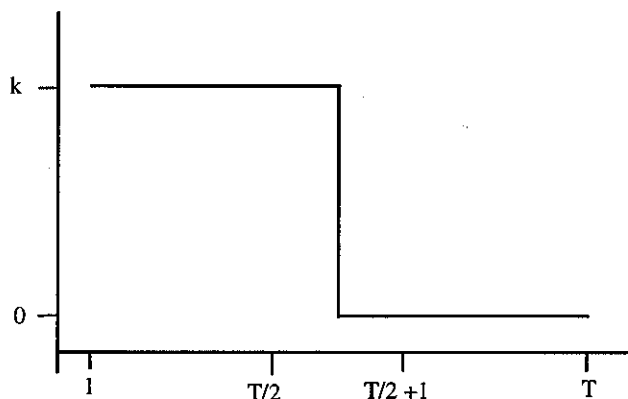
The change in power provided an objective criterion for measuring the effectiveness of the different methods as the trend signal in each scenario was increased. The results emphasized that the effectiveness of some methods is specific to only some patterns of temporal change. Linear regression and the Mann-Kendall test performed well for Scenarios 1 and 2, but very poorly for Scenario 3. The LOESS smoother performed well for Scenario 3 and reasonably well for Scenarios 1 and 2. The successive difference test performed moderately well throughout.

Although this assessment provided an effective basis on which to compare methods, the Workshop had queried the extent to which the methods would be affected by extreme values, or 'outliers'. The ACME therefore considered that the methods should also be compared in terms of their *robustness*, or sensitivity to 'outliers', and took note of the following study reported by WGS AEM.

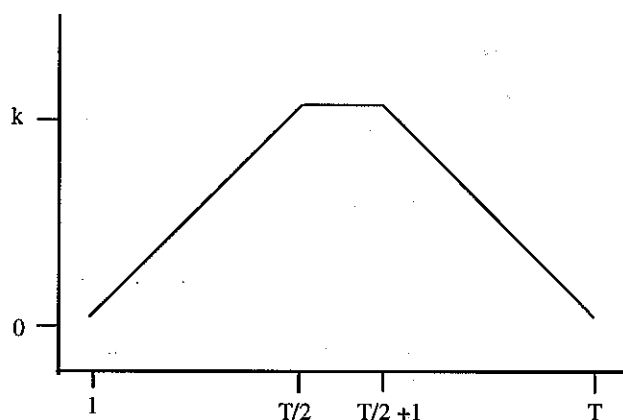
**Figure 4.7.2.1.a.** Scenario 1 shows a linear decrease over a period of  $T$  years.



**Figure 4.7.2.1.b.** Scenario 2 shows a step decrease occurring within a period of  $T$  years.



**Figure 4.7.2.1.c.** Scenario 3 shows a period of increase followed by a period of decrease occurring within  $T$  years.



The effect of an extreme observation will depend not only on the statistical method, but also on the signal-to-noise ratio of the underlying trend, the shape of this trend, and the length of the time series. It will also depend on the magnitude of the outlier, whether it lies above or below the underlying trend, and on its position in the series. The effect will also depend on the number of outliers present. To constrain this large number of variables, the study was restricted to:

- one positive outlier
- with a magnitude defined by the difference between the minimum and maximum value in the trend series, added to
- a trend signal that has a power of 90 % for a given method/scenario if this power is achievable, a range of  $6\sigma$  if it is not, for each of  $T = 10$  and  $T = 20$  years.

The power was then recalculated with an outlier present at each possible position in the time series.

The results were somewhat complex. However, broadly:

- the Mann-Kendall test is least affected under Scenario 1 (it does well the job it was designed to do);
- the effect of an outlier is reduced as the length of the time series increases;
- the difference between the methods is reduced as the length of the time series increases;
- for Scenarios 1 and 2, the effect of an outlier in the first half of the series (where a positive outlier tends to enhance a downward trend) is less than in the second half of the series (where a positive outlier tends to obscure the downward trend);
- for Scenarios 1 and 2, the effect of an outlier in the first half of the series may increase the power above 90 %.

The study also looked at the change in the notional 5 % significance level when an outlier of magnitude  $3\sigma$  is present at each position in a time series of length  $T = 10$  and  $T = 20$  years when there is no underlying long-term trend. Again, broadly:

- 1) the successive difference test was the most successful at maintaining a significance level of 5 %;
- 2) the other methods had significance levels below 5 % when the outlier occurred anywhere except in the first or last year;
- 3) this discrepancy decreased with increasing length of time series;
- 4) all methods had significance levels higher than the notional 5 % when the outlier occurred in the first or last year. For  $T = 10$ , the smoother was the most sensitive, then linear regression, then Mann-Kendall, and finally the successive difference test.

As with power, robustness as measured here is a tool to assist in choosing an appropriate method. Power and robustness should be used together with information about the context of the analysis and the monitoring objectives. For example, an outlier may correspond to a genuine short-term change or, at the end of a series, provide a warning of the beginning of a long-term change. Hence, if early warning is an important aspect of the monitoring programme, sensitivity to short-term changes may be desirable.

#### *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.

#### *Recommendations*

Specifically for the case of contaminant inputs, ICES recommends that regional monitoring organizations ensure that they select a statistical method that has appropriate properties. In particular, a method should be one

- 1) that is able to detect the patterns of change in contaminant inputs that are of interest;
- 2) for which the power to detect appropriate patterns of change has been evaluated and found to be adequate for the stated objectives;
- 3) for which robustness and sensitivity have been considered relative to the distributions of observations that are likely in contaminant input time series.

By exploiting the assessments of statistical power reported in 1997 (ICES, 1997) and the results presented

here on robustness, Member Countries can make an informed decision on appropriate statistical methods.

#### *Reference*

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

#### **4.7.3 Development of statistical methods for assessing trends in contaminant inputs**

##### *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 1998 report of the Working Group on Statistical Aspects of Environmental Monitoring (WGSAM) and ACME deliberations.

##### *Status/background information*

Following the 1997 ICES/OSPAR Workshop on the Identification of Statistical Methods for Trend Detection, WGSAM had discussed several statistical issues concerning the analysis of contaminant inputs data. The ACME noted three areas in which WGSAM had considered that techniques for assessing contaminant inputs need to be developed further. These are:

- 1) To evaluate the effectiveness of integrated trend assessment protocols. In particular, there is a need for such methods to:
  - a) assess the total significance level for a protocol, which may consist of a series of correlated tests of significance (the relationship between the total significance level and the significance level of the individual tests should be established);
  - b) provide guidance for controlling the total significance level of a protocol at a specified value;
  - c) assess the power of a protocol for a range of scenarios of changes in inputs;
  - d) develop a method for quantifying the *post-hoc* power for a protocol, e.g., for a particular data set, to compute the size of the trend that would have been detected with a 90 % power at a given significance level. This is particularly important for interpreting results that are not statistically significant, where environmentally significant

changes could be masked by high levels of variability or short time series.

- 2) To consider the appropriate temporal scale of measurement of contaminant inputs, particularly for riverine inputs. Specific aspects include the relative efficiency of monthly or annual data, how to analyse such data, the treatment of seasonal effects, and the effect of serial correlation in the data. Preliminary results have been presented by WGSAM on the relative efficiency of monthly and annual data. Although annual data or data summaries may offer some benefits such as simplicity and clarity (although it is usually best to report and archive disaggregated data), a statistical analysis on monthly data may provide more information on temporal trends with a potential increase in the power to detect them.
- 3) To develop methods to incorporate cofactors such as river flow for riverine inputs. When monitoring contaminant transport in rivers, an important issue is how the estimated transports could be corrected for variations in flow, both within and between years. Different approaches may be required depending on what the aims of the calculations are, and what kinds of data are available. One common goal (as in the OSPAR reporting of inputs to the North Sea) is to obtain the best estimates of the actual accumulated transport over a particular period, including variations from year to year. Another important purpose of monitoring is to estimate underlying temporal trends in the transports. In that case, there is an interest in reducing the incidental variance between years as much as possible, i.e., somehow to normalize transports to equal conditions with respect to factors that vary independently of variation in contaminant loads. Several methods were considered and reported by WGSAM.

#### *Need for further research or additional data*

The ACME considered that substantial progress has been made in the development of these methods and that further developments by WGSAM should be encouraged.

#### **4.7.4 Issues relevant to the assessment of temporal trends of contaminants in biota**

##### **4.7.4.1 Impact of expressing metal concentrations on a dry- or wet-weight basis**

#### *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 1998 report of the Working Group on Statistical Aspects of Environmental Monitoring (WGSAM) and ACME deliberations.

#### *Status/background information*

The ACME noted the results of a small statistical study reported to WGSAM which quantified the effect on the between-year variation (and hence the power for detecting temporal trends) of expressing trace metal concentrations on a wet-weight or a dry-weight basis in blue mussel (*Mytilus edulis*) soft body tissue. Although there were differences in the between-year variation, they were not consistent for all metals at the two sites investigated.

These results are reasonably consistent with results previously reported by WGSAM (ICES, 1996) for trace metals in fish liver. There, it was found that expressing trace metal concentrations on a *lean-weight* (=wet weight - fat weight) basis was more appropriate than on a *total wet-weight* or *fat-weight* basis. However, there seemed to be no gain in correcting for water and expressing metals on a *dry lean* basis. For liver, this may possibly be due to analytical errors in the measurement of *dry %* and *fat %*, which would not be a problem with blue mussels.

#### *Need for further research or additional data*

The ACME agreed that further data should be evaluated as and when they become available.

#### *Recommendations*

From this investigation it is not possible to give any recommendation concerning whether to use a wet-weight or a dry-weight basis for trace metal concentrations in biota, using the criterion of minimizing the between-year variation. Since changing the basis of a measured concentration may introduce additional measurement error, the ACME considered that sensible advice is to report concentrations relative to the basis on which they are measured, and that this should be consistent within the monitoring programme.

#### **Reference**

ICES. 1996. Report of the Working Group on Statistical Aspects of Environmental Monitoring. ICES CM 1996/D:1.

#### **4.7.4.2 Impact of deviations from sampling guidelines on temporal trend assessments**

##### *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 1998 report of the Working Group on Statistical Aspects of Environmental Monitoring (WGSAM) and ACME deliberations.

##### *Status/background information*

The ACME had requested WGSAM, at its 1998 meeting, to consider whether it would be possible to include, in data assessments, analytical data on contaminant concentrations in samples which did not fully meet the sampling guidelines (e.g., insufficient numbers of fish were available, or the available fish were of a different size range from that sampled previously) and the effect that the inclusion of such data would have on the confidence the assessors could have in the overall assessment, or whether such data should be excluded.

The ACME noted the deliberations of WGSAM on this topic of data quality and acceptability, as summarized in the following paragraphs.

If intermittent unexplained extreme values in a contaminant time series are a common feature which tends to obscure an otherwise regular trend, then a better estimate of this trend could be made by using a robust method, such as a robust form of smoothing. This would iteratively down-weight large residuals, making the smoother less responsive to single extreme data points.

Departures from the guidelines might have two different kinds of effects—those which affect the mean of the annual index (caused by shifts in the sampled population, e.g., fish of a different size), and those which affect its variance (e.g., a reduction in sampling intensity). Both of these effects may occur at once, e.g., by collecting fewer fish in only one size range. The effect will be as follows:

- 1) If they occur at random, both types of departure will reduce the probability that an existing trend will be detected.
- 2) Random shifts in the sampled population will tend to inflate the between-year component of variance.

3) A reduction in sampling intensity will increase the within-year component of variance. (Ideally, annual indices could then be weighted to reflect this varying precision.)

4) Although departures from the guidelines should obviously be avoided, the data should still be analysed, with annotation of the results to aid the interpretation of peculiar features of the observed time series.

5) The worst case would be non-random changes in the sampled population. For example, if the sampled population was consistently different in two sequences of years, there may be a spurious shift in the corresponding observed contaminant levels.

The ACME also noted the practice, in the analysis of the OSPAR Joint Monitoring Programme (JMP) data for temporal trends, of deleting data sets or data for particular years that fail corresponding quality assurance criteria. Again, randomly occurring biases and large within-batch standard deviations will tend to inflate between-year and within-year components of variance, with a corresponding loss of power.

The worst case would be if there were some non-random sequence of bias, e.g., induced by a change in analytical method with a corresponding shift in bias. However, this effect would not be detected by the current OSPAR practice of deleting data year by year. Again, a more sensible practice might be a shift to retaining more of the data, with some annotation of the results. Nicholson and Jones (1997) discuss further the interaction between monitoring and analytical targets.

##### *Need for further research or additional data*

It may be useful to explore in more detail the consequences of different forms of non-compliance with monitoring guidelines. Similarly, it may be useful to explore the effect on monitoring performance of different levels of analytical variability. This could be developed to establish sensible guidelines for using analytical quality control data for screening monitoring time series.

##### *Recommendations*

ICES has always recommended that monitoring guidelines be followed, and continues to do so. However, if difficulties in meeting guidelines persist, consideration should be given as to whether it would be appropriate to re-assess and revise them.

##### **Reference**

Nicholson, M.D., and Jones, B. 1997. Setting performance targets for analytical accuracy. *Marine Pollution Bulletin*, 35: 181–182.

#### 4.7.5 Use of seabird eggs in contaminant monitoring: variance components in analysis

##### *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 1998 reports of the Working Group on Statistical Aspects of Environmental Monitoring (WGSAM) and the Marine Chemistry Working Group (MCWG), and ACME deliberations.

##### *Status/background information*

Seabird eggs seem to be an appropriate matrix for temporal trend analysis of concentrations of mercury and organochlorine contaminants and possibly other organic compounds. They provide a well-defined sampling unit relative to other biological matrices, in that sex and physiological, nutritional and reproductive status are known. They reflect the status of the mother, and evidently the mothers are healthy, active and reproductive. Eggs also provide sufficient material for individual analysis, have a relatively high and stable fat content, and allow integrated biological effects monitoring of shell characteristics.

However, experience with spatial studies based on seabird eggs is limited, and it may prove difficult to find bird populations in all areas which do not migrate and, consequently, represent the area in which they were sampled. The target population must also be dense and stable, thereby allowing repeated sampling without depleting the population.

At present only Canada, Sweden, and Finland have programmes that include monitoring of contaminants in seabirds, although a trilateral programme involving common tern eggs from the Wadden Sea is to be undertaken by Denmark, Germany, and the Netherlands.

The ACME noted that good ornithological information is required to identify strictly marine feeding species with limited or well-defined migratory patterns. The habitat and nesting behaviour also need to be examined before a monitoring programme using eggs can be established. This programme would be most appropriate for specific, smaller areas within a larger regional area, and should be in addition to the usual monitoring of contaminants in fish, shellfish, and sediments. ACME advice regarding specific seabird species useful for monitoring contaminants in particular areas of the Northeast Atlantic is contained in the 1995 ACME report (ICES, 1995).

There is also a need to establish the levels of variability in the population, between and within clutches, and analytically both within and between laboratories. Requests have been made to WGSAM to provide information on levels of sampling and analytical variability, in particular:

variability between eggs within the same clutch,  $\sigma_c$ ;

variability between clutches within the same site,  $\sigma_s$ ;

variability between sites within the same year,  $\sigma_y$ ;

variability between years at the same site,  $\sigma_y$ ;

analytical variability within laboratory, within year,  $\tau_w$ ;

analytical variability within laboratory, between years,  $\tau_y$ ;

analytical variability between laboratories,  $\tau_l$ .

The ACME accepted the following summary providing indications of the likely magnitudes of these components of variance, as presented by WGSAM.

WGSAM exploited summary data from several reports (cf. ICES, 1997 (Annex 6); Barrett *et al.*, 1996; Becker *et al.*, 1989, 1991) and data on guillemot eggs collected within the Swedish monitoring programme (Bignert, 1997) to obtain estimates of some of these variance components. However, no information on analytical variance for contaminants in seabird eggs was available. Indications of potential levels of analytical variability from other tissues are discussed later.

##### **Estimation of Variance Components**

Where the available reports gave information on means and variances within sites, a pooled within-site variance was calculated. The between-site variance was calculated from the variance between the means adjusted for the pooled within-site variance. (It should be noted that in these reports the between-site variation is for sites separated on a large geographical scale, e.g., spread along the German North Sea coast.)

Where data were available on between-clutch and between-egg variability, pooled between-clutch and between-egg variance estimates were made. Note that the data in the tables below provide, for each egg position, the mean and the standard deviation computed over clutches within a site.

Also note that as pooled values for the within-group (sites or egg-position) variances have been used, negative estimates of the between-group variances (obtained by subtracting the within-group variance) can result. Any such negative estimates were set to zero.

Measurements at the same site and from the first egg in each clutch were available from the Swedish monitoring programme. A smoother was fitted to each series to



estimate the residual variance. This was adjusted for the between-clutch variance to give an estimate of between-year variance.

For some of the data provided, it was not possible to separate out all of the variance components. Also, as no data for analytical variability of contaminant measurements in bird eggs were available, it was not possible to remove its effect. Therefore, some variance estimates (between clutches and between years) include analytical variability.

The standard deviations corresponding to the estimated variance components for four contaminants in various species are given in Tables 4.7.5.1 to 4.7.5.4. All the variances are expressed on a log scale, i.e., as if log-concentration data were used. Hence, the standard deviations can be interpreted approximately as coefficients of variance.

**Table 4.7.5.1.a.** Between-egg, between-clutch, and between-site standard deviations for common tern; samples of three eggs have been taken from each of fifteen clutches (from Becker *et al.*, 1991).

Common tern	Between eggs	Between clutches	Between sites
	$\sigma_e$	$(\sigma_c^2 + \tau_w^2)^{0.5}$	$\sigma_s$
Hg	0.21	0.35	1.27
PCB	0.29	0.46	0.42
HCB	~0	0.38	1.28
DDE	0.23	0.41	0.77

**Table 4.7.5.1.b.** Between-egg and between-clutch standard deviations for oyster catcher; samples of three eggs have been taken from each of fifteen clutches (from Becker *et al.*, 1991).

Oyster catcher	Between eggs	Between clutches
	$\sigma_e$	$(\sigma_c^2 + \tau_w^2)^{0.5}$
Hg	~0	0.31
PCB	~0	0.32
HCB	0.08	0.33
DDE	0.12	0.36

**Table 4.7.5.1.c.** Between-egg and between-clutch standard deviations in herring gull (based on  $n = 24$ , from Becker *et al.*, 1989). Note that the between-clutch standard deviations include some yearly variation as the data combined measurements from 1979 and 1980.

Herring gull	Between eggs	Between clutches
	$\sigma_e$	$(\sigma_c^2 + \tau_w^2)^{0.5}$
Hg	0.14	0.42
PCB	~0	0.62
HCB	~0	0.62
DDE	~0	1.08

**Table 4.7.5.2.** Between-year standard deviations for data on common guillemot (from Bignert, 1997).

Common guillemot	Between years	No. of years
	$(\sigma_y^2 + \tau_y^2)^{0.5}$	
Hg	0.12	25
PCB	0.14	26
HCB	0.22	10
DDE	0.11	26

**Table 4.7.5.3.** Between-site standard deviations (from Barrett *et al.*, 1996).

	Herring gull (4 sites)	Kittiwake (4 sites)	Brunnich's guillemot (3 sites)
	$\sigma_s$	$\sigma_s$	$\sigma_s$
Hg	0.14	~0	0.53
PCB	0.42	0.31	0.35
HCB	0.41	0.17	0.21
DDE	0.59	0.34	0.13

**Table 4.7.5.4.** Within-site standard deviations (combined between-clutch and between-egg variations, from Barrett *et al.*, 1996).

	Herring gull (4 sites)	Kittiwake (4 sites)	Brunnich's guillemot (3 sites)
	$(\sigma_e^2 + \sigma_c^2 + \tau_w^2)^{0.5}$	$(\sigma_e^2 + \sigma_c^2 + \tau_w^2)^{0.5}$	$(\sigma_e^2 + \sigma_c^2 + \tau_w^2)^{0.5}$
Hg	0.41	0.50	0.31
PCB	0.35	0.28	0.16
HCB	0.25	0.17	0.15
DDE	0.21	0.54	0.32

The above tables indicate that, in general:

- 1) the between-site standard deviations are higher than the between-clutch standard deviations, which again are higher than between-egg standard deviations;
- 2) the between-site standard deviations depend on the range of contamination across the sites studied; for example, the high between-site standard deviations for the common tern (Table 4.7.5.1.a) are mainly caused by the high levels at one site;
- 3) the between-clutch standard deviations show some similarities between species and contaminants, ranging approximately from 0.3 to 0.6. The between-egg standard deviations range from approximately 0 to 0.3.

However, the tables also show large differences between species and areas of investigation. Furthermore, no consistent differences between the contaminants presented are seen.

**Using the estimated variance components to design an effective sampling strategy, taking account of systematic variation between eggs within a clutch**

Several papers investigating the differences between concentrations in the eggs within a clutch (e.g., Bignert *et al.*, 1995; Becker *et al.*, 1989) have suggested systematic patterns in concentrations with the laying order of the eggs.

This information was used to explore sampling strategies incorporating two ways of sampling eggs from each clutch: either an egg is chosen at random, or a specific egg is chosen in the laying order selected (often the first egg laid). It was assumed that using a specific egg removes all of the variation between eggs within a clutch.

The variance components calculated above can be used to compare the effect of these two strategies on the standard deviation of the estimated mean concentration. Three scenarios have been chosen:

- 1) sampling at the same site and the same year;
- 2) sampling at the same site for several years;
- 3) sampling at five sites in the same year.

Each scenario has been evaluated for samples from 5, 10, or 15 clutches at each site.

In general, assuming that all analyses are made within a single laboratory, the variance of the estimated annual mean log concentration when sampling is carried out by collecting a single egg in each of a number of years, sites, and clutches is:

$$\psi^2 = (\sigma_y^2 + \tau_y^2) + \sigma_s^2/n_s + (\sigma_c^2 + \tau_w^2)/n_s n_c + \sigma_e^2/n_s n_c$$

where:

$n_s$  is the number of sites;

$n_c$  is the number of clutches sampled per site.

Sampling and analytical variance components that cannot be separated using the estimates in the tables above have been bracketed. For the scenarios corresponding to a single year or site, the appropriate terms can simply be removed from the formula. Similarly, when an egg from a clutch is taken from the same position in the laying order, the term for variance between eggs within a clutch is removed.

To demonstrate, data from two species had to be combined. The estimated variance components from the common tern (Table 4.7.5.1.a) were used except for between-year variability, which is for the common guillemot (Table 4.7.5.2). Combining components of variance from several species is, of course, questionable since different feeding habits and migration behaviour of different species may lead to large differences in the component of variance.

Table 4.7.5.5 illustrates the effects of taking a random sample of one egg from each clutch or sampling the same egg in the order of laying on the estimated standard error of the mean log-contaminant concentrations for the three sampling scenarios and three sampling intensities.

In general, Table 4.7.5.5 shows that the gain in precision obtained by sampling the same egg in the order of laying compared with random sampling decreases when monitoring is conducted over several years and/or several

**Table 4.7.5.5.** Standard error for random sampling/standard error for sampling the same egg in the laying order for different combinations of sites sampled and years of sampling (using the estimated variance components from the common tern except for between-year variability, which is for the common guillemot).

No. of eggs (1 from each clutch)	Contaminant	1 site 1 year	1 site several years	5 sites 1 year
5	Hg	0.18/0.16	0.22/0.20	0.57/0.57
	PCB	0.24/0.21	0.28/0.25	0.21/0.21
	HCB	0.17/0.17	0.28/0.28	0.58/0.58
	DDE	0.21/0.18	0.22/0.21	0.35/0.36
10	Hg	0.13/0.11	0.18/0.16	0.57/0.57
	PCB	0.17/0.15	0.22/0.20	0.20/0.20
	HCB	0.12/0.12	0.25/0.25	0.57/0.57
	DDE	0.15/0.13	0.18/0.17	0.35/0.35
15	Hg	0.11/0.09	0.16/0.15	0.57/0.57
	PCB	0.14/0.12	0.20/0.18	0.20/0.20
	HCB	0.10/0.10	0.24/0.24	0.57/0.57
	DDE	0.12/0.11	0.16/0.15	0.35/0.35

sites. This happens because the relative importance of the between-egg variance component decreases. Increasing the number of clutches sampled at each site increases the gain, but only to a minor degree. When sampling from several sites, the dominance of the between-site variance component is so great that no gain at all is seen.

This discussion has focused on a relatively simple statistical criterion—the precision of the estimated mean log concentration. Further use of the calculated variances could be made by calculating the power to detect a trend (cf. Nicholson *et al.*, 1997).

### Analytical Variability

Although no values are available for egg tissue, levels of analytical variability in other tissues derived from various sources (QUASIMEME, ICES) may give a rough guide. Typical coefficients of variation for various CB congeners in fish liver oil were:

- 7–13 % between batch, approximately  $\tau_y$ , (based on  $n = 12$ );
- 10–14 % within-batch,  $\tau_w$ , (based on  $n = 5$ );
- 17–30 % between-laboratory,  $\tau_l$ .

Results for analyses of CBs in egg tissue, which contains a relatively high and stable amount of fat, would probably not be worse than this. For mercury in various tissues, between-laboratory CVs of 16–21 % have been reported (Berman and Boyko, 1992).

Note that the within- and between-batch standard deviations (for CBs) tend to fall in the upper half of the ranges in Table 4.7.5.4 and Table 4.7.5.2, respectively. This suggests that, for temporal monitoring at a site, analytical error would constitute a large proportion of the noise. However, given the larger component of between-site variation, this level of analytical variation would be compatible with monitoring across several sites.

### Conclusions

- 1) The extracted components from the available data sets show a large variability between species and areas.
- 2) In general, the between-site variation was the largest, followed by the between-clutch and the between-egg variation.
- 3) For combining data over a range of sites to provide, e.g., a regional average, the potentially larger component of between-site variation makes it less important to control the between-egg variation by sampling a certain egg in the laying sequence.
- 4) For temporal trend studies at the same site, sampling a certain egg in the laying sequence gives an improvement in the precision of the results, although

this will depend upon the magnitude of the within-batch analytical variance.

- 5) The relative magnitudes of these components of variance have been compared on a purely arithmetical basis, and no account has been taken of how well they have been estimated or whether they are statistically significant.

### Need for further research or additional data

No data were available on the analytical precision for measurements of contaminants in egg tissue. Hence, in the study reported above, it was not possible to quantify the components of analytical variation, or to separate them from the other estimated components. Samples of within-laboratory analytical variances or results from an intercalibration exercise including seabird eggs in the sample tissues are necessary to do this.

Finally, the ACME noted that the evaluation of the importance of the different components of variation in a monitoring programme based on seabird eggs was in terms of the standard error of an estimated mean. This study should be developed with further investigations of more focused objectives.

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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.

#### **4.8 Evaluation of the Effectiveness of Monitoring Programmes to Determine Trends against a Background of Natural Fluctuations**

##### *Request*

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

##### *Source of the information presented*

The 1998 reports of the Working Group on Shelf Seas Oceanography (WGSSO) and the Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM), and ACME deliberations.

##### *Status/background information*

The ACME has noted criticism with respect to insufficient sampling, unclear objectives, and the general status of some environmental monitoring programmes, as reflected in its 1996 report (ICES, 1996). In some cases, changes in monitoring strategy are already under way even before firm conclusions have been reached on the general functioning of the programme. Therefore, it is important to evaluate the effectiveness of individual environmental monitoring programmes in terms of their ability to determine possible trends against the natural variability.

There is a broad range of problems to consider. As an example, the Working Group on Shelf Seas Oceanography (WGSSO) has reviewed a Norwegian report which included some general points concerning the focus of monitoring programmes and emphasized the importance of considering the possibility of detecting changes as well as the statistical significance of monitoring results. A main aim of a monitoring programme is to detect a slowly developing trend against a background of large natural variations. The statistical basis for developing a programme that meets this aim

requires, at a minimum, answers to the following two questions:

- a) What size of changes in concentration is it important to be able to detect?
- b) Which probability level is desired for the conclusion to be correct?

The Norwegian report describes different observation techniques to address these concerns.

- 1) For most parameters, it is difficult to estimate realistic changes in yearly means based on traditional observations. This suggests that a year-to-year comparison should be made on a seasonal basis, for instance, for those seasons when the variability is low compared to the mean value. This indicates that a uniform observation frequency throughout the year for all variables is not advisable.
- 2) Cruises with good spatial coverage provide the form of surveillance giving the most thorough information for a given area and time. This kind of data gathering is most important for the characterization of water masses and spatial distributions. The spatial information from such cruises could be further enhanced by coordinating with other methods. Numerical models could be initialized with fields interpolated from cruise data and results from models used to evaluate the degree of representativity of measurements from fixed stations.
- 3) The main purpose of fixed sections is to monitor large-scale variability. Station spacing of about 10 km near shore and in frontal regions, and 20-30 km in more homogeneous water, seems to be adequate for resolving most of the spatial variance.
- 4) Measurements from ships of opportunity are a reasonable and effective method for gathering many different types of data from the marine environment, but the potential of the method is at present far from fully utilized.
- 5) Fixed coastal stations play a central part in most monitoring programmes, and long historical time series exist. It is therefore natural to build future monitoring programmes around such stations. The choice of variables, position and number of stations, and depths and frequency of observations are important. In the upper layers (such as in the Skagerrak), measuring once a week is recommended to include most of the variance, while in deeper water layers measurements 1-2 times per month seem suitable. Reducing the number of stations to allow for higher intensity at some selected locations must be considered.
- 6) Several of the most important environmental parameters have a significant part of their (near surface) variability at frequencies so high that they,

in practice, cannot be captured by traditional measurement methods. Automatic buoys can register most of the total variability; however, like fixed stations, the great spatial variability perpendicular to the dominant current pattern, even at short distances, leads to the measurements from single buoys being relevant only for small areas. With the cost of some of today's buoys, they are recommended for monitoring in straits, some fjords, and otherwise in situations where single buoys are adequate. To cover larger, more open areas, the use of buoys must be combined with other methods.

- 7) Satellites are able to give information about the sea surface with a relatively high resolution in both space and time. For many years, ocean currents have been estimated from satellite monitoring of the sea surface topography (using altimeters), and wind speed and wave height estimated from the same instrument are regularly used in weather forecasting. Sea surface temperature is the most commonly used remotely sensed parameter, and sea ice distribution has also been monitored for many years. Even accounting for shortcomings related to cloudiness and low solar altitude (a restriction for some of the parameters), remote sensing methods have a potential beyond that which is utilized in current surveillance programmes. The potential within coastal monitoring has lately been significantly enhanced by the introduction of the Sea WiFS sensor on one of the NASA satellites. Further advancement will follow in the near future when the MERIS is launched and later the ENVISAT. To utilize the large amount of information from satellites, it is necessary for some of the parameters to move beyond the usual pictures and to make the information available as reliable statistical material.
- 8) Three-dimensional circulation models, some coupled with a chemical-biological component, can give a valuable contribution to surveillance programmes. Still, such models need refinements and to be properly validated, and so far chemical-biological models are not much used in operational monitoring programmes. An important property of models is that they can be used to separate between anthropogenic and natural variability, and that probable effects of future management measures can be simulated.
- 9) Perhaps the least costly area for enhancement relates to the methods currently used for analysing sampled data. Many of the data series are undersampled compared to what is necessary to catch most of the variability. Methods taking this aspect into consideration are seldom used at present.
- 10) The great differences regarding strengths and weaknesses of monitoring programmes indicate that much can be gained by utilizing the best of several methods in close coordination. Numerical models should be used to a greater extent to put scattered data into a spatially and temporally continuous

context. The large amount of data from satellites can be made more reliable and valuable by linking them to data from research vessels, ships of opportunity, and automatic buoys.

The ACME noted that the above Norwegian strategy for monitoring environmental parameters was specifically relevant to the oceanographic characteristics of the northern Skagerrak and the Norwegian coast. More generally, monitoring of environmental parameters that act passively in sea water also has to take into account factors such as tidal streams and (weather-induced) water mass changes in space and time, which are commonly handled by normalization with respect to salinity. Examples of how the ICES Secretariat determines trends in nutrient data, as a contribution to OSPAR's Ad Hoc Working Group on Eutrophication (EUT) programme, were described in last year's ACME report (ICES, 1997). The analysis described was conducted on data collected from areas where natural variations predominate. The statistical basis of such analyses needs, however, to be elaborated in collaboration with WGSAM.

The ACME further noted that WGSAM has discussed various methods for assessing trends in contaminant time series (cf. Section 4.7.2, above). WGSAM has also considered a further potential method that may be appropriate for detecting trends against a background of natural fluctuations. This consists of the application of Singular Spectrum Analysis (SSA) to time series (Vautard and Ghil, 1989).

SSA is a non-parametric method that enables various characteristics of a time series, such as the trend and periodic components, to be described. It was illustrated by applying the method to the mean annual sea levels at Vlissingen from 1862 to 1990. The method is a type of Principal Component Analysis (PCA), modified by using a running window to expand the univariate time series into a multivariate time series. The mathematical principles underlying both methods are the same. The advantages of SSA over PCA are that using the window enables the method to be applied to a single time series and the various characteristics of the time series can be distinguished more satisfactorily.

The SSA method can also be applied to several time series simultaneously. This yields a type of average information for the various time series and gives a picture of the characteristics shared by the time series.

## References

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#### 4.9 Strategies for Monitoring the Pelagic Ecosystem

##### *Request*

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

##### *Source of the information presented*

The 1998 reports of the Working Group on Phytoplankton Ecology (WGPE) and the Working Group on Environmental Assessment and Monitoring Strategies (WGEAMS), and ACME deliberations.

##### *Status/background information*

High natural variability in space and time is typical for the pelagic marine ecosystem. This has been shown by the ICES-coordinated International Investigation of Patchiness in the Baltic Sea, 1986 (PEX86) and the Skagerrak Experiment, 1990 (SKAGEX), as well as in several papers presented at the International Symposium on the Temporal Variability of Plankton and their Physico-Chemical Environment, held in Kiel in March 1997. The ideal monitoring programme should be able to cover the range of natural variability in order to be able to detect possible long-term trends. This can be achieved by an intensive and extensive sampling programme which, however, is not possible in practice using traditional techniques, mostly owing to financial reasons.

There are several approaches in the different maritime regions and in ICES Member Countries to monitor the pelagic ecosystem. In 1998, WGPE concluded that a tremendous effort is being given to monitor pelagic parameters for a series of objectives, from purely scientific to commercial and/or user oriented. However, there are still very few attempts to coordinate national monitoring programmes so that a direct comparison between national data is possible on an international

level. Only parameters which are part of the HELCOM Cooperative Monitoring in the Baltic Marine Environment (COMBINE) programme and the OSPAR Joint Assessment and Monitoring Programme (JAMP) are subject to intercalibration and standardization, including the introduction of quality control procedures. In the HELCOM COMBINE programme, a strategy to use various techniques and methods to monitor several components of the pelagic ecosystem (bacteria, phytoplankton, mesozooplankton) is used. The Intergovernmental Oceanographic Commission (IOC) has published a manual dealing with monitoring strategies for harmful algae (Andersen, 1996). The work of WGEAMS has mainly been dealing with the monitoring of contaminants, and strategies concerning the pelagic ecosystem have only briefly been discussed. Proper statistical considerations for designing the overall monitoring strategy and, e.g., the sampling strategy have not been applied.

During recent years, new techniques have become available for monitoring pelagic ecosystems, such as automated platforms (buoys and voluntary observing ships (VOS)) equipped with, e.g., fluorometers, cytometers, nutrient analysers and water samplers, as well as new satellite and other remote sensors. The Global Ocean Observing System (GOOS) is planning to include these kinds of systems in its surveillance programme in combination with ecosystem models.

##### *Need for further research or additional data*

The development of new techniques and sampling strategies should be followed and evaluated by appropriate ICES Working Groups and reported to ACME.

Existing long-term data sets provide valuable information on long-term fluctuations in the various components of the marine ecosystem. The ACME shares the concern expressed at the International Symposium on the Temporal Variability of Plankton and their Physico-Chemical Environment, which strongly recommended the continuation of high quality long-term data sets.

The ACME noted the recommendation of WGPE that, owing to their high value, long-term ecological monitoring data require a proper analysis. This process deserves more attention to ensure that no important information is lost.

The ACME also noted that statistical methods provide a tool for designing monitoring programmes. Relevant statistical methods should be tested using existing data sets on pelagic biological variables to demonstrate their possibilities in the design of monitoring programmes. WGPE and the Working Group on Zooplankton Ecology

(WGZE) should identify suitable data sets (with proper descriptions) and discuss with WGSAM possible further joint uses of these data sets.

#### *Recommendations*

ICES encourages ICES Member Countries to ensure the continuation of existing or recently closed high quality

long-term data sets on various components of the marine ecosystem.

#### **Reference**

Andersen, P. 1996. Design and implementation of some harmful algal monitoring systems. IOC Technical Series, 44: 1–102.

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

#### *Request*

Item 2 of the 1998 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 1998 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 Baltic Monitoring Programme (BMP) has proceeded in a systematic manner since the establishment of the ICES/HELCOM Steering Group on Quality Assurance of Biological Measurements in the Baltic Sea (SGQAB) in 1992. The following progress has been made since the 1997 meeting of ACME:

- The HELCOM COMBINE Manual has been accepted by the Helsinki Commission and the QA sections, originally written by the ICES/HELCOM Steering Group on Quality Assurance of Chemical Measurements in the Baltic Sea (SGQAC), have been reviewed and revised slightly by SGQAB in such a way that they are applicable for both chemical and biological analyses. Specific QA instructions for the biological analyses are under development.
- The method for mesozooplankton sampling and analysis was rewritten.
- The 'Working manual and supporting papers on the use of a standardized incubator technique in primary production measurements' was reviewed and comments were forwarded to the ICES Working Group on Phytoplankton Ecology (WGPE) for their consideration in a possible revision of the manual.
- Coastal waters of the Baltic Sea are also included in the COMBINE Programme and a number of 'new' laboratories will deliver data to HELCOM. In-house QA manuals are also needed for these laboratories.
- The work of the taxonomic training courses on phytoplankton and benthos has been continued, but the training course on protozooplankton and microzooplankton had to be cancelled due to the small number of participants registered to attend.

#### *Need for further research or additional data*

The lack of biological reference materials (RMs) and the small number of intercalibrations/ring tests are an urgent matter of concern.

At present, certified reference materials (CRMs) are available only for chlorophyll *a*. The possibility of improving the QA in identification of the species in the monitoring samples would be increased if identification materials were available on the Internet. A Swedish institute provides identification material on Kattegat/Skagerrak phytoplankton on the Internet and the Finnish Institute of Marine Research has started to publish plankton sheets on its specific website ([www2.fimr.fi/algaline](http://www2.fimr.fi/algaline)). It is recommended that ICES Identification Leaflets for Phytoplankton also be made available via the Internet.

National ring tests are organized in several countries and they may provide the basis for the development of HELCOM-wide tests.

The new variables in the coastal programme, still under development, should be covered by proper QA instructions and evaluated by SGQAB.

Continuous updating of the taxonomic list of species is an important part of the biological QA programme.

#### *Recommendations*

ICES recommends that the ICES/HELCOM Steering Group on Quality Assurance of Biological Measurements in the Baltic Sea continue its work because proper QA procedures are a prerequisite for the HELCOM COMBINE Programme in order to ensure the comparability of the data produced by participating laboratories. Continuous updating of the in-house QA procedures and participation in the work of SGQAB, as well as participation in intercalibration exercises and training courses, should be mandatory for all Baltic laboratories participating in the COMBINE programme. These QA procedures involve costs which should be considered as part of the overall costs of the monitoring programme.

ICES endorses the SGQAB recommendations that the following basic principles for future quality assurance work should be observed:

- 1) the conduct of regular ring-tests on a five-year basis for all biological core and main variables, and mandatory participation for all laboratories which submit data to the database;



- 2) the necessity of mandatory training courses on sampling, analytical procedures, and particularly taxonomy;
- 3) development towards a taxonomic certification for experts in national laboratories and private companies.

Updated taxonomic species lists and biological data reporting formats are urgently needed.

The ICES Identification Leaflets for Phytoplankton should be made available via the Internet.

ICES also recommends that the QA sections of the COMBINE Manual that have been completed should be forwarded to HELCOM.

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

#### *Request*

Item 2 of the 1998 requests from the Helsinki Commission.

#### *Source of the information presented*

The report of the 1997 ICES/HELCOM Workshop/Training Course on Phytoplankton (WKPHYT), the 1998 report of the ICES/HELCOM Steering Group on Quality Assurance of Biological Measurements in the Baltic Sea (SQAB), and ACME deliberations.

#### *Status/background information*

The ICES/HELCOM Workshop/Training Course on Phytoplankton was held at the Tvärminne Zoological Station in Finland from 14–16 August 1997.

The ACME noted that the work of the Phytoplankton Expert Group in the Baltic Sea has continued in a systematic manner although there has been some confusion about its status. At its Nineteenth Meeting, the Helsinki Commission re-established the project 'Quality Assurance of Phytoplankton Monitoring in the Baltic Sea' (HELCOM, 1998) for the years 1998–2000.

Training courses on phytoplankton species identification have been organized annually. Also, other QA matters have been included in the programme (microscope techniques, data issues) and the group has participated in the development of the new COMBINE Manual.

During the Workshop/Training Course in 1997, problematic samples concerning cyanobacteria, in particular, and small flagellates were analysed and discussed under the guidance of experts. A common counting software was distributed. Some problems related to size classes and biomass factors were recognized and will be dealt with in the coming year. The species code will be maintained by the Finnish Institute of Marine Research until a new coding system is developed. In addition, a new species checklist for the Baltic Sea will be prepared by the Finnish Institute of Marine Research in 1998. Identification sheets will be designed by the Finnish Institute of Marine Research in cooperation with the Phytoplankton Expert Group and will be included on the website of the Institute.

In 1997, the Phytoplankton Expert Group recommended that, instead of taking separate water samples, a hose should be used to obtain integrated samples for phytoplankton determination. The sampling depth range should be changed from the earlier range of 0–10 m to 0–20 m since the euphotic layer extends to approximately 20 m and some species, concentrating in deep water layers, can be missed.

The next ICES/HELCOM Workshop/Training Course on Phytoplankton is scheduled to take place in Klaipeda from 12–16 October 1998.

#### *Need for further research or additional data*

The organization of taxonomic training courses and the expert evaluation of monitoring manuals are parts of a continuous process of QA. In addition, intercomparison exercises on analyses of phytoplankton samples should be arranged annually. There is a total lack of reference materials for phytoplankton species.

#### *Recommendations*

ICES stresses that taxonomic expertise is essential for phytoplankton monitoring in the Baltic Sea. 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 Mrs Gertrud Cronberg and Mr Guy Hällfors, the teachers in the training course.

#### **Reference**

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

## **5.1.2 Results of the ICES/HELCOM Benthos Taxonomic Workshop**

### *Request*

Item 2 of the 1998 requests from the Helsinki Commission.

### *Source of the information presented*

The report of the 1997 ICES/HELCOM Benthos Taxonomic Workshop (WKBT), the 1998 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 Benthos Taxonomic Workshop was a follow-up activity recommended on the basis of the outcome of the ICES/HELCOM Workshop on Quality Assurance of Benthic Parameters in the Baltic Sea (ICES, 1994). As one of the activities of this QA workshop, a ring-test was performed with pre-sorted samples. The results revealed considerable inconsistencies regarding the taxonomy of invertebrates in the western parts of the Baltic Sea. To resolve some of the taxonomic problems identified, the Benthos Taxonomic Workshop was held in Copenhagen from 4-7 November 1997. This was the first in an intended series of Taxonomic Workshops.

Twenty participants from five countries (Denmark, Sweden, Finland, Germany, and Poland) attended this workshop. They were mainly from the technical working level from both national institutes and private companies that carry out benthos studies on a consultancy basis. This was intended when this workshop was recommended.

The main species groups studied at this Workshop were polychaetes, sponges, and selected molluscs. The lectures and practical demonstrations were of a high level and on the forefront of taxonomy. Material from participants was also discussed during the lectures and practical activities. The hand-outs and newly designed regional species keys are annexed to the Workshop report. They include rules concerning how to deal with uncertain identifications and their inclusion in data banks.

### *Need for further research or additional data*

There is a need for a continuation of the training courses on benthos taxonomy and sampling procedures in order to improve the quality and comparability of the monitoring data. Furthermore, there is an urgent need for macrozoobenthos reference material.

### *Recommendations*

ICES stresses that taxonomic expertise is essential for macrozoobenthos monitoring in the Baltic Sea. The regular and systematic organization of training courses and the participation of expert staff from all laboratories delivering monitoring data to the HELCOM database are recommended.

### *Additional comments*

ICES expresses its gratitude to the lecturers in the Workshop, D. Eibye-Jacobsen, O. Tendal, M.E Petersen, B. Muus, and G. Hoeppe-Petersen, all from the Zoological Museum of the University of Copenhagen.

### *Reference*

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

## **5.2 Quality Assurance of Biological Measurements in the OSPAR Area**

### *Request*

Item 2.1 of the 1998 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.

### *Source of the information presented*

The 1998 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 took note of the review carried out by SGQAE on the information available regarding relevant biological monitoring programmes, and the quality assurance (QA) procedures associated with them, by OSPAR Contracting Parties and of the outcome of the

joint session between SGQAE and the ICES/HELCOM Steering Group on Quality Assurance of Biological Measurements in the Baltic Sea (SGQAB). The ACME further noted progress made by SGQAE on the production of a Quality Assurance Manual.

SGQAE identified the need to resolve any possible confusion arising from the use of QA terminology. For example, reference to the following terms is commonly encountered:

- quality assurance guidelines;
- quality assurance manuals;
- standard operating procedures (SOPs).

For SGQAE purposes, 'Standard Operating Procedures' (SOPs) are used to provide details on the conduct of specific sampling or analytical tasks, and are generally contained within QA manuals. The content of a QA manual is invariably developed for use at the level of the individual organization but, at the same time, would be expected to accurately reflect more general QA guidelines, where available.

SGQAE supported the production of in-house laboratory QA manuals as the most valuable practical expression of a national or international QA policy. The production of 'Standard Operating Procedures' for field and laboratory biological methods relevant to SGQAE interests is an essential part of QA manual preparation, and a high degree of consistency in their content among laboratories and among countries is to be expected, where guidelines have been fully adopted and correctly translated into local courses of action. An example of content specifications for SOPs is given here. In general agreement with the requirements of the international standard DIN EN 45001, procedures or methods are recommended which are published as international, regional or national standards. Regarding procedures or methods which are not standardized, it is recommended to prepare a description of operational procedures which should include the following topics:

- A) Scope of procedure used.
- B) Description of the study target.
- C) Variable to be determined.
- D) Equipment necessary, reference materials (e.g., voucher specimens\*), taxonomic literature used.
- E) Specification of working conditions required for effective sampling.
- F) Description of procedure/method with respect to the following aspects:
  - 1) Sampling and sample treatment, labelling, handling, transport and storage of samples, preparation for laboratory analysis;
  - 2) Instrument verification and calibration;

3) Recording of data;

4) Safety aspects.

G) Criteria to adopt or reject results/measurements.

H) Data to be recorded and methods for their analysis.

I) Assessment of uncertainty of measurements.

\* Representative specimen preserved and maintained for taxonomic verification.

This proposed content of a Standard Operating Procedure has been adapted from DIN EN 45001 (Chapter 5.4.3).

An evaluation of the content of SOPs across laboratories (or countries) for specified measures, in order to ensure that inconsistencies are resolved, would therefore appear to represent a potentially useful means to improve the quality of the resultant data. In the context of JAMP guidelines, harmonization of methodology by this means should significantly reduce the risk of data incomparability before major monitoring effort is expended.

SGQAE therefore stressed the importance that all laboratories engaged in OSPAR monitoring of the relevant biological variables should, as a minimum, ensure that their sampling and analytical procedures are fully documented in the form of SOPs, and that this activity is coordinated at a national level, preferably in conjunction with advice from the relevant specialist international Working Groups (e.g., ICES Working Groups). In practice, this may operate (and have benefits) in two directions:

- 1) 'Top-down': in some circumstances, it may already be feasible to produce specifications, at an appropriate level of detail, at a national level, which may then be used as a 'blueprint' for local (within-laboratory) application.
- 2) 'Bottom-up': the gathering and then critical review of laboratory SOPs will provide the means to iron out any inconsistencies; at the same time, the exercise may be extended to produce a generic (country-wide) specification.

As a means to resolve potential future problems in data acquisition, SGQAE advised that a limited number of representative SOPs (submitted anonymously, if necessary) should be evaluated in such a way as to establish the scope, if any, for errors in data acquisition, and their likely significance, arising from variation in the content, and to permit recommendations to be made in order to rectify inconsistencies. Such a process would then act as a trigger for appropriate action elsewhere at a local or national level, which may range from simple modifications to the procedures, to the conduct of intercomparison exercises on the effectiveness of different sampling or analytical approaches.

SGQAE was of the opinion that, because of the level of detail contained in the SOPs, the evaluation referred to above was more appropriate for consideration by expert groups (e.g., within ICES), while SGQAE could play a role in the further development of a general framework for the structuring of SOPs (subject to the outcome of the above exercise), alongside guidance on the preparation of quality manuals in their entirety, and in overall quality policy.

The ACME supported the view of SGQAE that there is a need for all laboratories engaged in OSPAR monitoring of the relevant biological variables to document their sampling and analytical procedures in the form of SOPs.

### *Recommendations*

ICES recommends the production of SOPs for field and laboratory biological variables and also encourages a close collaboration between SGQAE and SGQAB, as many issues are of joint interest. Finally, ICES shares the concern of SGQAE that it is extremely important for OSPAR to ensure a wider participation in the work of SGQAE. On the basis of experience gained in the Baltic Sea work, ICES emphasized the need for laboratories participating in the OSPAR monitoring programme on eutrophication parameters also to participate in the work of SGQAE: to have the QA guidelines accepted, the persons who will need to apply them must participate in their development. If not, there will not be a successful implementation of the guidelines.

### **5.3 Quality Assurance of 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 1998 reports of the Marine Chemistry Working Group (MCWG) and the Working Group on Phytoplankton Ecology (WGPE), and ACME deliberations.

#### *Status/background information*

Chlorophyll *a* as a biomass marker is often included in marine monitoring programmes. Slightly different methodologies are, however, used to measure the concentration of chlorophyll *a*. Two ICES Working Groups, MCWG and WGPE, have worked independently on this topic during the past year. One of the reasons for their work was to improve the quality and comparability of chlorophyll *a* data from different sources.

Each of the two Working Groups has prepared a paper on chlorophyll determinations, however, with slightly different focus. The MCWG paper mainly concentrates on the methodology for the routine determination of chlorophyll *a*. The paper was not intended as an analytical manual, but was meant 1) to summarize the background on the ecological importance of chlorophyll *a*, 2) to summarize the analytical principles and the procedures for its determination, and 3) to point out the critical and controversial aspects of the protocol. The WGPE paper has a wider focus covering not only chlorophyll *a*, but also other pigments. Both reports benefited from the recent publication of the SCOR Working Group 78 report on 'Phytoplankton pigments in oceanography' (Jeffrey *et al.*, 1997). This monograph served as a reference for the recommendations given in the two documents, although they were produced independently.

The main points of discussion in both reports concern the storage time of filtered samples at a temperature of  $-20^{\circ}\text{C}$  and the solvent used to extract the chlorophyll compounds.

#### *Storage time*

For the routine spectrophotometric or fluorometric determination of chlorophyll *a* only, Table 10.4 in the UNESCO monograph (Jeffrey *et al.*, 1997) shows that recovery after 60 days was  $100 \pm 10\%$  of the initial concentration, for both artificial mixtures of microalgae and natural populations. Consequently, both Working Groups considered that a storage time of the filtered samples of up to 2 months at  $-20^{\circ}\text{C}$  could reasonably be recommended for chlorophyll *a* samples. However, this does not apply to all the carotenoid and chlorophyll pigments tested, for which only a few days' storage can be recommended.

The ACME noted, however, that the HELCOM procedure for chlorophyll *a* determinations recommends that the filters not be stored or, if this is not possible, that they be kept in a desiccator at  $-20^{\circ}\text{C}$  for no more than 24 hours.

#### *Extraction solvent*

Although the SCOR work shows that 90 % acetone does not completely extract chlorophyll *a* from a few specific algae, tests on natural populations gave satisfactory results when compared with the reference solvent (dimethylformamide). Therefore, given that extinction coefficients are well established in 90 % acetone, and that this solvent has a low toxicity, the SCOR group recommended it (with grinding of the filter) for routine spectrophotometric and fluorometric determinations. This recommendation has been followed by WGPE and MCWG.

The HELCOM procedure for chlorophyll *a* recommends the use of 96 % ethanol. However, despite potential advantages of that method, the ACME agreed that it cannot presently be recommended for the ICES community since the complete methodology using this solvent is not available in the international literature.

#### *Need for further research or additional data*

There is a need for the HELCOM procedure using 96 % ethanol for the extraction of chlorophyll to be documented and made available in the international literature. Further work should be undertaken to compare the relative merits of the two extraction procedures when the ethanol method has been fully documented.

#### **Reference**

Jeffrey, S.W., Mantoura, R.F.C., and Wright, S.W. (Eds.) 1997. Phytoplankton pigments in oceanography: Guidelines to modern methods. UNESCO Publishing, ISBN 92-3-103275-5.

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

##### **5.4.1 Biological Effects Quality Assurance in Monitoring Programmes (BEQUALM)**

#### *Request*

There is no specific request; this is part of the continuing ICES work of coordinating quality assurance activities related to biological effects techniques and reporting on the results and their implications for monitoring programmes, such as the OSPAR Joint Assessment and Monitoring Programme (JAMP).

#### *Source of the information presented*

The 1997 and 1998 reports of the Working Group on Biological Effects of Contaminants (WGBEC), the 1998 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 above-mentioned reports providing information on progress with respect to the development of QA procedures for biological effects monitoring techniques.

Over the past two years, WGBEC has discussed the need to construct a quality assurance/quality control (QA/QC) infrastructure for marine biological effects techniques, similar to the QUASIMEME programme for chemical measurements. As an underpinning for this, a set of

QA/QC guidelines has been developed, mainly for those techniques recommended for inclusion in the biological effects component of the OSPAR Joint Assessment and Monitoring Programme (JAMP) (ICES, 1996).

In late 1997, a bid for funding under the EU Standards, Measurement and Testing Programme (SMT) entitled 'Biological Effects Quality Assurance in Monitoring Programmes' (BEQUALM) was prepared by scientists identified by WGBEC as project partners due to their expertise. The proposal was submitted to the European Commission (EC) by the project coordinator, Dr P. Matthiessen, CEFAS, Burnham Laboratory, UK (also Chairman of WGBEC), and was finally accepted by the EC with minor amendments in May 1998.

The programme will focus on the following major objectives:

Objective 1: Development of appropriate reference materials or type collections.

Objective 2: Development of an infrastructure for assessing the comparability of data from individual laboratories.

Objective 3: Demonstration that biological effects analyses are under statistical control and are of known quality.

To achieve these objectives, the project will set up a measurement infrastructure for a group of existing biological effects monitoring techniques (see Table 5.4.1.1). This infrastructure will organize the circulation of reference materials, the conduct of intercomparison exercises, and the coordination of practical training workshops. The ultimate aim is to develop a QA system over a period of three years which can then become self-financing, perhaps as part of the QUASIMEME quality assurance programme for chemical monitoring procedures in the marine environment (which already includes some chemical aspects of the OSPAR JAMP biological effects suite, but does not currently overlap with BEQUALM except with respect to imposex/intersex in gastropods).

Although the BEQUALM proposal has grown out of the needs of the JAMP, it seeks to implement an international QA programme for biological effects measurements which will be relevant to the needs of marine monitoring organizations throughout Europe. The project does not intend to be prescriptive about the methods to be used, but will instead be results-driven, i.e., differences in the techniques used are irrelevant providing that they give the correct results. In essence, BEQUALM will develop the tools required for a biological effects QA programme (e.g., reference materials, agreed procedures, robust statistical methods) and apply them via an infrastructure (see Table 5.4.1.1) composed of a Central Steering Group and Expert

Laboratories (each being a partner in the programme) which will run intercalibrations and training workshops for participating laboratories, and assess compliance with established norms.

The ultimate output of the BEQUALM programme will be as follows:

- 1) an agreed set of protocols for the biological monitoring methods included in the programme;
- 2) an agreement on acceptable limits of variation for the methods in question;
- 3) a system for monitoring the output of participating laboratories and assessing their compliance with agreed quality standards;
- 4) an understanding of the costs implicit in running such a QA programme;
- 5) finally, BEQUALM will have provided a QA infrastructure which can be continued as a self-financing service to European marine monitoring laboratories and programmes.

The work programme will be divided into tasks among partners according to their expertise, as shown in Table 5.4.1.1.

The biological effects techniques included in the programme cover many of the techniques of current

interest to European monitoring programmes, except for the measurement of PAH bile metabolites, which was originally considered too chemistry-oriented to fit into the programme, and the measurement of oxidative enzyme induction, for which an expert laboratory willing to participate could not be found in the time available.

The ACME supported the views of WGBEC and WGPDMO that there is a clear requirement for the implementation of a QA infrastructure for biological effects monitoring techniques and it, therefore, appreciated that the proposal has been accepted by the EC for funding. The ACME emphasized that the BEQUALM project constitutes an important continuation of long-term efforts by ICES to develop strategies and methodologies, including QA procedures, to be used in marine environmental monitoring programmes.

The ACME endorsed the view of WGBEC that there should be some flexibility in the BEQUALM project to incorporate more collaborators from other European expert laboratories in the work programme, as already planned for liver histopathology, liver nodules and external fish diseases as well as for phytoplankton assemblage analysis. In particular, it is desirable to include the measurement of PAH bile metabolites, via a Dutch expert laboratory, and oxidative enzyme induction, possibly via a French laboratory.

Table 5.4.1.1. Work programme and partnership details of the BEQUALM project.

Partner	Tasks
1-9	(1) Central Steering Group meetings
1 CEFAS Burnham, UK	(2) Water and sediment bioassays (2.1) Sediment bioassays (2.2) Water column bioassays
2 NIVA, Norway	(3) Metallothionein induction (4) ALA-D activity
3 Institute of Applied Environmental Research, Sweden	(5) DNA adduct induction
4 FRS Marine Laboratory, Aberdeen, UK	(6) P4501A induction (7) Imposex/intersex measurements
5 Plymouth Marine Laboratory, UK	(8) Lysosomal stability
6 CEFAS Weymouth, UK	(9) Liver histopathology, liver nodules and external fish disease measurements
7 Institute of Coastal Research, Sweden	(10) Fish reproductive success
8 FTZ Westküste, Germany	(11) Phytoplankton assemblage analysis (11.1) Phytoplankton (11.2) Chlorophyll
9 Institut für Meereskunde, Germany	(12) Benthic community analysis

It was further pointed out that the QA/QC infrastructure developed within BEQUALM has to reflect the current demands defined by the monitoring organizations, but it should remain flexible enough to encompass new validated biological effects techniques and methods as they become available.

### *Recommendations*

ICES recommends that laboratories in ICES Member Countries conducting biological effects monitoring programmes should be made aware of the BEQUALM project and should be encouraged to participate in the QA procedures planned. ICES further emphasized that there is also a need for establishing a QA infrastructure for other biological effects methods which are not covered by the BEQUALM project but which are commonly used in national monitoring programmes.

### **Reference**

ICES. 1996. Report of the Advisory Committee on the Marine Environment, 1996. ICES Cooperative Research Report, 217: 47-48; 105-119.

#### **5.4.2 Interlaboratory comparison of scope-for-growth (SFG) measurements**

##### *Request*

There is no specific request; this is part of the continuing ICES work of coordinating quality assurance activities and reporting on the results and their implications for monitoring programmes.

##### *Source of the information presented*

The 1998 report of the Working Group on Biological Effects of Contaminants (WGBEC) and ACME deliberations.

##### *Status/background information*

In 1993, the ACME stressed the importance of developing simple standard methods for biological effects monitoring and recognized the potential in the measurement of scope-for-growth (SFG) in mussels. The ACME fully endorsed the need for conducting an intercomparison exercise on this measurement, but progress has been slow mainly due to a lack of funding for the exercise at the coordinating institute, the Plymouth Marine Laboratory (PML), UK.

This exercise has now been conducted in a manner similar to that of interlaboratory comparisons of chemical analytical procedures, namely by sending mussels (*Mytilus edulis*) to the participants. A draft report

outlining the objectives, the strategy/approach, and a preliminary interpretation of the results was presented to and discussed at the 1998 meeting of WGBEC.

In summary, the manual/standard operating procedures for SFG measurement, together with data sheets, were prepared and sent to eleven participating laboratories in six European countries (Norway, Sweden, the Netherlands, England, Portugal, and Greece). In June 1997, mussels were collected from a clean and a polluted site and shipped by air to participants for SFG measurement. To date, results from ten laboratories have been received. All laboratories recorded a significant difference between the sites (i.e., clean >> polluted). However, the results showed slight differences in SFG and they could be divided into two groups based on the method used to measure feeding rate. Previous studies have shown no differences between these two methods, but on this occasion mussels measured in a 'closed' beaker system gave slightly higher rates than those in an 'open-flow' system. The likely explanation for this difference appears to be the lower food ration (algae) levels supplied to the mussels during the 24-hour period prior to measuring feeding rate in the closed beaker system. This hypothesis is being tested in new studies (spring 1998). In addition, there were two laboratories that were distinct outliers, one in each group, and these participating laboratories had the least experience in measuring SFG (assessed at the start by means of a questionnaire). The high outlier cannot readily be explained and the laboratory has not yet responded to specific technical queries. The reason for the low outlier is probably the poorer water quality in the region of this laboratory. This hypothesis will also be tested in future studies.

In the light of the results of this interlaboratory comparison, WGBEC requested an assessment of the possible advantages and disadvantages of standardizing on the slightly simpler 'closed' system for measuring clearance rate. This will be done by PML during 1998.

Finally, the reasons for measuring SFG rather than simply the feeding/clearance rate of mussels were also presented and discussed. The most important reason is to remove the effect of seasonal cycles in feeding and respiration rates, thus providing a near-absolute measurement (i.e., SFG) which allows a direct comparison of mussels from sites over a large spatial scale or latitudinal gradient. In addition, SFG provides an integrated measure of whole animal performance incorporating several primary mechanisms of toxicity. The benefits of determining SFG can be achieved with minimal additional costs or effort.

WGBEC therefore concluded that the SFG interlaboratory comparison has been a very successful exercise, particularly in relation to many previous

interlaboratory comparisons of other biological effects measurements. Many field studies have now been published and they clearly demonstrate that SFG, when combined with the measurement of contaminant levels in bivalves, is able to provide a sensitive and robust technique for pollution assessment over small and large spatial scales (i.e., the North Sea). It not only detects and quantifies pollution impact, but also provides a means of identifying the causes through quantitative toxicological interpretation of contaminant levels in mussel tissues (using a database and quantitative structure-activity relationships (QSARs)).

However, the ACME stressed that differences in physiological activities resulting from genetically different mussel populations may account for non-pollution-driven differences in SFG. For these reasons, the ACME recommends that careful consideration be given regarding the differences between the populations, and the age and the size of the animals, prior to designing a strategy for monitoring. For intercomparison exercises, the best approach would be to expose one population to a selected contaminant, or a mixture of contaminants, at different concentration levels, in order to perform a reliable SFG exercise.

#### *Need for further research or additional data*

The ACME identified factors that may influence the outcome of SFG and considered that there is a need for further research on possible effects of genetic differences in mussel populations on SFG.

The ACME also encouraged the conduct of further work on the following issues:

- 1) a comparison between clearance rates measured in both 'open-flow' and 'closed' systems after 24 hours under conditions of normal high food rations and low food rations (i.e., 1/30th) for the mussels;
- 2) the influence of the quality of the local sea water on intercomparison results;
- 3) assessment of the possible advantages and disadvantages of standardizing on the 'closed' system or beaker method for measuring clearance rate.

### **5.5 Quality Assurance of Chemical Measurements in the Baltic Sea**

#### *Request*

Item 2 of the 1998 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 1998 reports of the ICES/HELCOM Steering Group on Quality Assurance of Chemical Measurements in the Baltic Sea (SGQAC) and the Marine Chemistry Working Group (MCWG), and ACME deliberations.

#### *Status/background information*

The ACME reviewed the work of the ICES/HELCOM Steering Group on Quality Assurance of Chemical Measurements in the Baltic Sea (SGQAC) and the relevant sections of the 1998 Marine Chemistry Working Group report and noted that the general QA Guidelines for Chemical Measurements had been further reviewed and amended by SGQAC. In addition, further progress by SGQAC in the development of Technical Annexes to the Guidelines was noted. This includes:

- Annex H 'Technical notes on the determination of trace metals in sea water', that was accepted by the ACME in 1997, has been amended to include mercury.
- A new Annex J 'Technical notes on the determination of total mercury in marine biota by cold vapour atomic absorption spectrometry' has been completed.

The ACME accepted the revisions to the general section of the Guidelines on Quality Assurance of Chemical Measurements in the Baltic Sea and the two technical annexes (Annexes H and J) for transmission to HELCOM for inclusion in the integrated Baltic Monitoring Programme COMBINE manual.

The ACME noted that some technical annexes still need further development and advised the following:

- 1) In the development of technical notes on the analysis of chlorinated biphenyls (CBs) and organochlorine pesticides (OCPs) in sea water, SGQAC should use the experience of MCWG in preparing the final document.
- 2) In the development of technical notes on the determination of polycyclic aromatic hydrocarbons (PAHs) in sea water and biota, for the biota aspects SGQAC should use the Guidelines for the Determination of Polycyclic Aromatic Hydrocarbons (PAHs) in Biota contained in Annex 1 of this report (see also Section 4.2, above).

The ACME further noted that additional SGQAC activities include the development of an introduction to measurement uncertainty in chemical analysis, and the preparation of a questionnaire on analytical performance.

The ACME expressed its appreciation for the development of these comprehensive QA guidelines on chemical measurements. The ACME was of the opinion



that the procedure for measuring organic contaminants in sea water is of broader value and can be adopted for other purposes, e.g., for monitoring riverine inputs.

#### *Need for further research or additional data*

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

#### *Recommendations*

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

ICES agreed that the portions of the Guidelines on Quality Assurance of Chemical Measurements in the Baltic Sea that have been completed in 1998 (the revised Annex H 'Technical notes on the determination of trace metals in sea water' and the new Annex J 'Technical notes on the determination of mercury in marine biota by cold vapour atomic absorption spectrometry') should be forwarded to HELCOM for use in its monitoring programme.

### **5.6 Certified Reference Materials Available for Routine Monitoring of Organic Contaminants in the Marine Environment**

#### *Request*

There is no specific request; this is part of continuing ICES work relevant to quality assurance of marine monitoring.

#### *Source of the information presented*

The 1998 report of the Marine Chemistry Working Group (MCWG) and ACME deliberations.

#### *Status/background information*

Tables of information on certified reference materials (CRMs) available for use in marine monitoring were discussed at the 1998 MCWG meeting. The tables contain an overview of information on reference materials that are currently available for use in the routine monitoring of organic contaminants in the marine environment. The collated information covers biota (Table 5.6.1) and sediments (Table 5.6.2) for marine and fresh waters.

The following comments apply to the tables:

- values preceded by an asterisk (\*) are non-certified, all others are certified;
- certified calibration materials and standards were not included;
- the list does not purport to be complete and all CRMs listed may not be commercially available;
- users of CRMs should consult vendors for full and accurate information relating to individual CRMs;
- methyl mercury is not considered as an organic contaminant for this list.

The tables were discussed by MCWG, which made a number of further comments, as follows:

- 1) There is a lack of CRMs for some determinands in marine matrices.
- 2) While marine matrices are preferred as CRMs for marine monitoring programmes, freshwater sediments and biota may be suitable in the absence of appropriate marine-based materials.
- 3) The current tables are incomplete and will be updated for MCWG 1999.
- 4) There is sometimes a lack of information on CRMs. It can be difficult to compare CRMs used for routine monitoring due to a lack of information being presented on the method(s) used for the determination of assigned values, acceptable ranges, and the associated uncertainty.

The ACME also noted information about materials in preparation, as follows:

Code	Producer	Country	Analyte/matrix
LGC6114	LGC	UK	PCBs in marine sediment
LGC6156	LGC	UK	TBT (and metals) in marine sediment
HS-4B HS-3B	NRC	Canada	PAHs in marine sediment

The ACME agreed that the list of certified reference materials should be updated annually.

**Table 5.6.1.a.** Reference materials for organic contaminants in biota.

Code	SRM-1974a	SRM-1588	SRM-1945	MA-B-3/OC	MA-A-3/OC	MA-A-1/OC	IAEA-142
Organization	SRM-NIST	SRM-NIST	SRM-NIST	IAEA	IAEA	IAEA	IAEA
Country of origin	USA	USA	USA				
Matrix	Mussel tissue	Cod liver oil	Whale blubber	Garpiki tissue	Shrimp homogenate	Copepoda	Mussel homogenate
Units	$\mu\text{g kg}^{-1}$	$\text{ng g}^{-1}$	$\mu\text{g kg}^{-1}$	$\text{ng g}^{-1}$	$\text{ng g}^{-1}$	$\text{ng g}^{-1}$	$\text{ng g}^{-1}$ unless stated
expressed as	Dry weight		Wet weight	Dry weight	Dry weight	Dry weight	
[ $\pm$ ] expressed as	$\pm 95\%$ CI	$\pm 95\%$ CI	$\pm 95\%$ CI				(95 % CI of median)
Units of issue	3 x 15 g	5 x 1.2 ml/ ampoule	Set 2, 15 g/ ampoule	35 g	35 g	30 g	30 g
Form				Freeze-dried	Freeze-dried	Freeze-dried	Freeze-dried
		References 1,2,3	Reference 1				
<b>HYDROCARBONS</b>							
Resolved aliphatics							9.2 (6.2–16) $\mu\text{g g}^{-1}$
Unresolved aliphatics							100 (61–100) $\mu\text{g g}^{-1}$
n-C <sub>17</sub>							670 (510–910)
Pristane							170 (90–240)
Phytane							120 (50–180)
Sum alkanes (C <sub>14</sub> –C <sub>34</sub> )							5.2 (3.3–8.4) $\mu\text{g g}^{-1}$
Total aromatics							42 (30–48) $\mu\text{g g}^{-1}$
Unresolved aromatics							27 (25–48) $\mu\text{g g}^{-1}$
Acenaphthene							3.4 (1.9–7.1)
Anthracene	6.07 $\pm$ 1.76						4.3 (1.8–6.4)
Benz[a]anthracene	32.5 $\pm$ 4.8						15 (12–17)
Benzo[b]fluorathene	46.4 $\pm$ 3.9						19 (14–30)
Benzo[k]fluoranthene	20.2 $\pm$ 1.0						9.7 (6.0–13)
Benzo[a]pyrene	15.63 $\pm$ 0.77						3.5 (2.9–5)
Benzo[e]pyrene	84.0 $\pm$ 3.2						27 (22–30)
Benzo[ghi]perylene	22.0 $\pm$ 2.3						9.9 (8.3–13)
Biphenyl							7 (4.8–5.2)
Chrysene	44.2 $\pm$ 2.6						32 (21–46)
Fluoranthene	163.6 $\pm$ 10.3						73 (59–94)
Indeno[1,2,3-cd]pyrene	14.2 $\pm$ 2.9						6.5 (5.5–7.9)
1-Methylphenanthrene							20 (16–24)
2-Methylnaphthalene							23 (20–29)
Naphthalene	23.5 $\pm$ 4.4						

References from which additional information on OCs and PCBs can be obtained:

- 1) Hillery *et al.* 1995. Journal of High Resolution Chromatography, 18: 89.
- 2) Schantz *et al.* 1992. Chemosphere, 24(12): 1687.
- 3) Schantz *et al.* 1996. Chemosphere, 33(7): 1369.

Table 5.6.1.a. Continued.

Code	SRM-1974a	SRM-1588	SRM-1945	MA-B-3/OC	MA-A-3/OC	MA-A-1/OC	IAEA-142
Organization	SRM-NIST	SRM-NIST	SRM-NIST	IAEA	IAEA	IAEA	IAEA
Units	$\mu\text{g kg}^{-1}$	$\text{ng g}^{-1}$	$\mu\text{g kg}^{-1}$	$\text{ng g}^{-1}$	$\text{ng g}^{-1}$	$\text{ng g}^{-1}$	$\text{ng g}^{-1}$ unless stated
<b>HYDROCARBONS</b>							
Perylene	$7.67 \pm 0.35$						7.3 (6.1–9.3)
Phenanthrene	$22.2 \pm 1.7$						60 (41–82)
Pyrene	$151.6 \pm 5.7$						57 (39–81)
Triphenylene	$50.7 \pm 6.0$						
<b>PESTICIDES</b>							
Hexachlorobenzene		$148 \pm 21$	$32.9 \pm 1.7$	1.5 (0.9–2.1)	0.32 (0.2–0.44)		0.48 (0.32–0.70)
$\alpha$ -HCH		$86 \pm 19$	$16.2 \pm 3.4$	10 (0–24)	15	10 (1.6–18.4)	0.43 (0.21–0.65)
$\gamma$ -HCH		*25.5	$3.30 \pm 0.81$	3.4 (0–7.2)	3.2 (0–6.7)	8.2 (1.9–14.5)	0.97 (0.5–1.5)
Aldrin				1.8 (0.1–3.5)	0.7 (0.2–1.2)	14 (0–33)	
<i>trans</i> -Chlordane		$50 \pm 13$					
<i>cis</i> -Chlordane	*16	$158 \pm 8$	$46.9 \pm 2.8$				
Heptachlor epoxide		*33	$10.8 \pm 1.3$				
<i>trans</i> -Nonachlor	*15	$209 \pm 11$	$231 \pm 11$				
Dieldrin		$150 \pm 12$	* $37.5 \pm 3.9$				
<i>cis</i> -Nonachlor	*7	*94.4	$48.7 \pm 7.6$				
Oxychlordane			$19.8 \pm 1.9$				
2,4'-DDE	*5	*22.0	$12.28 \pm 0.87$				
4,4'-DDE	*50	$641 \pm 62$	$445 \pm 37$	160 (50–270)	4.7 (1.3–8.1)	6.1 (1.5–17.1)	8.2 (5.4–10)
2,4'-DDD	*15	*36.3	$18.1 \pm 2.8$				
4,4'-DDD	*45	$277 \pm 15$	$133 \pm 10$	46 (13–59)	0.81 (0.05–1.57)	5.5 (0–11)	4.3 (2.8–5.8)
2,4'-DDT	*9	$156 \pm 5$	$106 \pm 14$				
4,4'-DDT	*4	$529 \pm 45$	$245 \pm 15$	65 (16–114)	3.2 (0–6.7)	8.3 (3.4–13.2)	2.0 (1.0–3.1)
<b>PCBs</b>							
PCB18	*25		$4.48 \pm 0.88$				
PCB28	*80	*28.3	* $14.1 \pm 1.4$				1.3 (0.82–2.4)
PCB31		*8.3	* $3.12 \pm 0.69$				0.9 (0.57–1.4)
PCB44	*80	*35.1	$12.1 \pm 1.4$				
PCB49	*90	*29.8	$20.8 \pm 2.8$				2.3 (1.6–3.4)
PCB52	*120	*83.3	$43.6 \pm 2.5$				
PCB66	*100	*54.8	$23.6 \pm 1.6$				1.8 (1.1–2.0)
PCB95	*80	*36.5	$33.8 \pm 1.7$				
PCB99	*70	*213	$45.4 \pm 5.4$				4.3 (2.7–5.4)

\*Non-certified value.

Table 5.6.1.a. Continued.

Code	SRM-1974a	SRM-1588	SRM-1945	MA-B-3/OC	MA-A-3/OC	MA-A-1/OC	IAEA-142
Organization	SRM-NIST	SRM-NIST	SRM-NIST	IAEA	IAEA	IAEA	IAEA
Units	$\mu\text{g kg}^{-1}$	$\text{ng g}^{-1}$	$\mu\text{g kg}^{-1}$	$\text{ng g}^{-1}$	$\text{ng g}^{-1}$	$\text{ng g}^{-1}$	$\text{ng g}^{-1}$ unless stated
PCBs							
PCB101	*130	129 $\pm$ 5	65.2 $\pm$ 5.6	61 (27–94)			3.1 (2.7–5.0)
PCB105	*50	*60.2	30.1 $\pm$ 2.3				1.4 (1.1–2.1)
PCB110	*130	*75.8	23.3 $\pm$ 4.0				
PCB118	*130	*176	74.6 $\pm$ 5.1				3 (2.5–7.1)
PCB126							
PCB128	*20	*47.1	23.7 $\pm$ 1.7				1.5 (2.5–4.1)
PCB138	*130	261 $\pm$ 29	131.5 $\pm$ 7.4				5.6 (4.2–7.1)
PCB149	*90	*105	106.6 $\pm$ 8.4				3.7 (2.8–6.9)
PCB151	*25	*55.2	28.7 $\pm$ 5.2				
PCB153	*150	276 $\pm$ 40	213 $\pm$ 13	120 (86–154)			6.4 (4.9–8.7)
PCB156	*8		10.3 $\pm$ 1.1				0.5 (0.28–0.6)
PCB170	*6	45 $\pm$ 5	40.6 $\pm$ 2.6				
PCB180	*16	107 $\pm$ 4	106.7 $\pm$ 5.3				0.75 (0.55–1.4)
PCB187	*35	*35.3	105.1 $\pm$ 9.1				2.4 (2.1–3.7)
PCB194			39.6 $\pm$ 2.5				
PCB 206			31.1 $\pm$ 2.7				
PCB 209			10.6 $\pm$ 1.1				
Total PCBs							
Aroclor 1242						120 (67–173)	
Aroclor 1254				400 (170–630)	33 (0–67)		
Aroclor 1260				390 (140–640)		140 (70–210)	
DIOXINS/FURANS							
OCDF		*1.00					
1,2,7-TCDD		*0.32					
1,2,3,4-TeCDD		*0.38					
2,3,7,8-TCDD		*0.21					
1,2,3,6,7,8,-HxCDD		*0.39					
1,2,3,7,8,9-HxCDD		*0.22					
OCDD		*1.01					
Alpha-tocopherol		112					

\*Non-certified value.

**Table 5.6.1.b.** Reference materials for organic contaminants in biota.

Code	CARP-1	CRM 349	CRM 350	NIES11	MUS-1B	MUS-2	EDF-2525	EDF-2526
Organization	BCR	BCR	NIES	NRC	NRC	NRC	CIL	CIL
Country of origin	EC	EC	Japan	Canada	Canada	Canada	USA	USA
Matrix	Cod liver oil	Cod liver oil	Fish tissue	Common carp	Mussel tissue	Mussel Tissue	Fish	Fortified fish
Units	$\mu\text{g kg}^{-1}$	$\mu\text{g kg}^{-1}$	$\mu\text{g g}^{-1}$	$\text{ng kg}^{-1}$	$\mu\text{g g}^{-1}$	$\mu\text{g g}^{-1}$	$\text{ng kg}^{-1}$	$\text{ng kg}^{-1}$
expressed as				Wet weight			Wet weight	Wet weight
[ $\pm$ ] expressed as				$\pm 95\%$ CI				
Units of issue	2 g	2g	20 g	6 $\times$ 9 g	4 $\times$ 8 g	4 $\times$ 4 g	Set 1, 10 g/ ampoule	Set 1, 10 g/ ampoule
Form				Slurry			Slurry	Slurry
<b>PCBs</b>								
PCB28	68	22.5						
PCB44	*75	*44						
PCB52	124 $\pm$ 32	149	62					
PCB77							2376 $\pm$ 672	523 $\pm$ 45
PCB101		370	165					
PCB 101/90	124 $\pm$ 37							
PCB105	54 $\pm$ 24							144 $\pm$ 63
PCB118	132 $\pm$ 60	456	143					321 $\pm$ 70
PCB126							834 $\pm$ 277	521 $\pm$ 70
PCB128		*104	*41					
PCB138/163		*765	*274					
PCB138/163/164	102 $\pm$ 23							
PCB153	83 $\pm$ 39	938	317					
PCB169							181 $\pm$ 264	515 $\pm$ 44
PCB170/190	22 $\pm$ 8							
PCB180	46 $\pm$ 14	280	73					
PCB187/182	36 $\pm$ 16							
PCB194		*38						
<b>DIOXINS/FURANS</b>								
2,3,7,8-TCDF	11.9 $\pm$ 2.7						22 $\pm$ 1.6	17 $\pm$ 1.5
1,2,3,7,8-PCDF	5.0 $\pm$ 2.0						4.9 $\pm$ 0.56	40 $\pm$ 3.7
2,3,4,7,8-PCDF							14 $\pm$ 1.3	38 $\pm$ 3.5
1,2,3,4,7,8-HxCDF							8.2 $\pm$ 3.7	80 $\pm$ 8.4
1,2,3,6,7,8-HxCDF							2.7 $\pm$ 1.2	63 $\pm$ 5.5
1,2,3,7,8,9-HxCDF							0.76 $\pm$ 0.35	58 $\pm$ 7.0
2,3,4,6,7,8-HxCDF							2.3 $\pm$ 1.9	60 $\pm$ 5.5
1,2,3,4,6,7,8-HpCDF							4.4 $\pm$ 6.0	83 $\pm$ 9.2
1,2,3,4,7,8,9-HpCDF							0.63 $\pm$ 0.23	73 $\pm$ 7.7
OCDF							2.6 $\pm$ 1.3	190 $\pm$ 22
2,3,7,8-TCDD	6.6 $\pm$ 0.6						17 $\pm$ 1.4	19 $\pm$ 1.4
1,2,3,7,8-PCDD	4.4 $\pm$ 1.1						4.0 $\pm$ 0.57	40 $\pm$ 3.0

\*Non-certified value.

Table 5.6.1.b. Continued.

Code	CARP-1	CRM 349	CRM 350	NIES11	MUS-1B	MUS-2	EDF-2525	EDF-2526
Organization	BCR	BCR	NIES	NRC	NRC	NRC	CIL	CIL
Units	$\mu\text{g kg}^{-1}$	$\mu\text{g kg}^{-1}$	$\mu\text{g g}^{-1}$	$\text{ng kg}^{-1}$	$\mu\text{g g}^{-1}$	$\mu\text{g g}^{-1}$	$\text{ng kg}^{-1}$	$\text{ng kg}^{-1}$
<b>DIOXINS/FURANS</b>								
1,2,3,4,7,8,-HxCDD	$1.9 \pm 0.7$						$0.77 \pm 0.27$	$60 \pm 4.8$
1,2,3,6,7,8,-HxCDD	$5.6 \pm 1.3$						$3.0 \pm 1.2$	$56 \pm 4.8$
1,2,3,7,8,9-HxCDD	$0.7 \pm 0.4$						$0.79 \pm 0.26$	$60 \pm 4.4$
1,2,3,4,6,7,8-HpCDD	$6.5 \pm 1.8$						$1.4 \pm 0.53$	$76 \pm 5.9$
OCDD	$6.3 \pm 1.9$						$7.2 \pm 3.7$	$192 \pm 14$
<b>ANTIFOULING</b>								
Triphenyltin (as chloride)				*6.3				
Tributyltin (as chloride)				1.3				
<b>ALGAL TOXINS</b>								
Domoic Acid					38.3			
Okadaic acid						11		
Dinophysistoxin-1 (DTX-1)						1		

\*Non-certified value.

Table 5.6.2.a. Reference materials for organic contaminants in marine sediments.

Code	SES-1 <sup>1</sup>	HS-3	HS-4	HS-5	HS-6
Organization	NRC-CNRC	NRC-CNRC	NRC-CNRC	NRC-CNRC	NRC-CNRC
Country of origin	Canada	Canada	Canada	Canada	Canada
Matrix	Estuarine sediment	Harbour sediment	Harbour sediment	Harbour sediment	Harbour sediment
Units	$\text{mg kg}^{-1}$	$\text{mg kg}^{-1}$	$\text{mg kg}^{-1}$	$\text{mg kg}^{-1}$	$\text{mg kg}^{-1}$
Expressed as	Dry weight	Dry weight	Dry weight	Dry weight	Dry weight
( $\pm$ ) expressed as		90 % CI	90 % CI	90 % CI	90 % CI
Units of issue	200 g	200 g	200 g	200 g	200 g
Form	Freeze-dried	Freeze-dried	Freeze-dried	Freeze-dried	Freeze-dried
<b>PAHs</b>					
Acenaphthene	*7.21	$4.5 \pm 1.5$	<0.15	$0.23 \pm 0.10$	$0.23 \pm 0.07$
Acenaphthylene		$0.3 \pm 0.1$	<0.15	<0.15	$0.19 \pm 0.05$
Anthracene	*1.63	$13.4 \pm 0.5$	$0.14 \pm 0.5$	$0.38 \pm 0.15$	$1.1 \pm 0.4$
Benz[a]anthracene	*1.31	$14.6 \pm 2.0$	$0.53 \pm 0.05$	$2.9 \pm 1.2$	$1.8 \pm 0.3$
Benzo[b]fluoranthene		$7.7 \pm 1.2$	$0.70 \pm 0.15$	$2.0 \pm 0.15$	$2.8 \pm 0.6$
Benzo[k]fluoranthene		$2.8 \pm 2.0$	$0.36 \pm 0.05$	$1.0 \pm 0.4$	$1.43 \pm 0.15$
Benzo[a]pyrene	*1.21	$7.4 \pm 3.6$	$0.65 \pm 0.08$	$1.7 \pm 0.8$	$2.2 \pm 0.4$
Benzo[ghi]perylene	*1.21	$5.0 \pm 2.0$	$0.58 \pm 0.22$	$1.3 \pm 0.3$	$1.78 \pm 0.72$
Chrysene	*1.32	$14.1 \pm 2.0$	$0.65 \pm 0.08$	$2.8 \pm 0.9$	$2.0 \pm 0.3$
Dibenz[a,h]anthracene	*1.30	$1.3 \pm 0.5$	$0.12 \pm 0.05$	$0.2 \pm 0.1$	$0.49 \pm 0.16$
Fluoranthene	*1.58	$60 \pm 9$	$1.25 \pm 0.10$	$8.4 \pm 2.6$	$3.54 \pm 0.65$
Fluorene	*1.42	$13.3 \pm 3.1$	<0.15	$0.4 \pm 0.1$	$0.47 \pm 0.12$

Table 5.6.2.a. Continued.

Code	SES-1 <sup>1</sup>	HS-3	HS-4	HS-5	HS-6
Organization	NRC-CNRC	NRC-CNRC	NRC-CNRC	NRC-CNRC	NRC-CNRC
PAHs					
Indeno[1,2,3- <i>cd</i> ]pyrene	*1.28	5.4 ± 1.3	0.51 ± 0.15	1.3 ± 0.7	1.95 ± 0.58
Naphthalene	*3.62	9.0 ± 0.7	< 0.15	0.25 ± 0.07	4.1 ± 1.1
Phenanthrene	*1.37	85 ± 20	0.68 ± 0.08	5.2 ± 1.0	3.0 ± 0.6
Pyrene	*4.09	39 ± 9	0.94 ± 0.12	5.8 ± 1.8	3.0 ± 0.6

<sup>1</sup>Spiked concentrations given. Non-certified material.

\*Non-certified value.

Table 5.6.2.b. Reference materials for organic contaminants in marine sediments.

Code	CS-1	HS-1	HS-2	SRM 1941a	EC-1	EC-3	EC-4
Organization	NRC-CNRC	NRC-CNRC	NRC-CNRC	SRM – NIST	NWRI	NWRI	NWRI
Country of origin	Canada	Canada	Canada	USA	Canada	Canada	Canada
Matrix	Harbour sediment	Harbour sediment	Harbour sediment	Marine sediment	Harbour sediment	Niagara River Plume	Harbour sediment
Units	µg kg <sup>-1</sup>	µg kg <sup>-1</sup>	µg kg <sup>-1</sup>	µg kg <sup>-1</sup>	µg g <sup>-1</sup>	ng g <sup>-1</sup>	µg g <sup>-1</sup>
Expressed as	Dry weight	Dry weight	Dry weight	Dry weight			
(±) expressed as	± SD	± SD	± SD	95 % CI			
Units of issue	200 g	200 g	200 g	50 g	100 g	100 g	100 g
Form	Freeze-dried	Freeze-dried	Freeze-dried				
PAHs							
Acenaphthene						22 ± 9	*0.032
Acenaphthylene						*25 ± 8	*0.048
Anthracene				184 ± 14	1.2	*59 ± 11	*0.124
Benz[ <i>a</i> ]anthracene				427 ± 25	8.7	312 ± 28	*0.712
Benzo[ <i>b</i> ]chrysene				99 ± 20			
Benzo[ <i>a</i> ]fluoranthene				118 ± 11			
Benzo[ <i>b</i> ]fluoranthene				740 ± 110	7.8	*505 ± 88	*0.753
Benzo[ <i>k</i> ]fluoranthene				361 ± 18	4.4	*271 ± 104	*0.560
Benzo[ <i>a</i> ]pyrene				628 ± 52	5.3	386 ± 50	*0.675
Benzo[ <i>e</i> ]pyrene				553 ± 59	5.3	450 ± 49	*0.747
Benzo[ <i>ghi</i> ]perylene				501 ± 72	4.9	*348 ± 70	*0.576
Biphenyl				*175 ± 18			
Chrysene/Triphenylene					*9.2	*458 ± 59	*1.073
Chrysene				380 ± 24			
Dibenz[ <i>a,h</i> ]anthracene				73.9 ± 9.7	*1.3	*109 ± 17	*0.241
Dibenz[ <i>a,c</i> ]anthracene				43.1 ± 3.7			
Dibenz[ <i>a,j</i> ]anthracene				74.3 ± 6.8			
Fluoranthene				981 ± 78	23.2	558 ± 46	*1.087
Fluorene				97.3 ± 8.6		*42 ± 21	*0.088
Indeno[1,2,3- <i>cd</i> ]pyrene				525 ± 67	5.7	*359 ± 36	*0.564
1-Methylphenanthrene				101 ± 27			

\*Non-certified value.

Table 5.6.2.b. Continued.

Code	CS-1	HS-1	HS-2	SRM 1941a	EC-1	EC-3	EC-4
Organization	NRC-CNRC	NRC-CNRC	NRC-CNRC	SRM – NIST	NWRI	NWRI	NWRI
Units	$\mu\text{g kg}^{-1}$	$\mu\text{g kg}^{-1}$	$\mu\text{g kg}^{-1}$	$\mu\text{g kg}^{-1}$	$\mu\text{g g}^{-1}$	$\text{ng g}^{-1}$	$\mu\text{g g}^{-1}$
PAHs							
Naphthalene				1010 $\pm$ 140		*35 $\pm$ 20	*0.058
Perylene				452 $\pm$ 58	*1.1	*195 $\pm$ 21	*0.28
Phenanthrene				489 $\pm$ 23	15.8	293 $\pm$ 33	*0.732
Pyrene				811 $\pm$ 24	16.7	436 $\pm$ 47	*1.085
Picene				80.0 $\pm$ 9.0			
Triphenylene				197 $\pm$ 11			
PESTICIDES							
Hexachlorobenzene				70 $\pm$ 25	*5.4	279 $\pm$ 33.1	*2.2
cis-Chlordane				2.33 $\pm$ 0.56			
trans-Nonachlor				1.26 $\pm$ 0.13			
Dieldrin				*1.26			
cis-Nonachlor				*2.59			
2,4'-DDE				0.73 $\pm$ 0.11			
4,4'-DDE				6.59 $\pm$ 0.56			
2,4'-DDD				*20			
4,4'-DDD				5.06 $\pm$ 0.58			
4,4'-DDT				*1.25			
PCBs							
PCB18				*1.15	*47.4		*3.7
PCB28				*9.8	*48.7		*6.8
PCB31				*6.2			
PCB44				4.80 $\pm$ 0.62	*64.7		*7.5
PCB49				9.5 $\pm$ 2.1			
PCB52				6.89 $\pm$ 0.56	*99.4		*12.5
PCB66				6.8 $\pm$ 1.4			
PCB87					*44.9		*8.3
PCB95				7.5 $\pm$ 1.1			
PCB99				4.17 $\pm$ 0.51			
PCB101		1.62 $\pm$ 0.21	5.42 $\pm$ 0.34	11.0 $\pm$ 1.6	*109.4		*22.4
PCB105				3.65 $\pm$ 0.7	*34.2		*8.1
PCB110				9.47 $\pm$ 0.85	*120.1		*29.1
PCB118				10.0 $\pm$ 1.1	*79.8		*17.8
PCB128				1.87 $\pm$ 0.32	*14.5		*4.6
PCB137					*3.8		*1.7
PCB138		1.98 $\pm$ 0.28	6.92 $\pm$ 0.52	13.38 $\pm$ 0.97	*72.0		*28.7
PCB141					*19.4		*8.3
PCB149				9.2 $\pm$ 1.1			

\*Non-certified value.



Table 5.6.2.b. Continued.

Code	CS-1	HS-1	HS-2	SRM 1941a	EC-1	EC-3	EC-4
Organization	NRC-CNRC	NRC-CNRC	NRC-CNRC	SRM – NIST	NWRI	NWRI	NWRI
Units	$\mu\text{g kg}^{-1}$	$\mu\text{g kg}^{-1}$	$\mu\text{g kg}^{-1}$	$\mu\text{g kg}^{-1}$	$\mu\text{g g}^{-1}$	$\text{ng g}^{-1}$	$\mu\text{g g}^{-1}$
PCBs							
PCB151		$0.48 \pm 0.08$	$1.37 \pm 0.07$	*2.62	*16.6		*9.4
PCB153		$2.27 \pm 0.28$	$6.15 \pm 0.67$	$17.6 \pm 1.9$	*68.2		*27.3
PCB156				$0.93 \pm 0.14$			
PCB170		$0.27 \pm 0.05$	$1.07 \pm 0.15$	$3.00 \pm 0.46$	*16.8		11.8
PCB180		$1.17 \pm 0.15$	$3.70 \pm 0.33$	$5.83 \pm 0.58$	*44.9		*26.1
PCB183					*15.2		*8.4
PCB187				*7.0			
PCB194		$0.23 \pm 0.04$	$0.61 \pm 0.07$	$1.78 \pm 0.23$	*13.1		*6.9
PCB 196		$0.45 \pm 0.04$	$1.13 \pm 0.12$				
PCB 199		$0.57 \pm 0.07$	$1.39 \pm 0.09$				
PCB201					*7.3		*8.1
PCB206				$3.67 \pm 0.87$	*7.0		*3.2
PCB209		$0.33 \pm 0.1$	$0.90 \pm 0.14$	$8.34 \pm 0.49$	*1.4		*1.6
TOTAL PCBs							
Total	$1.15 \pm 0.60$	$21.8 \pm 1.1$	$111.8 \pm 2.5$		2.00	*660 $\pm$ 54	*0.577
OTHER CHLORINATED COMPOUNDS							
1,4-dichlorobenzene					*30.9	*108.2 $\pm$ 11.8	
1,3-dichlorobenzene					*5.9	105.4 $\pm$ 17.5	*6.8
1,2-dichlorobenzene					*4.9	20.7 $\pm$ 3.1	*6.8
1,3,5-trichlorobenzene					*2.7	113.6 $\pm$ 9.6	*4.4
1,2,4-trichlorobenzene					*3.4	*141.2 $\pm$ 13.7	*6.7
1,2,3-trichlorobenzene					*2.3	8.9 $\pm$ 1.2	*1.9
1,2,4,5-tetrachlorobenzene					*3.4	*155.6 $\pm$ 17.4	*2.4
1,2,3,4-tetrachlorobenzene					*1.5	44.3 $\pm$ 5.1	*1.6
1,2,3,5-tetrachlorobenzene					*0.76	*13.6 $\pm$ 1.3	*0.34
Pentachlorobenzene					*1.7	65.4 $\pm$ 8.2	*1.9
Hexachlorobutadiene					*0.66	61.3 $\pm$ 6.9	*0.55
Octachlorostyrene					*6.0	*41.0 $\pm$ 6.2	*1.04

\*Non-certified value.

**Table 5.6.2.c.** Reference materials for organic contaminants in marine sediments.

Code	DX-1	DX-2	RM424	EDF-2513 <sup>2</sup>
Organization	NWRI	NWRI	BCR	CIL
Country of origin	Canada	Canada	European Commission	USA
Matrix	Great Lakes blend	Lake Ontario sediment	Harbour sediment	Fortified soil
Units	pg g <sup>-1</sup>	pg g <sup>-1</sup>	µg g <sup>-1</sup>	ng g <sup>-1</sup>
Expressed as				Dry weight
Units of issue	50 g	50 g	25 g	10 g
<b>OTHER CHLORINATED COMPOUNDS</b>				
2,3,7,8-TCDF	*89 ± 44	*134 ± 61	20	0.45 ± 0.03
1,2,3,7,8-PCDF	39 ± 14	46 ± 10		0.87 ± 0.04
2,3,4,7,8-PCDF	62 ± 32	88 ± 28		0.86 ± 0.06
1,2,3,4,7,8-HxCDF	714 ± 276	825 ± 348		0.88 ± 0.05
1,2,3,6,7,8-HxCDF	116 ± 37	153 ± 61		0.95 ± 0.09
1,2,3,7,8,9-HxCDF	*28 ± 42	*36 ± 45		0.82 ± 0.06
1,2,3,4,6,7,8-HpCDF	*57 ± 36	*70 ± 47		0.91 ± 0.06
1,2,3,4,7,8,9-HpCDF	2398 ± 796	3064 ± 745		1.27 ± 0.11
OCDF	137 ± 62	152 ± 84		1.12 ± 0.12
1,2,7-TCDD	7122 ± 2406	7830 ± 3087		2.25 ± 0.15
2,3,7,8-TCDD	263 ± 53	262 ± 51		0.46 ± 0.03
1,2,3,7,8-PCDD	22 ± 8	28 ± 14		0.96 ± 0.06
1,2,3,4,7,8-HxCDD	23 ± 7	25 ± 8		0.90 ± 0.06
1,2,3,6,7,8-HxCDD	77 ± 27	85 ± 33		0.87 ± 0.05
1,2,3,4,7,8,9-HxCDD	53 ± 24	58 ± 19		0.90 ± 0.06
1,2,3,4,6,7,8-HpCDD	634 ± 182	757 ± 320		1.39 ± 0.10
OCDD	3932 ± 933	4402 ± 1257		3.51 ± 0.22

<sup>2</sup>Laurence *et al.* 1992. Chemosphere, 25(7–10): 1333.

\*Non-certified value.

## 5.7 Developments within QUASIMEME and QUASH

### *Request*

There is no specific request; this item is an ACME initiative to follow the development in these two QA projects owing to the long-standing ICES involvement in quality assurance matters.

### *Source of the information presented*

The 1998 report of the Marine Chemistry Working Group (MCWG) and ACME deliberations.

### *Status/background information*

The ACME took note that the QUASIMEME Laboratory Performance Studies (LPS) became available to all laboratories worldwide from June 1996 as a sequel to the EC-funded research programme QUASIMEME and has been open to all organizations making chemical measurements in the marine environment. Since the QUASIMEME LPS provides a continuing, routine support to laboratories, participation is on a subscription basis. The programme for the first year of the LPS, from June 1996 to May 1997, was designed specifically to support the chemical measurements required for the international marine monitoring programmes of the OSPAR Commission (OSPAR), the Helsinki Commission (HELCOM), and the Mediterranean Pollution Monitoring and Research Programme (MEDPOL) and national programmes, the National Monitoring Programme (NMP) of the United Kingdom Marine Pollution Management and Monitoring Group (UKMPMMG). In doing so, the needs of many other national and individual programmes were also served.

During the first year, there were four rounds. Each assessment has been completed and all participants have received a personal report with their own data from each round. The majority of sediment and biota samples were provided as natural or processed unspiked homogeneous samples. The seawater samples for each group of determinands were prepared to cover the range and concentrations of estuarine, coastal and open water sites.

The Laboratory Performance Studies were designed to support the quality management of participating chemical laboratories and assist in the improvement of the quality of measurements. The assessments provided by QUASIMEME also complement internal laboratory QA, and provide a support to laboratory accreditation in addition to the QA support to the environmental monitoring programmes. Laboratories from 27 countries subscribe to QUASIMEME.

## EU Project 'Quality Assurance of Sampling and Sample Handling' (QUASH)

The ACME was informed about the status of the QUASH project, funded under the EC Standards, Methods and Testing Programme, as described below. The aim of the project is to establish validated methods for sample handling and pretreatment, and to improve analytical results by identifying and reducing errors due to sampling and sample handling. At present, many of the guidelines or recommendations for a QA/QC programme related to sample handling or cofactors have not been verified and documented by interlaboratory trials.

The programme consists of six work packages (WP):

WP1) sampling and preservation of nutrients in sea water [Coordinator: Mr Stig Carlberg (SMHI, Sweden)]

The present status of sampling and preservation of nutrients in sea water has been evaluated through a questionnaire sent to the National Coordination Centres (NCCs) which participate in the programme and some other relevant laboratories, and also through a literature study on preservation methods. A practical workshop has been held at the Spanish Oceanographic Institute in Tenerife with participants from the NCCs. The workshop included the following critical steps: sampling of sea water, cleaning of bottles for subsampling, subsampling from the hydrocast bottles, pretreatment (filtration/centrifugation), preservation, storage, and transportation of seawater samples. A verification study of preservation methods has been initiated.

WP2) monitoring contaminants in biota: lipids and water as cofactors [Coordinator: Dr Jacob de Boer (RIVO-DLO, NL)]

An interlaboratory study on the determination of total lipids has been organized. In this study, a new method for the determination of total lipid content is being compared with existing methods. The study is to be evaluated and the results presented at a Workshop for WP2 and WP3, on lipid determination and sampling, in October 1998 in Galway, Ireland.

WP3) sampling of biological tissues [Coordinator: Dr Britta Pedersen (NERI, DK)]

The present status of sampling and sample handling has been evaluated through a questionnaire sent to the NCCs. Samples for an interlaboratory study, especially designed to determine the main sources of error due to

differences in homogenization procedures, have been sent to the NCCs. The results are planned to be presented at the workshop in Ireland, where also other critical steps in sampling and sample handling will be covered.

WP4) sample handling and cofactors in relation to normalization procedures for sediments [Coordinator: Dr Spyros Kornilios (IMBC, Crete)]

The present status of sample handling and cofactors in relation to normalization procedures for sediments has been evaluated through a questionnaire sent to the NCCs. An inter-laboratory study, using wet sediments, has been organized to study the sample handling and sieving procedures. A workshop is planned to be held at IMBC, Crete, where the outcome of the interlaboratory study will be discussed, as well as other relevant topics.

WP5) preparation of test material, laboratory and field performance studies [Coordinator: Dr Wim Cofino/Freek Ariese (IVM, NL)], and

WP6) laboratory and field performance studies [Coordinator: Dr David Wells (FRS, Aberdeen, UK)]

Relevant test materials have been prepared and tested and the QUASH database has been developed to support the other WPs. A QUASH workshop was held in Groningen, the Netherlands in 1997.

#### *Additional comments*

The ACME will continue to follow the progress of QUASIMEME LPS and QUASH.

## 6.1 Effects of Extraction of Marine Sand and Gravel on Marine Ecosystems

### *Request*

There is no specific request; this is part of the continuing ICES work of reporting information on the effects of extraction of marine sediments on the marine ecosystem.

### *Source of the information presented*

The 1998 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 the section of the report of the Working Group on the Effects of Extraction of Marine Sediments on the Marine Ecosystem containing information and discussions on effects of extraction of marine sand and gravel on marine ecosystems, including the extent of extractions and impacts on biota.

The ACME took note of and accepted the WGEXT review and update of the 1997 report on the effects of extraction of marine sand and gravel on the Baltic ecosystem (ICES, 1997). There was a view that the new information did not alter the general conclusions drawn in the 1997 review presented by the ACME and that a more thorough review would benefit from a joint meeting of WGEXT with the HELCOM Environment Committee's Working Group on Nature and Nature Conservation (EC NATURE).

The ACME noted that its information from 1997 has been combined with a background document (EC NATURE 8/98 9/2) compiled by Germany, which updates the available information on marine sand and gravel resources, their exploitation and environmental impacts, and legal regulations for marine sediment extraction in the Baltic Sea.

### **Extraction of Marine Sediments and Impacts on Biota**

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

#### Belgium

In 1997 extraction mainly took place at the northern part of the Kwintebank, as in previous years. Dredging activities have been monitored by a black box system since 1 January 1997. The Ministry of Economic Affairs has not yet made the data available.

#### *1998 ACME Report*

Additional extractions, under temporary licences for the extraction of about  $2 \times 10^6 \text{ m}^3$  of sand and gravel, have been carried out to obtain material for covering two new gas pipelines. Construction of the pipelines has been completed.

Since 1990, a geomorphological and sedimentological research programme (*Westbank*) has been carried out. Bathymetrical measurements (single beam), side-scan sonar and sedimentological sampling (van Veen grab) have been carried out.

#### Denmark

The extraction of marine sand and gravel represents 10–13 % of the total production of materials for construction and reclamation in Denmark. The amounts of dredged materials for construction increased slightly from 1992 to 1995, after which a slight decrease has been seen. The dredging of sand fill for land reclamation has increased markedly over the past ten years as a result of several large construction works in coastal areas. From 1989 to 1993, more than  $9 \times 10^6 \text{ m}^3$  of sand fill and till have been dredged for the construction of the Great Belt bridge and tunnel project. The consumption of sand for beach replenishment at the west coast of Jutland has shown an increase from  $40,000 \text{ m}^3$  in 1980 to more than  $3 \times 10^6 \text{ m}^3$  in 1997.

During the construction of the fixed link between Denmark and Sweden, up to  $3 \times 10^6 \text{ m}^3$  of sand fill will be dredged from the Kriegers Flak in the Baltic Sea. The dredging started in January 1996 and is expected to last four years. Until now,  $1.1 \times 10^6 \text{ m}^3$  has been dredged. During this period, up to  $7 \times 10^6 \text{ m}^3$  dredged material of glacial till and limestone will be used 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  $3 \times 10^6 \text{ m}^3$  of sand fill over the next two years.

No detailed forecast for future extractions has been prepared, but it is expected that the exploitation of marine sand and gravel will increase at the expense of land materials.

In the Sound between Denmark and Sweden, impact assessments have been carried out prior to the initiation of the tunnel and bridge project linking Denmark and Sweden. To date, minor effects arising from dredging operations have been demonstrated. The effects are in accordance with the forecasts and within the accepted limits. Sediment spill during dredging in glacial till and limestone with a large dipper dredger was about 4 % on average. The spill from the backhoe dredger is presently 4–6 % and the spill from cutter suction dredging 4.5 %

on average. To date, when more than 80 % of the dredging has been completed, the total spill has been 4.1 %.

A detailed resource assessment and environmental impact assessment (EIA) of the dredging of sandfill has been carried out on Kriegers Flak in the Baltic Sea. The assessment has been prepared in accordance with EC Directive 85/337. Preliminary results from the spill monitoring programme indicate that the spill rates are strongly influenced by the type of dredger being used. Spill rates range from 0.7 % to 4.8 %. The release of fines and nutrients is very low. For bottom fauna, the preliminary results indicate, in accordance with the EIA, that there is no environmental impact beyond 1000 m of the dredging area.

The effects of the dredging of sand for coastal protection off the west coast of Jutland have been studied. The study is based on a comparison with simultaneous changes in a reference area. A complete quantitative recovery including the number of species, the abundance and the biomass of the bottom fauna has occurred in less than one year after the sand extraction. However, the predominance of a supposed opportunistic species of polychaete (*Spio filicornis*) in the borrow area may indicate a pioneer recolonization. The impact of sand extraction on the predator populations is limited due to a patchy exploitation pattern leaving plenty of food in 70 % of the (undisturbed) bottom and a recovery of the benthic biomass in less than one year.

A three-year research project on the consequences of marine dredging was initiated in 1994. 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. Results from analyses of a very large number of samples from marine resources have shown that the content of silt and clay in only a few samples exceeds 5 %.

The environmental effects of dredging in gravel deposits have been studied in a highly dynamic area north of Læsø in the Kattegat. The dredged material was screened onboard, and the initial spill from the dredger was measured at between 70 % and 90 % of the material dredged. Most of the spill was sand, while only 3 % was silt and clay. Most of the spill sedimented very close to the dredger, where the vegetation was partly buried and, despite the large initial spill, only 5 % was still in suspension 500 m from the dredger.

The environmental impact of gravel dredging in the Limfjord area has been studied. During dredging, the sediments are screened and sand and finer particles are returned to the sea. Spill rates vary between 60 % and 90 %. Most of the spill is sand and less than 5 % consists of silt and clay. The spreading of the sand is restricted to

the dredging area, and sedimentation of fines outside the area is very limited.

## France

Production has been stable from 1990 to 1995, as follows:

Production (in 10 <sup>6</sup> tonnes)					
1990	1991	1992	1993	1994	1995
3.92	3.99	3.70	3.45	3.67	3.66

In 1996 and 1997, production stabilized at around  $3.6 \times 10^6$  tonnes per annum. Production from calcareous deposits is about one fifth that of siliceous aggregates.

In the Dieppe, biological monitoring of recolonization in a former dredging site where sand extraction activity stopped in 1994 was initiated in 1996 and continued in 1997. The results confirmed that the impact of sand deposited from overflow can be as great as the impact of dredging itself on benthic macrofauna. They also confirmed the recolonization process: (1) species richness is similar between recolonization and reference areas (46 and 50 species per station, respectively); (2) abundances have increased slightly, up to 1040 ind. m<sup>-2</sup>, corresponding to 40 % of the reference value; (3) biomass showed the largest increase, doubling between 1996 and 1997, up to 6 g m<sup>-2</sup>, representing 75 % of the reference biomass. The following rates of biological recovery at the site have been observed since the end of the dredging activities:

- after 16 months, species richness was fully restored and abundances recovered more rapidly than biomass (56 % and 35 %, respectively, of the reference values);
- after 28 months, the evolution was the opposite, with stabilization of densities (59 %) while biomass increased up to 75 % of the reference value.

Benthic Macrofauna Recovery (%)			
	T <sub>0</sub> (end of dredging)	T <sub>0</sub> + 16 months	T <sub>0</sub> + 28 months
Biomass	16.7	34.8	74.7
Abundance	14.3	56.1	58.6
Species richness	37	113	92

The nature of the recolonization community is closely linked to the type of substratum, with dominance of epifaunal species on the shingles and sand-dwelling species in the fine sands of the eastern sector. The western community is dominated by species characteristic of the coarse sands which are filling in the dredging site under the major influence of tidal currents,

as confirmed by side-scan sonar. This restoration process operating from the west explains the decreasing impact observed from the western part to the eastern part of the former dredging site. Four levels of impact can be distinguished:

- level 0: reference sediment and community; restoration achieved in the western sector (dominance of coarse sand-dwelling species);
- level 1: recolonization nearly achieved in the dredging site; low impact of overflow in the deposition area;
- level 2: recolonization in progress (lower abundances and biomass, dominance of opportunistic epifauna on bare shingles);
- level 3: maximal impact (lowest species richness-abundance-biomass; dominance of sand-dwelling species in the fine sands linked to dredging and/or overflow).

#### Germany

In the North Sea, the greatest amount of sediment extraction derives from maintenance dredging within the waterways inside estuaries. The annual dredging and dumping range between  $45 \times 10^6$  and  $55 \times 10^6$  tonnes. In 1997, sand extraction was continued for coastal protection of the island of Sylt. The extraction pit is situated 7 km west of Sylt at a water depth of 14 m. The maximum extraction volume is limited to  $2 \times 10^6$  m<sup>3</sup> per year. Commercial sand extraction is planned for the area of the Weisse Bank.

In the Baltic Sea, no extraction of marine sediments has taken place during the past ten years on the coastal shelf of Schleswig-Holstein, and no extraction is currently planned. On the coastal shelf of Mecklenburg-Vorpommern, there are seventeen extraction fields, mainly used for coastal defense purposes. They are periodically exploited. From four fields, sand and gravel is used as construction material. In 1997, aggregate extraction was  $2.3 \times 10^6$  m<sup>3</sup>. The total aggregate extraction from 1992 to 1997 was  $7 \times 10^6$  m<sup>3</sup>.

The effects of dredging on sensitive species in the Baltic Sea are being studied. The extraction field is situated close to Wurstrow (Darss-Zingst Peninsula) and was dredged in November 1997. The first side-scan sonars of the dredged area, in March 1998, showed a concentration of dredging tracks forming a 4.5-m deep trench in the area, with no macrofauna colonizing it so far.

A research project is planned to study the processes involved in the natural refilling of deep pits and large-scale extraction burrows in the North Sea, including the Wadden Sea, and in the Baltic Sea.

#### Ireland

There is currently only one existing license for sand and gravel extraction, which is located in an area off Wexford, southeastern Ireland, but there has been no commercial production from this site to date.

#### The Netherlands

The amounts of sand extracted from the North Sea in 1997 were as follows:

- Euro/Maas access-channel to Rotterdam  $8.3 \times 10^6$  m<sup>3</sup>,
- IJ-Access-Channel to Amsterdam  $4.2 \times 10^6$  m<sup>3</sup>,
- Dutch Continental Shelf  $10.3 \times 10^6$  m<sup>3</sup>,

giving a total sand extraction in 1997 of  $22.8 \times 10^6$  m<sup>3</sup>. About  $1.4 \times 10^6$  m<sup>3</sup> has been dredged on the Dutch shelf and used on the UK shelf for cable protection.

The main applications of the extracted sand are for beach nourishment ( $7.9 \times 10^6$  m<sup>3</sup> in 1997) and for land fill ( $13.5 \times 10^6$  m<sup>3</sup> in 1997).

A desk study estimated the demand for sand from the North Sea for the period 1996 to 2030 as  $919 \times 10^6$  m<sup>3</sup> for a minimum scenario and  $1736 \times 10^6$  m<sup>3</sup> for a maximum scenario. The total expected sand demand for a mean scenario is  $1171 \times 10^6$  m<sup>3</sup>, from which  $589 \times 10^6$  m<sup>3</sup> is for fill,  $432 \times 10^6$  m<sup>3</sup> is for beach nourishment, and  $150 \times 10^6$  m<sup>3</sup> is for construction purposes.

Gravel extraction did not take place in 1997 in the Dutch part of the Wadden Sea. A peak in the extraction of gravel is expected on the southeast coast of the Netherlands as a result of the implementation of the Delta Plan for the Major Rivers. Relatively large quantities of gravel will be extracted up to the year 2005. The total available quantities will increase from  $35 \times 10^6$  to  $60 \times 10^6$  tonnes due to this project.

Shell extraction from the Wadden Sea, North Sea (Voordelta), and Dutch internal waters from 1990 to 1997 is shown in Tables 6.1.1 and 6.1.2.

From 1999 on, new permissible amounts for the extraction of shells will be defined based on an environmental impact assessment which will be completed in 1998.

In autumn 1996 and winter 1996/1997, a beach nourishment was conducted on the central Dutch coast near Heemskerk/Wijk aan Zee. The borrow pit was located at a water depth of 7 m and reached a depth of 19 m below the sea floor. After the beach nourishment

was completed, the pit was filled with sand from an area in deeper water. The benthic fauna was surveyed before and after the nourishment. Two days after the refill of the pit, it was observed that the mollusc *Spisula subtruncata* had survived dredging and transferral to the pit. The recolonization of the pit area, especially by the worm *Phyllodoce mucosa*, was immediate following the refill (Van Dalfsen and Storm, 1998a). After three months, the biomass in the pit area was comparable with that immediately after the refill, but a change in the contribution of the different species occurred. *Spisula subtruncata* decreased. The contribution by worms increased, although *Phyllodoce mucosa* drastically decreased, showing it to be a really opportunistic species. There was no recovery of the molluscs or crustaceans (Van Dalfsen and Storm, 1998b).

Ecological effects of subaqueous sand extraction north of the Island of Terschelling have been studied. The effects on the benthic fauna of sand extraction at a water depth of 21 m were described (Essink, 1998; Van Dalfsen and Essink, 1998). After one year, a change in benthic community structure was observed due to a reduction in long-living species and recolonization by opportunistic species. After two years, the original structure had largely been restored. Nevertheless, adult specimens of longer-living species are rare. The total biomass is still less than that before the extraction. Complete recovery is predicted to take a further few years.

The enlargement of the Rotterdam harbour is planned. The amount of marine sand needed for that purpose will vary between  $400 \times 10^6$  and  $600 \times 10^6$  m<sup>3</sup>. Studies are under way to decide where and how to extract the sand. At present, the permitted extraction depth is defined at 2 m due to fishery concerns and to keep the same sediment at the seabed, as well as to facilitate the benthic fauna recolonization.

The construction of an airport on an artificial island at sea and land reclamation along the coast between Hoek van Holland and Scheveningen (The Hague) are also

planned. For these, a sand supply is needed of  $800 \times 10^6$  m<sup>3</sup> and  $400 \times 10^6$  m<sup>3</sup>, respectively.

#### Norway

In 1996 and 1997, very small amounts of marine sand and gravel were extracted for construction purposes.

#### Poland

In 1997, the amounts of extraction were as follows:

- $252 \times 10^3$  m<sup>3</sup> of sand was extracted from the seabed for beach nourishment on the Hel Peninsula;
- $1.9 \times 10^3$  tonnes of gravel was extracted from the Slupsk Bank area on the Polish EEZ.

About  $1.6 \times 10^6$  m<sup>3</sup> of sand was identified and geologically documented for beach nourishment in the western part of the Polish EEZ, in the vicinity of Dziwnów.

There is no new information available on the impacts of these extraction activities on biota.

#### Sweden

There were no marine aggregate extraction activities in 1997.

The effects of suspended sediments on cod eggs and larvae have been studied. The behaviour of adult herring and cod when affected by the sediment cloud during dredging has also been investigated. In general, the avoidance to clay and chalk suspension increases with the sediment load. Mortality studies showed that the larvae are more sensitive than the eggs to suspended particles. The sedimentation rate of the eggs increased due to sediment coating on the eggs.

**Table 6.1.1.** Total amount of shells ( $\times 10^3$  m<sup>3</sup>) extracted by the Netherlands in the Wadden Sea and Dutch sea-inlets of the North Sea.

	1990	1991	1992	1993	1994	1995	1996	1997
Wadden Sea	63,805	51,630	89,285	90,394	125,755	117,511	103,770	64,938
Sea-inlets	67,765	83,735	82,460	80,396	79,715	78,171	65,540	70,998
Total	130,970	135,365	171,740	170,790	205,470	195,685	169,310	135,936

**Table 6.1.2.** Total amount of shells ( $\times 10^3$  m<sup>3</sup>) extracted by the Netherlands in the southwestern part of the Netherlands (Zeeland), and the North Sea.

	1990	1991	1992	1993	1994	1995	1996	1997
Eastern Scheldt	12,524	490	0	1,475	5,575	300	750	0
Western Scheldt	15,802	21,225	25,175	14,390	4,158	26,850	21,025	28,340
Voordelta	21,282	10,975	12,415	23,750	6,750	20,505	22,500	48,415
Total	49,608	32,690	37,590	39,615	16,483	47,655	44,275	76,755



## United Kingdom

Production in 1997 fell to  $24.8 \times 10^6 \text{ m}^3$  from  $26.6 \times 10^6 \text{ m}^3$  in 1996.  $13 \times 10^6 \text{ m}^3$  was used by the construction industry, mainly for concrete;  $6.9 \times 10^6 \text{ m}^3$  went for export, primarily to the Netherlands and Belgium, with smaller quantities to France and Germany;  $4.9 \times 10^6 \text{ m}^3$  was for licenses for fill contracts and beach replenishments.

Marine sand and gravel continued to supply about 15 % of the total demand in Great Britain in 1997 and the estimated demand for the next fifteen years is expected to be on average  $21 \times 10^6$  tonnes per year.

There was no calcareous seaweed extracted from Crown Estate land in 1997, although a limited amount of extraction took place in the Falmouth Estuary. A limited amount of waste coal was extracted. Very small quantities of sand and gravel were extracted from non-Crown land.

In Scotland, periodic extraction of very small volumes of sand occur in the Tay River/Tay Estuary and in Spey Bay. Up to  $4 \times 10^3 \text{ m}^3$  of maerl is extracted annually from Wyre Sound in Orkney, from an area where dead maerl is accreting and material is being used for specialized wastewater purification and biological filtration applications rather than for general agricultural purposes.

Sediment transportation pathways and sediment movements have been studied since November 1996 in a dredging area southwest of the Isle of Wight. Numerical modelling was used to predict the effects of dredging on the area.

The recovery of an experimental dredging plot off northern Norfolk has been studied. Fish populations in the vicinity of the area were sampled and their stomach contents analysed. Plume dispersion was also analysed. The samples taken three years after dredging showed that the area has fully recovered both in terms of the stability of the sediment and the range of species, although numbers of animals are still lower than in the adjacent control areas.

A further four-year study was launched in January 1998 to investigate the potential for cumulative effects of multiple dredging activity on the seabed environment and fisheries. Feeding links between benthos and fish and/or shellfish populations will be investigated. There will also be an extension of the regional assessment of gravel communities and documentation of the scales of sensitivity in relation to future dredging activity.

A study of the sediment regime out to the 50 m depth contour has been undertaken from the Holderness coast to the Thames Estuary. A conceptual model of sediment movement in the survey area was created. The results of the development of the model will help to assess the impact of existing and future marine aggregate extraction in the coastal zone.

## *Recommendations*

ICES recommends that Member Countries follow the ICES Guidelines for Environmental Impact Assessment (ICES, 1994) and the ICES Code of Practice on Commercial Extraction of Marine Sediments (ICES, 1992) when granting permits for the extraction of marine sand and gravel.

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## 6.2 ICES Environmental Report

### *Request*

There is no request. On the basis of discussions over 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. This Environmental Report is intended for publication on the ICES website, when it is fully implemented. The first contributions to this Environmental Report are provided below.

### 6.2.1 Oceanographic conditions

#### *Source of the information presented*

The 1998 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. At its 1998 meeting, WGOH decided to prepare a summary, in order to provide fishery and environmental managers a brief report, describing the present status of physical conditions in the marine environment within the ICES area. The summary focuses on the upper layers of the ocean, which are most directly linked to fisheries. The deep ocean, however, is also briefly discussed.

The temperatures in both the Northwest Atlantic (the area between Greenland and Canada) and Icelandic waters have increased in recent years and are presently above the long-term averages. At the same time, ice formation has been late, appearing on the Labrador shelf late and receding early. The temperatures for the Northwestern European Shelf Edge and the northern North Sea also continued to increase during 1997 and are close to the highest values ever recorded. Further north, in the Norwegian Sea, there is also an increasing trend, with the same trend observed additionally in the Northern Iberian Peninsula and the Bay of Biscay. The Faeroes Plateau and the Barents Sea, however, have demonstrated a different trend than that mentioned above. At the Faeroes Plateau, the temperature in 1997 fell compared to 1996 and there has been a general cooling since 1960. In the Barents Sea, there has been a slight decrease during the last few years, and the temperature is now slightly below the long-term average.

The ICES standard sections and stations in the Nordic Seas have demonstrated dramatic changes in the water. It would appear that the production of deep water ceased in the 1980s, resulting in a drainage of deep water through the Faeroe-Shetland Channel. The implications of this change are not yet understood, but the inflow of warm

water into the Nordic Seas at the surface and the outflow of deep water below is part of the Global Ocean Conveyor Belt. Changes in this system could have profound effects on the climate of northern Europe. Continued monitoring is essential at this key period.

Many of the changes observed in the ICES standard sections and stations seem to be related to the variability of the North Atlantic Oscillation (NAO). It was pointed out that the NAO might influence a number of winter storms, storm tracks, precipitation, airflow, ice fluxes through the Fram Strait, and deep water formation. Climate change, its causes and effects, seem to be a highly relevant topic for the new ICES Five-Year Strategic Plan and will be considered in more detail by the Oceanography Committee.

### 6.2.2 Zooplankton monitoring results

#### *Source of the information presented*

The 1998 report of the Working Group on Zooplankton Ecology (WGZE) and ACME deliberations.

#### *Status/background information*

The WGZE updated and extended its overview of zooplankton monitoring programmes in ICES Member Countries. Results from monitoring activities were reviewed. Temporal trends in zooplankton biomass or community composition in various parts of the ICES area are summarized below.

#### Canada

Historical Continuous Plankton Recorder (CPR) data were analysed to detect differences in indices of phytoplankton and zooplankton abundance for different years between the eastern and western halves of the Scotian Shelf. All CPR data, from 1961 to 1994, were grouped into eastern or western Shelf regions, and the two regions compared over time. The phytoplankton greenness index (a measure of the amount of chlorophyll) was significantly higher in both regions of the shelf in 1991–1994 than during 1961–1975. The index of abundance of krill was higher on the eastern shelf during 1961–1975 than during 1991–1994. There was no significant difference between 1961–1975 and 1991–1994 in the krill index for the western shelf. Data were not collected between 1975 and 1991.

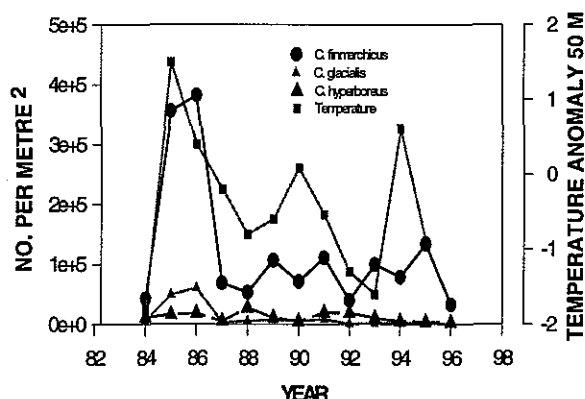
Acoustic data indicate a close relationship between the fish and the krill in Emerald Basin (off the coast of Nova Scotia). These data, collected over the past decade, have shown a close relationship between backscattering at 12 kHz and 200 kHz. The 12 kHz frequency data reflect the concentrations of pelagic fish in the basin and the 200 kHz frequency data provide an accurate estimate of the krill concentrations. The relationship between these two frequencies over the years 1985 to 1995 showed a

significant positive correlation. Both frequencies showed a general increase between 1985 and 1994, followed by a significant decrease in 1995.

Silver hake and redbfish, the two dominant pelagic species, feed primarily on krill in the Basin (Waldron, 1988). In June 1996, there was a large increase in the levels of the 200 kHz backscattering, indicating that the krill stocks had increased from the low values seen in 1995. Data at 12 kHz were not collected in June 1996.

The copepod *Calanus finmarchicus* accumulates in Emerald Basin during the summer and autumn, and remains in the deep water until the breeding season in the late winter and early spring. It is believed that the size of the autumn population of *C. finmarchicus* in the Basin is a good indicator of the size of the population on the Scotian Shelf during the previous spring and summer (Sameoto and Herman, 1990). The *C. finmarchicus* population declined between 1995 and 1996 to reach the historically low levels observed in 1984 and 1992. *C. glacialis* and *C. hyperboreus* (both Arctic species) had very low concentrations in the Basin in 1996. The temperature anomaly at 50 m in June and the numbers of *C. finmarchicus* appeared to be related, showing that, as the temperature increased, there was generally an increase in the size of the *C. finmarchicus* population (Figure 6.2.2.1).

**Figure 6.2.2.1.** Concentrations of three species of *Calanus* in Emerald Basin in relation to temperature, 1984–1996.



The eastern shelf has been influenced by abnormally cold bottom temperatures in recent years, and it is possible that this cold water has affected the size of the krill population in the area. Long-term time series data show that the levels of krill in the eastern region were lower in the 1990s than in the period between 1961 and 1975, when bottom temperatures were warmer.

The Emerald Basin *Calanus finmarchicus* data indicated that since 1987, population levels have been stable but

much lower than in 1985 and 1986. Zooplankton samples and the acoustic index showed that there was a gradual increase in both krill and fish populations in the Basin between 1984 and 1994, followed by a steep decline in their population size in 1995. The krill abundance increased in 1996 to levels observed between 1984 and 1994. The causes for the large fluctuations during 1994 and 1996 are not known.

It is postulated that if the bottom temperatures on the eastern shelf remain cold (i.e., approximately 1 °C), *C. finmarchicus* populations and possibly krill populations will remain low in the eastern region. Krill populations on the western half of the shelf are near their long-term average.

## USA

There has been a general decline in the salinity on Georges Bank from 1995 to 1997, as there has been an increased influence of the Labrador Current water on the Bank. In the spring of 1998 there were the largest level of cod spawning and numbers of cod larvae on the Bank that have been seen in decades. It is speculated that these levels may be as high as the gadoid outburst seen in the 1960s, a time when the Labrador Current also had a large influence on the Bank. A strong case has been presented for the continuation of long-term monitoring of physical and biological parameters on the Bank, if scientists hope to be able to understand the fluctuations in fish populations.

## Portugal

Ichthyoplankton and zooplankton are sampled during monthly surveys off the Portuguese coast. Recent results have shown an increase in the temperature and salinity of the waters of this region, resulting in a change in the distribution of fish and fish larvae, particularly a decrease in sardine numbers.

## Spain

The Instituto Español de Oceanografía (IEO) monitors zooplankton on seven transects off the Spanish coast, four in the ICES area and three in the Mediterranean Sea. This involves an extensive physical, chemical, and biological monthly sampling series at each site, with special attention to the sampling and analysis of hydrographic parameters, nutrients, chlorophyll *a*, and phytoplankton and zooplankton species. Depending on the transect, the time series extend from 1988 (La Coruña and Vigo), 1991 (Santander), and 1994 (Asturias) in the ICES area.

In the Cantabrian and Atlantic areas, the temperature shows an increasing trend throughout the time series.

This increase results in opposite long-term trends, with the water column thermal stratification following a clear upward trend, and the zooplankton species richness and diversity index showing a downward trend. This opposition in long-term trends is confirmed by the significant negative correlation between both sets of data. These results stress the importance that the lengthening of the period during which the water column remains stratified and an increase in the degree of stratification could have in limiting the interchange of nutrients from deeper to surface waters and, consequently, limiting the growth of phytoplankton, which then diminishes the abundance of zooplankton.

#### United Kingdom

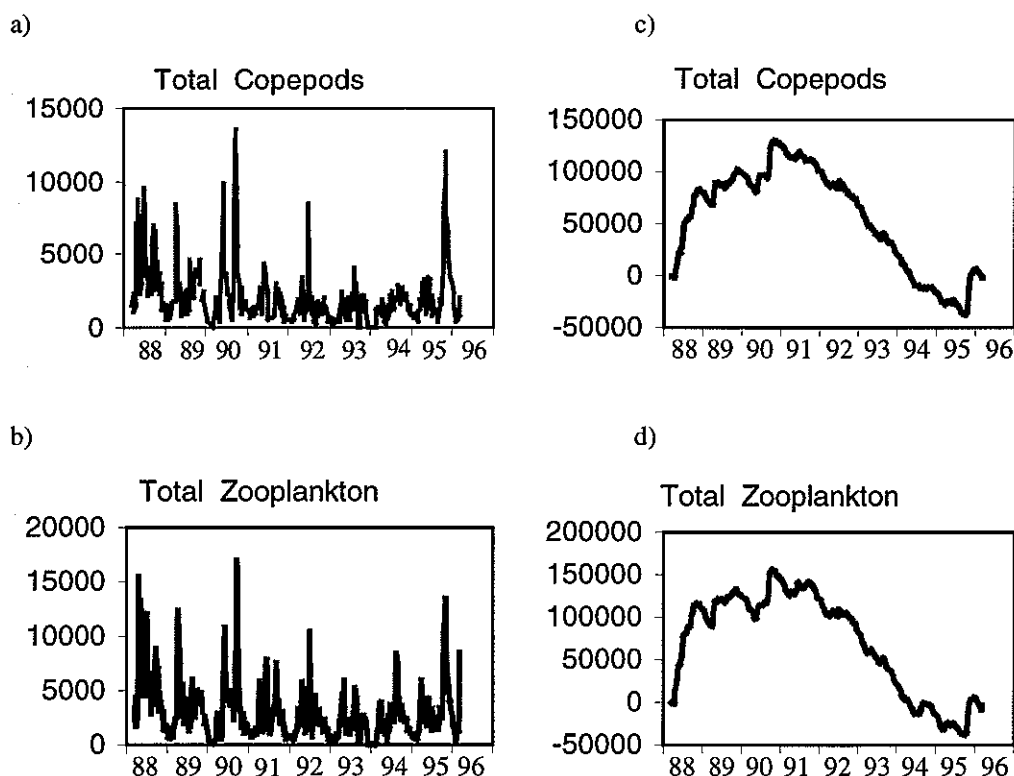
Zooplankton monitoring data are collected at a station off Plymouth. General trends in the zooplankton composition were analysed using the cumulated function suggested by Ibanez *et al.* (1993). The total zooplankton abundance and the copepod abundance off Plymouth show a decreasing trend from 1988 to 1995, with some recovery in 1996 (Figure 6.2.2.2). Applying the same approach to the percentage composition of the copepod population, changes in the population composition were identified and species presenting similar trends were grouped using a principal components analysis.

#### Germany

Trends for copepods at Helgoland Roads for the period 1975–1994 have been analysed. Multivariate analyses on 77 taxa revealed a change in community structure between 1979 (ortho-oblique clustering) and 1982 (multidimensional scaling (MDS)). The beginning and end of the community change period coincided with a decrease in sea surface temperature and salinity in 1979 and a severe decrease in salinity in 1981 (Figure 6.2.2.3, bottom graph). The trends for holoplankton (excluding *Noctiluca scintillans*) and total copepods as part of the holoplankton depict a similar development (Figure 6.2.2.3, top graph). Trends are calculated by subtracting the long-term mean from each single datum and then cumulatively summing up the differences. Until 1980, low abundance values prevailed for holoplankton and total copepods (steadily decreasing curve). *Noctiluca scintillans* already attained a massive increase in 1979. In 1981, abundances for total copepods and holoplankton began to increase; they were low in 1982, and then stayed on relatively high levels up to the beginning of the 1990s, when a further increase occurred. After 1991, abundances were again on a low level.

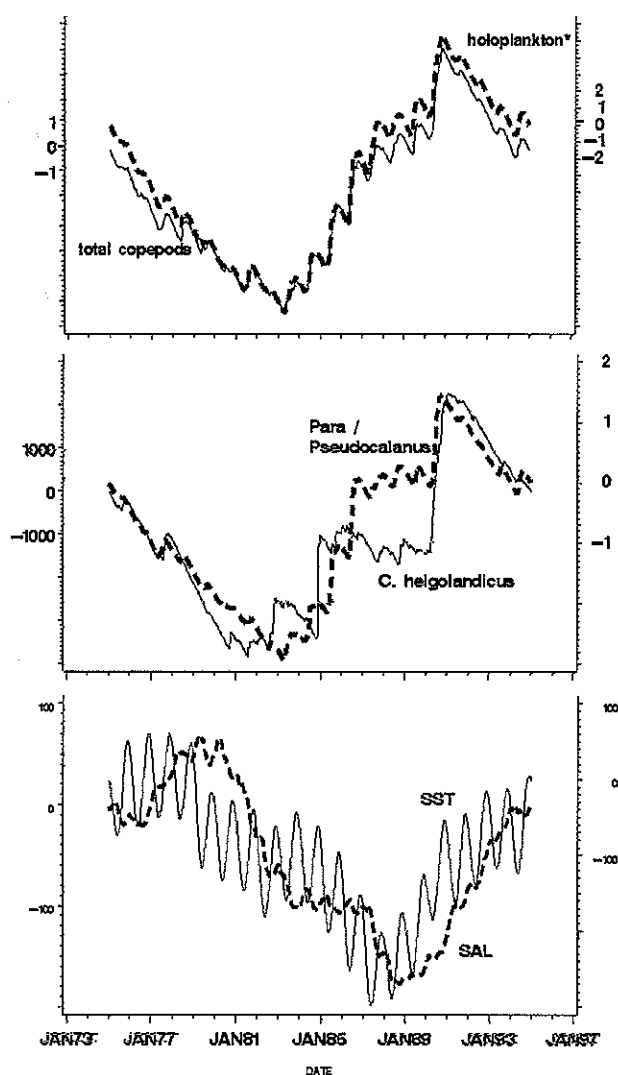
On a species level, the dynamics are more diversified (Figure 6.2.2.3, middle graph). The main turning point

**Figure 6.2.2.2.** Evolution of copepod and total zooplankton abundance from 1988 to 1996 off Plymouth. Real data are shown on the left (a and b) and cumulated deviations from the mean are shown on the right (c and d).



for *Calanus helgolandicus* occurred in 1980, after which the population density stayed on a higher level. For the *Para/Pseudocalanus* spp. group, the turning point occurred in 1983. Both species/groups showed a minor increase in 1978 and a minor decrease from 1986 to 1989.

**Figure 6.2.2.3.** Trend analysis for holoplankton (dashed line, \*except *Noctiluca scintillans*) and total copepods (solid line, top graph); *Calanus helgolandicus* (solid line) and *Para/Pseudocalanus* spp. (dashed line, middle graph); and sea surface temperature (SST) (solid line) and salinity (SAL) (dashed line, bottom graph) at Helgoland Roads. Cumulative deviations from the mean are plotted against time. Values for holoplankton, total copepods, and *Para/Pseudocalanus* are divided by 100,000.



From 1979 to 1982, sea surface salinity (SAL) and sea surface temperature (SST) values were low (negative slope of the curve (Figure 6.2.2.3, bottom graph)). From 1983 to 1986, values increased slightly. The turning points for plankton development fall into this period. Salinity and SST then decreased until 1989. This is

reflected in the lower values of *C. helgolandicus* and *Para/Pseudocalanus* at that time. However, this developmental pattern is not reflected in the values for holoplankton and total copepods.

#### Norway

Norwegian monitoring of zooplankton in the Barents Sea has been carried out since the early 1980s. In 1983–1984, there was a pronounced minimum in zooplankton biomass related to low advective transport of *Calanus finmarchicus* into the Barents Sea from the Norwegian Sea. Since then, the zooplankton biomass has increased to a maximum in the early 1990s, followed by a slight decrease in 1997. The zooplankton biomass is still high and the feeding conditions for pelagic fish in 1998 were considered good (Aure, 1998).

An inverse predator-prey relationship has been revealed between capelin and krill (Dalpadado and Skjoldal, 1996). A similar relationship also exists between capelin and pelagic amphipods (*Themisto* spp.). The capelin stock has been low in recent years. This has caused the stocks of krill and amphipods to increase, as indicated by high proportions of these organisms in the diet of cod.

#### References

- Aure, J. (Ed.) 1998. Havets miljø 1998. Fisken og Havet, særnr. 2: 1998. 90 pp. (In Norwegian, with English summary and legends to tables and figures.)
- Dalpadado, P., and Skjoldal, H.R. 1996. Abundance, maturity and growth of the krill species *Thysanoessa inermis* and *T. longicauda* in the Barents Sea. Marine Ecology Progress Series, 144: 175–183.
- Ibanez, F., Fromentin, J.M., and Castel, J. 1993. Application of the cumulated function to the processing of chronological data in oceanography. C.R. Academy of Sciences, Paris, Sciences de la Vie, 316: 745–748. (In French with an abridged version in English.)
- Sameoto, D.D., and Herman, A.W. 1990. Life cycle and distribution of *Calanus finmarchicus* in deep basins on the Nova Scotia shelf and seasonal changes in *Calanus* spp. Marine Ecology Progress Series, 66: 225–237.
- Waldron, D.E. 1988. Trophic behaviour of the silver hake (*Merluccius bilinearis*) population on the Scotian Shelf. Ph.D. Thesis, Dalhousie University, Halifax, N.S., Canada.

### 6.2.3 Harmful algal blooms

#### *Source of the information presented*

The 1998 report of the Working Group on Harmful Algal Bloom Dynamics (WGHABD) and ACME deliberations.

#### *Status/background information*

Decadal maps depicting the presence of algal toxins were updated by WGHABD on the basis of available data and are presented in Annex 3. These maps are intended to give a global and visual overview of harmful algal events (HAEs) in the ICES area. Nine ICES Member Countries did not report HAEs in the ICES area. This includes Iceland, where there is no monitoring for toxins, and the Netherlands, which failed to nominate an expert. To achieve a global overview, it is important that maps be as comprehensive as possible.

The previous format for national reports has been reviewed in the context of the development of a Harmful Algal Events Database (HAEDAT). HAEDAT will be managed by the Intergovernmental Oceanographic Commission (IOC) and a new reporting format has been

decided which includes a better description of abiotic conditions. In each country, a focal person will be sought who will be in charge of compiling the individual reports. It is expected that, in the coming years, the database will be updated regularly throughout the year. An automatic production of decadal maps will then be possible.

National reports have, however, been presented and compiled as usual, but it is the opinion of WGHABD that, in the future, they should be submitted in electronic form.

### 7.1 Overview on Mercury 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 1998 report of the Marine Chemistry Working Group (MCWG), the review note 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 1998 meeting, MCWG examined and accepted a review paper entitled 'Mercury in the marine environment—A review' by M. Leermakers (Belgium) which describes the global mercury cycle with particular reference to anthropogenic influences, the distribution of mercury in the marine environment, the biogeochemical behaviour of mercury, bioaccumulation pathways, and concentrations and trends of mercury in the North Sea and the North Atlantic.

The paper also provides a summary of the uncertainties and gaps in the information on mercury in the marine environment.

After review, the ACME accepted this paper for inclusion in its report; it is attached as Annex 4. The ACME agreed that it is an excellent and very informative review paper.

#### *Need for further research or additional data*

More information is needed about the inputs of mercury to the marine environment, including a better understanding of the processes occurring in estuaries and coastal areas, so that flux estimates to the ocean can be further refined. Information from Asia and the southern hemisphere is seriously lacking, taking into account the global spreading of mercury. There is also a need for new independent analytical methods in order to detect systematic errors in the methodologies. More relevant certified reference materials (CRMs) as well as field intercomparison exercises are also needed to verify the accuracy of the methodologies.

### 7.2 Influence of Biological Parameters on Concentrations of Trace Metals in Fish Liver

#### *Request*

There is no specific request; this is part of the continuing work of ICES on the monitoring of contaminants and the assessment of results from monitoring programmes.

#### *Source of the information presented*

The 1998 report of the Marine Chemistry Working Group (MCWG) and ACME deliberations.

#### *Status/background information*

The concentrations of many trace elements are higher in fish liver tissue than in fish muscle tissue and they are, therefore, generally easier to determine in the liver. For this reason, fish liver tissue is stipulated in monitoring programmes (e.g., OSPAR JAMP) for the determination of trace metals (except mercury) and organic contaminants. Large data sets have thus been acquired on concentrations of trace metals in fish liver. However, the size of the fish liver and its fat content are highly variable; the liver changes in response to both environmental influences and biological processes, e.g., food availability and reproduction. In the past, ICES Working Groups have examined the influence of variable fat content on metal concentrations but have not reached a clear conclusion, and in assessments of monitoring programmes, the influence of variable fat content has not been evaluated.

At its 1998 meeting, MCWG discussed a paper containing many new and interesting observations on the constituents of cod liver. The paper describes a three-year study on trace elements in Icelandic cod liver, where 454 cod were analysed for several trace elements, liver size, fat and water content, ash, nitrogen and phosphorus. In addition, the age, length, and weight of the fish were recorded. The main conclusion from the study indicates that fat and water content and trace element concentrations are very much dependent on liver size up to 100 grams; liver sizes less than 100 grams are common in the 30–45 cm length range of cod which are sampled in monitoring programmes. Livers heavier than 100 grams are relatively uniform in composition with respect to macroconstituents and many trace elements.

The ACME reviewed this paper and felt that it provided very useful information; accordingly, the ACME agreed to attach this paper as Annex 5.

### *Need for further research or additional data*

The Icelandic study shows how to perform the trace element normalization for Icelandic cod, which come from a single stock and a low-pollution environment. The liver weight, liver lean fraction, and fish age explain most of the concentration variations of macroconstituents and trace elements. For other areas and other fish species, it must be checked as to whether the same normalization procedures can be applied.

### *Additional comments*

The ACME noted that the reported information is relevant to a future assessment of data on trace metals in biota and that ultimately a change may be required in the monitoring guidelines.

### *Recommendations*

ICES recommends that similar studies be carried out by ICES Member Countries on cod and other fish species used for monitoring contaminants in other areas, in order to verify whether these results are generally applicable.

## **7.3 Appropriate Concentration Units for Nutrients and Oxygen in Sea Water**

### *Request*

This issue arises from the ICES Oceanographic Data Centre, in an attempt to obtain agreement concerning appropriate concentration units.

### *Source of the information presented*

The 1998 reports of the Marine Chemistry Working Group (MCWG), the Working Group on Marine Data Management (WGMDM), the Working Group on Oceanic Hydrography (WGOH), and the Working Group on Shelf Seas Oceanography (WGSSO), and ACME deliberations.

### *Status/background information*

Many organizations, especially academic institutes, and a number of global projects such as the Geochemical Ocean Sections Studies (GEOSECS) and the World Ocean Circulation Experiment (WOCE), have unilaterally adopted the unit  $\mu\text{mol kg}^{-1}$  for most chemical parameters in sea water. The trend for using this unit is increasing and an explanation for the scientific grounds for this, as well as advice to the data management community, is required. In particular, advice to the data community should take into account the fact that historical databases hold data only in  $\mu\text{mol dm}^{-3}$  or the equivalent.

It has for many years been customary to express concentrations of minor elements in sea water on a volume basis, e.g.,  $\text{mg m}^{-3}$ . But the concentrations of the major constituents of the salinity have, however, been expressed on a mass basis, e.g.,  $\text{g kg}^{-1}$ , as it was realized that the specific volume of sea water changes sufficiently with temperature, salinity, and depth to introduce significant uncertainty in results on these constituents when expressed on a volume basis.

However, treating the volumetric concentration has certain disadvantages in these circumstances. The concentration, as used in the advection-diffusion equation for dissolved substances, is the amount of dissolved substance per volume of sea water under *in situ* circumstances. This implies that the volumetric concentration may change when a water sample is brought to the sea surface.

Heating and cooling, or compression and expansion, and all the things that can occur when a water sample is brought from the cold abyss to the sea surface, will not change the numerical value of concentrations expressed on a per mass basis.

Chemists are aware of this problem and specify the concentration of a dissolved substance as the volumetric concentration at some specific laboratory conditions, but the corresponding metadata are not always reported. Moreover, scientists in different laboratories use different laboratory conditions when reporting their volume concentrations.

In the Operations Manual of the international WOCE Hydrographic Programme, the general rule is to record concentrations of dissolved nutrients, inorganic carbon, and dissolved gases on a mass basis.

During oceanic cruises, most chemical species are measured on board the research vessel. For producing satisfactory data, these measurements require frequent calibration using working standard solutions. However, for nutrients, for instance, the working standards are known to be unstable, so they should be prepared on board, shortly before use. As gravimetry is not reliable on board ships, only volumetry can be used to prepare working standards. Consequently, concentrations are not directly determined per mass of sea water (as required) but per litre.

Concentrations on a mass basis can then be arrived at by different routes, as indicated below:

- 1) Analysing a sample weighed (*in vacuo*) and standardizing with standards expressed on a mass basis.
- 2) Analysing a sample on a volume basis and converting to a mass basis using the equation of state for sea



water. For this, precisely volume-calibrated glassware must be used and the salinity and temperature of the sample at the time of analysis must be known.

- 3) For operations on historical data sets, convert data expressed on a volume basis, e.g., from a compilation or a data centre, to a mass basis. To do this accurately, either (i) the salinity and the temperature of the sample when it was measured must be known, or (ii) a single conversion factor can be used, which assumes the same salinity and the same temperature for all samples, e.g., a density of  $1025 \text{ kg m}^{-3}$  as practised at ICES ( $S = 35$ ,  $t = 20^\circ\text{C}$ ).

Routes 1 and 2 are reliable and either one can be selected by the investigator. Route 3(i) is troublesome since the temperature of measurement usually does not accompany the data. Route 3(ii) implies two assumptions: that the measurement temperature has been near  $20^\circ\text{C}$  and that the salinity has been near 35. The magnitude of the error introduced by assuming constant density therefore varies as the sample salinity departs from 35 and as the measurement temperature departs from  $20^\circ\text{C}$ .

The consequences in terms of precision are as follows. Sea water (of salinity 35) has a density of  $1.0248 \text{ kg dm}^{-3}$  at  $20^\circ\text{C}$ . This means that the value of a concentration expressed in  $\mu\text{mol kg}^{-1}$  is  $\sim 2.5\%$  lower than when expressed in  $\mu\text{mol l}^{-1}$ . A temperature difference of about  $\pm 4^\circ\text{C}$  generates a density difference of  $\pm 0.1\%$ , i.e., a rather low error which usually may be neglected. A salinity change as high as  $\pm 1.3$  (range 33.7–36.3) leads to a change in the final value by only  $\pm 0.1\%$ .

For coastal and estuarine studies, the problem is different from oceanic studies. These areas are under strong continental influence, therefore salinity (hence density) varies significantly in space and time. For that reason, if calibration of the methods is done, as usual, on a volumetric basis, each sample may have to be treated individually for density correction. This is obviously more complicated than in the case of oceanic studies. Generally, the more data that have to be corrected, the greater the risk of errors in final values.

Because of the problems induced by the use of concentrations expressed per mass of water, instead of per volume, in estuaries and coastal areas, it seems preferable not to recommend calibration in 'per kilogram' of water in such situations.

The issue of whether a standard unit for chemical measurements ( $\mu\text{mol kg}^{-1}$ ) could be adopted was originally raised by the ICES Oceanographic Data Centre. The Data Centre has, for some time, been receiving data in either one unit or the other and has, whilst preserving exactly the data values as supplied by the originator, developed procedures to store the data in its exchangeable data set using the factor 1.025,

representing the density of the standard ocean. This procedure was adopted on the advice of the World Data Centre for Oceanography (WDC-A), because ICES has obligations to conform with the standards and procedures as adopted in the data management standards of IOC, as specified by the IOC Working Committee on International Oceanographic Data and Information Exchange (IODE). In addition, all ICES data sets are, and will continue to be, published on CD-ROM by the WDC-A, for example, the World Ocean Data Base 1998 (see <http://www.noaa.nodc.gov>), necessitating conformity with their practices. In adopting the above procedure, however, the Data Centre has observed that the research community has, by and large, abandoned or forgotten the prescribed use of the reference temperature of  $20^\circ\text{C}$  to calibrate laboratory glassware volumes. This reference temperature is prescribed in the ICES format description and it can be regarded as a cause for concern that this standard may not have been followed in recent years. In addition, in all data sets so far submitted providing data in both units, including the European Union's Ocean-Margin Exchange (EU-OMEX) data set, the ratio between the two units is always the *in situ* density, i.e., the units given in  $\mu\text{mol kg}^{-1}$  are 'derived' directly from  $\mu\text{mol dm}^{-3}$ , giving  $\mu\text{mol kg}^{-1}$  the same status as 'potential temperature', which is also a derived quantity and therefore not preserved by data centres.

Consequently, it is necessary for the Marine Chemistry Working Group to develop precise guidelines on the exact procedures for producing data in  $\mu\text{mol dm}^{-3}$  and  $\mu\text{mol kg}^{-1}$ ; in particular, MCWG should develop advice on the analysis and conversion procedures that are required to ensure the precise conversion of values between these units.

#### Oxygen Saturation Units

In 1997, the ACME drew attention (ICES, 1997) to the fact that the latest Oxygen Saturation Tables were published by UNESCO in 1973. In 1986, a new algorithm was devised by the Joint Panel on Oceanographic Tables and Standards (JPOTS), which has been disbanded following the reorganization of the UNESCO Division of Marine Sciences under the auspices of the IOC. No plans to publish tables based on the new algorithm by any of the JPOTS co-sponsors (IOC/UNESCO, SCOR, IAPSO and ICES) have been forthcoming thus far. The ICES Oceanography Committee will, therefore, consider whether this issue is important enough to warrant the publication of a revised table.

#### Recommendations

ICES recommends that nutrients and oxygen data be stored in databases in either  $\mu\text{mol dm}^{-3}$  or  $\mu\text{mol kg}^{-1}$ , provided that the relevant information is submitted with the data to ensure the conversion between these units.

## Reference

ICES. 1997. Report of the ICES Advisory Committee on the Marine Environment, 1997. ICES Cooperative Research Report, 222: 44.

### 7.4 Endocrine Disruptors in the Marine Environment

#### *Request*

There is no specific request; this is part of the continuing work of ICES to keep under review marine contaminants and their effects.

#### *Source of the information presented*

The 1998 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*

The potential threat of anthropogenic endocrine disruptors in the marine environment to interfere with normal reproduction of marine organisms was first reviewed by ACME in 1996 (ICES, 1996). The ACME recognized the importance of this topic and recommended that further research be undertaken. In 1997, the ACME adopted lists, compiled by WGBEC, of biomarkers and bioassays currently used to investigate the reproductive or endocrine-disrupting effects of contaminants in the aquatic (mainly freshwater) environment and lists of research needs (ICES, 1997).

Since then, more research data which relate to endocrine-disrupting chemicals (EDCs) in the marine environment have become available and were discussed at the 1998 meetings of WGPDMO and WGBEC on the basis of a review paper. This review paper was considered by ACME, which accepted it as Annex 6 to this report.

On the basis of the information available, the ACME concluded that there are indications from an increasing number of studies that marine organisms from contaminated areas are exposed to a variety of contaminants with the potential to induce reproductive and non-reproductive disorders via changes in the endocrine system. However, so far the data are too limited to provide information on effects on fish, crustacea, and marine mammals on the population level. An exception to this is tributyltin (TBT) and its known serious effects (imposex and intersex) on mollusc populations.

## *Need for further research or additional data*

The ACME noted that major research programmes on effects of EDCs in the marine environment are now in progress in several ICES Member Countries. The results of these programmes will provide important information on the spectrum of marine species affected by EDCs and on the spatial distribution of EDC effects.

## *Recommendations*

ICES encourages Member Countries to conduct studies on the effects of endocrine-disrupting contaminants in the marine environment. These studies should be integrated with existing biological effects/fish disease monitoring programmes by including new parameters for the detection of general health effects and specific endocrine-disrupting effects. These new parameters should include, for example, gross reproductive/developmental disorders, gonado-somatic index (GSI), and a general histological screening of the gonads.

## References

ICES. 1996. Report of the ICES Advisory Committee on the Marine Environment, 1996. ICES Cooperative Research Report, 217: 91-95.

ICES. 1997. Report of the ICES Advisory Committee on the Marine Environment, 1997. ICES Cooperative Research Report, 222: 68-70.

### 7.5 Environmental Effects Monitoring Using Caged Bivalves

#### *Request*

There is no specific request; this is part of the continuing interest of ICES in new techniques to monitor the environmental effects of contaminants.

#### *Source of the information presented*

The 1998 report of the Working Group on Biological Effects of Contaminants (WGBEC) and ACME deliberations.

#### *Status/background information*

The ACME was informed about two Canadian environmental effects monitoring (EEM) programmes to quantify the impacts of liquid effluents from pulp and paper mills on receiving aquatic environments.

The ACME found these programmes interesting in that they are closely related to the use of scope-for-growth (SFG) measurements in marine monitoring (see also Section 5.4.2, above). In particular, automated measurements of a number of SFG parameters in a population of bivalves held *in situ* can be considered as a novel approach that avoids stressful manipulations of the animals, as is the case for classical measurements. Other advantages that can be identified include use of the natural food supply in both quality and quantity.

One component of the environmental effects monitoring compares growth, survival, and reproduction in fish living in the effluent versus those living in an unimpacted control area. Because fish surveys failed to produce useful data in marine and estuarine environments, a number of alternative approaches are being considered, one of which is to measure growth in caged bivalves. During the summer of 1997, Environment Canada sponsored a pilot project at a mill on northern Vancouver Island, BC, in which mussels were caged along a gradient from the pulp mill (minimum size range about 5 mm, compartmentalized cages for individual measurements, large replication with 100 animals per replicate). Endpoints measured included accumulation of pulp mill contaminants, and changes in whole-animal wet weights and end-of-test tissue weights. A similar pilot project is scheduled for a pulp mill located in Pictou Harbour, Nova Scotia, during the summer of 1998.

In addition, a second approach will be tested at Pictou Harbour in 1998. In this approach, an apparatus termed HABITRAP (Cranford *et al.*, 1995) will be evaluated; this apparatus is based on the measurement of bivalve food acquisition responses to ambient changes in environmental and contaminant conditions.

The HABITRAP provides continuous measurements of the clearance, ingestion, absorption, and egestion rates of a population of bivalves held *in situ*. The HABITRAP is a sequentially sampling sediment trap that quantitatively collects up to 39 samples of the biodeposits produced by a cohort of bivalves held over the mouth of the trap. Clearance, ingestion, and absorption rates are calculated from egestion rate data and measurements of the concentrations of absorbed (organic matter) and non-absorbed (ash) tracers in food and faeces samples (Cranford and Hargrave, 1994). HABITRAP sample collection periods are programmable so that population feeding activity can be integrated over time intervals ranging from hours to days. The HABITRAP is fully automated and can be moored in the water column or placed on the seabed such that the animals are exposed to ambient food supplies, flow conditions, and potential contaminants. A water sampling programme is conducted during HABITRAP deployments to characterize food supplies and environmental and contaminant exposure conditions. After the HABITRAPs are retrieved, the bivalves are sampled and the body burden of

contaminants determined to provide a measure of contaminant uptake over the sampling period. Such a sampling programme includes all three elements of an *exposure-dose-response* triad used in risk assessments. Chemical analyses of water samples define the exposure, the contaminants in bivalve tissues define the dose, and the time-series of bivalve feeding behaviour defines the response.

The first HABITRAP deployments will monitor short-term (hourly) feeding responses at a site previously shown to display large tidal cycle variations in pulp and paper mill effluent concentrations. Three HABITRAPs (two containing mussels and the third to control for sedimentation) will be moored at the site and programmed to provide hourly feeding responses over three consecutive tidal cycles. Later studies will provide time series of daily food utilization responses at several sites along a pollution gradient. During each deployment, simultaneous sequential water sampling is conducted from a boat and by instrument packages consisting of transmissometers for measuring particle concentrations, fluorometers for measuring chlorophyll *a* and dissolved organic matter levels (a measure of effluent concentration), and a McLane Water Transfer System for the autonomous *in situ* collection of particulate and extracted dissolved matter. Concentrations of biologically available contaminants will be determined at the end of the exposure period from contaminant body burden analysis.

The HABITRAP approach for monitoring food utilization by bivalve filter feeders has been extensively tested and the measurements found to be both precise and accurate. The high precision results from the ability of the method to 'average out' high-frequency changes in feeding behaviour and inter-individual variability. As a result, the method increases the capacity to define the 'normal' physiological state of populations. The high accuracy, as determined by comparing scope-for-growth and observed growth responses, results from the provision of natural food supplies and flow conditions. Feeding rates measured in the laboratory using algal cell monocultures have been found to be much less precise and can be highly inaccurate. However, it is recognized that laboratory-based scope-for-growth measurements are not intended to represent growth in nature, but rather to serve as an index of the animal's physiological state under standard conditions.

Numerous studies have shown that bivalve growth responses to a wide variety of contaminants depend primarily on changes in energy acquisition rather than utilization responses. Research on the seasonality of bivalve feeding responses, conducted using the HABITRAP, indicates that feeding behaviour is closely related to the metabolic demands of reproduction. As a result, for food acquisition to be compared between sites, the animals need to be in the same reproductive state and

the studies have to be conducted simultaneously. The automation of the HABITRAP allows a time series of bivalve food acquisition to be measured simultaneously at numerous sites along a pollution gradient or over larger spatial scales. A detailed comparison of responses, over temporal and spatial scales, to changes in contaminant levels and environmental conditions permits a diagnosis of the relative impact on food acquisition and growth of different natural and anthropogenic forcing functions. The increased knowledge of cause-effect relationships would enhance our capacity to predict the biological consequences of known contaminants.

## References

- Cranford, P.J., and Hargrave, B.T. 1994. *In situ* time-series measurement of ingestion and absorption rates of suspension-feeding bivalves: *Placopecten magellanicus*. *Limnology and Oceanography*, 39: 730-738.
- Cranford, P.J., Vass, W.P., and Reimer, D.P. 1995. HABITRAP: a new *in situ* technique using shellfish for monitoring biological effects of anthropogenic and natural changes in the coastal ecosystem. *In Proceedings of the 1995 Canadian Coastal Conference* (18-21 October 1995, Dartmouth, NS), Vol. 1, pp. 171-185.

## 7.6 Discharge of Produced Water by the Offshore Oil and Gas Industry

### Request

There is no specific request; this topic is of interest to many ICES Member Countries, and it has become an important issue on both sides of the Atlantic.

### Source of the information presented

The 1998 report of the Working Group on Environmental Assessment and Monitoring Strategies (WGEAMS) and ACME deliberations.

### Status/background information

The ACME noted a summary paper, forwarded by WGEAMS, on the amounts and composition of produced water from the offshore oil and gas industry, with some discussion of the possible impacts on the environment. The ACME felt that this paper provided useful information, and accepted it as Annex 7 to this report.

The ACME noted that the amounts of produced water that are continuously discharged into the North Sea are predicted to increase with the ageing of the oil wells. The possibility of effects on marine organisms from chemicals contained in produced water necessitates further research to determine the real impacts of these discharges.

### Need for further research or additional data

The ACME expressed concern that there is insufficient scientific evidence to make clear statements about the long-term sub-lethal effects of produced water in the wider receiving environment. Therefore, more studies on the effects of produced water should be conducted to investigate effects on:

- pelagic communities, from bacteria through phytoplankton to zooplankton;
- fish eggs and larvae;
- higher marine organisms;
- oxygen levels in the water column.

Additionally, discharge and dilution/transport models that can run for at least a year and that consider both particle distribution and the distribution of truly dissolved components should be developed. Such models may give a better picture of affected areas; they should also take into account possible overlapping with discharges from nearby oil and gas fields.

The following types of activities should be carried out in order to draw conclusions about the possible impacts of produced water:

- 1) verification of modelling results by field surveys;
- 2) development of more sensitive short-term tests;
- 3) development of tests for chronic effects on fish eggs and larvae;
- 4) mesocosm experiments and field experiments;
- 5) experimental studies on the development of fish eggs and larvae;
- 6) investigations of the effects of produced water on the activity and composition of bacterial communities;
- 7) field studies which take advantage of gradients and also take into account the variability of components found in produced water.

## 8.1 Results of Statistical Analysis of Data on Disease Prevalences in Wild Fish Stocks

### *Request*

There is no specific request; this is part of the continuing ICES work to develop methodologies for the statistical analysis of fish disease prevalence data. It is also of interest to the OSPAR Commission because externally visible fish diseases are included as one of the techniques for measuring general biological effects of contaminants under the OSPAR Joint Assessment and Monitoring Programme (JAMP).

### *Source of the information presented*

The 1998 report of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO), the 1997 report of the Study Group on Statistical Analysis of Fish Disease Data in Marine Stocks (SGFDDS), and ACME deliberations.

### *Status/background information*

The ACME reviewed information from WGPDMO on the progress made with respect to the statistical analysis of fish disease prevalence data held at the ICES Environmental Data Centre, as submitted by ICES Member Countries.

It was noted that, during the past year, a considerable amount of new data on diseases of dab (*Limanda limanda*) and flounder (*Platichthys flesus*) from the North Sea and adjacent areas (including the Baltic Sea) have been submitted to the ICES Environmental Data Centre according to the ICES standard procedures for the submission of disease data developed by the SGFDDS (ICES, 1997).

The present ICES fish disease database comprises information on single fish (length, sex, diseases) for 399,262 dab and 25,736 flounder covering the period 1981–1997. In total, reports for dab diseases are available from 128 ICES statistical rectangles (1–69 observations per rectangle) and for flounder from 29 ICES statistical rectangles (1–17 observations per rectangle).

These data were made available by the ICES Environmental Data Centre for an intersessional statistical analysis using methods for the analysis of spatial and temporal trends developed by the SGFDDS (ICES, 1997). The aim of the analysis was to identify location-specific temporal trends in the prevalence of the

diseases lymphocystis, epidermal hyperplasia/papilloma and skin ulcers in dab and lymphocystis and skin ulcers in flounder, as well as to compare the trends found at different geographical locations.

From the complete data set, only those rectangles were used for the analysis which fulfilled defined quality criteria with respect to the period covered, the number of regular observations, and the number of fish examined per size group. For dab, data from 33 rectangles, and for flounder, data from 12 rectangles met the requirements. The analysis focused on female fish of the size groups 20–24 cm for dab and 25–29 cm for flounder, as specimens of these size groups were represented in the samples with the highest frequency.

The statistical analysis was conducted using Generalized Logistic Models (GLM), with the disease prevalence as the dependent variable and the fish gender and size, the quarter of the year and the calendar time as explaining variables. Models were adapted to consider general non-linear trends and contained smooth functions. For each rectangle, the predicted prevalence of the above diseases together with a 95 % confidence interval were calculated. The estimation of the confidence intervals was done by a parametric bootstrap simulation.

WGPDMO noted that, from the results of the analyses, there is clear evidence of the presence of significant and consistent temporal trends in disease prevalences. However, the direction of these trends was not uniform for all areas. Some areas showed markedly distinct patterns which deserve particular attention in the coming years.

In the discussion of the results, WGPDMO concluded that an identification of possible causes for the observed spatial and temporal trends in disease prevalence is still not possible, partly due to the lack of availability of physical and biological data presently stored in ICES databanks. Therefore, WGPDMO re-emphasized the need for a holistic data analysis involving available environmental, oceanographic and fisheries data from the ICES databanks.

### *Need for further research or additional data*

The ACME regretted that the final report containing the results of the statistical analyses of fish disease data was not available for a detailed review at its meeting. However, emphasizing that the ICES fish disease data constitute valuable information for further use and should be considered, for example, by OSPAR for inclusion in the Quality Status Report (QSR) 2000, the ACME agreed that the final report on the statistical analyses should be

reviewed intersessionally when available. Accordingly, the ACME members reviewed this report in November/December 1998 and decided to attach it as Annex 8.

## Reference

ICES. 1997. Report of the Study Group on Fish Disease Data in Marine Stocks. ICES CM 1997/F:3.

### 8.2 Use of Liver Histopathology of Flatfish for Monitoring Biological Effects of Contaminants

#### *Request*

There is no specific request; this is part of the continuing work of ICES to identify and develop biological effects techniques suitable for monitoring programmes, such as the OSPAR Joint Assessment and Monitoring Programme (JAMP).

#### *Source of the information presented*

The 1998 reports of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) and the Working Group on Biological Effects of Contaminants (WGBEC), and ACME deliberations.

#### *Status/background information*

Studies on liver histopathology of flatfish are among the biological effects techniques designated for incorporation in the biological effects monitoring component of the OSPAR JAMP. Guidelines on sampling strategies and the type of liver lesions to be monitored have been developed, mainly based on the results of the 1996 ICES Special Meeting on the Use of Liver Pathology of Flatfish for Monitoring Biological Effects of Contaminants (ICES, 1997a).

Identifying that there was still a need for establishing quality assurance (QA) procedures for liver histopathological studies, WGPDMO proposed in 1997 to organize a Training and Intercalibration Programme (TIP) for the diagnosis of histopathological liver lesions of flatfish, with expert laboratories from Europe and North America participating. According to a recommendation from ICES ACME in 1997 (ICES, 1997b), the TIP has now become a part of the EU-funded project BEQUALM (see Section 5.4.1, above).

The ACME noted that the BEQUALM project will allow further integration of studies on liver histopathology into the OSPAR JAMP. The proposal for the liver histopathology intercalibration programme will require expert laboratories from various ICES Member Countries (lead laboratory: CEFAS Weymouth Laboratory, UK) to intercalibrate diagnoses of target liver lesions and

produce reference materials. Two workshops are proposed to establish protocols and provide exercises in diagnosis. In addition, a colour atlas of flatfish liver histopathology will be produced, and a methods description on liver histopathology is in preparation for publication in the *ICES Techniques in Marine Environmental Sciences* series.

With regard to the incorporation of liver histopathology data into the fish disease section of the ICES Environmental Data Bank, proposed by WGPDMO in 1997, the ICES Council has decided (ICES C.Res.1997/4:2) that the ICES Secretariat will adapt the ICES Fish Disease Data Entry Program and the Fish Disease Data Reporting Format, as appropriate. According to the results of the ICES Special Meeting (ICES, 1997a), WGPDMO recommended that five new categories of relevant liver lesions (unique degenerative lesions, foci of cellular alteration, benign neoplasms, malignant neoplasms, and non-specific lesions) should be added to the existing list of diseases of dab and flounder in the ICES Fish Disease Data Entry Program/Fish Disease Reporting Format and that ICES Member Countries should submit data on these categories of liver lesions.

#### *Need for further research or additional data*

The ACME endorsed the view expressed by WGPDMO and WGBEC that histopathological changes constitute an important integrated biomarker of contaminant effect and that, therefore, ICES Member Countries should be encouraged to include liver histopathology into their fish disease monitoring programmes. The observation of declining prevalences of grossly visible liver neoplasms in dab and flounder populations in various areas of the North Sea and Baltic Sea in recent years would also favour a histological approach. This will allow the detection of precursor and other contaminant-related lesions. These lesions may occur at higher prevalences, and in younger fish, than do neoplasms.

The ACME further agreed that data from liver histopathology studies should be incorporated into the ICES Environmental Data Bank and that the computer programs and database structure should be adapted accordingly. However, before data can be submitted, a quality assurance programme for the diagnosis of liver lesions, as outlined above and planned as part of the BEQUALM project (see Section 5.4.1, above), should be implemented. The ACME noted that, in order to fulfill the QA requirements, data will not be submitted to ICES before the QA project has been finalized.

#### *Recommendations*

ICES encourages Member Countries to conduct studies on fish liver histopathology according to ICES methodologies as part of biological effects monitoring programmes and to submit data on these lesions to the

ICES Environmental Data Centre once QA requirements for the diagnosis of lesions have been met.

## References

ICES. 1997a. Report of the ICES Special Meeting on the Use of Liver Pathology of Flatfish for Monitoring Biological Effects of Contaminants. ICES CM 1997/F:2.

ICES. 1997b. Report of the ICES Advisory Committee on the Marine Environment, 1997. ICES Cooperative Research Report, 222: 38–41.

### 8.3 Causes of the M-74 Syndrome in Baltic Salmon and Progress in Understanding Relevant Environmental Factors

#### *Request*

There is no specific request; this is part of the continuing ICES work of updating the present knowledge on this topic.

#### *Source of the information presented*

The 1998 report of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) and ACME deliberations.

#### *Status/background information*

The ACME reviewed information on the progress in understanding the environmental factors that influence the occurrence of the M-74 syndrome in Baltic salmon.

M-74 is a syndrome causing mortality in yolk-sac fry of Baltic salmon. Swedish and Finnish data on the percentage of females producing M-74-affected offspring in 1997 differed significantly. The Swedish prevalence data showed a steep decrease from 68 % in 1996 to 28 % in 1997, while the Finnish figures have remained at the same high level (approximately 70 %) for the last five years. This difference appears to be due to different methods of calculating the prevalence of the syndrome. In Swedish hatcheries, females with visible signs of M-74 (behavioural disturbances and pale pigmentation of eggs) were not used as spawners and were excluded from the calculations in 1997, whereas the Finnish data included all females.

So far, there are no reports of the occurrence of M-74 (or equivalent reproductive disorders) in areas other than the Baltic Sea (Sweden, Finland, Estonia) and the Great Lakes region in North America (United States and Canada), where the equivalent disorder is termed Early Mortality Syndrome (EMS).

As a result of the scientific collaboration between the United States, Canada, Finland, and Sweden, three hypotheses for the development of EMS/M-74 have been formulated:

- the salmon diet is deficient in thiamine;
- the salmon diet is rich in thiaminase;
- the diet contains a contaminant interfering with thiamine.

Research on the aetiology of M-74 has, therefore, focused on thiamine dynamics as well as on the impact of environmental contaminants.

Many of the symptoms observed in affected brood stocks and fry (neurological behavioural disturbances) are similar to signs of thiamine deficiency in mammals. Recent Finnish investigations on the thiamine dynamics showed that muscle tissue was the main site of thiamine reserves of the female salmon. Bath treatment of eggs with thiamine concentrations ranging from 0.1 % to 0.3 % gave a positive dose-dependent response.

Eggs from females with visible signs of M-74 appear pale, indicating a lack of carotinoids. Low pigmentation may be a sign of a high load of organic contaminants since the pigments, i.e., the carotinoids, are metabolized by the same cell system (the cytochrome P450 system) that is activated by some organic compounds. Intraperitoneal injections of astaxanthin (a carotinoid) given to females significantly increased the astaxanthin content in their eggs, but had no influence on the survival of the yolk-sac fry.

At present, data indicate that no infectious agents are involved in the aetiology of M-74.

Finnish investigations have demonstrated a significant positive correlation between M-74 and the levels of certain planar PCBs. However, a clear cause-effect relationship between these contaminants and the development of the syndrome has not been demonstrated.

The ACME noted that, although intensive research is going on in order to determine the aetiology of the M-74 syndrome (and the equivalent EMS in North America), no significant breakthrough in this field occurred during 1997. However, the ACME considered it likely that the M-74 syndrome, like many other fish disease problems, has a multifactorial aetiology.

#### *Recommendations*

ICES recommends that Member Countries continue to monitor salmonid populations for the occurrence of reproductive disorders similar to the M-74 syndrome and that countries in areas where the M-74 syndrome occurs

standardize their methods for monitoring the disease levels in order to ensure comparability of data on trends.

#### **8.4 Status of *Ichthyophonus* in Herring**

##### *Request*

There is no specific request; this is part of the continuing ICES work of updating the present knowledge on fish disease prevalence.

##### *Source of the information presented*

The 1998 report of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) and ACME deliberations.

##### *Status/background information*

The ACME reviewed information on the status of *Ichthyophonus hoferi* infections in herring.

In 1990, an epizootic of *Ichthyophonus hoferi* occurred in herring from the Norwegian and the Barents Seas, the

North Sea, the Skagerrak and Kattegat, and the Baltic Sea. Since that time, the prevalence has declined in all areas. At present, the most heavily affected herring stock seems to be the Atlanto-Scandian herring, in which the prevalence of *Ichthyophonus hoferi* ranges from 2 % to 3 %. In other herring stocks monitored, the prevalence is below 1 %.

On the Pacific coast of the United States, the prevalence of *Ichthyophonus hoferi* in Pacific herring showed a decreasing trend from 25 % in 1996 to 18 % in 1997. In contrast to observations from the North Atlantic area, the infection in Pacific herring did not seem to be associated with mortalities.

*Ichthyophonus hoferi* infections still persist in some stocks of Atlantic herring, but generally at a lower or decreasing prevalence level.

##### *Recommendations*

ICES recommends that Member Countries continue to monitor the prevalence of *Ichthyophonus hoferi* infections in herring as a part of the fish stock assessment work.



### 9.1 Status of On-going 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 1998, but the status of on-going and proposed introductions and transfers was reviewed.

#### Source of the information presented

The 1998 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. These releases include eggs and fish of established cultured species of salmon, trout, pike, bass and perch in Canada; hatchery-reared juvenile Pacific cupped oysters *Crassostrea gigas* in Germany, Ireland, England and France; Manila clams *Ruditapes philippinarum* in Norway, Ireland and England; juveniles of the shrimp *Penaeus japonicus* and flat oysters *Ostrea edulis* in France; salmon and sea trout elvers in Sweden; and the red alga *Porphyra yezoensis* in the State of Maine, USA.

The ACME noted the outcome of studies, in both 1996 and 1997, on the effects of the introduction of *P. yezoensis* in coastal waters off the State of Maine, which showed that while asexual monospores were found in the intertidal areas close to the farm site, none survived through the winter. The risk of natural spreading of this alga is, therefore, considered to be low.

The ACME noted that accidental introductions and transfers of non-native species continue to be documented in many ICES Member Countries. These include:

- the seaweed *Codium fragile* on the coast of Prince Edward Island, Canada, which is causing significant problems for oyster cultivation at sites in Malpeque Bay;
- the Japanese shore crab *Hemigrapsus penicillatus* in waters off Le Havre, France, which is considered to have the potential to spread to the UK and North Sea coasts of Europe;
- the shipworm *Teredo navalis*, which continues to spread in the Baltic Sea, with evidence of reproduction of the species in some areas of the Baltic;
- the parasitic copepod *Mytilicola orientalis*, introduced with imports of oysters from France, which has now become established in Dungarvan Bay, Ireland;
- the zebra mussel *Dreissena polymorpha*, which has recently become established in the Shannon River, Ireland;
- the seaweed *Caulerpa racemosa*, found recently in the port of Marseilles, France, and *Caulerpa taxifolia*, now present in the Ligurian Sea, Tuscany, Sicily, and the Croatian side of the Adriatic Sea;
- the cladoceran zooplankton species *Cercopagis pengoi*, which continues to spread in the Baltic Sea and has had mass occurrences in the Gulf of Finland and the Gulf of Riga;
- the round goby *Neogobius melanostomus*, which is now well established in the Gdansk Bay area of the Baltic Sea;
- the Pacific red alga *Grateloupia doryphora*, which is now established off Rhode Island and Cape Cod on the Atlantic coast of the USA;
- an unidentified South African sabellid worm, which has been infecting the Californian abalone industry since the late 1980s;
- the polychaete *Marenzelleria* spp. in the estuarine and coastal waters of the North Sea and Baltic Sea. This polychaete is native to estuarine and coastal waters of the Atlantic coast of North America. Investigations by the University of Rostock (Germany) have shown that there are two species of *Marenzelleria* now present in Europe. In the Baltic Sea (Germany to the Bothnian Bay), the species has been identified as *Marenzelleria* cf. *viridis*, while in the North Sea (from Denmark to Belgium to the United Kingdom), the species has been identified as *Marenzelleria* cf. *wireni*. For each of these species, genetically similar parent populations have been

identified in North America, but the populations in the North Sea may be due to range extensions of Arctic populations. Work is going on to further elucidate the taxonomic position of the North Sea populations. The organism has a high reproductive potential and *Marenzelleria* sp. has developed populations of several thousand individuals per square metre in many estuarine and coastal habitats in Europe. Although negative correlations have been found between the abundance of *Marenzelleria* sp. and native benthic fauna, there is no clear evidence that *Marenzelleria* sp. is outcompeting the native fauna. Populations of *Marenzelleria* sp. may provide a food source for demersal fish species, e.g., plaice *Pleuronectes platessa* and flounder *Platichthys flesus*.

In addition to the above, the ACME noted range expansions and population increases of long-established alien species in the North Atlantic and the Mediterranean Sea, e.g., the tubeworm *Ficopomatus* in France and Ireland, the alga *Caulerpa racemosa* in the northern Mediterranean, the expansion of the alga *Codium fragile tomentosoides* in the Gulf of St. Lawrence, the population increases of the Chinese mitten crab *Eriocheir sinensis* in Germany and England, and the continued expansion of the alga *Sargassum muticum* in Scandinavia.

There is now a considerable trade in the import/export of live finfish and shellfish among ICES Member Countries. The ACME continues to express concern in relation to these transfers of fish and shellfish stocks relative to the potential for the accidental introduction of exotic species. Increasing movements and transfers of fish and shellfish could potentially increase the frequency of accidental introductions of alien species (e.g., *Mytilicola orientalis*).

#### *Need for further research or additional data*

The ACME noted that few data are available on the transfer, or potential transfer, of non-native species and, in particular, harmful algal species associated with the movement of shellfish stocks in and between ICES Member Countries. Such data are necessary for the development of sound, scientifically based management and control strategies.

#### *Recommendations*

ICES recommends that Member Countries continue to monitor the introduction and transfer of non-native species and to alert ICES to range expansions of established alien species.

ICES recommends that the ICES Code of Practice on the Introductions and Transfers of Marine Organisms (ICES, 1995) be followed in all cases involving the transfer of fish or shellfish stocks.

#### **Reference**

ICES. 1995. ICES Code of Practice on the Introductions and Transfers of Marine Organisms 1994/Code de conduite du CIEM pour les introductions et transferts d'organismes marin 1994. 12 pp.

### **9.2 Ballast Water 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 1998 report of the ICES/IOC/IMO Study Group on Ballast Water and Sediments (SGBWS) and ACME deliberations.

#### *Status/background information*

The ACME reviewed the outcome of the second meeting of the ICES/IOC/IMO Study Group on Ballast Water and Sediments, held in The Hague, The Netherlands on 23–24 March 1998. The meeting was co-chaired by Dr J. Carlton (ICES), Dr M. Nauke (IMO), and Dr C. Bolch (IOC).

#### **Sampling**

Very large volumes of ballast water are transported globally and these volumes are likely to increase with increasing global trade. While extensive data sets on ballast water volumes and the organisms associated with ballast water and sediments are now being developed, it was agreed that standardization of these data sets was needed to ensure comparability and facilitate cross-analysis. The importance of obtaining data on both imported and exported ballast water and sediments was stressed. The need for global intercalibration of sampling techniques was also emphasized.

With respect to sampling techniques, the need to distinguish between sampling programmes for scientific research and sampling programmes for management and compliance monitoring was stressed. Different sampling methods may need to be employed, depending on the type of data and information required.

Understanding the physical and chemical environment within the ballast tank and how it changes with time was considered to be a fundamental requirement for understanding how the ballast environment promotes or depresses biological transfer. With a better understanding of these factors, it may be possible to manipulate the environment to depress biotic success and hence control introductions.

### Management strategies

Management strategies to minimize the uptake of organisms within donor regions need to be more thoroughly investigated. Such strategies include:

- moving the ship to a higher salinity region of an estuary to minimize the uptake of larvae of oligohaline or freshwater species (e.g., zebra mussel *Dreissena*);
- moving the ship away from sewage outfalls;
- moving the ship away from active algal blooms;
- avoiding regions where dredging is taking place;
- avoiding or reducing ballasting when targeted nuisance species are abundant in the water column.

The successful implementation of such management strategies necessitates the development of systems for the rapid and timely dissemination of up-to-date information and data derived from port surveys and other previous and ongoing monitoring programmes, e.g., toxin monitoring programmes. Such systems could include the issuing of notices to mariners as well as frequent and regular reports to harbour authorities in both 'donor' and 'receiving' ports.

The exchange of ballast water also needs to be more fully investigated. In order to assess the effectiveness of the exchange process, information is required on:

- which tanks or holds were exchanged;
- how much water was exchanged;
- when and where the water was exchanged;
- what organisms escape the exchange process.

There are a variety of experimental approaches to the treatment of ballast water and these have been documented in the 1997 ACME report (ICES, 1997). As many of these approaches are still in the experimental stage, much more work is required before they can be adopted for routine use.

### Risk Assessment

Research is continuing on risk assessment and decision systems relating to ballast management. Up-to-date information, from continuing port surveys, is required in order to provide information on the species of concern that may be taken up with ballast water. Such information is critical for the management of ballast in receiving ports.

There is continuing interest in other ship-associated transport mechanisms, e.g., fouling of ships' hulls and in sea chests. The removal of hull fouling with mechanical cleaners while vessels are berthed in the dock may be one means whereby a new inoculation of an exotic species may take place, and much work needs to be done to distinguish between ballast-associated introductions and other ship-mediated introductions.

### International Cooperation

The SGBWS continued to stress the need for international cooperation on this issue. The SGBWS noted that the attendance at its 1998 meeting was more than double that at the first meeting in 1997, and that the number of countries represented had trebled and considerable amounts of new data were presented. This clearly reflects the international and global interest in ballast water management issues.

The SGBWS pointed out the need for rapid and broad dissemination of information on management strategies of ballast water and sediments.

### Recommendations

ICES recommends that international cooperation for ballast water management and control should continue.

### Reference

ICES. 1997. Report of the ICES Advisory Committee on the Marine Environment, 1997. ICES Cooperative Research Report, 222: 97-99.

### 9.3 Other Issues Relevant to Introductions and Transfers of Marine Organisms

#### Request

There is no specific request; this topic is of interest to ICES Member Countries and other international organizations.

### Source of the information presented

The 1998 report of the Working Group on Introductions and Transfers of Marine Organisms (WGITMO) and ACME deliberations.

### Status/background information

#### **Multinational initiatives and programmes**

The ACME noted that several multinational studies on topics related to the introduction and transfer of marine organisms are presently under way. These include the following studies listed below.

#### EU Concerted Action Plan

The EU Concerted Action Plan on testing monitoring systems for risk assessment of harmful introductions by ships to European waters is a programme that involves six European countries (Germany, Finland, Ireland, Sweden, the United Kingdom, and Lithuania), as well as scientists from North America, Asia, some Mediterranean countries, Australia, and the International Maritime Organization (IMO). The programme is coordinated by Germany and funded by the EU. The subject areas of the study include:

- determination of the state-of-the-art of ballast water studies;
- evaluation of sampling methods;
- validation of sampling methods through inter-calibration workshops and assessment of in-transit survival;
- development of a set of intercalibrated monitoring systems for use by EU countries and international organizations, e.g., ICES, IOC, IMO.

#### Baltic Marine Biologists (BMB)

- 1) The BMB Working Group on Non-indigenous Estuarine and Marine Organisms (NEMO) held meetings in 1995 and 1996, but since then it has been working by correspondence. BMB NEMO is focused on developing and maintaining a database on the alien species in the Baltic Sea; initial entries in this database are listed on the BMB NEMO home page: <http://www.ku.lt/nemo/mainnemo.htm>. A number of scientists from all countries bordering the Baltic Sea are involved.
- 2) A Baltic Marine Biologists Symposium will be held at the University of Klaipeda, Lithuania, in June 1999, and will include the topic of alien species in brackish water ecosystems.

### Risk Assessment for Marine Alien Species in the Nordic Area 1997–1998

This programme, funded by the Nordic Council of Ministers, was launched in 1997 to evaluate, *inter alia*, whether:

- resources were at risk and vulnerable to invasions of non-indigenous species;
- Nordic marine areas were particularly sensitive to the introduction of non-indigenous species;
- organisms, or categories of organisms, have the potential to cause large-scale environmental problems, particularly impacting on biodiversity.

Economic losses as a result of the introduction of non-indigenous species will also be considered.

#### **New publications**

The ACME noted the initiation of a new journal 'Biological Invasions', published by Kluwer. The journal will bring together information from terrestrial, freshwater and marine fields of research on this topic.

#### **Genetically modified organisms**

The ACME took note of the updated information presented by members of WGITMO on issues related to genetically modified organisms (GMOs).

In Canada, work is under way to develop transgenic salmonids for the aquaculture industry. The federal Department of Fisheries and Oceans is continuing to develop a policy aimed at providing guidelines for the development and use of transgenic and other genetically modified fish and shellfish.

The Swedish Environmental Protection Agency has finalized a policy on the introduction and spread of non-native and genetically modified organisms. With respect to genetically modified organisms, the objectives are that:

- a) any use of GMOs which conflicts with the objectives of conserving biological diversity, protecting human health or ensuring the sustainable use of natural resources must not be allowed;
- b) GMOs should only be permitted to be introduced into the environment when it is clear that they can neither multiply nor spread outside the area of introduction, nor spread genes that are likely to multiply outside the area of introduction.

In the United Kingdom, any proposal to release GMOs to the wild would have to include a thorough risk assessment analysis under EU Directive 90/220.

### 10.1 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 1998 report of the Working Group on Phytoplankton Ecology (WGPE) and ACME deliberations.

#### *Status/background information*

Nutrients in coastal areas have different origins, including both natural sources and anthropogenic inputs. The importance of anthropogenic inputs relative to natural fluxes is often difficult to assess.

The effect of nutrient reduction plans is not clear. For example, the most recent background documentation for the North Sea dates from 1995. It appears that most of the North Sea countries have met their commitments for the reduction of phosphorus inputs, but are far from meeting the requirements in terms of nitrogen. Discussing the evolution of phytoplankton communities in coastal waters in this context would require a recent assessment of the nutrient reduction plans.

WGPE reported on the evolution of phytoplanktonic communities in seven geographical areas. Most of the reports emphasized the great difficulty in extracting trends due to the strong influence of meteorological variability on phytoplankton (such as the events which led to anoxia in the Kattegat in 1997).

A clear linkage between algal bloom events and nutrient enrichment has not been established for the Eastern coastal waters of the United States.

Concerning the Baltic Sea, between 1970 and 1980, intensive blooms indicated an increase in eutrophication. Since input reductions have taken place, however, symptoms of eutrophication have decreased in some near-coastal areas but, in the open sea, no clear changes have been observed. From modelling conclusions, both nitrogen and phosphorus would have to be reduced to counteract eutrophication.

The 36-year-long time series in the German Bight, at Helgoland, clearly shows an increase in small flagellates (< 10 µm), but no change in the spring diatom bloom.

Despite the increase (3 to 4 times) in nitrate concentrations and the doubling of phosphorus concentrations in the German Bight, the effect of increasing levels and N:P ratios on phytoplankton could not be shown at Helgoland to the level expected according to eutrophication theory.

Modelling of Dutch coastal waters predicts a decrease in eutrophication-related problems following a reduction in nitrogen inputs. However, this conclusion, drawn from mesocosm experiments, is not backed up by field measurements.

#### *Need for further research or additional data*

There is a clear need to identify the gaps in knowledge about eutrophication theory, such as limits in the extrapolation of mesocosm experiments to natural conditions, and in the modelling approach, which is very often poorly validated against measurements.

### 10.2 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 1998 report of the Working Group on Harmful Algal Bloom Dynamics (WGHABD) and ACME deliberations.

#### *Status/background information*

The dynamics of harmful algal blooms (HABs) cannot be understood following a single approach, as growth strategies differ from one species to another. The evolution of a toxic event depends on the species of interest. One challenge is to apply results acquired on one species to others, if possible. One way of simplifying the diversity is to attempt a functional or a phenomenological grouping of species. This approach could not be achieved solely through bibliographic reviews, such as the one produced in 1998 by NATO (Anderson *et al.*, 1998). In order to obtain information about the oceanographic regimes and the sequence description of one event, a questionnaire has been distributed to different scientists around the world. This questionnaire includes items which are often overlooked, such as a description of the hydrodynamic regimes. It is expected that a synthesis of the answers will make apparent some gaps in knowledge, thereby allowing a strategic planning of research. One of

the foreseeable outcomes in the near future will be to scale the relative importance of anthropogenic nutrient inputs in the development of toxic events.

The ultimate goal of research and monitoring efforts on HABs and their impacts is to protect public health, fisheries resources, ecosystem structure and function, and coastal aesthetics, as well as other recreational values. A fundamental understanding of the many factors that regulate the dynamics of HABs is necessary, but does not provide sufficient protection. WGHABD detailed the different strategies for mitigation and control which could be envisaged in the light of present knowledge. It appears that ecological knowledge would not be sufficient, even if all the control processes were known, and that engineering expertise as well as economic studies are required.

## Reference

Anderson, D.M., Cembella, A.D., and Hallegraeff, G.M. (Eds.) 1998. Physiological ecology of harmful algal blooms. NATO ASI Series, Vol. G41. Springer-Verlag.

## 10.3 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 1998 report of the Working Group on Zooplankton Ecology (WGZE) and ACME deliberations.

### *Status/background information*

#### **Zooplankton methodology manual**

WGZE reviewed the progress in preparing the ICES zooplankton methodology manual. Most chapters have now been drafted and are in review or revision. Outstanding chapters will be reviewed and finalized during summer 1998. Publication arrangements with Academic Press are being made, with the aim of having the manual published in spring 1999.

#### **Zooplankton taxonomic skills**

WGZE have underlined the importance of the use of taxonomy in zooplankton ecology studies. There is a general trend of more emphasis on behaviour and population dynamics in ecological studies, e.g., in GLOBEC programmes. There is also the issue of biodiversity in pelagic ecosystems. Therefore, there is a strong case not only to preserve but also to improve

zooplankton taxonomic skills within the ICES community.

WGZE has considered a programme on zooplankton biogeography in the North Atlantic as a task on its long-term plan. Another issue raised by WGZE is the fate of old samples. The information contained in the samples collected throughout the years is formidable as it reflects the evolution of the pelagic community structures over those years. There is a need to keep archived samples in good condition and in a state where they would be available to the scientific community. This will allow the experts to conduct intercomparisons on the biogeography at decadal time scales and develop inferences on effects of climatic oscillations.

### *Need for further research or additional data*

The relationship between species diversity and functions of marine ecosystems is a wide and important area of research.

The ACME requested WGZE to produce a list of zooplankton taxonomic experts within the ICES area, as a basis for coordinated management of taxonomic skills. WGZE should also produce a checklist of the zooplankton species in the different parts of the ICES area, and a list of reference materials, as a contribution to quality assurance work on zooplankton species identification.

The ACME recommends that a special workshop on zooplankton taxonomy be held to prepare plans and proposals for future collective actions in this field. WGZE was requested to prepare a detailed proposal for the subjects and conduct of this workshop.

## 10.4 Effects of Fishing Activities on Benthos

### *Request*

There is no specific request; this is part of the continuing ICES work on studies of benthos in the marine environment.

### *Source of the information presented*

The 1998 report of the Benthos Ecology Working Group (BEWG), the 1997 report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO), and ACME deliberations.

### *Status/background information*

The ACME noted that the final report of the EU-AIR-funded project 'The effects of different types of fisheries on the North Sea benthic communities' (IMPACT II) has been completed and published (Lindeboom and de Groot,

1998). The recommendations given in this study are summarized below.

Specific recommendations from IMPACT II include the following:

- Mortality in invertebrate populations due to commercial trawl fisheries depends on (i) the spatial distributions of species and trawling effort of the different fleets, and (ii) the total direct mortality estimate. Management measures to reduce this fishing mortality have to be centred on reduction of trawling effort, spatial restriction (e.g., zonation) of a particular trawling effort, and reduction of the direct mortality rate (e.g., alternative gear design).
- The use of sampling gears suitable for specific fractions of the benthic fauna in monitoring studies of invertebrate populations in the North Sea will provide more appropriate data for the analysis of long-term changes. Traditional gears such as box corers and grab samplers are appropriate for small-sized infauna and epifauna, fine-meshed small beam trawls for fish and larger epifauna, and the Triple-D benthos dredge for larger-sized infauna and epifauna in sandy sediments. More attention should be devoted to the development of appropriate sampling gears for other types of sediments such as stony and very silty areas.
- To understand the long-term impact on the occurrence of individual species, more information on population dynamics of these species (effects of recruitment and size distribution, recovery time, succession patterns, etc.) should be collected.
- The extraction of more detailed information on long-term effects from historical data series should be continued.

General recommendations from IMPACT II include the following:

- Studies on the direct effects of fishing in areas which have been continually trawled in recent decades are inconclusive. Rare and long-lived species may already have disappeared, while relatively resistant species may predominate in the present-day fauna. More conclusive evidence for the long-term effects of beam trawling on the benthic ecosystem can only be obtained by studying relatively large areas closed to fisheries for many years.
- Research should be encouraged to reduce the destruction of potentially valuable undersized fish, as well as benthos and habitats. Alternative fishing methods should be developed.
- Studies on commonly overlooked parts of the benthic fauna, i.e., large and rare infauna and epifauna that may be vulnerable to fisheries, should be encouraged.

- For future studies examining the effects of fishing, more detailed information on the distribution of fishing effort in time and space is needed. It should be considered to equip all vessels with 'black boxes' to independently register their fishing activities.
- The development and application of indirect methods to estimate fishing intensity (marks in the shells of bivalves, lost arms of echinoderms) should be encouraged.
- Fisheries management should not only be concerned with fish stocks of commercial value, but also with ecosystem management.

The ACME noted that the equipment of fishing vessels of 24 m length and longer with 'black boxes' has been initiated as a result of an EU directive in connection with fisheries management, and is scheduled for implementation in mid-1998.

The ACME further noted the progress in an ongoing EU-FAIR project 'Monitoring epibenthic biodiversity in the North Sea using groundfish surveys', during which a protocol for monitoring epibenthic biodiversity in the North Sea and Skagerrak has been developed. While the analysis of the project results is still under way, a proposal for an epibenthos biodiversity monitoring project, to be carried out jointly with the third quarter International Bottom Trawl Survey (IBTS) groundfish survey (coordinated by ICES), will be submitted to the EU in 1998. As an important result of the project, otter trawl and beam trawl effort in the North Sea was collated from national databases by ICES statistical rectangle, covering the period since the early 1980s for which such information is available (Jennings *et al.*, 1998).

While fishing is an important direct and indirect source of benthos mortality, predation mortality by benthic-feeding fish is even more important. Changes in the structure of the North Sea fish population, with a decrease in the abundance of gadoids and an increase in flatfish, are likely to have increased the predation pressure on benthos. At present population levels, it is estimated that 20–45 % of the benthic production is consumed by the eight principal benthic-feeding fish species. Although two years of conducting a fish stomach sampling programme in the North Sea and research associated with the Multispecies Assessment Working Group (MAWG) are beginning to give a better picture of the consumption rates of many fish species, a principal source of uncertainty in these estimates lies in the limited knowledge about benthos production.

Fishing gears may cause damage to substrates and benthic habitats by altering sediment structure and destroying benthic organisms; the magnitude of the effects depends on the type of gear and the type of habitat. Models of fish consumption and predation do not

account for physical changes to benthic habitats caused by trawls and dredges. The homogenization of substrates and removal of biogenic structures, particularly long-living epifauna, can alter benthic habitats, reduce their complexity, and reduce sheltering opportunities for benthic and demersal organisms. The destruction or loss of benthic species can reduce the complexity of food webs, making predators more dependent on fewer types of prey. These structural changes may have long-term, adverse effects on benthic community structure and production rates, which may ultimately affect the productivity of higher trophic levels.

The ACME also took note of the ICES Symposium on 'Benthos Ecosystem Dynamics: Environmental and Fisheries Impacts', to be held on Crete, Greece, from 5-7 October 1998. The programme includes the following:

- evaluation of direct and indirect effects of fisheries-related and other anthropogenic perturbations on the benthos;
- ecosystem modelling of fisheries-related impact on the benthos;
- food chains and energy flow in fisheries-perturbed benthic ecosystems;
- long-term and short-term consequences of fishery activities upon the benthos and the benthic environment;

- disturbance due to natural and anthropogenic causes, and recovery of benthos communities after cessation of disturbance;
- temporal variation in benthos communities: mechanisms and causes (including experimental approaches).

#### *Recommendations*

As the need for ecosystem management will become more pressing in the near future, ICES recommends that research projects developing tools for ecosystem management in the ICES area be supported by ICES Member Countries to the largest possible extent.

#### **References**

- Lindeboom, H.J., and de Groot, S.J. (Eds.) 1998. The effects of different types of fisheries on the North Sea and the Irish Sea benthic ecosystems. Netherlands Institute of Sea Research (NIOZ) Report 1998(1).
- Jennings, S., Alvsvåg, J., Cotter, A.J., Ehrich, S., Greenstreet, S., Jarre-Teichmann, A., Mergardt, N., Rijnsdorp, A., and Smedstad, O. 1998. International fishing effort in the North Sea: an analysis of spatial and temporal trends. Fisheries Research (in press).



**11.1 Schemes for the Identification of Priority Contaminants***Request*

There is no specific request; this is part of the continuing ICES work on contaminants, and is also of interest to the OSPAR Commission and the Helsinki Commission.

*Source of the information presented*

The 1998 report of the Working Group on Environmental Assessment and Monitoring Strategies (WGEAMS) and ACME deliberations.

*Status/background information*

The identification of priority contaminants is a necessary aspect of the strategic planning of marine monitoring programmes to ensure that resources are allocated to the most important groups of substances. The ACME prepared a review of the available prioritization schemes in 1992 (ICES, 1992) and is aware that prioritization has been the subject of considerable discussion at several recent HELCOM and OSPAR Working Group and Committee meetings. The OSPAR strategy document on hazardous substances covers a very full list of candidate hazardous substances which has been developed by collating lists from a number of different sources. The ACME noted that the OSPAR Programmes and Measures Committee (PRAM) was to consider the formation of an OSPAR Ad Hoc Working Group specifically to develop a dynamic selection and prioritization mechanism (DYNAMEC) for hazardous substances, and the OSPAR Working Group on Concentrations, Trends and Effects of Substances in the Marine Environment (SIME) has developed a series of action steps to aid the prioritization process.

The ACME made several observations on the OSPAR strategy:

- The prioritization scheme places considerable weight on acute toxicity effects, for example, through the use of toxicity threshold levels in accordance with the procedure for the development of Ecotoxicological Assessment Criteria (EACs). In the light of the central role that field biological observations have played in the identification of significant problem chemicals, such as TBT and environmental oestrogens, the ACME noted that the prioritization scheme should take account of field observations in addition to desk-based assessments.
- OSPAR documents define 'toxicity' in a very broad way to include carcinogenicity, mutagenicity, and teratogenicity in a classification to encompass acute, sub-chronic, and chronic effects. OSPAR has

developed a special strategy to ensure the adequate assessment of endocrine disruptors. The ACME endorsed this broad view of toxicity and recommended that OSPAR ensure the effectiveness of the proposed prioritization scheme in relation to compounds which exert chronic toxicity, for example, carcinogenicity.

*Additional comments*

The definition of an 'endocrine disruptor' used by OSPAR is broad, but the evaluation criteria for endocrine disruptors are strongly directed to sex hormones. Tools are required to detect the disruption of other hormone (e.g., thyroid) systems.

The ACME noted that some of the terminology in the OSPAR list of candidate compounds could usefully be updated, as it reflects the age of some of the source documents. For example, the generic term 'PAHs' is not very useful; it would be better to refer to specific compounds or subsets of compounds.

The ACME considered it unlikely that the proposed prioritization system could take account of synergism, antagonism, and other similar interactions between contaminants.

**Reference**

ICES. 1992. Report of the ICES Advisory Committee on Marine Pollution, 1992. ICES Cooperative Research Report, 190: 62-67.

**11.2 Procedures to Assess the Combined Effects of Exposure of Organisms to Contaminants***Request*

There is no specific request; this is part of the continuing ICES work on monitoring the biological effects of contaminants, and is also of interest to the Commissions.

*Source of the information presented*

The 1998 report of the Working Group on Environmental Assessment and Monitoring Strategies (WGEAMS) and ACME deliberations.

*Status/background information*

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. The regulation of waste discharges to the sea often depends upon controls on the concentrations of individual substances, based on some kind of assessment of the toxicity and other properties of the substances concerned. Only rarely are interactions, synergistic or antagonistic, taken into account. One recent advancement in this direction is the gradual development of discharge regulations based on the toxicity of whole effluents rather than the detailed (and usually incomplete) chemical composition of the effluents.

This issue is particularly important in OSPAR, where schemes are being developed to control the discharge of complex chemical streams from offshore oil platforms, i.e., produced water (see also Annex 7), using hazard and risk assessment methods (e.g., Chemical Hazard Assessment and Risk Management Model (CHARM)). These methods presently treat the component chemicals separately when there is no knowledge about synergistic or antagonistic effects.

WGEAMS approached this topic by reviewing several studies on interactions between groups of contaminants. The following information contains examples of procedures that have been tested to investigate these interactions.

The first example concerns groups of contaminants which produce similar responses in biological test systems. In such circumstances, there is the possibility of assessing additive effects, provided that the relative potency of the different compounds is known. The example described below is based on the P4501A1 response to xenobiotics. It is known that dioxins effectively induce this enzyme system, as do some chlorobiphenyl (CB) congeners. A system of Toxic Equivalent Factors (TEFs) has been developed, using 2,3,7,8-tetrachlorodibenzodioxin (2,3,7,8-TCDD) as the reference compound, to allow the total effective toxicity of mixtures of dioxins and CBs to be estimated. However, it is also known that some polycyclic aromatic hydrocarbon (PAH) compounds can induce the same system. There is a system of factors linking PAH compounds, but this is based on benzo[a]pyrene as the reference compound and, therefore, is not directly linkable to the dioxin-based system. A recent study (Hess, 1998) on samples from the Firth of Clyde in Scotland revealed the following:

'A common mechanism of action in the toxicity of planar aromatic compounds was their interaction with the P4501A enzyme system. A model inducer of the P4501A system is 2,3,7,8-TCDD. The toxicities of PCDD/Fs and CBs had been previously expressed as toxic equivalent factors (TEFs) for individual congeners relative to 2,3,7,8-TCDD. No TEF system had been proposed that related PAHs to 2,3,7,8-TCDD.

A comparison of benzo[a]pyrene (B[a]P) and 2,3,7,8-TCDD was carried out using the H4IIE rat liver cell line. The ED<sub>50</sub>s resulting from this comparison were used to calculate a TEF for B[a]P, which equalled 0.001, relative to 2,3,7,8-TCDD.

Thus, an existing TEF system that compared the toxicity of individual PAHs relative to B[a]P could be converted to a TEF system relative to 2,3,7,8-TCDD thereby creating a comparative system for all the planar aromatic compounds considered in this thesis. This system was used to express the chemical concentrations of individual compounds in sediments as toxic equivalents (TEQs). These individual TEQs were summed up to represent the total toxicity of a group of compounds to the P450 enzyme system, such as CBs, PCDD/Fs or PAHs or all the planar aromatic compounds measured (CHEM-TEQ).

The contribution of CBs to the total CHEM-TEQ was < 1 %. The group of PCDD/Fs was the next highest contributor with up to 5 % of the total. PAHs accounted for the largest part of dioxin-like toxicity (94–99 %) on the eight stations sampled.

Enzyme induction measured in cell culture tests was used to estimate the integrative toxic threat which the contaminants pose to the environment.

The dioxin-like toxicity measured in this way was of the order of 100 ng kg<sup>-1</sup> to several µg kg<sup>-1</sup> dry sediment, called EROD-TEQ. The highest EROD-TEQ was found in sediments originating from the sewage sludge dump site. This site was also the most contaminated with all groups of compounds.

The EROD-TEQ and the CHEM-TEQ showed positive correlation with each other at significance levels over 90 %. Including PAHs in the assessment of toxicity resulted in explanation of two-thirds of the measured toxicity.

The prediction interval on the correlation between EROD-TEQ and CHEM-TEQ was wide. The error on the prediction was partly due to the absence of TEFs for alkylated PAHs and partly to the high analytical error on the EROD-TEQ. The prediction of biological effects from chemical measurements alone is difficult, but can be helpful for regulatory purposes. The fact that PAHs, especially B[a]P, accounted for the largest part of the toxicity to the P450 enzyme system was one of the major conclusions of this thesis. This suggests that the significance of PAHs should be more studied in the environment. The relative toxicities

of alkylated PAHs which occur at concentrations similar to the parent compounds are unknown and warrant further study.'

The above example concerns a single biological effect and the additive nature of the interactions of members of three major groups of contaminants (dioxins, CBs, and PAHs). These compounds may be considered similar in that they are flat (planar) in shape and appear to interact with the same set of receptors. The second example of interactions also concerns EROD (P4501A1) induction, but it is an inhibitory rather than an additive interaction. There are a number of published accounts of inorganic substances inhibiting the activity of the P450 system. For example, Viarengo *et al.* (1997) reported that ionic copper, mercury, and methylmercury at nanomolar concentrations partly inhibit EROD activity in fish liver microsomes treated with  $\beta$ -naphthoflavone or benzo[a]pyrene as inducers. Activity was completely inhibited at micromolar concentrations, and the effect of the metals was reduced by the addition of GSH. Similarly, George (1989) demonstrated that cadmium ions reduced the P450 response in plaice, confirming previous studies in the same species (George and Young, 1986) and in rainbow trout.

The final example from the WGEAMS report concerns an investigation of the causes of an unusual biological effect observed in a wild fish population. Adult Atlantic salmon migrating up the Don River in Scotland in the 1970s and early 1980s were found to develop a non-infectious haemolytic anaemia with associated jaundice. The condition was striking from external examination in that yellow/orange/red pigments accumulated in the salmon skin, particularly around the eyes, fin bases, and the ventral surface. Extensive field and laboratory investigations into the cause of the effect were carried out. It was concluded that the condition arose from the exposure of non-feeding salmon to water-soluble fractions of diesel oil from various industrial sources and resin acids from wood pulp mill effluents. The effect was not caused by either of these two groups of substances in isolation, but arose after sequential or simultaneous exposure to diesel oil and resin acids (Croce and Stagg, 1997; Croce *et al.*, 1997).

Another large-scale field study examined interactions between biomarkers and sets of contaminants in the field. This report from Chang *et al.* (1996) associates liver lesions in winter flounder *Pleuronectes americanus* with chemical contaminants in sediments from the northeastern United States. Samples of sediments and demersal fish tissues were collected at ten estuarine and coastal sites, ranging from grossly polluted to relatively unimpacted. The data set consists of 54 contaminant variables measured in sediments (18 PAHs, 15 pesticides, 8 CBs, and 13 metals) and ten lesions in winter flounder liver sections. The following statistical procedures were used to analyse the data: multivariate

statistical methods described by Cooley and Lohnes (1971) were used to explore associations between the liver lesions and chemical contaminants. Factor analysis was used to reduce the original variables to factors (i.e., principal components) representing linear combinations of the original variables. Canonical correlation analysis then used the results of the factor analysis to explore possible interrelationships between liver lesion factors and chemical contaminant factors. Bartlett's procedure (1947) was used for testing for interdependency of the data sets, indicated as correlations between the two sets of variables.

In general, inflammatory liver lesions showed strong positive associations with low molecular weight petroleum-derived PAHs, tri- to hexachlorobiphenyls and the heavy metals chromium, cadmium, lead, thallium and selenium, but were negatively associated with DDT-type pesticides. Neoplastic lesions showed strong positive associations with most PAHs measured, whether petroleum or combustion derived, the pesticides dieldrin, *trans*-nonachlor and  $\alpha$ -chlordane, and silver, copper, antimony and tin, but no associations with PCBs were found. Necrotic lesions showed strong positive associations with hepta- to nonachlorobiphenyls and arsenic, zinc, nickel and mercury, and strong negative associations with high molecular weight combustion-derived PAHs and DDT metabolites.

The authors concluded that the analysis reported here strongly implicates certain chemical contaminants as the causative agents of hepatic lesions in the winter flounder.

The ACME emphasized that these examples illustrate the complexity of the matter and noted that the nature of the interactions between contaminants is by no means always predictable and that elucidation of interactions normally required a well-planned and careful experimental programme. These processes have generally not yet been taken into account in the development of discharge control standards or international monitoring programmes.

#### *Need for further research or additional data*

WGEAMS intends to revisit this important topic at its 1999 meeting. The ACME encourages this initiative.

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### 12.1 Assessment of the Impact of Incidental Mortality on Marine Mammals

#### *Request*

Item 4.2 of the 1998 Work Programme from the OSPAR Commission: to provide information on incidental mortality of marine mammals owing to fishing activities on a species (and gear type) basis for each of the five OSPAR regions.

#### *Source of the information presented*

The 1998 report of the Working Group on Marine Mammal Population Dynamics and Trophic Interactions (WGMMPD); ACFM and ACME deliberations.

#### *Status/background information*

There are very few estimates available of by-catches of marine mammals based on well-designed programmes in the five OSPAR regions. The available estimates are given in Table 12.1.1, below. These estimates do not cover all fisheries in the respective regions and they are, therefore, not representative of the total by-catch of the species in the region. However, the information reviewed here is probably the best available at present. An overall description of the main fisheries in each region is given in Section 13.1, below.

WGMMPD made the following comments on their findings:

#### Region I. Arctic Waters

Fisheries off Iceland (where more than 200 incidentally-caught harbour porpoises were collected per year for biological studies) and fisheries by small vessels in Norway (where more than 56,000 harp seals were incidentally caught in one exceptional year) deserve closer investigation of by-catches.

#### Region II. Greater North Sea

The estimated annual by-catch of 4450 harbour porpoises in the central and southern North Sea comprises more than 2.6 % of the estimated number of porpoises in this area. This estimate is based on a Danish project and relates only to the Danish bottom-set net fishing effort, which is part of a larger fishery for sole, cod, and turbot in this area. The by-catch in the Swedish Skagerrak is likely to exceed 4 % of the population and this, coupled with evidence of declining numbers of porpoises in this area, would indicate that action is now needed to reduce the by-catch.

#### Region III. The Celtic Seas

The by-catch in fixed bottom-set nets in the seas south of Ireland is likely to place the population of harbour porpoises in this area at risk. The by-catch in the pelagic trawl fishery in the same area may be placing the populations of other delphinids at risk, also.

**Table 12.1.1.** Estimates of by-catches of marine mammals in the five OSPAR regions.

OSPAR Region	Species	Number by-caught per year	Gear type	Comments
I. Arctic Waters				There have been no direct assessments of by-catches of marine mammals in this region during the 1990s
II. Greater North Sea	Harbour porpoise	4450 (95 % confidence limits 2580–6320)	Bottom-set nets	Observer programme, Central North Sea, Danish fishery
		113	Gillnets	Observer programme, Swedish Skagerrak
III. The Celtic Seas	Harbour porpoise	2200 (95 % confidence limits 900–3500)	Bottom-set nets	Observer programme, estimate does not cover smaller boats
IV. Bay of Biscay and Iberian Coast				No reports of direct assessments of by-catches of marine mammals in this region
V. Wider Atlantic	Common dolphin	330–400	Drift nets	Observer programme, French fishery
	Striped dolphin	1135–1160	Drift nets	Observer programme, French fishery

#### Region IV. Bay of Biscay and Iberian Coast

There is evidence of by-catches in marine fisheries in this region, and trawl fisheries and gillnet fisheries should receive further investigation.

#### Region V. Wider Atlantic

Information from only one fishery (observer programme on a French drift net fishery for albacore tuna) was available from this area.

Some of the estimates above, although incomplete with regard to total by-catch, may give cause for concern about the populations of marine mammals impacted.

The ACME acknowledges the review undertaken by the WGMMPD on incidental mortality of marine mammals in fishing operations in the five OSPAR regions, noting the scarcity of data and varying quality of information available to the Working Group. The full review is attached as Annex 9.

#### *Recommendations*

ICES encourages Member Countries to monitor their fisheries to identify gear types, areas, and seasons where by-catches of marine mammals occur. In fisheries where the incidental mortality of marine mammals is significant, continued monitoring is recommended.

To assess the impact of by-catches on marine mammal populations, ICES further recommends that robust estimates of abundance and information on the distribution (stock identity) of affected species are obtained in addition to estimates of total by-catch.

### **12.2 Concentrations and Effects of CBs in Marine Mammals**

#### *Request*

Item 3.1 of the 1998 Work Programme from the Oslo and Paris Commissions: to collect information and data on concentrations of non-*ortho* and mono-*ortho* CBs in marine mammals and on any relevant biological effects, and to prepare a report on the findings and potential implications.

#### *Source of the information presented*

The 1998 reports of the Working Group on Marine Mammal Habitats (WGMMA), the Working Group on Environmental Assessment and Monitoring Strategies (WGEAMS), the Working Group on Biological Effects of Contaminants (WGBEC), and the Marine Chemistry Working Group (MCWG), and ACME deliberations.

#### *Status/background information*

Global contamination of the marine environment by persistent organochlorine compounds, such as chlorobiphenyls (CBs) and organochlorine pesticides (OCPs) including DDTs, is well documented. As a result of their lipophilicity and persistence, these compounds bioaccumulate in the food chain resulting in high concentrations in top predators such as cetaceans and pinnipeds. Some information on concentrations of CBs recorded in marine mammal tissues is given in Annex 10.

An experimental study on harbour seals (*Phoca vitulina*) showed that reproduction in this species was significantly impaired by feeding the seals diets of Wadden Sea fish containing high concentrations of both PCBs and DDTs. Reproductive failure in Baltic Sea grey seals (*Halichoerus grypus*) and ringed seals (*Phoca hispida*) has been ascribed to high concentrations of organochlorine contaminants. However, in these cases critical reviews concluded that it was impossible to confirm a cause-effect relationship linking the effects to specific contaminants.

Other effects attributed to organochlorine contaminants are related to cytochrome P4501A (CYP1A) enzyme induction, immune dysfunctions such as suppression of natural killer cell activity, and lower thyroid hormone and vitamin A concentrations. The biological effects of individual compounds are reviewed in Annex 10.

A number of studies have demonstrated a relationship between structure-binding and structure-activity relationships for several classes of halogenated aromatic compounds. These observations form the basis for the development of toxic equivalent factors for individual compounds, related to a compound with a well-documented toxic effect. The toxic equivalent factors are then used as multipliers for each congener identified in a sample, the summation of which gives an overall toxicity equivalence (TEQ). For example, there is clear experimental evidence that immune function in harbour seals is reduced when blubber lipid TEQ (mainly derived from mono-*ortho* CBs) exceeds about 70 pg g<sup>-1</sup>.

An estimation of TEQs in blubber lipid of harbour seals (from the Irish east coast, UK east coast, and Svalbard; data provided by the ICES Environmental Data Centre) showed TEQ values below critical levels suggested for this species. However, the results were not very conclusive due to the incomplete nature of the data and the small amount of dose-response information available.

It is the combined effects of toxic compounds and additional environmental factors that influence the populations of marine mammals. The review contained in Annex 10 provides a summary of the present knowledge,

and identifies further data and knowledge that are a prerequisite for a better understanding of this complex problem. This review paper is provided to OSPAR as the ICES response to item 3.1 on the 1998 Work Programme from OSPAR.

#### *Need for further research or additional data*

More experimental research is needed to improve the knowledge on dose-response relationships for marine mammals exposed to specific compounds, and comprehensive long-term studies are required on cause-effect relationships to link population-level effects to

environmental contaminants. More data on concentrations of organochlorine contaminants in marine mammals are also needed.

#### *Recommendations*

ICES recommends that Member Countries send data on concentrations of CBs in marine mammals, along with QA information and other relevant metadata, to the ICES Environmental Data Centre in order that a more complete assessment of the effects of CBs on marine mammals may be made in the future.

### 13.1 Impact of Fishing on Age/Size Distribution and Spatial Distribution of Five Fish Species in Five OSPAR Areas

#### *Request*

Item 4.1 of the 1998 Work Programme from the OSPAR Commission: to provide information on the impact of fishing activity on the growth and spatial distribution of the target fish population for commercially exploited stocks of fish and shellfish in the five OSPAR regions that are subject to regular assessment. OSPAR and ICES have agreed on the following species to be covered under this request: cod, herring, sole, mackerel, and hake.

#### *Source of the information presented*

The 1997 report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO), and ACFM and ACME deliberations.

#### *Status/background information*

It was noted that these requests from OSPAR appeared to be straightforward, but in fact they posed several problems. The interpretation of most information assembled required some knowledge of the present state and recent histories of the stocks and fisheries in the corresponding OSPAR region. OSPAR did not ask for specific narrative on these contextual matters, but the ACME felt strongly that such information should be provided. Hence, each portion of the response to this request by OSPAR region begins with a section on context. Presenting this context once in an integrated way greatly reduces the text needed for all the subsequent observations, as the context applies equally to the regional material in Sections 13.2 and 13.3, below, as well as in this section. **It is important that this context section be read in association with the respective parts of Sections 13.2 and 13.3, below, as well as with this section.**

The boundaries of the OSPAR regions are clearly differentiated (Figure 13.1.1). Unfortunately, the boundaries ICES uses in delineating Sub-areas and Divisions do not coincide with the OSPAR regions (Figure 13.1.2). WGECO assigned ICES Sub-areas and Divisions to OSPAR regions as shown in Table 13.1.1.

Biologically, the boundaries among fish stocks do not always (or often) coincide with the OSPAR regional boundaries. In some cases, single OSPAR regions contain several stocks. In those cases, WGECO felt strongly that data should not be combined across stocks due to the potential for errors of interpretation in both

directions. Combining data across stocks might conceal important information, by swamping trends in one stock with data from other stocks showing dissimilar trends, or a similar trend at a different time. It also might create the artificial impression that a trend in some trait existed when there was no trend. For example, if two stocks in a region differed in size at age, the appearance of a trend in size (or growth) could result from changes in the relative abundance of the two stocks whose size data were being combined.

**Table 13.1.1.** The relationship between OSPAR regions and ICES Sub-areas and Divisions used in this report.

OSPAR Region	ICES Sub-area/Division
I	I, IIa, IIb, Va, Vb1, Vb2, XIVa, XIVb
II	IIIa, IVa, IVb, IVc, VIIId, VIIe
III	VIa, VIIa, VIIb, VIIf, VIIg, VIIh, VIIj
IV	VIIIa, VIIIb, VIIIc, VIId, IXa
V	VIb, VIIc, VIIk, VIIle, IXb, X, XII

The reciprocal problem also exists. Particularly with highly migratory stocks such as mackerel and horse mackerel, a single stock might be present for all or various parts of the year in more than one region. The systems for collecting fisheries statistics usually try to capture the area being fished, so some portions of this request, as well as those covered in Sections 13.2 and 13.3, below, can be addressed directly for these stocks. To address other portions of the request on a regional basis would require making highly uncertain (and sometimes fictitious) partitioning among the regions. In those cases, the relevant data series are described and interpreted fully in one of the OSPAR regions, highlighting that a pattern is being discussed at a spatial scale different from the regional boundaries. In each of the regional sections, the data and information specific to the region are presented and discussed as far as can be done with reasonable scientific rigour, and readers are pointed to the regional section which contains the full treatment of the issue.

In a few cases, WGECO found the wording of the requests challenging. Some requests, if taken exactly as worded, would require very large amounts of work to fulfil. WGECO assumed that summaries of dominant patterns would provide the information generally needed, and prepared responses in that context. Nonetheless, it may be surprising how often the interpretation of a superficially simple request is actually quite complex. It may also be surprising how often there are very few sound data regarding topics which should be known much better.



Figure 13.1.1. Regions of the OSPAR maritime area (OSPAR, 1995).

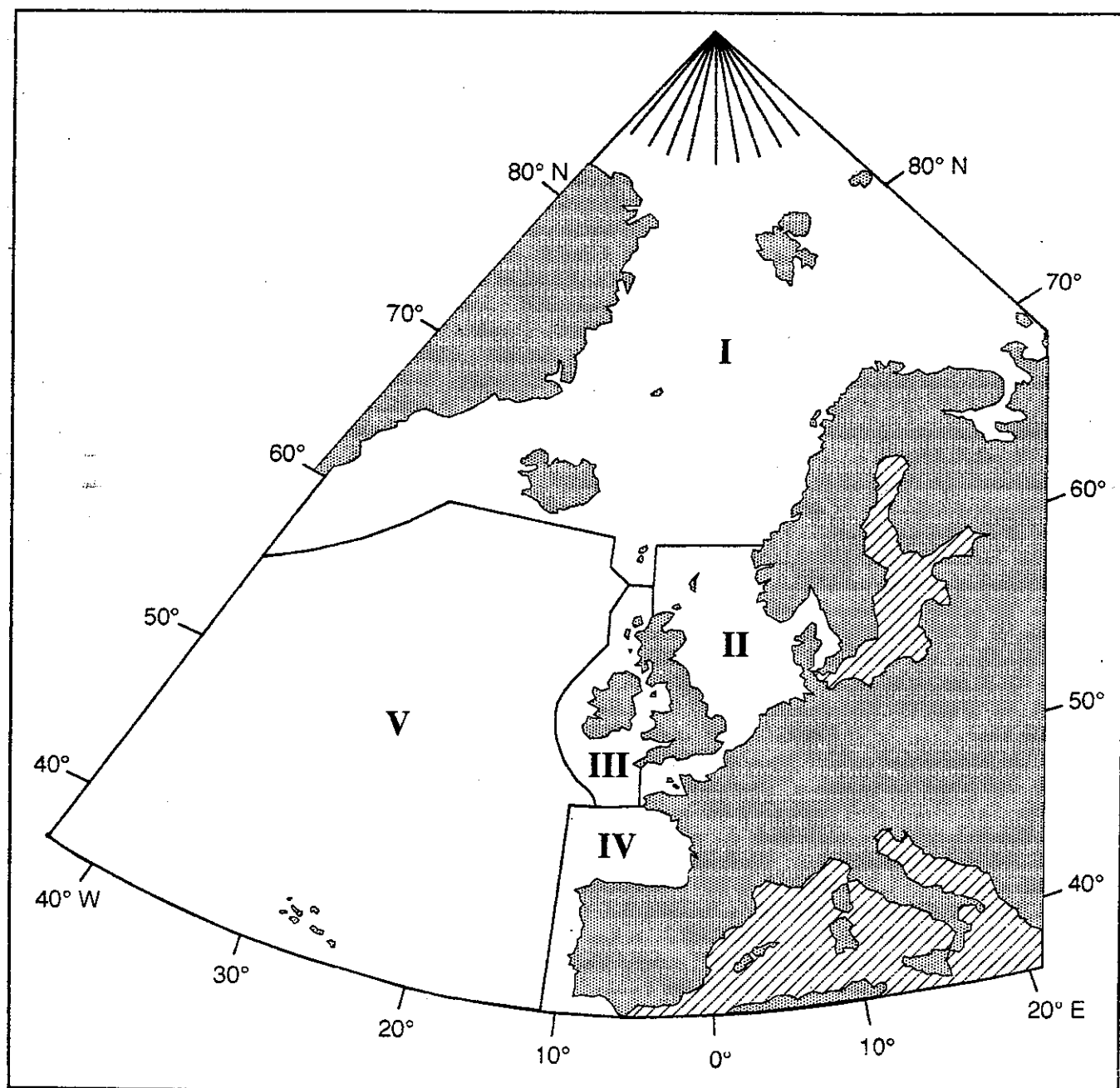
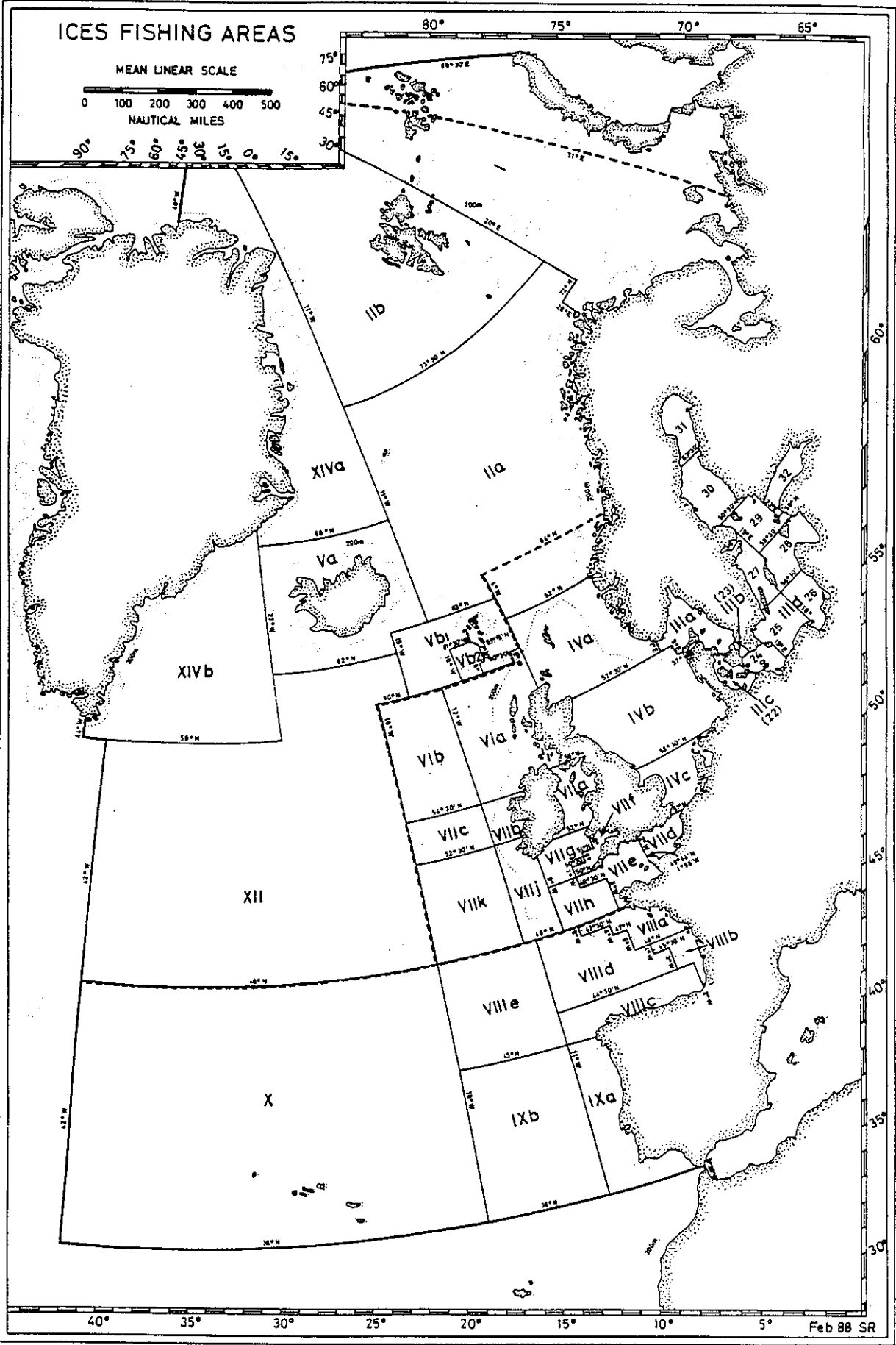


Figure 13.1.2. ICES Sub-areas and Divisions in the Northeast Atlantic.



Figures 13.1.1.1.1.2 and 13.1.1.1.1.3 illustrate the information that will be presented for each stock for which changes in age or size composition over time have been quantified. A full explanation of these figures can serve as a guide for the interpretation of all of the figures which follow in this section. In this case, the data on numbers of fish at age (used to create Figure 13.1.1.1.1.2) and biomass by age (Figure 13.1.1.1.1.3) come from estimates of the size of this stock, based on catch data and research survey data going back to 1946. Generally, age composition data will come from similar sources, although in a few cases, data will only come from a research survey series, because complete analytical assessments using catch and research data are not possible. As young cod in this stock are not well represented in fisheries, the information on numbers of fish at each age starts at age 5, and no information can be presented for younger ages.

The y-axis in Figure 13.1.1.1.1.2 represents the percentage of the stock at a particular age (or, in some cases, groups of ages), according to the legend provided. The x-axis represents time, in years. Although the percentages are treated as continuous, the estimates are made only once per year, for the same month, with linear interpolation between the annual estimates. The absolute stock size changed substantially over the period 1946–1997, as explained in the context, but those changes are not pictured in the figures. Rather, because the percentages at age always sum to 100 for a single year, the dynamics in the figure only represent changes in age composition, which was the information requested.

In 1946, at the end of the Second World War, the Northeast Arctic cod stock had been nearly unfished for several years. Correspondingly, nearly 40 % of the numbers of individual fish and over 60 % of the biomass consisted of fish age 9 or older. (Because older fish are larger than younger fish, they will always comprise a higher percentage by weight than by numbers.) However, by 1954, fish age 9 and older comprised less than 10 % of the population, both in numbers and in weight. This reduction in the percentage of older fish was almost certainly a consequence of the rapid escalation in fishing after the war and of fishing mortality, which has been generally high since the 1950s. High fishing pressure results in high total mortality for a stock which, consequently, reduces life expectancy of individuals. This, in turn, reduces the relative occurrence of older ages.

In many of the figures which follow, this change in proportion of older fish as a consequence of fishing will not be obvious. This is likely to be because the time series are shorter and the abundance of older fish was greatly reduced before comprehensive sampling data on age composition became available. Where possible, WGECO has estimated the age composition that the stock would have had if it were not fished (i.e., fishing mortality = 0.0) and if natural mortality at age were exactly the rate assumed in the assessments of the stocks

at present. Although this assumption may not be completely sound, in that it implicitly assumes that there would be no compensation in other sources of mortality if fishing were eliminated, the estimates give a general picture of what the unfished condition might have been.

The part of the figure covering the period since the mid-1950s is more comparable to the other figures in this section. Here, the major variability in stock dynamics has been occasional pulses of strong or weak recruitment, which are respectively the large peaks and the troughs in the lowest band in each figure. The extreme values of year class sizes can be tracked over time; for example, the very poor year classes which recruited as age 5s in 1970 and 1971 (i.e., the very low percentage of the stock at age 5 in those years) also appear as particularly narrow bands (lower percentages) of age 6s in 1971 and 1972, narrower bands yet (even lower percentages) of age 7s in 1972 and 1973, etc.

Although there is a high variability in the recruitment signal over the forty years covered by the figures, after the mid-1950s there seems to be little signal which could be attributed to fishing. Because all the ages have to total 100 %, when recruiting year classes are very weak or very strong, older fish have to comprise proportionately slightly more or less, respectively, of the population. Otherwise, there are no strong patterns present over time in the older age groups. Especially in the figure for biomass (Figure 13.1.1.1.1.3), there may be a slight tendency for older fish to comprise somewhat less of the population in recent decades but, statistically, such a tendency is weak and indistinct compared to the recruitment signal.

These two important factors will dominate many other figures throughout this section. Specifically, the variability in fishing mortality over the time period is generally much smaller than the contrast in the recruitment signal. Also, most of the contrast in intensity of fishing occurred prior to the mid-1950s, so time series which start after that period may convey little information about how fishing may have affected age composition historically. Rather, what time series from the second half of this century are likely to demonstrate is that recruitment has been highly variable and older fish are consistently relatively uncommon. Many studies indicate that very strong fluctuations in recruitment have a strong environmental component (e.g., effects of climate) (Cushing, 1996), and since these signals swamp the small variability in fishing pressure over the same period, age and size composition data of individual stocks over recent decades are likely to contain little extractable information about the effects of fishing. Only if fishing intensity were changed abruptly, by a large amount, and then held at the new level for some years, would it be likely that identifiable patterns of response would appear in the age or size composition data from single stocks.

## References

Cushing, D. 1996. Towards a science of recruitment in fish populations. Excellence in Ecology, Vol. 7. Ecology Institute, Oldendorf/Luhe, Germany. 175 pp.

OSPAR. 1995. Assessment and Monitoring: The Joint Assessment and Monitoring Programme. Oslo and Paris Commissions. London, UK. 21 pp.

### 13.1.1 OSPAR Region I: Arctic Waters

This region includes the northeast Arctic (Sub-areas I and II), the Faroe Islands (Division Vb), and northwestern areas (Division Va and Sub-area XIV).

The total landings of fish and invertebrates in the northeast Arctic in 1996 were 2.5 million t. The major demersal fish stocks include cod (*Gadus morhua*), haddock (*Melanogrammus aeglefinus*), saithe (*Pollachius virens*), redfish (*Sebastes marinus*), and Greenland halibut (*Reinhardtius hippoglossoides*) and in 1995 landings of 1.1 million t were taken from these stocks. The main fleets exploiting the demersal species are factory and freezer trawlers, fresh fish trawlers, a fleet of vessels using conventional gears (gillnets, longlines, handlines, and Danish seines) and small purse seiners targeting saithe in coastal waters. The last two fleets account for approximately 30 % of the landings of demersal stocks. The major pelagic stocks are Norwegian spring-spawning herring (*Clupea harengus*) and capelin (*Mallotus villosus*), and they are exploited by purse seines and pelagic trawls. In 1996 the landings of herring were 1.2 million t, while there is no fishing on capelin at present.

The major demersal stocks at the Faroe Islands are cod, haddock, and saithe. The main gears used are bottom trawls, longlines, and jigs. The total demersal catches decreased from 120,000 t in 1985 to 65,000 t in 1994, but have since increased again to 83,000 t in 1996.

In the northwestern areas, the main demersal stocks are cod at Greenland and Iceland, saithe, and oceanic redfish. The demersal species are mainly exploited by stern trawlers, but considerable fisheries for cod are also carried out by longlines, gillnets, handlines, and Danish seines. There is also a purse seine fishery on Icelandic summer-spawning herring and capelin in the Iceland/East Greenland/Jan Mayen area.

## 13.1.1.1 Cod

### 13.1.1.1.1 Northeast Arctic cod

#### Context - Pattern of numbers, biomass, landings, and fishing mortality

From a level of about 900,000 t in the mid-1970s, landings (yield) declined steadily to around 300,000 t in 1983–1985 (see Figure 13.1.1.1.1.1). Landings increased to above 500,000 t in 1987 before dropping to 212,000 t in 1990, the lowest level recorded in the post-war period. The catches increased rapidly from 1991 onwards, and were stable around 750,000 t in 1994–1996 (ICES, 1998a), while the Total Allowable Catch (TAC) for 1997 was 850,000 t. The average rate of mortality inflicted by the fishery (commonly designated as  $F^*$ ) on fish aged 5–10 years (age ranges are chosen to include the sizes of fish likely to be taken by the fishing gears) increased almost continuously from a level of about 0.2 in 1946 to 0.9 at the end of the 1970s. In the years 1981–1989, the average  $F$ s were in the range 0.7 to 1.0. In 1990 fishing mortality dropped to 0.28 as a result of management measures brought into effect to control the amount of fishing effort. Age 5–10  $F$  then increased, reaching 0.76 in 1994, but dropped again to 0.58 in 1996. The present exploitation is well above both the  $F_{\max}$  and  $F_{\text{med}}$  levels of 0.26 and 0.46, respectively ( $F_{\max}$  is a reference fishing mortality derived from analysis of the theoretical potential yield of the stock;  $F_{\text{med}}$  is a reference fishing mortality based on the past history of the stock and fishery), which means that there is a potential for increased yields by lowering the fishing mortality. The minimum biologically acceptable level (MBAL) for the spawning stock biomass (SSB) of Northeast Arctic cod is 500,000 t. Since 1991 the SSB has been above this level, and most of the year classes produced in the 1990s have been strong at the 0-group stage. With the 1997 level of exploitation ( $F_{97} = 0.67$ ), however, the SSB is expected to drop below 500,000 t in the year 2000. ICES recommended that the fishing mortality should be reduced to below  $F_{\text{med}}$ , corresponding to landings in 1998 of no more than 514,000 t. The TAC agreed by the management authorities, however, was 654,000 t.

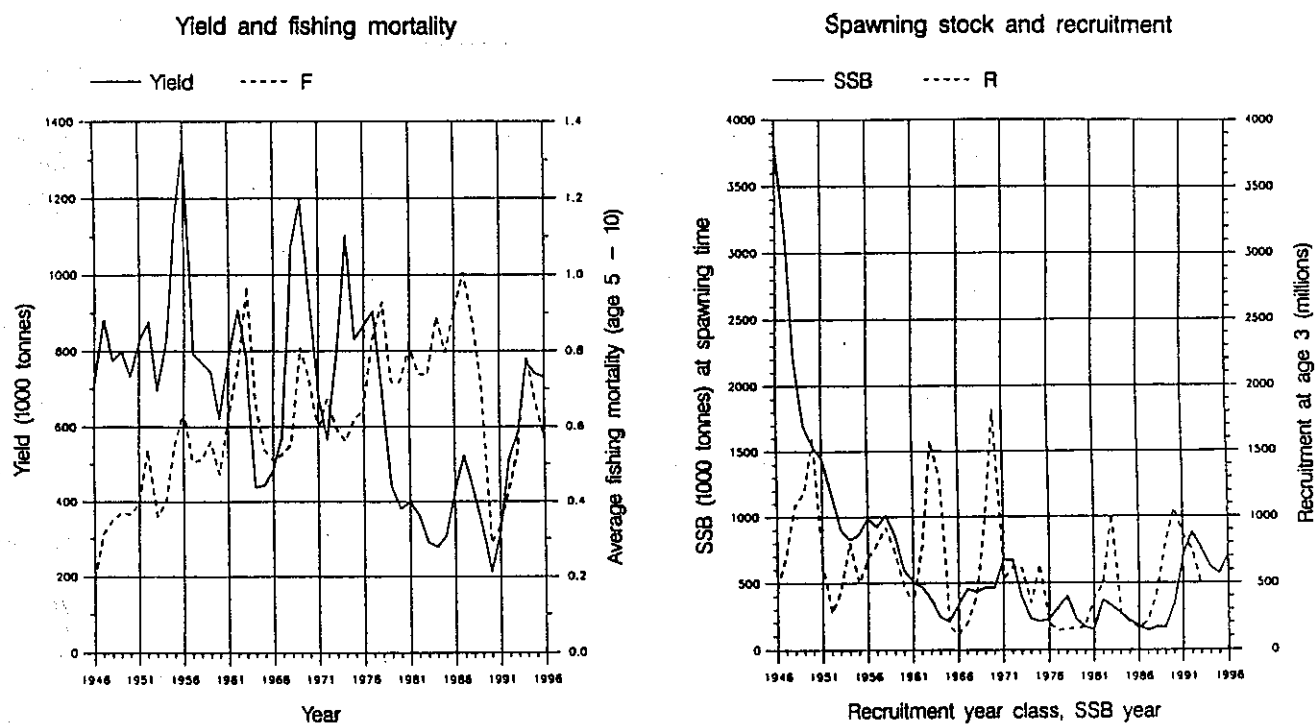
#### Changes in size distribution and/or age composition

Current landings are dominated by the strong 1990 year class. In recent years, landings have usually been concentrated on one or two strong year classes. The

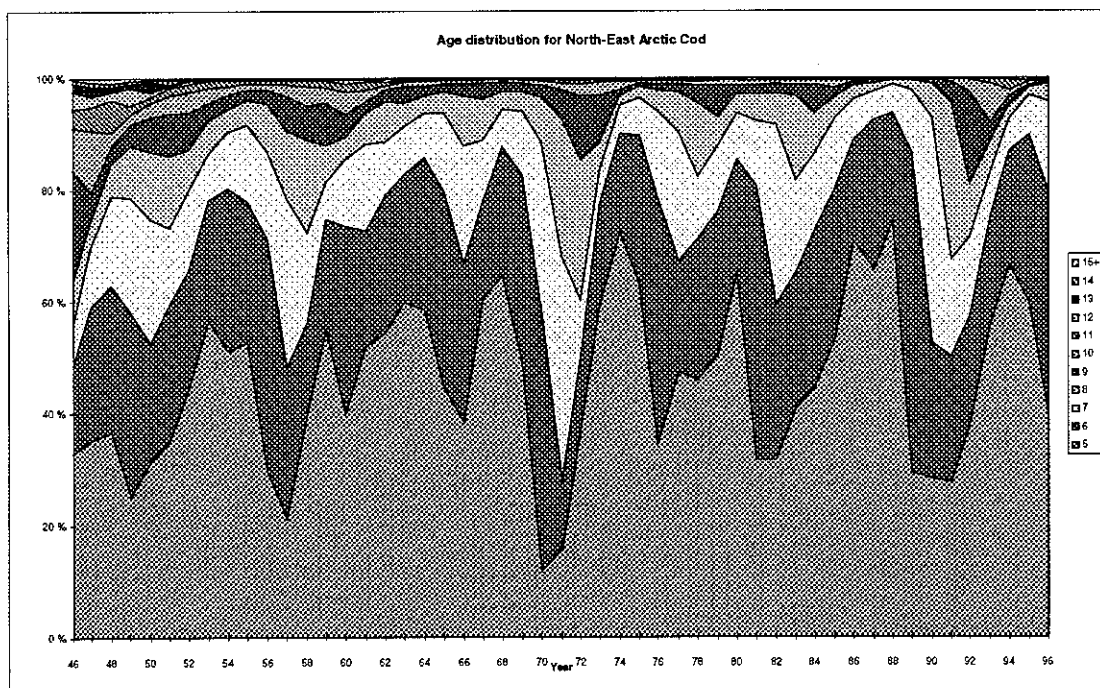
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\* Fishing mortality is a measure of fishing intensity. The fishing mortality is defined to be between zero and infinity; zero corresponds to no fishing, while an infinite fishing mortality corresponds to the removal of all fish from the stock.

**Figure 13.1.1.1.1.1.** Landings (yield), fishing mortality, spawning stock biomass, and recruitment for cod in the Northeast Arctic from 1946–1996.



**Figure 13.1.1.1.1.2.** Observed age distribution Virtual Population Analysis (VPA) for ages 5 and up of Northeast Arctic cod from 1946–1996. The key indicates age in years.



strong year-to-year variation in year class strength leads to changes in size and age distributions of such a magnitude that it, to some degree, will mask the effects of changes in total fishing effort or fishing mortality. The numbers at age and biomass (by age) distributions for the stock are shown in Figures 13.1.1.1.1.2 and 13.1.1.1.1.3, respectively. The distributions are clearly dominated by the large changes in recruitment. The effect of the fishery can somewhat be explained by looking at Figure 13.1.1.1.1.4. This graph shows the average age and biomass distributions for three different periods. The first period is 1946–1947, which represents the closest one can come to an unexploited stock through this time series. This period is just after the Second World War when the fishing effort in the Barents Sea was negligible and the only fishery with any impact was the fishery in Lofoten during the spawning season. The period from 1974–1975 represents a time with very high fishing effort, while the last period (1992–1993) represents the stock just after a short period of reduced effort following the collapse in the late 1980s.

#### **Changes in spatial distribution**

The Northeast Arctic cod is a highly migratory stock with both feeding and spawning migrations. Recent survey results have shown that the area occupied by the stock increases with increasing stock size (Jakobsen *et al.*, 1997). This increase in area should also be expected to be related to climatic conditions. That is, favourable climatic conditions (higher temperatures) seem to coincide with good recruitment, high growth, and increasing stock size. Due to the strong migrations, a sustainable fishery is not likely to affect the spatial distribution directly. However, if the spawning stock is largely reduced and fewer recruits are produced, the extension of both the spawning areas and nursery areas will be reduced. An attempt to illustrate the changes in distribution together with changes in abundance is presented in Figure 13.1.1.1.1.5. The yearly (February) abundance indices from the Norwegian bottom trawl survey for age groups 3 to 8 are shown together with the corresponding estimated percentages of the stock occupying the eastern part of the Barents Sea (the survey area east of 30°E). The impact of the strong 1983 and 1990 year classes dominates the abundance indices shown. The easterly distribution has an increasing trend from the late 1980s to the mid-1990s, with a tendency for local 'peaks' in 1989 and 1994. The data on percentages distributed in the east are also shown in Figures 13.1.1.1.1.6 and 13.1.1.1.1.7. The first figure shows the year class (cohort) effects, while the second figure clearly demonstrates the age effect. That is, older fish are distributed further west than younger year classes. It is stressed that the distributional data presented represent only year-to-year effects between the February observations. Seasonal variations showing spawning and feeding migrations are not presented.

#### **13.1.1.1.2 Icelandic cod**

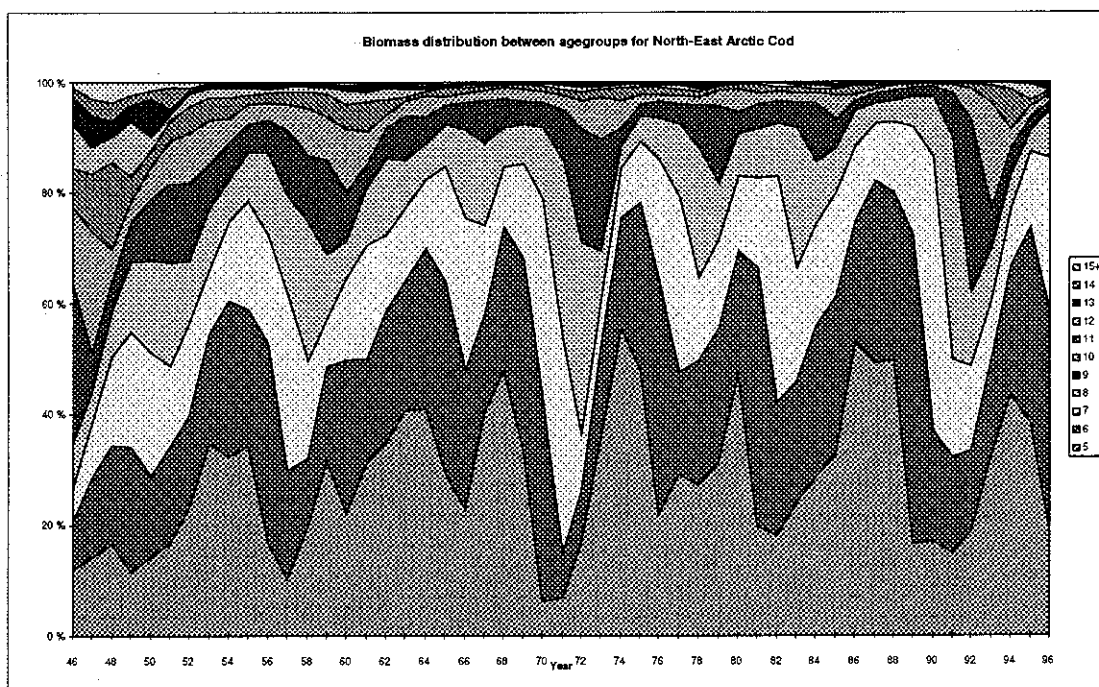
The fleet fishing for cod at Iceland operates throughout the year (ICES, 1997). The gears used for catching cod are longlines, bottom trawls, gillnets, handlines, and Danish seines. The fishing vessels are of different sizes but can, however, be grouped into three main categories: trawlers (> 300 GRT) (gross registered tonnes), multi-gear boats (< 300 GRT), and small boats (< 20 GRT). The trawlers operate throughout the year outside the 12-mile limit. They follow the spawning and feeding migration patterns of cod and fish on spawning grounds off the southwestern and southern coasts during the spawning season, but move to feeding areas off the northwestern coast during the summer time. During the autumn, this fleet is more spread out. The multi-gear boats operate mainly using gillnets during the spawning season in winter and spring along the southwestern coasts, but in recent years this fleet has also used gillnets in late autumn. Part of this fleet uses longlines during autumn and early winter. During summer some of these boats trawl along the coast out to the 3-mile limit. Others fish with Danish seines close to the shore. Most of the smaller boats operate with handlines mainly in shallow waters during the summer and autumn periods.

#### **Context - pattern of numbers, biomass, landings, and fishing mortality**

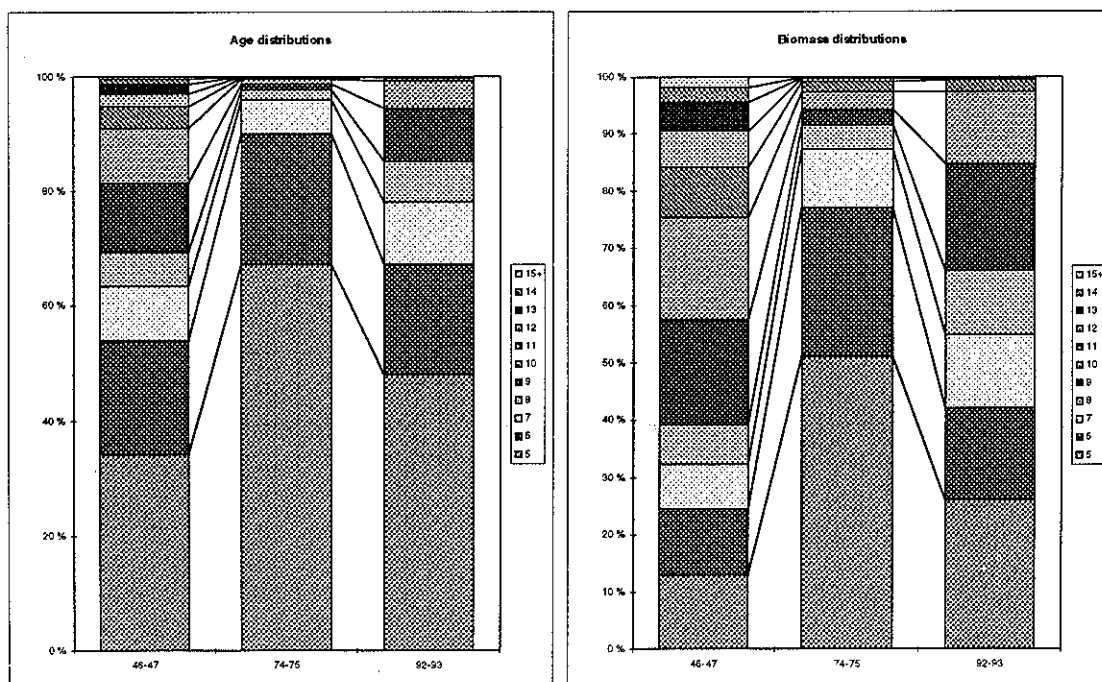
In the period 1978–1981, landings of cod increased from 320,000 t to 469,000 t due to immigration of the strong 1973 year class from Greenland waters combined with an increase in fishing effort. Catches then declined rapidly to only 280,000 t in 1983. Although cod catches have been regulated by quotas since 1984, catches increased to 392,000 t in 1987 due to the recruitment of the 1983 and 1984 year classes to the fishable stock in those years. Since 1988, all year classes entering the fishable stock have been well below average, or even poor, resulting in a continuous decline in the landings. The 1995 catch of only 170,000 t is the lowest catch level since 1942. Effort on cod in 1994 decreased compared to 1993. This trend continued in 1995 and a marked reduction in effort against cod has taken place in the most recent years due to further reductions in quotas and a diversion of the effort towards other stocks and areas. Due to an increase in the fishable stock biomass, the quota for the 1996/1997 fishing year was set at 186,000 t. Landings in 1996 increased accordingly to 182,000 t. This led to a slight increase in effort by the gillnet fleet, but the effort of the longliners declined compared to 1995, and the effort of the trawlers was unchanged between these years.

The Icelandic cod stock reduced in numbers from about 600 million to 300 million individuals during the period 1977 to 1994 (Figure 13.1.1.1.2.1), but increased to 400 million in 1996. The SSB was below 300,000 t during large parts of this period, but since 1993 the SSB has been growing and has now exceeded 300,000 t.

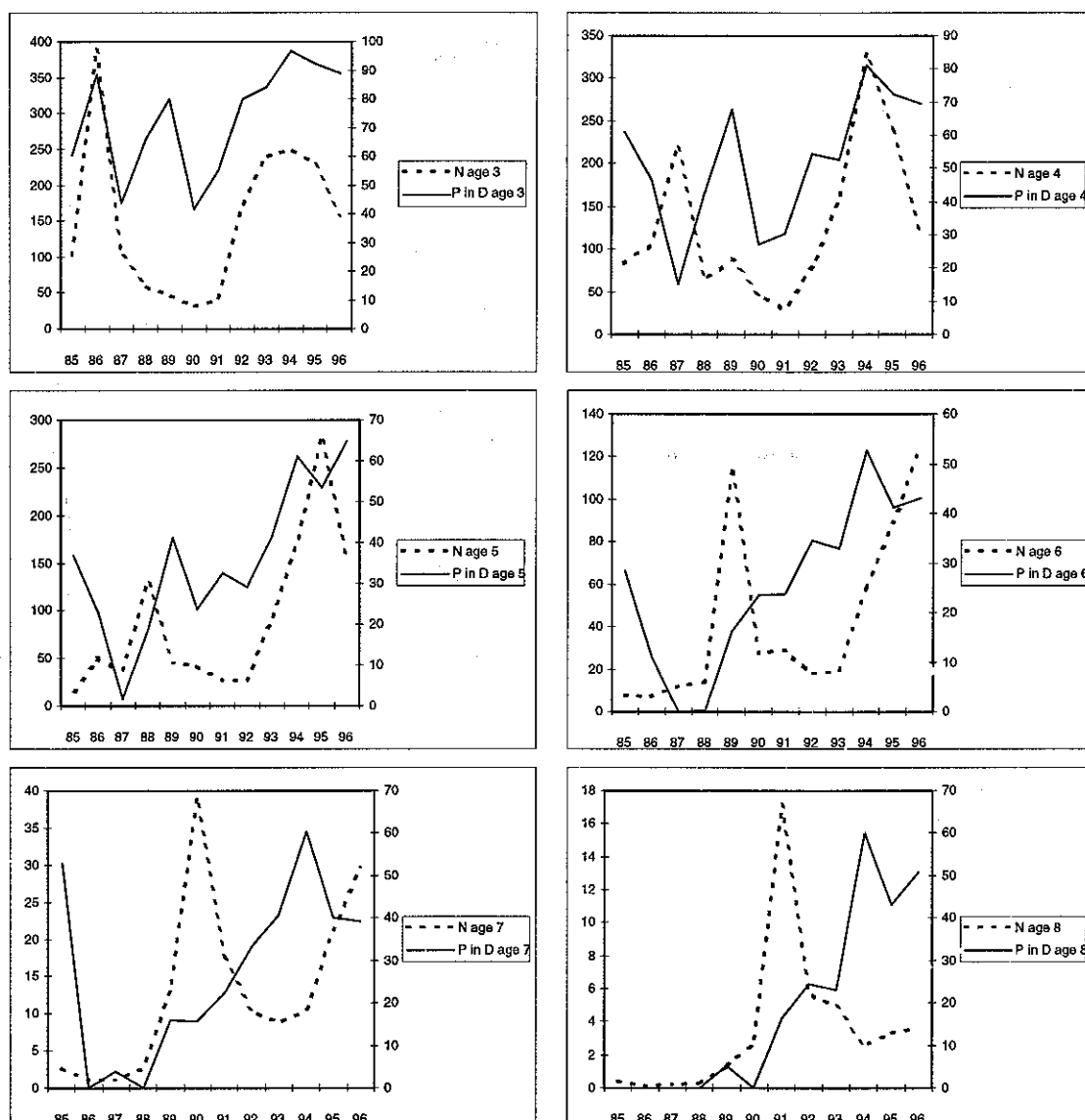
**Figure 13.1.1.1.3.** Observed biomass distribution Virtual Population Analysis (VPA) for ages 5 and up of Northeast Arctic cod from 1946–1996. The key indicates age in years.



**Figure 13.1.1.1.4.** Average age and biomass distributions for the three periods 1946–1947, 1974–1975, and 1992–1993. The keys indicate age in years.



**Figure 13.1.1.1.1.5.** Abundance indices of Northeast Arctic cod (N [= numbers] at ages 3–8) together with the estimated percentage of the year class in the eastern part of the survey area (p [= percent] in D [= area D, the eastern portion of the range]) from 1985–1996.



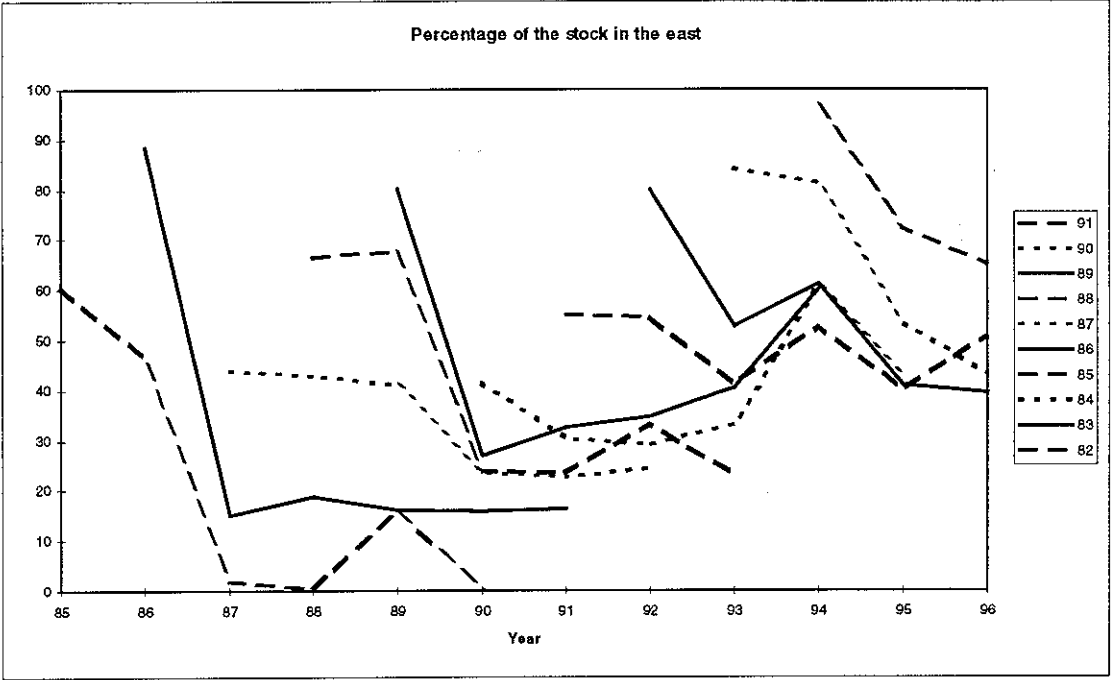
For the past twenty years fishing mortality has ranged between 0.43 and 0.96 (Figure 13.1.1.1.2.2), but since 1993 there has been a substantial reduction in the fishing mortality and in 1995 it was at  $F = 0.52$ . The fishing mortality associated with the trawlers increased in 1996 (Figure 13.1.1.1.2.3), which can be explained by the increased catch rate for this fleet especially in 1996. The current estimate of  $F$  (0.45) is at the  $F_{med}$  level. In spite of poor recruitment in recent years, the spawning stock has shown the first signs of recovery from the historically low levels in most recent years. This is a result of recent catch restrictions (25 % of fishable stock size) combined with an increase in maturity at age.

#### Changes in size distribution and/or age composition

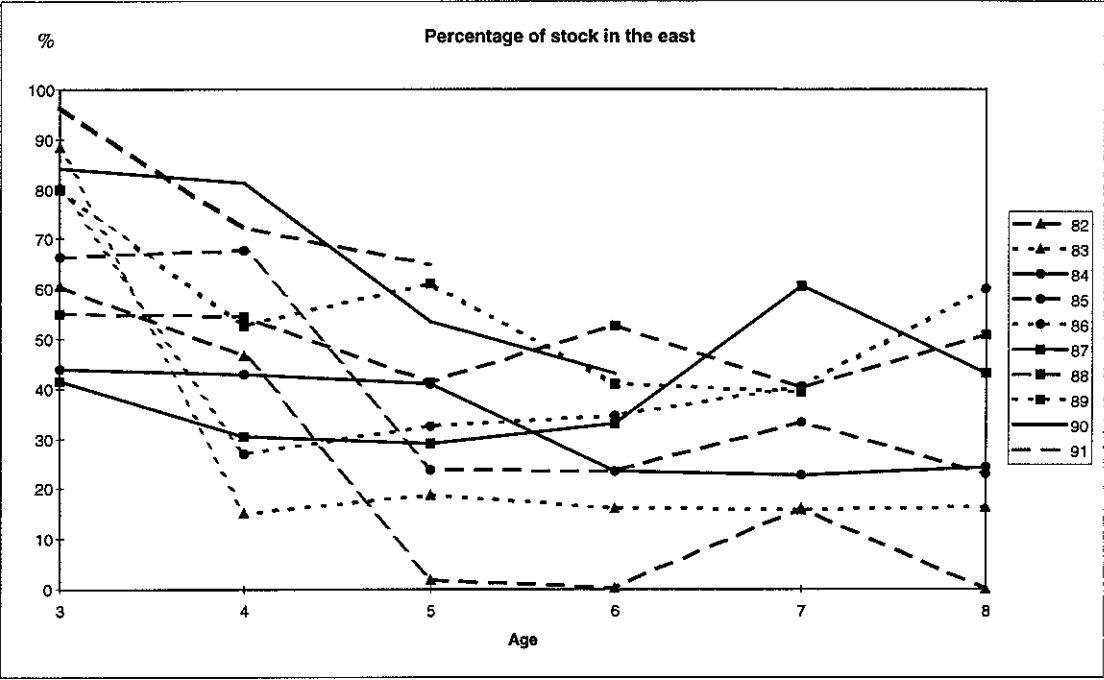
Fishing mortality by age (Figure 13.1.1.1.2.4) for the gillnetters and the Danish seiners shows that these fleets exploit mainly the oldest age groups (8–12), whereas the longliners and especially the handliners exploit the younger ages. The average size of cod increased from 44 cm to 53 cm in 1985–1989, but a gradual decline in mean size occurred during 1989–1994 (53 cm and 44 cm, respectively). During the last few years, the mean length of Icelandic cod has increased again and was at 54 cm in 1997 (Figure 13.1.1.1.2.5.a). This is reflected in the size composition of the cod stock, as the general



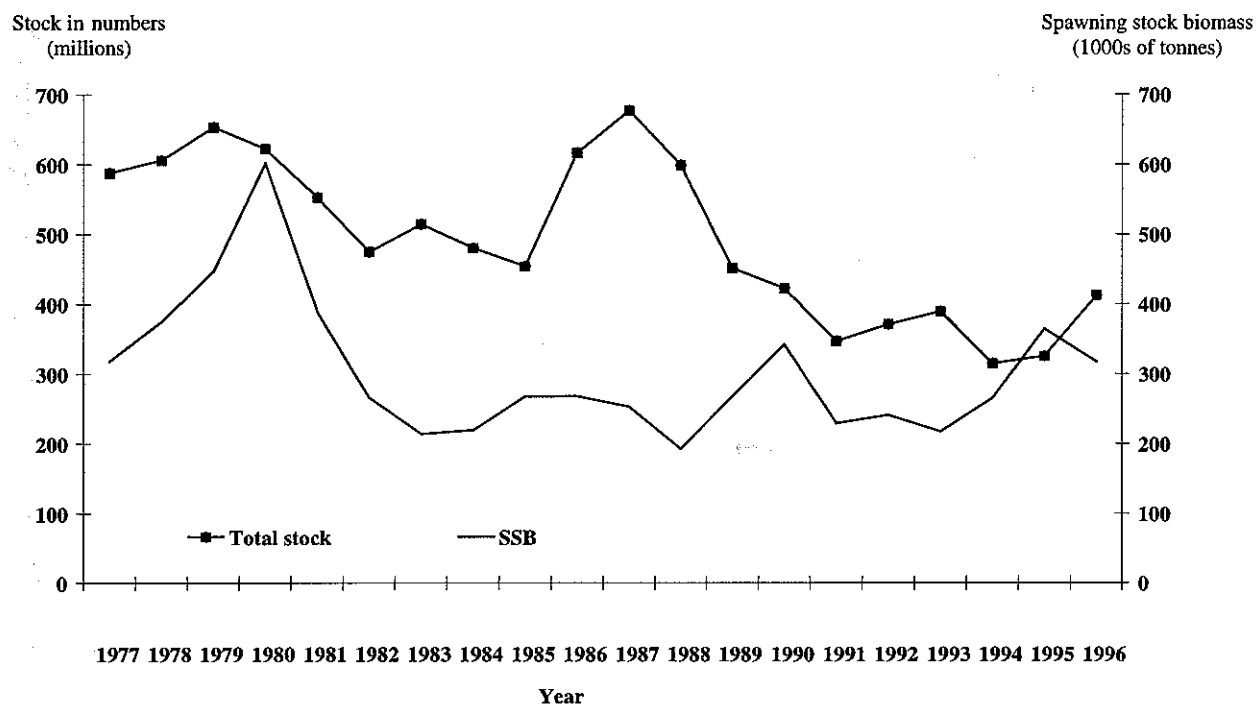
**Figure 13.1.1.1.6.** Estimated percentage of Northeast Arctic cod in the eastern part of the survey area by year and cohort. The key indicates cohort.



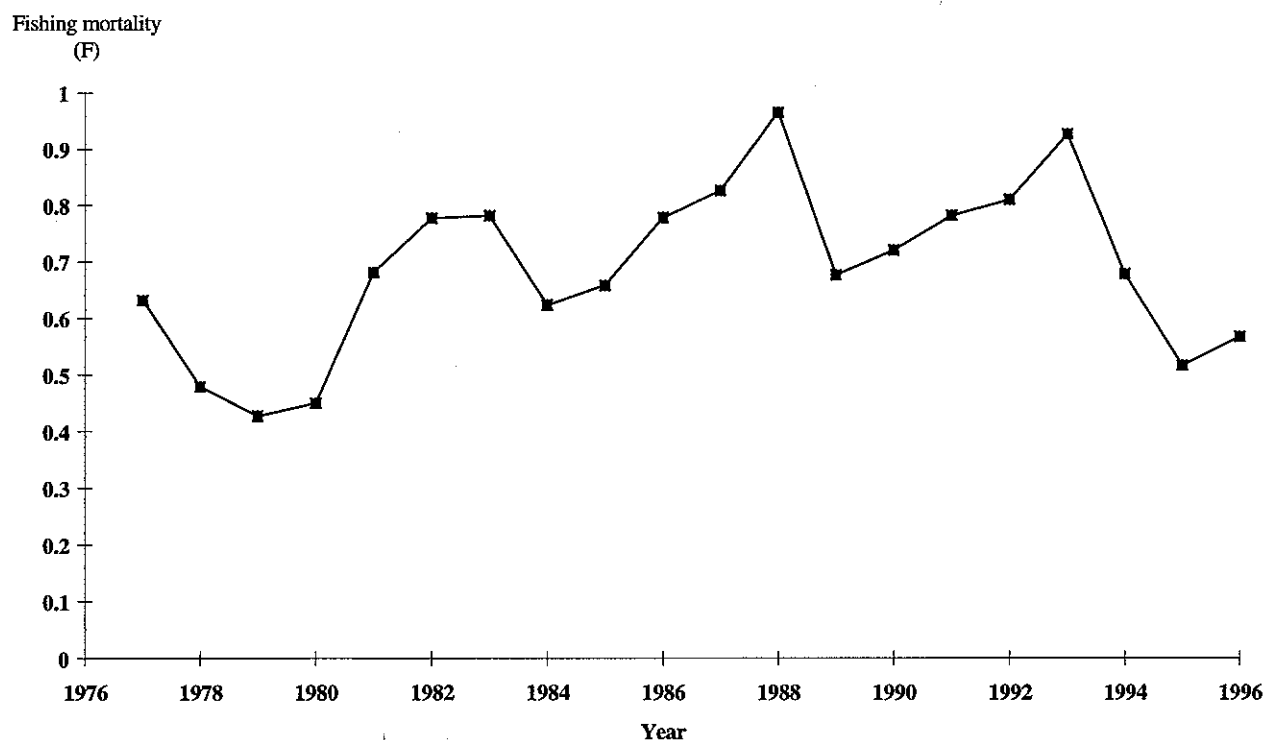
**Figure 13.1.1.1.7.** Estimated percentage of Northeast Arctic cod in the eastern part of the survey area by age and cohort. The key indicates cohort.



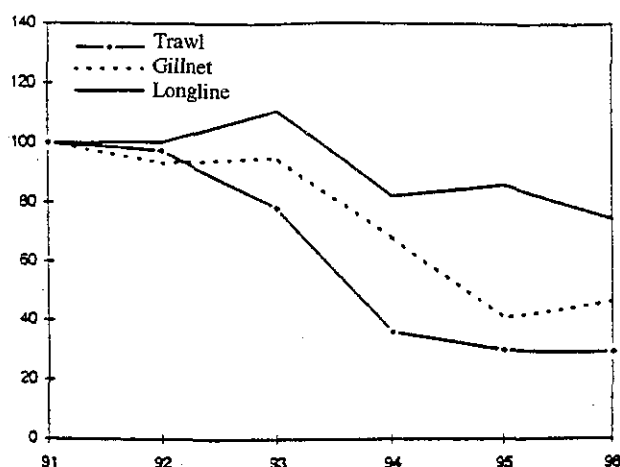
**Figure 13.1.1.2.1.** Cod at Iceland. Total stock in numbers (millions of individuals) and spawning stock biomass (thousands of tonnes) from 1977–1996 (data from ICES, 1997a).



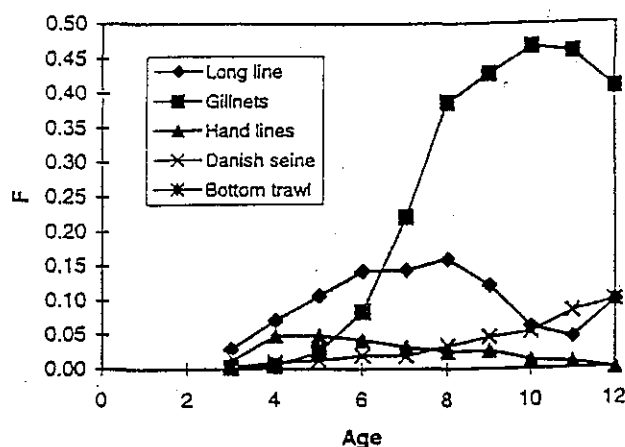
**Figure 13.1.1.2.2.** Cod at Iceland. Fishing mortality (unweighted average for age 5–10 years) from 1977–1996 (data from ICES, 1997a).



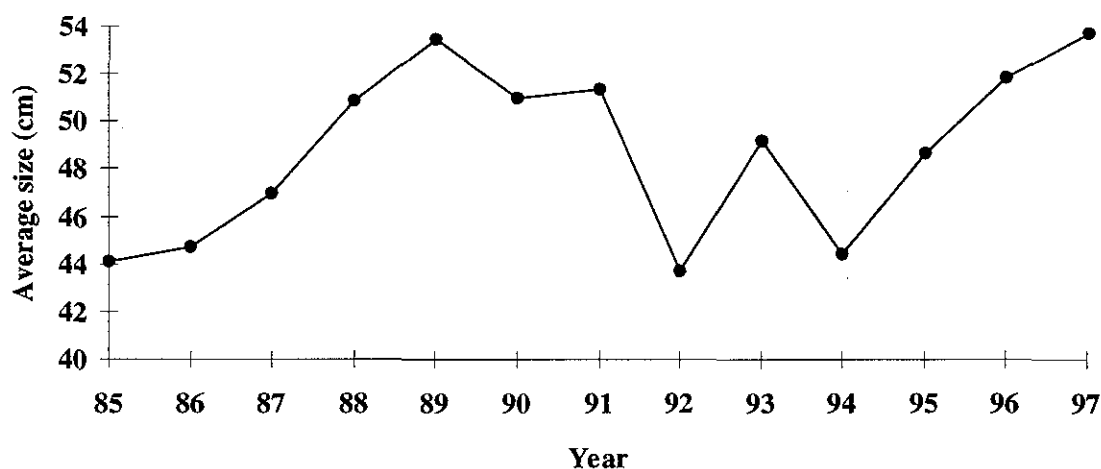
**Figure 13.1.1.1.2.3.** Trends in relative effort (1991 = 100) by fishing gear in the Icelandic cod fishery during 1991–1996 (from Anon., 1997).



**Figure 13.1.1.1.2.4.** Fishing mortality by gear and age in the Icelandic cod fishery, showing the average over the years 1992–1996 (data from ICES, 1997a).



**Figure 13.1.1.1.2.5.a.** Average size of cod in Icelandic waters, 1985–1997.



trend in the proportional changes of cod > 40 cm showed an increase from 55 % to 75 % during 1985–1997 (Figure 13.1.1.1.2.5.b). During this period, there were no strong year classes in the recruitment. The change in size distribution occurred because of severe catch restrictions.

#### Changes in spatial distribution

Only small, local-scale changes in spatial distribution are thought to have occurred over the years. These would be very difficult to disentangle from the large seasonal migrations of the stock, and would require data on finer space and time scales than those which are available.

#### 13.1.1.1.3 East Greenland cod

Tagging experiments show that an interrelationship exists among the cod stocks in East Greenland, West Greenland, and Icelandic waters. Due to migration effects, the offshore components of East and West

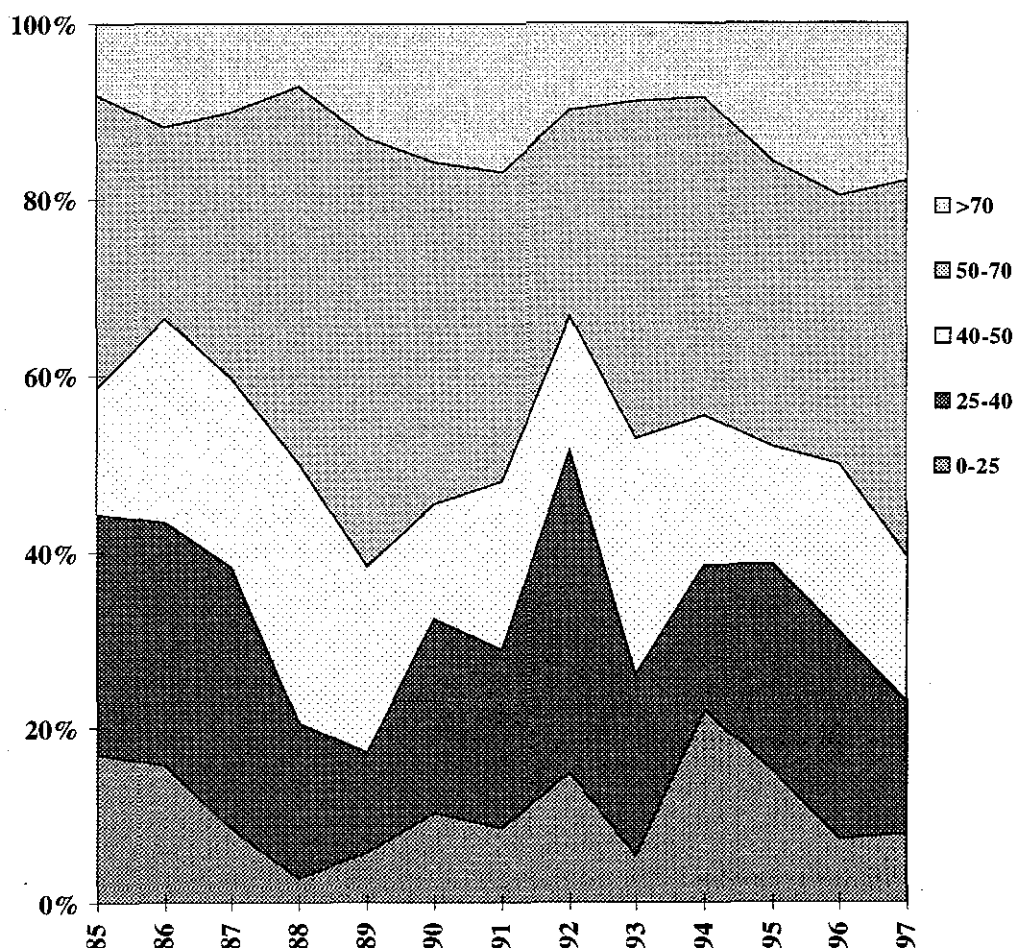
Greenland were first assessed in 1996 as one stock unit and distinguished from the inshore populations.

Cod were mainly exploited by stern trawlers using otter trawls, but considerable fisheries were also carried out using gillnets, longlines, and handlines (miscellaneous gears).

The officially reported data also include the inshore catches (ICES, 1998b). The highest catches recorded since 1955 were reported in the 1960s. From 1968, the catches decreased sharply from about 450,000 t to about 50,000 t in the mid-1970s. Due to two recruiting medium-sized year classes of 1973 and 1984, the catches increased to 100,000 t and 130,000 t, respectively, but then decreased to 1000 t in 1996.

Before 1975, offshore catches dominated the total figures by more than 90 %. Thereafter, this proportion decreased to 40–50 % and the most recent yields have been

**Figure 13.1.1.2.5.b.** Relative size composition of the Icelandic cod stock from 1985–1997. Size classes (as indicated in the key) are 0–25 cm, 25–40 cm, 40–50 cm, 50–70 cm, and > 70 cm total length.



dominated by inshore landings during the years after 1993 (Figure 13.1.1.3.1).

The directed cod fishery was given up in 1992 due to the severely depleted status of the offshore stock component. Since then, no adequate data have been available to update the analytical assessment. Therefore, the data series for the spawning stock biomass ended in 1992. Figure 13.1.1.3.2 shows the dramatic decrease in spawning stock biomass from 1.8 million tonnes in 1955 to 20,000 t in 1977. After that, it has varied within a range of 20,000 t to 100,000 t.

The dramatic collapse of the offshore component of the stock was associated with emigration, high fishing mortalities, and changes in environmental conditions. The interaction between the East Greenland and Irminger currents during the early 1970s and 1980s has apparently rendered climatic conditions unsuitable for offshore cod (ICES, 1997a). Due to climatic conditions after 1970, the offshore cod has been unable to reproduce successfully and all major cod recruitment has occurred by larval drift from Iceland.

#### Changes in size distribution and/or age composition

As an analytical assessment has not been calculated since 1992, no information on stock in number per age is available. Age disaggregated abundance indices (for age groups 1–3, 4–6 and 7+), derived from the German groundfish survey (ICES, 1997a; Hvingel *et al.*, 1996a, 1996b) and showing the age composition from 1982 to 1996, are mainly related to the recruitment pattern and not to the fishery (Figure 13.1.1.3.3).

Rätz (1997) has analysed the structures and changes of the demersal fish assemblage off Greenland. During the period 1982–1996, he found fundamental shifts in species composition in coherence with dramatic changes in stock abundance, biomass, and size structure of ecologically and economically important species.

Figures 13.1.1.3.4. and 13.1.1.3.5 show these effects very clearly. In both areas, cod has nearly disappeared compared with the relatively high abundance values before 1990. The mean individual weight of cod off West Greenland has decreased from around 1.5 kg to nearly

Figure 13.1.1.1.3.1. Cod catches (tonnes) from 1955–1995 off Greenland, divided into inshore and offshore components.

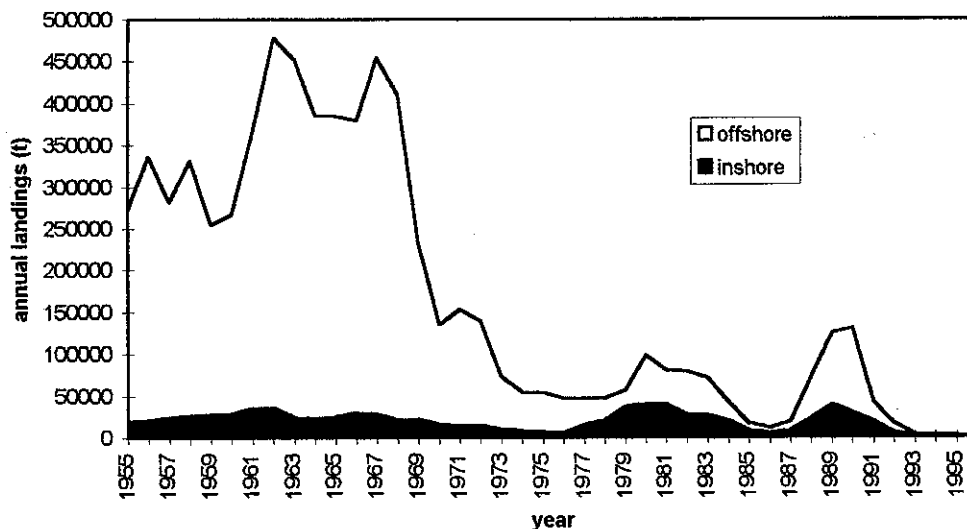


Figure 13.1.1.1.3.2. Greenland cod (offshore component). Trends in spawning stock biomass and fishing mortality (mean of age groups 5–8) from 1955–1992.

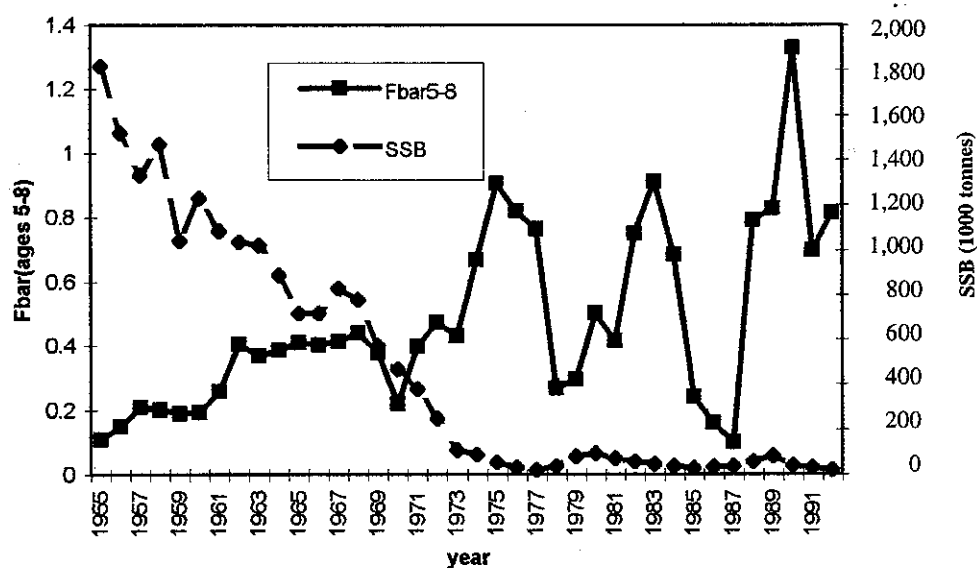


Figure 13.1.1.1.3.3. Greenland cod (offshore component). Percent distribution of age groups from survey data, 1982–1996.

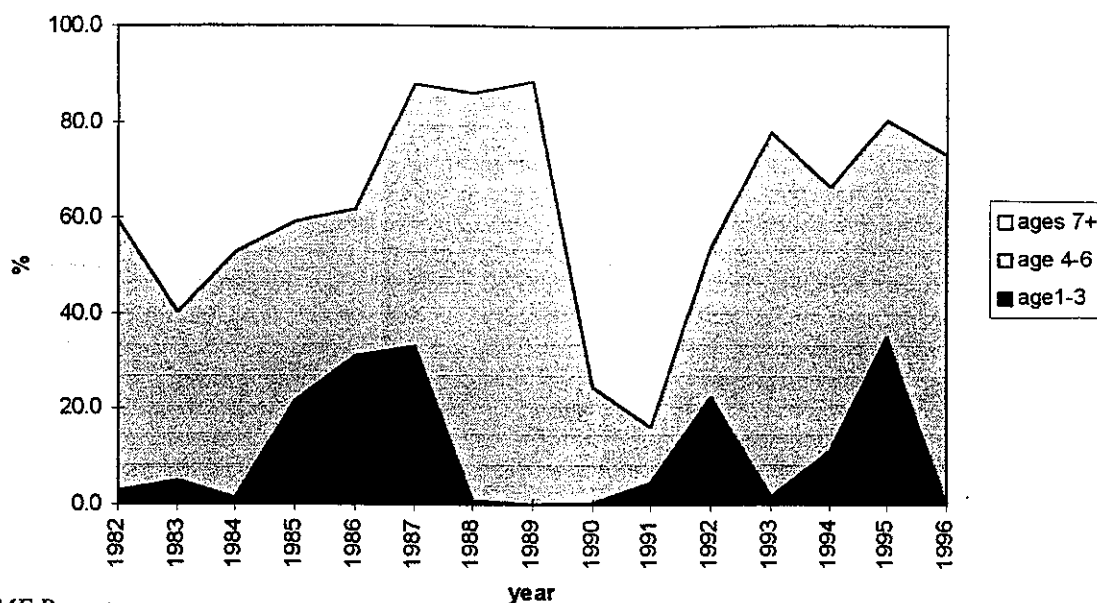


Figure 13.1.1.1.3.4. Abundance indices off West and East Greenland, and total, for Atlantic cod from 1982–1996.

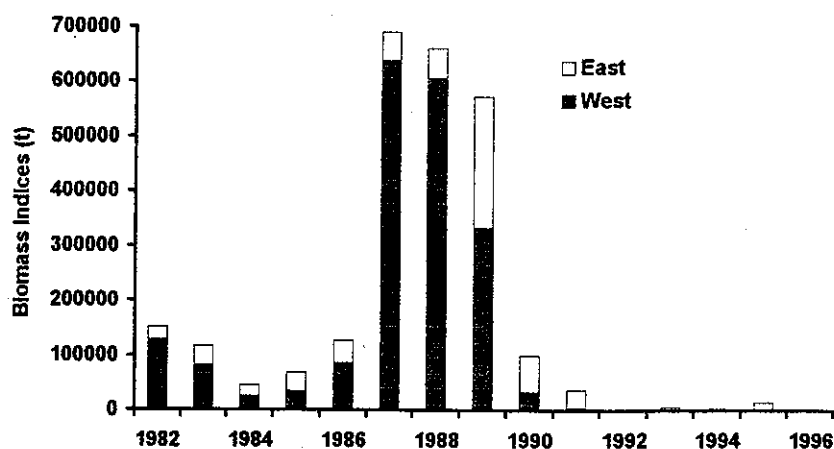
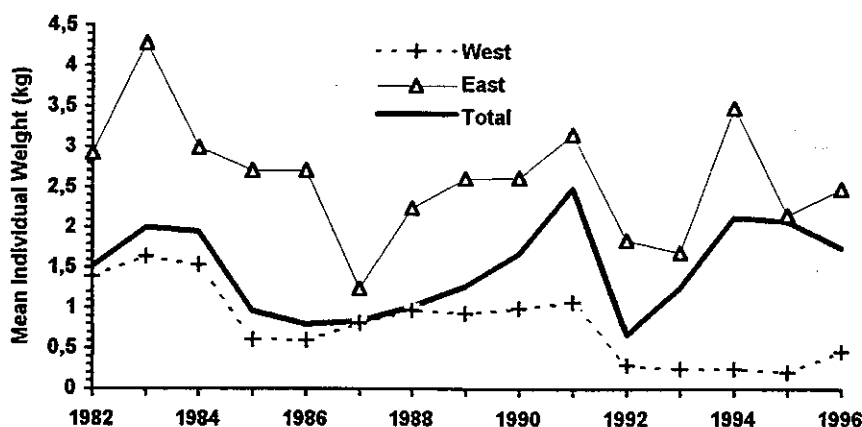


Figure 13.1.1.1.3.5. Mean individual weight of cod (kg) off West and East Greenland, and total for Atlantic cod from 1982–1996.



0.3 kg, whereas off East Greenland the individual weight varies greatly from year to year by about 2.5 kg but without a trend.

#### Changes in spatial distribution

Figure 13.1.1.1.3.4 also gives some indication of a geographical change in abundance. The losses in individuals were less pronounced off East Greenland (94 %) than off West Greenland (100 %). Changes in spatial distribution of cod within the area off East Greenland can hardly be investigated due to the present very low abundance.

#### 13.1.1.1.4 Faroe Plateau cod

##### Context - Pattern of numbers, biomass, landings, and fishing mortality

The landings of Faroe Plateau cod steadily decreased from 35,000 t in 1986 to 6000 t in 1993, the lowest catch

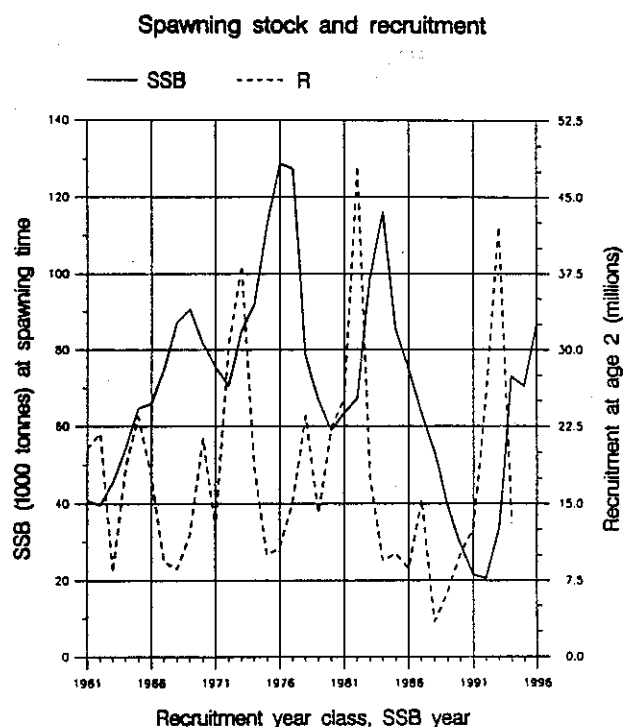
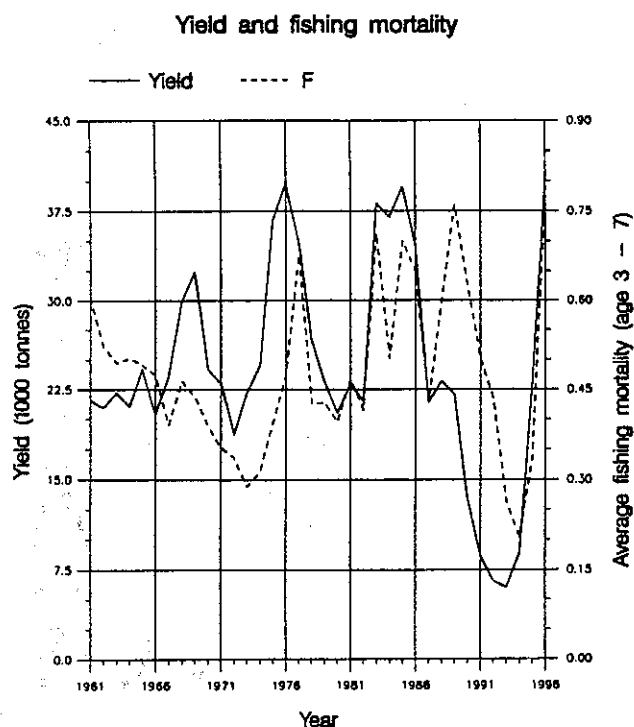
on record (see Figure 13.1.1.1.4.1 and ICES, 1997a). In 1995 the catches increased to 19,000 t and in 1996 to 40,000 t, the highest value during the 1961 to 1996 period. The average age 3–7 fishing mortality for the whole period is 0.48, with the lowest value in 1994 (0.20) and the highest value in 1996 (0.79). The present level of fishing mortality is more than twice the estimated  $F_{max}$  (0.31) and  $F_{med}$  (0.37).

Due to poor recruitment from 1984 to 1991 and high fishing mortalities, the spawning stock biomass declined steadily from 1983 to 1992, when it was the lowest on record at 20,000 t. Since then, it has increased sharply to almost 87,000 t in 1996. The SSB is expected to decrease in the medium term. No minimum biologically acceptable level (MBAL) has been estimated for the SSB, but only one strong year class has been produced at SSBs lower than 70,000 t. ICES recommends that fishing mortality be reduced to sustainable levels (below  $F_{med}$  = 0.37).

**Figure 13.1.1.1.4.1.** a) Landings (yield) (thousands of tonnes) and fishing mortality (ages 3–7); b) spawning stock biomass (thousands of tonnes) and recruitment (millions) for cod in the Faroe Plateau from 1961–1996.

a)

b)



#### Changes in size distribution and/or age composition

Historic landings are dominated by the catches of 3–5 year olds. The observed changes in size and age distribution are dominated by both changes in recruitment and in fishing effort. High effort shows a clear tendency to reduce the mean age in the stock (3 years and older), while a reduction in effort, as observed in 1991–1994, clearly produces an increase in mean age.

#### Changes in spatial distribution

There were no data available to analyse whether there have been any changes in spatial distribution.

#### 13.1.1.1.5 Faroe Bank cod

This is a very small stock and no detailed analytical assessment is made on it. The catches declined from 5000 t in 1973 to 330 t in 1992. Since then there has been a very strong regulation of the fishery. The predicted catch for 1997 is 2000 t and ICES recommends that the fishing effort in 1998 should not exceed the present level.

#### 13.1.1.2 Herring

##### 13.1.1.2.1 Norwegian spring-spawning herring

#### Context - pattern of numbers, biomass, landings, and fishing mortality

The catches of Norwegian spring-spawning herring decreased from 1.3 million t in 1951 to record low levels of around 10,000 t in the first part of the 1970s and then increased to 232,000 t in 1993 (see Figure 13.1.1.2.1.1 and ICES, 1997b). The international fishery on herring recommenced in 1994 with catches of 479,000 t and the catches increased to 900,000 t in 1995, 1.2 million t in 1996, and probably 1.5 million t in 1997. The average age 5–13 weighted fishing mortality is presently at about 0.16. When the stock collapsed around 1970, the average  $F$  was about 1.5. In the late 1960s and the early 1970s, the catch in numbers of Norwegian spring-spawning herring consisted largely of small herring taken in northern Norwegian fjords. This fishery was then closed and has not been reopened. The spawning stock biomass is estimated to have been about 14 million t in 1950, decreasing to below 10,000 t in the beginning of the

1970s, and then slowly recovering, passing 1 million t in 1987, and 5 million t in 1994. The SSB is presently estimated to be about 9 million t and is expected to increase further to 9.6 million t in 1998. From 1998 to 1999 the SSB will decrease for  $F_s$  above 0.06 in 1998 due to weak year classes after 1993. The current management strategy is based on  $F = 0.15$ , with a catch ceiling of 1.5 million t and a minimum SSB of 2.5 million t (MBAL). The ICES advice is to keep the SSB above MBAL and to adapt the catch control rule to this.

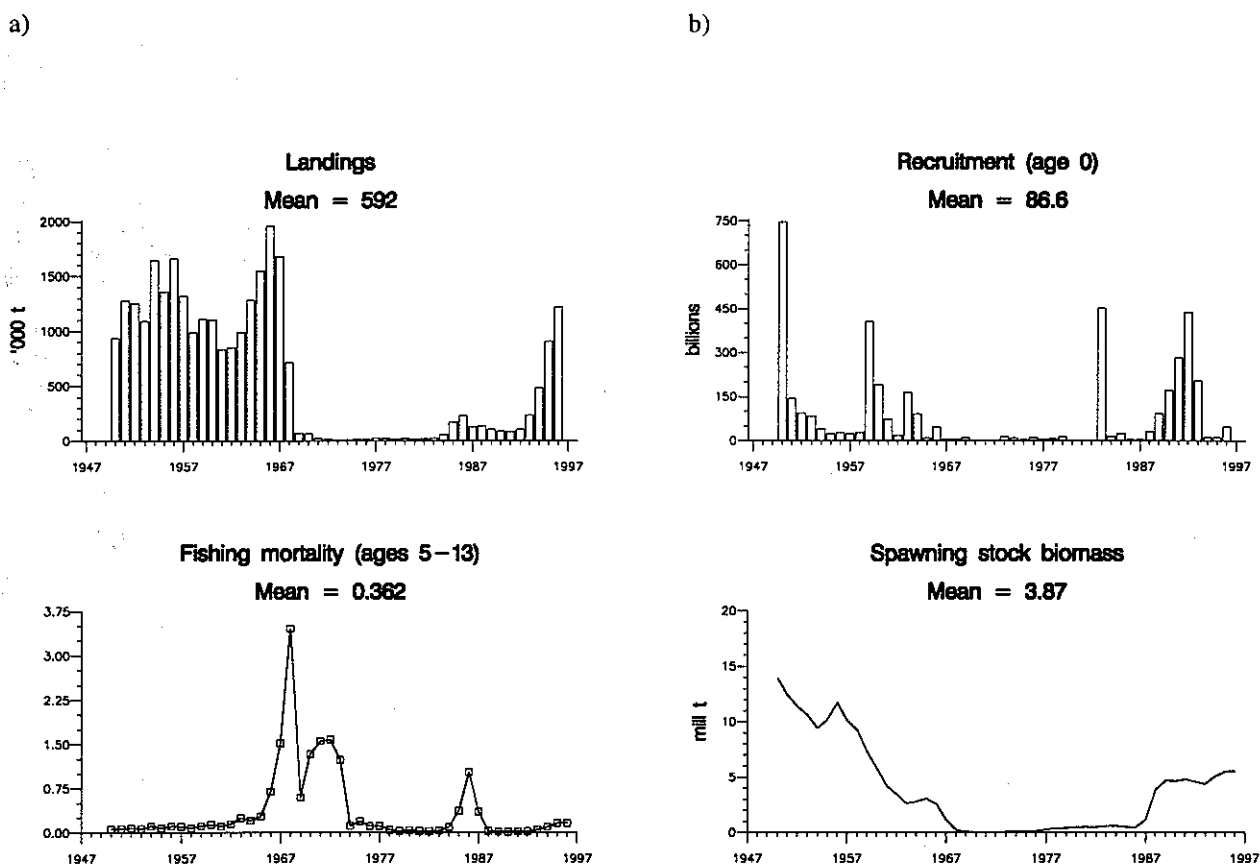
### Changes in size distribution and/or age composition

Both the landings and the stock itself are dominated by very few strong year classes. The fishing mortality of 0- and 1-group herring was high in the 1950s and until the early 1970s, but has been close to negligible after the recovery of the stock in the late 1980s. As long as the stock is being harvested at the current levels and patterns of exploitation, the size and age distribution will be dominated by the varying recruitment, meaning that the mean age in the landings will tend to increase with close to one year for each new year until a strong new year class enters the fishery.

### Changes in spatial distribution

Before the herring stock collapsed around 1970, the wintering took place in oceanic waters off East Iceland (Devold, 1963), and the Norwegian Sea was the main feeding area. Spawning has traditionally taken place at different grounds along the Norwegian coast. Since the herring stock recovered in 1988, the wintering area has been located in fjords of northern Norway and mainly in the Vestfjorden area (Slotte and Johannessen, 1997). The nursery areas are in the southern Barents Sea and northern Norwegian fjords. In the early 1990s, the herring reoccupied the Norwegian Sea as its main feeding area (Vilhjálmsen *et al.*, 1997). The collapse of the stock has been attributed to excessively high fishing mortality. Therefore, the change to a very restricted distribution, which occurred during the subsequent period when stock size was very low, can be considered an indirect consequence of fishing as well. Hence, according to that argument, fishing was also responsible for the total change in distribution from oceanic waters (the Norwegian Sea) to Norwegian coastal waters and fjords.

**Figure 13.1.1.2.1.1.** a) Landings (thousands of tonnes) and fishing mortality (ages 5–13); b) recruitment (billions) and spawning stock biomass (millions of tonnes) of Norwegian spring-spawning herring from 1950–1995.





### 13.1.1.2.2 Icelandic summer-spawning herring

#### Context - pattern of numbers, biomass, landings, and fishing mortality

Icelandic summer-spawning herring are mainly fished by Danish seines during the autumn along the southeastern and eastern coasts of Iceland.

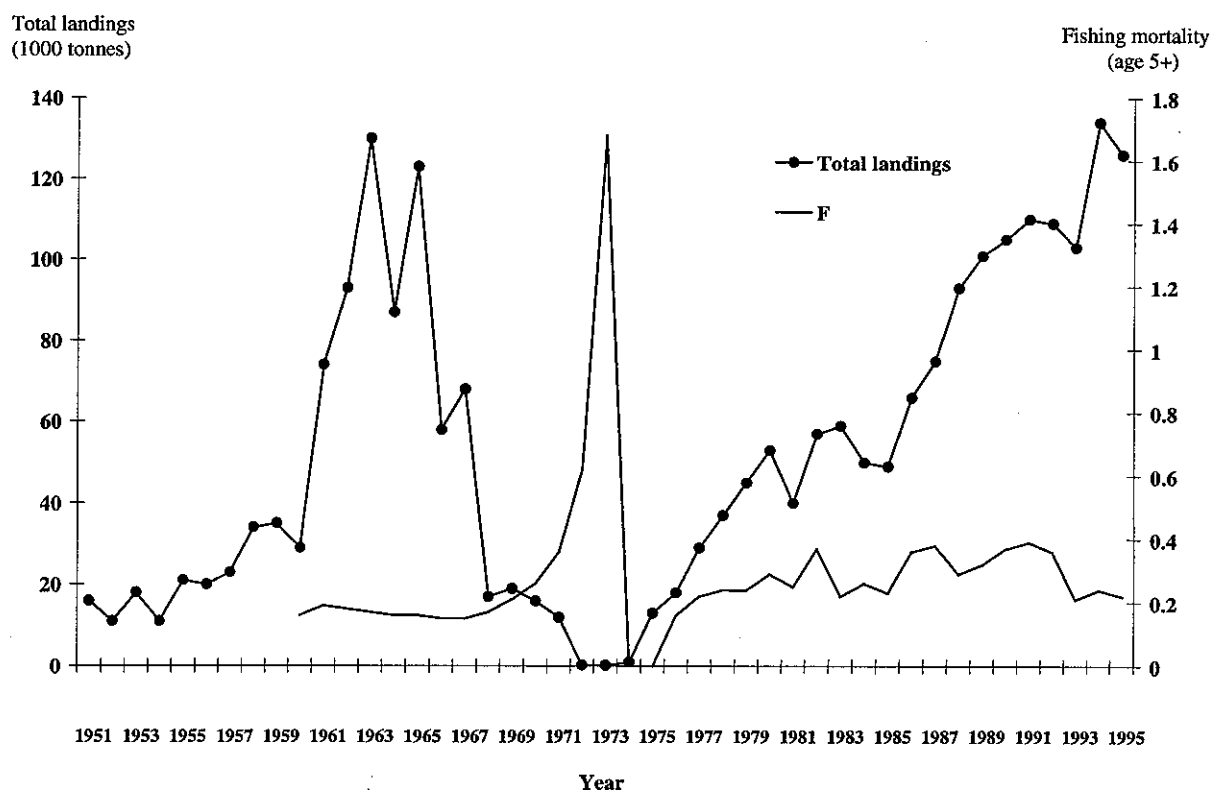
Total landings of Icelandic summer-spawning herring increased sharply from 16,000 t to 130,000 t in the period 1951–1963 (Figure 13.1.1.2.2.1). For a few years the annual catch remained high, but a total collapse in the herring fishery followed in the latter part of the 1960s, with only a few tonnes landed in 1972 and 1973. During the past twenty years, the Icelandic summer-spawning herring stock has been managed at levels corresponding fairly closely to fishing at  $F_{0.1}$  (a reference fishing mortality derived from theoretical analysis of the relationship of yield to effort). During that period, the annual landings have gradually increased from 250 t to the previous high landings of around 126,000 t. The stock size of Icelandic summer-spawning herring has increased from 1500 million individuals in 1977 to more

than 3000 million in 1996 (Figure 13.1.1.2.2.2). During the same interval, the spawning stock biomass increased from 130,000 t to 600,000 t.

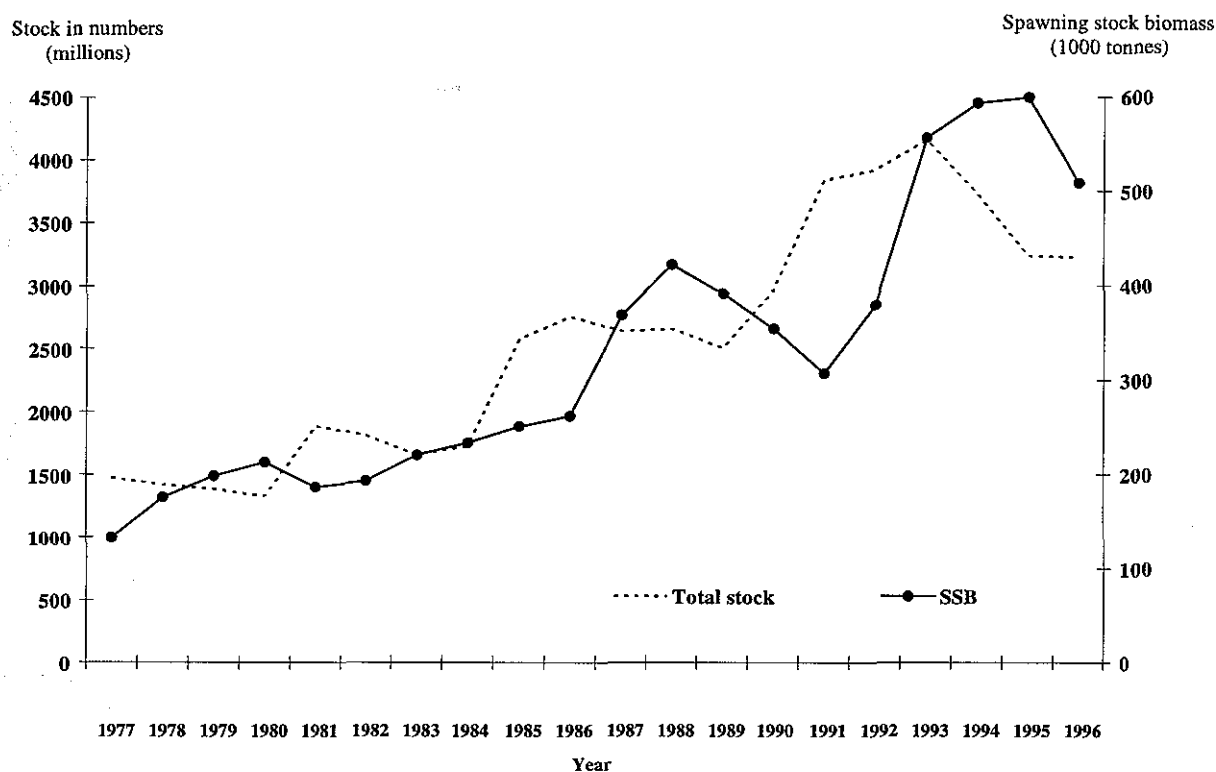
#### Changes in size distribution and/or age composition

In the years immediately after the fishery was reopened in 1975, the 1971 year class was the most abundant. During the period 1979–1982, the 1974 and 1975 year classes predominated in the catches. During the period 1983–1986, the fishery was dominated by the strong 1979 year class. On the other hand, the fishery in 1987 and 1988 was based on a number of year classes ranging from 4–10 years of age. In the period 1989–1991, the 1983 year class predominated in the catch. The 1988 year class was also well represented in the 1991 catches and predominated during the 1992 season. In 1993 the age distribution was dominated by the strong 1989 year class, although the 1988 year class was also well represented. In 1994/1995, the catches were distributed over four year classes, i.e., those of 1988–1991. The catch in numbers of three-year-old herring has never been higher and yielded some 25 % of the total numbers in the 1994/1995 season.

**Figure 13.1.1.2.2.1.** Total landings (thousands of tonnes) of Icelandic summer-spawning herring from 1951–1995 and fishing mortality (5+) during the period 1960–1995. Data from Jakobsson (1980) and ICES (1997b).



**Figure 13.1.1.2.2.2.** Total stock size of Icelandic summer-spawning herring (millions of individuals) and spawning stock biomass (thousands of tonnes) from 1977–1996.



During the period 1977–1996, few changes have taken place in stock numbers at age (Figure 13.1.1.2.2.3; Anon., 1997). In the 1995 landings, 50 % of the catch was two- to five-year-old herring. The proportional abundance of these age classes has decreased from 80 % to 60 % during that period.

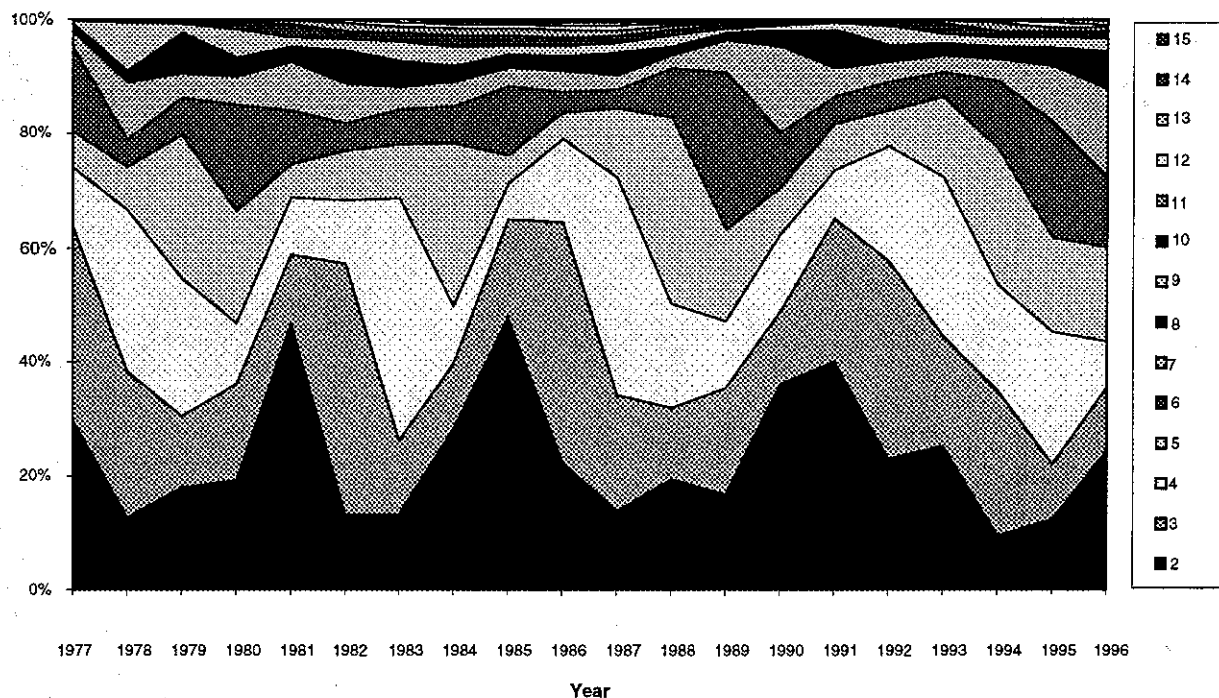
#### Changes in spatial distribution

The distribution of the Icelandic summer-spawning herring during the fishing season in autumn and winter is mainly constricted to the southeastern and eastern coasts of Iceland. There is no evidence to suggest that the commercial exploitation of herring in this region has altered the spatial distribution of the species.

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**Figure 13.1.1.2.2.3.** Relative stock number at age of Icelandic summer-spawning herring from 1977–1996. The key indicates age in years.



Jakobsson, T., Korsbrekke, K., Mehl, S., and Nakken, O. 1997. Norwegian combined acoustic and bottom trawl surveys for demersal fish in the Barents Sea during winter. ICES CM 1997/Y:17. 26 pp.

Rätz, H.-J. 1997. Structures and changes of the demersal fish assemblage off Greenland and trends in near-bottom temperature, 1982–1996. NAFO SCR Doc. 97/5, Serial No. N2830.

Slotte, A., and Johannessen, A. 1997. Spawning of Norwegian spring-spawning herring (*Clupea harengus* L.) related to geographical location and population structure. ICES CM 1997/CC:17. 36 pp.

Vilhjálmsen, H., Misund, O.A., Arrhenius, F., Holst, J.C., Gislason, A., Gudmundsdottir, A., Jacobsen, J.A., Krysov, A., Malmberg, S., and Reid, D. 1997. Report on surveys of the distribution, abundance and migrations of the Norwegian spring-spawning herring, other pelagic fish and the environment of the Norwegian Sea and adjacent waters in late winter, spring and summer of 1997. ICES CM 1997/Y:4. 32 pp.

### 13.1.2 OSPAR Region II: Greater North Sea

Demersal fisheries in the North Sea consist of human consumption fisheries and industrial fisheries. Human consumption fisheries target a mixture of roundfish

species (cod, haddock, whiting (*Merlangius merlangus*), or a mixture of flatfish species (plaice (*Pleuronectes platessa*) and sole (*Solea solea*)) with a by-catch of roundfish. The industrial fisheries use small mesh trawls and target mainly sandeels (*Ammodytes* sp.), Norway pout (*Trisopterus esmarki*), and sprat (*Sprattus sprattus*). The industrial catches also contain a by-catch of other species, including herring, haddock, and whiting.

In the North Sea, all stocks of roundfish and flatfish species have been exposed to high levels of fishing mortality for a long period. For most of these stocks, their lowest observed spawning stock size has been seen in recent years. The present assessments for roundfish indicate a decline in fishing mortality in recent years for cod, haddock, and whiting. The reason for this decline is unclear. The decline is somewhat supported by a reduction in effort by some of the major fleets in the last few years and by a diversion of effort to *Nephrops* and anglerfish (*Lophius piscatorius*). However, this decline may also be artificial, since it appeared in the assessment for the first time. Fishing mortality on plaice and sole has been varying at a high level over a long period with no trend. Sandeel landings averaged around 600,000 t annually between 1976 and 1986, increasing to 800,000 t between 1987 and 1996, and have increased to record levels in the first half of 1997. The sandeel spawning stock has fluctuated around one million tonnes with no obvious trend. The 1996 sandeel year class appears to have been very strong. Norway pout landings decreased over the period 1974 to 1988 and thereafter fluctuated at

around 200,000 t. The spawning stock declined to its lowest level in 1986 and has since increased, partly due to a good year class in 1994.

The Skagerrak/Kattegat area is, to a large extent, a transition area between the North Sea and the Baltic Sea. Several of the stocks in the Skagerrak show close affinities to the North Sea stocks, including cod, haddock, whiting, saithe, hake, plaice, and Norway pout, in terms of population dynamics. Tagging experiments suggest extensive migration between the two areas. Both human consumption and industrial fisheries operate in the Skagerrak/Kattegat. Human consumption fleets include gillnetters and Danish seiners exploiting flatfish and cod as well as demersal trawlers targeting roundfish, flatfish, *Nephrops*, and *Pandalus*. Demersal trawling is also used in the industrial fisheries for Norway pout and sandeel. Pelagic trawlers exploit herring, mackerel (*Scomber scombrus*), horse mackerel (*Trachurus trachurus*), and sprat. The main herring stocks exploited in the area are North Sea autumn-spawners and the stock of spring-spawners spawning in the western Baltic Sea and the southern part of Division IIIa. Most of the stocks are assessed in conjunction with the stocks in the neighbouring areas: cod and autumn-spawning herring with the North Sea stocks, spring-spawning herring with the western Baltic stocks.

A large proportion of the Eastern Channel is in the coastal zone (within the 12-mile limit), which is exploited by small-scale fisheries. The major fleets operating in this area are: a French inshore fleet consisting mainly of small vessels using a variety of gears; an English inshore fleet using fixed gear; English and Belgian offshore beam trawlers; and French offshore otter trawlers. The beam trawl fleets target sole, with a significant plaice by-catch. Sole are also taken by the two inshore fisheries using trammels and, in the case of the French inshore fleet, otter trawls. Plaice are targeted by the French offshore trawlers, with sole taken as a by-catch, while the major part of the cod landings originates from French offshore trawlers and inshore gillnetters. Whiting are caught in mixed fisheries inshore and offshore by French trawlers. Effort directed at flatfish increased consistently and considerably in all fleets from 1975, reaching a peak in 1989–1990, after which it has stabilized. A pelagic fishery operates in winter during the herring spawning season.

Up to and including 1995, cod, whiting, plaice, and sole in Division VIIId were assessed as separate stocks. Review of the stock identity of these species indicated that sole should continue to be considered as a separate stock, but that there were strong links between the Eastern Channel and the southern North Sea for cod, plaice, and whiting. The pelagic fish species—herring (Downs herring), horse mackerel, mackerel, and sprat—are subject to TACs set over larger areas.

### 13.1.2.1 Cod

#### 13.1.2.1.1 North Sea

##### Context - pattern of numbers, biomass, landings, and fishing mortality

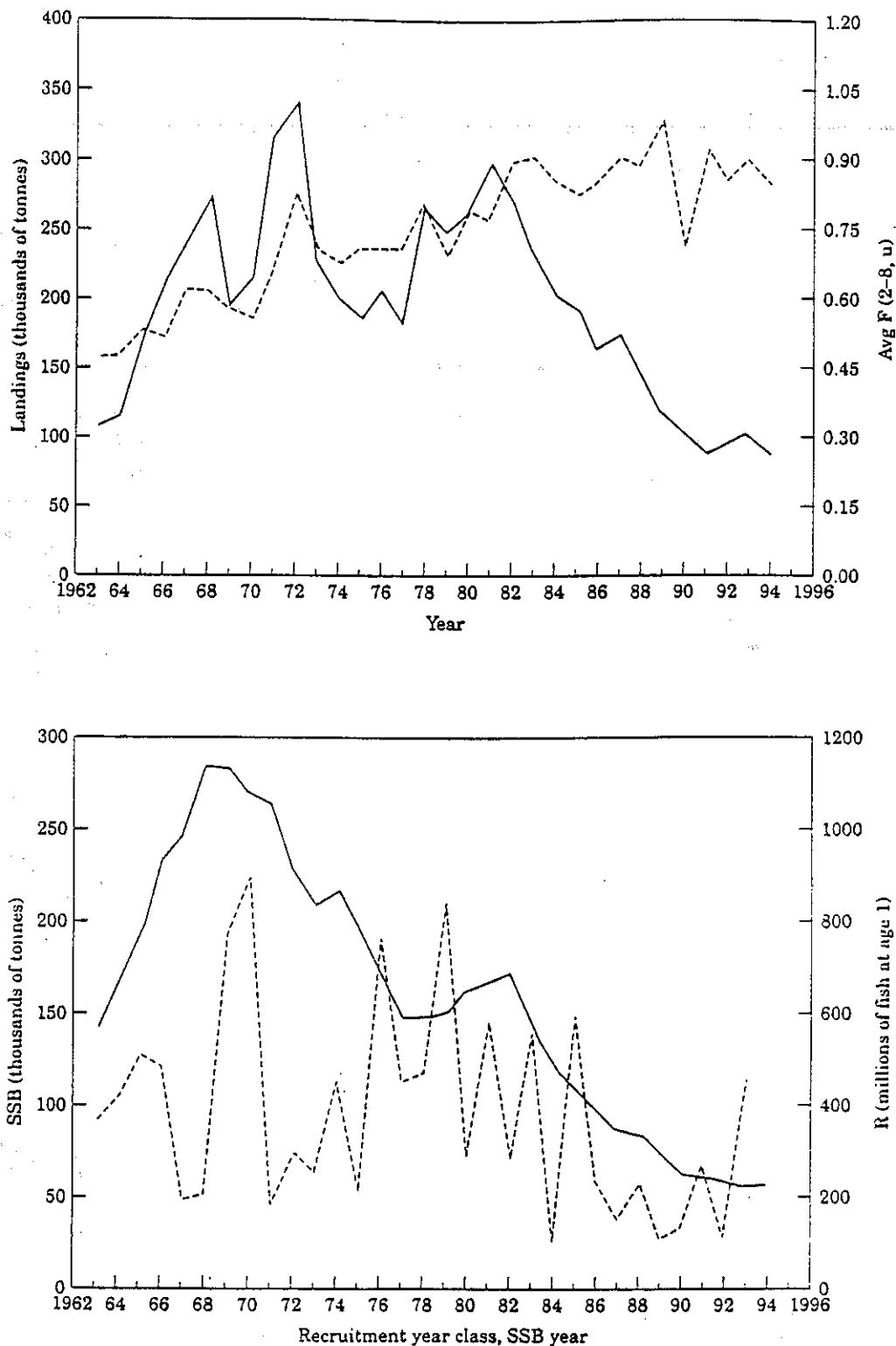
Landings of cod have increased from 108,000 t in 1963 to 341,000 t in 1972, fluctuated around the 200,000 t to 250,000 t level from 1972 to 1981, then declined steadily to 88,000 t in 1994 (Figure 13.1.2.1.1.1; Serchuk *et al.*, 1996). Over the last thirty years, fishing mortality has increased almost continuously, doubling over the period 1963–1989, until stabilizing at around  $F = 0.9$ . Since 1981, fishing mortality has been in excess of the biological reference point,  $F_{med}$ , indicating recruitment overfishing, and the landings have become dominated by two-year-old immature fish. Furthermore, discards of one-year-old cod have been considerable in some years. Since 1983, the spawning stock biomass has been below MBAL, the level of spawning stock size below which the probability of poor recruitment increases as spawning stock size decreases (ICES, 1992), and the year class strength was below average between 1987 and 1993. A good year class in 1993, however, resulted in an increase of spawning stock biomass to around 100,000 t in 1996 and landings have again increased to 126,000 t. Another good year class in 1996, together with an apparent drop in fishing mortality in recent years, are both favourable indicators for the status of the stock in the short term (ICES, 1998). WGECC notes, however, that in the recent past it has been widely reported that recruitment of strong year classes of gadoids to the fishery has been accompanied by high levels of discarding. This risk of losing the potential benefit of the strong 1996 year class, combined with the very precarious state of the stock, as shown by Cook *et al.* (1996), leaves WGECC pessimistic about the likelihood of noteworthy improvement of the stock in the medium term.

##### Changes in size distribution and/or age composition

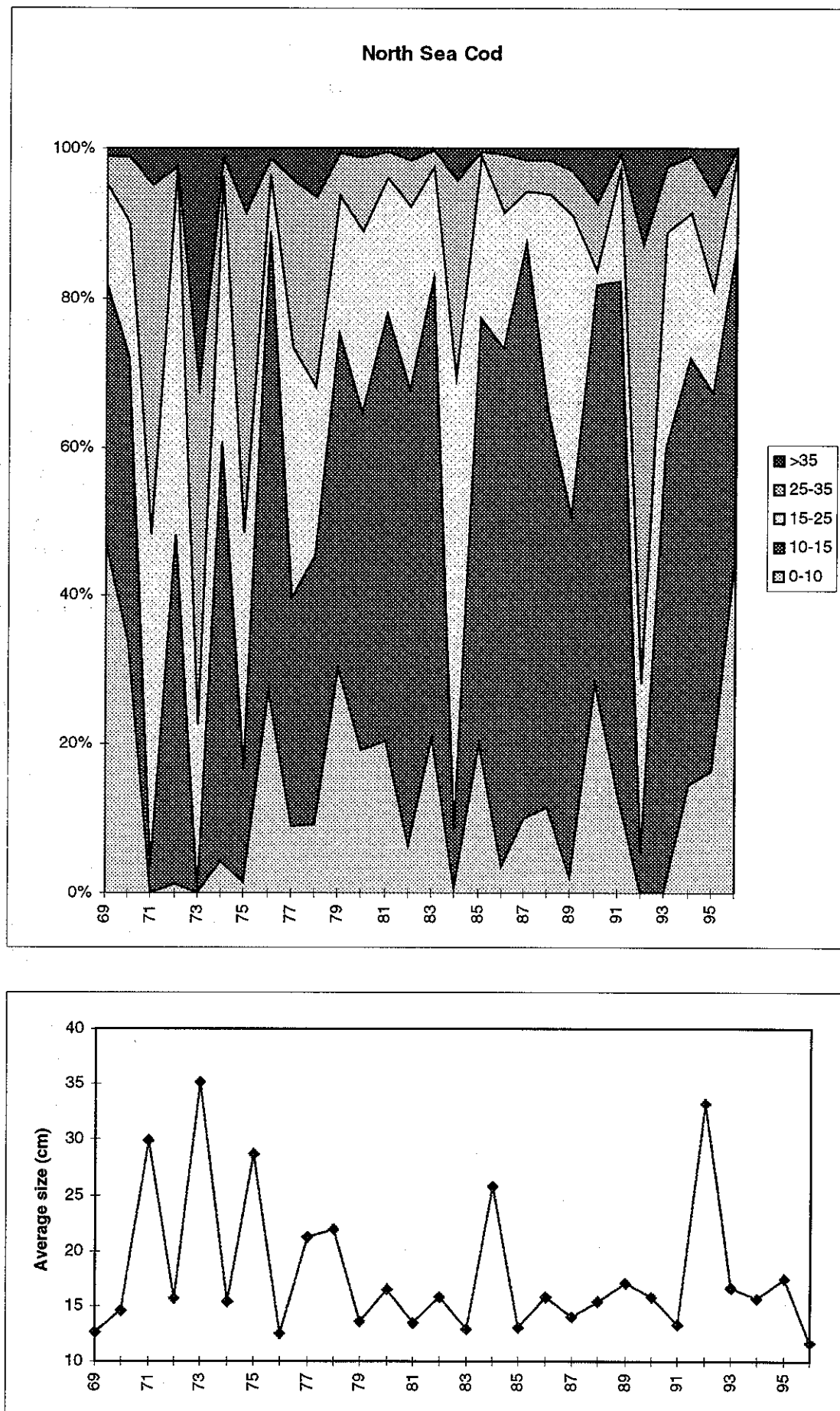
Figure 13.1.2.1.1.2 shows variations in the proportion of cod sampled in the International Bottom Trawl Survey (IBTS) that belonged to different length classes. Clear trends are not obvious, though since 1976 smaller fish have tended to dominate the catches. The important point to note is that from 1976 larger cod are rarely encountered. The two points where cod ( $> 15$  cm) again dominated the survey catches after 1976 probably reflect years of poor recruitment more than anything else.

Plotting the trends in the numbers of cod at age as determined by the Single Species Virtual Population Analysis (SSVPA) reveals a clear shift towards younger fish (Figure 13.1.2.1.1.3). This trend is associated with the steady increase in fishing mortality over this period.

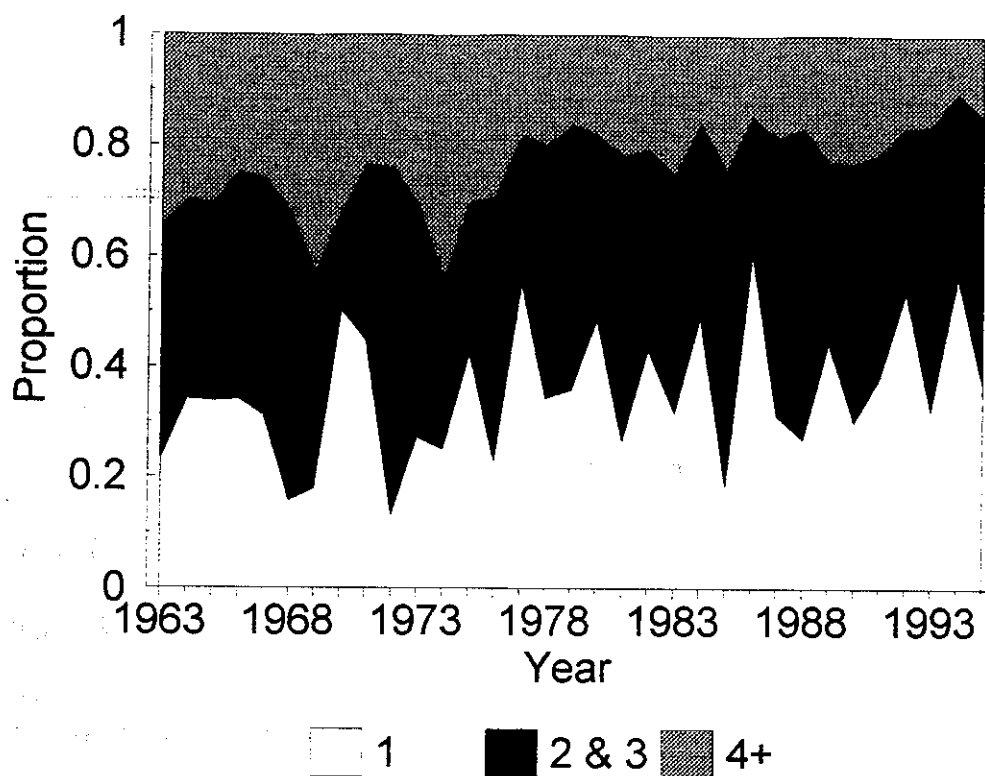
**Figure 13.1.2.1.1.1.** Trends in landings (—) and fishing mortality (---) (top) and spawning-stock biomass (—) and recruitment (---) (bottom) for North Sea cod from 1963–1994. Data from ICES (1996).



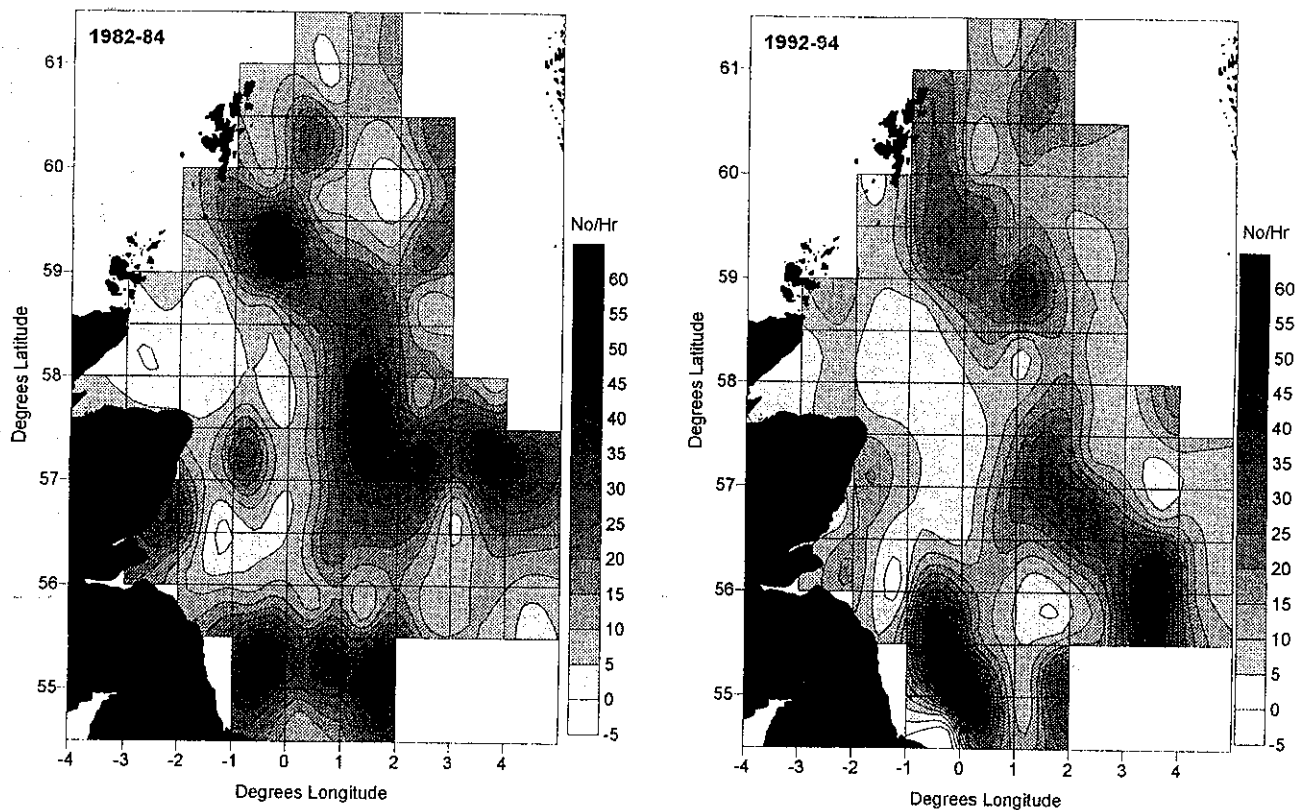
**Figure 13.1.2.1.1.2.** Changes over time in the proportion of cod at different length classes in the IBTS dataset (top) and trends in average size (bottom) from 1969–1996. The key shows length in centimeters.



**Figure 13.1.2.1.1.3.** Variation in the proportion of cod at age over the duration of the VPA (1963–1995). The key indicates age in years.



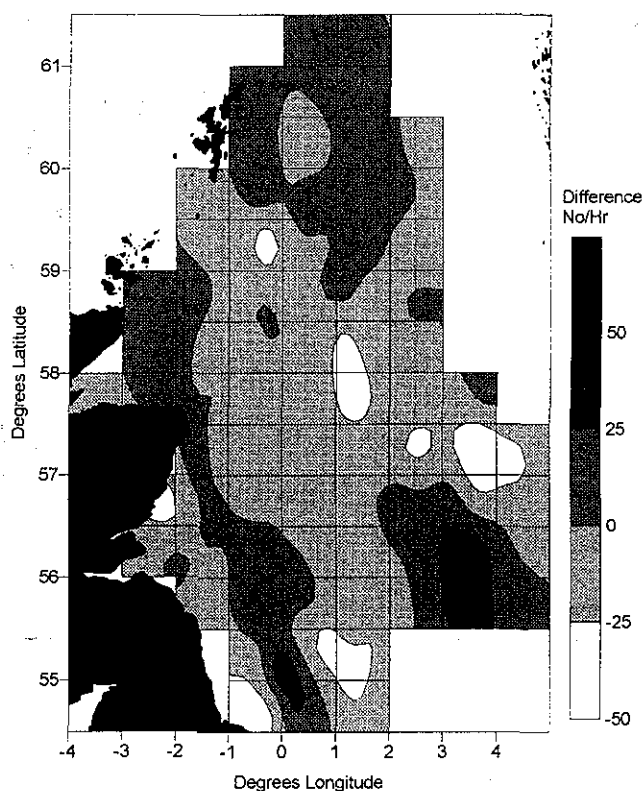
**Figure 13.1.2.1.1.4.** The distributions of cod sampled by the Scottish AGFS in 1982–1984 and 1992–1994.



### Changes in spatial distribution

Scottish Annual Groundfish Survey (AGFS) data were analysed to investigate changes in the distribution of cod in the northwestern North Sea. Data for two three-year periods, 1982–1984 and 1992–1994, were extracted from the database. The North Sea cod stock declined from around 130,000 t to 60,000 t over the intervening period. Mean catch rates in each statistical rectangle were determined, after any rectangle sampled less often than twice in one of the three-year periods was excluded from the analysis. The data were gridded using a multiquadric radial basis function based on the mean trawl positions in each rectangle in each of the three-year periods. The distributions of cod in each period are shown in Figure 13.1.2.1.1.4. Although there is a superficial similarity between the two distributions, a plot of the difference between them indicates marked changes (Figure 13.1.2.1.1.5). Catch rates in areas of high density in 1982–1984 have decreased markedly, while catch rates in nearby low-density areas have increased. The population distribution has shifted, but it is difficult to relate this to fishing activity in the absence of effort data. Certainly, no evidence of a retraction of the distribution associated with the decline in the North Sea cod biomass is indicated.

**Figure 13.1.2.1.1.5.** Chart showing the difference in the abundance of cod between the two time periods, 1982–1984 and 1992–1994.



Cod are widely distributed over the North Sea. Data from the International Bottom Trawl Survey in February from 1971–1991 show that age group 1 is most abundant in the southern part of the North Sea, although in certain years most of the catch of this age group was taken in the central part. Two-year-old cod are more evenly distributed, and age three and older fish are mainly found in the northern North Sea (Heessen, 1993). Compared to earlier years, a smaller proportion of the juvenile cod is now found in the southern North Sea and German Bight. It is not known whether this is due to the low number of juveniles produced by the stock in recent years or to changes in environmental parameters. With respect to temperature, a previous analysis of changes in cod distribution does not suggest a temperature preference, at least not for the juveniles (Heessen and Daan, 1994).

More information regarding changes in cod distribution is given in Section 13.1.3.1, below.

#### 13.1.2.1.2 Skagerrak and Kattegat

##### Context - pattern of numbers, biomass, landings, and fishing mortality

The state of cod in the Skagerrak and Kattegat is uncertain, but indications are that cod stocks have been declining for two decades until recently. Landings in the Skagerrak in 1996 were 16,400 t (plus a by-catch of 900 t from the industrial fishery), compared with 12,100 t in 1994. Landings from the Kattegat were 6100 t in 1996.

##### Changes in size distribution and/or age composition

No information on size distribution or age composition was available.

##### Changes in spatial distribution

No information on spatial distribution was available.

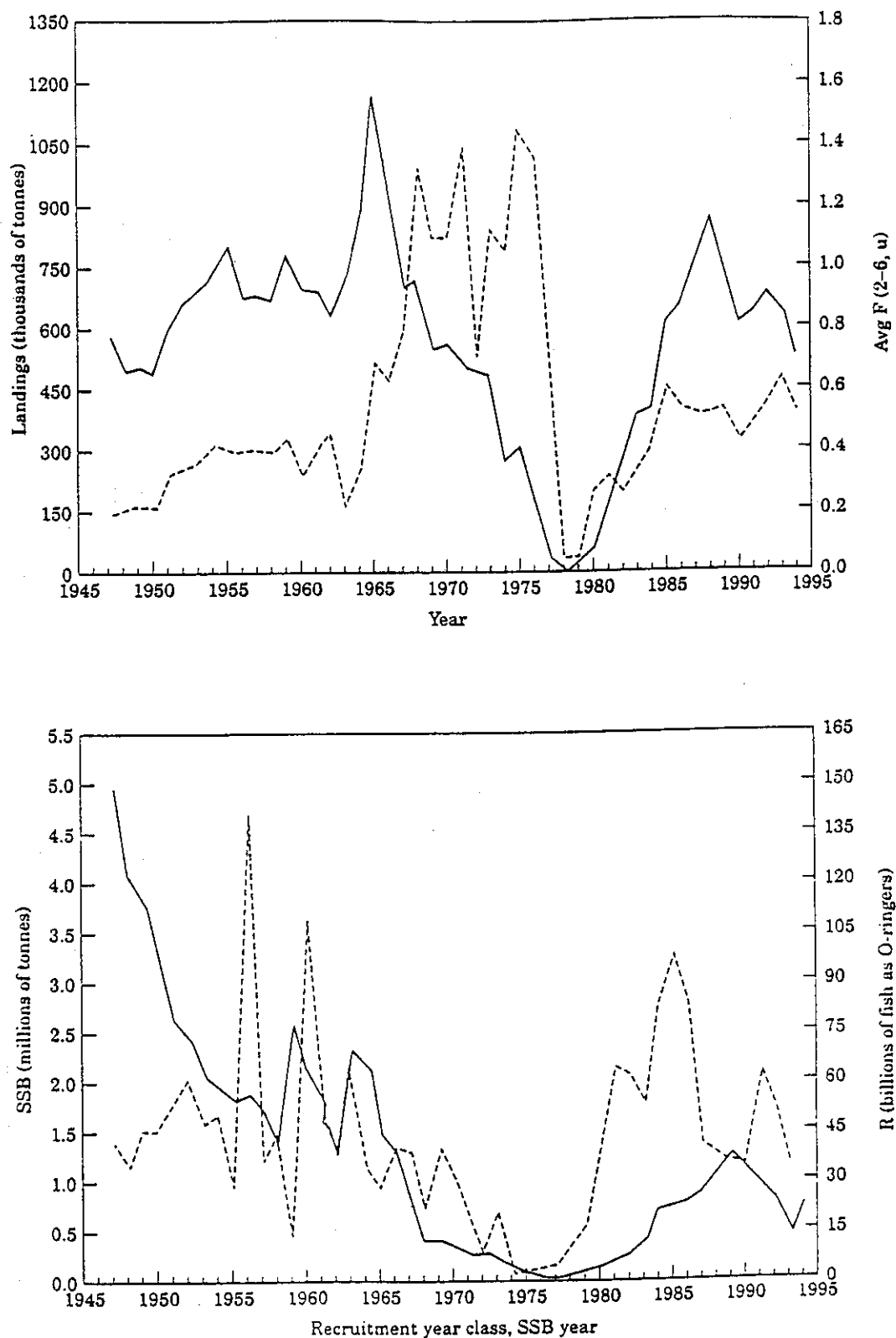
#### 13.1.2.2 Herring

##### Context - pattern of numbers, biomass, landings, and fishing mortality

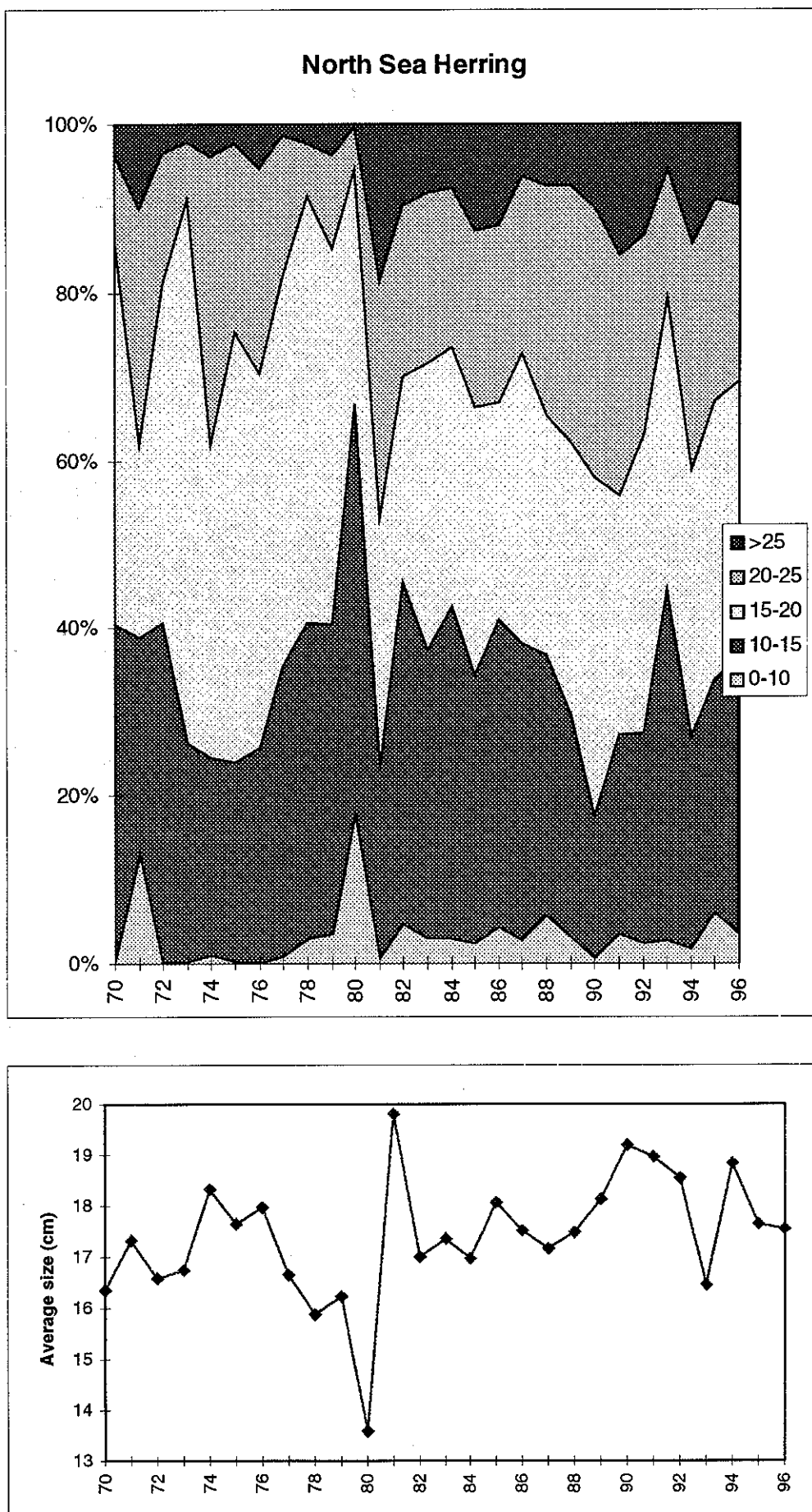
Herring are taken in several different fisheries in this region including, for example, the directed herring fishery (mainly for human consumption) in the North Sea as well as the industrial fishery in the Kattegat (Figure 13.1.2.2.1; Serchuk *et al.*, 1996). From 1951 to 1963 landings fluctuated between 600,000 t and 800,000 t, during which time fishing mortality was relatively stable at around  $F = 0.4$ . Landings then peaked briefly in 1965, at approximately 1.2 million t, then declined markedly to exceptionally low levels of less than 50,000 t in 1978. This coincided with a marked and sudden increase in fishing mortality, to values exceeding  $F = 1.0$  for most of the 1968 to 1976 period, associated with the rapid



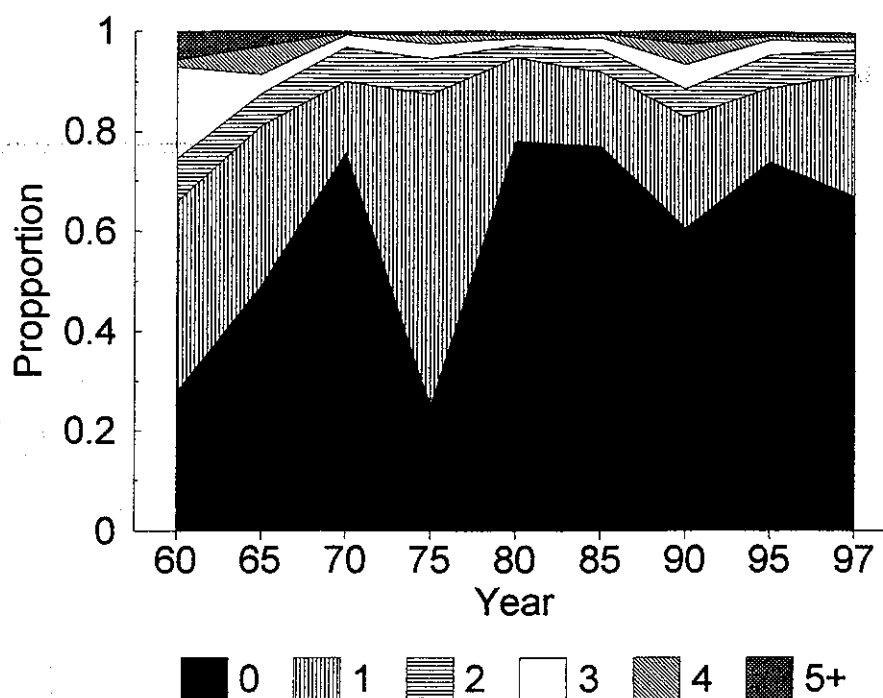
**Figure 13.1.2.2.1.** Trends in landings (—) and fishing mortality (---) (top) and spawning stock biomass (—) and recruitment (---) (bottom) for North Sea herring from 1947–1994. Landings include all North Sea autumn-spawning herring. Data from ICES (1996).



**Figure 13.1.2.2.2.** Changes over time in the proportion of herring at different length classes in the IBTS dataset (top) and trends in average size (bottom) from 1970–1996. The key shows length in centimeters.



**Figure 13.1.2.2.3.** Variation in the proportion of herring at age over the duration of the VPA (1960–1997). The key indicates age in years.



expansion of the purse seine fishery. The fishery for herring in the North Sea was closed in 1977 to allow stocks to recover and, on reopening in the early 1980s (1981 southern North Sea, 1983 northern North Sea), landings increased quickly to nearly 900,000 t in the late 1980s before dropping back again. Landings in 1994 and 1995 were less than 600,000 t, but in 1996 they dropped to 264,000 t. Since 1986 fishing mortality has fluctuated around  $F = 0.6$ , although exploitation of juvenile herring in the small mesh sprat fishery has increased substantially in recent years, reducing the long-term yield of adult herring and diminishing the future reproductive potential of the stock. The herring spawning stock in the North Sea has been below the MBAL level of 800,000 t since 1992. In 1995 ACFM decided that the herring stock was outside safe biological limits and that, if current levels of exploitation were maintained, it would be unlikely that MBAL would be regained.

#### Changes in size distribution and/or age composition

The IBTS data suggest that the proportion of herring caught that belong to the smaller size classes has declined slightly over the 26-year period covered by this survey, albeit with some marked year-to-year fluctuations. The mean size appears to have increased somewhat (Figure 13.1.2.2.2). This appears contradictory to the VPA proportion at age data where the proportion of one-year old and older fish in the population declined steadily during the 1960s. Data for only every fifth year, and

1997, were analysed here to detect trends. Apart from an increase in one-year olds in 1975, this pattern has remained rather constant during the 1980s and 1990s (Figure 13.1.2.2.3).

#### Changes in spatial distribution

The WGECO did not have adequate information to address this issue.

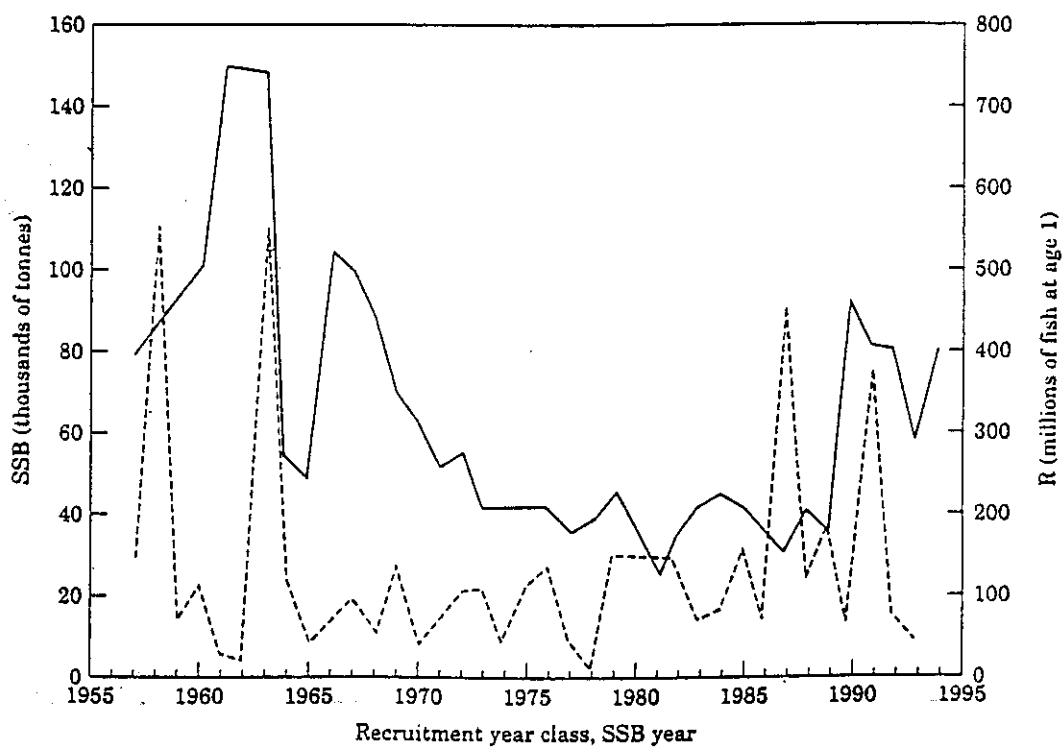
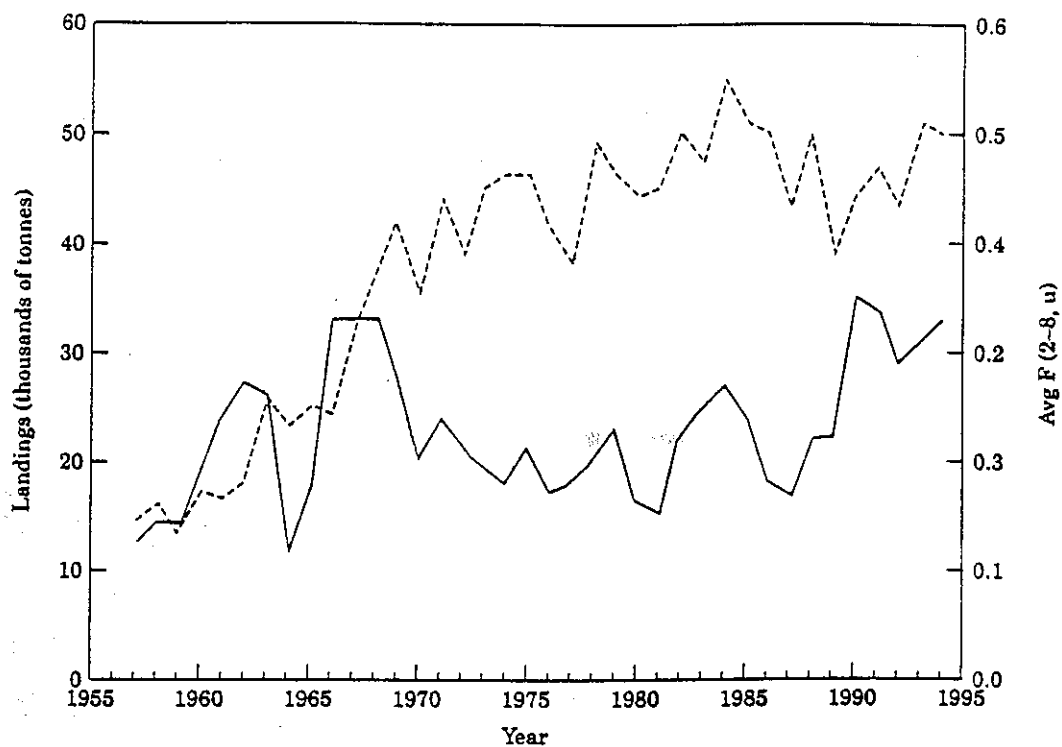
#### 13.1.2.3 Sole

##### 13.1.2.3.1 North Sea

#### Context - pattern of numbers, biomass, landings, and fishing mortality

Sole are exploited in a mixed (with plaice) beam trawl fishery in the southern North Sea and in a directed gillnet fishery in coastal areas. Landings doubled, from around 12,000 t to 26,000 t, from 1957 to 1962, coinciding with a peak in the spawning stock biomass of 150,000 t which was due to the recruitment of the outstanding 1958 year class (Figure 13.1.2.3.1.1; Serchuk *et al.*, 1996). High natural mortality associated with the severe 1962/1963 winter weather conditions caused a marked drop in the spawning stock biomass in 1964 and 1965 and this was reflected in a sharp fall in landings to 10,000 t in 1964. However, another good year class in 1963 resulted in an

**Figure 13.1.2.3.1.1.** Trends in landings (—) and fishing mortality (---) (top) and spawning stock biomass (—) and recruitment (---) (bottom) for North Sea sole from 1957–1994. Data from ICES (1996).



increase in both spawning stock biomass and landings in 1967 to 100,000 t and over 30,000 t, respectively. Over this period, fishing mortality increased sharply and continuously, from  $F < 0.2$  to  $F > 0.4$ . Over the next 15 years fishing mortality continued to increase steadily, reaching record levels in 1985 of  $F > 0.5$ . The spawning stock declined to around 40,000 t in 1974, while landings fell back to around 20,000 t in 1970 and both continued to fluctuate around these levels until 1989. A strong year class in 1987, coinciding with a slight drop in fishing mortality, back to  $F = 0.4$ , resulted in a marked increase in the spawning stock biomass to around 80,000 t in 1990, although landings increased to record high levels of approximately 35,000 t. A further good year class in 1991 sustained both spawning stock biomass and landings, despite a fishing mortality which again increased to  $F > 0.5$  in 1993. Fishing mortality has exceeded both  $F_{\max}$  (0.23) and  $F_{\text{med}}$  (0.33) since the late 1960s, suggesting both growth and recruitment overfishing. Despite this, however, the sole stock was considered to be within safe biological limits and the spawning stock biomass remained above MBAL and was expected to remain so over the short to medium term. However, extra natural mortality in the 1995/1996 winter appears to have affected the stock size considerably. Because the mortality level could not be quantified, the present state of the stock is uncertain, but it is believed to be below an agreed MBAL of 35,000 t in 1997. Landings decreased in 1996 to 22,500 t.

#### **Changes in size distribution and/or age composition**

The proportion of the sole population in each length category in the IBTS data set has fluctuated markedly over time. Likewise, average length has also shown considerable variation (Figure 13.1.2.3.1.2). In neither representation of the data was any particular trend apparent over time. However, the VPA data suggest a clear decrease in the proportion of fish six years of age and older in the population and this trend has been steady since the late 1960s (Figure 13.1.2.3.1.3). The proportion of two- and three-year-old fish has increased. These trends are associated with the increases in fishing mortality discussed in the previous section.

#### **Changes in spatial distribution**

Changes in the distribution of sole are addressed in Section 13.1.3.3, below.

##### **13.1.2.3.2 Skagerrak and Kattegat**

#### **Context - pattern of numbers, biomass, landings, and fishing mortality**

The catches of sole in the Skagerrak and Kattegat in 1996 amounted to 1059 t. The stock size is not known

precisely, but data from the fishery and surveys indicate that the stock was exceptionally large in the period 1988–1996. Recruitment now seems to be back to the pre-1988 level.

#### **Changes in size distribution and/or age composition**

No information on size distribution or age composition was available.

#### **Changes in spatial distribution**

No information on changes in spatial distribution was available.

##### **13.1.2.3.3 Eastern Channel**

#### **Context - pattern of numbers, biomass, landings, and fishing mortality**

Fishing mortality of sole in the Eastern Channel has increased from 0.36 in 1995 to 0.48 in 1996. After an increase following strong recruitment in the period 1989–1991, the spawning stock has decreased for two years, but stays above the historical minimum of 7000 t. In recent years, TACs for sole have not been restrictive. However, at the current level of fishing mortality, there is a relatively high probability (65 %) of the spawning stock biomass falling below 7800 t.

#### **Changes in size distribution and/or age composition**

Changes in size distribution and/or age composition of sole in the Channel are addressed in Section 13.1.3.3, below.

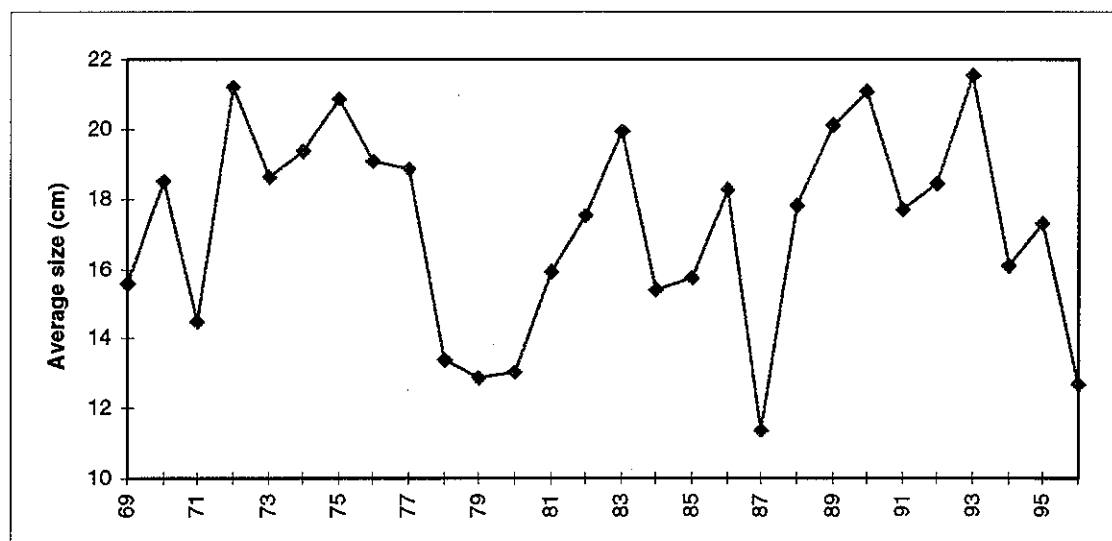
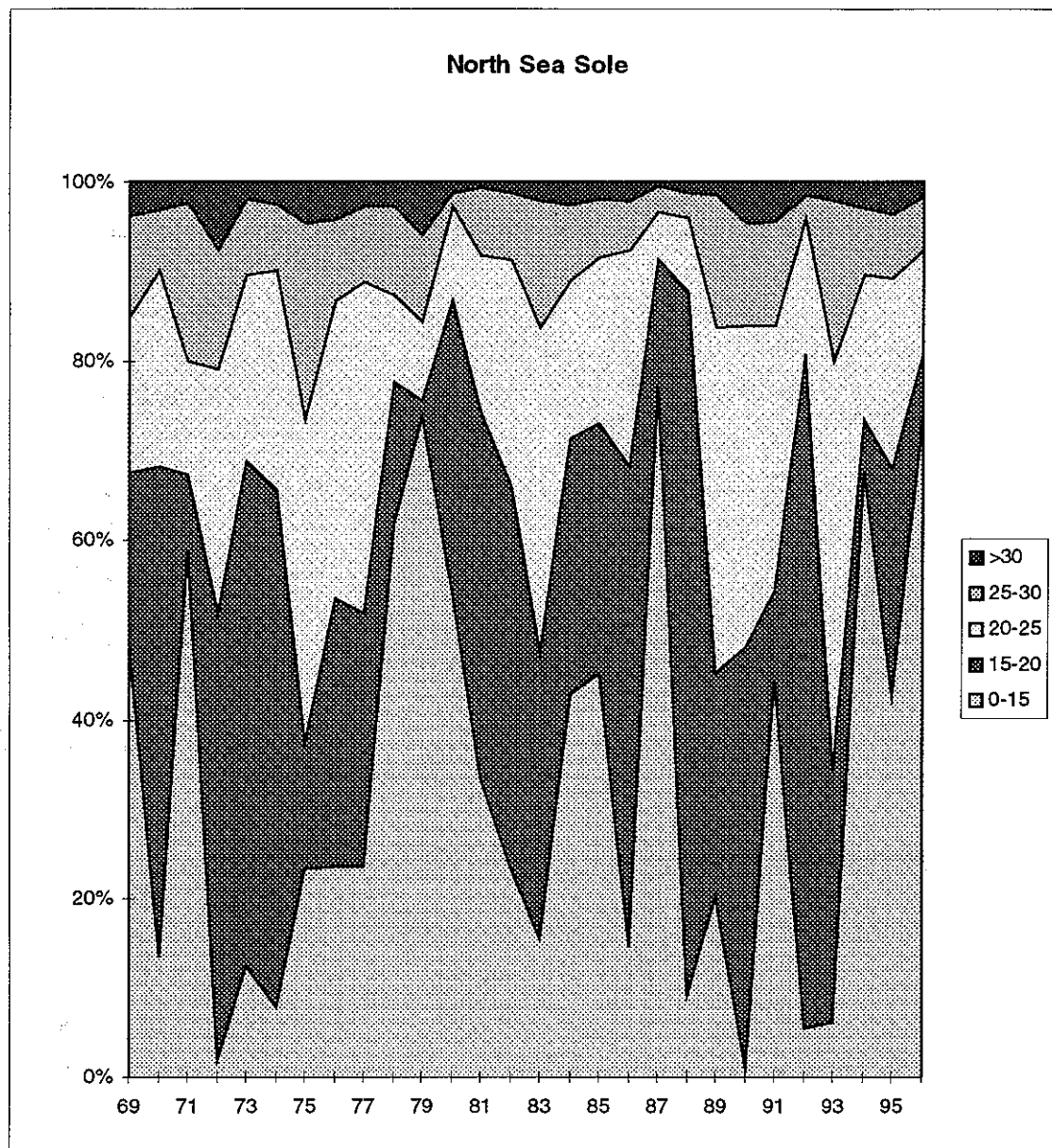
#### **Changes in spatial distribution**

Changes in the distribution of sole are addressed in Section 13.1.3.3, below.

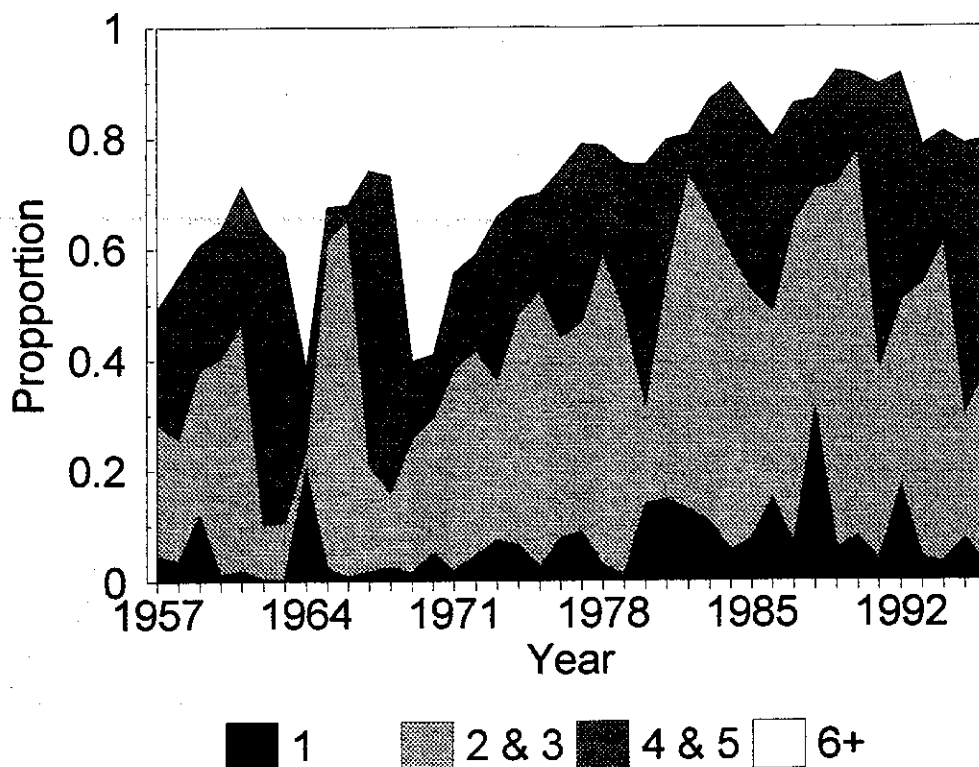
##### **13.1.2.4 Mackerel**

Two mackerel stocks are exploited within this region, the Western and the North Sea stocks. Mackerel, a highly migratory species, are harvested at different times of the year in different OSPAR regions (Serchuk *et al.*, 1996). The bulk of the catch of the combined stocks (North Sea, Western, and Southern) is taken in OSPAR Regions II and III. Changes in the size/age composition and spatial distribution of all three stocks are considered below in Section 13.1.4.3.

**Figure 13.1.2.3.1.2.** Changes over time in the proportion of sole at different length classes in the IBTS dataset (top) and trends in average size (bottom) from 1969–1996. The key indicates length in centimeters.



**Figure 13.1.2.3.1.3.** Variation in the proportion of sole at age over the duration of the VPA (1957–1996). The key indicates age in years.



#### 13.1.2.4.1 North Sea stock

##### Context - pattern of numbers, biomass, landings, and fishing mortality

The recent trends in this stock are shown in Figure 13.1.2.4.1.1. Before 1964, annual landings of the North Sea mackerel stock were less than 100,000 t. Spawning stock biomass at this time exceeded 3 million t (Jones, 1983). The development of purse seine technology in the early 1960s allowed an almost ten-fold increase in landings, to more than 900,000 t in 1967. This was followed by a drastic decline in landings to less than 200,000 t in 1971, during which time spawning stock biomass fell by 80 %. Strong recruitment from the 1969 cohort resulted in a relatively small, and temporary, rise in spawning stock biomass and landings, but by 1980, spawning stock biomass had again fallen to less than 200,000 t and during 1979 to 1986 landings ranged between 25,000 t and 66,000 t. After 1985 SSB has been estimated at between 50,000 t and 100,000 t. Since 1989 it has proved impossible to allocate catches of mackerel taken in the North Sea to either the North Sea or the Western stocks; catches from the North Sea stock have been assumed to be 10,000 t annually. The North Sea stock is considered to be outside safe biological limits and to require the maximum possible protection. Since 1980 ICES has recommended that no catches be taken, however, this can only be achieved by closing all mackerel fisheries in areas where North Sea mackerel occur. Consequently, ICES has recommended since 1991 that: no fishing for mackerel be allowed in Divisions IIIa,

IVb, and IVc at any time of the year; there should be no fishing for mackerel in Division IVa between 1 January and 31 July; and the minimum landing size of 30 cm and existing by-catch regulations in Division IIIa and Sub-area IV should be maintained. These regulations may encourage misreporting and discarding.

##### Changes in size distribution and/or age composition

Figure 13.1.2.4.1.2 shows variation in the proportion of mackerel at length in the IBTS data set. The proportion of small fish in the catches has been consistently high since 1986. Prior to this, it was more variable. Average length was also variable up to 1985, then declined in 1986 and has remained low since. For further details see Section 13.1.4.3, below.

##### Changes in spatial distribution

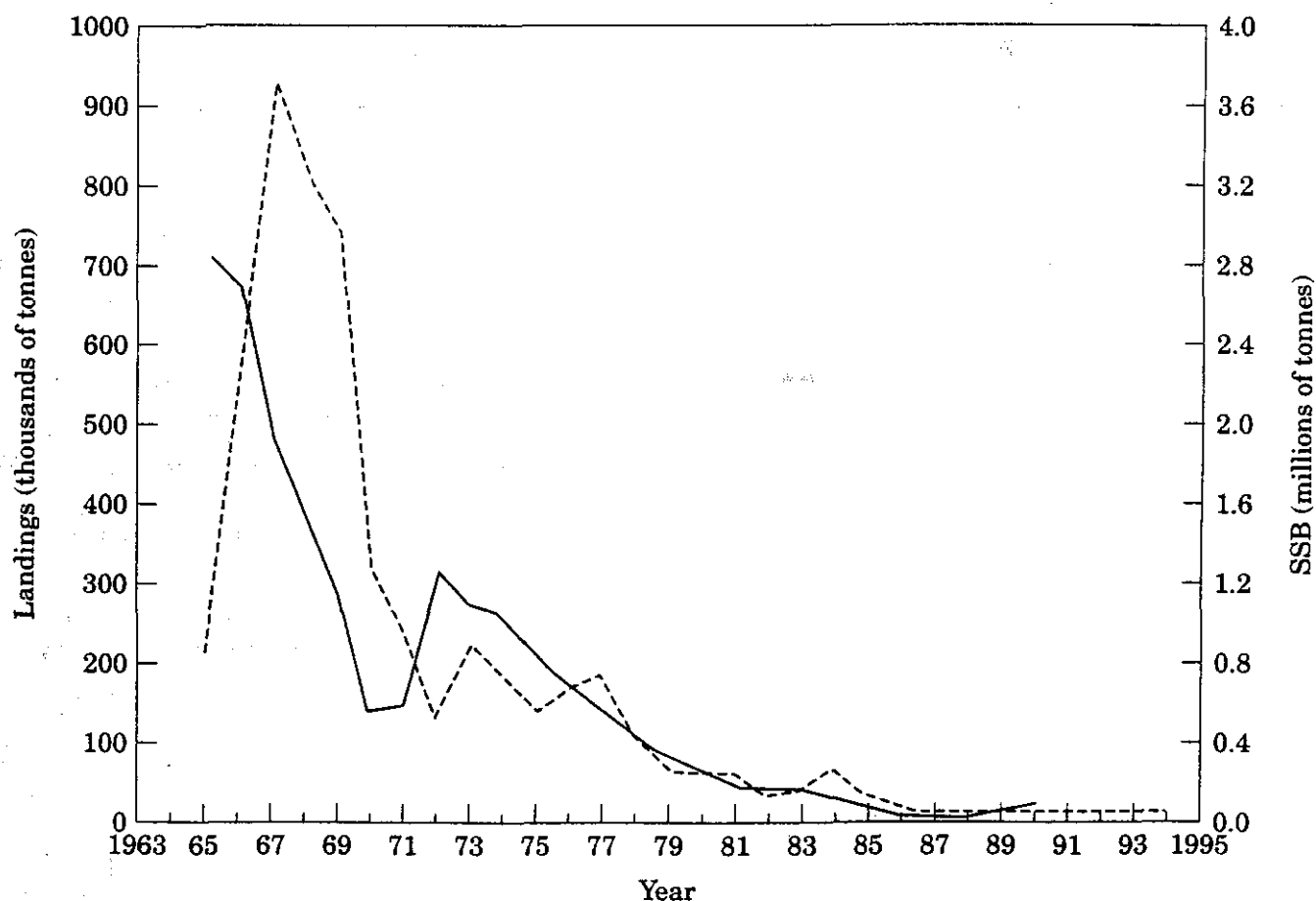
Changes in spatial distribution of North Sea mackerel are considered in Section 13.1.4.3, below.

#### 13.1.2.4.2 Western stock

##### Context - pattern of numbers, biomass, landings, and fishing mortality

Despite the collapse of the North Sea mackerel stock, landings of mackerel from the North Sea increased from 50,000 t in 1985 to nearly 475,000 t in 1994 due to a shift in the annual migration pattern in the Western

**Figure 13.1.2.4.1.1.** Trends in landings (---) and spawning stock biomass (—) for North Sea mackerel from 1965–1990. Data from ICES (1996).



mackerel stock. Since 1986, landings of Western mackerel taken in the North Sea have accounted for over 50 % of the total landings from this stock; in earlier years this proportion was less than 10 % (Figure 13.1.2.4.2.1). Up to 1994 the Western mackerel stock was considered to be well within safe biological limits. However, the spawning stock biomass has since declined to a record low level, while fishing mortality levels have reached a record high. Landings in 1995 declined to 322,000 t. The combined mackerel stock (Southern, Western and North Sea, of which the Western stock is the dominant component) may now be outside safe biological limits. ICES has determined that a significant reduction in fishing mortality in all areas where mackerel are caught, including international waters, is necessary to reverse the decline in spawning stock biomass; a 40 % reduction in fishing mortality is needed to prevent the spawning stock from declining further.

#### **Changes in size distribution and/or age composition**

Changes in the size/age composition of the Western mackerel stock are addressed in Section 13.1.4.3, below.

#### **Changes in spatial distribution**

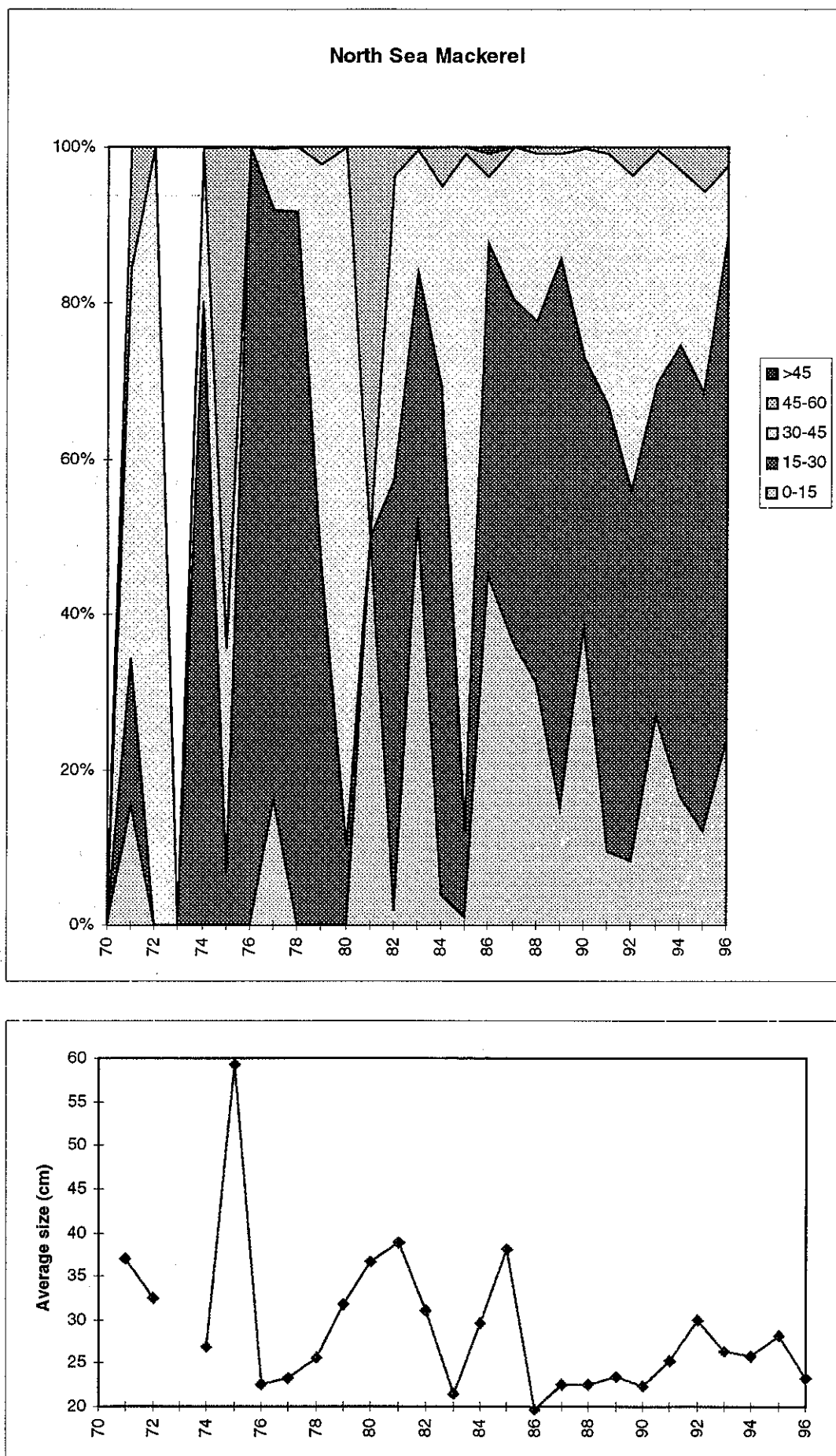
Changes in the spatial distribution of mackerel are described in Section 13.1.4.3, below.

#### **13.1.2.5 Hake**

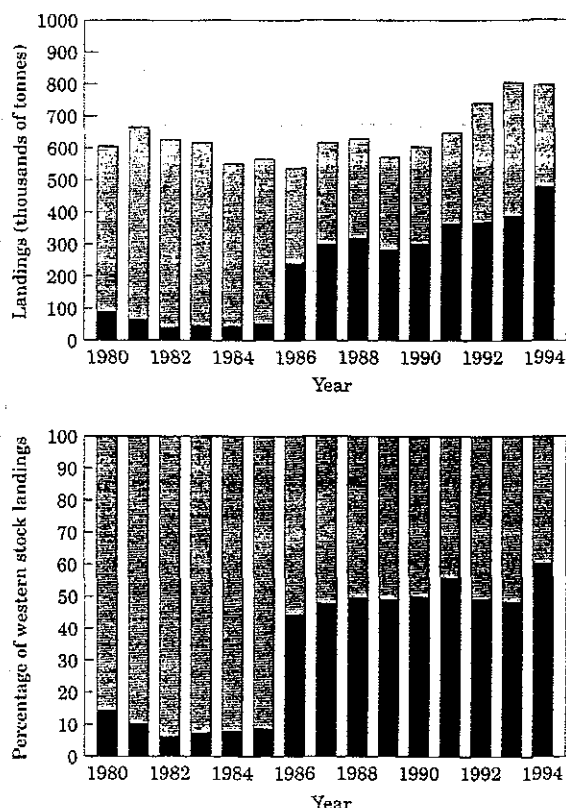
Hake are not targeted by the fisheries in this region.



**Figure 13.1.2.4.1.2.** Changes over time in the proportion of mackerel at different length classes in the IBTS dataset (top) and trends in average size (bottom) from 1970–1996. The key indicates length in centimeters.



**Figure 13.1.2.4.2.1.** Total landings of Western mackerel (■) and landings of Western mackerel taken in the North Sea (■) (top) and percentage of total Western mackerel landings taken in the North Sea (■) (bottom) from 1980–1994. Data from ICES (1996).



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Serchuk, F.M., Kirkegaard, E., and Daan, N. 1996. Status and trends of the major roundfish, flatfish, and pelagic fish stocks in the North Sea: thirty-year overview. *ICES Journal of Marine Science*, 53: 1120–1129.

## 13.1.3 OSPAR Region III: The Celtic Seas

The demersal fisheries in Division VIa are predominantly conducted by otter trawlers fishing for cod, haddock and whiting, with by-catches of saithe, anglerfish, megrim (*Lepidorhombus whiffiagonis*) and lemon sole (*Microstomus kitt*). These trawlers use mesh sizes of 80–100 mm depending on the area and may at times discard considerable quantities of young haddock and whiting. The majority of these vessels are locally based Scottish trawlers using light trawls, but vessels from Ireland, Northern Ireland, England, France and Germany also participate in this fishery. The pelagic fishery for herring is mainly operated by UK vessels in the north, and by Irish vessels in a roe fishery in the south. There is a directed fishery for blue whiting (*Micromesistius poutassou*), mackerel and horse mackerel in the area. The industrial fisheries in Division VIa are much smaller than those in the North Sea and are based on the irregular Scottish sandeel fishery which peaked in the late 1980s.

In the Irish Sea, the roundfish fisheries are conducted primarily by vessels from the bordering countries (UK and Ireland). The majority of vessels are otter trawlers fishing for cod, whiting and plaice, with by-catches of haddock, anglerfish, hake and sole. Since the early 1980s, there has been a development of semi-pelagic trawling for cod and whiting, predominantly by vessels from Northern Ireland. Although some of the otter trawlers also take part in the fishery for sole, there has been a growing number of beam trawlers, particularly from southern England and from Belgium exploiting this stock. The most important by-catches of this fleet are plaice, rays (*Raja* spp.), brill (*Scophthalmus rhombus*), turbot (*Scophthalmus maximus*) and anglerfish. A fleet of vessels, primarily from Northern Ireland and Ireland, takes part in a targeted *Nephrops* fishery, and all boats take a considerable by-catch of whiting, most of which is discarded. These discards comprise mainly juveniles, as the distribution of *Nephrops* coincides with the main nursery grounds of whiting. The main pelagic fishery in the Irish Sea is for herring, although the size of the fleet, mainly pair trawlers from Northern Ireland, has declined in recent years.

Most of the demersal fisheries in this area have a mixed catch. Although it is possible to associate specific target species with particular fleets, various quantities of cod, whiting, hake, anglerfish, megrim, sole, plaice and *Nephrops* are taken together, depending on gear types. In the Celtic Sea and Western Channel, fisheries for demersal species, mainly cod, whiting, sole and plaice, are conducted by Belgium, France, Ireland and the UK. The principal gears used are otter trawls and beam trawls. The targeting of sole and plaice using beam trawls became prevalent during the mid-1970s, leading to an increase in the landings of these two species. A gradual replacement of otter trawls by beam trawls has occurred in the Belgian and UK fleets. In the Bay of Biscay there has been a substantial increase in the coastal gillnet fishery targeting sole. A trawl fishery for anglerfish by Spanish and French vessels developed in the Celtic Sea and Bay of Biscay in the 1970s and expanded until 1990. The fishery has become dependent on small juvenile fish for which there is no minimum landing size. In addition, a gillnet fishery has developed in the Celtic Sea in the last decade.

*Nephrops* is an important component of the fisheries in this area. These fisheries developed in the 1970s and 1980s and effort increased continuously until recent years. Landings increased initially as effort increased, but they have tended to stabilize or decline at continuing high effort levels. The mesh size when fishing for *Nephrops* can lead to a significant by-catch of juvenile fish, notably hake.

There are separate trawl fisheries targeting herring in the Celtic Sea and mackerel and horse mackerel in the whole area. The herring fishery is principally a 'roe' fishery and discard rates have at times reached very high levels. There is also a small directed fishery for sprat in the Channel.

### 13.1.3.1 Cod

#### 13.1.3.1.1 Division VIa

##### Context - pattern of numbers, biomass, landings and fishing mortality

Over the past twenty years, fishing effort in ICES Division VIa (west of Scotland) has generally increased in the Scottish light trawl fleet and *Nephrops* trawl fleet, although the latter has shown some reductions in very recent years, and effort has also declined for the Scottish seine fleet. In 1994, Scottish trawl effort declined to a particularly low level, but has since risen to levels of the late 1980s and early 1990s. In addition to Scottish fleets, French trawlers have also been important in this area, and landed over 15,000 t in 1996. With estimates of misreporting since 1992 included in the assessment, the spawning stock biomass in 1996 is estimated to have been 15,600 t, well below the long-term mean of the series (26,200 t). Mean fishing mortality in 1996 (0.86) is

also well above the long-term average (0.77) and exceeds both  $F_{max}$  (0.274) and  $F_{med}$  (0.586) (Figure 13.1.3.1.1.1) (ICES, 1998a).

##### Changes in size distribution and/or age composition

The highest levels of fishing mortality are on three- and four-year-old fish (1.03 and 1.07, respectively), and on all ages over two the fishing mortality is above 0.84. For Division VIa cod, the strong 1996 year class is thought to contribute approximately 40 % to landings in 1998, and 35 % to SSB. Figure 13.1.3.1.1.2 shows the cod population age composition (in percent) according to number of fish in this area from 1966 to 1985. For this stock, the maturity ogive suggests that only 52 % of the two-year-old fish, and 86 % of the three-year-old fish, are considered to be mature.

##### Changes in spatial distribution

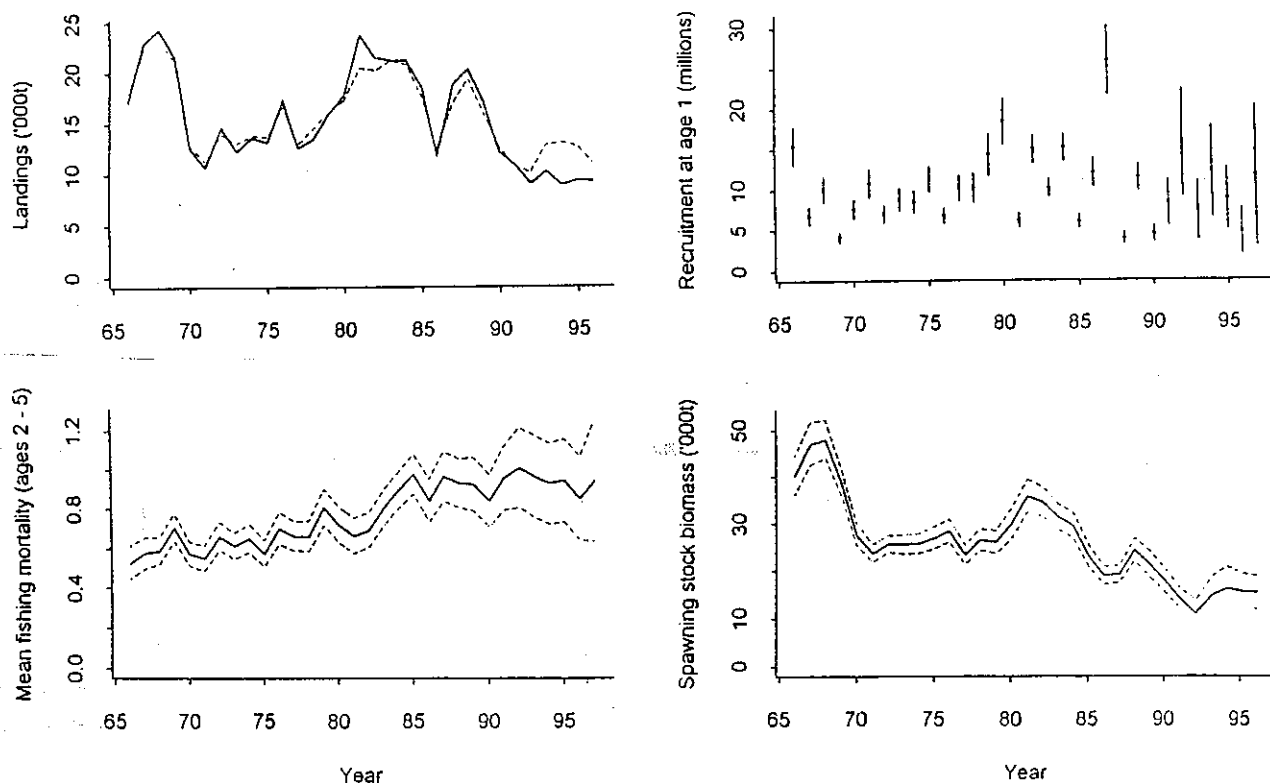
Cod is a species which generally shows a northern Arctic/boreal distribution, and is abundant in the northern part of this OSPAR region. The majority of landings are from the coastal waters off the west coast of Scotland close to the Hebrides, and beyond the western edge of the continental shelf, and the shelf waters to the west of Ireland. The 1996 landings from Division VIa were 9331 t and from Division VIb were 327 t. There is no information to suggest that the distribution of cod in this region has been altered as a direct result of commercial fishing activity (ICES, 1998a).

#### 13.1.3.1.2 Division VIIa

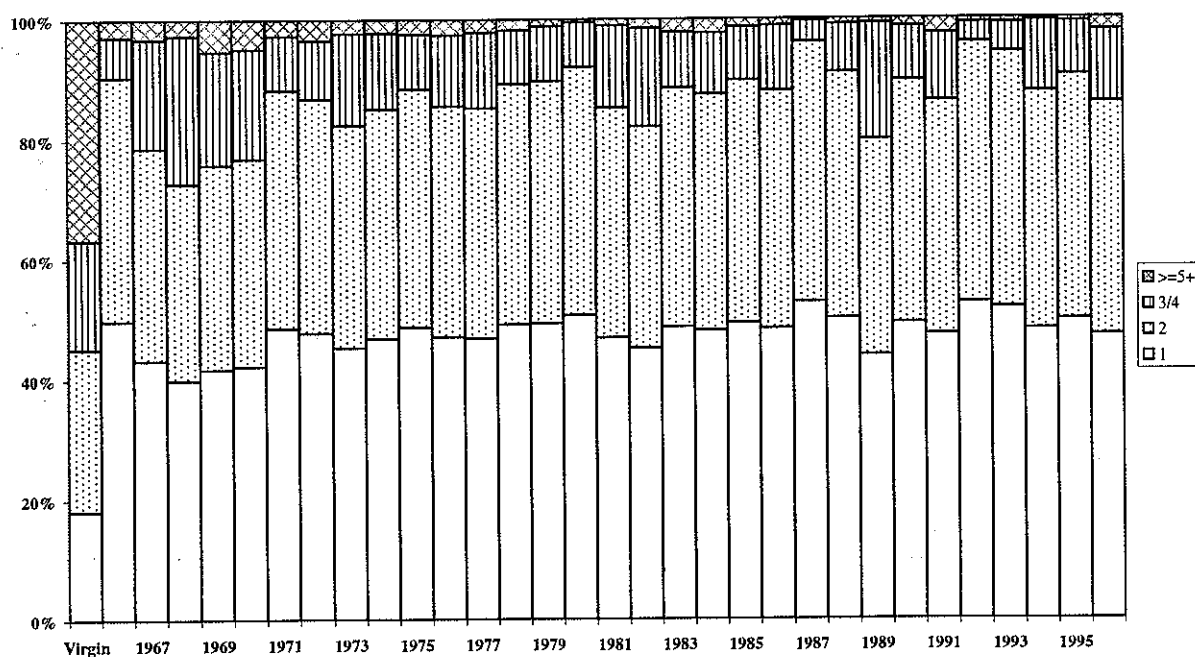
##### Context - pattern of numbers, biomass, landings and fishing mortality

The catch per unit of effort (CPUE) of the spring Northern Ireland fishery exploited by otter and pelagic trawl fleets in the western Irish Sea has declined continuously since 1986, although there was a sharp increase in CPUE of this fleet in 1996. The England and Wales otter trawl CPUE declined throughout the 1990s, and is still at a level of approximately half that of 1989. Fishing effort in all of the three main fleets has shown a marked decline since 1989 and, as a result, landings have declined from a peak of 14,000 t in 1988 to present levels of 4800 t (Figure 13.1.3.1.2.1). The SSB reached its lowest point in 1995 (Figure 13.1.3.1.2.1), and due to the poor state of this stock, the TAC was reduced substantially from 11,000 t in 1993 to 5800 t in 1995. The current estimate of  $F$  for this stock (0.58) exceeds  $F_{max}$  but lies below  $F_{med}$ , and sensitivity analysis suggests that the probability of the spawning stock biomass falling below  $B_{loss}$  in 1999 is negligible.  $B_{loss}$  is defined as the biomass where models suggest that the ability of the stock to recover is jeopardized. Nevertheless, cod are taken in a mixed fishery with haddock, whiting and plaice, and the implications of increased effort directed

**Figure 13.1.3.1.1.1.** Trends in landings (solid line is based on main estimate; dashed lines give alternate estimate) and recruitment (estimates, with error bars) (top), fishing mortality and spawning stock biomass (estimates in solid lines, with error limits in dashed lines) (bottom) for cod in Division VIa from 1966–1996.



**Figure 13.1.3.1.1.2.** Cod in Division VIa - population age composition (in percent) according to the number of fish estimated for each age group, from 1966–1985. The key indicates age in years.



on the large haddock year class entering the Irish Sea fishery should be considered (ICES, 1998a).

#### **Changes in size distribution and/or age composition**

Landings of cod are predominantly of two- to four-year-old fish, as landings of one-year-old cod have declined since 1991 and are the second lowest of the time series, and landings of five-year-old fish have also been the poorest on record. The 1995 and 1996 year classes (two- and three-year olds) will contribute 65 % to the 1998 landings (Figure 13.1.3.1.2.2). Maturity at age for this stock has been revised as a result of recent studies and data used in the assessments now indicate that 38 % of two-year-old fish, and 100 % of three-year-old fish, are mature (ICES, 1998a).

#### **Changes in spatial distribution**

Cod are found throughout the Irish Sea, but occur in the greatest abundance in the coastal and offshore waters of Ireland and in Liverpool Bay (Figure 13.1.3.1.2.3.a). There is insufficient evidence to confirm that the commercial exploitation of cod in this region has altered the spatial distribution of the species, as results from the beam trawl surveys show generally comparable distributions in western waters (i.e., the Irish Sea) between 1990 and 1996 (Figures 13.1.3.1.2.3.a and 13.1.3.1.2.3.b).

##### **13.1.3.1.3 Celtic Sea stocks**

#### **Context - pattern of numbers, biomass, landings, and fishing mortality**

Western Channel cod are now assessed with stocks in the Celtic Sea, so this description includes cod from Divisions VIIe, VIIf, VIIg, VIIh, VIIj, and VIIk. Only Divisions VIIe and VIIk are considered to lie outside OSPAR Region III.

The very strong 1986 year class resulted in high landings in 1988–1990, but since 1991, landings have returned to the levels recorded in the period 1981–1987, and for 1996 were 11,900 t for the entire region. The majority of these landings (80 %) were taken by France, but England, Wales, Belgium and Ireland also landed significant quantities of fish from this fishery. The spawning stock biomass of Celtic Sea cod reached a peak of 24,000 t in 1989 and subsequently decreased sharply to 6600 t in 1992 due to high fishing mortality and poor recruitment (Figure 13.1.3.1.3.1). With recruitment of the relatively good 1990 and 1991 year classes, SSB increased to 13,700 t in 1994 and 15,000 t in 1996, which is currently above the mean of the longest time series. After reaching the highest value of fishing mortality of over 1.0 in 1991,  $F$  decreased slightly until 1995 ( $F = 0.73$ ) and was estimated at 0.75 in 1996, which is above the mean, and

slightly below  $F_{med}$  (ICES, 1998b). Cod is a fast-growing and early maturing fish, and the predicted SSB in these assessments is heavily dependent on the assumed values of recruitment for the incoming year classes, and landings from this OSPAR region are increasingly dependent on strong year classes entering the fishery. Fishing mortalities are high and, at such levels, the contribution of good year classes to the SSB is very transitory.

#### **Changes in size distribution and/or age composition**

No data are available that identify changes in size or age composition as a result of exploitation.

#### **Changes in spatial distribution**

Cod are found throughout the region but at a reduced abundance relative to the unexploited stock. There are insufficient data on cod in this region to identify changes in the spatial distribution of the species resulting from commercial exploitation. Some year-to-year variation in distribution is observed, but this is attributed to the variation in recruitment: large year classes may be somewhat more widely distributed than small ones. Beyond this factor, no changes in spatial distribution over time have been reported.

##### **13.1.3.2 Herring**

###### **13.1.3.2.1 Division VIa (North)**

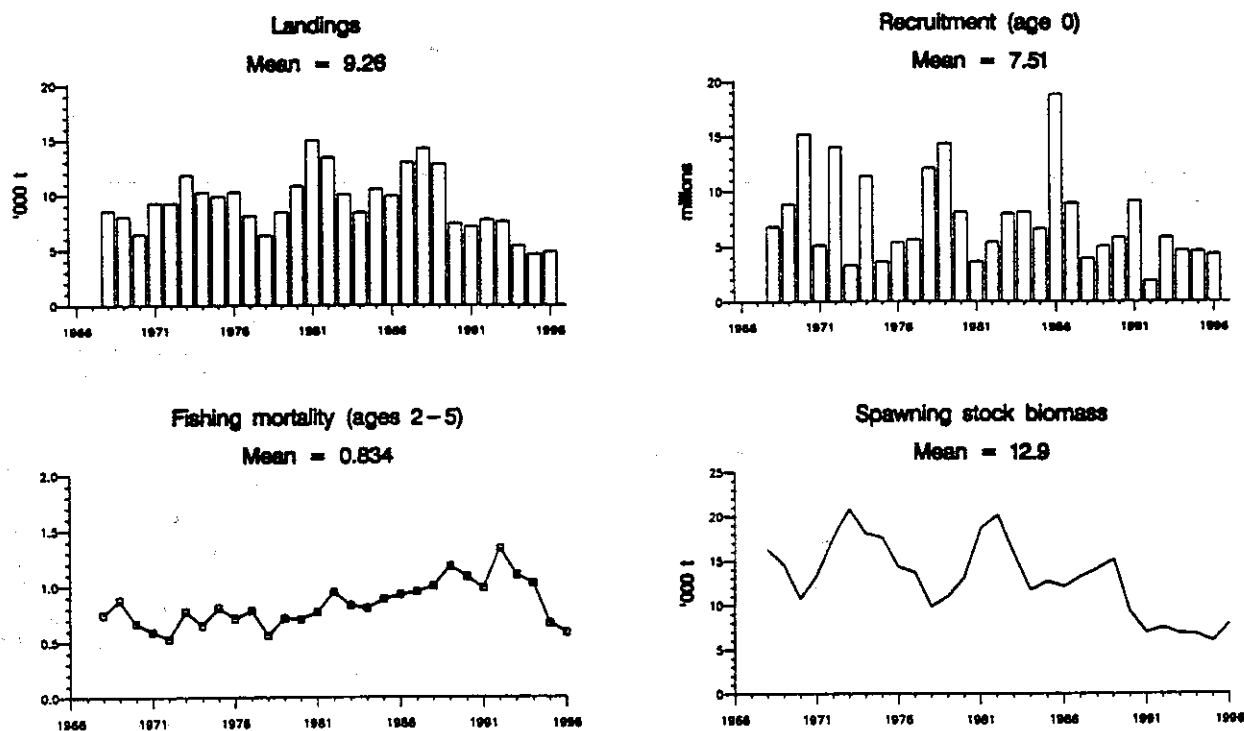
#### **Context - pattern of numbers, biomass, landings, and fishing mortality**

Continued difficulties with catch reporting exist for this stock, and misreporting (reporting the catch as being from a different area than that in which it was actually caught, e.g., the North Sea) is thought to be approximately 68 % of the total catch (ICES, 1997). The problem is particularly acute during the peak months of the herring fishery around Shetland (August to October). Acoustic surveys have been used to estimate SSB at 370,000 t, which is lower than values derived in 1993, which was an exceptionally high stock estimate thought to have been affected by an influx of populations from other regions. Assessment of this stock suggests that it is lightly exploited, with little risk of a stock decline at the current levels of exploitation.

#### **Changes in size distribution and/or age composition**

Current estimates of stock size are still influenced by poor sampling of weight at age, and additional weight information needs to be collected from offshore regions for future assessments. Until more complete data are available, it is unclear how the continued exploitation of this stock has affected the age composition.

**Figure 13.1.3.1.2.1.** Trends in landings and recruitment (top), fishing mortality and spawning stock biomass (bottom) for cod in the Irish Sea (Division VIIa) from 1969–1996.



**Figure 13.1.3.1.2.2.** Cod in the Irish Sea - population age composition (in percent) according to the number of fish estimated for each age group, from 1977–1996. The key indicates age group in years.

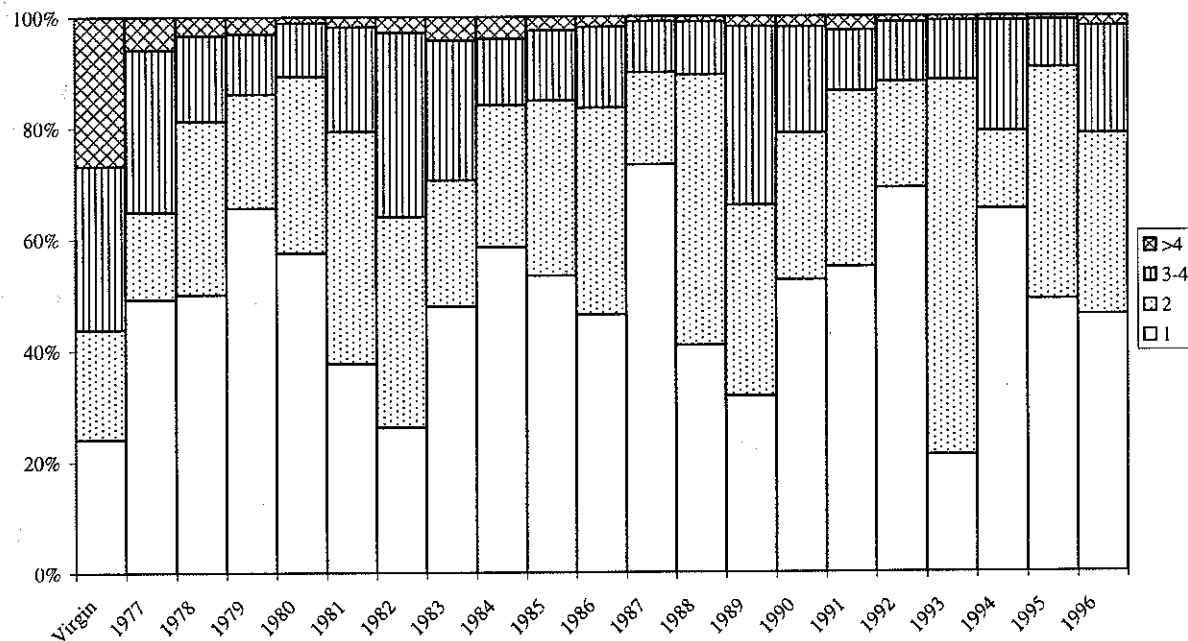


Figure 13.1.3.1.2.3.a. Distribution of cod, as determined by the International Beam Trawl Surveys, in the third quarter of 1990.

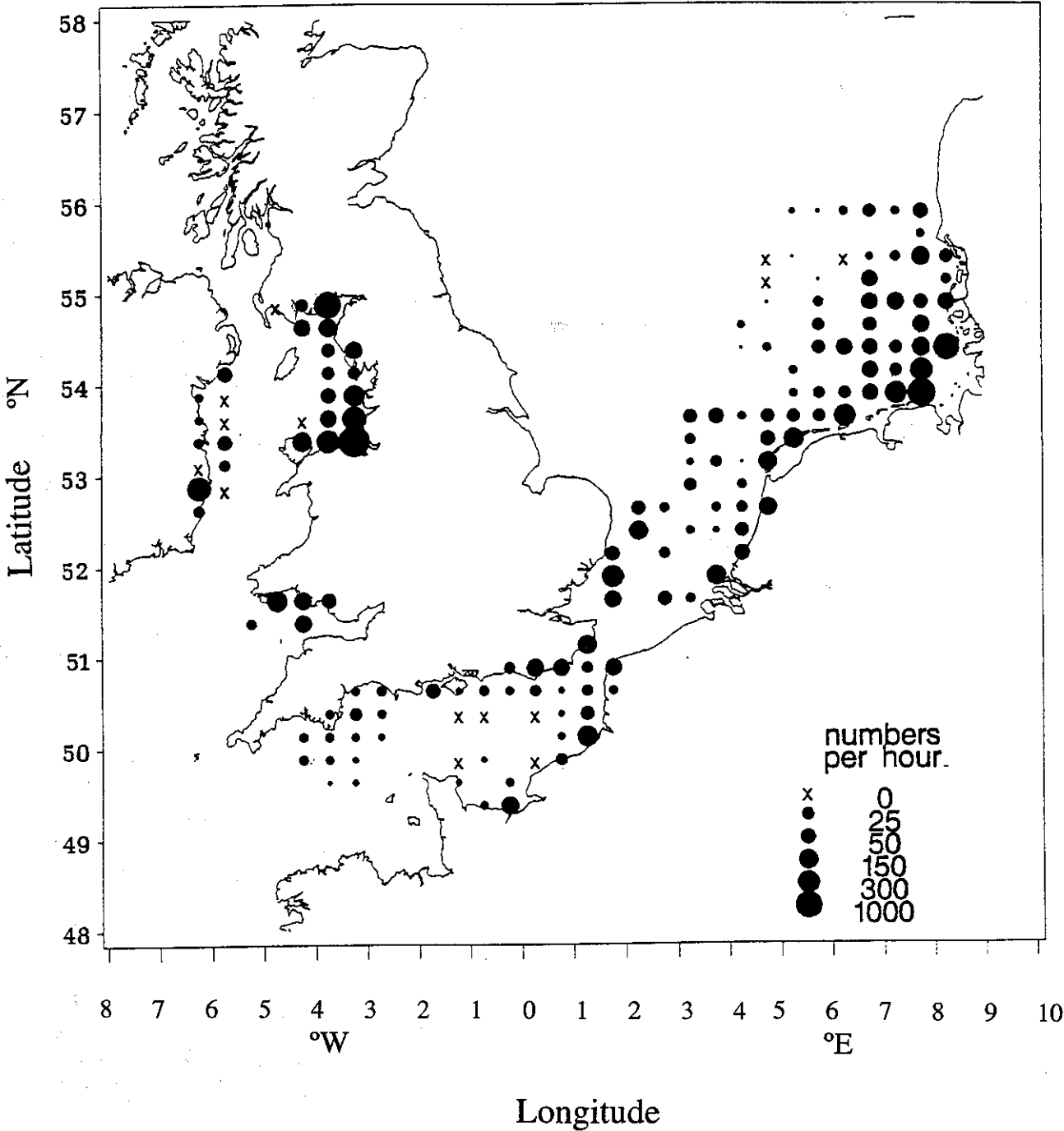
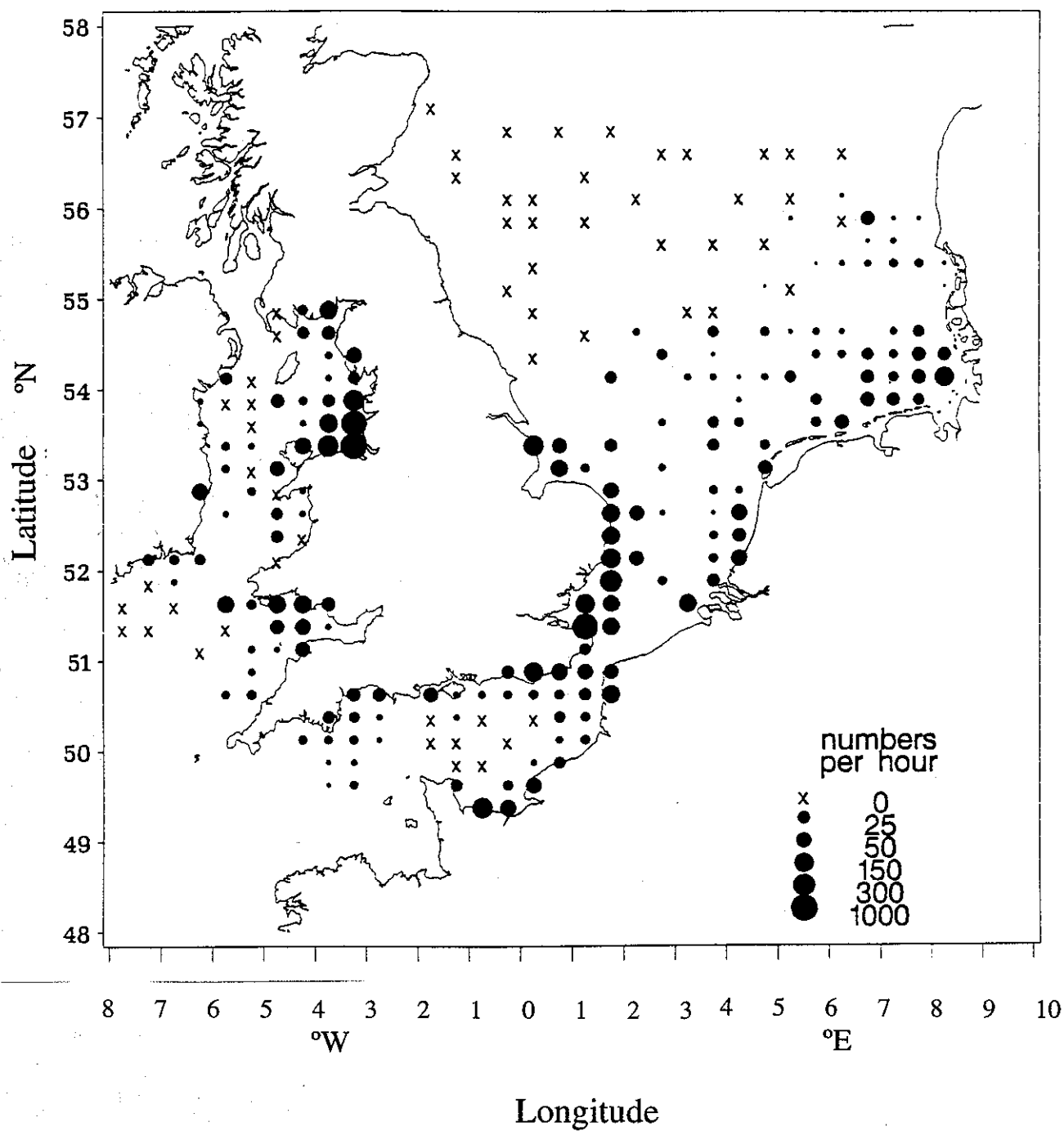
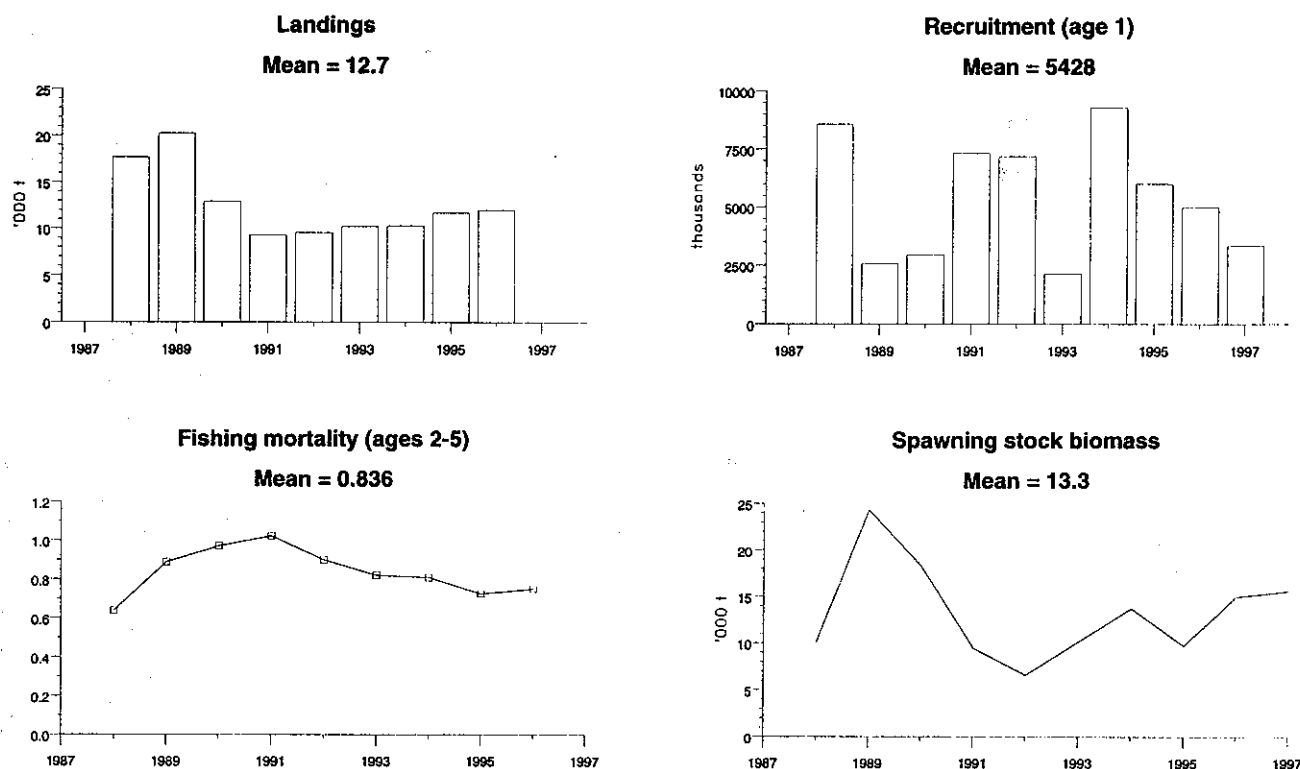


Figure 13.1.3.1.2.3.b. Distribution of cod, as determined by the International Beam Trawl Surveys, in the third quarter of 1996.





**Figure 13.1.3.1.3.1.** Trends in landings and recruitment (top), fishing mortality and spawning stock biomass (bottom) for Celtic Sea and Western Channel cod from 1971–1995.



### Changes in spatial distribution

The herring fishery in the northern part of Division VIa takes place in two main areas: certain vessels fish inshore for small, younger herring, whilst other vessels fish offshore in deeper waters where the fish are larger and older. The distribution of herring in the fourth quarter in this region and other parts of OSPAR Region III is shown in Figure 13.1.3.2.1.1. There are no data which can be used to identify how the spatial distribution of the species has altered as a result of commercial exploitation.

#### 13.1.3.2.2 Clyde Herring

##### Context - pattern of numbers, biomass, landings, and fishing mortality

Management of this stock is complicated by the presence of two virtually indistinguishable stocks: a resident spring-spawning population and the immigrant autumn-spawning component from Division VIa. In recent years, management has been directed at rebuilding the highly depleted spring-spawning component to historical levels, using closed areas and seasons. Historically, this spring-spawning stock supported a fishery with catches of up to 15,000 t per year in the 1960s, and landings generally began to decline through the 1970s and 1980s, until at present the landings have fluctuated at below 1000 t. As

there are no fishery-independent surveys and no stock separation of the catches, the current state of the spring-spawning stock is unknown (ICES, 1997).

##### Changes in size distribution and/or age composition

No data are available which identify changes in size or age composition as a result of exploitation.

##### Changes in spatial distribution

Spatial changes in the distribution of the autumn-spawning component can be explained by environmental factors affecting the distribution of migrating species, but there are no data to suggest that exploitation of the spring-spawning component has affected the spatial distribution. Herring spawn on coarse sand and gravel sediments and the location of these substrates influences spawning distribution.

#### 13.1.3.2.3 Divisions VIa (South), VIIb, and VIIc

##### Context - pattern of numbers, biomass, landings, and fishing mortality

There have been no recent analytical assessments of this stock, but there is some evidence that the stock has declined in recent years and is now at a comparatively

low level. There has been no substantial recruitment to the stock in recent years and the very strong 1985 year class has now passed through the fishery (ICES, 1997).

#### **Changes in size distribution and/or age composition**

No data are available which identify changes in size or age composition as a result of exploitation.

#### **Changes in spatial distribution**

The scarcity of herring in this region may be due to a combination of a decline in the stock, accentuated by a more northerly distribution of the stock in recent years resulting from environmental factors.

#### **13.1.3.3 Sole**

##### **13.1.3.3.1 Division VIIa**

#### **Context - pattern of numbers, biomass, landings, and fishing mortality**

Sole is at the northern limit of its distribution in the Irish Sea and the major fisheries for the species are in the Irish Sea, the Bristol Channel and the Celtic Sea, where it is mainly taken in a beam trawl fishery with plaice as a by-catch. In the Irish Sea, sole spawning stock biomass is currently (spawning time in 1996) at its historically lowest level, and is only 60 % of the average observed over the period 1970–1996 (Figure 13.1.3.3.1.1). Since the good 1989 year class, there have been five consecutive below-average recruitments. The prediction for spawning stock biomass in 1997 is 17 % below  $B_{loss}$ , and sensitivity analysis would suggest that the probability of spawning stock biomass remaining below  $B_{loss}$  in 1999 is about 45 %. Whilst this stock is considered to be outside safe biological limits, the population contains a broader distribution of year classes than cod, for example, and would be expected to decline slowly. Recent studies of sole maturity have suggested that for western sole stocks a maturity rate of about 70 % applies to three-year-old fish, and fish are considered fully mature (98 %) at age 5 (ICES, 1998a).

#### **Changes in size distribution and/or age composition**

Although no specific studies are known of changes in the age composition of sole in this Division, data from Working Group assessments show that the 1994 and 1995 year classes (two- and three-year olds) would form approximately 20 % of the catch in 1997 and approximately 50 % of the catch in 1998. This confirms that the age structure of Irish Sea sole is dominated by young fish and the fishery is heavily dependent on them (Figure 13.1.3.3.1.2).

#### **Changes in spatial distribution**

Flatfish abundance data from the Irish Sea have been collected since 1988 by the UK as part of a programme to monitor variation in recruitment of sole stocks on the south and west coasts of England. The International Beam Trawl Surveys collate these data with those of other surveys in the North Sea, and data for 1990 and 1996 are shown in Figures 13.1.3.3.1.3.a and 13.1.3.3.1.3.b (ICES, 1990; Rogers *et al.*, 1997). Although the spatial distribution of the surveys has varied slightly during the seven-year period, most notably with the inclusion of the central North Sea in recent years, the centres of peak abundance of sole in Liverpool Bay and Solway Firth remain constant. Data collected from a longer time period would be required to compare the distribution of sole at a time when fishing activity was lower, but the requirements of this species for specific fine and productive sand/mud sediments suggest that changes in distribution would be limited, and are governed by environmental factors (Symonds and Rogers, 1995).

##### **13.1.3.3.2 Celtic Sea**

#### **Context - pattern of numbers, biomass, landings, and fishing mortality**

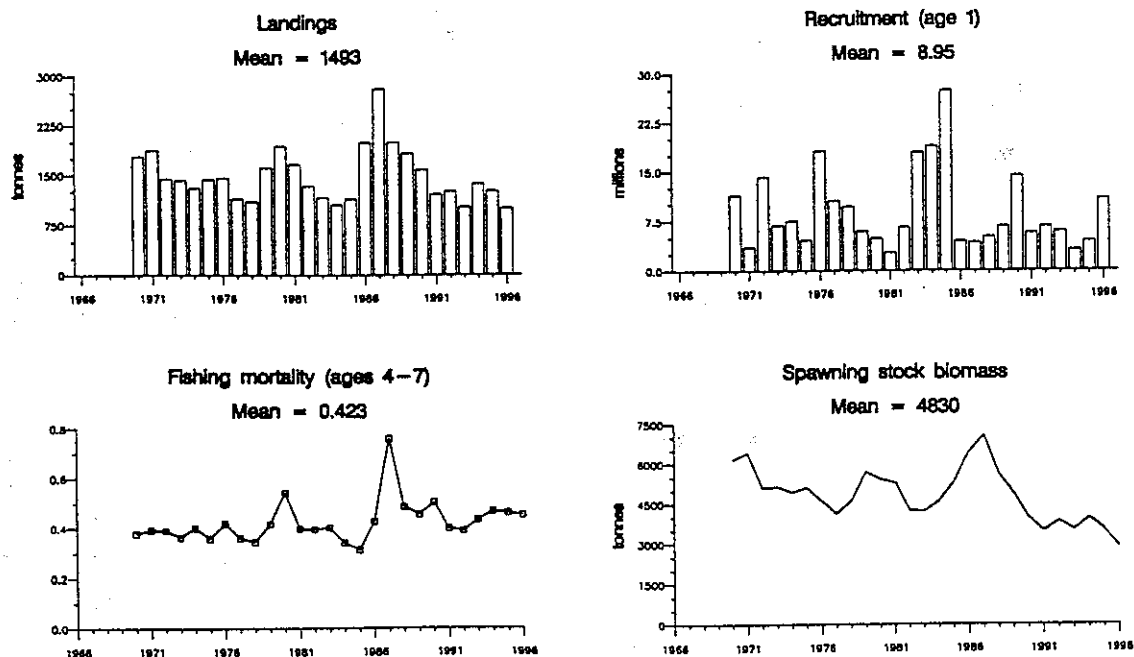
Sole stocks in Divisions VIIf and VIIg (the Bristol Channel) are assessed by ICES. Total international landings were 994 t in 1996, and the largest proportion of these landings were taken by Belgian beam trawlers. Assessments have shown that fishing mortality has increased from around 0.29 in the 1970s to a peak of 0.65 in 1990, and  $F$  is currently at 0.48 (Figure 13.1.3.3.2.1). This value of  $F$  is 41 % above  $F_{med}$  and 2 % above  $F_{high}$ . At the current fishing mortality, there is a greater than 50 % probability that SSB will fall below  $B_{loss}$  in 1999, and that a reduction of 25 % in fishing mortality is required in order to ensure that SSB remains above  $B_{loss}$  in 1999. A slight decline in  $F$  during the early 1990s has been explained by the greater time spent by beam trawl fleets in other fishing areas. Recruitment has fluctuated without any trend, however, and the 1989 year class was outstanding and equivalent only to that of 1970. Sole is taken mainly in a beam trawl fishery, with plaice as a by-catch, and to a lesser extent in the otter trawl fisheries, so management advice needs to take into account measures proposed for plaice (ICES, 1998b).

#### **Changes in size distribution and/or age composition**

Although no specific studies are known of changes in the age composition of sole in the Celtic Sea, data from Working Group assessments suggest that young fish are becoming an increasingly larger proportion of the landings from this fishery (Figure 13.1.3.3.2.2).

[illegible]

**Figure 13.1.3.1.1.** Trends in landings and recruitment (top), fishing mortality and spawning stock biomass (bottom) for sole in the Irish Sea (Division VIIa) from 1966–1996.



**Figure 13.1.3.1.2.** Sole in the Irish Sea (Division VIIa) - population age composition (in percent) according to the number of fish estimated for each age group, from 1977–1996. The key indicates age group in years.

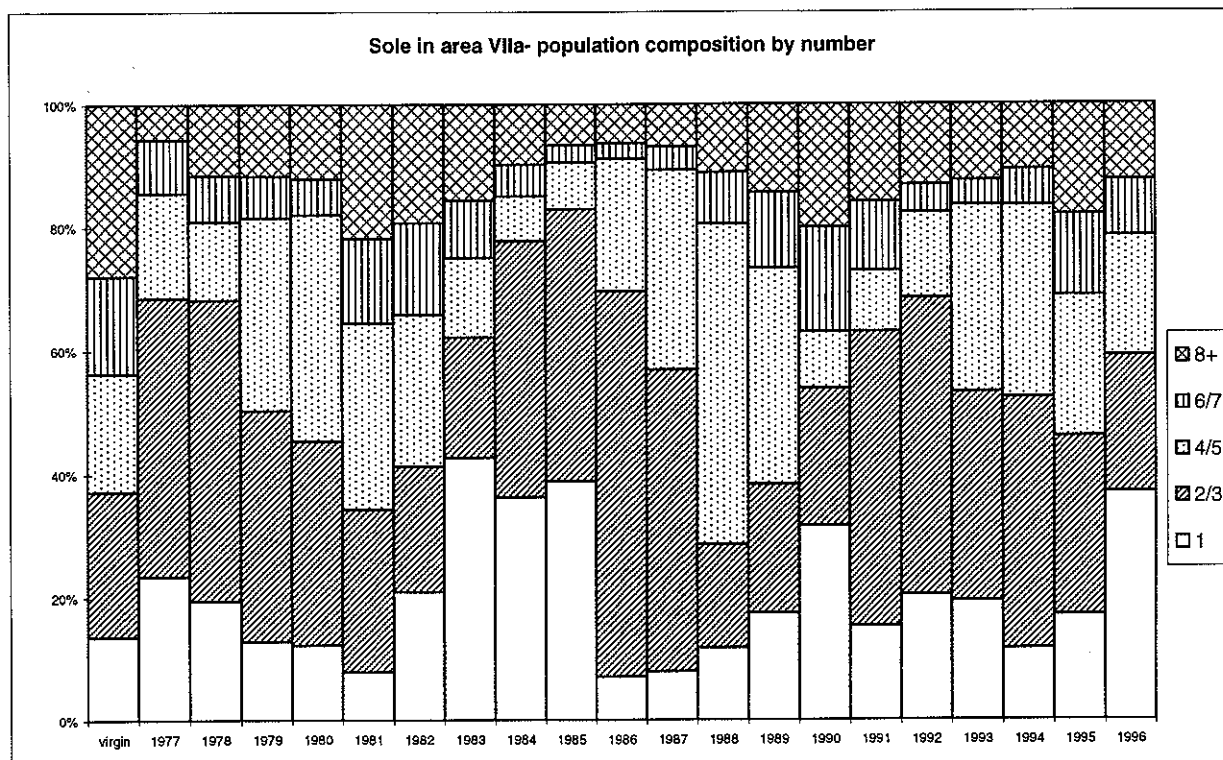


Figure 13.1.3.1.3.a. Distribution of sole, as determined by the International Beam Trawl Surveys, in the third quarter of 1990.

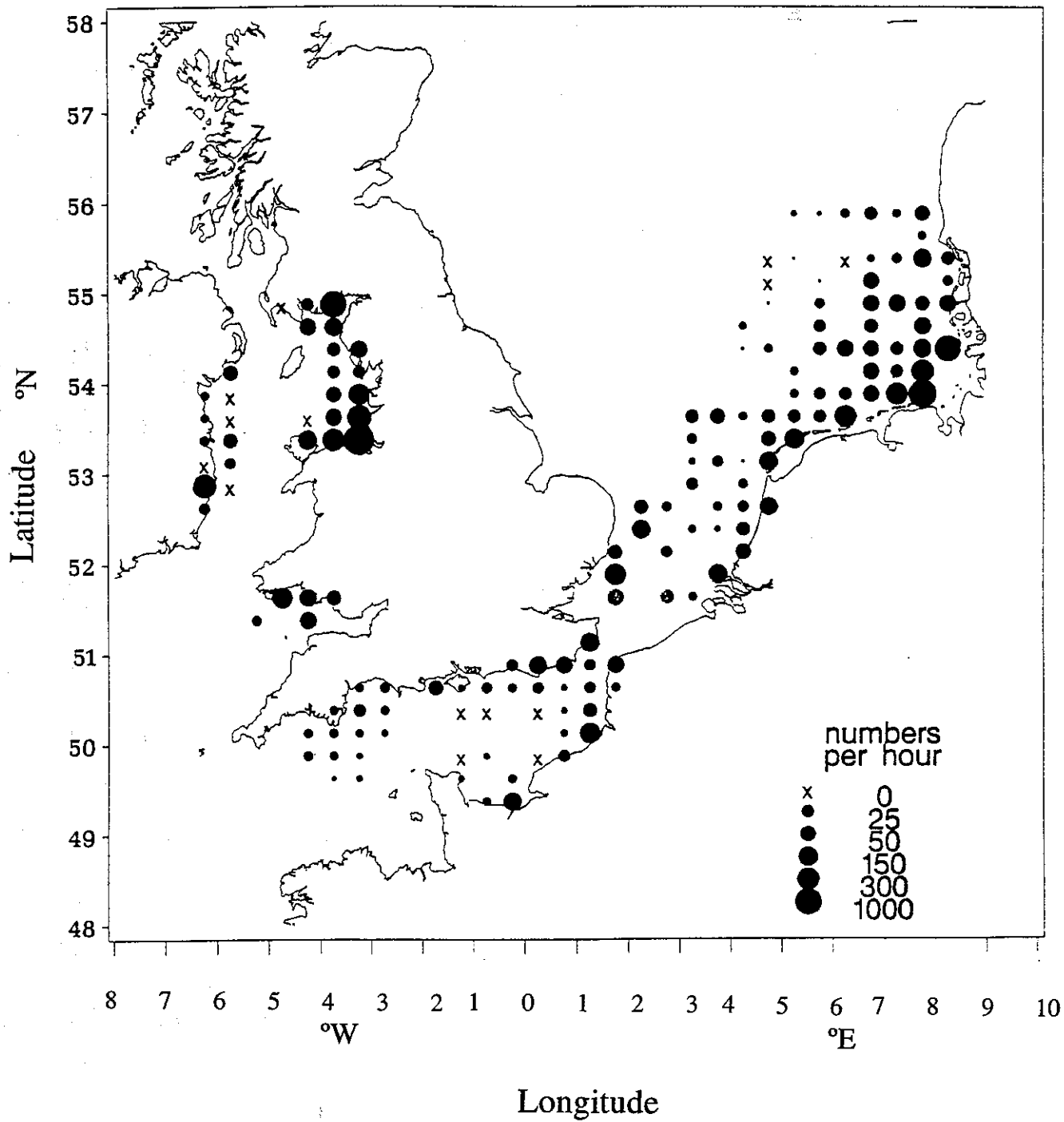
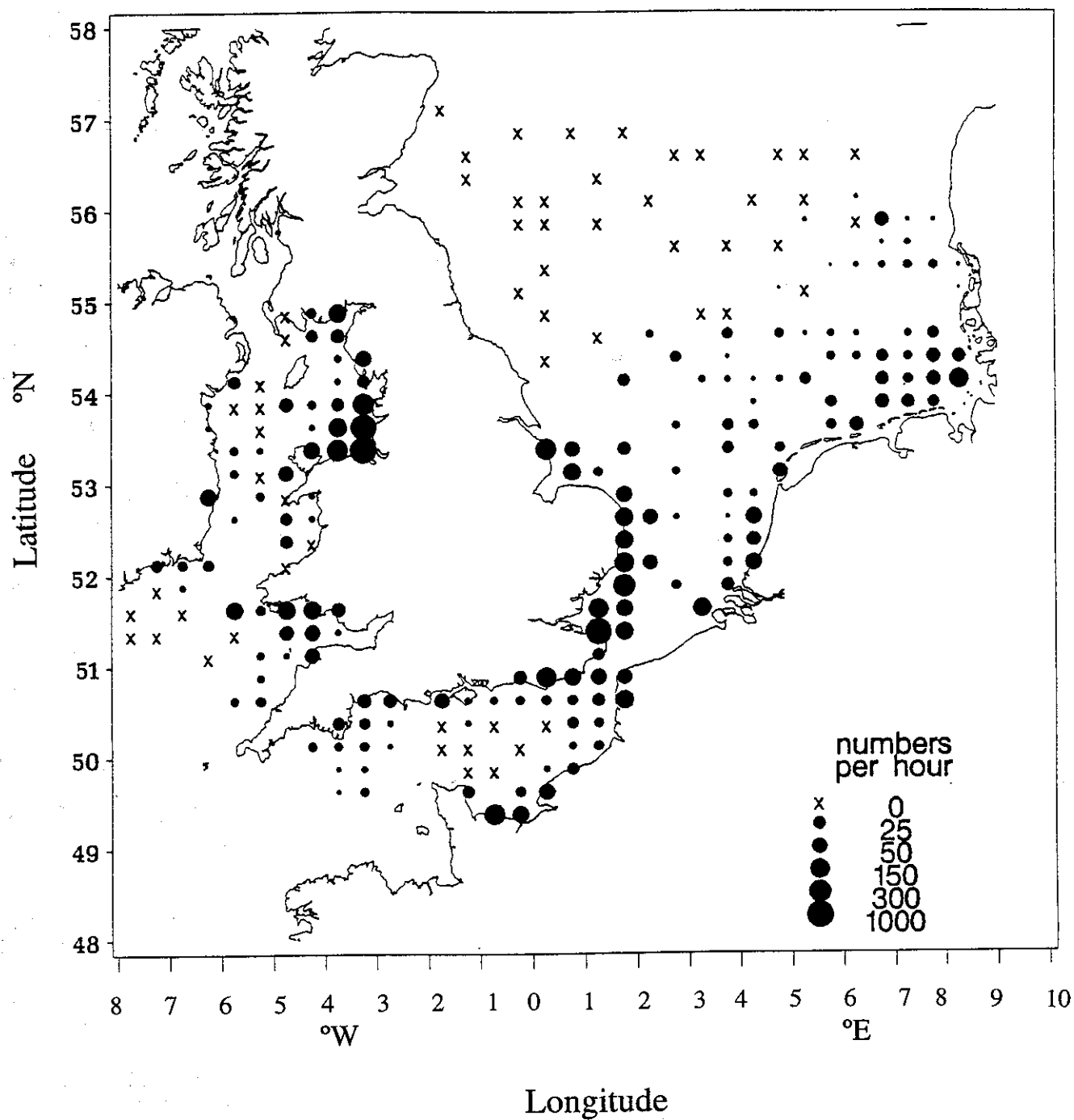
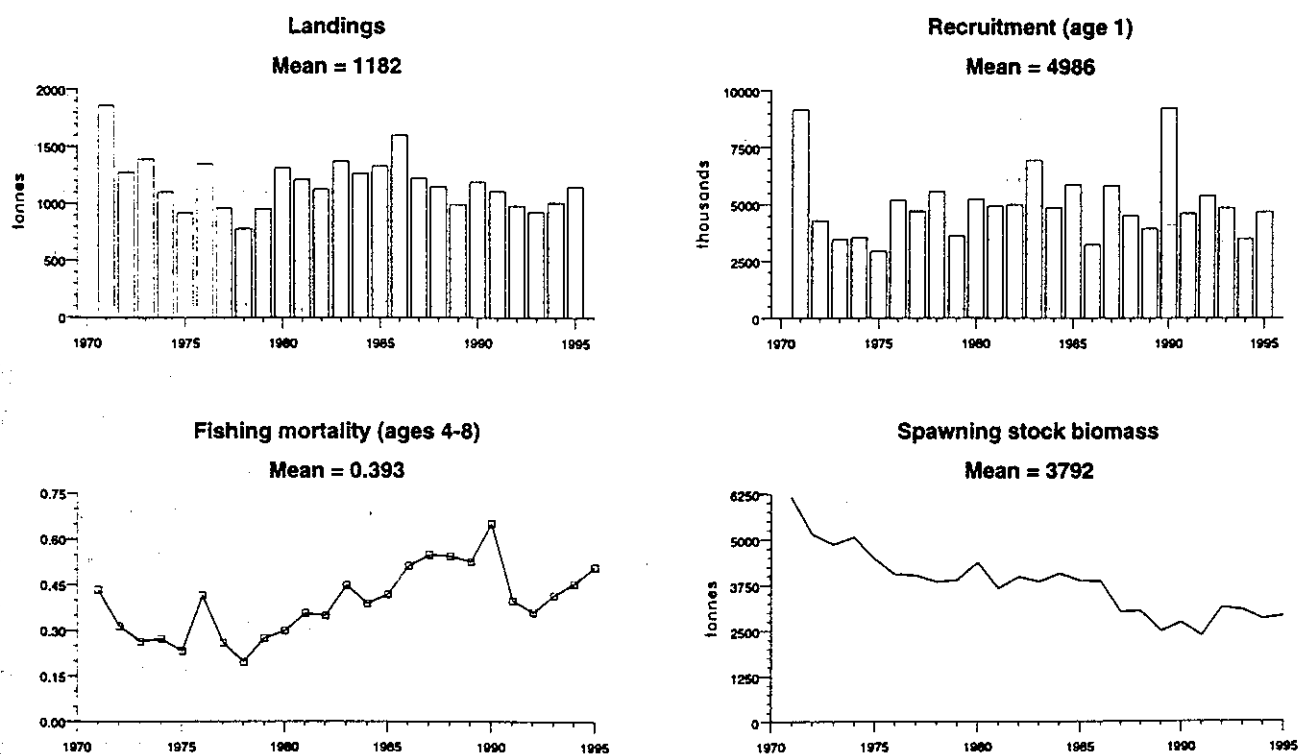


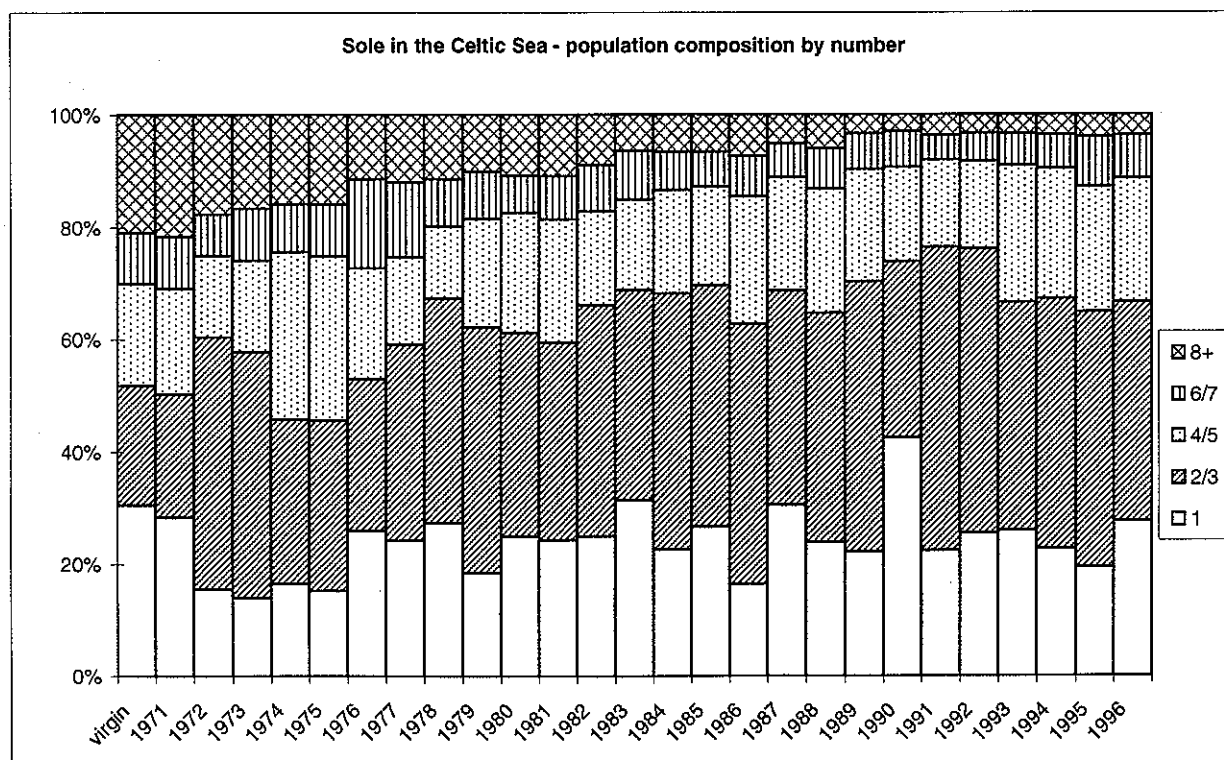
Figure 13.1.3.3.1.3.b. Distribution of sole, as determined by the International Beam Trawl Surveys, in the third quarter of 1996.



**Figure 13.1.3.3.2.1.** Trends in landings and recruitment (top), fishing mortality and spawning stock biomass (bottom) for Celtic Sea sole from 1971–1995.



**Figure 13.1.3.3.2.2.** Sole in the Celtic Sea - population age composition (in percent) according to the number of fish estimated for each age group, from 1971–1996. The key indicates age group in years.



## Changes in spatial distribution

This region is close to the Irish Sea, and many of the comments above, in Section 13.1.3.3.1, also apply to the Celtic Sea sole stock. For those parts that have been surveyed, which represent the main centres of abundance of sole in the Celtic Sea, the population is largely within the Bristol Channel, where it occupies sheltered substrates in the bays of the area (see Figures 13.1.3.3.1.3.a and 13.1.3.3.1.3.b). An analysis of the spatial distribution of sole age groups in the Bristol Channel by Symonds and Rogers (1995) showed that juveniles (0–2 group fish) remained generally within the shallow 0–20 m depth contours, but recruit fish became more widely dispersed in the Outer Bristol Channel and Celtic Sea. This species has particular requirements for nursery, feeding and spawning grounds, which are governed by local hydrography and substrate types.

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- Symonds, D.J. and Rogers, S.I. 1995. The influence of spawning and nursery grounds on the distribution of sole *Solea solea* L. in the Irish Sea, Bristol Channel and other areas. *Journal of Experimental Marine Biology and Ecology*, 190: 243–261.
- 13.1.4 OSPAR Region IV: Bay of Biscay and Iberian Coast**
- 13.1.4.1 Context - Description of the fisheries**
- 13.1.4.1.1 Fisheries in the northern parts of the Bay of Biscay (Divisions VIIIa,b,d,e)**
- Most of the demersal fisheries in this area have a mixed catch of species. Although it is possible to associate specific target species with particular fleets, various quantities of cod, whiting, hake, anglerfish, megrim, sole, plaice and *Nephrops* are taken together, depending on gear type.
- In the Bay of Biscay, Celtic Sea and Western Channel, fisheries for demersal species, mainly cod, whiting, sole, plaice, hake, and *Nephrops*, are conducted by Spain, Belgium, France, Ireland, and the UK. The principal gears used are otter trawls and beam trawls. The targeting of sole and plaice using beam trawls became prevalent during the mid-1970s, leading to an increase in the landings of these two species. The gradual replacement of otter trawls by beam trawls has occurred in the Belgian and UK fleets. In the Bay of Biscay, there has been a substantial increase in the coastal gillnet fishery targeting sole.
- A trawl fishery for anglerfish by Spanish and French vessels developed in the Celtic Sea and Bay of Biscay in the 1970s and expanded until 1990. The fishery has become dependent on small juvenile fish for which there is no minimum landing size. In addition, a gillnet fishery has developed in the Celtic Sea during the last decade.
- Nephrops* are an important component of the fisheries in this area. These fisheries developed in the 1970s and 1980s and effort increased continuously until recent years. Landings increased initially as effort increased, but they have tended to stabilize or decline at continuing high effort levels. The mesh size when fishing for *Nephrops* can lead to a significant by-catch of juvenile fish, notably hake.
- The assessment units used for demersal stocks in this area are small and catches deriving from them are generally in the region of 10,000 t or less. However, the TACs set for the stocks often cover many assessment units. In addition, for a number of units, there are insufficient data for adequate assessments. This means



that TACs which cover a number of heavily exploited stocks comprise a summation across units of analytical forecasts and average catches which offer no effective management control of the exploitation rate. Since a number of stocks affected by this problem are regarded as being close to or outside safe biological limits, there is a need to reconsider the areas on which TACs are set if management is to improve. In 1997, the assessment areas for cod and whiting have been expanded to include Division VIIj,k.

A notable feature of the demersal fisheries in this area is their mixed nature. The use of measures to reduce fishing mortality directly, such as effort reductions in fleets, is likely to avoid a number of the disadvantages of catch controls in regulating the exploitation rate.

The fisheries in the Celtic Sea are very similar to the fisheries in the Bay of Biscay and some of the same fleets operate in both areas. However, the technical measures in the two areas differ. The minimum mesh sizes in the Celtic Sea are often different from those in the Bay of Biscay. This difference makes enforcement more difficult since vessels can carry multiple mesh sizes and may fish in the Celtic Sea using the lower mesh sizes without being detected. It should be noted, however, that the recent European Commission proposal to revise the existing conditions on technical measures attempts to eliminate this problem.

Two stocks of anchovy are considered in the Iberian region, one in Sub-area VIII and one in Division IXa. The Spanish and French fleets fishing for anchovy in Sub-area VIII are well separated geographically and in time (the Spanish fleet operates in Divisions VIIIb and VIIIc in spring and the French fleets in Division VIIIa in summer and autumn and in Division VIIIb in winter and summer). Changes in the catch-at-age composition between the 1984–1996 period and the earlier years could be related to a higher dependence of catches on recruitment in recent years and a change in the seasonality in this fishery. The number of Spanish purse seiners for anchovy has remained stable since 1990 and a slight increase in the number of French purse seiners has been observed in the past five years. A sharp increase in fishing effort for anchovy in the Bay of Biscay has occurred since 1987, mainly due to the increased effort in the French pelagic trawl fleet.

#### **13.1.4.1.2 Fisheries in the Iberian Region (Divisions VIIIc and IXa)**

The Iberian region along the eastern Atlantic shelf is an upwelling area with high productivity; this phenomenon takes place during late spring and summer due to the northerly winds and typical oceanographic system in the area.

The fisheries in the region are of a mixed nature. Different kinds of Spanish and Portuguese fleets operate in the Iberian region: one is the trawl fleet (single, pair and crustacean trawlers) fishing for species such as hake, blue whiting, horse mackerel, megrim, anglerfish, mackerel, *Nephrops*, and cephalopods as the main species. Other fleets fishing for different target species are longliners fishing for hake and mackerel, gillnetters targeted on hake, anglerfish and mackerel, and purse seiners which target sardine and anchovy and, secondarily, horse mackerel and mackerel.

Many bottom trawlers are fishing in the southern part of Division IXa (Gulf of Cadiz); these trawlers are smaller than those operating in the northern parts of the Iberian region. The composition of their catches is also different. They are fishing for hake as well as for crustaceans and cephalopods.

The number of trawlers in the Iberian region has been decreasing since the early 1980s, which has resulted in a decreasing trend in the overall effort in the Portuguese and Spanish fleets. The fleets operating with gillnets and longlines have also decreased in terms of the number of boats in recent years. Spanish and Portuguese boats using trawls, longlines or gillnets are currently subjected to a controlled and restricted system of reduced days of entrance into the harbour to land the catch for the market, with the objective of decreasing the total fishing effort.

Traditionally, the anchovy fishery in Division IXa has been located in the Gulf of Cadiz (Sub-division IXa South), except in 1995 when the bulk of the fishery was located to the north of Portugal and to the west of Galicia (Sub-division IXa North) and was very reduced in the Gulf of Cadiz. This was because of the exceptional availability of anchovy in the northern part of Division IXa owing to environmental factors.

The catches of horse mackerel in Divisions VIIIc and IXa have been relatively stable over the past ten years. However, the proportion of landings by different gears has changed, i.e., trawl catches are decreasing, while purse seine catches are increasing.

The fisheries in the Iberian region are managed by a TAC system and technical measures. Common mesh sizes for trawls are 65 mm, except for trawlers directed at blue whiting or horse mackerel which use 40 mm mesh sizes. In the Gulf of Cadiz, the legal trawl mesh size is 40 mm. The technical measures are minimum landing sizes and seasonal closures to protect juvenile hake.

Of the species of interest to OSPAR, in Region IV only sole, hake, and mackerel support directed fisheries.

#### 13.1.4.2 Sole

##### **Context - pattern of numbers, biomass, recruitment, landings, and fishing mortality**

Catches of sole in the Bay of Biscay have increased continuously in the past two decades, from 3000 t in the late 1970s up to 7000 t at the beginning of the 1990s. In recent years, international landings and catches have decreased gradually. The 1996 landings, at 5850 t, are below the level of the previous three years. Since 1984, catches of sole by French small-mesh shrimp trawlers have decreased markedly, and the gillnet and trammel-net fisheries have expanded (ICES, 1998b).

The mean spawning stock biomass has been 13,000 t. The SSB decreased from 15,100 t in 1987 to 13,200 t in 1990, increased again to 16,700 t in 1993, and then decreased to 12,900 t in 1996. A succession of below-average recruitments in 1991–1993 has contributed to this decline in SSB since 1993. Recruitment has fluctuated around 50 million 0-groups over the period, though the 1995 year class is estimated to be more than 40 % above the mean (72 million 0-groups).

Over the period 1979–1996, fishing mortality has steadily increased, reaching its highest level in 1994 (0.54). Mean  $F$  on ages 2–6 is estimated to have been 0.51 in 1995 and 0.47 in 1996. Trends in yield, fishing mortality, spawning stock biomass and recruitment are plotted in Figure 13.1.4.2.1 (from ICES, 1998b).

Landings of sole in the Bay of Biscay have remained at a high level since 1988, even though fishing mortality increased continuously until 1994. This is a consequence of an exploitation pattern which has been improving over the same period and rather stable recruitment. As four of the five most recent year classes are below average, SSB has declined from its peak in 1993. At the current  $F$ , the probability that SSB in 1999 will fall below the lowest level ( $B_{loss}$ ) in the time series for which the assessment is reliable (1984–1996) is less than 0.05 (ICES, 1998b).

##### **Changes in size distribution and/or age composition**

The proportion of age groups 0, 1 and 2 (juvenile fish) and age group 3+ (mature fish) in the French catches has changed over the period 1979–1996. Since 1984, with the evolution of the different métiers in the French fishery, the proportion of young sole (age 3 and under) in the catches has declined.

Figure 13.1.4.2.2 shows the evolution of the proportion of the number at age of sole in this area for the period 1979 to 1996.

#### Changes in spatial distribution

No marked changes in fishing grounds have occurred during the period over which data are reliable.

#### 13.1.4.3 Mackerel

The mackerel caught in Northeast Atlantic waters are treated as a combined stock composed of three components: the North Sea component (Sub-area IV, Division IIIa), the Western component (Divisions VIIa,b,e, Sub-areas VI and VII, and Division IIa) and the Southern component (Divisions VIIIc and IXa), attributed to three major spawning areas (ICES, 1996). The Southern and Western mackerel components are known to undertake large-scale migrations between summer feeding grounds in the North Sea and Norwegian Sea and spawning areas north of Spain and south and west of Ireland (ICES, 1998a).

The schools of this highly migratory species are harvested at various times of the year in different fisheries using different gears, across all OSPAR regions. The bulk of the catches are taken in OSPAR Region III and in OSPAR Region II. The gears used in each fishery are briefly described below.

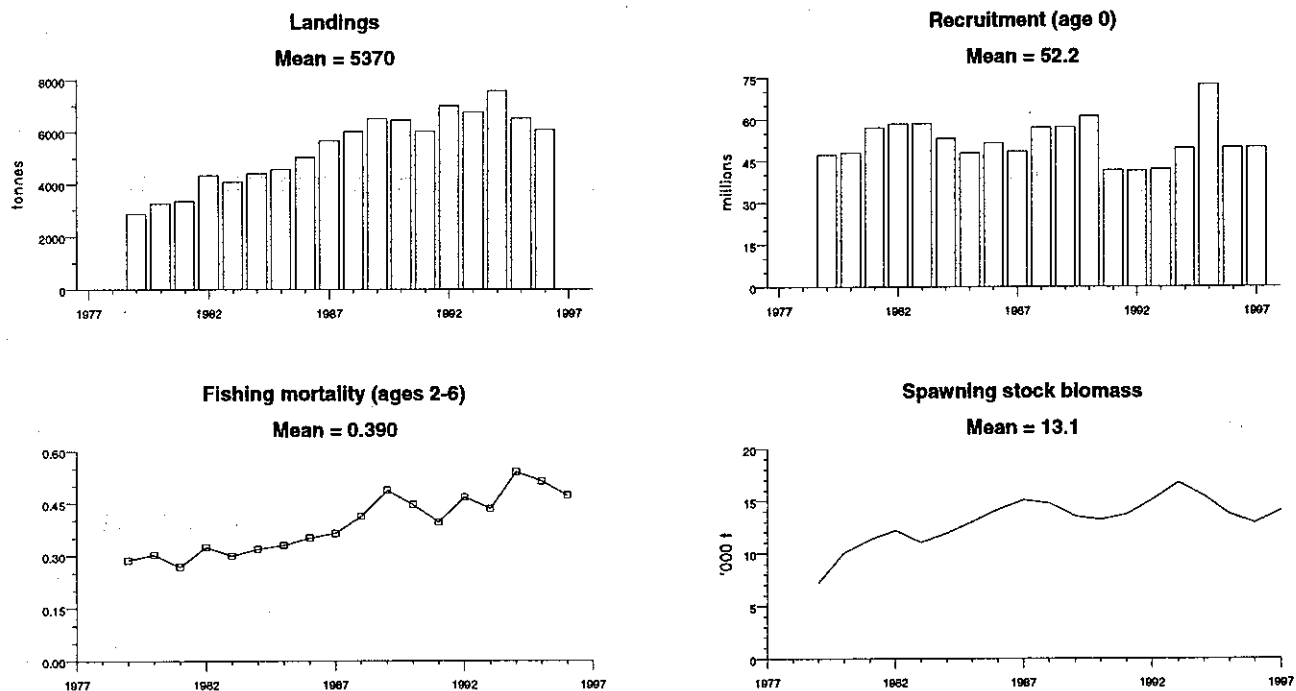
##### **Context - pattern of numbers, biomass, landings, and fishing mortality**

From 1975 to 1996, the total estimated catch of the combined stock has been rather constant around a mean value of 700,000 t (Figure 13.1.4.3.1). In 1996 the total catch showed a reduction of about 200,000 t, compared with 1995. In 1996 fishing mortality decreased, which seems to be due to TAC constraints applied to the combined stock. ICES (1998c) considers the Northeast Atlantic mackerel (i.e., the combined stock) to be within safe biological limits at present.

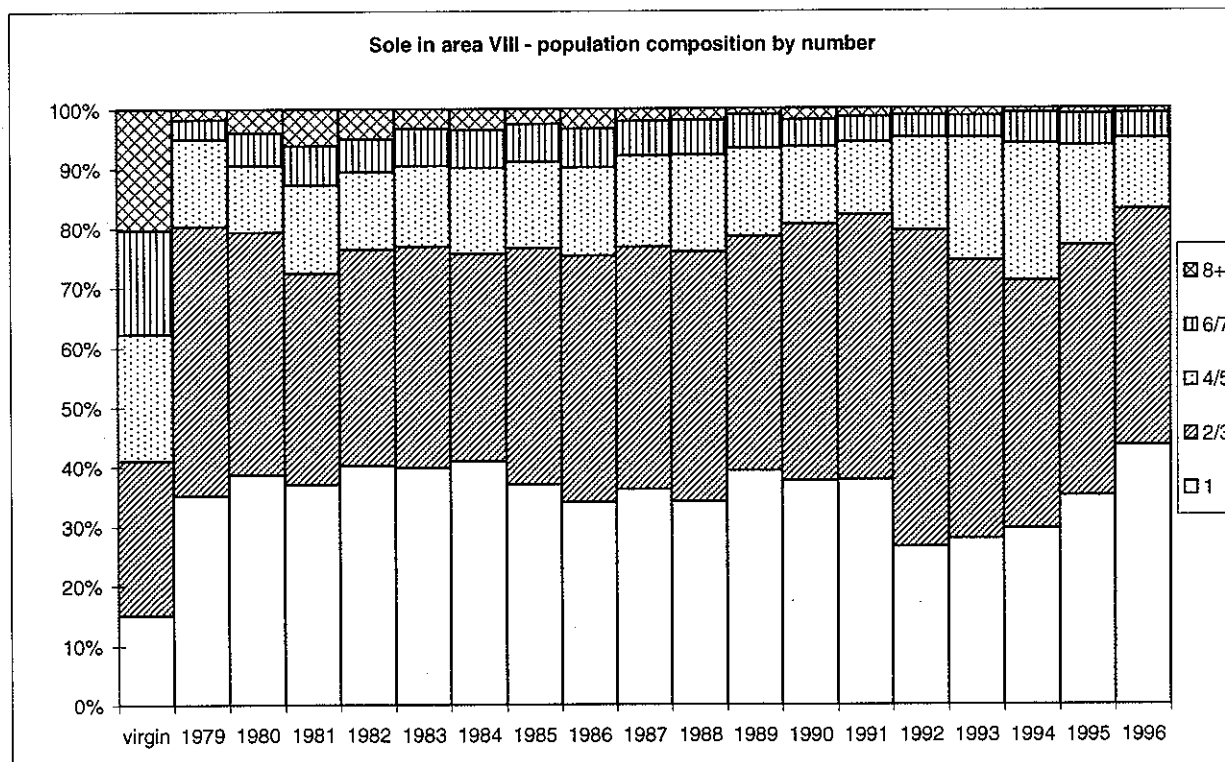
The North Sea mackerel component is treated under OSPAR Region II (Section 13.1.2.4.1, above).

In the Western component, the catches developed from low levels in the 1960s to more than 800,000 t in 1993. The main catches are taken in directed fisheries by purse seiners and midwater trawlers. Large catches were taken in the northern North Sea and the Norwegian Sea from 1989–1995. The SSB declined in the 1970s from 3.5 million t and has remained stable since then at above 2.3 million t. Recruitment (at age 0) was highly variable during the 1970s, and has remained consistently high since that time. There is no separate ICES advice for the Western component of the stock, which dominates with 85 % of the catches of the total combined stock (ICES, 1998a).

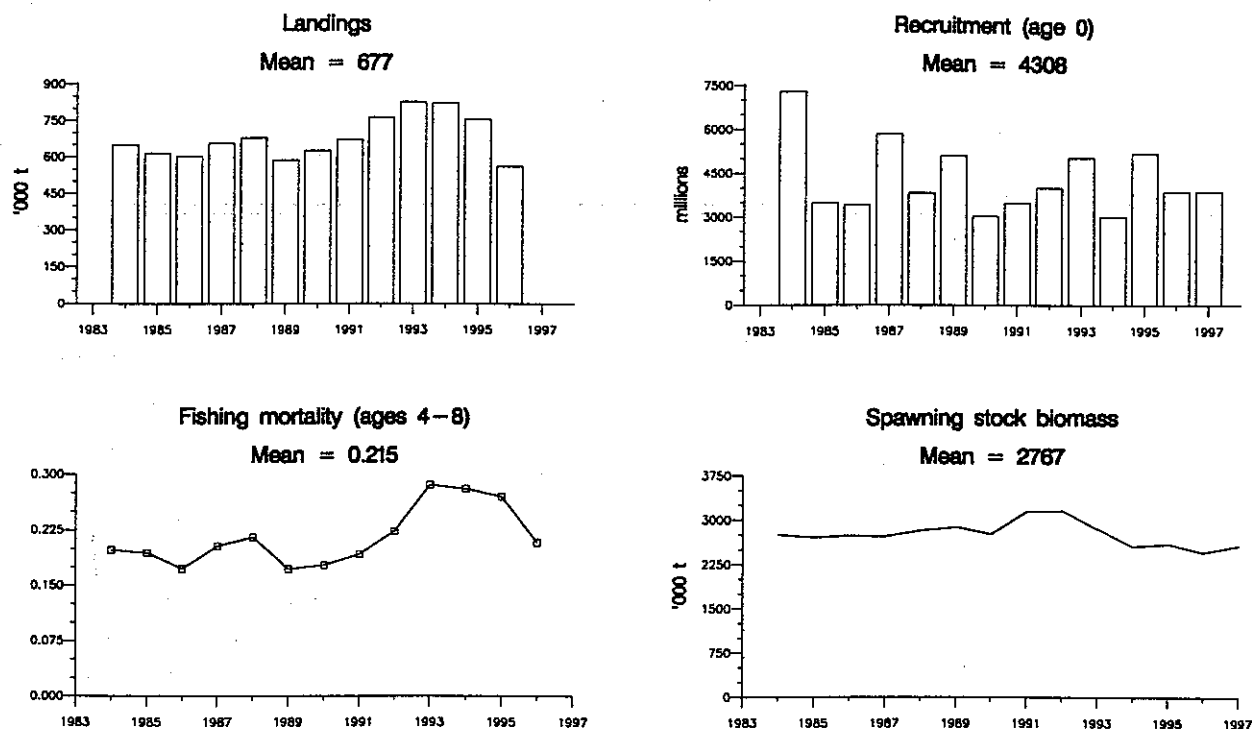
**Figure 13.1.4.2.1.** Trends in landings and recruitment (top), fishing mortality and spawning stock biomass (bottom) for sole in the Bay of Biscay from 1979–1996.



**Figure 13.1.4.2.2.** Sole in the Bay of Biscay (Sub-area VIII) - population age composition (in percent) according to the number of fish estimated for each age group, from 1979–1996. The key indicates age group in years.



**Figure 13.1.4.3.1.** Trends in landings and recruitment (top), fishing mortality and spawning stock biomass (bottom) for Northeast Atlantic mackerel from 1984–1996.

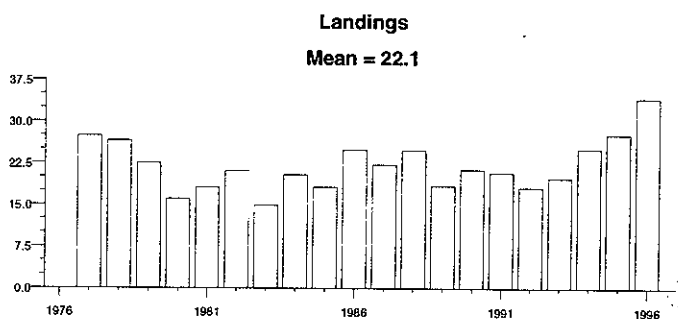


Mackerel in the Southern component is a target species for the Spanish handline fleet during the spawning season in the Bay of Biscay (eastern part of Division VIIIc). This fishery takes about one third of the total catches of this component. In Division IXa, adult mackerel is a by-catch in hake fisheries using gillnets, exploited on the slope of the continental shelf. On the continental shelf of Division IXa, juvenile mackerel is a by-catch of the bottom trawl fishery directed at hake and horse mackerel and of the purse seine fishery directed at sardine and horse mackerel. The SSB of this component was estimated by the 1995 egg survey as 300,000 t (ICES, 1997). The landings have remained relatively stable during the past twenty years (1976 to 1996) at around 22,000 t. Figure 13.1.4.3.2 shows the trends in landings of mackerel in the Southern component (ICES, 1998a).

The largest catches (80 %) from the southern component are taken in the first half of the year, mainly from Division VIIIc, and consist of adult fish (mainly 2–7 years old). In the second half of the year, catches consist of juveniles (0- and 1-group) and are mainly from Division IXa.

**Figure 13.1.4.3.2.** Trends in landings of Southern mackerel from 1977–1996.

#### Southern Mackerel



The assessment and management advice on the Northeast Atlantic mackerel fisheries over the period 1999–2002 may be greatly influenced by the results of the international mackerel egg surveys scheduled to take place in the Southern and Western areas in 1998 and in the North Sea in 1999.

In the Western component, the catches have mainly been composed of two- to seven-year-old fish, which constitute 72 % of the total catches of this component.

### Changes in size distribution and/or age composition

Figure 13.1.4.3.3 shows the age composition in percent of the Western mackerel population, as determined according to the number of individual fish estimated for each age group, for the period 1980–1994.

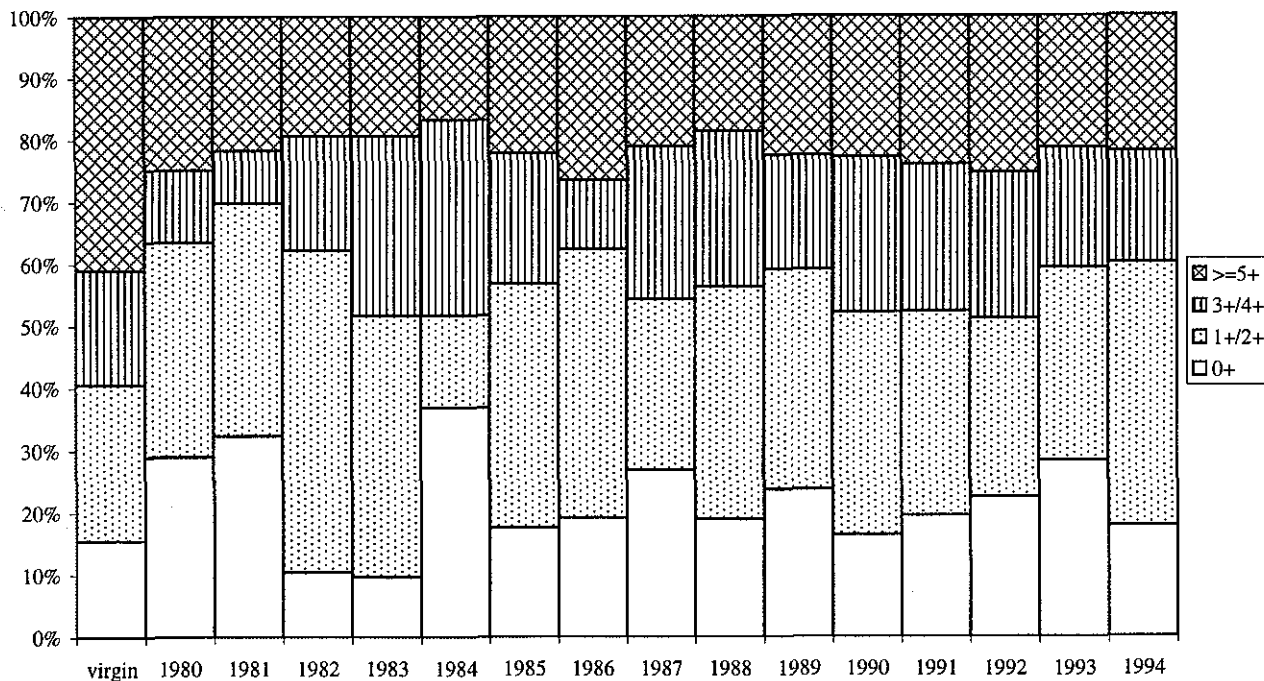
### Changes in spatial distribution

Unlike the North Sea and the Southern mackerel stock components, the Western mackerel component is known to undertake large-scale migrations between summer feeding grounds in the North Sea and the Norwegian Sea and its spawning areas south and west of Ireland.

Previous studies have shown that the timing and pattern of the post-spawning northerly migration has been relatively stable. The return southerly migration, however, has changed dramatically in both timing and route over the past twenty years (Walsh and Martin, 1986; ICES, 1981, 1986, 1988a, 1998b).

During the 1970s and early 1980s, this southerly migration occurred in late summer and early autumn, with the fish moving through relatively shallow waters and giving rise to a very substantial fishery in the Minch (west of Scotland, 57°N–58°N 6°W). Since then, the migration has occurred progressively later in the year, but has stabilized since 1992. The fish now do not cross the 4°W line until mid-January, with the fish being found west of Scotland and Ireland in February. The timing of the migration across the 4°W line is of considerable importance to commercial fishermen, since this longitude separates two management areas, and fishing to the east of it is subject to severe quota restrictions. The later the fish arrive, therefore, the shorter the fishing season for many fishermen. Walsh and Martin (1986) suggested, based on commercial catch data, that this change may have been related to changes in the hydrography of the area following the 1970s salinity anomaly.

**Figure 13.1.4.3.3.** Western mackerel - population age composition (in percent) according to the number of fish estimated for each age group, from 1980–1994. The key indicates age group in years.



Recent work on the migration of mackerel has suggested that water temperature is the major environmental parameter controlling the direction and speed of migration (Walsh and Martin, 1986; Castonguay *et al.*, 1992; Walsh *et al.*, 1995; Reid *et al.*, 1997). The winter migration of Western mackerel from their feeding grounds in the North Sea and the Norwegian Sea to spawning areas south and west of Ireland occurs in the months of December to March. The migration path follows the shelf edge for most of its route, with the fish being found generally between the 100 m and 250 m contours (Walsh *et al.*, 1995). These authors showed that the migration around the north of Scotland appears to follow a track which coincides with a tongue of warmer water transported northwards up the shelf edge by the shelf edge current (SEC). Observations made during an acoustic survey in January 1995 (Reid *et al.*, 1997) indicated that when the migrating mackerel encountered an intrusion of unusually warm water onto the shelf, the fish stopped their active migration and adopted different schooling behaviour. This led to the hypothesis that the spawning migration of this species may be influenced by 'enviroregulation'. This is a process by which the fish select their immediate environments by behavioural means (Neill, 1984). If the fish find themselves in some 'non-preferred' temperature, they may swim faster or deeper in an attempt to gain more preferred temperatures. For example, Olla *et al.* (1975) showed that mackerel swam faster at water temperatures below 7°C. Migration may be triggered when water temperature drops below a threshold, and the subsequent migration route constrained by the narrow tongue of warm water derived from the northward flowing current.

If environmental regulation is constraining the distribution and migration of the mackerel, then it would be expected that immediately prior to the start of migration, the fish would tend to concentrate in the warmest areas of the North Sea, leaving those areas which cool earlier or faster. As these areas of aggregation also cooled, migration would commence. The shelf edge area adjacent to Viking Bank is likely to be one area where the water will stay warmer longer due to Atlantic inflow along the shelf margin.

An exceptionally large number of juvenile mackerel (1996 year class) was observed in the southern North Sea and adjacent areas during 1997, and its spawning component attribution remains unknown (ICES, 1998b). Figures 13.1.4.3.4.a and 13.1.4.3.4.b indicate the distribution of juvenile mackerel as observed in the International Bottom Trawl Surveys covering the entire Northeast Atlantic, carried out during the fourth quarter of the year by each nation. These databases were created, assembled, and processed under the EC FAIR project Shelf Edge Fisheries and Oceanography Studies (SEFOS) from 1993–1996.

With regard to the OSPAR interest in relating changes in distribution to fishing, for the large Western component of mackerel, there has been a major change in migration

pattern and in summer feeding distribution. The changes in migration routing seem fully accounted for by hydrographic factors, without invoking an additional effect due to fishing. However, there are no data to explore hypotheses that the initial invasion of the North Sea and subsequently the Norwegian Sea for feeding in summer was a consequence of the collapse of the North Sea component of the mackerel stock.

#### 13.1.4.4 Hake

Hake is distributed over all the eastern Atlantic coasts from the south of the Iberian Peninsula up to northern Scotland. Two components are distinguished for assessment purposes: the Northern and Southern stocks.

The Northern stock (ICES Division IIIa, Sub-areas IV, V, VI, and VII, and Divisions VIIIa,b) is found in the north of OSPAR Region IV and the south of OSPAR Region III; the distribution of hake does not allow the subdivision of its biological features between OSPAR Regions III and IV.

Sub-area IV and Division IIIa are in OSPAR Region II, but there the catches of hake represent the smallest proportion in relation to catches in the other OSPAR regions where hake are caught.

Consequently, the Northern stock analysis is given here in the context of OSPAR Region IV, according to the main importance of the Bay of Biscay nursery areas in relation to those south of Ireland.

##### 13.1.4.4.1 Northern hake stock (Division IIIa, Sub-areas IV, VI and VII, and Divisions VIIIa, b)

Since the 1930s, hake has been the main species supporting trawl fleets on the Atlantic coasts of France and Spain, and is present in the catches of nearly all fisheries in Sub-areas VII and VIII. Spain and France take 60 % and 25 % of the landings, respectively, and the UK reports about 10 %. After a decline in landings from the mid-1980s, an increase was observed for the first time in 1995, though they decreased again in 1996 to the lowest level observed. Hake are caught throughout the year, the peak landings being made in the spring and summer months. The three main gear types used by vessels fishing for hake as a target species are longlines (England and Wales, Spain), fixed nets (England and Wales, Spain, and France) and otter trawls (all countries). By-catches of mainly juvenile hake are taken in the *Nephrops* fisheries in the northern Bay of Biscay.

#### Context - pattern of numbers, biomass, landings, and fishing mortality

The stock is considered to be close to safe biological limits. SSB increased steadily between 1978 and 1987, coincident with a period of relatively constant

Figure 13.1.4.3.4.a. Distributions of juvenile mackerel (age 0) in the fourth quarter from 1993–1996.

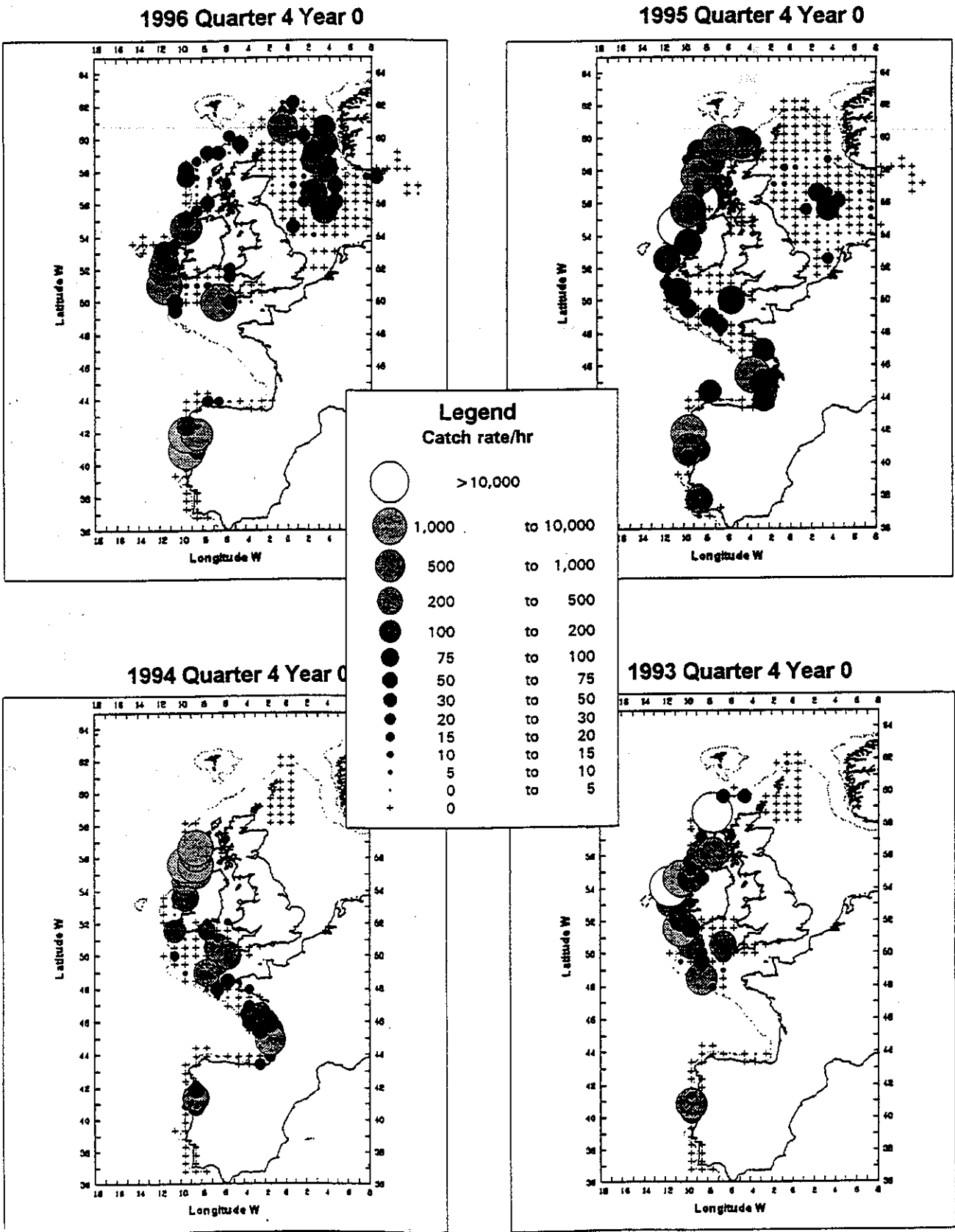
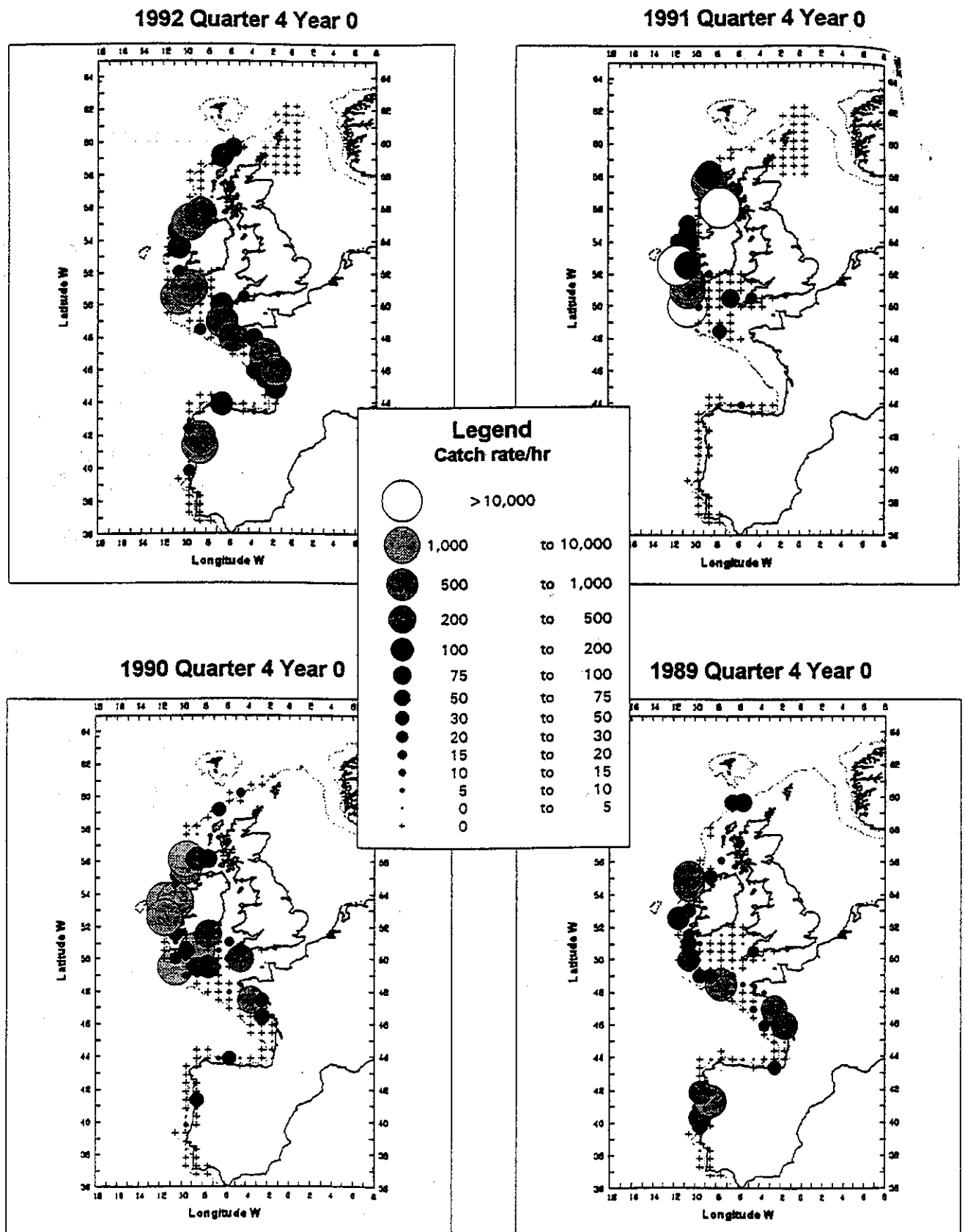


Figure 13.1.4.3.4.b. Distributions of juvenile mackerel (age 0) in the fourth quarter from 1989–1992.





exploitation, but a subsequent increase in fishing mortality was associated with a substantial decline in spawning stock biomass until 1994 (120,000 t, the lowest value in the series). Spawning stock biomass increased slightly in 1995 and 1996. Recruitment has been relatively stable, oscillating around the mean value of 313 million 0-group fish since the highest recorded recruitment in 1985 (500 million). Figure 13.1.4.4.1.1 shows the trends in landings, recruitment, fishing mortality, and spawning stock biomass (ICES, 1998b).

#### **Changes in size distribution and/or age composition**

Hake catches have been mainly composed of 0–4 age groups. Figure 13.1.4.4.1.2 indicates that there is substantial year class variation in this stock, but no long-term trend towards reductions in the proportions of older or younger fish in the stock.

#### **Changes in spatial distribution**

Hake movements are indicated by the seasonal distribution of catches in the fishery. From the beginning of the year until March/April, hake are present in the northern part of the Bay of Biscay. They appear on the shelf edge in the Celtic Sea in June and July. Between August and December, the hake fishery is centred to the west and southwest of Ireland, with a decline in catch rates in shallower waters.

These patterns are well explained by ontogenetic migrations which appear in data from national bottom trawl surveys (ICES, 1994). Hake spawn from February through July along the shelf edge, the main areas extending from north of the Bay of Biscay to the south and west of Ireland. 0-groups descend to the seabed (at depths in excess of 200 m), moving to shallower water with a muddy seabed (75–120 m) by September. There are two major nursery areas: in the Bay of Biscay and off southern Ireland. When they are three years old, hake begin to move into the shallower regions of the Bay of Biscay and the Celtic Sea, but as they approach maturity they disperse to offshore regions.

Analysis of the spatial distribution of hake in the Bay of Biscay (Petitgas *et al.*, 1991) shows that the local and surface waters occupied by adult hake do not change over the years studied, while the densities do change, reflecting variation in year class sizes.

Figures 13.1.4.4.1.3.a and 13.1.4.4.1.3.b show the spatial distribution of the Northern hake stock by age group for the years 1990 and 1992.

#### **13.1.4.4.2 Southern hake stock in the Iberian region (Division VIIIc and Sub-areas IX and X)**

##### **Context - pattern of numbers, biomass, landings, and fishing mortality**

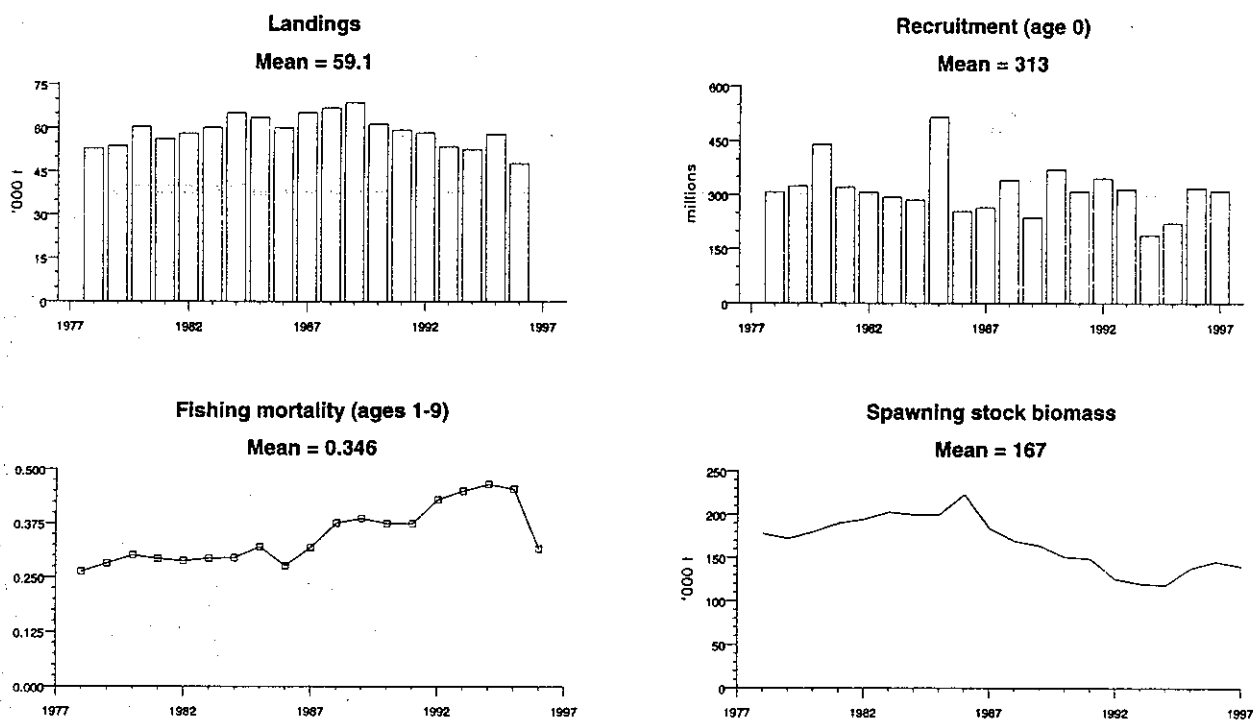
Hake are found on the shelf and the slope in the depth interval 30–700 m around the Iberian Peninsula. The greatest concentration of juveniles (< 17 cm) is found on the shelf between 150 m and 300 m. According to Portuguese surveys, juveniles of hake are mainly concentrated in the closed area of Milfontes-Arrifana, mainly from December to March. Recruitment occurs off the northern coast of Portugal in August. Off the northwest coast of Spain, where the main nursery grounds are located, hake recruit continuously during spring and early summer, and recruitment can be considered finished by September, when small hake around 11 cm are encountered at depths less than 200 m.

In Portuguese waters, spawning takes place on the shelf at depths between 150 m and 300 m. The main spawning season is from December to March, although spawning females can be caught throughout the year. Off the northwest coast of Spain, the spawning season is between December and April, with a peak in February–March. In northern and eastern areas, the spawning time is later, from April to July.

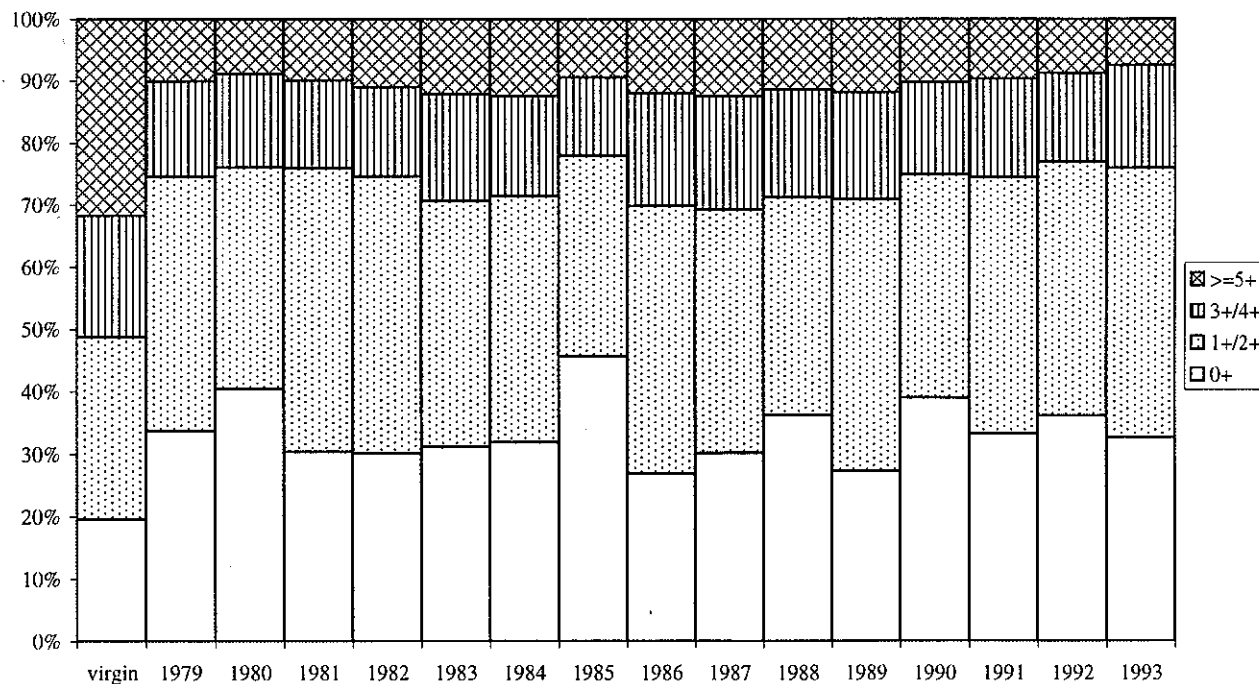
The Southern hake stock is considered to be outside safe biological limits. Landings have declined almost continuously since 1983 (23,000 t), reaching the lowest catch on record in 1996 (10,000 t). Spawning stock biomass decreased very sharply between 1984 and 1986, from 59,000 t to 26,000 t, and the 1996 spawning stock biomass was near to the lowest recorded, which occurred in 1995 (15,000 t). Recruitment has declined steadily since 1984 (around 120 million 0-group) and, with the exception of one year (1992), has been poor since 1989 on average about 80 million 0-group). The assessment indicates that there has been a decreasing trend in fishing mortality since 1986. Figure 13.1.4.4.2.1 shows the trends in landings, recruitment, fishing mortality, and spawning stock biomass (ICES, 1998b).

Fishing mortality in 1996 was at  $F_{med}$  (0.23). There is evidence of reduced recruitment below a spawning stock biomass of 23,000 t. At the current fishing mortality, the probability of SSB in 1999 remaining below the minimum biologically acceptable level (MBAL) is more than 50 % (ICES, 1998b).

**Figure 13.1.4.1.1.** Trends in landings and recruitment (top), fishing mortality and spawning stock biomass (bottom) for the Northern stock of hake from 1978–1996.



**Figure 13.1.4.1.2.** Northern hake - population age composition according to the number of fish estimated for each age group, from 1979–1993. The key indicates age group in years.



**Figure 13.1.4.1.3.a.** Distribution of hake age groups (0–5+) observed in autumn 1990 during the French bottom trawl survey in the Bay of Biscay.

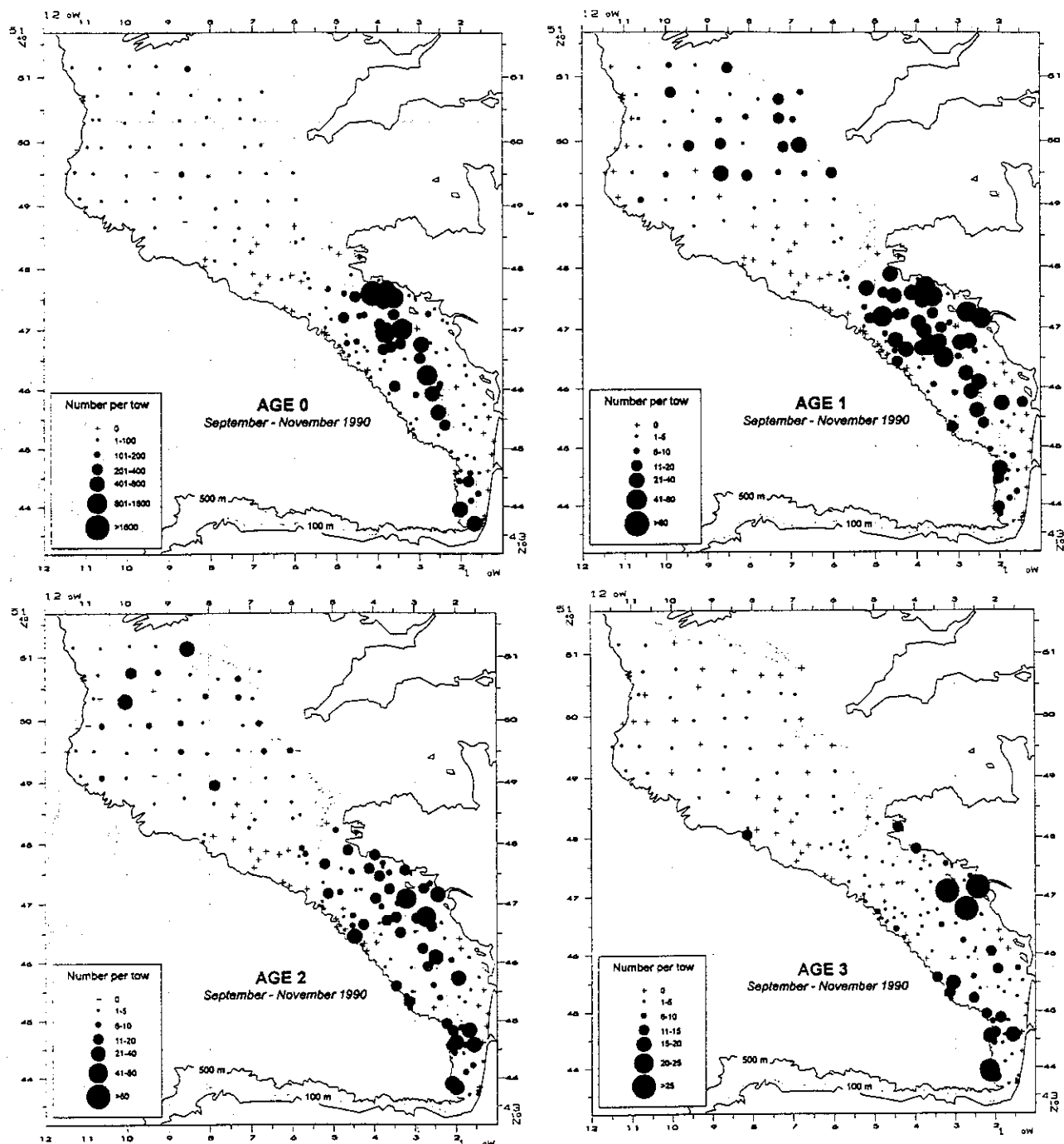


Figure 13.1.4.4.1.3.a. Continued.

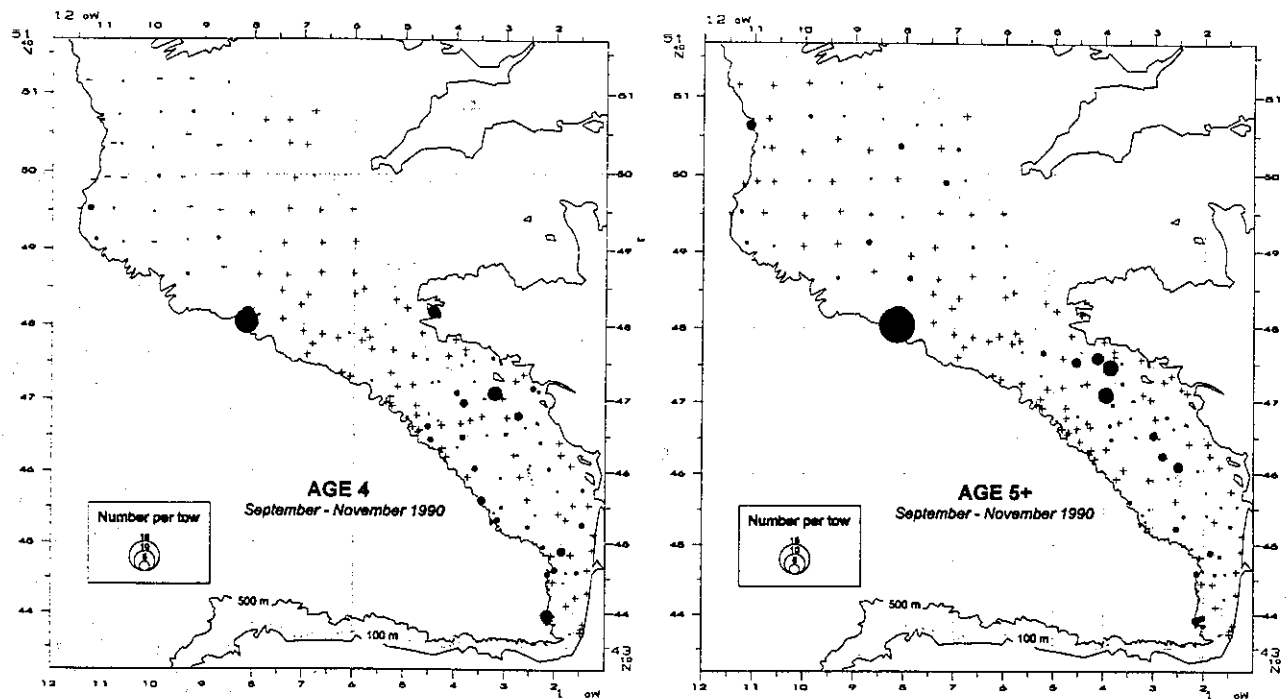


Figure 13.1.4.4.1.3.b. Distribution of hake age groups (0-5+) observed in autumn 1992 during the French bottom trawl survey in the Bay of Biscay.

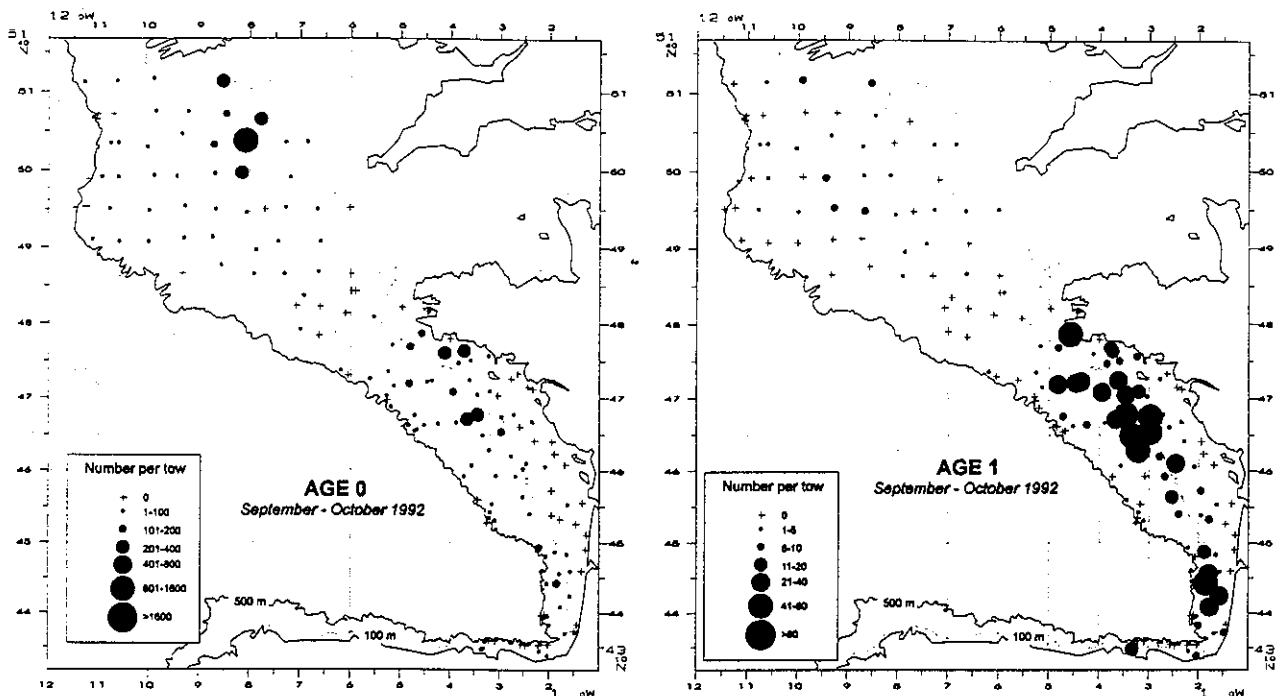
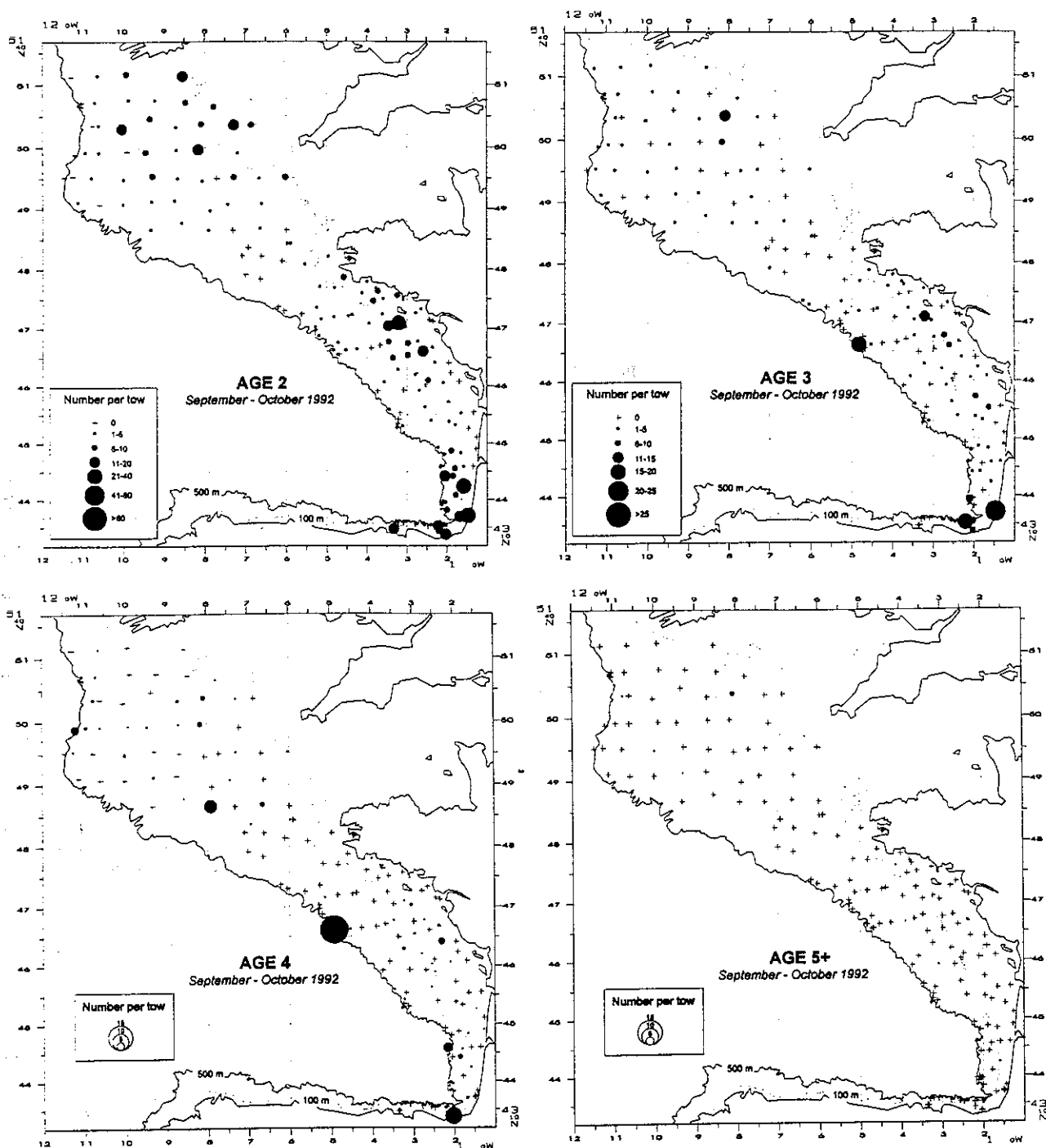
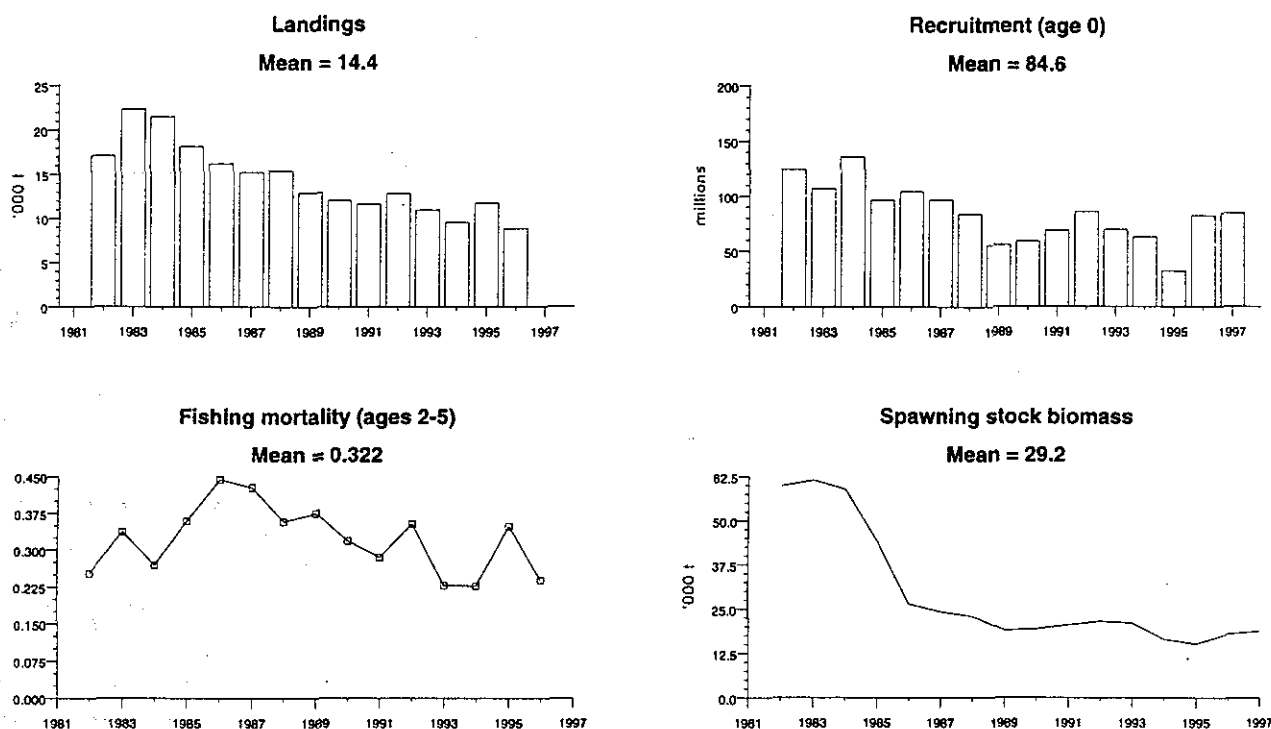


Figure 13.1.4.4.1.3.b. Continued.



**Figure 13.1.4.4.2.1.** Trends in landings and recruitment (top), fishing mortality and spawning stock biomass (bottom) for the Southern stock of hake from 1982–1996.



#### Changes in size distribution and/or age composition

In a time series covering the period 1978–1996, a change in the length composition of the landings appears after 1982. There has been a decrease in the estimated number of fish smaller than 30 cm (ICES, 1998b). However, the change is attributed by ICES (1998b) to a change in fishing practices, following the enforcement in 1989 of a minimum legal landing size (27 cm). This size limit made it more difficult to obtain samples of small fish (ages 0 and 1), thus changing the apparent size composition of the stock. This interpretation is supported by an increase in mean weight in landings during the period. This explanation suggests that the biological feature of population age composition has not actually changed, but that the populations were sampled with different size selectivities in each of the two periods. Figure 13.1.4.4.2.2 depicts the evolution of the proportion of the numbers at age from 1982 to 1996.

Although changes in the size abundance of adult hake are expected for this depleted stock, there is no evidence from the analysis that this is occurring. One possible explanation is that the adult hake are mainly living on the slope in non-trawlable areas, where both the groundfish survey and the commercial trawlers cannot fish.

#### Changes in spatial distribution

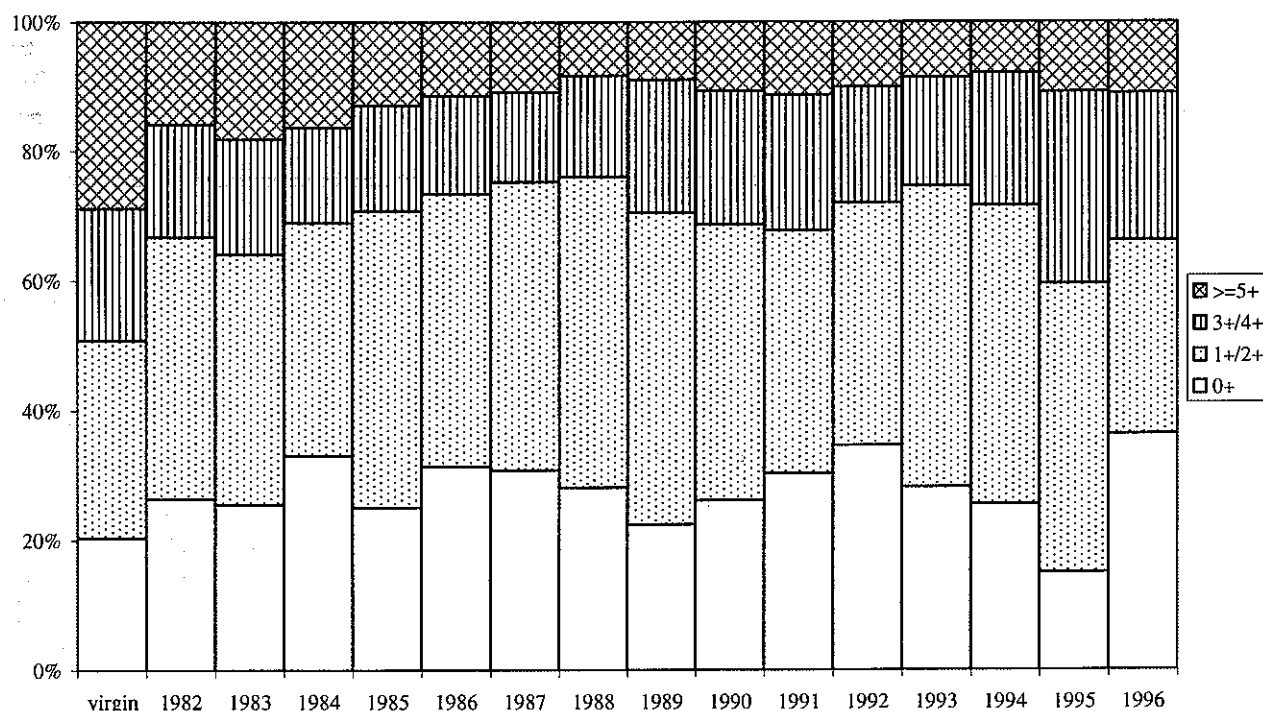
The Iberian region is an important nursery ground for hake. An area closed to fishing has been enforced off the southwest coast of Portugal to prevent trawlers from catching juveniles during the autumn and winter, when recruitment peaks. The closed area has led to short-term increases in hake numbers in the closed area, because they experience lower mortality during the period of closure. However, any additional changes in spatial distribution over time appear to be short-term responses to local environmental conditions.

Figure 13.1.4.4.2.3 depicts the abundance distribution of hake in OSPAR Region IV during the fourth quarter of 1990.

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**Figure 13.1.4.4.2.2.** Hake in the Southern area (Divisions VIIIc and IXa) - population age composition (in percent) according to the number of fish estimated for each age group, from 1982–1996. The key indicates age group in years.



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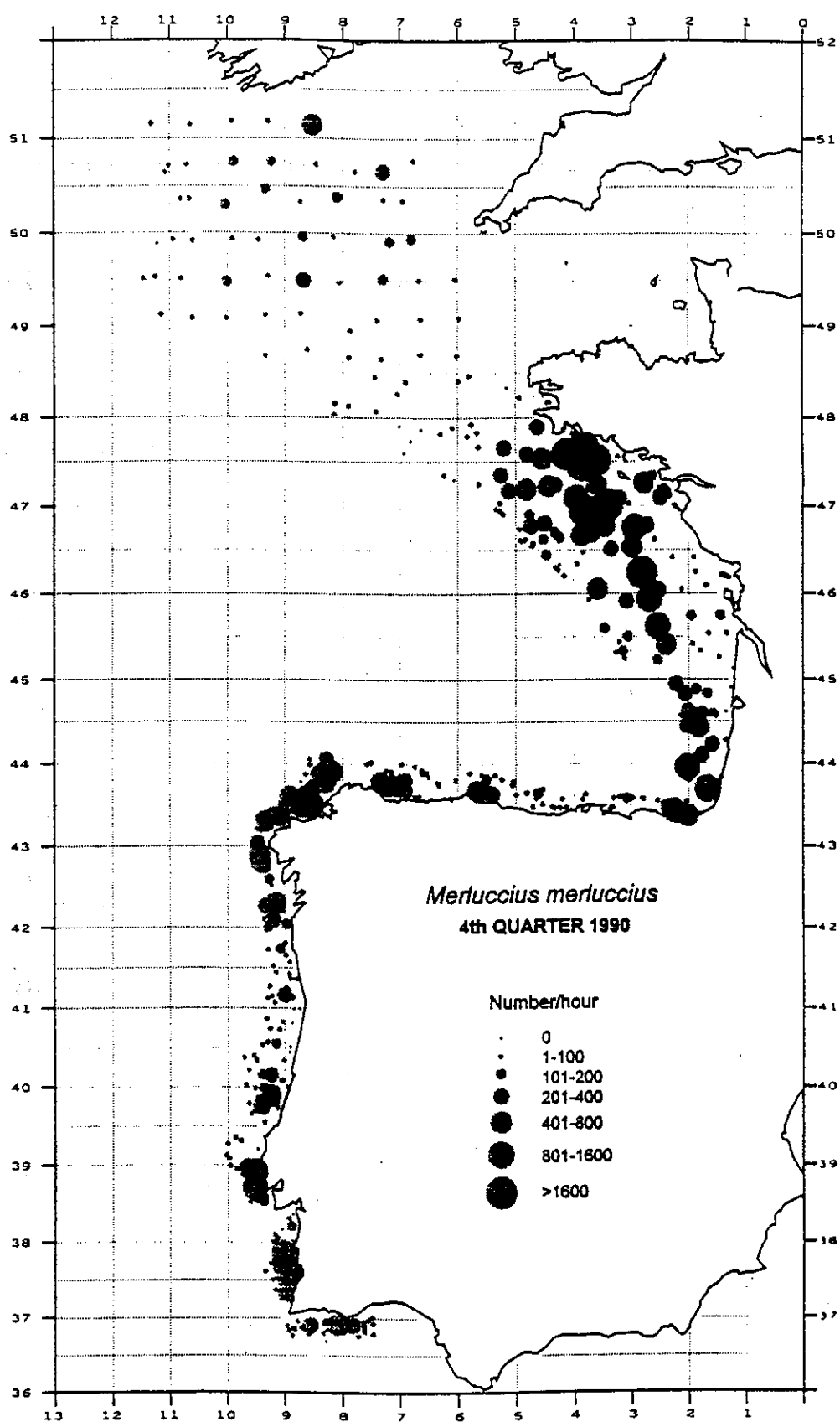
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Figure 13.1.4.4.2.3. Hake distribution observed in the fourth quarter of 1990.





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### 13.1.5 OSPAR Region V: The Wider Atlantic

#### 13.1.5.1 Description of the fisheries

WGECO identified four classes of fisheries in this OSPAR region. First, particularly in the southern and central portions, fisheries have been prosecuted on large pelagic tuna and tuna-like stocks for many years. Advice on these fisheries and stocks is provided by the International Commission on Conservation of Atlantic Tunas (ICCAT), rather than by ICES. ICCAT reports on by-catches in these fisheries are included in Section 13.2, below. Otherwise, the information here does not consider these fisheries. Second, there are several longline and trawl fisheries in OSPAR Regions I, III, and IV which are active primarily in deep waters on the continental slopes, targeting species such as ling (*Molva molva*), tusk (*Brosme brosme*), argentine (*Argentina sphyraena*), grenadiers (*Macrourus berglax* and *Coryphaenoides rupestris*), and various species of deep-water sharks. In some years, these fisheries may extend into OSPAR Region V. However, to keep the features of the major fisheries together, catches, by-catches, and discards in these fisheries on the continental slopes are included in the information reported for the appropriate coastal OSPAR region, where these fisheries conduct the large majority of their harvesting.

The third group of fisheries are directed fisheries prosecuted within this OSPAR region. Most of OSPAR Region V comprises distant and deep waters. Fisheries for demersal and pelagic stocks (other than tunas and related species) have generally developed recently, and many of them are expanding rapidly. They target species whose biology is generally poorly known. However, some knowledge is becoming available as studies expand in this area (Gordon and Duncan 1985, 1987; Gordon *et al.*, 1996). The limited knowledge available suggests that these stocks can sustain only low exploitation rates, due to their longevity, late age of maturity, and apparently low fecundity (ICES, 1995, 1996). In the northern portion of OSPAR Region V, the primary fishery has been trawling for redfish (*Sebastes marinus* and *Sebastes mentella*), a large resource which has attracted rapidly increasing effort. There are indications that *S. marinus* includes a genetically distinct component, 'giant' *S. marinus*, and *S. mentella* is considered to consist of at least a deep-sea and an oceanic stock. At least thirteen

fleets have joined this fishery, but the main fleets are from Russia, Germany, Iceland and Norway. Redfish catches in Region V peaked in 1994 and 1995, at 94,000 t and 127,000 t, respectively. These increases have come through continued expansion of the areas and depths fished. New trawl fisheries are developing along the mid-Atlantic Ridge for golden-eye perch (*Beryx splendens*), orange roughy (*Hoplostethus atlanticus*), black scabbard fish (*Aphanopus carbo*), and wreckfish (*Polyprion americanus*). Catch per unit effort (CPUE) is not thought to be an informative index of stock status for these fisheries. However, annual increases in the depths fished and decreases in the size of redfish taken in areas fished over several years support the concern of ICES that fisheries on these stocks have expanded too rapidly. Attempts to extend these fisheries to other deep-water stocks of fish and sharks confront the same problems: there is little biological information available on the stocks being targeted, but general life history features suggest that only very low exploitation rates could be sustained and very long recovery times are required after over-exploitation has occurred. Owing to the lack of information on by-catches and discards in these deep-water fisheries, it is not known whether present levels of by-catch mortality are below the total sustainable level.

The fourth group of fisheries are traditional longline, handline and gillnet fisheries around the Azores and adjacent seamounts. These target a variety of species, featuring red sea-bream (*Pagellus bogaraveo*), wreckfish, conger eel (*Conger conger*), greater forkbeard (*Phycis blennoides*), bluemouth (*Helicolenus dactylopterus*), golden-eye perch, alfonso (*Beryx decadactylus*), kitefin shark (*Dalatias licha*), and gulper shark (*Centropomus granulosus*). The fishery is prosecuted by vessels up to 30 m in length, with total landings under 5000 t in recent years. Recently there has been interest in expanding these fisheries to other species, and some exploratory surveys have been conducted in waters adjacent to those traditionally fished.

#### 13.1.5.2 Trends in the five selected species

There are no fisheries for cod, herring, sole, hake or mackerel in Region V. Occasional small catches of cod, hake, and mackerel taken on the eastern boundaries of ICES Divisions VIb and VIIC,k are included in the descriptions relevant to OSPAR Regions III and IV (see Sections 13.1.3 and 13.1.4, above).

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## 13.2 Discards in Fisheries in the Northeast Atlantic

### *Request*

Item 4.3 of the 1998 Work Programme from the OSPAR Commission: to provide information on quantities of discards by gear type for commercially exploited stocks of fish and shellfish in the maritime area subject to regular assessment.

### *Source of the information presented*

The 1997 report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO), based on material from the 1997 reports of the following Working Groups: Arctic Fisheries Working Group, Northern Pelagic and Blue Whiting Fisheries Working Group, Herring Assessment Working Group for the Area South of 62 °N, Northwestern Working Group, Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak, Working Group on the Assessment of Mackerel, Horse Mackerel, Sardine and Anchovy, Working Group on the Assessment of Northern Shelf Demersal Stocks, Working Group on the Assessment of Southern Shelf Demersal Stocks, Working Group on North Atlantic Salmon, *Pandalus* Working Group, Working Group on *Nephrops* Stocks, Study Group on the Biology and Assessment of Deep-Sea Fisheries Resources, and Study Group on the Assessment of Other Fish and Shellfish Species; ACFM and ACME deliberations.

### *Status/background information*

Discards appear in most fisheries. Fish are discarded because they are either of no economic interest or illegal to land. Discards due to economic interests usually consist of non-commercial or low-value commercial fish or fish sizes. Discards due to legal restrictions usually consist of undersized or over-quota fish, but at times fish are discarded due to an economic optimization of the vessel quota. Discarding due to quotas and minimum landing size is originally for protection purposes, but discarding may contribute to an increased mortality for the fish discarded. The significance of the discard mortality is under investigation.

Discards have been recorded since the 1920s, but most of the existing data are not quantitative. Several national and international sampling schemes have recently been or are now being conducted. New sampling schemes are planned to include several countries around the North Sea.

The information on discards contained in this section has been compiled on the basis of reports and information that have been available to the relevant ICES Assessment Working Groups and WGECO. WGECO is aware that this information is not complete both in terms of

information contained in reports that are not available to ICES and in reports that have not been made publicly available. In addition, some data were not reported because they were still preliminary at the time this work was carried out. The European Commission has funded a number of programmes to study discards, but the results of these studies will not be available for several years (e.g., three to five years for programmes just beginning). When the results of these studies have been reported, there will be a great deal of new information available, such that it is not worthwhile to spend a large effort to try to compile further information at the present time.

For each region, the material presented here must be taken together with the corresponding contextual information contained in Section 13.1, above.

### 13.2.1 Discards in OSPAR Region I: Arctic Waters

#### 13.2.1.1 Information on discards by gear type

The Assessment Working Groups have provided very little information on discards for OSPAR Region I, either because the problem is expected to be small in some fisheries or because very little data were available. Table 13.2.1.1.1 summarizes some of the main fisheries by ICES Sub-area/Division and the discard information provided.

In addition to the fisheries included in Table 13.2.1.1.1, there is also some known discarding of by-catch species in the deep-water fisheries of this region.

#### 13.2.1.2 Commentary on quality of data and collection programmes

WGECO was not aware of any programmes for collecting information on discards in this region. Some participants in Stock Assessment Working Groups for the region, however, believe that there is some discarding in these areas, based on observations. Also, from the historical data provided for tuning assessments, it may be concluded that unaccounted mortality probably has been large in some periods (ICES, 1998). In the 1980s, the Institute of Marine Research in Bergen, Norway, wanted to hire a commercial trawler for fishing together with the fleet, but the request was unsuccessful. The suggestion may be proposed again due to problems with the assessment of Northeast Arctic cod.

There also exist earlier studies on by-catches and discards for a few of the fisheries in the region. Høyen and Jacobsen (1987) estimated the by-catch of cod in the Norwegian shrimp fishery north of 69 °N (Sub-areas I and II). The estimates were based on commercial landing statistics, data from the surveillance of the shrimp fishery, and cod and shrimp research surveys. They found that the number of 1- to 3-year-old cod taken as

**Table 13.2.1.1.1.** Information on discards by fishery/species, gear and ICES Sub-area/Division in OSPAR Region I provided by ICES Assessment Working Groups (NA = no data available).

Fishery/ species	Gear	Species discarded	Sub-area/Division			
			I+IIa+IIIb	Va	Vb	XIVb
Atlantic salmon	Longline	Atlantic salmon	No fishery	No fishery	1.8–15.6 % of catch in 1982–1991 <sup>1</sup>	
Capelin	Purse seine trawl		Not supposed to be a problem	Not supposed to be a problem	No fishery	No fishery
Herring	Purse seine trawl		Not supposed to be a problem	Not supposed to be a problem	No fishery	No fishery
Cod, redfish, haddock	Trawl, longline, gillnet		NA	NA	NA	NA
Shrimp	Trawl	Cod	NA	NA	NA	32 t or 100,000 ind. in 1984 <sup>2</sup>
		Others	NA	NA	NA	NA
<i>Nephrops</i>	Trawl	<i>Nephrops</i>		31.5 % of catch in 1996 <sup>3</sup>		
		Others		NA		

<sup>1</sup>ICES, 1993.

<sup>2</sup>ICES, 1997.

<sup>3</sup>Unpublished Icelandic data.

by-catch in the shrimp fishery was relatively high compared to the number taken by the human consumption fishery (on average about 800 and 10 times higher for cod aged one and two, respectively). Because only fish of commercial size are landed from the shrimp fishery, most of the by-catch of these age groups was discarded. The long-term loss in yield was estimated at 20,000 t and 30,000 t for the 1982 (average strength) and 1983 (strong) year classes, respectively. In order to limit the by-catch of cod and haddock, areas have been closed in periods when the by-catch of undersized fish exceeded ten specimens (of cod and haddock) per 10 kg of shrimp. The introduction of a sorting grid in 1993 has further reduced the by-catch problem in the Norwegian shrimp fishery.

McBride and Fotland (1996) made estimates of unreported catch of cod in the Norwegian commercial trawl fishery in the Barents Sea (Sub-area I) in 1989. They used information from the catch statistics, length samples from the landings, length samples from bottom-trawl surveys, and data on codend selectivity for Norwegian bottom trawlers. Of the estimated total numbers of fish caught in the area, 6.9 % was found to be discarded or not reported because they were below the minimum market size. These authors concluded that their estimates were conservative relative to the peak discard rates estimated for 1953–1954 (40 % by number and 20 % by weight) by Garrod (1967).

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## 13.2.2 Discards in OSPAR Region II: Greater North Sea

A project partially funded by the EU was started in 1995 to estimate discards for all Danish fisheries including

those in the North Sea. The project will run for three years. The EU has also partially-funded a two-year project starting in June 1997 to place observers on board Scottish and Norwegian purse seiners. These projects cover discards in the herring, mackerel, and horse mackerel fisheries.

### 13.2.2.1 Herring

#### North Sea and Eastern Channel

As outlined in Section 13.1.2.2, above, herring are caught by purse seines and midwater trawls in the North Sea. Only the Netherlands has provided information on discard amounts in their herring fishery, which only forms a small part of the overall North Sea fishery (Table 13.2.2.1.1). Most of the Netherlands fishery uses pelagic trawls, so there is at present no information on discard rates from purse seines, the other main gear used in this fishery. In addition, no information is available on the practice of 'slippage'. 'Slippage' occurs when the purse seine is closed and pulled tight, but the fish are released from the net and not brought on board, usually because the fish are not of the desired size. Fish released in this way are usually dead or moribund.

#### Skagerrak/Kattegat

No estimates of discards were available for Division IIIa. Preliminary data from the Danish study described above indicate that there is very little discarding in the

Kattegat. However, discarding may be high in the Skagerrak, especially during summer when there is a demand for high quality herring for the Dutch market.

### 13.2.2.2 Mackerel

Mackerel are caught by purse seines and pelagic trawls in the North Sea. Only the Netherlands has provided information on discard amounts from their fishery, which is mostly by pelagic trawls (Table 13.2.2.2.1). There is therefore no information on discard rates from purse seines or on the practice of 'slipping' (see Section 13.2.2.1, above). Discarding to land the highest grade of fish (high-grading) and slipping has been a problem in the past when large year classes arrive in the fishery. This may become a particular problem again when the comparatively strong 1996 year class arrives in the fishery.

### 13.2.2.3 Horse mackerel

Horse mackerel are caught by purse seines and pelagic trawls in the North Sea. Only the Netherlands has provided information on discard amounts from their fishery, which is prosecuted mostly by pelagic trawls. There is therefore no information on discard rates from purse seines or on the practice of 'slipping' (see Section 13.2.2.1, above). The part of the western horse mackerel stock that is fished in the Western Channel (Division VIIe) could not be disaggregated from the fisheries in other waters of the region, or to the west of the Channel.

**Table 13.2.2.1.1.** Herring discards (tonnes) from the Netherlands fishery, total catches (tonnes) according to Divisions of the North Sea and Eastern Channel, and proportions of this total landed by the Netherlands. Discard amounts were provided only by the Netherlands (from ICES, 1997a).

Area		1990	1991	1992	1993	1994	1995	1996 <sup>1</sup>
IVa West	NL discards	750	883	850	825	550	0	356
	Total catch	141,780	152,767	157,265	128,662	177,877	196,365	99,866
	NL %	21 %	19 %	19 %	22 %	9 %	13 %	3 %
IVa East	NL discards	0	0	0	0	0	0	0
	Total catch	174,747	126,627	115,775	100,154	85,469	109,562	38,115
	NL %	0	0	0	0	0	0	0
IVb	NL discards	2,560	1,072	1,900	245	460	0	592
	Total catch	175,474	225,448	202,229	210,473	131,008	165,455	77,916
	NL %	16 %	12 %	13 %	15 %	30 %	18 %	24 %
IVc & VIId	NL discards	5,350	2,662	2,200	2400	2400	0	521
	Total catch	61,082	60,685	73,981	84,878	74,078	62,905	49,565
	NL %	19 %	32 %	25 %	23 %	27 %	38 %	28 %
Total	NL discards	8,660	4,617	4,950	3,470	2,510	0	1,469
	Total catch	553,082	565,527	549,249	524,020	467,534	534,281	264,868
	NL %	13 %	13 %	14 %	15 %	16 %	15 %	13 %

<sup>1</sup> Provisional data.

**Table 13.2.2.2.1.** Mackerel discards (tonnes) from the Netherlands fishery, total catches (tonnes) in the North Sea including the Skagerrak/Kattegat, and the proportion of this total landed by the Netherlands. Discard amounts were provided only by the Netherlands (from ICES, 1998a).

	1990	1991	1992	1993	1994	1995	1996
NL discards	4,300	7,200	2,980	2,720	1,150	730	1,387
Total catch	305,100	365,900	367,164	390,558	475,980	323,400	212,838
NL %	4.5 %	1.3 %	1.8 %	2.0 %	0.8 %	0.4 %	0.9 %

Table 13.2.2.3.1 shows North Sea horse mackerel discards from the Netherlands fishery, total catches in the North Sea including the Skagerrak/Kattegat, and the proportion of this total landed from the areas fished (Divisions IVb,c) by the Netherlands. No breakdown by nationality was available for Divisions IVb and IVc. Discard amounts were provided only by the Netherlands (from ICES, 1998a). Landings from the northern North Sea (Division IVa and some Norwegian IVb) of the western horse mackerel management stock are included for information, but no discard data were available.

#### 13.2.2.4 Demersal stocks

Information on demersal stocks and discard information for some fisheries targeting these stocks has been assembled by the ICES Working Group on the Assessment of Southern Shelf Demersal Stocks (ICES, 1998b) and the Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak (ICES, 1998c). Within the demersal fisheries, there are a number of gears catching a variety of species and, conversely, species are caught by a variety of gears. Ideally, it would be very useful to report discards either by gear type for all species or by species for all gear types. Unfortunately,

this is not possible, and information is available only for some national fisheries and very often only for the main target species of that fishery. Discards of non-target species, where they occur, may not be reported.

#### ICES Division VIIe

Table 13.2.2.4.1 compiles information from ICES (1998b) for Division VIIe (Western Channel). The discards in this area have been calculated by raising the sampled catches in these fisheries by the proportion discarded within nearby Irish fisheries (Connolly and Wheatley, 1997). This treatment assumes that the fishing practices and the mix of species and sizes of fish are similar between the Irish fisheries and fisheries in the Western Channel.

In the UK otter trawl fishery, the main species discarded are dab, gurnard (*Eutrigla gurnardus*) (more than half the total is these two species), lesser spotted dogfish (*Scyliorhinus canicula*) and whiting. The discards are very variable seasonally, with more than half of the dab discards occurring in the first quarter (Q1), but when seasonality of landing is considered, the ratio of by-catch to landings is particularly bad for dab in the last quarter (Q4).

**Table 13.2.2.3.1.** Horse mackerel discards (tonnes) from the Netherlands fishery in the North Sea, total catches (tonnes) in the North Sea including the Skagerrak/Kattegat, and the proportion of this total landed from the area fished by the Netherlands.

	1990	1991	1992	1993	1994	1995	1996
NL discards			400	930	630	30	212
Total catch	146,387	78,594	118,769	148,445	114,402	107,381	36,987
% taken in NL fisheries area	11.9 %	14.5 %	11.7 %	2.6 %	2.2 %	7.4 %	20.4 %
Western stock landings from North Sea	112,753	63,869	101,752	134,908	106,911	90,527	18,356

**Table 13.2.2.4.1.** Summary of information on discards (tonnes) in UK fleets in ICES Division VIIe (ICES, 1998b).

Country/gear type	Main target species	Landings (of main target species*)	Discards of main target species	Total discards (all species)
UK otter trawl (1995)	plaice, lemon sole, and whiting	1271	179	1314
UK pair trawl (second and third quarters, 1995)	whiting	117	31.3	169
UK beam trawl (1995)	plaice	588	9	1402
UK fixed nets (1992/1993)	hake	30	0.4	16
UK fixed nets (1992/1993)	cod	93	0.2	0.2
UK fixed nets (1992/1993)	turbot/anglerfish	100	6	29

\*Note: It was unclear from the information reported whether the landing figures in tonnes related only to the main target species or to total landings.

There was no sampling in the first and fourth quarters of 1995 in the UK demersal pair trawl fleet fishing for whiting. In the other quarters of that year, about 117 t of fish were landed (i.e., brought to land for sale) and 169 t discarded at sea. This was equivalent to approximately one half the year's landings. There were also very large differences in the amounts discarded between the two quarters recorded (68 % of the total catch in the second quarter, 29 % in the third). The main species discarded were dab, lesser spotted dogfish, whiting, horse mackerel and gurnard.

The UK beam trawl fleet fishing for plaice in 1995 had a very high level of discards in the first quarter of the year. About a quarter of this was of cuttlefish (*Sepia* spp.); the other main species discarded were Norway pout, whiting (Q1, Q2, Q3), dab (Q1) and gurnard (Q1). Around 50 t of spider crab (*Maja squinado*) were discarded in the first half of the year.

Three gillnet fleets were monitored in 1992 and 1993. These gillnet fleets differ in their location and the mesh size of nets used. Although the fleet targeting cod appears to have a low by-catch, this fishery has a high by-catch of porpoise (*Phocoena phocoena*).

#### Demersal stocks in the North Sea and Skagerrak

Information on discards from the demersal fisheries in the North Sea is very poor at present. This is despite the fact that these fisheries form a very large part of the North Sea fishing fleet, and probably generate a very large quantity of discards (Garthe *et al.*, 1996). The ACME noted that ICES is not receiving up-to-date information from studies currently under way. It is of particular concern that some studies are now complete but the data have not been made available to ICES nor to the wider scientific community. There are also a number

of on-going studies (e.g., Project EC 95/094 of DG XIV) and the ACME looks forward to being able to use their results in due course.

#### Discards from three German fleets

Three German fishing fleets were sampled between 1995 and 1997: 1) the fishery predominantly for saithe in ICES Division IVa, 2) the mixed gadoid fishery in ICES Divisions IVb and IVc, and 3) the fishery by small beam trawlers for flatfish in Divisions IVb and IVc. Only information on saithe, cod, whiting, haddock, plaice and sole was compiled by ICES (1998c). Information on discards of other commercially important species has been released, but was not forwarded to the Working Group. Discard sampling also did not continue throughout all of the years of the study, thus making estimates of annual amounts less accurate than desirable.

A total of 795 t of the six commercially important species recorded was discarded by the German saithe fishery in 1996, from a total catch of these species of 12,925 t (Table 13.2.2.4.2). The proportion discarded was comparatively high during the fourth quarter of the year and amounts of whiting discarded often exceeded amounts landed.

An estimated total of 164 t of these six species was discarded by the mixed gadoid fishery in ICES Divisions IVb and IVc, from a total catch of around 7000 t (Table 13.2.2.4.3). Cod predominated in both catches and discards. Lack of a discard recording scheme meant that discard amounts (and therefore total catch) had to be extrapolated from landed amounts, using the average proportion discarded relative to landings from the third and fourth quarters. The accuracy of these extrapolations depends on the similarity of discarding practices between the first and second half of the year.

**Table 13.2.2.4.2.** Discards and total catches (tonnes) of selected species from German saithe fisheries in Division IVa (from ICES, 1998c).

Quarter	Species	Discards	Catch
Q1 (1996)	Saithe	34	3,205
	Cod	8	293
	Whiting	29	35
	Haddock	50	212
	Plaice	0	0
	Sole	0	0
Q2 (1996)	Saithe	70	2,477
	Cod	7	275
	Whiting	9	21
	Haddock	71	407
	Plaice	0	0
	Sole	0	0
Q3 (1996)	Saithe	41	2,636
	Cod	5	420
	Whiting	8	14
	Haddock	74	154
	Plaice	0	0
	Sole	0	0
Q4 (1996, scaled from 1995 results)	Saithe	259	3,622
	Cod	30	695
	Whiting	67	96
	Haddock	33	363
	Plaice	0	0
	Sole	0	0
Total		795	12,925

**Table 13.2.2.4.3** Discards and total catches (tonnes) of selected species from German mixed gadoid fisheries in Divisions IVb and IVc (the numbers in parentheses are extrapolations) (from ICES, 1998c).

Quarter	Species	Discards	Catch
Q1 (1997 scaled from Q3/Q4 results)	Cod	(13)	(546)
	Whiting	(2)	(19)
	Haddock	(1)	(98)
	Plaice	(0)	(0)
	Sole	(0)	(0)
Q2 (1996 scaled from Q3/Q4 results)	Cod	(60)	(2,539)
	Whiting	(3)	(29)
	Haddock	(1)	(142)
	Plaice	(0)	(0)
	Sole	(0)	(0)
Q3 (1995)	Cod	48	2467
	Whiting	6	26
	Haddock	2	145
	Plaice	0	0
	Sole	0	0
Q4 (1996)	Cod	27	709
	Whiting	1	35
	Haddock	0	249
	Plaice	0	0
	Sole	0	0
Total		(164)	(7,004)

The German beam trawl fleet for flatfish was sampled in 1995. There were no records from the first quarter, as catches are apparently landed by Dutch fishermen under a German flag at this time of the year. A large amount of discards is released (7513 t) compared with the amounts caught (13,854 t) (Table 13.2.2.4.4). Expressed in catch per hour, the German beam trawl fishery in 1995 caught 68.7 kg/hour of marketable fish, and 148 kg/hour of non-marketable fish and shellfish were discarded, i.e., a discard rate of 68 %. Estimated numbers of fish discarded by the German fleet alone exceeded 60 million in 1995.

**Table 13.2.2.4.4.** Discards and total catches (tonnes) of selected species from German flatfish fisheries in Divisions IVb and IVc (from ICES, 1998c).

Quarter	Species	Discards	Catch
Q2 (1995)	Cod	229	
	Whiting	36	
	Haddock	0	
	Plaice	2,693	4,365
	Sole	168	911
Q3 (1995)	Cod	105	
	Whiting	29	
	Haddock	0	
	Plaice	2,156	4,592
	Sole	51	498
Q4 (1995)	Cod	53	
	Whiting	20	
	Haddock	0	
	Plaice	1,931	3,213
	Sole	42	275
Total		7,513	13,854

#### Dutch pair, otter and beam trawl

Little information was available to the Working Group on by-catch levels in these fleets (see Tables 13.2.2.4.5 and 13.2.2.4.6). A discard sampling scheme has been undertaken, partly funded by the EU, but data from this study were not available for review. The pattern of discards is apparently very variable between seasons, making it difficult to prepare quantitative annual estimates of discards. However, in view of the importance of the beam trawl fleet and the known concern about discard levels, the Working Group made a very approximate estimate of tonnage of discards of the three species.

Information on discards from the beam trawl fishery for flatfish in the North Sea is available in terms of percentage by numbers caught for the periods 1978–1982 and 1989–1990 by the Netherlands fleet (ICES, 1998c). Table 13.2.2.4.6 shows the percentage of catch discarded and numbers of fish discarded per 100 fishing hours for the main species. Discards are dominated by dab, plaice and sole. In order to convert the discards in numbers to discards by weight, an average weight of discards of dab



was assumed to be 0.092 kg (the weight of a dab of length 21 cm), plaice to be 0.122 kg (assumed length = 23 cm), and sole to be 0.106 kg (assumed length = 22 cm). These weights are crude but reasonable approximations based on minimum landing size and size at 50 % retention by the fishing gear. The numbers discarded were then related to the numbers landed. For plaice and sole, the percentage discarded can be related to the numbers landed as tabulated in the Assessment Working Group report (ICES, 1998c). The total weight of dab discards was estimated by multiplying the weight of plaice discards by the ratio of the number of dab discards to the number of plaice discards and taking account of the differences in mean discard weight. The results of the calculation (as shown in Table 13.2.2.4.7) should only be taken as a very crude indication of the discard level of the total international beam trawl fishery in this area. Overall, the discarded weight of these three species is roughly the same as the landed weight of these species. These species represent the great majority of fish being caught by that fleet.

**Table 13.2.2.4.7.** Estimates of discards of plaice, sole and dab from the beam trawl fleet in the North Sea (see text for sources and methods of calculation) for the years between 1978 and 1990.

Species	Discards (tonnes)	Total Catch (tonnes)
Plaice	42,000	181,000
Sole	1,600	22,600
Dab	110,000	> 110,000

## Danish fleets

Little information on discards is yet available from a three-year programme that commenced in 1995; some results are shown in Table 13.2.2.4.8.

## English fisheries

Discard rates of three species (whiting, haddock, and cod) in English fly seine, *Nephrops*, otter and pair trawl fisheries from 1994–1997 are presented in Tables 13.2.2.4.9, 13.2.2.4.10, and 13.2.2.4.11. Unfortunately, it is impossible with the information available to ICES to scale these figures up to provide tonnage estimates for total discards in these fisheries.

## Scottish demersal fleet

The Scottish discard sampling scheme in the North Sea has been in operation since 1975. This time series is used in assessments of the haddock and whiting stocks as being representative of the overall discarding practice. Quantities of various species of fish discarded by Scottish demersal vessels in the North Sea are presented in Table 13.2.2.4.12. No data on overall catches were available to the Working Group, so it was not possible to expand these estimates to entire fisheries.

**Table 13.2.2.4.8.** Discards in percent of catch for selected Danish fishing fleets.

	Danish Vessel Groups								
	Trawlers			Danish Seiners	Gillnetters				
	Demersal North	Demersal South	Industrial		Cod	Plaice	Turbot	Hake	Common sole
Number of hauls 1995+1996	7	64	75	50	110	15	25	16	3
% discard	25.5	8.0	0.0	18.7	4.8	8.2	20.1	0.0	38.2
Monk	<0.1			<0.1			0.2		
Whiting									2.1
Dab	0.4	0.2		2.6	<0.1	2.7			34.4
Haddock		<0.1		4.3	<0.1				
Hake					<0.1				
Mackerel		<0.1					1.3		
Turbot		<0.1				0.2	4.6		
Plaice	<0.1	2.6		<0.1		0.8	0.4		0.6
Saithe	24.7	<0.1		<0.1	0.1		<0.1		
Cod	0.2	4.9		1.1	2.2	2.5	5.5		
Common sole							<0.1		

**Table 13.2.2.4.5.** Discards of selected species in the Dutch pair trawl and other trawl fisheries (percentages of discards and numbers per 100 fishing hours). Average percentages are weighted over total catch numbers.

	No. of samples	Area	Cod		Whiting		Haddock		Bib		Plaice		Dab	
			%	No.	%	No.	%	No.	%	No.	%	No.	%	No.
Average	45	North Sea	20	6,179	53	27,944	29	303	88	949	84	3,647	84	20,499
	10	North Sea	44	7,827	80	44,671	29	1,101	na	459	Na	4,878	na	85,780

**Table 13.2.2.4.6.** Discards of selected species in the Dutch beam trawl fishery (percentages of discards and numbers per 100 fishing hours). Average percentages are weighted over total catch numbers.

	No. of samples	Area	Plaice		Sole		Dab		Flounder		Whiting		Cod	
			%	No.	%	No.	%	No.	%	No.	%	No.	%	No.
Average	49	North Sea	49	29,064	16	239	98	100,953	81	1,411	85	8,508	66	2,762
	8	North Sea	41	16,910	9	912	98	75,310	100	197	93	12,926	41	1,591
	9	North Sea	55	27,101	5	908	99	103,525	100	64	71	3,508	49	1,735
	9	North Sea	58	50,052	5	246	96	80,832	80	1,547	70	5,860	71	8,217
	8	North Sea	59	42,494	21	2,125	96	92,940	84	6,152	81	6,454	87	5,520
	5	North Sea	20	7,503	29	5,214	98	54,234	0	0	86	15,663	49	630
	6	North Sea	46	32,972	22	8,263	99	202,118	54	640	96	12,469	61	32
		Plaice box	83	83,433	8	935	99	160,347	100	201	88	3,852	80	10,215
		Non-plaice box	36	17,489	18	2,658	97	75,979	81	1,647	85	8,953	51	1,308

**Table 13.2.2.4.9.** Discards rates by weight and number for whiting in selected English fisheries.

Whiting	Year	Q1			Q2			Q3			Q4		
		% by No.	% by WT	No. hauls	% by No.	% by WT	No. hauls	% by No.	% by WT	No. hauls	% by No.	% by WT	No. hauls
Fly seine	1994	No data	No data	0	95	90	15	98	97	12	No data	No data	0
Nephrops	1994	94	84	10	100	98	3	97	94	7	99	98	11
Nephrops	1995	96	90	11	No data	No data	0	No data	No data	0	94	88	16
Nephrops	1996	85	75	12	98	93	12	No data	No data	0	93	86	17
Nephrops	1997	78	77	5	No data	No data	0						
Otter	1994	47	29	27	64	52	28	91	83	34	53	37	18
Otter	1995	66	58	12	26	18	33	48	35	17	48	34	15
Otter	1996	33	23	29	62	52	23	100	100	17	24	17	22
Otter	1997	19	18	22	53	39	64						
Pair	1994	76	59	8	39	32	12	59	46	6	92	87	6
Pair	1995	No data	No data	0	80	66	8	92	87	13	71	55	16
Pair	1996	No data	No data	0	42	30	11	60	47	8	No data	No data	0
Pair	1997	36	24	5	100	100	6						

Table 13.2.2.4.10. Discard rates by weight and number for haddock in selected English fisheries.

Haddock	Gear type	Q1			Q2			Q3			Q4		
		% by No.	% by WT	No. hauls	% by No.	% by WT	No. hauls	% by No.	% by WT	No. hauls	% by No.	% by WT	No. hauls
	Fly seine	No data	No data	0	93	88	15	99	98	12	No data	No data	0
	Nephrops	93	81	10	21	13	3	100	100	7	97	82	11
	Nephrops	82	46	11	No data	No data	0	No data	No data	0	81	67	16
	Nephrops	78	72	12	89	79	12	No data	No data	0	65	49	17
	Nephrops	79	78	5	No data	No data	0						
	Otter	69	59	27	41	26	28	66	50	34	28	18	18
	Otter	No data	No data	12	38	19	33	93	84	17	85	72	15
	Otter	70	48	29	58	46	23	67	57	17	30	19	22
	Otter	74	74	22	39	36	64						
	Pair	100	100	8	61	50	12	74	60	6	0	0	6
	Pair	No data	No data	0	84	77	8	73	53	13	53	31	16
	Pair	No data	No data	0	39	26	11	32	21	8	No data	No data	0
	Pair	0	0	5	50	48	6						

Table 13.2.2.4.11. Discard rates by weight and number for cod in selected English fisheries.

Cod	Gear type	Q1			Q2			Q3			Q4		
		% by No.	% by WT	No. hauls	% by No.	% by WT	No. hauls	% by No.	% by WT	No. hauls	% by No.	% by WT	No. hauls
	Fly seine	No data	No data	0	98	93	15	96	88	12	No data	No data	0
	Nephrops	30	9	10	43	8	3	18	8	7	69	41	11
	Nephrops	76	25	11	No data	No data	0	No data	No data	0	45	12	16
	Nephrops	40	8	12	34	8	12	No data	No data	0	80	41	17
	Nephrops	92	38	5	No data	No data	0						
	Otter	30	11	27	38	13	28	66	35	34	38	23	18
	Otter	12	2	12	12	6	33	47	22	17	33	14	15
	Otter	17	4	29	19	4	23	4	1	17	47	21	22
	Otter	39	19	22	22	8	64						
	Pair	10	2	8	45	21	12	57	32	6	18	4	6
	Pair	No data	No data	0	18	5	11	23	8	8	No data	No data	0
	Pair	No data	No data	0	18	5	11	23	8	8	No data	No data	0
	Pair	10	1	5	11	3	6						

**Table 13.2.2.4.12.** Annual estimates of discards (total biomass in tonnes) for species caught by Scottish demersal vessels in the North Sea. (The tabulated estimates are obtained using the weighted ratio estimator under the fill-in (from Stratoudakis, 1997)).

Species	1988	1989	1990	1991	1992	1993
Anglerfish	64	182	172	431	125	362
Cod	2,230	11,846	8,338	3,727	2,586	5,946
Common dab	8,947	3,528	2,712	1,351	1,919	4,360
Cuckoo ray	436	132	194	218	269	360
Grey gurnard	2,030	1,617	3,018	2,058	3,380	4,632
Haddock	42,289	29,742	24,569	33,027	43,709	60,714
Hake	7	4	208	41	40	16
Herring	1,159	478	168	3,393	95	1,465
Horse mackerel	81	530	664	304	130	657
Lemon sole	2,690	3,388	1,095	3,064	1,415	1,005
Lesser argentine	37	50	75	212	58	419
Lesser spotted dogfish	462	178	114	295	261	734
Long rough dab	1,122	425	511	1,305	1,187	1,135
Mackerel	14	193	231	711	55	257
Megrim	130	47	79	22	43	70
Norway pout	90	110	74	9,489	1,778	863
Plaice	2,344	1,915	946	520	580	1,696
Poor cod	34	29	27	70	56	54
Red gurnard	25	18	13	423	95	376
Saithe	137	863	332	1,698	1,533	9,510
Spotted ray	21	0	0	10	0	6
Starry ray	4,850	637	1,821	1,951	2,657	2,854
Whiting	26,672	27,576	25,520	32,251	24,332	32,165
Witch	668	577	192	486	246	262
Other	643	929	830	779	1,021	1,066
Landings of cod	41,382	31,480	31,120	28,748	28,204	28,191

### 13.2.2.5 *Nephrops*

The sampling of discards from the *Nephrops* fishery in the North Sea has been comparatively good, although sampling levels and strategies vary between the various grounds (see Table 13.2.2.5.1). The data reporting programmes include some indication of which fish species were discarded along with undersized *Nephrops*. There is a wide variation in amounts discarded among *Nephrops* fisheries, which is probably related to variations in the gear used (for example, UK vessels use a net with a panel that allows the escape of small fish). As with other fisheries, market demand also influences the type of fish discarded. Undersized *Nephrops* formed the bulk of most discards, but there was a wide mixture of fish discarded as well. In the Farn Deep, whiting was the major species discarded, while dab and long-rough dab were the main species discarded in the Kattegat.

Notable *Nephrops* grounds not sampled included the Noup and Fladden Ground.

### 13.2.2.6 *Pandalus* in Divisions IIIa and IVa east

ICES (1998d) records discards of *Pandalus borealis* from *Pandalus* fisheries in the main North Sea fishery in ICES Divisions IIIa and IVa (east) (see Table 13.2.2.6.1). These estimates were based on proportions of *Pandalus* in the Norwegian catch with a carapace (exoskeleton) length of less than 15 mm. There are no data on discards of species other than *Pandalus* for this fishery. High grading occurs with the discard of medium-sized fresh shrimps and retention of large boiled shrimps.

### 13.2.2.7 German brown shrimp (*Crangon*) fishery

Walter (1997) described the amounts of discards in the brown shrimp (*Crangon crangon*) fishery off Lower Saxony. Commercial shrimp represented 11 % of the weight of the catch. The majority of the catch by weight was undersized shrimp (64 %), other invertebrates (8 %), and fish (11 %) (Table 13.2.2.7.1). The highest discard ratio was in August, with much lower ratios in spring and

**Table 13.2.2.5.1.** *Nephrops* discards and total catches (tonnes) from some North Sea fishing grounds. (from ICES, 1997b). The parentheses indicate that a value is not a true total.

Area		Quantity in tonnes						
		1990	1991	1992	1993	1994	1995	1996*
Skagerrak	Discards					2,642	3,171	1,373
	Total catch					3,811	4,611	3,511
Kattegat	Discards					2,648	926	
	Total catch					2,736	1,549	
Firth of Forth	Discards	383	245	303	553	1,498	596	886
	Total catch	2,294	1,634	2,016	2,797	2,938	2,194	2,317
Moray Firth	Discards	191	289	308	214	152	464	463
	Total catch	2,287	1,808	1,880	2,037	1,756	1,601	1,727
Farn Deepes	Discards	1,040	820	756	383	1,166	530	990
	Total catch	3,278	2,883	2,219	3,413	4,863	3,098	3,469
Botney Gut/ Silver Pit	Discards			203	268	331	281	187
	Total catch			1,101	1,296	1,326	1,469	1,084
Total of fisheries	Discards	(1,614)	(1,354)	(1,570)	(1,418)	8,437	5,968	(3,899)
	Total catch	(7,859)	(6,325)	(7,216)	(9,543)	17,430	14,513	(12,108)

\*Provisional data

**Table 13.2.2.6.1.** *Pandalus* discards and total catches (tonnes) in ICES Divisions IIIa and IVa (east) in the northeastern North Sea (from ICES, 1998d).

Year	Quantity in tonnes						
	1990	1991	1992	1993	1994	1995	1996
Discards	1,723	765	713	1,340	426	642	1,282
Total catch	11,881	12,362	13,728	14,059	12,076	13,938	15,515

**Table 13.2.2.7.1.** Total catch and discards (tonnes) from the German brown shrimp fishery off Lower Saxony in 1993 (Walter, 1997).

Month	Total catch	Discarded shrimp	Discarded fish	Other discarded invertebrates
April	2,240	1,155	496	67
May	3,520	1,969	476	550
June	3,794	2,741	303	264
July	4,701	3,614	483	147
August	7,127	5,289	743	572
September	8,015	5,888	962	387
October	6,308	5,166	356	111
November	2,305	1,754	221	17
Total	38,009	27,576	4,040	2,114

autumn. Plaice were present in all samples of discards, with herring (73 % of samples) being the next most common commercial species in the discards. They also formed the majority of the fish discards by mass.

### 13.2.2.8 Other fisheries

*The above section includes a summary of all information made available to WGECO by the Assessment Working Groups. There are evidently considerable areas within the fisheries covered by the Assessment Working Groups where there has been no attempt to estimate discards. In addition, there are fisheries where there appears to be no careful evaluation of catches. Examples are the fishery for Norway pout, gillnet fisheries in the North Sea, and fisheries using seine nets.*

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## 13.2.3 Discards in OSPAR Region III: The Celtic Seas

### 13.2.3.1 Herring

#### Herring discards in the Celtic Sea and Division VIIj

The estimated herring catches from 1987–1996 for the combined areas by year and by season (1 April–31 March) are given in Tables 13.2.3.1.1 and 13.2.3.1.2, respectively. The reported catch taken during the 1996/1997 season, including the estimates of herring discards and unallocated landings, was over 17,000 t compared with 23,300 t during the previous season. The decrease was mainly due to marketing difficulties during early 1997 and a reduced level of discarding.

The level of discarding in this fishery is believed to have decreased in recent years as fishermen have become more expert in identifying suitable shoals for roe for the Japanese market and in controlling the amounts of fish in their nets. Nevertheless, discards may on occasion reach a high level, particularly if the fishery is allowed to remain open despite marketing difficulties; also, WGECO was not aware of specific studies on the proportion of catch lost by slippage. During the first quarter of 1997, the landings from Division VIIa (South) and Division VIIg were raised by 10 % to include discards as in previous years. The level of discards for the remainder of the season is not believed to have been significant.

**Table 13.2.3.1.1.** Discards and total catches (tonnes) of herring in the Celtic Sea and Division VIIj by calendar year for 1987–1996 (data from ICES, 1997a). These figures may not in all cases correspond to the official statistics and cannot be used for management purposes.

Year	Discards	Total Catch
1987	4,200	27,300
1988	2,400	19,200
1989	3,500	22,700
1990	2,500	20,200
1991	1,900	23,600
1992	2,100	23,000
1993	1,900	21,100
1994	1,700	19,100
1995	700	19,000
1996 <sup>1</sup>	3,000	21,800

<sup>1</sup> Preliminary

**Table 13.2.3.1.2.** Discards and total catches (tonnes) in the Celtic Sea and Division VIIj by season (1 April–31 March) for 1987/1988–1996/1997 (data from ICES, 1997a). These figures may not in all cases correspond to the official statistics and cannot be used for management purposes.

Year	Discards	Total Catch
1987/1988	4,000	26,200
1988/1989	3,400	20,400
1989/1990	3,600	23,100
1990/1991	1,700	18,600
1991/1992	2,100	25,600
1992/1993	2,000	21,200
1993/1994	1,800	18,600
1994/1995	1,900	19,300
1995/1996	3,000	23,300
1996/1997	600	17,400

#### **Herring discards in Divisions VIa (North), VIa (South) and VIIb,c**

The main catches in 1996 from this fishery in Division VIa (North) were taken by the UK (Scotland); in Division VIa (South) Ireland took over 95 % of the total allocated catch. The total amount of unallocated catches

in 1996 was over 8600 t, which was considerably higher than that recorded for 1995.

The catches taken in this area from 1986–1996 are shown in Table 13.2.3.1.3. There were no estimates of discards reported in 1995–1996, and there are no indications that discarding is a major problem in this fishery, even though substantial catches from this fishery in recent years have been taken in a 'roe' fishery. Reports, however, have been received of quantities of discarded herring taken by bottom trawlers fishing in the areas adjacent to known spawning grounds, but it has not been possible to quantify the amounts.

#### **Quality of data and collection programmes**

Reasonably reliable data appear to be available for the quantities of herring discarded in these targeted herring fisheries. Official data collection programmes do not occur in all fisheries, but the results of a recent EU-funded project (EU Project BIOECO/93/17) indicated that the overall discard rate of 10 %–20 % used by previous Working Groups for the Celtic Sea and Division VIIj fishery was realistic.

**Table 13.2.3.1.3.** Discards and total catches (tonnes) of herring in Division VIa (North) in 1982–1996 and in Divisions VIa (South) and VIIb,c, in 1986–1996 (ICES, 1997a). (The termed 'misreported' implies that catches that have been reported as taken in Division VIa were actually from the North Sea.)

Year	Division VIa (North)			Division VIa (South) and Divisions VIIb,c	
	Discards	Misreported	Total Catch	Discards	Total Catch
1982	-		92,630		
1983	-		63,523		
1984	-	19,142	63,864		
1985	-	4,672	38,994		
1986	-	10,935	71,078		28,785
1987	-	18,647	44,105		48,600
1988	-	11,763	35,516		29,100
1989	1,550	19,013	33,945	1,000	29,200
1990	1,300	25,266	44,774	2,530	43,969
1991	1,180	22,079	32,388	3,400	37,700
1992	200	22,593	28,888	100	31,850
1993	820	24,397	32,020	250	36,800
1994	700	30,234	24,619	700	33,900
1995		36,687	33,794		27,792
1996		56,007	26,105		32,500

### 13.2.3.2 Mackerel

In some fisheries, e.g., those in Sub-areas VI and VII, mackerel is taken as a by-catch in the horse mackerel fisheries. Reports from these fisheries have suggested that discarding may be significant because of the low mackerel quota relative to the high horse mackerel quota—particularly in those fisheries carried out by freezer trawlers. In autumn 1997, an EU-funded programme involving Norway and Scotland commenced with the intention of studying the performance of the purse seine fisheries for herring and mackerel in OSPAR Region II. This programme will provide data on discards for these fleets. At present only one country—the Netherlands—provides information on mackerel discards. The information from the Netherlands fleet is not applied to any other fleets outside the region, because Spanish studies (see below) indicate that discard rates may vary greatly according to fishing practices and areas fished. The discarding of small mackerel may again become a problem in all areas if the 1996 year class is very strong, as seems possible at present.

Information available on catches and discards of mackerel in Sub-area VI is provided in Table 13.2.3.2.1.

An EU programme carried out by Spain studied the rate of discards of all species taken by the Spanish fleets fishing in Sub-areas VI, VII and Division VIIIc. The results of this study (Pérez *et al.*, 1994) showed that the discard rates varied by species, area and fishing fleet. The observed levels of discards were between 0.2 % and 25.7 % for horse mackerel, and between 0.1 % and 8.1 % for mackerel.

**Table 13.2.3.2.1.** Discards and total catches (tonnes) of mackerel in Sub-area VI. Discards not estimated prior to 1978. (Data from ICES, 1998a)

Year	Discards	Total Catch
1978	15,100	166,900
1979	20,300	223,600
1980	6,000	224,700
1981	2,500	337,600
1982	4,100	344,500
1983	22,300	337,400
1984	1,600	307,700
1985	2,735	390,875
1986	-	104,100
1987	-	183,700
1988	3,100	118,700
1989	2,600	123,900
1990	5,800	120,600
1991	10,700	120,200
1992	9,620	151,526
1993	2,670	136,167
1994	1,390	135,728
1995	74	145,700
1996	255	130,150

### 13.2.3.3 Horse mackerel

Spain, Portugal, Ireland, Denmark, and the Netherlands have directed trawl and/or purse seine fisheries for horse mackerel. In OSPAR Region III, the western horse mackerel stock is caught. Only one country provides data for discards. Therefore, the quantities of discards given in Table 13.2.3.3.1 are not representative for the total fishery. In the discard study described by Pérez *et al.* (1994), observed levels of discards between 0.2 % and 25.7 % for horse mackerel were found.

### 13.2.3.4 Demersal trawl fisheries

For stocks assessed by the Working Group on the Assessment of Southern Shelf Demersal Stocks, a distinction can be made between discard data which are used in the Working Group assessments of particular stocks, and data resulting from discard sampling programmes in which all (or most) species in the catch have been recorded.

Tables 13.2.3.4.1 to 13.2.3.4.3 give the quantities of discards from Irish trawl fleets operating during 1996 in ICES Divisions VIIb, VIIg and VIIj, respectively, estimated by raising the sampled catches in relation to the quarterly landed catch of target species by the corresponding fleets. A detailed description of the Irish discard sampling scheme is given in a working document (Connolly and Wheatley, 1997; ICES 1998b). A range of species is discarded depending on the area and target species. In Division VIIb, dogfish, grey gurnard and haddock comprise 60 % of the total discards by the Irish trawl fishery (Table 13.2.3.4.1); in Division VIIj dogfish, haddock, megrim and whiting comprise 60 % of the total discards (Table 13.2.3.4.2), and in Division VIIg discards are dominated by whiting and haddock (Table 13.2.3.4.3).

Discard sampling is not a routine part of sampling for any of the countries contributing to the Working Group on the Assessment of Southern Shelf Demersal Stocks and is mostly dependent on external funding.

Estimates of quantities of fish discarded were available for the following stocks which are assessed by the Working Group on the Assessment of Northern Shelf Demersal Stocks:

Species	ICES Division	Fleets	Years
Whiting	VIa	All Scottish trawlers	1976–1996
Whiting	VIIa	<i>Nephrops</i> trawlers	1981–1996
Haddock	VIa	All Scottish trawlers	1976–1996



**Table 13.2.3.3.1.** Total landings and discards (tonnes) of the complete western horse mackerel fisheries. Specific landings for Divisions VIa and VIIa-c,e-k are included.

Year	Discards	Total landings	Landings Division VIa	Landings Divisions VIIa-c, -k
1982	-	41,587	6,283	32,231
1983	-	64,862	24,881	36,926
1984	500	73,625	31,716	38,782
1985	7,500	80,551	33,025	35,296
1986	8,500	105,665	20,343	72,761
1987	-	157,240	35,197	99,942
1988	3,740	188,100	45,842	81,978
1989	1,150	268,867	34,870	131,218
1990	9,930	373,463	20,794	182,580
1991	5,440	333,555	34,415	196,926
1992	1,820	37,050	40,881	180,937
1993	8,600	433,145	53,782	204,318
1994	3,935	388,875	69,546	194,188
1995	2,046	510,597	83,486	320,102
1996	16,870	396,652	81,259	252,823

**Table 13.2.3.4.1.** Discards (tonnes) in the Irish otter trawl fleet operating in ICES Division VIIb during 1996 (data from ICES, 1998b). Mesh size > 80 mm. -

Quarter	Q1	Q2	Q3	Q4	Total
Target species	Megrim	Megrim	Megrim	<i>Nephrops</i>	1996
Landings of target species (tonnes)	237	166	163	100	
Total discards (tonnes)	264	760	1,677	270	2,972

**Table 13.2.3.4.2.** Discards (tonnes) in the Irish otter trawl fleet operating in ICES Division VIIj during 1996 (data from ICES, 1998b). Mesh size > 80 mm.

Quarter	Q1	Q2	Q3	Q4	Total
Target species	Megrim	Whiting	Whiting	Whiting	1996
Landings of target species (tonnes)	364	383	260.5	121	
Total discards (tonnes)	121	326	62	107	618

**Table 13.2.3.4.3.** Discards (tonnes) in the Irish otter trawl fleet (OTB) and beam trawl fleet (TBB) operating in ICES Division VIIg during 1996 (data from ICES, 1998b). Mesh size: OTB > 80 mm; TBB > 80 mm.

Gear	OTB	OTB	TBB	TBB
Quarter	Q2	Q3	Q2	Q4
Target Species	<i>Nephrops</i>	Whiting	Anglerfish	Anglerfish
Landings of target species (tonnes)	277	600	45	9
Total discards (tonnes)	71	649	7	2

**Table 13.2.3.4.4.** Estimates of the weight of discards of whiting and haddock (tonnes) by trawl fleets operating in Divisions VIa and VIIa. Percentages of the total catches of the sampled fleets are given where available (data from ICES, 1998c).

Year	Whiting				Haddock	
	VIa all fleets		VIIa N. Ireland <i>Nephrops</i> fleets		VIa all fleets	
	Discards (t)	% of total catch	Discards (t)	% of total catch	Discards (t)	% of total catch
1993	11,855	54 %	2,702	-	16,904	47 %
1994	18,964	59 %	1,186	-	11,192	44 %
1995	15,944	54 %	2,153	57 %	8,794	42 %
1996	11,776	48 %	3,494	67 %	11,826	47 %

Discarding by Scottish trawlers in Division VIa is estimated by the Marine Laboratory in Aberdeen, based on an observer programme. Estimates of discarding from Division VIIa *Nephrops* fleets are obtained by analysing samples of discard material provided by skippers (Fisheries Research Centre (FRC), the Marine Institute, Dublin and the Department of Agriculture for Northern Ireland (DANI), Belfast). An EU-funded programme to sample Irish vessels including those in the Irish Sea ran from 1995 to 1997 (FRC, Dublin). A further EU-funded programme, which commenced in 1996 and is coordinated by CEFAS, Lowestoft, includes estimation of discarding by sectors of the Northern Ireland trawl fleet previously unsampled for discards (DANI, Belfast). Other estimates of discarding are available from studies carried out by the English Sea Fish Industry Authority (SFIA) (Hepples, 1993; Emberton *et al.*, 1995) in the Irish Sea in 1992 and again in 1993/1994.

Table 13.2.3.4.4 gives the quantities of Divisions VIa and VIIa whiting and VIa haddock discarded annually by sampled fleets since 1993, and the percentage of the total catch of these fleets that was discarded. More detailed information for the different Scottish trawl fleets operating in Division VIa is collected by the Marine Laboratory, Aberdeen, but was not available to ICES at the time of the WGECO meeting.

The discard sampling carried out by the SFIA (UK) in 1993 and 1994 provided estimates of discard rates of plaice, sole and whiting by English otter trawlers, *Nephrops* trawlers, beam trawlers and anchor seines. The results are summarized below.

For plaice and sole, discarding was confined mainly to fish below the minimum landing size, and was influenced by depth and geographical location. An average of 63 % of whiting above the minimum landing size of 27 cm was discarded. Discarding of whiting was more variable than discarding of the flatfish and was controlled mainly by marketing factors, as whiting is of comparatively low value. Overall discard rates (expressed as percent of the total catch) were as follows:

Species	Number of hauls sampled	Discarded (%)	Standard error	Main age range discarded
Sole	131	8	1.2	3-4
Plaice	162	55	2	2-3
Whiting	147	88	1.9	1-3

The percentage discarded by fleet and species is given as:

Species	Whitefish otter trawl	<i>Nephrops</i> trawl	Beam trawl
Sole	5 %	< 1 %	3 %
Plaice	58 %	54 %	67 %
Whiting	72 %	97 %	93 %

Data on discards of non-commercial species in the Northern Ireland *Nephrops* fleet for 1996 are shown in Table 13.2.3.4.5. This fleet fishes using otter trawls with 70 mm codends. Square mesh panels have been mandatory in recent years.

**Table 13.2.3.4.5.** Discards (kg) of non-commercial species by the Northern Ireland *Nephrops* fleet fishing the western Irish Sea in 1996 (data from ICES, 1997b).

Species	Total discarded
Aphrodite	8,346
Argentine	636
Alloteuthis	7,292
Bib	16,629
Mud balls	5,065
Cuckoo ray	4,973
Spurdog	6,145
Eledone	77,986
Flounder	3,158
Four-bearded rockling	454
Fries goby	1,933
Common goby	48
Grey gurnard	182,848
Red gurnard	35,796
John Dory	592
Lemon sole	24,832
Swimming crab	5,801
Lesser spotted dogfish	62,735
Norway pout	248,089
Pogge	155
Queen scallop	824
Snake blenny	4,297
Homelyn ray	3,900
Spotted dragonet	4,784
Solenette	130
Sprat	727
<i>Sepiolo</i> spp.	3,187
Thickback sole	6,014
Tub gurnard	13,218
Lesser weever	181
Blue whiting	2,557
Buccinum	26,016

In 1994 Ireland commenced an EC-funded programme to sample discards at sea. Estimates of discarding in 1996 for the main Irish fleets operating in Divisions VIIa and VIa are shown in Table 13.2.3.4.6.

**Table 13.2.3.4.6.** Estimated discards (tonnes) from the main Irish fleets operating in Divisions VIIa and VIa during 1996.

Gear	Division VIIa			Division VIa	
	Otter trawl	Twin rig	Beam trawl	Seiner	Otter trawl
Total (t)	687	433	52	162	3,011

### 13.2.3.5 *Nephrops*

Details of the *Nephrops* sampling procedures used by different countries were given in the report of the Study Group on Life Histories of *Nephrops* (ICES, 1996). This Study Group reviewed *Nephrops* sampling because quite wide variations in sampling levels had been identified in the context of a presentation of sampling levels by Functional Unit (a Functional Unit is a sub-area of a Management Area, used in the management of *Nephrops* stocks).

The Study Group attempted to identify good practice and optimum sampling strategies. The sampling mainly targets vessels directed at *Nephrops*. Although *Nephrops* landings are nearly always adequately sampled, discards have to be sampled at sea and this has resourcing implications which limit the frequency of sampling. In

some sampling programmes, fish discards are also sampled.

A summary of the availability of discard sampling from *Nephrops* trawlers within OSPAR Region III is given in Table 13.2.3.5.1. This identifies by *Nephrops* Management Area and/or Functional Unit the availability of *Nephrops*, commercial fish, non-commercial fish and benthos discard samples from *Nephrops* trawlers. In nearly all cases, the benthos is not recorded. Only in Scotland, Spain, and Portugal are the non-commercial discards sampled. In Scotland, the fish discard data are collected specifically for the ICES area-based fish assessment Working Groups, and are aggregated to match fish stock assessment areas. These data are available from the Working Groups on the Assessment of Northern Shelf Demersal Stocks, Southern Shelf Demersal Stocks, and Demersal Stocks in the North Sea and Skagerrak and from the various Working Groups assessing pelagic stocks.

Table 13.2.3.5.2 shows the discards of fish in the Functional Units used for *Nephrops* assessment within OSPAR Region III. Some of these data were collected during the course of one-off sampling programmes, e.g., the EC-funded project BIO/ECO/93/003, while other discard programmes are currently being funded by the EC. Information on by-landings, i.e., the fish landed from *Nephrops* trawlers, is readily available. While this does not quantify discards, where discard data are not available, it would indicate the commercial species most likely to be discarded if data were available.

**Table 13.2.3.5.1.** Details of the discard sampling available from *Nephrops* trawlers.

ICES Division	Functional Unit (FU)	<i>Nephrops</i>	Commercial fish	Non-Commercial fish	Benthos
VIa	N Minch (11)	Y	Y	Y	N
	S Minch (12)	Y	Y	Y	N
	Firth of Clyde	Y	Y	Y	N
VIIa	Irish Sea E (14)	Y	Y	N	N
	Irish Sea W (15)	Y	Y	Y	Y
VIIb,c,j,k	Porcupine Bank (16)	Y	Y	Y	N
	Aran ground (17)	N	N	N	N
VIIIf,g,h	Celtic Sea (20–22)	Y	Y	N	N

**Table 13.2.3.5.2.** Discards and landings (tonnes) of *Nephrops* in the Functional Units (FUs) within OSPAR Region III (ICES, 1997b).

***Nephrops* North Minch (FU11)**

Year	Discards	Landings
1990	252	2,201
1991	388	2,440
1992	312	3,220
1993	27	2,920
1994	1,541	3,410
1995	741	3,166
1996	267	2,341

***Nephrops* South Minch (FU12)**

Year	Discards	Landings
1990	639	4,202
1991	109	3,998
1992	343	3,819
1993	529	3,891
1994	414	4,160
1995	398	3,988
1996	384	3,430

***Nephrops* Firth of Clyde (FU13)**

Year	Discards	Landings
1990	395	2,745
1991	215	2,697
1992	85	2,471
1993	240	2,866
1994	309	2,180
1995	548	3,680
1996	555	3,681

***Nephrops* Irish Sea East (FU14)**

Year	Discards	Landings
1991	461	859
1992	151	495
1993	217	618
1994	39	514
1995	33	504
1996	36	450

***Nephrops* Irish Sea West (FU15) (Northern Ireland fleet)**

Year	Discards	Landings
1991	217	6,024
1992	731	5,112
1993	892	5,355
1994	667	5,841
1995	839	5,401
1996	711	5,601

***Nephrops* Irish Sea West (FU15) (Ireland fleet)**

Year	Discards	Landings
1991	817	3,371
1992	546	2,370
1993	739	2,715
1994	354	1,768
1995	743	3,247
1996	631	2,255

***Nephrops* Irish Sea West (FU15) (Total International Catch)**

Year	Discards	Landings
1991	1,034	9,395
1992	1,277	7,482
1993	1,631	8,070
1994	1,021	7,609
1995	1,582	8,648
1996	1,342	7,856

***Nephrops* Celtic Sea (FU20-22)**

Year	Discards	Landings
1991	3,047	3,295
1992	3,874	4,165
1993	3,436	4,586
1994	4,237	5,130
1995	4,555	5,922
1996	3,921	4,889

It should be remembered that the exploitation pattern generated on fish by *Nephrops* trawls is likely to be quite different from that generated by directed finfish vessels. The *Nephrops* mesh size permitted is considerably smaller than that permitted for fish. In NEAFC Regions 1 and 2 the *Nephrops* mesh size is 70 mm, while the fish mesh size ranges from 80 mm to 100 mm. In NEAFC Region 3 the *Nephrops* mesh size is generally 55 mm, and the fish mesh size is 65 mm. Sampling programmes looking at fish discards need to ensure that all *Nephrops* vessel categories are sampled. These smaller mesh sizes are only permitted if certain catch composition conditions are met. EC Council Regulation 3094/86 specifies for Regions 1, 2 and 3 that a minimum of 30 % by weight in the retained catch must be *Nephrops*, and that the proportion of protected (Annex II) species must not exceed 60 %. In the UK, national technical measures specify that square mesh panels of a mesh size of 80 mm (75 mm in Sub-area VII) must be fitted to *Nephrops* trawls. Square mesh panels allow small fish, particularly whiting and haddock, to escape before reaching the codend, and significantly reduce the quantities of small fish which have to be discarded. Quantities of the major discard species in the Northern Ireland *Nephrops* fleet are listed in Table 13.2.3.5.3.

Fish discarding from *Nephrops* trawlers is subject to a wide range of factors. While technical measures, such as by-catch limits and minimum landing sizes, determine discarding practice, other factors also come into play such as the market demand for certain species and the seasonal nature of the *Nephrops* fisheries. In the eastern Irish Sea (FU 14), whiting which exceed the minimum landing size can be discarded through lack of market demand for this species, while other species like plaice

and sole are carefully sorted and only undersized fish are discarded. Some *Nephrops* fisheries are prosecuted in the winter, others mainly in the summer. The fish species composition can differ significantly with the seasons. Estimates of the fish discards need to be weighted by the seasonal directed *Nephrops* fishing effort.

### 13.2.3.6 Other species

#### Deep-water fisheries

There have been relatively few discard studies in the deep-water fisheries of the ICES area. One Irish study which compares the discards of trawl and longline fisheries in ICES Division VIa was published in 1996 (Connolly and Kelly, 1996), and new data are being collected from commercial vessels by observers from several countries.

Other discard studies are being carried out as part of the EC FAIR Project (95/655) 'Developing deep-water fisheries: data for their assessment and for understanding their interaction with and impact on a fragile environment'. The Marine Laboratory, Aberdeen, has undertaken two trips on Scottish commercial vessels to observe discards of deep-water species in ICES Division VIa. These data are still in a raw format and a preliminary analysis is not yet available. French studies have estimated the discards of *Coryphaenoides rupestris* from Sub-areas VI and VII as 34 % of the landings (V. Allain, IFREMER, unpublished). Norway has collected some data from the Reyknes Ridge in 1996 and further sampling was undertaken in 1997. These data will be available in due course.

**Table 13.2.3.5.3.** Estimated quantities (in tonnes) of fish and invertebrates discarded in the Northern Ireland *Nephrops* fisheries in 1996 (data from ICES, 1997b).

Species		Tonnes discarded
Whiting	<i>Merlangius merlangus</i>	2,494
Dublin Bay prawn (Norway lobster)	<i>Nephrops norvegicus</i>	711
Norway pout	<i>Trisopterus esmarkii</i>	248
Dab	<i>Limanda limanda</i>	211
Plaice	<i>Pleuronectes platessa</i>	206
Starfish	<i>Asteroidea</i> spp.	194
Grey gurnard	<i>Eutrigla gurnardus</i>	183
Haddock	<i>Melanogrammus aeglefinus</i>	139
Edible crab	<i>Cancer pagurus</i>	107
Horse mackerel	<i>Trachurus trachurus</i>	97
Poor cod	<i>Trisopterus minutus</i>	87
Curly octopus	<i>Eledone cirrosa</i>	78
Common dragonet	<i>Callionymus cirrosa</i>	65
Long rough dab	<i>Hippoglossoides platessoides</i>	63
Lesser spotted dogfish	<i>Scyliorhinus canicula</i>	63
Other species		297
Total		5,742

## Marine mammals

A programme to assess the cetacean by-catch in the Irish and UK set gillnet fisheries in the Celtic Sea was conducted from 1992–1994 using volunteer observers. Observers were present for the hauling of over 2500 km of net which caught 43 harbour porpoises and four common dolphins. The by-catch rate was 7.7 porpoises per 10,000 km.hour of net immersion. The estimated total annual by-catch of 2200 porpoises (95 % confidence limits 900–3500) is 6.2 % of the estimated number of porpoises in the Celtic Sea and there is serious cause for concern about the ability of the population to which they belong to sustain this level of by-catch (Tregenza *et al.*, 1997).

Eleven fisheries were investigated in a study of the by-catch of marine mammals in pelagic trawl fisheries of the Northeast Atlantic. In one fishery (Irish herring trawling), four grey seals were caught in approximately 100 hours of trawling. Eleven different pelagic trawl fisheries (Dutch horse mackerel, French tuna, French hake, French sea bass) caught a total of eighteen dolphins during a total of 1300 hours of trawling. The extent of observation and the number of observed by-catches were insufficient to make a reliable estimate of overall catch rates by gear type, but the average total catch rates were between 1.1 and 1.5 cetaceans per 100 hours of trawling (Morizur *et al.*, 1997).

A number of other studies in the Northeast Atlantic were identified by the Working Group as containing important information on discards, but which were not reviewed (Goujon *et al.*, 1993a, 1993b, 1996; Antoine *et al.*, 1997).

## References

Antoine, L., Goujon, M., and Massart, G. 1997. Captures accidentelles de dauphins dans les filets dérivants à thon en Atlantique Nord Est. ICES CM 1997/Q:10. 8 pp.

Connolly, P.L., and Kelly, C.J. 1996. Catch and discards from experimental trawl and longline fishing in the deep water of the Rockall Trough. Journal of Fisheries Biology, 49 Supplement A: 132–144.

Connolly, P., and Wheatley, S. 1997. The Irish discard sampling scheme. Working document at the 1997 meeting of the ICES Working Group on the Assessment of Southern Shelf Demersal Stocks.

Emberton, M., Course, G., and Lart, W. 1995. Irish Sea finfish and *Nephrops* discard study 1993/94. SFIA Consultancy Report No. CR99.

Goujon, M., Antoine, L., Collet, A. and Fiffas, S. 1993a. Approche de l'impact écologique de la pêche thonière

au filet maillant dérivant en Atlantique nord-est. Report submitted to the Commission of the European Community (24th Report of the Scientific and Technical Committee for Fisheries, Brussels, 15–17 November 1993). RI-DRV 93.034 RH/Brest, 47 pp.

Goujon, M., Antoine, L., and Collet, A. 1993b. Incidental catch of cetaceans in the French albacore tuna driftnet fishery: preliminary results. ICES CM 1993/N:13. 17 pp.

Goujon, M., Antoine, L., and Leroy, B. 1996. Captures de la pêche germonière française au filet maillant dérivant dans l'Atlantique nord-est. Albacore research programme, SCRS/94/176. Vol. Scient. Papers Rec. Doc. Sci. ICCAT, 191–205.

Hepples, S. 1993. MAFF-funded discard study—Irish Sea VIIa—January to December 1992. SFIA: Seafish Report No. SR428.

ICES. 1996. Report of the Study Group on Life Histories of *Nephrops*. ICES CM 1996/K:2.

ICES. 1997a. Report of the Herring Assessment Working Group for the Area South of 62°N. ICES CM 1997/Assess:8.

ICES. 1997b. Report of the Working Group on *Nephrops* Stocks. ICES CM 1997/Assess:9.

ICES. 1998a. Report of the Working Group on the Assessment of Mackerel, Horse Mackerel, Sardine and Anchovy. ICES CM 1998/Assess:6.

ICES. 1998b. Report of the Working Group on the Assessment of Southern Shelf Demersal Stocks. ICES CM 1998/Assess:4.

ICES. 1998c. Report of the Working Group on the Assessment of Northern Shelf Demersal Stocks. ICES CM 1998/Assess:1.

Morizur, Y., Tregenza, N., Heessen, H., Berrow, S., and Pouvreau, S. 1997. Incidental mammal catches in pelagic trawl fisheries of the Northeast Atlantic. ICES CM 1997/Q:5.

Pérez, N., Pereda, P., Uriarte, A., Trujillo, V., Olaso, I., and Lens, S. 1994. Discards of the Spanish fleet in ICES Sub-areas VI and VII. Study contract DGXIV, Ref.N:PEM/93/005.

Tregenza, N.J.C., Berrow, S.D., Hammond, P.S., and Leaper, R. 1997. Harbour porpoise (*Phocoena phocoena* L.) by-catch in set gillnets in the Celtic Sea. ICES Journal of Marine Science, 54: 896–904.

#### 13.2.4 Discards in OSPAR Region IV: Bay of Biscay and Iberian Coast

There have been few published studies of discard practices in this region; the largest was a project partly funded by the EU (EC DGXIV PEM/93/005) reported on by Pérez et al. (1995).

Discards of species in the Spanish trawl fisheries can reach up to 60 % of the total catch (in weight) of all species (Table 13.2.4.1). These levels of discarding are in broad agreement with the levels reported by Pérez et al. (1995) for Sub-area VII.

Nearly all non-target species caught in the Spanish trawl fishery in Division VIIIc are discarded at a rate of 100 % (Table 13.2.4.2) (Olaso et al., 1996).

There is no information available on discards in the southern (Portuguese) part of Division IXa. The bottom trawl fisheries may discard undersized hake, horse mackerel, sardine, and Nephrops as these species have a minimum landing size.

##### References

- Olaso, I., Velasco, F., Pereda, P., and Pérez, N. 1996. Importance of the blue whiting discard in the diet of lesser spotted dogfish in the Cantabrian Sea. ICES CM 1996/Mini:2.
- Pérez, N., Pereda, P., Uriarte, A., Trujillo, V., Olaso, I., and Lens, S. 1995. Final report on the project 'Discards of the Spanish fleet in ICES Divisions'. Study contract DG XIV, ref. no. PEM/93/005.

#### 13.2.5 Discards in OSPAR Region V: The Wider Atlantic

There are no discard data reported for fisheries on groundfish in OSPAR Region V. Recent studies have recorded by-catches in two trips for redfish in this region, using midwater pelagic trawls, but these data are in preliminary form and have not been released. Due to the very low intensity of coverage of this study to date, these data should not be considered representative of by-catches and discarding in this fishery. Within several years, the existing programmes may be able to provide some picture of the incidental catches in this fishery.

Species discarded in the Azores line and gillnet fisheries include various scabbard fish (*Aphanopus* sp.), skates, greater forkbeard, morid cod, anglerfish (*Lophius piscatorius*), and rabbit fish (*Chimaera monstrosa*). There are no quantitative estimates of the amounts of discards for any of these species.

Particularly after its 1994 annual meeting, the International Commission for the Conservation of Atlantic Tunas

(ICCAT) has sponsored programmes to monitor by-catches and discarding in fisheries for tunas and tuna-like fish. Their reporting regions do not correspond directly with the OSPAR regions, and ICCAT (1996) contains only lists of species and qualitative descriptions of by-catch. Moreover, for the 1996 report, responses to the ICCAT inquiries about by-catches had been received from fewer than 20 of 95 possible fisheries, so coverage is incomplete everywhere, including in OSPAR Region V. Results of these ongoing studies will be reported in future ICCAT annual reports. Based on this preliminary information, the main species taken as by-catch in fisheries for tuna and related species in the North Atlantic are sharks, particularly blue and mako sharks, and there is some catch of sea-turtles and seabirds in these fisheries.

##### Reference

- ICCAT. 1996. International Commission for the Conservation of Atlantic Tunas; Report for the Biennial Period 1994–1995, Part II (1995), Volume 2. 237 pp.

#### 13.2.6 Discussion of the information on discards

Discards and by-catches clearly constitute one of the major ecosystem effects of fishing, sometimes even potentially greater than the direct harvest of the target species. The quantity and quality of data on discards need to be improved, both to enable evaluation of the ecosystem effects of fishing, and to increase the reliability of present single-species and multispecies assessments. Two types of improvements are needed. More programmes must be implemented to quantify discards, and the data which are collected need to be handled better. Instituting additional discard monitoring programmes can have major cost implications, particularly because the programmes must be scientifically credible. However, cost is not a major factor in improving the quality of handling and presenting the data which have been collected.

The requirements for scientifically sound programmes for collecting and handling data on cetacean by-catches have been reviewed in depth in Northridge (1996). That source reviews experience from a number of practices which are generally applicable, and not only specific to monitoring the by-catch of cetaceans. The experience gained indicates the following:

- Self-reported magnitudes of by-catches are often unreliable, and rarely credible.
- Except for non-target species whose populations are routinely reported as numbers instead of weight (seabirds, marine mammals), by-catches and discards should be reported in absolute weight (or percent of catch, if the catch is recorded in weight on the same record), not in numbers.

**Table 13.2.4.1.** Percentage of discards/total discards and discards/total catch of the Spanish trawl fleets in ICES Divisions VIIIa,b,c and IXa in 1994. From Pérez *et al.* (1995).

<b>a) Trawlers in ICES Divisions VIIIa,b</b>		
<b>Species</b>	<b>% Discards/Total Discards</b>	<b>% Discards/Total Catch</b>
Horse mackerel	52.4	25.7
Poor cod	6.5	3.1
Mackerel	5.4	2.6
Blue whiting	5.1	2.4
Dragonet	3.4	1.6
<i>Trachurus</i> spp.	3.2	1.5
Dogfish	2.3	1.1
Black-mouthed dogfish	2.2	1.0
<i>Munida</i> spp.	1.9	0.9
Red gurnard	1.6	0.8
Blue-leg swimcrab	1.1	0.5
Hake	1.0	0.5
Cuckoo ray	1.0	0.5
Other species	11.1	5.3
<b>Total</b>		<b>47.4</b>
<b>b) Trawlers in ICES Division VIIIc</b>		
<b>Species</b>	<b>% Discards/Total Discards</b>	<b>% Discards/Total Catch</b>
Blue whiting	24.4	12.2
Dogfish	10.4	5.2
Horse mackerel	9.5	4.7
Black-mouthed dogfish	7.2	3.6
Silver pout	6.2	3.1
<i>Munida</i> spp.	5.4	2.7
Curly octopus	5.4	2.7
<i>Geryon longipes</i>	3.1	1.5
<i>Polybius henslowii</i>	3.0	1.5
<i>Actinauge richardi</i>	3.0	1.5
<i>Opistoteuthis agassizi</i>	2.5	1.2
Board fish	2.4	1.2
Sea cucumber	1.9	1.0
Blue-leg swimcrab	1.8	0.9
Hermit crab	1.4	0.7
Four-spot megrim	1.3	0.7
Roughnose rattail	1.0	0.5
Other species	10.0	5.0
<b>Total</b>		<b>49.9</b>
<b>b) Trawlers in ICES Division IXa</b>		
<b>Species</b>	<b>% Discards/Total Discards</b>	<b>% Discards/Total Catch</b>
Snipe fish	35.5	20.7
<i>Polybius henslowii</i>	20.7	12.3
Blue whiting	16.3	9.7
Dogfish	3.5	2.1
<i>Munida</i> spp.	2.9	1.7
Holothuroidea undetermined	2.4	1.4
Silver pout	1.5	0.9
Echinoidea undetermined	1.4	0.8
European sardine	1.1	0.6
Other species	15.2	9.0
<b>Total</b>		<b>59.2</b>



**Table 13.2.4.2.** Fauna caught and discarded (kg/100 fishing hours) by the Spanish fleet in ICES Sub-area VIIIc in 1994 (from Olaso *et al.*, 1996).

TAXA	Catch	Discards	Discards/Catch (%)
CRUSTACEA			
DECAPODA			
Anomura			
<i>Munida</i> undetermined	138	138	100
Paguridae undetermined	1	1	100
<i>Pagurus alatus</i>	8	8	100
<i>Pagurus prideauxi</i>	33	33	100
Brachyura			
<i>Liocarcinus depurator</i>	41	41	100
<i>Macropipus tuberculatus</i>	4	4	100
<i>Polybius henslowii</i>	83	83	100
Decapoda undetermined	7	7	100
Macrura			
<i>Polycheles typhlops</i>	1	1	100
Natantia			
<i>Chlorotocus crassicornis</i>	1	1	100
<i>Dichelopandalus bonnieri</i>	3	2	67
<i>Plesionika heterocarpus</i>	1	1	100
<i>Processa</i> spp.	1	1	100
<i>Solenocera membranacea</i>	4	4	100
MOLLUSCA			
CEPHALOPODA			
Decapoda			
<i>Alloteuthis</i> spp.	2	2	100
<i>Rossia macrosoma</i>	10	10	100
Sepiidae undetermined	4	1	25
Octopoda			
Octopoda undetermined	2	1	50
CNIDARIA			
Anthozoa undetermined	71	71	100
POLYCHAETA			
<i>Aphroditae aculeata</i>	2	2	100
FISH			
ANACANTHINI			
<i>Antonogadus macrophthalmus</i>	1	1	100
<i>Gadiculus argenteus</i>	135	135	100
<i>Micromesistius poutassou</i>	2,775	530	19
<i>Trisopterus luscus</i>	96	4	4
<i>Trisopterus</i> spp.	3	2	67
<i>Merluccius merluccius</i>	473	16	3
GOBIOIDEI			
<i>Callionymus</i> undetermined	10	10	100
<i>Argentina sphyraena</i>	26	12	46
<i>Sardina pilchardus</i>	1	1	100
MYCTOPHOIDEI			
Myctophidae undetermined	2	2	100
NOTIDANOIDEI			
<i>Scyliorhinus canicula</i>	931	225	24
PERCOIDEI			
<i>Trachurus trachurus</i>	578	10	2
<i>Cepola rubescens</i>	16	16	100
<i>Trachinus draco</i>	5	5	100
PLEURONECTOIDEI			
<i>Amoglossus laterna</i>	5	5	100
<i>Lepidorhombus boscii</i>	119	31	26
SCOMBROIDEI			
<i>Scomber scombrus</i>	301	20	7
ZEOMORPHI			
<i>Capros aper</i>	63	63	100

- Gear and units of effort associated with by-catches and discards should always be reported.
- Position information should always be reported.
- By-catches and discards should be disaggregated to species.
- If estimates are based on a sample of the catch, the sampling fraction and strategy should be reported. The strategy for selecting the subsample should be unbiased.
- Coverage of the programme relative to the whole fishery should be reported, as should the scheme for allocating coverage. Coverage also should be wide enough, and apportioned in an unbiased manner among vessels, such that the data are representative of the fishery. (These last two practices are statistical design issues, and should be dealt with empirically.)
- Questionnaires do not work.

This is not an exhaustive list of features required by a programme to collect data on by-catch and discards, but it highlights some major considerations. If ICES is to conduct credible investigations in this important area of ecosystem effects of fishing, the requirement for these programmes must be given priority by ICES Member Countries and by the management agencies it advises.

In this context, the ACME notes with concern that interim data, and in a few cases even reports of completed studies, are not making their way to ICES, and are not being taken note of in assessment working groups. Now that many national laboratories and agencies are implementing programmes to quantify this important aspect of the effects of fishing, ICES should be making full use of the information being collected.

### *Recommendations*

Iceland and Norway have both introduced special regulations to reduce the numbers of discards. Unfortunately, they have no discard monitoring and sampling programmes to evaluate the effects of these regulations.

ICES recommends that its Member Countries collect and analyse data that would make it possible to evaluate the effectiveness of these and other management measures taken to reduce the number of discards.

ICES encourages Member Countries to improve the quality and quantity of data on discards, both to enable an evaluation of the ecosystem effects of fishing and to improve the reliability of single-species and multispecies assessments. ICES encourages Member Countries to implement additional programmes to quantify discards. In addition, improvements should be made to the quality of handling and presenting the data that have been collected.

Finally, ICES encourages Member Countries to make available to ICES the information on by-catches and discards collected nationally and under various international programmes.

### **Reference**

Northridge, S.P. 1996. A review of marine mammal by-catch observer schemes with recommendations for best practice. JNCC Report No. 219, Aberdeen, UK. 42 pp.

### 13.3 Changes in Abundance of Individual Species of Non-target Fish owing to Fishing Activities

#### Request

Item 4.4 of the 1998 Work Programme from the OSPAR Commission: to provide information on changes in abundance of individual species of non-target fish in the maritime area owing to fishing activities.

#### Source of the information presented

The 1997 report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO), and ACFM and ACME deliberations.

#### Status/background information

ICES has accepted the information below as a response to this request. For each region, the material presented here must be taken together with the corresponding contextual information contained in Section 13.1, above.

#### 13.3.1 OSPAR Region I: Arctic Waters

Information on trends in abundance of non-target species in OSPAR Region I is only available separately for the different shelf and fishing areas within this region, especially for the Barents Sea and for the shelf areas around Iceland and Greenland.

All three studies presented here represent boreal systems within OSPAR Region I.

##### 13.3.1.1 East Greenland case study

Before the collapse of the cod stock, the fishery was mainly carried out by large stern trawlers and factory ships, which used otter trawls equipped with very heavy ground gears to protect nets against damage because of the rough fishing grounds. The by-catch was mainly discarded, except for commercially valuable species such as catfishes.

Since 1992, the shrimp fishery has expanded to all traditional fishing areas off West Greenland and, to a lesser extent, off East Greenland (Hvingel *et al.*, 1996a, 1996b). This fishery uses nets with smaller mesh openings than the traditional cod fishery. Data on by-catches and discards are not available, but it seems likely that this increasing fishery on shrimp could negatively influence the recovery of the cod stocks and could also be responsible for a large amount of discards.

#### Description of information source

Rätz (1997) derived abundance indices for non-target species in this area from annual groundfish surveys

covering the shelf areas and the continental slopes of West and East Greenland. These surveys were primarily designed for the assessment of cod stocks; they commenced in 1982 and were carried out by the German R/Vs 'Walther Herwig II' and 'Walther Herwig III'. A standard fishing gear and standard survey design were used to make the catch data comparable over the time period.

This data set was not available to WGECO and could not be further evaluated during the meeting. Therefore, this section only deals with the changes of the non-target species already described by Rätz (1997).

#### Results and interpretation

Long rough dab (*Hippoglossoides platessoides*) (Figure 13.3.1.1.1)

In comparison with the mean indices of the 1980s, the most recent estimates of long rough dab abundance off East Greenland increased by a factor of 2.7, whereas long rough dab became less abundant in West Greenland waters. A similar reversal was observed for the geographical distribution pattern of the stock. In the early 1980s, only one tenth of the total stock was distributed off East Greenland. Since then, the stock has recovered off East Greenland and these waters are now inhabited by the largest portion of the combined stock.

Catfishes (Figures 13.3.1.1.2 and 13.3.1.1.3)

The abundance indices of both common catfish (*Anarhichas lupus*) and spotted catfish (*Anarhichas minor*) show comparable trends over the time period 1982–1996, but the trend is more pronounced for the spotted catfish than for the common catfish. For both species, an increasing trend in abundance off East Greenland and a decreasing trend off West Greenland can be observed, in combination with a geographical shift of the main part of the stock from West to East Greenland.

Starry ray (*Raja radiata*) (Figure 13.3.1.1.4)

The abundance of the starry ray shows no clear trend over the observed time period (1982–1996), as was the case for the other three non-target species. The abundance off East Greenland is more or less stable, but the dramatic decrease in abundance within West Greenland waters should be a reason for concern.

Summarizing the findings, there is an increasing trend in abundance for these four non-target species in East Greenland waters. Climatic changes should not be excluded as being responsible for this development, but it seems more obvious that the collapse of the cod fishery in 1992 and the low fishing effort during the following years have influenced these positive trends.

### 13.3.1.2 The Barents Sea as a case study

This study presents results from the Norwegian bottom trawl survey in the Barents Sea (Jakobsen *et al.*, 1997). Most of the results shown here have not been presented before. The study is separated into two parts: one for the commercially exploited species mainly caught as by-catches in fisheries targeted at other species and, in addition, long rough dab and polar cod (*Boreogadus saidi*). The other part addresses species of little or no commercial value.

#### Species of commercial interest

##### Greenland halibut (*Reinhardtius hippoglossoides*)

The fishery on Greenland halibut increased in the mid-1960s with the introduction of international trawlers. Landings reached as high as 80,000 t in the early 1970s. Since 1992 the fishery has been regulated by allowing only direct fisheries from longliners and gillnetters. Trawl catches are limited to by-catch only. The maximum by-catch as the percentage of Greenland halibut onboard at any time was set at 10 % initially, but this was reduced to 5 % in 1994. Nonetheless, the by-catch from the trawlers constitutes the bulk of the total landings. The survey results presented in Figures 13.3.1.2.1 and 13.3.1.2.2 cover only a fraction of the Greenland halibut stock and the results are probably dominated by a higher proportion of young fish. Within the survey area, the abundance seems to have increased until 1989–1990 and decreased since then. The observed mean length shows the opposite trend, with a minimum in 1990 and an increase since then.

##### Spotted catfish and Common catfish

These two stocks are not assessed within the ICES system. There exists a small direct fishery by longliners, but it is reasonable to believe that most of the catches are taken as by-catches by commercial trawlers. Common catfish are more abundant closer to the coastal areas in the west. The two species show a remarkable similarity in trends for both abundance indices and mean length. Both species reached a minimum in abundance in 1988 and have increased since then. And both species show an increased mean length in the past few years.

##### Halibut (*Hippoglossus hippoglossus*)

The stock size is at a very low level and the observed variations in both abundance indices and mean length can be explained by sampling variability.

##### Long rough dab

This species has been targeted only in experimental fisheries by international trawlers. The by-catch of long rough dab is expected to be quite low. Long rough dab is

quite abundant in the Barents Sea and the trends shown in Figure 13.3.1.2.1 demonstrate some of the dynamics of the stock. A minimum abundance index of 147 million was observed in 1987, while a maximum of 944 million was observed in 1993. The trend in abundance is related to the spatial distribution. Only 60 % of the stock was observed in the eastern subarea (D) in 1987–1988, while as much as 90 % of the stock was observed there in 1993.

##### Polar cod

This species has been targeted by Russian trawlers in some periods. The survey covers only a relatively small proportion of the stock and most of the observed variations may well be related to changes in geographical distribution.

#### Species with little or no commercial value

Trends in abundance indices for twelve species or species groups are presented in Figure 13.3.1.2.3. All the species show large variations. Most of the species seem to exhibit some kind of peak around 1990, which coincides with very high recruitment for cod and haddock that year. Some of the species seem to reach an overall maximum for the period in 1995 or 1996. Some of the species showed a low abundance for the period 1987–1988, when the overall temperature in the water mass was at low levels. Whether the described variations are due to environmentally induced changes in distribution, catchability or availability to the sampling trawl or to changes in the effort in the trawl fleets operating in the Barents Sea is not known.

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Figure 13.3.1.1.1. Abundance indices off West Greenland and East Greenland, and the total for long rough dab.

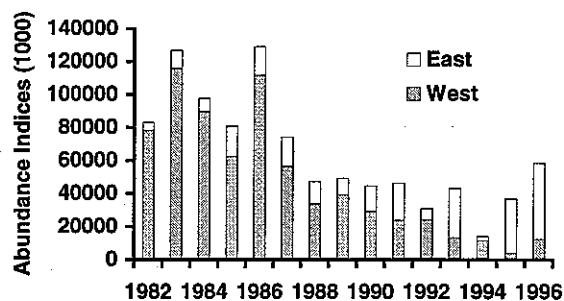


Figure 13.3.1.1.2. Abundance indices off West Greenland and East Greenland, and the total for common catfish.

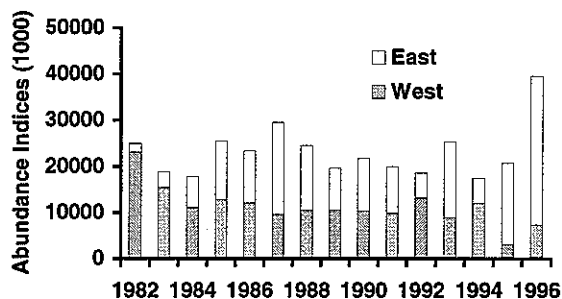


Figure 13.3.1.1.3. Abundance indices off West Greenland and East Greenland, and the total for spotted catfish.

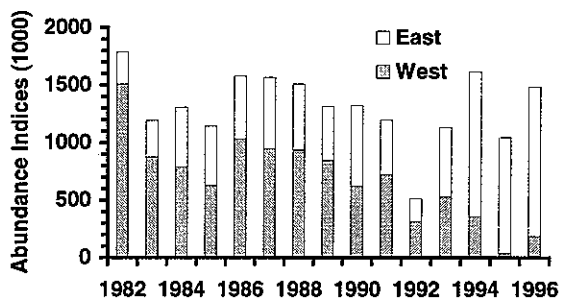
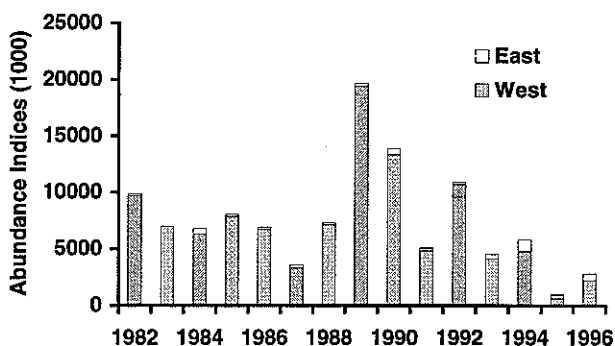
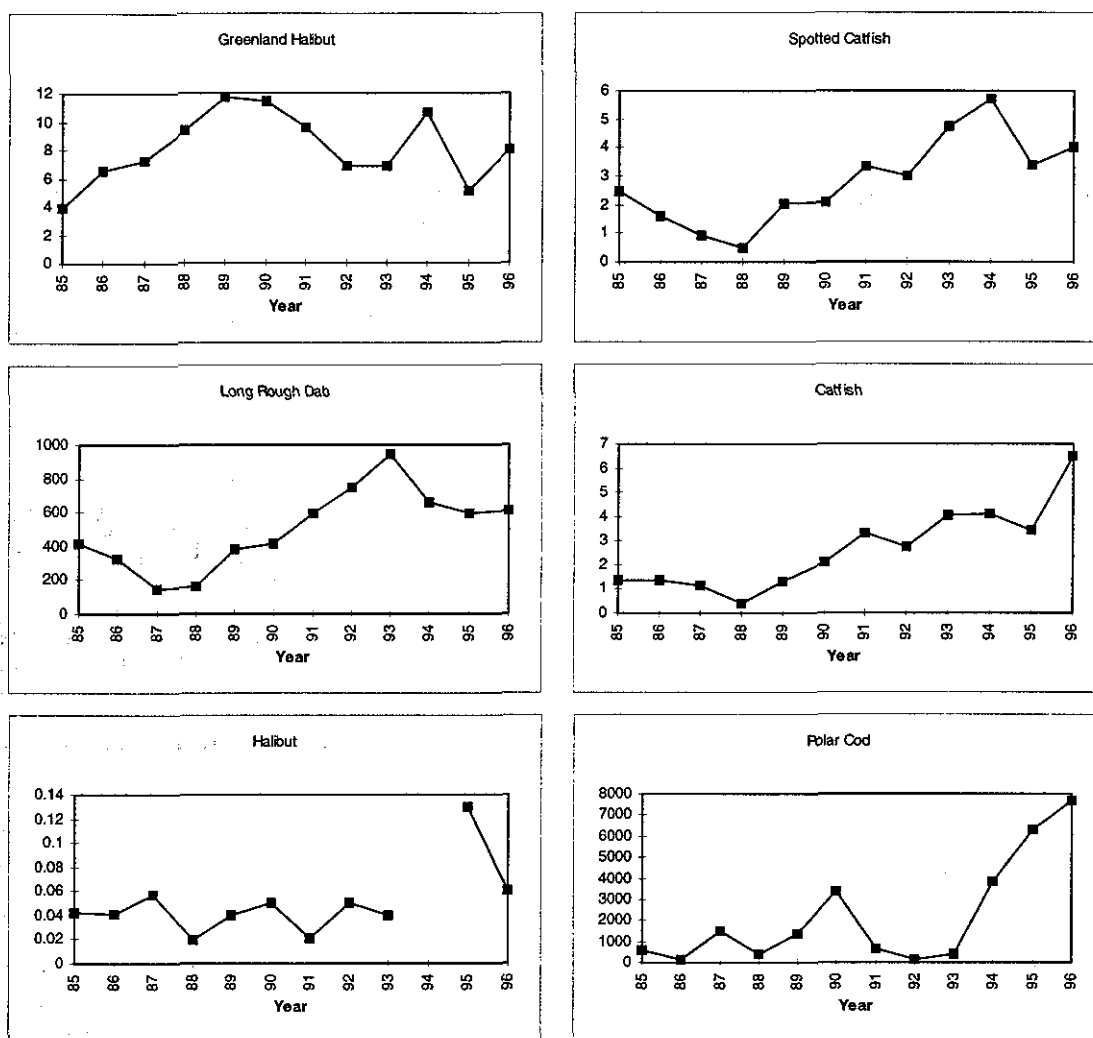


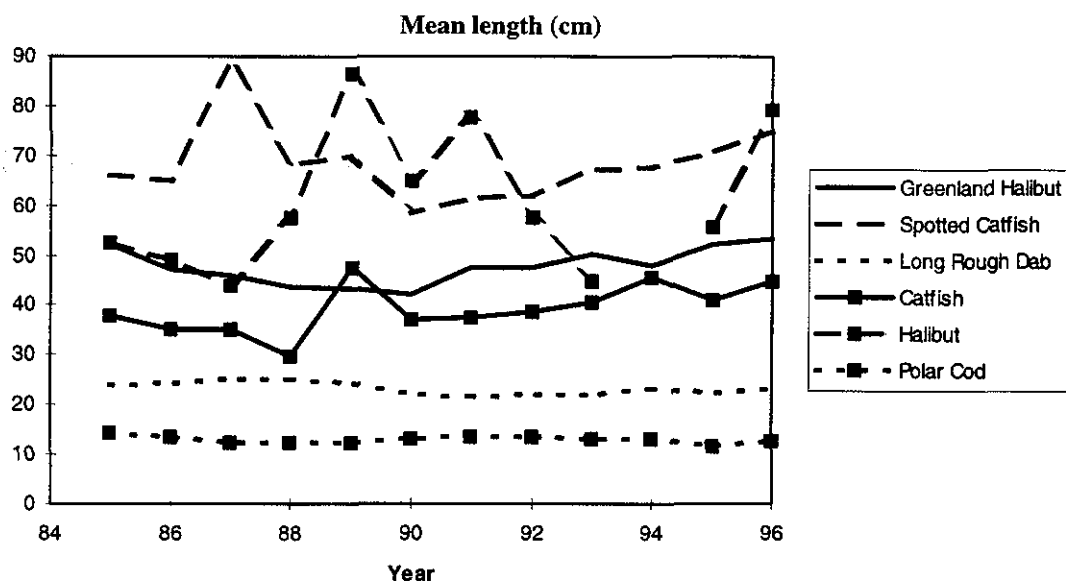
Figure 13.3.1.1.4. Abundance indices off West Greenland and East Greenland, and the total for starry ray.



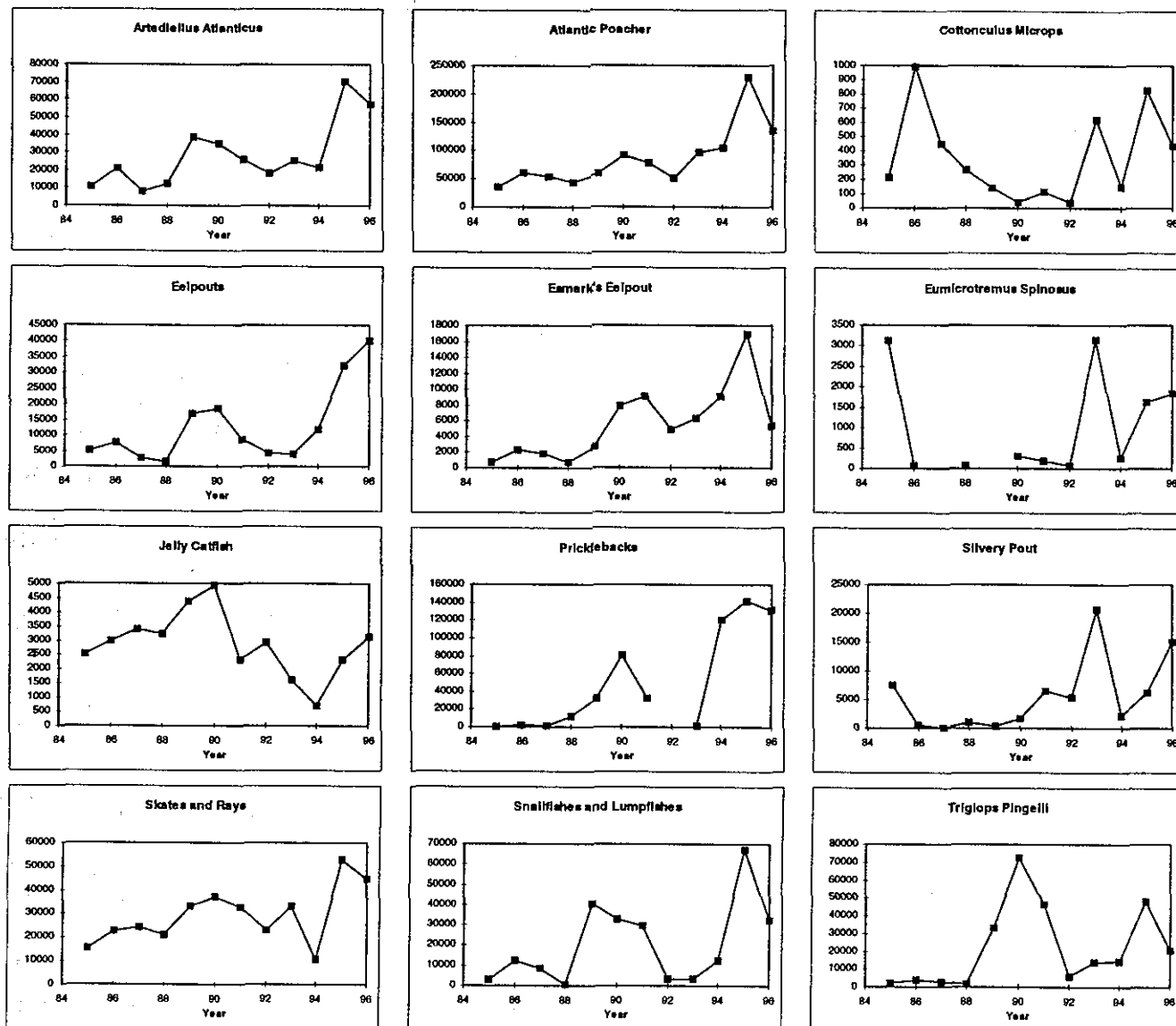
**Figure 13.3.1.2.1.** Trends in abundance indices for some selected species in the Barents Sea bottom trawl survey (Sub-areas A, B, C and D).



**Figure 13.3.1.2.2.** Trends in estimated mean length (cm) for some selected species in the Barents Sea bottom trawl survey (Sub-areas A, B, C and D).



**Figure 13.3.1.2.3.** Trends in abundance indices for twelve species with very low or no commercial value. Indices are estimated using data from the Norwegian bottom trawl survey in the Barents Sea (Sub-areas A, B, C and D).



Several recent studies have been published that show trends in the abundance of fish species which are not the target of specific fisheries. Each study will be presented individually and then an overall summary will be provided. Common names in English and Latin species names for most of the species mentioned in this subsection are listed in Index Tables 13a and 13b based on Knijn *et al.* (1993). These Index Tables can be found at the end of Section 13.3.

#### Rijnsdorp *et al.* (1996)

This study compared the standardized catch rates of a suite of species taken by five gears at two different time periods, an early period (1906–1909) and a more recent period (1990–1995). For the purposes of this OSPAR request, the catch rates for the two most comparable gears used in the different periods, a 20-foot otter trawl (40 mm mesh) used in the early period and the GOV trawl (20 mm mesh) used more recently, were compared (Table 13.3.2.1). Of nineteen non-target species for which trend data were available, eighteen species appear to have decreased in abundance. In making these comparisons, the potential effects of using different fishing gears must be borne in mind. This is particularly emphasized by the relative catch efficiencies of the two gears. The GOV trawl is approximately 40 times more efficient than the 20-foot otter trawl, so all the catches in the earlier period have been multiplied by 40 to make them comparable with catches in the later period. However, to counter this, the mesh size used between 1990 and 1995 was half that used in 1906–1909, thus greater numbers of the smaller fish should have been taken by the gear in the later period.

#### Heessen and Daan (1996)

This study analysed long-term trends, over the period 1970 to 1993, in the International Bottom Trawl Survey data on ten non-target North Sea fish species (Figure 13.3.2.1). Clear increasing trends are apparent in six of these species: starry ray, poor cod (*Trisopterus minutus*), grey gurnard (*Eutrigla gurnardus*), bullrout (*Myoxocephalus scorpius*), long rough dab and lemon sole (*Microstomus kitt*), over the entire period, while common dab shows an increasing trend from 1982. Variation in the abundance of bib (*Trisopterus luscus*) and four-bearded rockling (*Enchelyopus cimbrius*) has been highly variable, but no strong temporal trend is apparent. Variation in the abundance of spurdog (*Squalus acanthias*) has been dominated by the exceptional peak in abundance in 1978. From 1970 to 1976 and from 1981 to 1993, the abundance of this species appears to have fluctuated at relatively low levels.

#### Heessen (1996)

Time series data for forty fish species sampled by the International Bottom Trawl Survey (IBTS) in the North Sea over the period 1970 to 1993 are given in this study. From this, time series data for a further 26 non-target species can be extracted (Figure 13.3.2.2). Many of these species appear and disappear from the data set periodically during this time period. For some species, however, clearer trends are apparent.

#### Walker and Heessen (1996)

This study specifically examines trends in the populations of some of the skate and ray species in the North Sea. Data from the February IBTS over the period 1970 to 1993 for six species of ray are presented (Figure 13.3.2.3). Data are also presented indicating changes in the amount of skates and rays landed from various parts of the North Sea (Figure 13.3.2.4). Few trends are obvious over this time period, with the exception of the steady increase in catch rates of the starry ray. This species is one of the few skates and rays with no commercial value. It is invariably discarded when caught. The most heavily targeted ray, the thornback ray, has all but disappeared from the southeastern North Sea (see Table 13.3.2.1).

#### Greenstreet and Hall (1996)

This study of long-term changes in the groundfish species assemblage structure in the northwestern North Sea presents species abundance data for the periods 1929 to 1953 and 1980 to 1993 for ten species identified as either typifying the species assemblages of three sub-areas, or discriminating between them. Seven of these are non-target species (Figure 13.3.2.5). The data analysed were from the Scottish August Ground Fish Survey for the period 1980 to 1993. The earlier data (1929 to 1953) were collected using the same trawl gear by Scottish fisheries research vessels over the months July to September in each year. Catch rates were adjusted to take into account the differing trawl speeds of the various vessels used. The data suggest that variability in the abundance of non-target species was much greater over the period 1929 to 1953. Long-term trends in the abundance of several of the non-target species are also apparent. Spurdog were more abundant in the early period, and seem still to be declining during the later period. On average, common dab and long rough dab were no more abundant in the 1980s than in the period 1929–1953, however, common dab abundance has increased steadily throughout the recent period. Lemon sole and Norway haddock (*Sebastes viviparus*) are more abundant now than they were during 1929–1953, while grey gurnard are scarcer now than in 1929–1953, although their abundance has increased steadily throughout the period 1980–1993.



**Table 13.3.2.1.** Changes in catch rate (number per hour fishing) of nineteen non-commercial species between 1906–1909 and 1990–1995 (Rijnsdorp *et al.*, 1996).

Species	1906–1909	1990–1995	Direction of trend
Spurdog	10.0	0.1	Decreased
Thornback ray	2.8	< 0.05	Decreased
Poor cod	21.6	6.8	Decreased
Three-bearded rockling	1.2	< 0.05	Decreased
Five-bearded rockling	11.2	0.2	Decreased
Tub gurnard	1.2	0.6	Decreased
Grey gurnard	90.0	13.3	Decreased
Red gurnard	0.4	< 0.05	Decreased
Bullrout	0.8	0.2	Decreased
Hooknose	2.4	1.1	Decreased
Red mullet	0.0	1.0	Increased
Lesser weever	808.8	9.4	Decreased
Greater weever	178.0	0.0	Decreased
Dragonet	74.0	1.7	Decreased
Scaldfish	186.8	0.1	Decreased
Long rough dab	96	0.3	Decreased
Common dab	975.2	176.8	Decreased
Lemon sole	0.4	0.2	Decreased
Solenette	457.6	0.5	Decreased

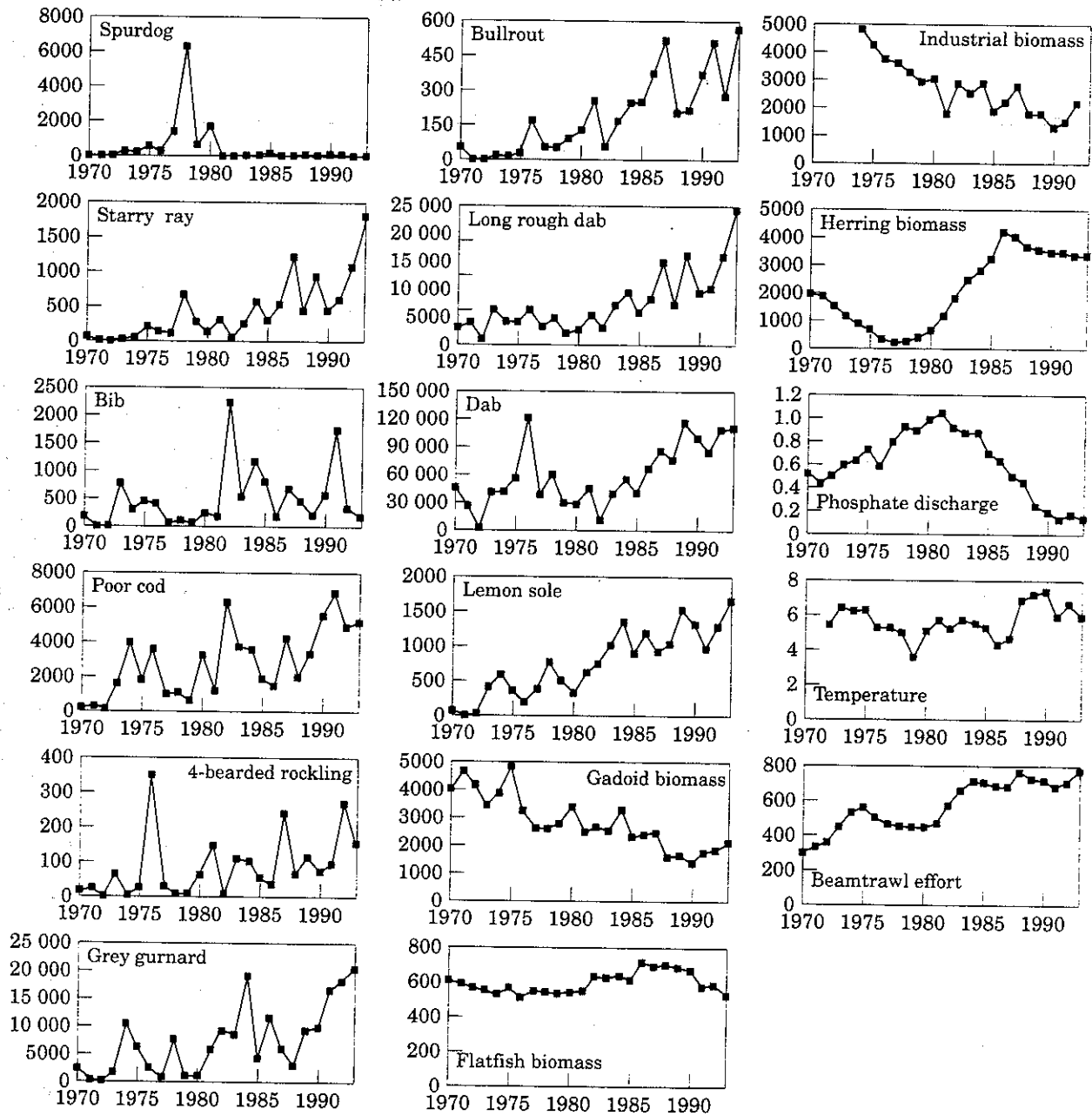
#### Rogers and Millner (1996)

This study presents data on changes in the abundance of eight non-target fish species over the period 1973 to 1995 for three separate regions around the southeast coast of England. The area divisions are shown in Figure 13.3.2.6, while the trends in abundance are given in Figure 13.3.2.7. The data indicate considerable fluctuations in abundance and few consistent trends. In recent years, eelpout appear to have declined in all three areas, while sea snail (*Liparis liparis*) numbers have increased in area 3. Changes in abundance do not seem to correlate closely between areas.

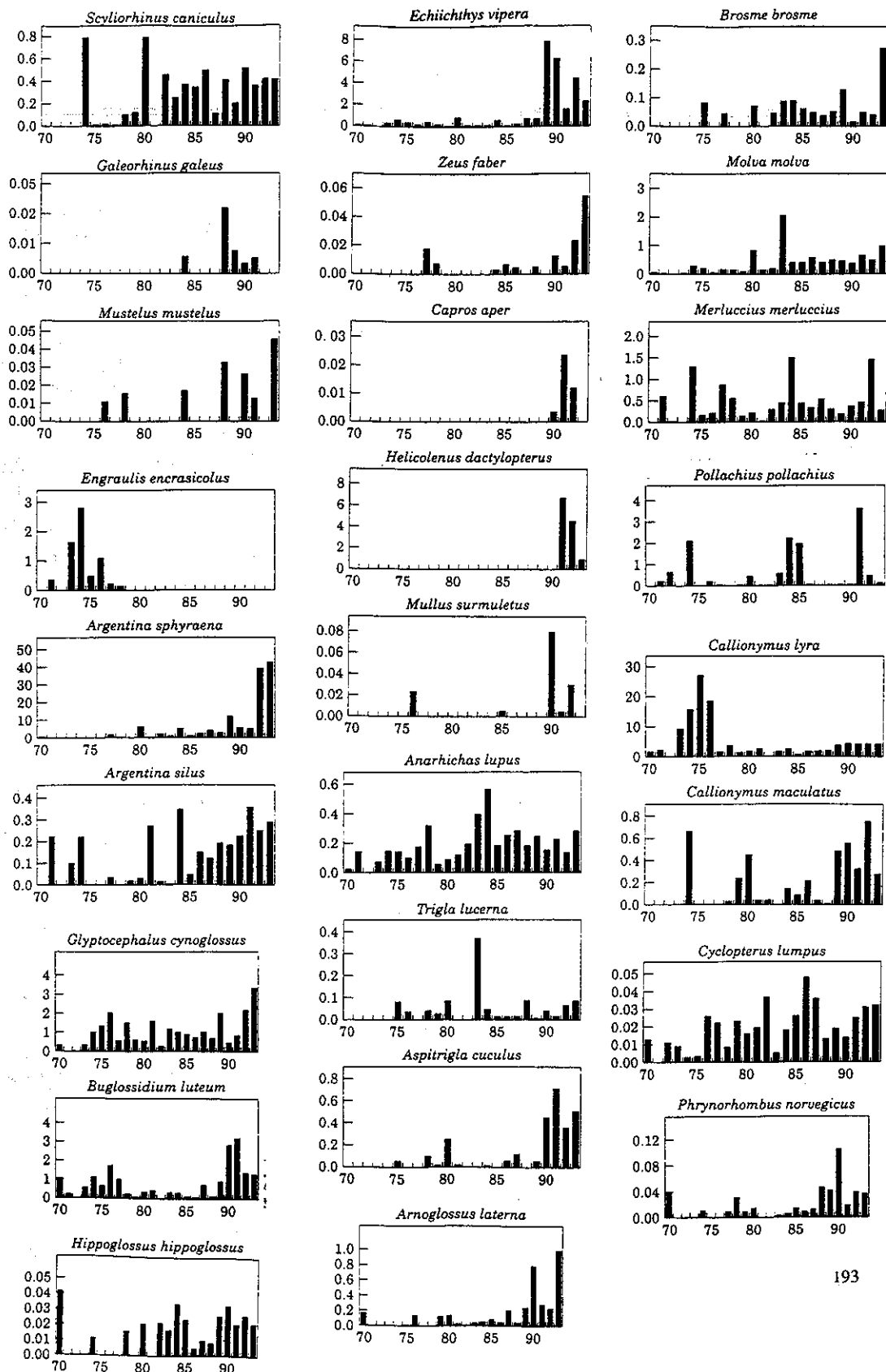
#### Corten and van den Kamp (1996)

This study provides trends in abundance data for twelve species over the period 1970 to 1994 in the southern North Sea (corresponding to ICES Roundfish Areas 5 and 6). The data analysed come from the International Bottom Trawl Survey. Again, considerable fluctuations in abundance were apparent, but few long-term trends were obvious (Figure 13.3.2.8). The exception to this was the clear increase in the abundance of lesser weever (*Echiichthys vipera*) since 1989.

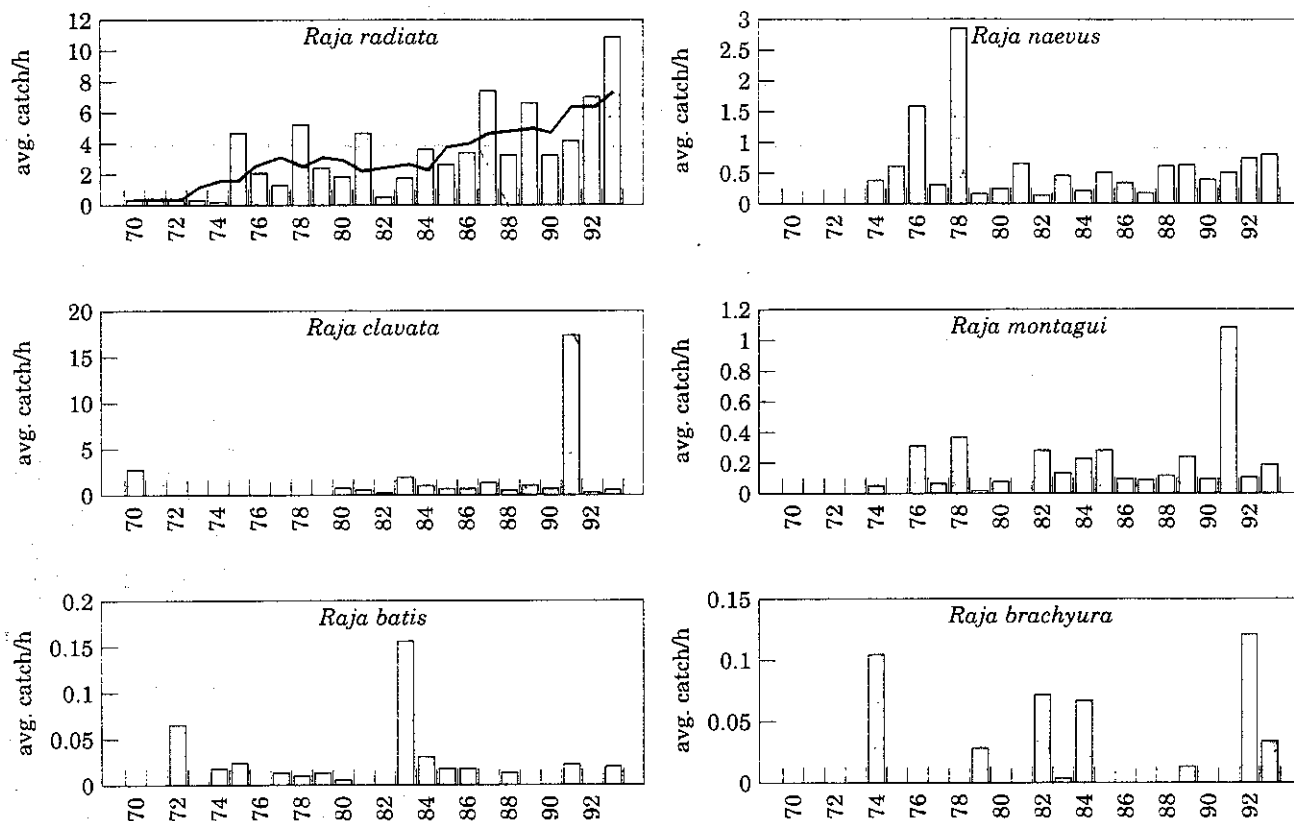
**Figure 13.3.2.1.** Trends in the abundance of ten non-target fish species in the North Sea and time series of four classes of biomass, beam trawl effort, phosphate discharges, and water temperature (Heessen and Daan, 1996).



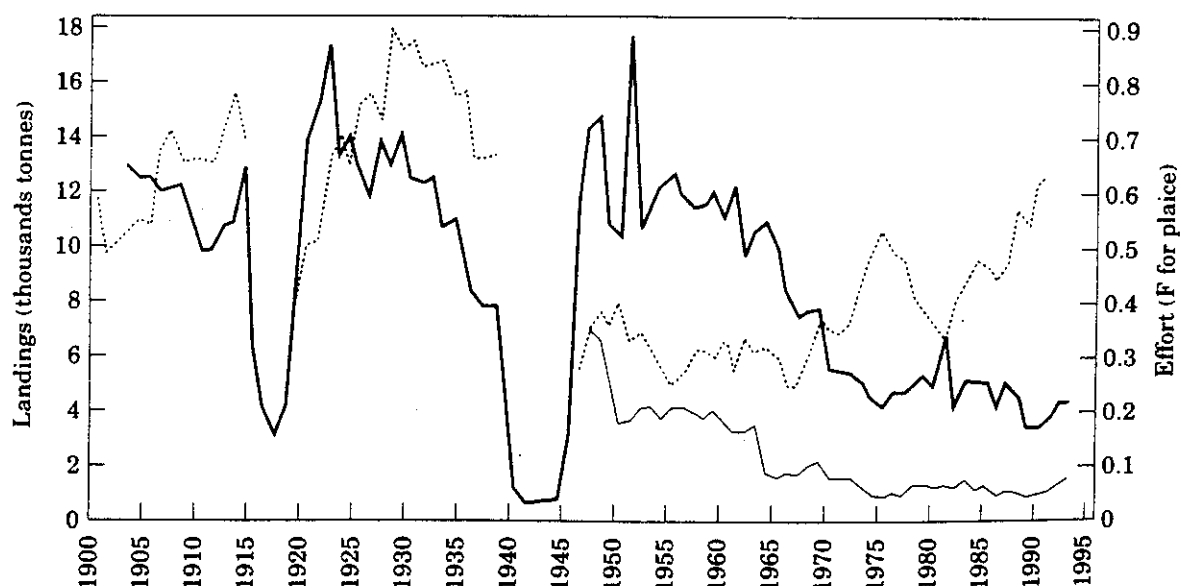
**Figure 13.3.2.2.** Trends in the abundance (measured as average catch per one hour fishing) of 26 non-target fish species in the North Sea (Heessen, 1996).



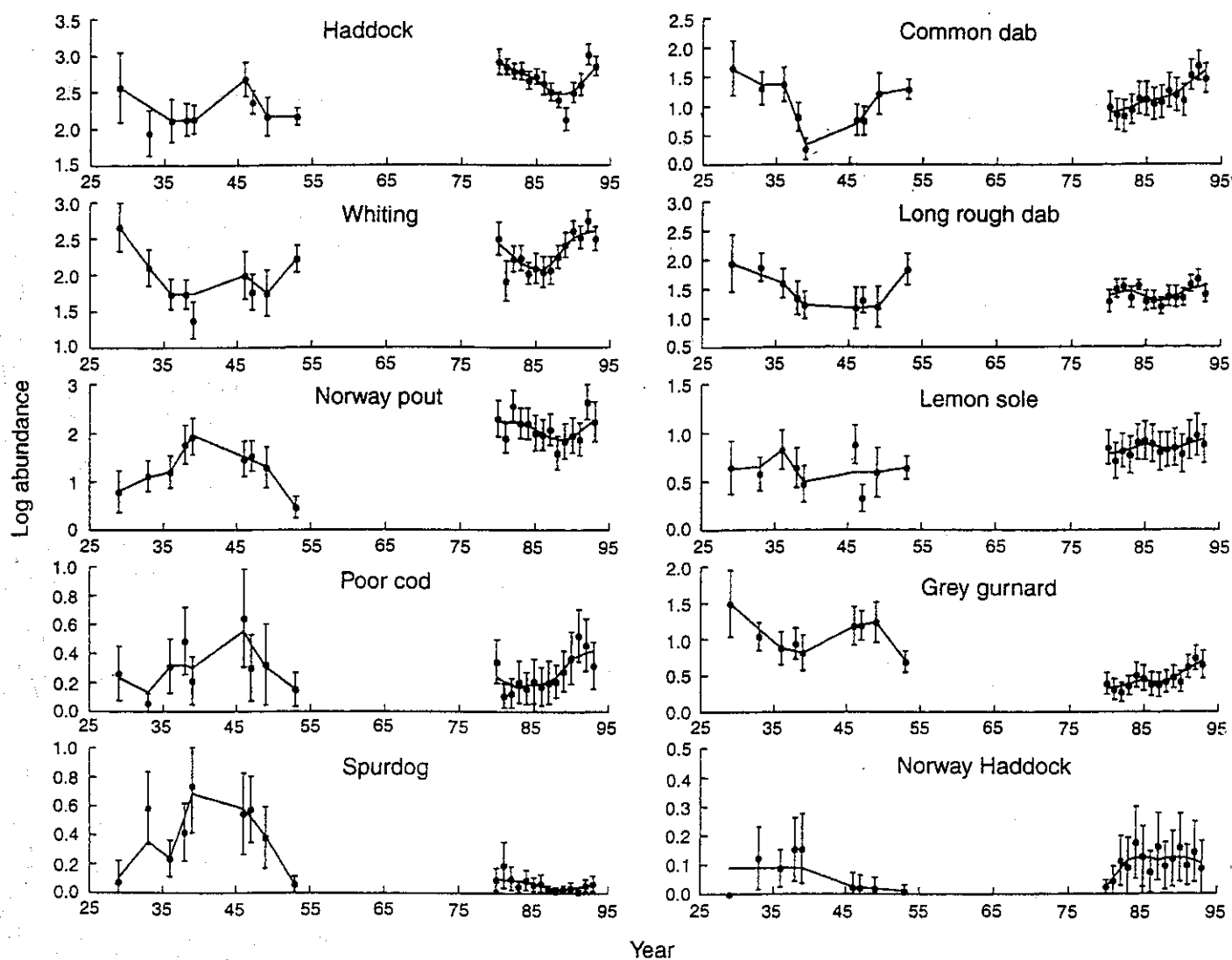
**Figure 13.3.2.3.** Trends in the catch rates of six ray species in the IBTS from 1970 to 1993. A five-year running mean is shown for the starry ray *Raja radiata* (Walker and Heessen, 1996).



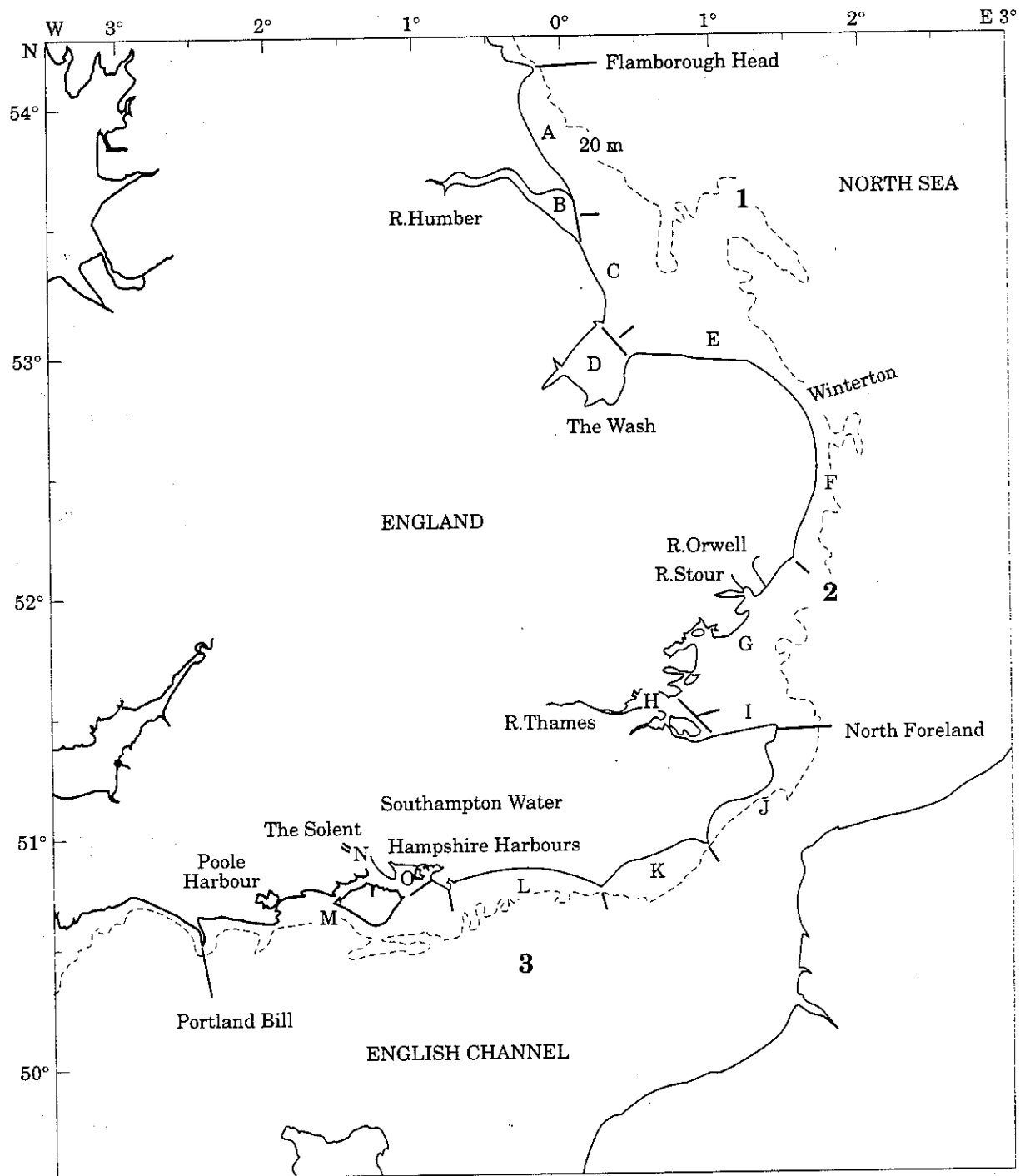
**Figure 13.3.2.4.** Landings of skates and rays from the southern (light solid line) and total North Sea (heavy solid line). The trend in plaice fishing mortality (dashed line) is shown as an index of fishing effort.



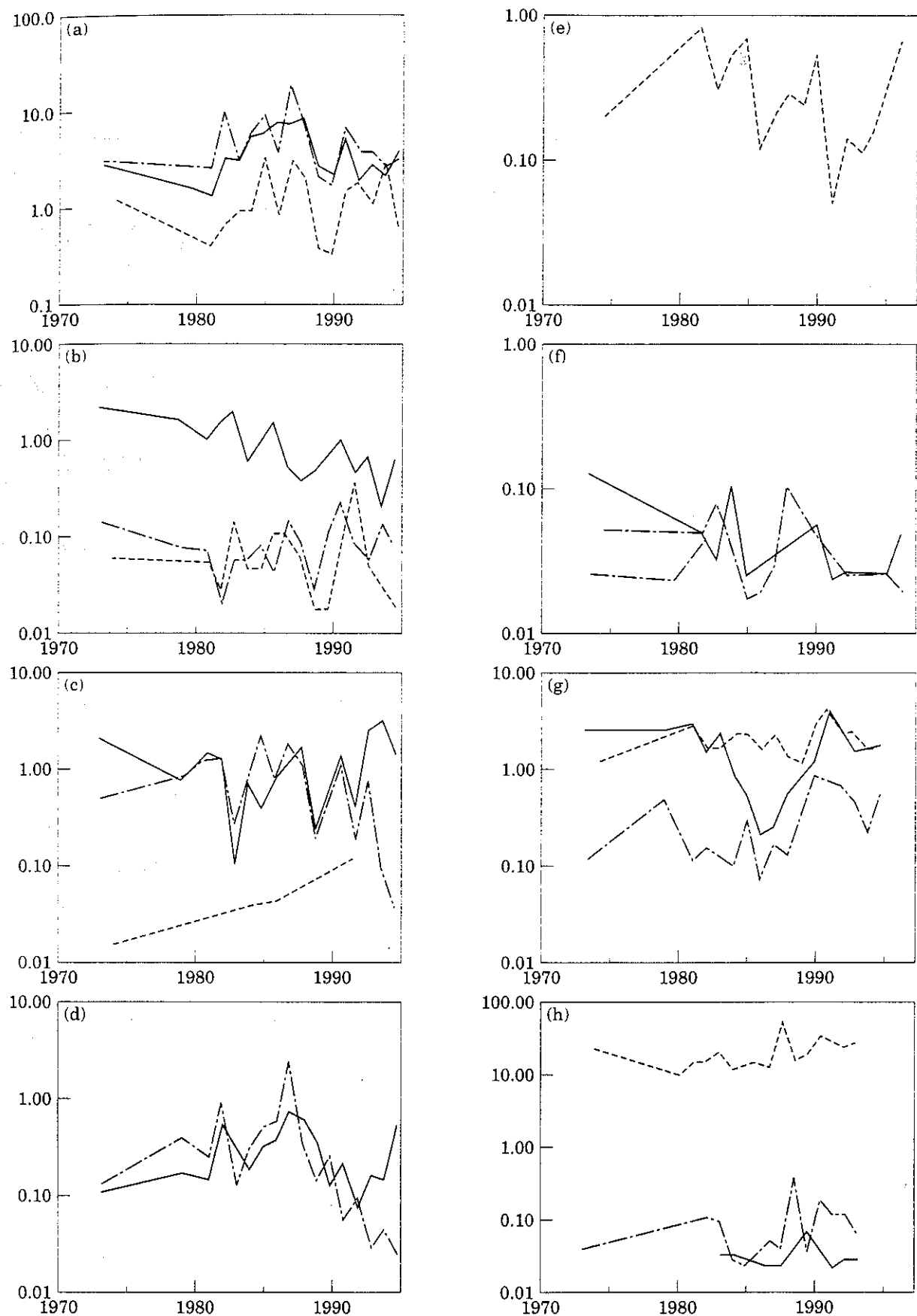
**Figure 13.3.2.5.** Long-term variations in the density of three target species and seven non-target species in the northwestern North Sea. Log catch per hour is shown and the lines are fitted by LOESS smoother.



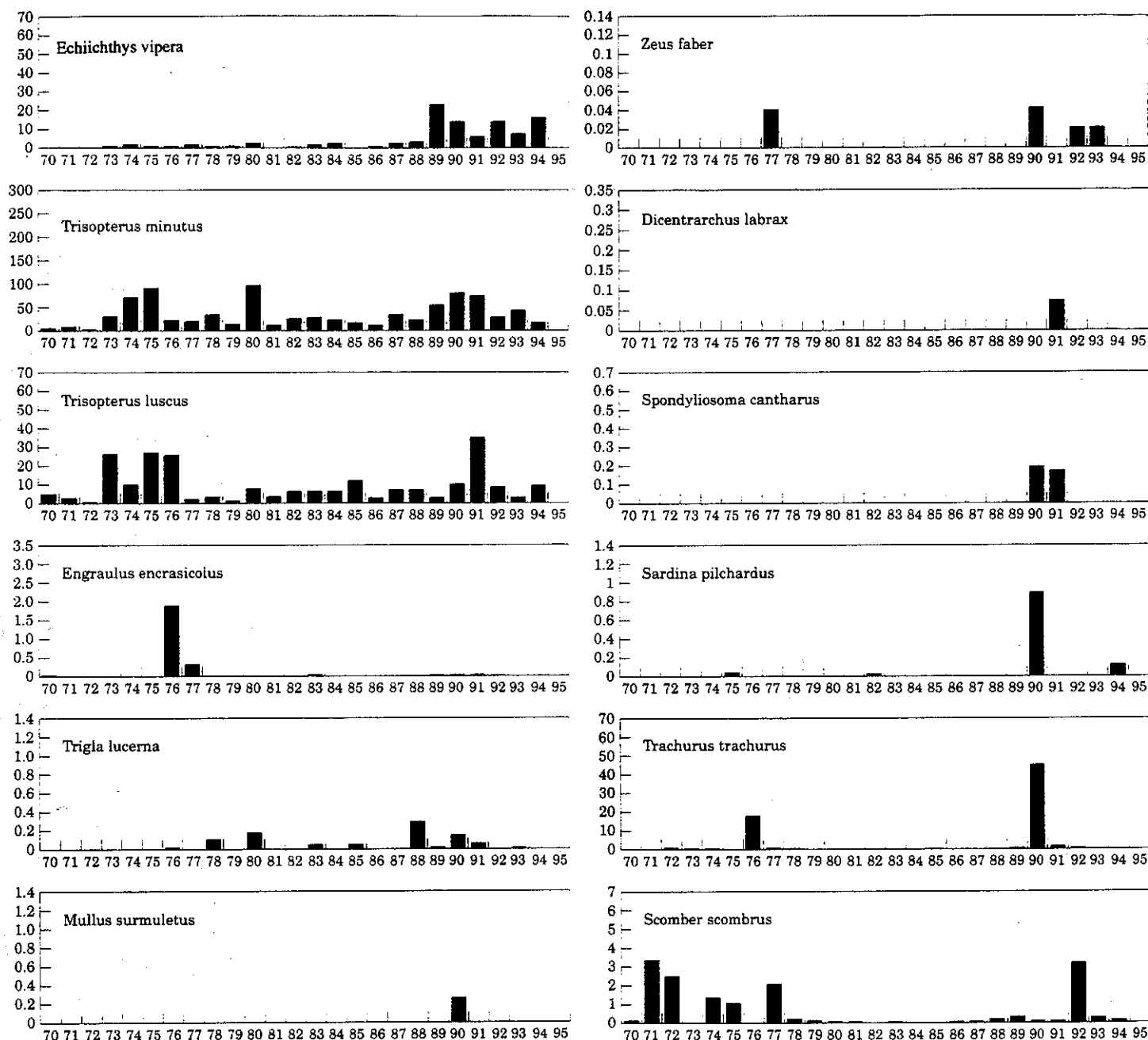
**Figure 13.3.2.6.** Map showing three areas included in the analysis by Rogers and Millner (1996).



**Figure 13.3.2.7.** Mean catch (numbers/1000 m<sup>2</sup>) of hooknose (a), butterfish (b), sea snail (c), eelpout (d), balan wrasse (e), spotted ray (f), lesser weever (g), and solenette (h) in the period 1973–1995 for area 1 (continuous line), area 2 (dot dash line), and area 3 (dashed line) (Rogers and Millner, 1996).



**Figure 13.3.2.8.** Average abundance of twelve fish species (numbers per hour fishing) in the southern North Sea over the period 1970–1994 (Corten and van den Kamp, 1996).



## Conclusions

A considerable amount of effort has been spent collecting groundfish data over many years and analysing them for time series trends. Few trends are apparent and they are rarely consistent for any particular species over different areas and between different studies. Rarely is it possible to relate changes in the abundance of particular species to changes in the fishing regime. However, some consensus across studies is reached with respect to trends in the abundance of the skates and rays. With the exception of the starry ray, declines in the populations of these species seem to be indicated by most studies. The greater weever (*Trachinus draco*) is another species which seems now to be almost absent from the southeastern North Sea. When comparing studies in this way, some regard must be given

to the possible confounding effects of variations in the sampling efficiency between studies. This is due to a number of factors, such as variation in the type of fishing gear, each differing in its catch efficiency for different species, or differences in the way samples are sorted once on board ship.

The effect of fishing on non-target species is clearly of great importance if the ecosystem effects of fishing are going to be given serious consideration. Greenstreet and Hall (1996) demonstrated significant differences between the species composition of the non-target component of the groundfish species assemblage between two time periods, 1929–1953 and 1980–1993. These were not associated with changes in species diversity, except in the area where fishing effort had been highest for the longest



period of time. This suggests that sustained fishing pressure can have effects on the relative abundance of non-target species. This is perhaps an area where more integrated and collaborative research is required.

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### 13.3.3 OSPAR Region III: The Celtic Seas

#### 13.3.3.1 Sampling the non-target species of OSPAR Region III

Time series of otter trawl groundfish surveys in this OSPAR region are described in Table 13.3.3.1 (ICES, 1997) and range in duration from sixteen years (Scottish Groundfish Survey and English Celtic Sea and Western Approaches Survey) to surveys initiated in 1997. None of these data sets were available to WGECCO for the analysis

of time-series trends for specific non-target species. As all the ICES Divisions within this region have experienced fishing effort for many years before the start of these demersal surveys, it is unlikely that trends in decreasing abundance of non-target species could be directly attributed to fishing activity, owing to the lack of data on the abundance of these species before fishing effort began. In the North Sea, analyses of time series from the beginning of the Twentieth Century have been necessary to observe changes in the abundance of non-target species (see Section 13.3.2, above).

#### Abundance of non-target species using data from the International Beam Trawl Surveys

The catch rates of all fish sampled by the English Beam Trawl Surveys in the western waters of the British Isles have been described in reports of the Study Group on Beam Trawl Surveys (ICES, 1994, 1996), and the results have been analysed and discussed by Rogers *et al.* (1998). As explained above, it is unlikely that recent changes will be observed in the abundance of non-target species because of the short duration of this data set. Nevertheless, plots of mean abundance of dab, dragonet (*Callionymus* sp.), dogfish, poor cod, and solenette (*Buglossidium luteum*) are shown in Figure 13.3.3.1. Fluctuations in annual abundance are likely to be attributable to changes in year class strength, even in those species, such as the dogfish, which have been considered to be particularly vulnerable to fishing activity.

#### Disappearance of the common skate from the Irish Sea

The best documented example of the effects of fishing activity on species abundance in this OSPAR region is the decline in abundance of the common skate *Raja batis* (Brander, 1981). In 1902, the skate was reported to be common and was frequently landed by trawl and line fisheries; it was also present as a by-catch in shrimp nets. Abundance began to decline in the 1950s and since the mid-1970s the species has been considered to be rare. During the ten-year period from 1969 to 1979, the landings of skate at Concarneau declined by 82 %. Several aspects of the biology of the skate make it vulnerable to fishing, particularly the slow growth rate and high age at maturity. The rate of egg laying of skate is not known, but can be estimated as 40 per year; the age at first maturity is thought to be 11 years, and the stock is likely to collapse under conditions of total mortality which exceed approximately 0.37. The only effective protection for the skate is probably a complete halt to all types of demersal fishery in which it is caught. As this is unrealistic, it must be accepted that this species, and others like it, will be fished out as a consequence of the exploitation of other demersal fish (Brander, 1981). In areas where relict populations exist, the closure of these areas, which is an approach advised by ICES for the North Sea, should be considered.

**Table 13.3.3.1.** Overview of bottom trawl surveys in 1996 in Sub-areas VI, VII, and VIII and Division XIa.

The Scottish Groundfish Survey in Division VIa (code: SGF6a)

Start: 1981  
 Gear: 36/47 GOV trawl, large rubber bobbins, 20 mm liner  
 Timing: quarter 1 (March since 1986)  
 Target: cod, haddock, whiting, saithe and herring  
 Stratification: by rectangle  
 Depth strata: no  
 No. of hauls: 40  
 Continuation: continued in 1997 and 1998  
 Contact: Andrew Newton, FRS, Aberdeen, Scotland, UK

The Scottish Groundfish Survey in Division VIb (code: SGF6b)

Start: 1985  
 Gear: 48 ft Aberdeen trawl, large rubber bobbins, 35 mm cover  
 Timing: quarter 3 (September)  
 Target: haddock  
 Stratification: by rectangle  
 Depth strata: no  
 No. of hauls: 45  
 Continuation: continued in 1997 and 1998  
 Contact: Andrew Newton, FRS, Aberdeen, Scotland, UK

The Scottish Mackerel Recruit Survey (code: SMR)

Start: 1985  
 Gear: 36/47 GOV trawl, large rubber bobbins, 20 mm liner  
 Timing: quarter 4 (November/December)  
 Target: mackerel only until 1995 (cod, haddock, whiting, herring added in 1996)  
 Stratification: by rectangle  
 Depth strata: no  
 No. of hauls: 50  
 Continuation: Long-term, area redefined in 1997  
 Contact: Andrew Newton, FRS, Aberdeen, Scotland, UK

West Coast Groundfish Survey (Code: WCGS)

Start: 1990  
 Gear: commercial trawl, rockhoppers, 20 mm liner  
 Timing: quarter 4 (October/November)  
 Stratification: by rectangle  
 Depth strata: no  
 Target: commercial species  
 No. of hauls: 71  
 Continuation: 1997 and 1998  
 Contact: Paul Connolly, FRC, Dublin, Ireland

The Irish Sea Recruit Survey (code: ISRS)

Start: 1983  
 Gear: 3-bridle otter trawl, close contact groundgear, 20 mm codend  
 Timing: quarter 2 (June) and quarter 3 (September)  
 Stratification: fixed stations  
 Depth strata: no  
 Target: cod, whiting, haddock and plaice  
 No. of hauls: 28 each survey  
 Continuation: continued in 1997, will be discontinued in 1998  
 Contact: Paul Connolly FRC, Dublin, Ireland

The Irish Sea and Celtic Sea Groundfish Survey (code: ISCSGS)

Start: 1997  
 Gear: 20/25 GOV trawl, standard groundgear, 20 mm liner  
 Timing: quarter 4 (October)  
 Stratification: by rectangle  
 Depth strata: <50, 50–100, 100–150, 150–200, 200–250, >250 m  
 Target: commercially important species  
 No. of hauls: 50  
 Continuation: will commence in 1997  
 Contact: Paul Connolly, FRC, Dublin, Ireland.

The West and South Coast of Ireland Recruit Survey (code: WSCRS)

Start: 1992  
 Gear: dual purpose otter trawl, medium bobbins, 20 mm codend  
 Timing: quarter 3 (July)  
 Stratification: by depth, fixed stations  
 Depth strata: no  
 Target: inshore juvenile fish  
 No. of hauls: 74  
 Continuation: continued in 1997  
 Contact: Paul Connolly, FRC, Dublin, Ireland

The Celtic Sea and Western Approaches Groundfish Survey (code: CSGF)

Start: 1981  
 Gear: Portuguese high-headline trawl, medium rubber bobbins, 20 mm liner, tickler  
 Timing: quarter 1 (March)  
 Stratification: by depth and latitude  
 Depth strata: 0–89, 90–114, 115–139, 140–179, >180 m  
 Target: mackerel and commercially important species  
 No. of hauls: 75  
 Continuation: continuing in 1997 and 1998  
 Contact: John Nichols, CEFAS, Lowestoft, England, UK

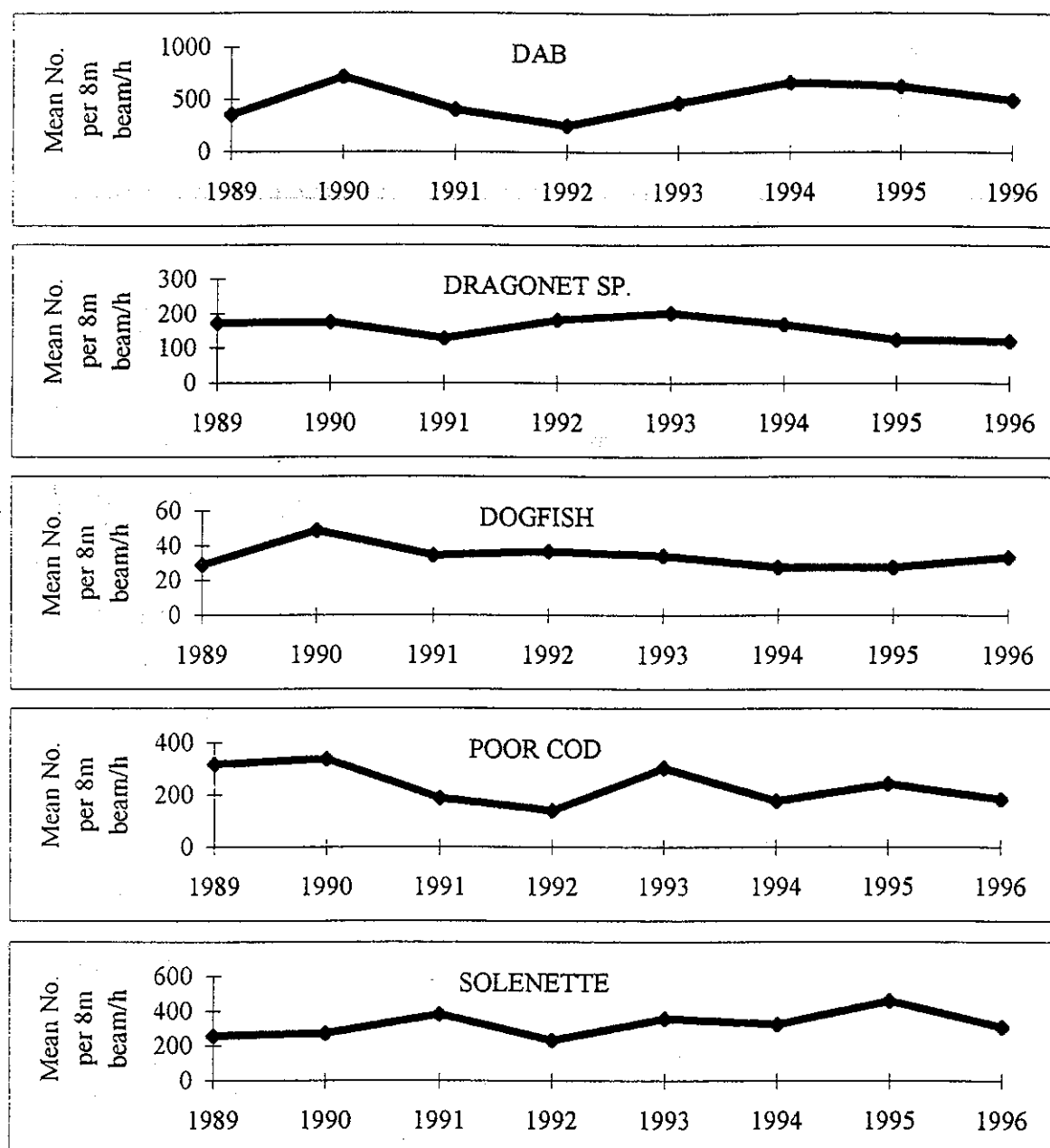
The Northern Ireland Groundfish Survey in Division VIIa (code: NIGFS)

Start: 1991  
 Gear: Otter trawl, rockhoppers, 20 mm liner  
 Timing: quarter 1 (March), quarter 3/4 (September/October) (also June 1991–1994)  
 Stratification: by depth, area and bottom type (7), fixed stations  
 Depth strata: <50 m, ≥50 m  
 Target: commercially important species  
 No. of hauls: 45 per survey  
 Continuation: March and September surveys to be continued in 1997 and 1998  
 Contact: Mike Armstrong, DANI, Belfast, Northern Ireland, UK

The German Survey in the Western Waters (code: GSWW)

Start: 1991  
 Gear: 36/47 GOV trawl, standard groundgear, 20 mm liner  
 Timing: quarter 2 (April)  
 Stratification: by rectangle  
 Depth strata: no  
 Target: commercially important species  
 No. of hauls: 40  
 Continuation: continued in 1996, not in 1997; again in 1998 as part of the mackerel/horse mackerel egg surveys and subsequently triennial  
 Contact: Nils Hammer, BFA-ISH, Hamburg, Germany

**Figure 13.3.3.1.** Mean annual abundance of selected non-target species sampled during the Beam Trawl Survey in Division VIIa (1989–1996).



## References

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

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Rogers, S.I., Rijnsdorp, A.D., Damm, U., and Vanhee, W. 1998. Demersal fish populations in the coastal waters of the UK and continental N.W. Europe from beam trawl survey data collected from 1990 to 1995. *Journal of Sea Research*, 39: 79–102.

## 13.3.4 OSPAR Region IV: Bay of Biscay and Iberian Coast

A stratified random bottom trawl survey has been carried out by Spain since 1983 (except in 1987) along the Cantabrian Sea continental shelf and in Galician waters. Each year, from 50 to 70 tows of 30 minutes' duration have been taken in depths from 30 m to 500 m.

Abundances and biomasses of all species have been recorded. Details of the survey procedures are summarized in Cardador *et al.* (1997) and Olaso *et al.* (1998).

The difficulties in applying the data from this survey to this OSPAR request are the same as for other studies and other OSPAR regions. The surveys quantify the trends in non-target species under the standard assumptions of constant catchability, etc. However, to determine that part of the trends caused by fishing, rather than by random variation or environmental forcing, requires both good data on the intensity and distribution of fishing over the period of the survey, and the ability to partition variance among possible causal factors. Neither the data on fishing nor the ability to determine causation of changes are available. Also, as with other OSPAR regions, the surveyed areas have been fished for many years prior to the survey. Therefore, the largest effects of fishing on the non-target species may have happened before the survey began in recent decades.

From the Spanish survey, data on thornback ray (*Raja clavata*) and lesser spotted dogfish (*Scyliorhinus canicula*) were converted into indices of abundance over time. These species were selected because they were

thought to be likely to show direct impacts of fishing on the abundance of non-target species (ICES, 1996). Other potentially vulnerable species highlighted in that reference occurred in numbers too low for the estimation of trends in biomass over time.

## Results

### Lesser spotted dogfish

As shown in Figure 13.3.4.1a, the abundance was variable throughout the 1980s, and then showed a consistent decline from 1990 to 1995. This trend seems to have reversed in the two most recent years, with the biomass in 1997 very similar to the biomass in 1990.

### Thornback ray

Figure 13.3.4.1b shows that the abundance appears stable in the early 1980s, although between 1986 and 1990 there is substantial variation in abundance. These fluctuations are so great that it is unlikely that they are caused by changes in the total size of the population, but must reflect changes in distribution or in local availability to the fishery. As with lesser spotted dogfish, there appears to be a declining trend from 1990 to 1995, with a substantial increase in the two most recent years.

Figure 13.3.4.1a. Time series of biomass indices (kg/30 min.) and standard error for dogfish in the Cantabrian Sea.

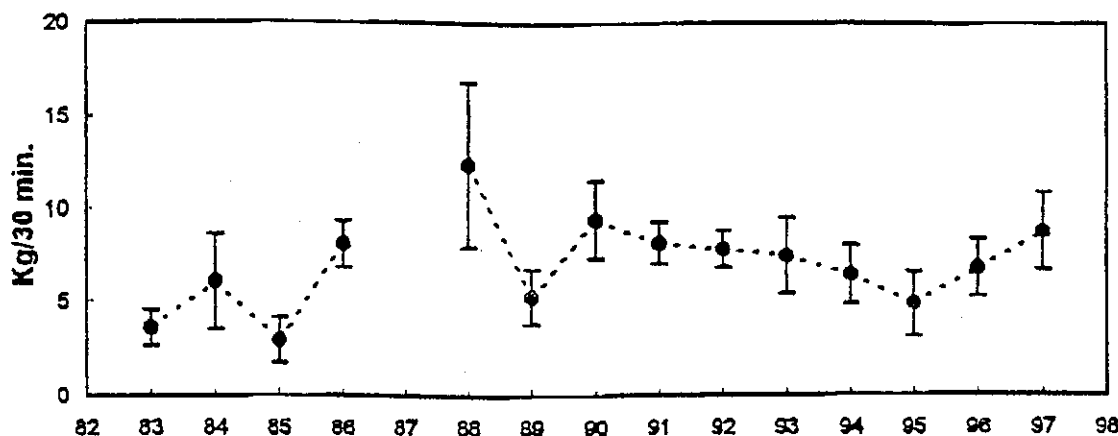
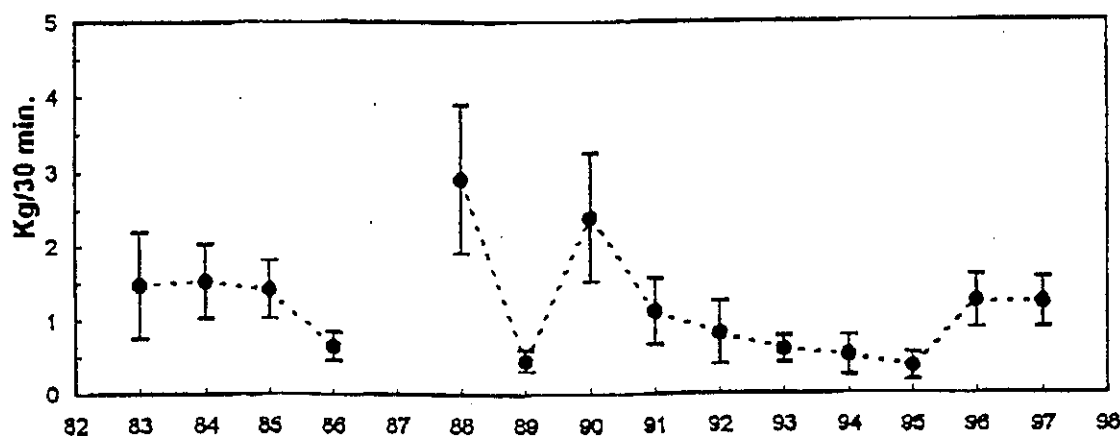


Figure 13.3.4.1b. Time series of biomass indices (kg/30 min.) and standard error for thornback ray in the Cantabrian Sea.



## Discussion

Both of these species are most abundant in the shallowest depth stratum (30–100 m). Trawling is prohibited by regulation inshore of the 100 m contour, although some illegal fishing is known to occur in these areas. Even if these species are taken in fisheries, their survival rate when discarded has been measured at over 90 % (Kaiser and Spencer, 1995). Because of the prohibition on fishing and the high survivorship if discarded, neither species would be expected to show noteworthy changes in abundance due to trawling. In that context, it is interesting that the declining trend in abundance through the 1990s was reversed the second year after artificial barriers were placed inside the 100 m contour, to make illegal trawling much more difficult to conduct. This could have two interpretations. Possibly illegal fishing was so intense that the populations of these two elasmobranchs were unable to maintain themselves with the level of by-catch mortality they were suffering. Alternatively, it is reported that the survey gear has had to align trawl tracks in proximity to the artificial reefs which were constructed, in order to sample in the stratum. If the artificial reefs are attractive to these species, then catch rates might be elevated in the years since 1994. In that case, changes in availability to the fishing gear continue to influence greatly the trend over time. Only more detailed work at fine spatial scales can begin to disentangle these possible causes of the observed pattern.

There is also a trawl survey conducted in waters from 20–500 m off the coast of Portugal. This survey began using a stratified random design in the mid-1980s, although the design changed to fixed stations in 1989. Gomes and Serrão (1997) reported on the multispecies assemblages identified by various multivariate analyses of the catch data. Because these assemblages were defined in part (usually primarily) by species for which there are directed fisheries, they are not appropriate for looking at changes in the abundance of non-target species. However, the data series may contain such information for some non-targeted species. The data should be analysed further, first for non-target species which are sampled reliably by the survey, and then for trends in the abundance of those species.

## References

- Cardador, F., Sánchez, F., Pereiro, F.J., Borges, M.F., Caramelo, A.M., Azevedo, M., Silva, A., Pérez, N., Martins, M.M., Olaso, I., Pestana, G., Trujillo, V., and Fernandez, A. 1997. Groundfish surveys in the Atlantic Iberian waters (ICES Divisions VIIIc and IXa): history and perspectives. ICES CM 1997/Y:08. 36 pp.
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- Kaiser, M.J., and Spencer, B.E. 1995. Survival of by-catch from a beam trawl. *Marine Ecology Progress Series*, 126: 31–38.
- Olaso, I., Velasco, F., and Pérez, N. 1998. Importance of discarded blue whiting (*Micromesistius poutassou*) in the diet of lesser spotted dogfish (*Scylliorhinus canicula*) in the Cantabrian Sea. *ICES Journal of Marine Science*, 55: 331–341.

### 13.3.5 OSPAR Region V: The Wider Atlantic

In the areas where the groundfish and midwater pelagic fisheries operate in OSPAR Region V, there are only a few opportunistic and partial surveys. No time series data exist on the abundance of the target species, let alone non-target species. However, records of exploratory cruises by several countries in the 1970s provide potentially important information on species composition, size composition, and ages of many species, particularly in the areas around Porcupine and Rockall Banks (Ehrich, 1983; Gordon, 1986; Gordon and Duncan, 1987; Merritt *et al.*, 1991a, 1991b). Repeat surveys of these areas, or full monitoring of catch composition from fisheries, might give data useful for comparative analyses in the light of the recent expansion of these fisheries. A series of surveys on Rockall Bank has been conducted since the mid-1980s. Unfortunately, neither data nor scientific reports from that series have been published yet, nor have they been made available to ICES. Concerns about the possible over-exploitation of some target species in this region (see Section 13.1.5, above) suggest that any species taken regularly as by-catch in these fisheries may also be impacted. However, there are no data to shed light on this matter.

For the traditional fisheries around the Azores, Portugal initiated a series of surveys starting in 1993. These surveys focused on collecting abundance and biological information on the species targeted by the fisheries, for comparison with data from surveys in the early 1980s (ICES, 1996). Information was recorded on a few non-target species in the recent surveys, but it has not been established as to whether comparable data can be recovered from the earlier surveys.

There are no surveys which would give information about changes in the abundance of species taken as by-catch in the fisheries for tunas and tuna-like fishes, although the by-catch monitoring programme which has been implemented may give useful information if continued for an adequate period. There are particular concerns about impacts on some species of sharks, and there are discussions among

FAO, ICES, and ICCAT with regard to expanding programmes to monitor the status of sharks in several areas, including the mid-Atlantic.

## References

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- Gordon, J.D.M., Merrett, N.R., Bergstad, O.A., and Swan, S.C. 1996. A comparison of the deep-water fish assemblages of the Rockall Trough and Porcupine Sea Bight, eastern North Atlantic: continental slope to rise. Journal of Fish Biology, 49 (suppl. A): 217–238.
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- Merrett, N.R., Haedrich, R.L., Gordon, J.D.M., and Stehmann, M. 1991b. Deep demersal fish assemblage structure in the Porcupine Sea Bight (Eastern North Atlantic): results of single warp trawling at lower slope to abyssal soundings. Journal of the Marine Biological Association UK, 71: 359–373.

Index Table 13.a. Index of common names.

English Name	Scientific Name
Alfonsine	<i>Beryx decadacylus</i>
Anchovy	<i>Engraulis encrasicolus</i>
Anglerfish	<i>Lophius piscatorius</i>
Balan wrasse	<i>Labrus bergylta</i>
Bass	<i>Dicentrarchus labrax</i>
Bib	<i>Trisopterus luscus</i>
Black scabbard fish	<i>Aphanopus carbo</i>
Black sea-bream	<i>Spondylusoma cantharus</i>
Blonde ray	<i>Raja brachyura</i>
Blue whiting	<i>Micromesistius poutassou</i>
Bluemouth	<i>Helicolenus dactylopterus</i>
Boar-fish	<i>Capros aper</i>
Brill	<i>Scophthalmus rhombus</i>
Bullrout	<i>Myoxocephalus scorpius</i>
Butterfish	<i>Pholis gunellus</i>
Capelin	<i>Mallatus villosus</i>
Cod	<i>Gadus morhua</i>
Common catfish	<i>Anarchichas lupus</i>
Common dab	<i>Limanda limanda</i>
Common dragonet	<i>Callionymus lyra</i>
Common skate	<i>Raja batis</i>
Conger eel	<i>Conger conger</i>
Cuckoo ray	<i>Raja naevus</i>
Dragonet	<i>Callionymus</i> sp.
Eelpout	<i>Zoarces viviparus</i>
Five-bearded rockling	<i>Ciliata mustela</i>
Four-bearded rockling	<i>Enchelyopus cimbrius</i>
Greater argentine	<i>Argentina silus</i>
Greater forkbeard	<i>Phycis blennoides</i>
Greater weever	<i>Trachinus draco</i>
Greenland halibut	<i>Reinhardtius hippoglossoides</i>
Grey gurnard	<i>Eutrigla gurnardus</i>
Gulper shark	<i>Centropgorus granulosus</i>
Haddock	<i>Melanogrammus aeglefinus</i>
Hake	<i>Merluccius merluccius</i>
Halibut	<i>Hippoglossus hippoglossus</i>
Herring	<i>Clupea harengus</i>
Hooknose	<i>Agonus cataphractus</i>
Horse mackerel	<i>Trachurus trachurus</i>
John Dory	<i>Zeus faber</i>
Kitefin shark	<i>Dalatias licha</i>
Lemon sole	<i>Microstomus kitt</i>
Lesser argentine	<i>Argentina sphyraena</i>
Lesser-spotted dogfish	<i>Scyliorhinus canicula</i>
Lesser weever	<i>Echiichthys vipera</i>
Ling	<i>Molva molva</i>
Long rough dab	<i>Hippoglossoides platessoides</i>
Lumpsucker	<i>Cyclopterus lumpus</i>
Mackerel	<i>Scomber scombrus</i>
Megrim	<i>Lepidorhombus whiffiagonis</i>
Norway haddock	<i>Sebastes viviparus</i>
Norway pout	<i>Trisopterus esmarki</i>
Norwegian herring	<i>Clupea harengus</i>
Norwegian topknot	<i>Phrynorhombus norvegicus</i>
Ocean perch	<i>Sebastes marinus</i>
Orange roughy	<i>Hoplostethus atlanticus</i>
Pilchard	<i>Sardina pilchardus</i>

English Name	Scientific Name
Plaice	<i>Pleuronectes platessa</i>
Polar cod	<i>Boreogadus saidi</i>
Pollack	<i>Pollachius pollachius</i>
Poor cod	<i>Trisopterus minutus</i>
Red gurnard	<i>Aspitrigla cuculus</i>
Red mullet	<i>Mullus surmuletus</i>
Red sea-bream	<i>Pagellus bogaraveo</i>
Redfish	<i>Sebastes mentella</i>
Rough grenadier	<i>Macrourus berglax</i>
Roundnose grenadier	<i>Coryphaenoides rupestris</i>
Saithe	<i>Pollachius virens</i>
Sandeel	<i>Ammodytes</i> sp.
Scaldfish	<i>Arnoglossus laterna</i>
Sea snail	<i>Liparis liparis</i>
Smooth hound	<i>Mustelus mustelus</i>
Sole	<i>Solea solea</i>
Solenette	<i>Buglossidium luteum</i>
Spotted catfish	<i>Anarchichas minor</i>
Spotted dragonet	<i>Callionymus maculatus</i>
Spotted ray	<i>Raja montagui</i>
Sprat	<i>Sprattus sprattus</i>
Spurdog	<i>Squalus acanthias</i>
Starry ray	<i>Raja radiata</i>
Thornback ray	<i>Raja clavata</i>
Three-bearded rockling	<i>Gaidropsarus vulgaris</i>
Tope	<i>Galeorhinus galeus</i>
Tub gurnard	<i>Trigla lucerna</i>
Turbot	<i>Scophthalmus maximus</i>
Tusk	<i>Brosme brosme</i>
Whiting	<i>Merlangius merlangus</i>
Witch	<i>Glyptocephalus cynoglossus</i>
Wreckfish	<i>Polyprion americanus</i>

Index Table 13.b. Index of species names.

Scientific Name	English Name
<i>Agonus cataphractus</i>	Hooknose
<i>Ammodytes</i> sp.	Sandeel
<i>Anarchichas lupus</i>	Common catfish
<i>Anarchichas minor</i>	Spotted catfish
<i>Aphanopus carbo</i>	Black scabbard fish
<i>Argentina silus</i>	Greater argentine
<i>Argentina sphyraena</i>	Lesser argentine
<i>Arnoglossus laterna</i>	Scaldfish
<i>Aspitrigla cuculus</i>	Red gurnard
<i>Beryx decadacylus</i>	Alfonsine
<i>Beryx splendens</i>	Golden-eye perch
<i>Boreogadus saiidi</i>	Polar cod
<i>Brosme brosme</i>	Tusk
<i>Buglossidium luteum</i>	Solenette
<i>Callionymus lyra</i>	Common dragonet
<i>Callionymus maculatus</i>	Spotted dragonet
<i>Callionymus</i> sp.	Dragonet
<i>Capros aper</i>	Boar-fish
<i>Centropogon granulosus</i>	Gulper shark
<i>Ciliata mustela</i>	Five-bearded rockling
<i>Clupea harengus</i>	Herring
<i>Clupea harengus</i>	Norwegian herring
<i>Conger conger</i>	Conger eel
<i>Coryphaenoides rupestris</i>	Roundnose grenadier
<i>Cyclopterus lumpus</i>	Lumpsucker
<i>Dalatias licha</i>	Kitefin shark
<i>Dicentrarchus labrax</i>	Bass
<i>Echiichthys vipera</i>	Lesser weever
<i>Enchelyopus cimbrius</i>	Four-bearded rockling
<i>Engraulis encrasicolus</i>	Anchovy
<i>Eutrigla gurnardus</i>	Grey gurnard
<i>Gadus morhua</i>	Cod
<i>Gaidropsarus vulgaris</i>	Three-bearded rockling
<i>Galeorhinus galeus</i>	Tope
<i>Glyptocephalus cynoglossus</i>	Witch
<i>Helicolenus dactylopterus</i>	Bluemouth
<i>Hippoglossoides platessoides</i>	Long rough dab
<i>Hippoglossus hippoglossus</i>	Halibut
<i>Hoplostethus atlanticus</i>	Orange roughy
<i>Labrus bergylta</i>	Balan wrasse
<i>Lepidorhombus whiffiagonis</i>	Megrim
<i>Limanda limanda</i>	Common dab
<i>Liparis liparis</i>	Sea snail
<i>Lophius piscatorius</i>	Anglerfish
<i>Macrourus berglax</i>	Rough grenadier
<i>Mallotus villosus</i>	Capelin
<i>Melanogrammus aeglefinus</i>	Haddock
<i>Merlangius merlangus</i>	Whiting
<i>Merluccius merluccius</i>	Hake
<i>Micromesistius poutassou</i>	Blue whiting
<i>Microstomus kitt</i>	Lemon sole
<i>Molva molva</i>	Ling
<i>Mullus surmuletus</i>	Red mullet
<i>Mustelus mustelus</i>	Smooth hound
<i>Myoxocephalus scorpius</i>	Bullrout
<i>Pagellus bogaraveo</i>	Red sea-bream
<i>Pholis gunellus</i>	Butterfish

Scientific Name	English Name
<i>Phrynorhombus norvegicus</i>	Norwegian topknot
<i>Phycis blennoides</i>	Greater forkbeard
<i>Pleuronectes platessa</i>	Plaice
<i>Pollachius pollachius</i>	Pollack
<i>Pollachius virens</i>	Saithe
<i>Polyprion americanus</i>	Wreckfish
<i>Raja batis</i>	Common skate
<i>Raja brachyura</i>	Blonde ray
<i>Raja clavata</i>	Thornback ray
<i>Raja montagui</i>	Spotted ray
<i>Raja naevus</i>	Cuckoo ray
<i>Raja radiata</i>	Starry ray
<i>Reinhardtius hippoglossoides</i>	Greenland halibut
<i>Sardina pilchardus</i>	Pilchard
<i>Scomber scombrus</i>	Mackerel
<i>Scophthalmus maximus</i>	Turbot
<i>Scophthalmus rhombus</i>	Brill
<i>Scyliorhinus canicula</i>	Lesser-spotted dogfish
<i>Sebastes marinus</i>	Ocean perch
<i>Sebastes mentella</i>	Redfish
<i>Sebastes viviparus</i>	Norway haddock
<i>Solea solea</i>	Sole
<i>Spondylusoma cantharus</i>	Black sea-bream
<i>Sprattus sprattus</i>	Sprat
<i>Squalus acanthias</i>	Spurdog
<i>Trachinus draco</i>	Greater weever
<i>Trachurus trachurus</i>	Horse mackerel
<i>Trigla lucerna</i>	Tub gurnard
<i>Trisopterus esmarki</i>	Norway pout
<i>Trisopterus luscus</i>	Bib
<i>Trisopterus minutus</i>	Poor cod
<i>Zeus faber</i>	John Dory
<i>Zoarces viviparus</i>	Eelpout



### **13.4 Measures of Evaluating Ecosystem Effects of Fishing Activities**

#### *Request*

There is no specific request; this is part of on-going ICES work on developing management reference points, and evaluating direct and indirect effects of fishing on marine ecosystems.

#### *Source of the information presented*

The 1997 report of the Working Group on Ecosystem Effects of Fishing Activities (WGECO) and ACME deliberations.

#### *Status/background information*

The ACME reviewed and accepted initial information from WGECO concerning the behaviour of community metrics, the effects of fishing on the level of predation on benthos by fish, and reference points that include ecosystem considerations. This material is contained in the sub-sections below.

#### **13.4.1 Theory on the behaviour of community metrics: introductory discussion**

##### **Integrating information on North Sea assemblages from different surveys**

Long time series of marine fisheries survey data are important, and can provide invaluable insights into the temporal changes in fish populations. Some of these data sets extend back almost to the beginning of the Twentieth Century, and relate to a period which experienced less extensive fishing impact than the present day. The purposes of this section are to identify all such time series of data available for the North Sea ecosystem which relate specifically to fisheries assemblages, and to evaluate the feasibility of concatenating separate series into larger, integrated data sets.

##### **Research vessel surveys**

The earliest research vessel survey data from the North Sea that have been computerized relate to a series by Dutch and English vessels from 1906 to 1909. For each haul, the numbers of the larger fish species caught are available for 10-cm groups, and some information on the bottom fauna was also recorded. The distribution of the fishing stations in the North Sea was fairly uneven, but the southeastern North Sea was well covered by most surveys.

The Scottish August Groundfish Survey (AGFS) has taken place each year since 1980. The gear used in the AGFS is identical to that used by the Marine Laboratory, Aberdeen, in groundfish survey work extending back to

the 1920s. The data for the entire AGFS, as well as for the months of July to September from the earlier survey covering the years 1929 to 1953, are available in electronic format.

One of the longest North Sea time series for demersal species is provided by the first quarter International Bottom Trawl Survey (IBTS), which began in 1960/1961 and has been carried out annually in February since 1965. Initially, the survey coverage was restricted to the southern and central North Sea, but the coverage was extended in 1974. Since 1969, the Skagerrak and Kattegat have also been sampled. For the period 1970–1982, the records are incomplete and many data are still in paper format, stored in different laboratories.

By 1988, a number of countries which border the North Sea had developed beam trawl surveys which targeted different age ranges of flatfish. Collation and analysis of some of the data derived from these surveys were initially focused on the North Sea and eastern Channel, but during the early 1990s all surveys in Sub-areas IV and VII were included (ICES, 1991). Six surveys were modified following recommendations of the Study Group on Beam Trawl Surveys to develop a more standardized sampling protocol (ICES, 1994).

##### **Problems with combining gear catches**

Different fishing gears vary in catch efficiency for different sizes of fish, and this is the main problem encountered when comparing catch data collected between one survey and another. As all fishing gears are selective and the catchabilities of fish at size vary, standardization to a common swept area of the gear does not resolve all the problems, and relative catchabilities can only be obtained when all gears are fished simultaneously on the same ground.

The selectivity and catchability of a demersal trawl are influenced by the way that the net is rigged, the type of ground gear, the length of the towing warp and otter trawl sweeps, the mesh size in the codend, and the speed at which the gear is towed. In addition, the ground over which the gear is towed and the tidal conditions during towing will also influence catch rates of fish (ICES, 1996). Also, the fishing characteristics of the different research vessels used in the collection of the data can be important. The absolute numbers of each species sampled may be affected by differences in the areas swept by the fishing gear as a result of vessels of differing horse power towing the gear at different speeds.

It is possible that beam trawls of the same design but of different widths may not show a linear relationship in their catch rates of all demersal species, and that the use of different attachments (chain mat, flip-up ropes, etc.) will also affect the gear efficiency. During surveys in 1990 and 1991, catch ratios of dab, sole and plaice between gears were consistently different (ICES, 1993),

suggesting that it was not possible to derive raising factors to convert the catch numbers of one gear into those of another gear.

Some institutes have changed the duration of tows, causing potential problems with the analysis of long-term trends in species diversity. The issue of the sample size dependence of some community metrics is particularly relevant when it comes to considering the effect of variation in trawl duration.

A final consideration in comparing different data sets, again related to the problems of sampling effort dependence, is the possible consequences of the protocols used for handling catches once they are brought on board the vessel. It is frequently impossible to sort and handle every single fish in a large catch. Subsampling is necessary. Straightforward proportional division of the catch, sorting one fraction and discarding the rest, effectively reduces sampling effort at that station, but it also reduces the probability of finding rare fish. In biodiversity studies, it is important that not only the haul duration is standardized, but also that the entire catch is sorted in such a way as to obtain a reasonably accurate estimate of even the rarest species.

Comparative gear trials suggest that the levels of standardization currently used in the IBTS database are important to ensure that catch data are collected in a similar way, and that catch comparisons between gears are important. They also illustrate how difficult it is to combine catches from similar gears. The relative catchability of different species by different gears is an important consideration in deciding which species to include in the species suite in a particular analysis. To combine the catch rates of fish between, for example, the otter trawl catches of the IBTS and the beam trawl catches of the beam trawl surveys in the North Sea, will require extensive species-by-species knowledge of the selectivity of each gear. These data are not yet available.

### **Comparative analyses of faunal assemblages**

Results of analyses calculating and contrasting diverse multivariate metrics and modelling approaches for several different data sets on fish assemblages were reviewed. Additional studies in the primary literature were also considered. The results and comparisons led to several conclusions with regard to the use of various metrics in studies of community structure, function, and dynamics.

It is clear from the analyses conducted or reviewed by WGECO that the many metrics available for quantifying community-level change (whether in response to fishing or not) are partially redundant and overlapping in content. However, few are identical, and using multiple metrics in a single study may be necessary to obtain a usefully complete picture of how the community or ecosystem is changing, whatever the cause.

Unfortunately, the more metrics which are applied in a single study, the greater the interpretational flexibility in terms of what receives attention when conclusions are drawn. This interpretational flexibility can undermine the goals of empiricism and objectivity, which are much of the original motivation for applying the quantitative metrics.

The results underscored a second problem as well. The signals we are trying to pick up (fishing effects on communities or ecosystems) have been in place for a long time. Over the past few decades, when the large majority of all our data (especially on non-target species) has been collected, fishing intensity has been consistently high in most fisheries. Therefore, the variation in fishing intensity (which is the statistical contrast in the process) has been low compared to the longer-term changes from no fishing to the situation at, say, the end of the third quarter of this century. The variation, or contrast, in fishing over recent decades is also low compared to the overarching effects of environmental forcing on fundamental biological processes such as recruitment, growth, and distribution in space. Clearly, it will be very difficult to detect empirically and unambiguously the effects of fishing on community-level properties of marine ecosystems using time series which start in the 1960s or later, or short-term data sets collected many years apart.

These difficulties must be added to the differential progress that has been made in developing the theoretical framework for these community metrics, compared to the application of metrics to data sets. The work reviewed in this section suggests that sound theory linking fishing to size spectrum metrics is developing the most rapidly. There are potentially promising developments in mass balance model approaches also, but it is too early to evaluate the strengths and difficulties of the dynamic version of this approach, which will be necessary to link changes in metrics specifically to fishing. Progress on providing a theoretical framework for linking fishing to community metrics is the slowest in the traditional multivariate metrics of community structure, where applications are the most numerous. Given the roots of many of these metrics in statistics or general community ecology, it is not surprising that there is little theory specifically linking fishing to the multivariate metrics, but theory is essential. Without the theoretical framework, we cannot have unambiguous predictions of how particular metrics should change with fishing, and without such unambiguous predictions, we cannot test hypotheses rigorously. It is not even clear whether all metrics can make usefully specific predictions, either in the sense of being rejectable with data sets which are realistic to collect, or in the sense of being able to make clearly different predictions from different views of fishing effects on an ecosystem property.

In addition to the conclusion that the scientific community must focus more on the theory linking fishing to ecosystem properties, the ACME also concluded that a

better picture of the state of the unfished marine systems needs to be developed. Once we do have a theory we can use with confidence, we will need to have confidence in our picture of the starting conditions as well. Those two things together will enable us to make significant strides in evaluating the effects of fishing on communities, and testing diverse scientific hypotheses.

In our hindcasting, however, we will have to be vigilant to the dangers of circularity. If we develop our view of the unfished state through reliance on a particular theoretical approach to community dynamics, chosen for preference or convenience, analysis methods drawn from the same framework run the risk of merely confirming that we can get back as fishing effects whatever ideas we put into reconstructing the theoretical unfished state.

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### 13.4.2 Fishing effects on the level of predation on benthos by fish

#### Introduction

The effect of fisheries on target species is well established (Pope and Macer, 1996; Rijnsdorp and Millner, 1996). The direct mortality of benthos arising from the use of heavy gears has also been demonstrated (e.g., Lindeboom and de Groot, 1997). The indirect effects of fishing on the ecosystem are less clear. These include changes in nutrient cycling caused by physical disturbance of the sediment-water interface and the addition of offal to the system, the consequences of the continued removal of fixed carbon from the marine to the terrestrial system, and the changes in the food chain arising from manipulation of the density and size structure of the target populations.

In the North Sea, populations of the benthic-feeding gadoids, cod, haddock and whiting, have declined over the last thirty years or so (Pope and Macer, 1996; Serchuk *et al.*, 1996). During the same period, populations of long rough dab, common dab and lemon sole have increased (Heessen and Daan, 1996). Therefore, there is a need to provide an assessment of the consequences of these changes for the level of predation pressure exerted on the benthos.

## Methodology and approach

For eight benthic-feeding species, time series of abundances (biomass) and feeding rate data were available. These species are: sole (*Solea solea*), plaice (*Pleuronectes platessa*), cod (*Gadus morhua*), haddock (*Melanogrammus aeglefinus*), whiting (*Merlangius merlangus*), long rough dab (*Hippoglossoides platessoides*), common dab (*Limanda limanda*), and lemon sole (*Microstomus kitt*).

Virtual Population Analysis (VPA) estimates of total stock biomass and stock biomass at age for sole, plaice, cod, haddock, and whiting were available from the Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak (ICES, 1998). Temporal trends in stock biomass prior to the time period covered by the assessment group were taken from Pope and Macer (1996) for cod, haddock and whiting, and from Rijnsdorp and Millner (1996) for plaice. Several corrections were applied, to account for discarding and other factors. Data on diet and ration were taken from the ICES stomach sampling programmes in 1981 and 1991.

## Results

Trends in the biomass of the stocks were reconstructed back to 1920. Most species of benthic feeders underwent substantial changes in abundance over this period, with different species peaking in different intervals. In the most recent two decades, roundfish have generally been at relatively low biomasses, whereas flatfish, particularly dab, have been at high biomasses.

The diets of plaice and sole were rather similar. Fish contributed less than 10 % in the diet. Of the benthic invertebrates, annelids were dominant in the diets of both plaice and sole. The proportion of annelids decreased with the age of the fish, whereas that of echinoderms and molluscs increased. The proportion of crustaceans was relatively small (< 10 %). The diet of dab was dominated by echinoderms (mainly ophiuroids) and, to a lesser extent, crustaceans. Long rough dab mainly took crustaceans and annelids. The diet of lemon sole was dominated by annelids. The proportion of benthic invertebrates in the diet of the gadoids showed a clear decline from about 50 % in age 1 to less than 10–30 % in age group 6+. Haddock showed the highest relative proportion of benthic invertebrates in their diet, followed by cod and whiting. Within the benthos, there was some difference in the prey types selected. Cod and whiting mainly took crustaceans, whereas haddock took crustaceans, echinoderms, and annelids in nearly equal proportions. Molluscs were insignificant in all three gadoid diets.

The quantity of benthos consumed by flatfish in the North Sea has increased steadily over the period 1970–1993. While the quantity removed by plaice has remained fairly constant, the expansion of the common

dab population has led to dab being the dominant consumer in this group. There are no clear trends in the composition of the fauna consumed, with year-to-year variations in the relative abundance of the various species of flatfish driving the patterns of prey taken.

Gadoid consumption of food peaked in the late 1960s at around 39 million tonnes annually, of which approximately 17 million tonnes was benthos. Quantities of food consumed declined from 1968 to 1989 in line with the decreasing stock sizes. The values for gadoid consumption of benthos calculated here are higher than those estimated by the Multispecies Assessment Working Group.

The benthic prey were dominated by crustaceans (~55 % of the benthic food, by weight), but annelids and echinoderms also made important contributions to the diet (~20 % each). Molluscs represented a minor dietary component (~5 %) (Figure 13.4.2.1). Comparisons of the taxonomic composition of the consumed material must be regarded as suggestive only, given the underlying assumptions. Since species- and size-specific diets were assumed to be constant, variation in the total consumption of different prey types was caused entirely by variation in the abundance and size composition of the gadoids over the time series. Given this, the consumption of echinoderms during the period from the Second World War to the early 1960s was only about half that for the remainder of the time series (Figure 13.4.2.1). The echinoderms were mainly replaced by crustaceans.

#### Combined effects and implications for system productivity

The data sets considered here allow an evaluation of the fish predation pressure over the period 1970–1993 for eight of the most abundant demersal species. In spite of the declines in target fish populations (gadoids and plaice), the overall level of predation on the benthos has increased from around 23 million tonnes per year in 1970 to 29 million tonnes per year in 1993 (Figure 13.4.2.2). In addition, there are indications of a decrease in the proportion of crustaceans and molluscs in the diet and an increase in the importance of echinoderms (primarily ophiuroids).

To evaluate the potential effect on the benthic fauna from the predation by the eight fish species, estimates were compared of production and consumption. Due to the limited extent of published data, it was possible to make this comparison only at a coarse taxonomic level. Production estimates for North Sea benthos were derived from two sources: Christensen (1995, using his groups 'echinoderms', 'polychaetes', and 'other macrobenthos') and Greenstreet *et al.* (1997). Based on these data, the predation rate of fish on benthos appears high (20–45 % of the benthic production being used by the eight fish species included in this analysis, see Table 13.4.2.1).

**Table 13.4.2.1.** Annual production rates of benthic invertebrates and consumption rates by fish in the North Sea (1970–1993). The consumption-to-production ratio is given in the right column.

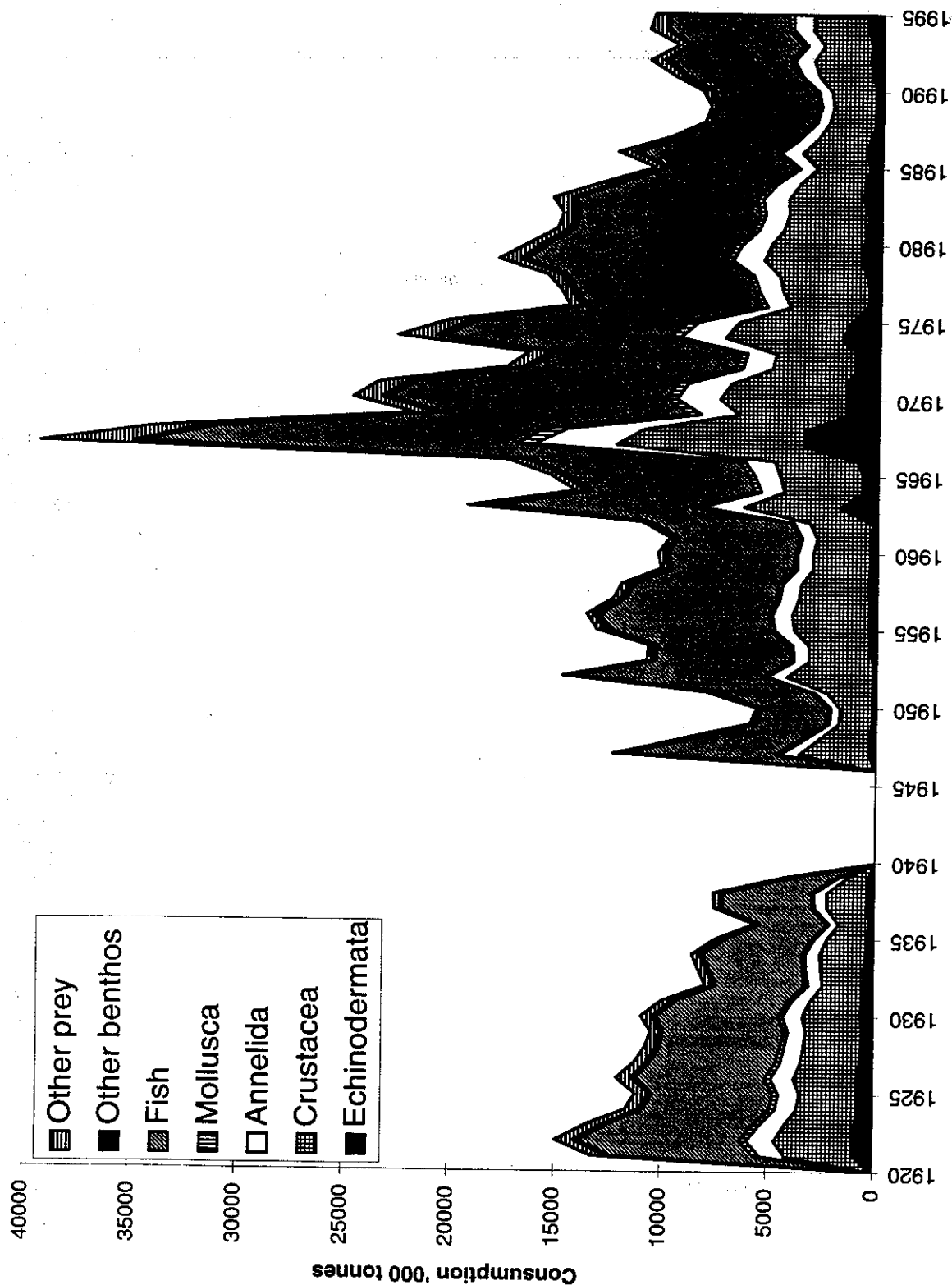
	Rate per year (1000 t)	consumption production
<b>Benthic production estimates</b>		
based on Christensen (1995)	119,700	0.19
based on Greenstreet <i>et al.</i> (1997)	51,152	0.44
<b>Consumption by fish</b>		
based on basic calculations for the eight species included in this study	22,698	

#### Discussion

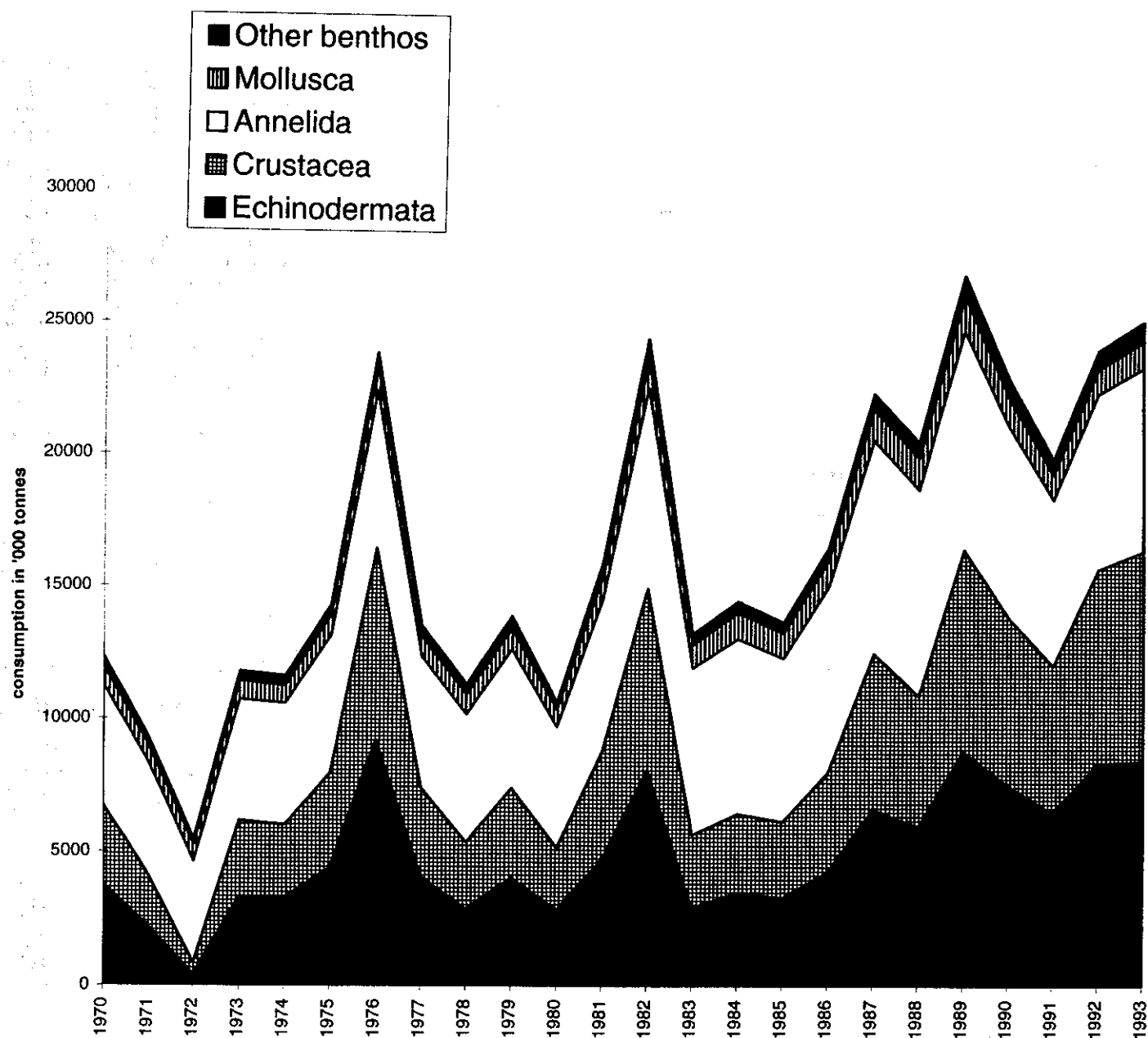
This study has demonstrated that the consumption of North Sea benthos may have changed as fish stock sizes have changed. The principal factor influencing the stock size of exploited fish species is fishing, and the expansion of the non-target dab population may be due to competitive or predatory release. There is, therefore, a case for believing that the observed changes in benthos consumption have resulted from the increase in fishing mortality on the target species. Given that demersal fish biomass has decreased, the increase in predation on the benthos may seem surprising. However, fishing has removed the larger gadoids, whose diet was principally piscivorous, and allowed the expansion of flatfish and young gadoids, which prey upon benthos to a greater extent. However, the differences in diet of the various species would also appear to have influenced the composition of the benthos consumed. Overall, crustaceans have declined in importance, while echinoderms (predominately ophiuroids) have increased.

This finding must be interpreted with caution, as the composition of the benthos in the diet used in the model formulation is based on studies which have sampled in the recent past (Ntiba and Harding, 1993; Greenstreet, 1996; Rijnsdorp and Vingerhoed, in prep.). Therefore, they take no account of long-term changes in the composition of the benthos arising from natural temporal trends, climate-driven variation, the changing levels of fish predation or direct impacts of fishing activities. Results from studies which have compared the composition of fauna in the early part of this century in various parts of the North Sea with the contemporary composition support this concern (Riesen and Reise, 1982; Reise, 1982; Kroncke, 1990). These studies demonstrated shifts in the composition of the fauna towards increased dominance by species which share opportunistic life history traits.

Figure 13.4.2.1. Estimated consumption of prey by gadoids from 1920–1995 according to prey type.



**Figure 13.4.2.2.** Estimated total consumption of benthic organisms by eight species of benthic-feeding fish in the North Sea from 1970–1993.



Three principal assumptions underpin our models. They are that: (i) the composition of the diet has not changed over time (discussed above); (ii) the biomass at age estimates derived from VPA and survey data are valid; and (iii) the total biomass can be used to predict predation levels outside the period covered by the VPA. In the basic calculations, we have only included data for the age groups reported in stock assessment reports. Impacts of predation on benthos by fish younger than the youngest ages in the assessments are not included in these estimates. Thus, it appears that the conclusion that fish predation on benthos is intensive and has increased during the past twenty years, as fishing has reduced gadoid stocks and dab populations have boomed, is likely to be conservative.

Estimates of benthic productivity in the North Sea are generally of the order of 51 million to 120 million tonnes wet weight per year (Greenstreet *et al.*, 1997; Christensen, 1995). Our estimates of the amount of this material consumed by the eight dominant benthivorous fish species (23 million tonnes per year) amounts to less than 45 % of this production.

It must be remembered that, in addition to the indirect effects outlined here, there are direct effects of fishing on the benthos. Recent studies under the EC-funded IMPACT projects (see also Section 10.4, above) showed that the direct mortality caused by beam trawling, estimated as the total mortality associated with one fishing event, was species dependent and varied from 10–40 % in gastropods, starfish, crustaceans, annelid worms and sea mouse, from 10–50 % for the sea urchin *Echinocardium cordatum* and the masked crab *Corystes cassivellanus*, and from 30–80 % for a number of bivalves. At the population level, the mortality imposed by the trawl fishery will depend on the level of direct mortality, the trawling frequency, and the overlap in spatial distribution between the fishery and the benthic organisms. Taking account of the patchy distribution of the beam trawl fisheries, annual fishing mortality rates on benthic invertebrates in the heavily trawled southern North Sea were estimated between 7–45 %. Compared to the estimated percentage of the benthic production that is consumed by fish predators (~45 %), the estimated fishing mortality rates are lower. In combination, therefore, direct fishing mortality rates and indirect changes in predation pressure further support the hypothesis that intensive trawling may have caused shifts in benthic assemblages from large, slowly reproducing species to small species with a high reproductive rate. As such, trawling may have played a role in the increase in growth rate observed in bottom-dwelling flatfish (de Veen, 1976; Millner and Whiting, 1996; Rijnsdorp and van Leeuwen, 1996).

The data produced here show that the consumption of benthos by fish predators has changed in both quantity and composition during the period when fish biomass has been altered by fishing. Alterations, at the ecosystem scale, in the distribution of biomass between

compartments and species within ecosystem compartments are likely to have further indirect effects on ecosystem function. These include alteration of the movement of nutrients and carbon around the system and potential changes in the balance of top-down and bottom-up control of the system.

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#### 13.4.3 Reference points including ecosystem considerations

WGECO was requested to examine and develop potential reference points which might be used for including ecosystem considerations in relation to the precautionary approach. This task specifically concerned reference points, and not the broader management objectives for which the quantitative reference points are developed and used. The material which follows will be readily interpretable in the context of current approaches to fisheries. However, a much broader framework was considered than just traditional fisheries management objectives. Many other types of objectives already influence fisheries practices, from those at very local scales (for example, the protection of specific bivalve beds close to shore-based viewpoints, because they attract concentrations of seaducks) to very large ones (the objective of protecting ecosystem diversity, for example). It is important that the following arguments are viewed as applying in all of those contexts, and not simply as serving traditional fisheries management objectives. Likewise, it is important that specific objectives be

discussed and set by society in many contexts, in addition to fisheries.

#### 13.4.3.1 Statement of the issue

The precautionary approach (FAO, 1995; Doullman 1995; Garcia, 1996) has been accepted as a guiding principle in fisheries management. It covers biological, social, and economic aspects of fisheries. Until now, the practical implementation of the precautionary approach has led to the establishment of limit and target reference points for commercial species. These reference points are recommended as quantitative management objectives. At the current exploitation pattern of fish stocks, the short-term objective is to have a low probability of fish stocks falling below limit reference points, to ensure a long-term sustainability (ICES, 1997a). Target reference points are viewed as long-term objectives.

An additional aspect of the precautionary approach is the integration of fisheries management and ecosystem management. An ecosystem approach in the management and assessment of fisheries involves considering all relevant physical, chemical, and biological ecosystem variables (Anon., 1997). It implies thereby a widening of the current implementation of the precautionary approach. The question at stake is whether reference points being developed for commercial species are sufficient to ensure an effective ecosystem management. This section reviews the ecosystem considerations of different potential reference points, including single-species reference points for target and non-target species, multi-species and ecosystem reference points, mass-balance perspectives, and community metrics.

Even though ICES acknowledges that the Convention on Biodiversity and the FAO Code of Conduct on Responsible Fishing imply that fisheries should be managed in a manner which ensures that ecosystems are sustainable, in the sense that the likelihood that fishing causes a species to become extinct is low, little work has so far been done on how to define reference points in an ecosystem context. Naturally, such definitions would not only be restricted to fish, but would need to include other components of the fauna such as benthos, seabirds and marine mammals, for some of which reference points relating anthropogenic impact to population status have either been defined elsewhere or are non-existing. In addition, they would need to consider not only how fishing mortality affects individual species and their genetic make-up, but also how discarding and physical seabed disturbance affect the system.

One of the great effects of fishing is the harvesting of target species. If it were the case that reference points were used as intended in management, fisheries would already be much further on the way to meeting any specified ecosystem objectives. On the other hand, commercially important species are, by their nature, often highly productive components of the ecosystem.



Reducing their abundances through fishing may have particularly great impacts on the dynamics of the food web. Also, just because they often are less productive, non-targeted species can be much more vulnerable to mortality caused by fishing than are many commercially important species. It has been proposed that these sensitive species could be useful indicators for the state of the ecosystem. With respect to the single-species approach, attention is given to the usefulness of such signal species as a basis for additional reference points for fisheries management.

Multispecies models contain more ecosystem considerations than their single-species counterparts. The multispecies models used by ICES account for predator/prey relationships. In work completed to date, they have led to more conservative estimates of reference points, and they estimate lower fishing mortality rates for sustainable fisheries than single-species models (ICES, 1997b). In that sense, they require more conservative fisheries to achieve an equal degree of precaution.

### 13.4.3.2 Specific reference points considerations

#### What ICES already advises

ICES considers a stock to be within safe biological limits if the spawning stock biomass (SSB) is above the minimum biologically acceptable level (MBAL) with high likelihood, and there is a low likelihood of SSB falling below MBAL in the medium term, at status quo fishing mortalities. MBAL plays a key role in ICES advice. It is estimated in a variety of ways, but is generally considered to be the SSB at which either the probability of poor recruitment is increased or the probability of good recruitment is decreased markedly. The total allowable catches (TACs) advised by ICES are based upon fishing mortalities. ICES does not advise one TAC level, but gives 'short- and medium-term forecasts (if possible) of the stock development at different exploitation levels. The responsibility for using a precautionary approach in setting the definitive level of a TAC is vested in the fisheries management agencies receiving advice from ICES.

Within ICES, several Working Groups and Study Groups are discussing biological reference points which can be used in the ICES advice in the near future. For a description of these discussions on reference points for commercial species, the reports of the ACFM Study Group on the Precautionary Approach to Fisheries Management (ICES, 1997a) and the Comprehensive Fishery Evaluation Working Group (ComFIE) (ICES, 1997c) are of interest. In addition to the work produced by these two groups, the Multispecies Assessment Working Group (MSAWG) compared the difference in

the above recommendations between a single-species approach and a multispecies approach (ICES, 1997b).

While ICES has made steady progress in developing precautionary reference points, the implementation of ICES advice on single-species harvesting leaves room for improvement. Although there are encouraging trends towards TACs consistent with ICES advice in some fisheries, there are many stocks for which TACs are set higher than ICES advises, and which are fished harder than managers intend. Because of the difficulties in reducing the present intensity of fishing in many areas, conservation of even the individual targeted stocks is at risk in many fisheries. Therefore, discussion of the possible benefits of fisheries management using reference points based on the state of the ecosystem rather than on the states of individual harvested stocks is largely speculative. On the other hand, such a discussion might identify compelling reasons at the ecosystem level for fisheries management to practice greater caution.

To begin this speculative discussion, the first question to pose is 'If all fisheries were managed so that there was a high probability of achieving conservation objectives for the target fish stocks, would there be a high likelihood of achieving conservation objectives for ecosystems?' Current knowledge makes the answer to this question clearly 'No' for at least four reasons:

- 1) The genetic diversity of a target stock might be at risk, even in management regimes which complied with single-species reference points for biomass and fishing mortality;
- 2) The conservation of non-target species could be at risk due to direct mortality from fishing activities;
- 3) The conservation of dependent predatory species could be at risk due to local depletion of prey aggregations, even if conservation of the prey stock were being achieved on a much larger spatial scale;
- 4) The conservation of some species could be placed at risk through the abundance of scavenging species increasing due to discarding in fisheries.

It is not a coincidence that in all four of these situations, the reference points which must be added are still single-species reference points. In those cases, the principles and criteria parallel most closely the existing approaches to reference points for target stocks. However, it is stressed that the issue does not end with single-species reference points. The weight of scientific evidence suggests that there are additional reasons at the ecosystem level for why the answer would be 'No'. Examples of these reasons include documented changes to nutrient cycling and remineralization rates and pathways, caused by impacts of fishing gear on substrates (Rowe *et al.*, 1975; Prins and Smaal, 1990), and diverse consequences on food web structure and function, caused by fisheries changing the absolute and relative abundances of target and non-target species.

<sup>1</sup>  $F_{msy}$  is the fishing mortality which produces maximum sustainable yield (see Beverton and Holt, 1957).

## **Additional reference points for species, from an ecosystem perspective**

### Genetic reference points for exploited stocks

In some studies it has been demonstrated that even short periods of intensive exploitation can alter the genetic makeup of an exploited population. Longer periods of exploitation, possibly at rates sustainable with regard to target stock size, may induce genetic responses as well (Lande, 1993; Stokes *et al.*, 1994; Waples, 1995). On a case-by-case basis, however, it is often problematic to differentiate phenotypic responses of life history or morphological traits from a loss of genetic characteristics from the population (e.g., Rijnsdorp, 1993). Nonetheless, the loss of genetic diversity is a possible consequence of sustained or episodic intensive fishing, and is not addressed in existing biological reference points based on biomass and fishing mortality. The Convention on Biological Diversity explicitly recognizes the need for management to conserve the genetic diversity of stocks, so additional single-species reference points are necessary to fulfill this responsibility.

### Reference points for non-target species

Despite a reduction in the fishing mortality rate of commercial species which would result from full implementation of the current ICES management advice, there may still remain unwanted effects for a number of reasons. Fisheries kill organisms other than the target species. The by-catch mortality may be unsustainable for a non-target species for two different reasons. First, direct exploitation may be too high. Species such as elasmobranchs and cetaceans and some structure-building benthos may only be able to withstand much lower mortality rates than the target fishing mortalities for directed fisheries. Commercial species may, by their nature, be more resilient to exploitation. Specific management targets should be set for the more vulnerable components of the ecosystem. Even low levels of by-catch mortalities may require reference points for specific species such as some seabirds and marine mammals. This is because of their inability to withstand high mortality rates or their potentially high vulnerability to incidental mortality due to, at least periodically, their forming very large aggregations. Secondly, because the EU management sets single-species TACs, a fishery targeting a mix of commercial species may continue fishing, and thus generating additional mortality, as long as not all TACs have been taken. ICES advice acknowledges this potential problem in the text of the annual advice. However, the estimation and application of single-species reference points may have to include aspects of multi-species relationships explicitly, to provide a higher likelihood of achieving conservation objectives for stocks taken in mixed fisheries.

Downward or upward trends in populations of many non-target species have been shown for the North Sea and

other intensively fished areas (Heessen and Daan, 1996; Anon., 1997). Still, not all these species are suitable as potential reference points in an ecosystem consideration in fisheries management because, to be useful as a reference point, it is desirable to have a very well-defined and clear relation of stock status with fishing activities. Otherwise, it will not be possible to formulate effective management measures. The status of top predators, species which serve as main sources of food; structure-building organisms or representatives of a vulnerable group of species may be particularly useful as reference points. From recent ecosystem and fisheries research, two potential indicators or potential reference points are the harbour porpoise (*Phocoena phocoena*) and the thornback ray (*Raja clavata*).

### Reference points for ecologically dependent species

In Antarctic waters, the breeding success and even the survivorship of a number of predators, including several species of seabirds and marine mammals, are affected greatly by the status of krill (Laws, 1984; Croxall and Prince, 1987). Correspondingly, the requirements of these ecologically dependent predators play a major role in the management of krill fisheries in that region (SC-CAMLR, 1992). Recently, the Scientific Working Group of the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR) reviewed elements required for a precautionary approach to the management of krill fisheries, in light of the expanding ideas about the precautionary approach, and progress in the development of reference points. The associated analyses indicate that, although a precautionary overall catch limit is necessary for large geographical areas, that limit is not sufficient to safeguard some of the dependent predators. A management approach has been proposed which requires geographical subdivision of the overall catch according to the varying requirements of predator populations, and uses information on the predator populations and their physiological needs in setting harvest levels (Everson and de la Mare, 1996).

For the Northeast Atlantic, ICES has received requests for advice about possible management measures which might be necessary to protect local aggregations of sandeels near sensitive wildlife concentrations. The request clearly stems from the same concern, i.e., that there may be ecologically related species whose conservation is not assured by a management approach which places the stock being targeted at negligible risk overall. Also, the fishery for capelin in the Barents Sea is managed under an approach which gives the feeding requirements of cod (and possibly other predators) priority over human harvests.

Specific types of biological reference points have not been proposed for such ecologically related species, nor have the links between the reference points and specific management actions been specified. Nonetheless, in at least a few cases, such as colonial seabirds and their prey fish stocks, cod and capelin, and Antarctic top predators

and krill, the relationships have been studied extensively, and the management needs are recognized. The knowledge base might be an adequate foundation for the development, testing, and implementation of such reference points linked among species.

#### Reference points for species affected by scavengers feeding on discards and offal

Populations of many scavenging seabirds have grown in recent years (e.g., Lloyd *et al.*, 1991). Some of this growth may be due to recovery following a long period of persecution which ended in the early part of the current century, but it is likely that much of the growth of the populations of some species is due to the increased food supply deriving from fishery wastes (e.g., Fisher, 1952; Furness and Barrett, 1985). This growth appears to be continuing in many populations.

Owing to the requirement of seabirds to breed in areas that are free (or virtually free) of mammalian predators that can take eggs or young, there is frequently competition for the limited habitat that meets this requirement. In many cases, this leads to displacement either into nearby sub-optimal habitats or away from the area entirely (Howes and Montevecchi, 1993). In many cases, this displacement may not be desired by local wildlife managers (and may locally reduce biodiversity).

#### Summary of reference points at the species level

Suppose that biologically sound reference points for genetic diversity were added to the existing biomass and mortality reference points, and that reference points were also identified for all non-target species and for species ecologically dependent on aggregations being fished. Furthermore, suppose that fisheries complied with these reference points, such that there was a high likelihood of achieving all single-species conservation objectives. Would conservation and sustainability of the ecosystem be achieved with at least an equal likelihood? If the answer to this core question is no, there are two ancillary questions. First, what multispecies properties might still be at an unacceptable level of risk? Second, how should these properties be monitored and/or modelled, in order to identify and evaluate the effectiveness of actions taken to reduce the risk?

#### **Biological reference points from an ecosystem perspective**

The answer to the first question in the preceding paragraph is that we do not know whether conservation and sustainability of the ecosystem as a whole would be achieved. We do know that, without question, fishing has changed the size composition of fish in some, possibly many, exploited systems (Pope and Knights, 1982; Pope *et al.*, 1988; Dayton *et al.*, 1995), and in the North Sea in particular (ICES, 1996; Rice and Gislason, 1996). Regardless of the trophic model considered, changing the

size composition of predators in the ecosystem has, with high likelihood, changed the way that predation pressure is distributed among lower trophic levels in the ecosystem. The uncertainty is in the magnitude of the change, and its consequences for the ecosystem. We also know that the flux and residency of nutrients within the system must also have changed, as the numbers and biomass in different trophic levels as well as features of the benthos have been changed (Rowe *et al.*, 1975; Prins and Smaal, 1990). Again, it is the magnitude and ecosystem consequences which are uncertain. Even if present knowledge is inadequate to answer the first question, it is adequate to highlight that a truly precautionary approach to the possible consequences, as outlined below, should be of serious concern.

A number of multispecies or ecosystem models have been developed which can be used to investigate this question. At this time, though, different models make very different predictions about the ecosystem consequences (or lack thereof) of changing the distribution of predation pressure among sizes (and undoubtedly species) of prey. We also know too little about the flux of nutrients at lower trophic levels, and among the benthic, pelagic, and demersal parts of the ecosystem, to know even how the flux of nutrients has changed as a result of reducing the numbers and biomass of large predators, let alone the consequences of the changes. Therefore, it is premature to draw inferences about the impact of the changes in size composition of predatory fish on the sustainability and conservation of the larger ecosystem as a unit, and on the larger question of the need for additional precautionary reference points.

Primary production in marine ecosystems away from the coastal zone is generally controlled by the availability of nutrients and usually nitrogenous forms. In stratified regions, the rate-controlling step is the regeneration of nutrients by zooplankton and the excretion of ammonia by fish. In vertically well-mixed areas, the flux of nutrients from the benthos is also important—decomposers in the benthos being responsible for the ammonification of organic nitrogen and the reduction of nitrate to ammonia (Sørensen, 1978). The high productivity of coastal waters may be dependent on this benthic-pelagic coupling (Rowe *et al.*, 1975). The flux rate of this coupling is dependent on the biological activity in the sediments, and in particular the nature of the benthic fauna (Prins and Smaal, 1990; Josefsen and Schlüter, 1994). Fishing has the potential to alter these rates by causing (i) alterations in the benthic fauna, (ii) resuspension of benthic material by towed bottom gears, (iii) alterations in the chemical status of bottom sediments, e.g., exposure of anoxic material, and (iv) alterations in the sizes of the food web compartments.

Although we cannot evaluate the likelihood of achieving ecosystem-level objectives using a strategy of achieving all single-species conservation objectives, we do note some important considerations with regard to ecosystem-

level reference points. First, it is well established that the dynamics of individual stocks and populations which are connected trophically contain time lags and buffers (e.g., age structure, density-dependent growth) which can slow down the rate at which the consequences of perturbations of a food web may be manifested. Therefore, we may not yet be observing the full impacts on the ecosystem of past levels of fishing. Moreover, if there were to be changes in major ecosystem properties, most models suggest that the changes could be difficult and slow to reverse, and would aggravate the loss in total yield of fish, beyond the yield already foregone directly due to overfishing the target stocks.

Although we are not in a position to recommend that ecosystem reference points are necessary, beyond the reference points which would assure sustainability and conservation of all populations killed directly by fishing, neither are we prepared to confirm that single-species reference points are enough to ensure a precautionary approach. This is a complex problem, with important implications, and many more investigations of model (and ecosystem) dynamics are required. For example, although WGECO has clearly documented that the slope of the biomass spectrum of the North Sea has changed over the past twenty years, we cannot advise what a maximum tolerable slope would be, what a 'good' target slope would be, or even whether these are reasonable concepts to consider.

A commitment to a precautionary approach to fisheries management and the conservation of biodiversity has to include a commitment to pursue these types of questions much further. Relevant programmes would have to answer the following questions:

- a) What ecosystem properties require more than just the conservation of the individual component species?
- b) Which of the properties in a) could be placed at risk by fisheries?
- c) What management measures would be necessary to have a high likelihood of achieving conservation of the properties in b)?
- d) How could the properties potentially at risk be measured and monitored?

Some of these questions have fueled research and debate among community ecologists for decades, and rapid resolutions are unlikely. Future meetings of WGECO could address the state of knowledge on these questions more intensively, but would require attendance by diverse specialists, and the opportunity to focus significant time on these questions. However, the need for some ecosystem-level reference points is real. Even if different theoretical frameworks suggest different properties for ecosystem-level reference points (often just because the different frameworks use different biological 'currencies'), in internally consistent ways, every framework indicates that such properties exist.

In relation to fisheries impacts, much of the discussion on the implications of using the precautionary approach has focused on how to define target and limit reference points using traditional single-species fisheries models to make predictions of the impact on the target species (e.g., ICES, 1997c, 1998). For non-target species, seabirds, and benthos, reference points have not been set and, in particular for benthos, the present knowledge has not, with few exceptions, yet crystallized into models which could readily be used to predict the consequences of fishing for individual species or assemblages.

Little effort has been spent on investigating how reference points could be defined by models which allow the species to interact. Multispecies fish stock models include species interaction in the form of fish predation and are available for some areas, but have seldomly been used for providing management advice. Some of the multispecies models have been extended to include marine mammals and seabirds. Often this has been in terms of the impact mammals and seabirds have on the commercially exploited species; only very rarely has the reverse question been asked. At present, the models are therefore of limited use for defining reference points in relation to fisheries-generated food limitations for seabirds and mammals. However, simpler models have been used to estimate exploitation levels on prey species which take the needs of their predators into account, e.g., the models used to arrive at precautionary catch limits for krill in the Antarctic (Everson and de la Mare, 1996).

Few models describe how community or ecosystem properties would change in response to fishing, and often the existing metrics, such as species diversity indices or slopes of size spectra, are difficult to connect to the perceived state of the affected system. For this reason, such metrics have not yet been used to define limits and target reference points. The models that are available describe either overall metrics, such as the slope of the size composition of the fish assemblage, or consider energy flow among trophic compartments. Of the latter type, mass balance models offer a range of possible measures that could be used for defining reference points. Another possibility is to utilize more conceptual tools, such as trophic cascade models (Carpenter *et al.*, 1985; Carpenter and Kitchell, 1988; McQueen and Post, 1988a, 1988b; Leavitt *et al.*, 1989; Brönmark *et al.*, 1992; Martin *et al.*, 1992; Christoffersen *et al.*, 1993; Schindler *et al.*, 1993). However, in each instance, the challenge is not to derive the metric, but to relate it to changes in the affected system that are of relevance to society.

#### 13.4.3.3 Concluding remarks

This section has been developed by starting from existing practice and asking what must be added. One necessary addition to present practice is reference points for non-target species. However, the task does not stop there. Implicitly, present practice assumes that explicit conservation objectives have been set by management agencies, to justify the development of even the reference

points used at present. As recent ICES advice makes clear, even that assumption is not absolutely true. Nonetheless, in endorsing the precautionary approach, governments and management agencies have clearly stated their commitment to the conservation of all species affected directly or indirectly by fishing (FAO, 1995; Garcia, 1996). Much of the internal debate within ICES has centred on what additional commitments are implicit in this approach, because there are strong theoretical reasons to expect that certain ecosystem properties may be altered by fishing activities.

Will society (and biology) be served by objectives to conserve particular configurations of an ecosystem being fished? Do the diverse international agreements summarized in FAO (1995) require such objectives to be adopted? What does it mean for an ecosystem to be 'at risk', and can an ecosystem be 'at risk' if the species which comprise it are not at risk? Although ACME looked forward to exploring these fundamental questions at future meetings, it stressed that they must be discussed in many other fora as well, both within and outside ICES.

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### 14.1 General Trends in Mariculture

#### *Request*

There is no specific request; this is part of the continuing ICES work concerning environmental interactions of mariculture.

#### *Source of the information presented*

The 1998 report of the Working Group on Environmental Interactions of Mariculture (WGEIM) and ACME deliberations.

#### *Status/background information*

Marine fish production continues to increase in many countries. For salmon, the increase in the production is usually about 20 % per year for Norway and Scotland. The production of turbot, seabream, and sea bass is developing in southern Europe (France, Spain) and in some Mediterranean countries. The aquaculture production of molluscs (oysters, mussels, and scallops) is quite stable in the major producing countries (France, Spain, the Netherlands), but is progressing in Scotland and Canada.

For salmon, there is a tendency towards larger cage farms at sites in Norway and particularly in Scotland. In several countries, there is a tendency to test the potential of new species of both finfish and shellfish for aquaculture production.

#### **Emerging environmentally friendly aquaculture practices**

Sustainable development of aquaculture requires that the environmental impact of this industry be limited as much as possible. Several environmentally friendly initiatives and research programmes are being undertaken in several countries, including:

- marine finfish cultivation in recirculating systems (pilot scale operations for sea bass (100 tonnes/year) have been built in France, Iceland and Italy);
- for land-based operations, depuration treatment of wastes is now widely applied with reservoirs where phytoplankton blooms occur which allow associated production of molluscs;
- fallowing strategies in fish cage farming are increasingly being applied in some countries (Scotland, Norway, and Ireland) for two main reasons: firstly, to break disease cycles and, secondly, to allow a period of time for the recovery of the benthic environment. The duration of fallowing is increasing to one or two years, as a function of the

recovery rate of benthos colonization, which is a function of hydrochemical conditions;

- applying minimum distances between mariculture installations. To minimize the cumulative effects on the water column and on the benthos and to avoid the spreading of disease, some ICES Member Countries have been applying a minimum distance between fish farms (1000 m in Norway, 5 miles in Scotland, etc.). Regulations on the required distance are made empirically by the countries, and there is a need to develop research programmes to assess whether these distances are optimal to minimize the environmental impacts.
- to fight against sea lice infestation, which is a very prevalent disease of Atlantic salmon, alternatives to medicinal chemical treatment have been developed; the use of 'cleaner fish', e.g., wrasse, is increasing. It has been estimated that 10 million wrasse will be required annually in the future, only for Norway. If the increasing demand for wrasse is to be met, it is likely that cultivation of that species will soon become necessary.
- the development of different acoustic systems, that are now very effective in repelling marine mammals such as seals, and thereby reducing predation on cultured fish.

### 14.2 Quantitative Information on the Escape of Fish from Mariculture Operations

#### *Request*

Item 4.1 of the 1997 Work Programme from the Oslo and Paris Commissions requested ICES to provide quantitative information on the escape of fish from mariculture operations. The 1997 ACME report contains the response to this request, but some supplemental information is presented here.

#### *Source of the information presented*

The 1998 report of the Working Group on Environmental Interactions of Mariculture (WGEIM), the Proceedings of the 1997 ICES/NASCO Symposium on Interactions between Salmon Culture and Wild Stocks of Atlantic Salmon (ICES, 1997a), and ACME deliberations.

#### *Status/background information*

The OSPAR request in 1997 was to provide quantitative information on the escape of fish from mariculture operations in the context of disease transfer, genetic composition in relation to wild stocks, and competition for food and habitat, and means by which these impacts can be controlled.

To supplement the response made in 1997 (ICES, 1997b), the ACME reviewed the relevant sections on fish escapes from mariculture in the 1998 WGEIM report and in the Proceedings from the ICES/NASCO Symposium. As this topic was not the subject of a specific request in 1998, only information from Norway was relevant to the OSPAR area.

### Records of Escapees in Norway

Investigations of a number of types of Atlantic salmon populations show that the relative proportion of escaped cultivated fish in coastal areas is high (34–54 %) and significantly higher than in the fjords (see Table 14.2.1). The proportion of escapees in the rivers as a whole is relatively small, while it increases among brood stock populations in the rivers. It seems to be a trend that the proportion is highest in areas with high aquaculture activity.

**Table 14.2.1.** The relative proportion of cultivated salmon in catches from various areas in Norway, given as the percentage of the total salmon catch for each area, in the period 1989–1996.

Year	Coast	Fjords	Angling, river	Brood stock
1989	45	14	7	35
1990	48	15	7	34
1991	49	10	5	24
1992	44	21	5	26
1993	47	20	4	20
1994	37	19	4	23
1995	42	17	4	26
1996	54	16	7	31

The total number of escaped Atlantic salmon reported from Norwegian fish farms in 1996 was 455,900, of which the first year generation constituted about 53 % (Table 14.2.2). The short-term environmental objective is to keep the number of escapees per year below 400,000. As can be seen in Table 14.2.2, this objective was not fulfilled in 1996. Furthermore, the gap between the desired objective and its achievement was probably much larger because the reported number is an underestimate; the actual number of escapees may be as high as 700,000.

**Table 14.2.2.** Reported numbers of escaped salmon from Norwegian fish farms in 1996 by age groups in production (generation) and as a summed total.

Generation	1994	1995	1996	Total
Number	40,800	168,800	241,000	455,900

As the number of cultured Atlantic salmon in Norway is about 180 times the number of wild salmon, even a relatively small proportion of refugees may represent a threat to wild stocks. When revising the environmental

objectives for Norwegian aquaculture in 1997, reducing salmon escapement was again given top priority.

Most of the fish escape during management operations, which means that there is a continuous leakage of fish from the farms. To reduce the danger of catastrophic losses due to storms, the authorities have implemented a system for approving the technical standard of fish farms. There is further compulsory monitoring to detect fish escapes and, in cases of a mass escape, the fish farmer must have a plan for recapture.

Tagging experiments to study the migration of Norwegian escaped salmon have clearly demonstrated that Norwegian salmon from mariculture are also caught in the Baltic countries, as well as in Iceland, France, Spain, and even Canada. These results show that the problem is not local, but must be studied on a wider scale.

A document on the distribution and numbers of Atlantic salmon on Canada's west coast has been published (McKinnel *et al.*, 1997). From 1991 to 1995, a total of 97,799 fish were reported to have escaped in British Columbia.

In the USA (Washington State), it has been reported that a single escape in 1995 introduced more than 100,000 farmed salmon into the environment when a farm broke up under extreme storms and tidal conditions.

### Need for further research or additional data

The ACME considered that, despite the lack of hard scientific evidence, the threat that escapes might have adverse effects is serious with regard to disease transfer and the genetic composition of the metapopulation of wild salmon. The ACME emphasized that appropriate technical measures to minimize chronic escapes from mariculture operations should be applied to the extent possible.

### Recommendations

ICES encourages Member Countries to conduct studies aimed at quantifying the escapes of fish and shellfish from mariculture operations.

ICES recommends that technical measures that aim at minimizing escapes of organisms from mariculture operations be developed and applied in order to reduce the potential risk of adverse effects of escapes on disease transfer, genetic composition in relation to wild stocks, and competition for food and habitat.

ICES further recommends that, in order to prevent disease transfer from farmed and escaped farmed organisms to wild stocks, the guidelines provided in the Office International des Epizooties (OIE) Aquatic



Animal Health Code and in the 1994 ICES Code of Practice on the Introductions and Transfers of Marine Organisms (ICES, 1995) should be followed.

## References

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McKinnel, S., Thomson, A.J., Black, E.A., Wing, B.L., Guthrie, C.M., Koerner, J.F., and Helle, J.H. 1997. Atlantic salmon in the North Pacific. Aquaculture Research, 28: 145–157.

## 15 DATA HANDLING

### 15.1 Activities of the ICES Environmental Data Centre

#### 15.1.1 Ad Hoc Working Group on Monitoring of the OSPAR Commission

The ICES Environmental Data Centre serves as the data centre for OSPAR's monitoring programme on contaminants in the marine environment. As a part of the 1998 work programme, the ICES Data Centre was requested to compile and analyse data on temporal trends in contaminants in biota for the 1998 meeting of the OSPAR Ad Hoc Working Group on Monitoring (MON98).

The deadline for the submission of historical (i.e., pre-1996) data was 1 May 1997. A total of 185 historical data submissions (data files) were received from Belgium, Denmark, France, Germany, Iceland, the Netherlands, Norway, Spain, Sweden, and the UK; 126 of these were received before the agreed deadline, while 59 (more than 30 %) were received after the agreed deadline. The last historical submission was received on 21 November 1997, almost seven months after the agreed deadline.

The deadline for the submission of new (i.e., 1996) data was 1 August 1997. A total of ten new data files were received from the above-mentioned countries. Three of these submissions were received before the agreed deadline, while seven were received after the agreed deadline. The last new data submission was received on 21 November 1997, almost four months after the agreed deadline.

Two quality assurance (QA) assessors were appointed by OSPAR. Their task was to review the QA information that accompanied field data when they were reported to the Data Centre. There were three types of QA information: (1) information about participation in intercomparison exercises, (2) results of analyses of reference materials, and (3) written material about the analytical method(s) applied. The Data Centre passed on this information to the QA experts. A significant amount of the material was in digital form, thus allowing the QA assessors to partially automate the review process.

The agreed data set comprised measurements of trace elements (As, Cd, Cr, Cu, Hg, Pb, Zn), PAHs (anthracene, phenanthrene, fluoranthene, pyrene, benz[a]-anthracene, chrysene, benzo[a]pyrene, indeno[1,2,3-cd]pyrene, benzo[ghi]perylene), and organochlorines (PCBs,  $\alpha$ -HCH,  $\gamma$ -HCH, HCB, DDT, DDE, TDE, dieldrin, and *trans*-nonachlor) in fish liver, fish muscle, and the soft body of blue mussel. More than 11,000 data (medians) were assessed, using the method suggested by Nicholson *et al.* (1998).

For the first time, data inventories and preliminary data products were made available to the participants of the MON98 Working Group on closed (password-protected) pages on the World Wide Web. This allowed participants to familiarize themselves with the data and to make preliminary assessments prior to the meeting.

For a number of the parameters, background/reference concentrations (BRC) and/or ecotoxicological assessment criteria (EAC) were available, and this allowed MON98, for the first time, to compare observed concentrations with target concentrations.

#### Reference

Nicholson, M.D., Fryer, R.J., and Larsen, J.R. 1998. Temporal trend monitoring: Robust method for analysing contaminant trend monitoring data. ICES Techniques in Marine Environmental Sciences, No. 20. 22 pp.

#### 15.1.2 Fish disease data assessment

A subgroup of the Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) has, in recent years, evaluated data on fish disease prevalence as compiled and stored at the ICES Environmental Data Centre on an annual basis. Following a decision at the ICES Annual Science Conference in 1997, this work now comes directly under the remit of WGPDMO.

The Data Centre made a data set available to selected members of WGPDMO prior to the 1998 meeting of the group. Five laboratories from four ICES Member Countries provided data (Denmark, Germany, the Netherlands, and the UK).

The data set contained information about a total of 424,998 specimens of dab (*Limanda limanda*, n=399,262) and flounder (*Platichthys flesus*, n=25,736). The fish were sampled between 1981 and 1997 at positions within 131 different ICES statistical rectangles.

The data set was analysed prior to the meeting of the WGPDMO, and the results of the analysis were reviewed at the meeting (see Section 8.1, above). The report on the results is attached as Annex 8.

#### 15.1.3 Development of biological databases

Under item 2 of the 1997 data handling requests from the OSPAR Commission, ICES was asked to begin to establish a data bank for phytobenthos, zoobenthos, and phytoplankton species. In addition, the Helsinki Commission requested ICES to update and further

elaborate the ICES Biological Data Reporting Format (covering macrozoobenthos, phytobenthos, phytoplankton, and zooplankton) to meet the revised HELCOM Baltic Monitoring Programme (BMP) Guidelines and recommendations on quality assurance procedures arising from the ICES/HELCOM Steering Group on Quality Assurance of Biological Measurements in the Baltic Sea (see also Section 5.1, above).

The development of the biological databases and associated reporting formats and software was initiated via an e-mail conference. The purpose of the conference was to identify and take into account current national and/or institutional practices and trends in the handling of biological data.

The e-mail conference was organised as a series of WWW pages (<http://www.ices.dk/env/biorep>), providing background information for the conference, and access to the individual contributions.

The conference was scheduled to run from 15 October to 15 December 1998. A total of thirteen contributions was received between 30 October 1997 and 25 February 1998. The contributions submitted had not been reviewed by the time of the June 1998 ACME meeting.

#### 15.1.4 European Environment Agency

The ICES Environmental Data Centre responded to a request from the European Environment Agency (EEA) for data on trace metals and PCBs in fish, blue mussels, and sediments for use in the preparation of the Dobris +3 report.

#### 15.1.5 Web access to data inventories

Web access to inventories of data held at the Environmental Data Centre was installed during 1997 and early 1998. Two different types of inventories are currently available.

Static inventories contain information about individual data submissions. For each submission, the web pages provide information about the status, size, number of stations, number of parameters, etc., for each particular submission. The intention of these web pages is to give data suppliers immediate feedback regarding the status and content of a particular data submission.

Dynamic inventories allow direct search in the databases. Search criteria can be defined in terms of time, space, parameter, matrix, reporting laboratory, and (for biota only) species.

In the beginning of 1997, all contributing institutes were asked whether they would give permission for their data

to appear in the dynamic inventories. Most institutes gave positive replies during 1997; one institute gave a negative reply, and a number of institutes did not reply. The institutes that did not reply were contacted again in a second round in Spring 1998, and were again asked to give their permission for inclusion. This time it was indicated that no reply would be taken as an affirmative response. This round produced no negative replies, thereby indicating that all institutes have now given their permission for the inclusion of their data in the inventories.

#### 15.1.6 Web access to reporting formats

The transfer of the annexes included in the 'ICES Environmental Data Reporting Formats' to the ICES WWW pages was completed in April 1998. The maintenance of the web pages is automated so that updates are produced whenever the underlying databases are modified. The ultimate aim of phasing out the production of printed versions of the annexes has therefore been achieved.

### 15.2 Handling of Nutrient Data for the OSPAR Commission

In 1997, the ACME noted with concern the rapidly deteriorating position with regard to the delivery of nutrient data to the ICES Oceanographic Data Centre (ICES, 1997).

This year, the situation with regard to the submission of nutrient data to ICES remains much the same as last year, and can be summarized as follows for data collected in the North Sea:

Year	Number of Stations	Number of stations with nutrients	% Nutrient Stations
1990	10,708	6,034	56
1991	6,828	4,077	60
1992	7,122	4,800	67
1993	7,554	4,929	65
1994	5,104	2,913	57
1995	5,006	2,192	44
1996	3,792	1,392	37
1997	1,104	58	5
1998	187	0	0

This table indicates that many oceanographic station data are not being delivered to ICES until several years after collection, with the lag time for the delivery of nutrient data being particularly marked. As a result, the Oceanographic Data Centre contains only 20 % of the

quantity of nutrient data for 1996 that it holds for 1990. The situation with regard to other sea areas is much worse than this.

For OSPAR-specific estuarine data, submissions have been received for 1996 from Germany, the Netherlands, and Belgium. Most other countries deliver OSPAR nutrient data directly to the Oceanographic Data Centre as part of their research data set, so OSPAR-specific data are difficult to identify. In some cases, data have been of poor quality and have not yet been merged into the data bank, pending the resolution of a number of issues. These particular problems often arise from instrumental difficulties during data collection.

The OSPAR EUT work in 1996 identified many gaps in nutrient data for various OSPAR regions, and a call was put out for additional data. As yet, this call has yielded a very small amount of new data.

## Reference

ICES. 1997. Report of the ICES Advisory Committee on the Marine Environment, 1997. ICES Cooperative Research Report, 222: 115.

## 15.3 Taxonomic Code Systems

### *Request*

This topic is relevant to item 2.1 of the requests for scientific advice and item 2 of the data handling requests on the 1998 Work Programme from the OSPAR Commission, and also to items 2 and 4 of the 1998 requests from the Helsinki Commission; it is also of interest to ICES Member Countries.

### *Source of the information presented*

The 1998 reports of the ICES/HELCOM Steering Group on Quality Assurance of Biological Measurements in the Baltic Sea (SGQAB), the Working Group on Phytoplankton Ecology (WGPE), and the ICES/IOC Working Group on Harmful Algal Bloom Dynamics (WG HABD), and ACME deliberations.

### *Status/background information*

Taxonomic coding of species is necessary in the databases where biological data are stored. Several coding systems exist:

- 1) The RUBIN code has been used by HELCOM, but it is no longer being updated by the originators. This code will, however, still be updated by the Finnish Institute of Marine Research for the purposes of the phytoplankton counting software used for the HELCOM monitoring work until a new coding system becomes available.
- 2) The U.S. National Oceanographic Data Centre (NODC) Taxonomic Code (Version 7) is used in some countries, but the updating of this system has also ceased.
- 3) The NODC Taxonomic Code has been replaced by the U.S. Interagency Taxonomic Information System (ITIS). The coding is flexible and recoding is possible, but there is at the moment no procedure for updating the code and marine species have low priority in the development of the system. The numeric code is not hierarchical.
- 4) Several additional national or institutional coding systems exist.

According to SGQAB and WGPE, the scientific name of the species/taxon should always be the basis for coding.

Several taxonomic species lists exist in the HELCOM and OSPAR areas.

A list of known checklists and phytoplankton identification literature was extended by WGPE, which also discussed possible ways of preparing a practical 'ICES Checklist of Phytoplankton' and its content. Several smaller checklists covering specific regions should be extracted from this list.

The updating of the HELCOM phytoplankton species checklist is under way and will be available on the Internet together with Baltic Sea phytoplankton identification sheets (<http://www2.fimr.fi/algaline>).

A taxonomic checklist for zooplankton is under preparation by WGZE.

### *Need for further research or additional data*

The ACME concluded that ICES will need checklists of scientific names of plankton and zoobenthos taxa, including synonyms used, which will form the basis for the taxonomic coding for the ICES biological database. The coding should be flexible enough to allow the changes that will result on the basis of taxonomic development.

A procedure has to be found for how the various species checklists on plankton and zoobenthos in the ICES area can be made available for use by ICES Member Countries and updated regularly.

In the HELCOM monitoring work, a common software for plankton species data entry with taxonomic names is used. This must be noted when biological data reporting formats and data entry programs are developed by ICES.

### *Recommendations*

ICES recommends that coding of the plankton and zoobenthos species in the ICES database should be based on the scientific name of the species/taxon. For computational purposes, ICES can make or adopt a flexible numeric code. The ICES system should not preclude the use of other systems on a national basis.

### 16.1 Global Ocean Observing System (GOOS)

In 1997, the ACME considered in detail its views on an appropriate potential input by ICES to the Global Ocean Observing System (GOOS) (ICES, 1997). The focus of this input was operational fisheries oceanography. During the inaugural meeting of the new ICES Oceanography Committee at the 1997 ICES Annual Science Conference, the Committee debated at length its future remit and the progress it would be making towards the development of an Action and Strategic Plan in the intersessional period. The Oceanography Committee regards GOOS as a fundamental focus for its activities and, against the background of the information contained in the 1997 ACME report, agreed to support the establishment of a Steering Group on GOOS.

The Oceanography Committee considered that the implementation of the GOOS programme will have to be based on national and regional contributions (e.g., EuroGOOS and US-GOOS) through bilateral and multi-lateral agreements. The five GOOS modules were viewed as devices which served their purpose in the planning stage. However, in the implementation phase, the thrust would be thematic rather than modular, as there is a high degree of commonality in the data needs, data handling, and modelling tools required within each module.

In recommending the establishment of the Steering Group on GOOS, the Oceanography Committee requested it to prepare an Action Plan for how ICES should take an active and leading role in the further development and implementation of GOOS. It requested that input to this process come via the Chairmen of all of the Working Groups under the Oceanography, Marine Habitat, and Living Resources Committees, thus fostering the potential for wide inter-Committee collaboration, as well as establishing the scope of ICES relevance to GOOS. The Committee also recognized that ICES cannot develop a role in GOOS without the active support of its Member Countries, and it hoped that the Steering Group would be able to provide an appropriate stimulus to Delegates to take account of the potential role of ICES when they make their commitments at the First GOOS Agreement Meeting, to be held in conjunction with the IOC Assembly in 1999. The aim of this meeting is to obtain the greatest possible international interest in and commitment to GOOS. This is intended to be a meeting on a high political level. It is important that ICES, in due time before that meeting, has clarified what role it would like to play in GOOS directly as well as through ICES coordinated national contributions to GOOS.

The ACME considered that the ICES Environmental Report (see Section 6.2, above) may form a nucleus for developing operational GOOS products by ICES. The hindcast and nowcast descriptions of environmental conditions will need to be made at different geographical and temporal scales for different purposes. Descriptions of climate oscillations or changes may appropriately be done for the whole ICES area, whereas the impacts of climatic variability on biological production and fish stocks need to be addressed more specifically at the regional or ecosystem level. Operational fisheries oceanography is expected to facilitate the further integration of environmental and fisheries issues for the ICES advisory functions. This is the case for the better use of environmental information in assessments and predictions of fish stocks, as well as for the assessment of environmental impacts by human activities (e.g., fisheries, pollution) against the backdrop of natural variability.

#### Reference

ICES. 1997. Report of the ICES Advisory Committee on the Marine Environment, 1997. ICES Cooperative Research Report, 222: 110–111.

### 16.2 Global Ocean Ecosystem Dynamics (GLOBEC)

#### Request

There is no specific request; this is part of continuing ICES work on marine issues.

#### Source of the information presented

The 1998 reports of the Workshop on Prediction and Decadal Scale Ocean Climate Fluctuations of the North Atlantic, the Workshop on Application of Environmental Data in Stock Assessment, the Working Group on Cod and Climate Change (WGCCC), the third ICES/GLOBEC Backward Facing Workshop, and the ICES/GLOBEC North Atlantic Regional Coordination Group, and ACME deliberations.

#### Status/background information

Much of the work carried out by the above-mentioned groups has been to evaluate how fish stocks are affected by environmental factors at scales from hours to decades, what the possibilities are for predicting the consequences of environmentally induced changes on fish stocks, and

how to improve the quality of advice on fisheries by applying information about the environment.

The ICES/GLOBEC programme is the North Atlantic regional component of the International GLOBEC programme. The Workshops and Theme Sessions on GLOBEC-related issues have attracted many participants from outside the normal ICES community. To date, the applications for the science carried out within ICES/GLOBEC have been mainly on fisheries, but the programme has always acknowledged the part it could play on other marine ecosystem issues. One of these is by

helping with the design and interpretation of monitoring for programmes such as the Living Marine Resources module of GOOS.

*Need for further research or additional data*

The 1998 meeting of the Working Group on Cod and Climate Change set out a five-year plan for further research, which will be considered by the Oceanography Committee. It will also be included in the International GLOBEC Implementation Plan, which is scheduled for publication in late 1998.

## ANNEX 1

### GUIDELINES FOR THE DETERMINATION OF POLYCYCLIC AROMATIC HYDROCARBONS IN BIOTA

#### 1 INTRODUCTION

Polycyclic aromatic hydrocarbons (PAHs) consist of a variable number of fused aromatic rings. By definition, PAHs contain at least three fused rings, although in practice related compounds with two fused rings (such as naphthalene and its alkylated derivatives) are often determined and will be considered in these guidelines. PAHs arise from incomplete combustion processes and from both natural and anthropogenic sources, although the latter generally predominate. PAHs are also found in oil and oil products, and these include a wide range of alkylated PAHs formed as a result of diagenetic processes, whereas PAHs from combustion sources comprise mainly parent (non-alkylated) PAHs. PAHs are of concern in the marine environment for two main reasons: firstly, low-molecular weight (MW) PAHs can be directly toxic to marine animals; secondly, metabolites of some of the high-MW PAHs are potent animal and human carcinogens—benzo[*a*]pyrene is the prime example. Carcinogenic activity is closely related to structure, however, and benzo[*e*]pyrene and four benzofluoranthene isomers (all six compounds have a molecular weight of 252 Da) are much less potent. Some compounds (e.g., heterocyclic compounds containing sulphur, such as benzothiophenes and dibenzothiophenes) may also cause taint in commercially exploited fish and shellfish and render them unfit for sale.

PAHs are readily taken up by marine animals both across gill surfaces and from their diet, and may bioaccumulate, particularly in shellfish. Filter-feeding organisms such as bivalve molluscs can accumulate high concentrations of PAHs, both from chronic discharges to the sea (e.g., of sewage) and following oil spills. Fish are exposed to PAHs both via uptake across gill surfaces and from their diet, but do not generally accumulate high concentrations of PAHs as they possess an effective mixed-function oxygenase (MFO) system which allows them to metabolize PAHs and to excrete them in bile. An assessment of the exposure of fish to PAHs therefore requires also the determination of PAH-metabolite concentrations in bile samples, as turnover times can be extremely rapid. Thus, the analysis of PAHs in fish muscle tissue should normally only be undertaken for food quality assurance purposes (Law and Biscaya, 1994).

There are marked differences in the behaviour of PAHs in the aquatic environment between the low-MW compounds (such as naphthalene; 128 Da) and the high-MW compounds (such as benzo[*ghi*]perylene; 276 Da) as

a consequence of their differing physico-chemical properties. The low-MW compounds are appreciably water soluble and can be bioaccumulated from the 'dissolved' phase by transfer across gill surfaces, whereas the high-MW compounds are relatively insoluble and hydrophobic, and can attach to both organic and inorganic particulates within the water column. PAHs derived from combustion sources may actually be deposited to the sea already adsorbed to atmospheric particulates, such as soot particles. The majority of PAHs in the water column will eventually be either taken up by biota or transported to the sediments, and deep-water depositional areas may generally be regarded as sinks for PAHs, particularly when they are anoxic.

#### 2 APPROPRIATE SPECIES FOR ANALYSIS OF PAHs

##### 2.1 Benthic Fish and Shellfish

All teleost fish have the capacity for rapid metabolism of PAHs, thereby limiting their usefulness for monitoring temporal or spatial trends of PAHs. Shellfish (particularly molluscs) generally have a lesser metabolic capacity towards PAHs, and so they are preferred because PAH concentrations are generally higher in their tissues.

For the purposes of temporal trend monitoring, it is essential that long time series with either a single species or a limited number of species are obtained. Care should be taken that the sample is representative of the population and can be repeated annually. There are advantages in the use of molluscs for this purpose as they are sessile, and so reflect the degree of contamination in the local area to a greater degree than fish which are mobile. The analysis of fish tissues is often undertaken in conjunction with biomarker and disease studies, and associations have been shown between the incidence of some diseases (e.g., liver neoplasia) in flatfish and the concentrations of PAHs in the sediments over which they live and feed (Malins *et al.*, 1988; Vethaak and ap Rheinallt, 1992). The exposure of fish to PAHs can be assessed by the analysis of PAH-metabolites in bile, and by measuring the induction of mixed-function oxygenase enzymes which effect the formation of these metabolites. At offshore locations, the collection of appropriate shellfish samples may be problematic if populations are absent, sparse or scattered, and the collection of fish samples may be simpler. Generally the analysis of PAHs in fish muscle tissue should only be considered for the purposes of food quality assurance.



## 2.2 First Choice Shellfish Species

### *Mytilus* spp. (mussel)

The blue mussel (*Mytilus edulis*) occurs in shallow waters along almost all coasts of the Northeast Atlantic. It is therefore suitable for monitoring in nearshore waters. No distinction is made between *M. edulis* and *M. galloprovincialis* because the latter species, which may occur along Spanish and Portuguese coasts, fills a similar ecological niche. A sampling size range of 30–70 mm shell length is specified to ensure availability throughout the whole maritime area. In some areas (e.g., the Barents Sea), other species may be considered. Recent monitoring studies have indicated a seasonal cycle in PAH concentrations (particularly for combustion-derived PAHs) in mussels, with maximum concentrations in the winter prior to spawning and minimum concentrations in the summer. It is particularly important, therefore, that samples selected for trend monitoring and spatial comparisons are collected at the same time of year, and preferably in the first months of the year before spawning.

## 3 SAMPLING

Two alternative sampling strategies can be used: sampling to minimize natural variability and length-stratified sampling. Only details of length-stratified sampling are described in this document, as this strategy has been used in monitoring programmes for temporal trends of contaminants in biota in the Northeast Atlantic.

### 3.1 Shellfish

For shellfish, the upper limit of shell length should be chosen in such a way that at least 20 mussels in the largest length interval can easily be found. The length stratification should be determined in such a way that it can be maintained over many years for the purposes of temporal trend monitoring. The length interval shall be at least 5 mm in size. The length range should be split into at least three length intervals (small, medium and large) which are of equal size after log transformation. For example, if the length range is 40–70 mm, then the interval boundaries could be as follows:

small: 40–48 mm shell length;  
medium: 49–58 mm shell length;  
large: 59–70 mm shell length.

### 3.2 Fish

Fish are not recommended for spatial or temporal trend monitoring of PAHs, but can be useful as part of biological effects studies or for food quality assurance purposes. The sampling strategy for biological effects monitoring is described in the OSPAR Joint Assessment and Monitoring Programme (JAMP) Guidelines for Monitoring Contaminants in Biota.

## 4 TRANSPORTATION

Fish samples should be kept cool or frozen (at a temperature of –20 °C or lower) as soon as possible after collection. Live mussels should be transported in closed containers at temperatures between 5 °C and 15 °C, preferably below 10 °C. For live animals it is important that the transport time is short and controlled (e.g., maximum of 24 hours). Frozen fish samples should be transported in closed containers at temperatures below –20 °C. If biomarker determinations are to be made, then it will be necessary to store tissue samples at lower temperatures, for example, in liquid nitrogen at –196 °C.

## 5 PRETREATMENT AND STORAGE

### 5.1 Contamination

Sample contamination may occur during sampling, sample handling, pretreatment and analysis, due to the environment, the containers or packing materials used, the instruments used during sample preparation, and from the solvents and reagents used during the analytical procedures. Controlled conditions are therefore required for all procedures, including the dissection of fish organs on-board ship. In the case of PAHs, particular care must be taken to avoid contamination at sea. On ships there are multiple sources of PAHs, such as the oils used for fuel and lubrication, and the exhaust from the ship's engines. It is important that the likely sources of contamination are identified and steps taken to preclude sample handling in areas where contamination can occur. A ship is a working vessel and there can always be procedures occurring as a result of the day-to-day operations (deck cleaning, automatic overboard bilge discharges, etc.) which could affect the sampling process. One way of minimizing the risk is to conduct dissection in a clean area, such as within a laminar-flow hood away from the deck areas of the vessel. It is also advisable to collect samples of the ship's fuel, bilge water, and oils and greases used on winches, etc., which can be used as fingerprinting samples at a later date, if there are suspicions of contamination in particular instances.

### 5.2 Shellfish

#### 5.2.1 Depuration

Depending upon the situation, it may be desirable to depurate shellfish so as to void the gut contents and any associated contaminants before freezing or sample preparation. This is usually applied close to point sources, where the gut contents may contain significant quantities of PAHs associated with food and sediment particles which are not truly assimilated into the tissues of the mussels. Depuration should be undertaken in controlled conditions and in clean sea water; depuration over a period of 24 hours is usually sufficient. The aquarium should be aerated and the temperature and

salinity of the water should be similar to that from which the animals were removed.

### 5.2.2 Dissection and Storage

Mussels should be shucked live and opened with minimal tissue damage by detaching the adductor muscles from the interior of at least one valve. The soft tissues should be removed and homogenized as soon as possible, and frozen in glass jars at  $-20^{\circ}\text{C}$  until analysis.

When samples are processed, both at sea and onshore, the dissection must be undertaken by trained personnel on a clean bench wearing clean gloves and using clean stainless steel knives and scalpels. Stainless steel tweezers are recommended for holding tissues during dissection. After each sample has been prepared, all tools and equipment (such as homogenizers) should be cleaned.

## 5.3 Fish

### 5.3.1 Dissection and storage

Ungutted fish should be wrapped separately in suitable material (e.g., aluminium foil) and frozen. If plastic bags or boxes are used, then they should be used as outer containers only, and should not come into contact with tissues. Organ samples (e.g., livers) should be stored in pre-cleaned containers made of glass, stainless steel or aluminium, or should be wrapped in pre-cleaned aluminium foil and shock-frozen quickly in liquid nitrogen or in a blast freezer. In the latter case, care should be taken that the capacity of the freezer is not exceeded (Law and de Boer, 1995). Cold air should be able to circulate between the samples in order that the minimum freezing time can be attained (maximum 12 hours). The individual samples should be clearly and indelibly labelled and stored together in a suitable container at a temperature of  $-20^{\circ}\text{C}$  until analysis. If the samples are to be transported during this period (e.g., from the ship to the laboratory), then arrangements must be made which ensure that the samples do not thaw out during transport. Subsamples for biomarker determinations should be collected immediately after death (maximum 1 hour) in order to minimize post-mortem changes in enzymatic and somatic activities, and stored in suitable vials in liquid nitrogen until analysis.

When samples are processed, both at sea and onshore, the dissection must be undertaken by trained personnel on a clean bench wearing clean gloves and using clean stainless steel knives and scalpels. Stainless steel tweezers are recommended for holding tissues during dissection. After each sample has been prepared, all tools and equipment (such as homogenizers) should be cleaned.

### 5.3.2 Subsampling

To sample fish muscle, care should be taken to avoid including any epidermis or subcutaneous fatty tissue in the sample. Samples should be taken underneath the red muscle layer. In order to ensure uniformity, the right side dorso-lateral muscle should be sampled. If possible, the entire right side dorsal lateral fillet should be homogenized and subsamples taken for replicate PAH determinations. If, however, the amount of material to be homogenized would be too large, a specific portion of the dorsal musculature should be chosen. It is recommended that the portion of the muscle lying directly under the first dorsal fin is used in this case.

When dissecting the liver, care should be taken to avoid contamination from the other organs. If bile samples are to be taken for PAH-metabolite determinations, then they should be collected first. If the whole liver is not to be homogenized, a specific portion should be chosen in order to ensure comparability. Freeze-drying of tissue samples cannot be recommended for PAH determination, due to the contamination which may result from back-streaming of oil from the rotary pumps used to generate the vacuum.

When pooling of tissues is necessary, an equivalent quantity of tissue should be taken from each fish, e.g., 10 % from each whole fillet.

## 6 ANALYSIS

### 6.1 Preparation of Materials

Solvents, reagents and adsorptive materials must be free of PAHs and other interfering compounds. If not, then they must be purified using appropriate methods. Reagents and adsorptive materials should be purified by solvent extraction and/or by heating in a muffle oven as appropriate. Glass fibre materials (e.g., Soxhlet thimbles) should be cleaned by solvent extraction, and filter papers should be thoroughly solvent-rinsed before use. It should be borne in mind that clean materials can be re-contaminated by exposure to laboratory air, particularly in urban locations, and so storage after cleaning is of critical importance. Ideally, materials should be prepared immediately before use, but if they are to be stored, then the conditions should be considered critically. All containers which come into contact with the sample should be made of glass, and should be pre-cleaned before use. Appropriate cleaning methods would include washing with detergents, rinsing with water, and finally solvent-rinsing immediately before use. Heating of glassware in an oven (e.g., at  $400^{\circ}\text{C}$  for 24 hours) can also be useful in removing PAH contamination.

## 6.2 Lipid Determination

Although PAH data are not usually expressed on a lipid basis, the determination of the lipid content of tissues can be of use in characterizing the samples. The lipid content should be determined on a separate subsample of the tissue homogenate, as some of the extraction techniques used routinely for PAH determination (e.g., alkaline saponification) destroy lipid materials. The total fat weight should be determined using the method of Bligh and Dyer (1959) or an equivalent method.

## 6.3 Dry Weight Determination

Generally PAH data are expressed on a wet weight basis, but sometimes it can be desirable to consider them on a dry weight basis. Again, the dry weight determination should be conducted on a separate subsample of the tissue homogenate, which should be air-dried to constant weight at 105 °C.

## 6.4 Extraction and Clean-up

PAHs are lipophilic and so are concentrated in the lipids of an organism, and a number of methods have been described for PAH extraction (see, e.g., Ehrhardt *et al.*, 1991). The preferred methods generally utilize either Soxhlet extraction, or alkaline digestion followed by liquid-liquid extraction with an organic solvent. In the case of Soxhlet extraction, the wet tissue must be dried by mixing with a chemical agent (e.g., anhydrous sodium sulphate), in which case a time period of several hours is required between mixing and extraction in order to allow complete binding of the water in the sample. Alkaline digestion is conducted on wet tissue samples, so this procedure is unnecessary. In neither case can freeze-drying of the tissue prior to extraction be recommended because of the danger of contamination from oil back-streaming from the rotary pump (which provides the vacuum) into the sample. Apolar solvents alone will not effectively extract all the PAHs from tissues when using Soxhlet extraction, and mixtures such as hexane/dichloromethane may be effective in place of solvents such as benzene and toluene, used historically for this purpose. Alkaline digestion has been extensively used in the determination of PAHs and hydrocarbons and is well documented. It is usually conducted in alcohol (methanol or ethanol), which should contain at least 10 % water, and combines disruption of the cellular matrix, lipid extraction and saponification within a single procedure, thereby reducing sample handling and treatment. For these reasons, it should be the method of choice. Solvents used for liquid-liquid extraction of the homogenate are usually apolar, such as pentane or hexane, and they will effectively extract all PAHs.

Tissue extracts will always contain many compounds other than PAHs, and a suitable clean-up is necessary to remove those compounds which may interfere with the subsequent analysis. Different techniques may be used,

both singly or in combination, and the choice will be influenced by the selectivity and sensitivity of the final measurement technique and also by the extraction method employed. If Soxhlet extraction was used, then there is a much greater quantity of residual lipid to be removed before the analytical determination can be made than in the case of alkaline digestion. An additional clean-up stage may therefore be necessary. The most commonly used clean-up methods involve the use of alumina or silica adsorption chromatography, but gel permeation chromatography and similar high performance liquid chromatography (HPLC) based methods are also employed (Nondek *et al.*, 1993; Nyman *et al.*, 1993; Perfetti *et al.*, 1992). The major advantages of using HPLC-based clean-up methods are their ease of automation and reproducibility.

## 6.5 Pre-concentration

The sample volume should be 2 ml or greater to avoid errors when transferring solvents during the clean-up stages. Evaporation of solvents using a rotary-film evaporator should be performed at low temperature (water bath temperature of 30 °C or lower) and under controlled pressure conditions, in order to prevent losses of the more volatile PAHs such as naphthalenes. For the same reasons, evaporation to dryness should be avoided at all costs. When reducing the sample to final volume, solvents can be removed by a stream of clean nitrogen gas. Suitable solvents for injection into the gas chromatograph (GC) or GC-MS include pentane, hexane, heptane and *iso*-octane, whereas for HPLC analyses acetonitrile and methanol are commonly used.

## 6.6 Selection of PAHs to be Determined

The choice of PAHs to be analysed is not straightforward, both because of differences in the range of PAH compounds resulting from combustion processes and from oil and oil products, and also because the aims of specific monitoring programmes can require the analysis of different representative groups of compounds. PAHs arising from combustion processes are predominantly parent (unsubstituted) compounds, whereas oil and its products contain a much wider range of alkylated compounds in addition to the parent PAHs. This has implications for the analytical determination, as both HPLC-based and GC-based techniques are adequate for the determination of a limited range of parent PAHs in samples influenced by combustion processes, whereas in areas of significant oil contamination and following oil spills only GC-MS has sufficient selectivity to determine the full range of PAHs present. The availability of pure individual PAHs for the preparation of standards is problematic and limits both the choice of determinands and, to some degree, the quantification procedures which can be used. The availability of reference materials certified for PAHs is also rather limited. A list of target parent and alkylated PAHs suitable for environmental monitoring is given in Table A1.1, and this differs both

**Table A1.1.** Compounds of interest for environmental monitoring for which the guidelines apply.

Compound	MW	Compound	MW
Naphthalene	128	C <sub>2</sub> -Phenanthrenes/Anthracenes	206
C <sub>1</sub> -Naphthalenes	142	C <sub>3</sub> -Phenanthrenes/Anthracenes	220
C <sub>2</sub> -Naphthalenes	156	Fluoranthene	202
C <sub>3</sub> -Naphthalenes	170	Pyrene	202
C <sub>4</sub> -Naphthalenes	184	C <sub>1</sub> -Fluoranthenes/Pyrenes	216
Acenaphthylene	152	C <sub>2</sub> -Fluoranthenes/Pyrenes	230
Acenaphthene	154	Benz[ <i>a</i> ]anthracene	228
Biphenyl	154	Chrysene	228
Fluorene	166	2,3-Benzanthracene	228
C <sub>1</sub> -Fluorenes	180	Benzo[ <i>a</i> ]fluoranthene	252
C <sub>2</sub> -Fluorenes	194	Benzo[ <i>b</i> ]fluoranthene	252
C <sub>3</sub> -Fluorenes	208	Benzo[ <i>j</i> ]fluoranthene	252
Dibenzothiophene	184	Benzo[ <i>k</i> ]fluoranthene	252
C <sub>1</sub> -Dibenzothiophenes	198	Benzo[ <i>e</i> ]pyrene	252
C <sub>2</sub> -Dibenzothiophenes	212	Benzo[ <i>a</i> ]pyrene	252
C <sub>3</sub> -Dibenzothiophenes	226	Perylene	252
Phenanthrene	178	Indeno[1,2,3- <i>cd</i> ]pyrene	276
Anthracene	178	Benzo[ <i>ghi</i> ]perylene	276
C <sub>1</sub> -Phenanthrenes/Anthracenes	192	Dibenz[ <i>ah</i> ]anthracene	278

from the list previously developed within ICES specifically for intercomparison purposes, and the historic list of Borneff. In both cases, the lists were concentrated on a subset of parent (predominantly combustion-derived) PAHs due to analytical limitations. This approach completely neglects the determination of alkylated PAHs, which allows the interpretation of PAH accumulation from multiple sources including those due to oil inputs. It will not be necessary for all of these PAH compounds and groups to be analysed in all cases, but an appropriate selection can be made from this list depending on the specific aims of the monitoring programme to be undertaken.

## 6.7 Instrumental Determination of PAHs

Unlike the situation for chlorobiphenyls (CBs), where GC techniques (particularly GC-ECD) are used exclusively, two major approaches based on GC and HPLC are followed to an equal extent in the analysis of PAHs. The greatest sensitivity and selectivity in routine analyses are achieved by combining HPLC with fluorescence detection (HPLC-UVF) and capillary gas chromatography with mass spectrometry (GC-MS). In terms of flexibility, GC-MS is the most capable technique, as in principle it does not limit the selection of determinands in any way, while HPLC is suited only to the analysis of parent PAHs. In the past, analyses have also been conducted using HPLC with UV-absorption detection and GC with flame-ionization detection, but neither can be recommended because of their relatively poor selectivity. Both in terms of the initial capital cost of the instrumentation, and the cost per sample analysed, HPLC-UVF is cheaper than GC-MS. With the advent of

high-sensitivity benchtop GC-MS systems, however, this cost advantage is now not as marked as in the past, and the additional information regarding sources available makes GC-MS the method of choice.

Intercomparison exercises have demonstrated a serious lack of comparability between specific hydrocarbon concentrations measured in different laboratories and using both analytical approaches described above (Farrington *et al.*, 1986). An interlaboratory performance study has recently been carried out within the QUASIMEME laboratory testing scheme in order to assess the current level of comparability among laboratories conducting PAH analyses and to identify improvements in methodology, but samples of biota have not yet been distributed in this series of exercises (Law and Klungsøyr, 1996; Law *et al.*, 1998).

Limits of determination within the range of 0.1 to 0.5  $\mu\text{g kg}^{-1}$  wet weight for individual PAH compounds should be achievable by both GC-MS and HPLC-UVF techniques.

## 6.8 HPLC

Reversed-phase columns (e.g., octadecylsilane (RP-18)) 15–30 cm in length are used almost exclusively, in conjunction with gradient elution using mixtures of acetonitrile/water or methanol/water. A typical gradient may start as a 50 % mixture, changing to 100 % acetonitrile or methanol in 40 minutes. This flow is maintained for 20 minutes, followed by a return to the original conditions in 5 minutes and 5–10 minutes'

equilibration before the next injection. The use of an automatic injector is strongly recommended. Also, the column should be maintained in a column oven heated to 10–30 °C. The systems yielding the best sensitivity and selectivity utilize fluorescence detection. As different PAH compounds yield their maximum fluorescence at different wavelengths, for optimum detection of PAHs the wavelengths of the detector should be programmed so that the excitation/emission wavelengths detected are changed at pre-set times during the analytical determination. For closely eluting peaks, it may be necessary to use two detectors in series utilizing different wavelength pairs, or to effect a compromise in the selected wavelengths if a single detector is used. As the fluorescence signals of some PAHs (e.g., pyrene) are quenched by oxygen, the eluents must be degassed thoroughly. This is usually achieved by continuously bubbling a gentle stream of helium through the eluent reservoirs, but a vacuum degasser can also be used. Polytetrafluorethylene (PTFE) tubing must not then be used downstream of the reservoirs as this material is permeable to oxygen; stainless steel or polyetheretherketone (PEEK) tubing is preferred.

## 6.9 GC-MS

The two injection modes commonly used are splitless and on-column injection. Automatic sample injection should be used wherever possible to improve the reproducibility of injection and the precision of the overall method. If splitless injection is used, the liner should be of sufficient capacity to contain the injected solvent volume after evaporation. For PAH analysis, the cleanliness of the liner is also very important if adsorption effects and discrimination are to be avoided, and the analytical column should not contain active sites to which PAHs can be adsorbed. Helium is the preferred carrier gas, and only capillary columns should be used. Because of the wide boiling range of the PAHs to be determined and the surface-active properties of the higher PAHs, the preferred column length is 25–30 m, with an internal diameter of 0.15 mm to 0.3 mm. Film thicknesses of 0.3 µm to 1 µm are generally used; this choice has little impact on critical resolution, but thicker films are often used when one-ring aromatic compounds are to be determined alongside PAHs, or where a high sample loading is needed. No stationary phase has been found on which all PAH isomers can be resolved; the most commonly used stationary phase for PAH analysis is 5 % phenyl methylsilicone (DB-5 or equivalent). This will not, however, resolve critical isomers such as benzo[*b*], [*j*] and [*k*]fluoranthenes, or chrysene from triphenylene. These separations can be made on other columns, if necessary. For PAHs there is no sensitivity gain from the use of chemical ionization (either positive or negative ion), so analyses are usually conducted in electron-impact mode at 70eV. The choice of full-scan or multiple-ion detection is usually made in terms of sensitivity. Some instruments such as ion-trap mass spectrometers exhibit the same sensitivity in both modes, so full-scan spectra are collected, whereas for quadrupole instruments greater

sensitivity is obtained if the number of ions scanned is limited. In that case, the masses to be detected are programmed to change during the analysis as different PAHs elute from the capillary column.

## 7 CALIBRATION AND QUANTIFICATION

### 7.1 Standards

A range of fully-deuterated parent PAHs is available for use as standards in PAH analysis. The availability of pure PAH compounds is limited. Although most of the parent compounds can be purchased as pure compounds, the range of possible alkyl-substituted PAHs is vast and only a limited selection of them can be obtained. In HPLC, where the resolving power of the columns is limited and the selectivity less than that which can be obtained using MS detection, only a single internal standard is normally used (e.g., phenanthrene-*d*<sub>10</sub>), although fluoranthene-*d*<sub>10</sub> and 6-methyl chrysene, among others, have also been used. If GC-MS is used, then a wider range of deuterated PAHs can be utilized, both because of the wide boiling range of PAHs present and because that allows the use of both recovery and quantification standards. Suitable standards could range from naphthalene-*d*<sub>8</sub> to perylene-*d*<sub>12</sub>. Crystalline PAHs of known purity should be used for the preparation of calibration standards. If the quality of the standard materials is not guaranteed by the producer or supplier (as for certified reference materials), then it should be checked by GC-MS analysis. Solid standards should be weighed to a precision of 10<sup>-5</sup> grams. Calibration standards should be stored in the dark because some PAHs are photosensitive, and ideally solutions to be stored should be sealed in amber glass ampoules. Otherwise, they can be stored in a refrigerator in stoppered measuring cylinders or flasks that are gas tight to avoid evaporation of the solvent during storage.

### 7.2 Calibration

Multilevel calibration with at least five calibration levels is preferred to adequately define the calibration curve. In general, GC-MS calibration is linear over a considerable concentration range but exhibits non-linear behaviour when the mass of a compound injected is low due to adsorption. Quantification should be conducted in the linear region of the calibration curve, or the non-linear region must be well characterized during the calibration procedure. For HPLC-UVF, the linear range of the detection system should be large, and quantification should be made within the linear range. External standardization is often used with HPLC due to the relatively limited resolution obtainable with this technique as generally employed.

### 7.3 Recovery

The recovery of analytes should be checked and reported. Given the wide boiling range of the PAHs to be determined, the recovery may vary with compound

group, from the volatile PAHs of low molecular weight to the larger compounds. For GC-MS analysis, deuterated standards can be added in two groups: those to be used for quantification are added at the start of the analytical procedure, whilst those from which the absolute recovery will be assessed are added prior to GC-MS injection. This ensures that the calculated PAH concentrations are corrected for the recovery obtained in each case. In the case of HPLC, where only a single deuterated PAH standard is used, it is more common to assess recovery periodically by carrying a standard solution through the whole analytical procedure, then assessing recovery by reference to an external standard. This technique does not, however, correct for matrix effects, and so may be used in conjunction with the spiking of real samples.

## 8 ANALYTICAL QUALITY CONTROL

Planners of monitoring programmes must decide on the accuracy, precision, repeatability, and limits of detection and determination which they consider acceptable. Achievable limits of determination for each individual component are as follows:

- for GC-MS measurements:  $0.2 \mu\text{g kg}^{-1}$  ww;
- for HPLC measurements:  $0.5\text{--}10 \mu\text{g kg}^{-1}$  ww.

Further information on analytical quality control procedures for PAHs can be found elsewhere (Law and de Boer, 1995). A procedural blank should be measured with each sample batch, and should be prepared simultaneously using the same chemical reagents and solvents as for the samples. Its purpose is to indicate sample contamination by interfering compounds, which will result in errors in quantification. The procedural blank is also very important in the calculation of limits of detection and limits of quantification for the analytical method. In addition, a laboratory reference material (LRM) should be analysed within each sample batch. The LRM must be homogeneous and well-characterized for the determinands of interest within the analytical laboratory. Ideally, stability tests should have been undertaken to show that the LRM yields consistent results over time. The LRM should be of the same matrix type (e.g., liver, muscle, mussel tissue) as the samples, and the determinand concentrations should be in the same range as those in the samples. Realistically, and given the wide range of PAH concentrations encountered, particularly in oil spill investigations, this is bound to involve some compromise. The data produced for the LRM in successive sample batches should be used to prepare control charts. It is also useful to analyse the LRM in duplicate from time to time to check within-batch analytical variability. The analysis of an LRM is primarily intended as a check that the analytical method is under control and yields acceptable precision, but a certified reference material (CRM) of a similar matrix should be analysed periodically in order to check the method bias. The availability of biota CRMs certified for

PAHs is very limited, and in all cases the number of PAHs for which certified values are provided is small. At present, only NIST 1974a (a frozen wet mussel tissue) and NIST 2974 (a freeze-dried mussel tissue) are available. At regular intervals, the laboratory should participate in an intercomparison or proficiency exercise in which samples are circulated without knowledge of the determinand concentrations, in order to provide an independent check on performance.

## 9 DATA REPORTING

The calculation of results and the reporting of data can represent major sources of error, as has been shown in intercomparison studies for PAHs. Control procedures should be established in order to ensure that data are correct and to obviate transcription errors. Data stored on databases should be checked and validated, and checks are also necessary when data are transferred between databases. Data should be reported in accordance with the latest ICES reporting formats.

## 10 ACKNOWLEDGEMENT

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## ANNEX 2

### DETERMINATION OF MONO-, DI- AND TRIBUTYLtin IN SEDIMENTS: ANALYTICAL METHODS

#### 1 INTRODUCTION

This Technical Annex provides advice on the determination of butyltin compounds in total sediment, sieved fractions, and suspended particulate matter. Although monitoring programmes often only include tributyltin, the determination of mono- and dibutyltin compounds is considered here because of their importance for fate and biological effects studies. However, current methodology often fails in the quantitative determination of monobutyltin (and, to a lesser extent, dibutyltin), and methods for optimal leaching/extraction of monobutyltin are still under development. With respect to different sample types, recoveries tend to depend on the nature and contamination level of the sediment. Although including mono- and dibutyltin in these guidelines may seem somewhat premature, the analytical performance for tributyltin (TBT) will surely benefit from all measures necessary to include analysis of mono- and dibutyltin in the analytical procedures.

The determination of TBT in sediments includes extraction, often derivatization, clean up, chromatographic separation, and detection. Care must be taken to use a suitable extraction and clean-up procedure. The extraction and derivatization can easily be incomplete and, therefore, the addition of adequate recovery standards is indispensable for quality assurance. For separation, both GC and HPLC methods can be applied. A wide range of very selective detection methods is available. A schematic overview of the different methods dealt with in these guidelines, according to the most common analytical pathways, is provided in Figure A2.1.

These guidelines are not intended as a complete laboratory manual, but are designed to assist analytical chemists in setting up procedures for the determination of butyltin compounds in sediments. Chemists already performing such analyses are encouraged to critically reconsider their methods and to use these guidelines to improve their procedures and/or the associated quality control measures, where necessary. In addition, guidance can be sought from specialized laboratories.

Whichever procedure is adopted, each laboratory must demonstrate the validity of each step of the procedure used. The use of a second (and different) method, carried out concurrently to the routine procedure, is recommended for that validation. Analyses have to be carried out by experienced staff.

In order to assess the analytical results of butyltin compounds in sediments, covariables must be measured as potential normalizers (e.g., grain-size distribution, organic carbon content, carbonate content).

#### 2 SAMPLING AND STORAGE

Storage of samples is preferably done in glass, but containers of other materials such as polycarbonate or aluminium are also suitable. Nevertheless, possible adsorption of and contamination by organotin compounds need to be checked. Because photochemical degradation during storage has been reported for the aqueous phase (Quevauviller and Donard, 1991), the samples should be protected from light. During transportation, the samples should be kept cool; a temperature of 25 °C should not be exceeded (Seligman *et al.*, 1996). If samples are not analysed within 48 hours after sampling, they should be stored at 4 °C (short-term storage). For longer-term storage, the samples should be placed in a freezer (below -20 °C) with or without freeze drying. Under these conditions, samples can be stored for over a year (Gomez-Ariza *et al.*, 1994).

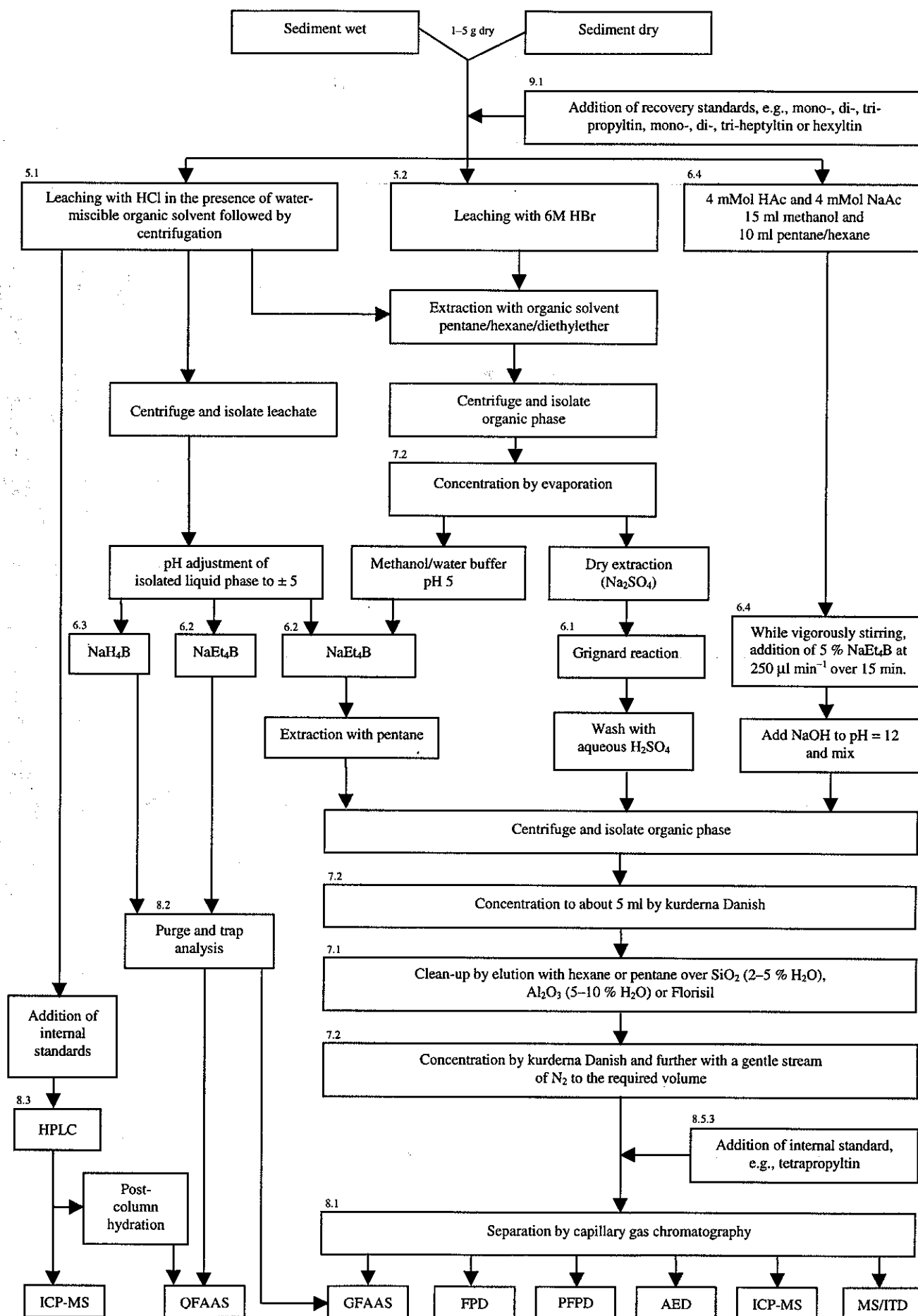
#### 3 BLANKS AND CONTAMINATION

The complete analytical procedure should be checked for blank values. Although butyltin compounds are not likely to occur in the laboratory environment or in solvents or most chemicals, commercial Grignard reagents sometimes contain significant concentrations of various (butyl)tin species. This can be solved by purchasing from other suppliers or by preparing the reagent in the laboratory. There are no procedures known to remove alkyltin compounds from Grignard reagents. Other derivatizing reagents, e.g., sodium tetrahydroborate and sodium tetraethylborate, can also show contamination.

Normal laboratory glassware cleaning procedures are usually sufficient to avoid memory effects and diminish blank values and other interferences. When high blanks do occur, glassware should be treated thoroughly with concentrated HNO<sub>3</sub> and rinsed with organotin-free water and acetone prior to use. In addition, all solvents, chemicals, and adsorptive materials should be checked to localize the source of contamination or interference and measures must be taken accordingly (e.g., cleaning, different brand of chemicals, etc.). Typical procedural blank values are 0.3 ng Sn per compound or less.



**Figure A2.1.** Schematic view of the most common analytical pathways (the numbers on the boxes refer to sub-sections of the text).



Before taking a subsample for analysis, samples should be sufficiently homogenized. Especially samples from ship-docking areas can contain paint particulate matter irregularly distributed in the sample, thereby affecting the representativity of the subsample. This can only be avoided when intensive mixing techniques (ballmill) are applied. Homogeneity can be checked by analysing several subsamples (e.g., five). Sediment samples from the marine environment are more homogeneous than those from harbour areas, as contamination in marine sediments usually derives from the water phase as mediated by tidal water movements. Less polluted samples are often more homogeneous than highly polluted samples. Because the size of the sample intake for analysis is inversely related to the pollution level, the intake will be small when the risk for heterogeneity is high. For this reason, multiple analyses might be appropriate for the higher concentration levels. The sample intake is usually around 1–5 g (dry weight), but some methods do not allow the use of more than 1 g.

Most extraction methods can deal with wet as well as dry samples. Analysis of wet samples saves laborious drying procedures, but dry samples are more easily homogenized and stored. In general, butyltins can often be analysed in the same sample used to monitor other substances such as PCBs. Since mono-, di-, and tributyltin are ionic compounds and strongly sorbed to the sediment, there is no danger of losses through evaporation during air drying or freeze drying. Air drying has been reported possible up to 50 °C, but because other related compounds (i.e., phenyltins) decompose, freeze drying is preferred (Gomez-Ariza *et al.*, 1994). Whichever drying procedure is used, the suitability with regard to cross-contamination and losses should always be tested (Quevauviller and Donard, 1991).

## 5 LEACHING AND EXTRACTION

Butyltin compounds are strongly bound to particulate matter. The binding forces to the sediment have a dualistic character. Whereas tributyltin is mainly bound by hydrophobic forces, mineral binding dominates for monobutyltin because of its high electrical charge (e.g., the binding characteristic of trace metals). To obtain extraction, the butyltin compounds have to be released from the sediment, i.e., the binding must be diminished and the solubility in the extraction solvent must be maximized.

Several principally different approaches can be applied to extract butyltins from sediments:

- 1) Soxhlet extraction with medium polar solvents or mixtures, with acetic or hydrochloric acid added to promote solubility;

- 2) leaching with acidified (co-solvent assisted) aqueous mixtures followed by an off-line determination of the butyltin compounds in the leachate;
- 3) leaching under acidic conditions with simultaneous extraction of the compounds to an organic phase, as applied with different acids and solvents;
- 4) *in situ* derivatization with simultaneous extraction to an organic phase.

Soxhlet extraction (dichloromethane, hexane/methanol) recovers tributyltin, but the leaching capabilities of the organic solvents used are far from adequate for di- and monobutyltin. Therefore, Soxhlet extraction will not be addressed further. The other techniques mentioned are elaborated below. To maintain a logical order, '*in situ* derivatization' will be discussed as a derivatization technique (see Section 6.4, below) and not as an extraction technique. Furthermore, the use of recovery internal standards (RIS) to check the procedural steps is discussed separately in Section 9, below.

### 5.1 Leaching in Combination with Co-solvents

A decrease in mineral binding can be achieved by the addition of strong acids (e.g., HCl), resulting in positively charged ionic butyltin species. A quite high acid concentration is needed to complete the leaching of monobutyltin. To promote the solubility of tributyltin, a co-solvent (e.g., 50–100 % (m)ethanol) can be added (Cai *et al.*, 1993a, 1993b). However, a high acid concentration in the leachate hinders a direct chromatographic analysis and the acid also interferes when a derivatization for further analysis is applied. The acidity achieved is generally not sufficient for the complete leaching of monobutyltin. In a test, none of ten different acidic leaching procedures examined was able to demonstrate a useful recovery for monobutyltin (Zhang *et al.*, 1991). Despite this disadvantage, the method offers a practical advantage to laboratories already having analytical methods available for surface water analysis. By considering the leachate as a water sample, the same method of quantification as is used for water analysis can be applied. It is a prerequisite to perform the separation of sediment and leachate (centrifugation) in a strongly acidic environment, as re-adsorption will occur when buffering to pH 4 (Cai *et al.*, 1993b). Provided that extra attention is paid to validation with respect to differences in sample matrices, tributyltin and dibutyltin can be determined in this way.

### 5.2 Leaching and Subsequent Extraction to an Organic Phase

When extracting organotin compounds with an organic phase immiscible with water (e.g., DCM, diethylether, hydrocarbons, etc.), much higher acid concentrations (6 M HCl) can be applied without obstructing the

derivatization. High acid concentrations will definitely leach most of the monobutyltin from the sediment, but the high electrical charge of the monobutyltin<sup>3+</sup> ion will not allow complete extraction to an organic phase. Under these strongly acidic conditions, the addition of complexing agents, e.g., tropolone (2-hydroxy-2,4,6-cycloheptatrienone) or diethyldithiocarbamate (Zhang *et al.*, 1991; Quevauviller, 1996) is not expected to have much effect as, just like the sediment, the agent will be protonated and consequently lose (much of) its complexation ability. When applied, the effectiveness of complexing agents should be critically evaluated. Furthermore, large amounts of agents in the extract may affect the chromatography. Quantitative extraction of all butyltin compounds to pentane is possible only under strongly acidic conditions when HBr (6 M) is used (Gomez-Ariza *et al.*, 1995). The presence of bromide ions is essential to promote the extraction to the organic phase (pentane) through the formation of neutral ion-pairs. For tributyltin, it was shown that the distribution coefficient between octanol and water increased from 10<sup>2</sup> to 10<sup>6</sup> after the addition of 1 M bromide (Weidenhaupt, 1995).

Gomez-Ariza *et al.* (1995) used a 'sediment:6 M HBr: pentane' ratio of 1:5:10 g/v/v for extraction. The leaching time was set at one hour, followed by an extraction of one hour. For completeness a second extraction with pentane is recommended. The pentane extract obtained can safely be concentrated, as the ionic butyltin compounds will not evaporate easily. This low risk for evaporation also allows transfer to other solvents if required for derivatization or analysis. The residue can be subjected to chromatographic methods such as high performance liquid chromatography (HPLC) that directly analyse the butyltin compounds in their ionic form. For gas chromatographic (GC) methods, the butyltin compounds are derivatized to their hydride or tetra-alkyl form.

## 6 DERIVATIZATION

### 6.1 Grignard Reaction

A Grignard reaction with alkylmagnesium bromide (Grignard reagent) can only be applied in aprotic solvents (hydrocarbons or ethers) and is therefore only applicable after extraction with an apolar solvent (see Section 5.2). If dichloromethane or chloroform are used as extraction solvents, they should be replaced. The organic extract must be dried over Na<sub>2</sub>SO<sub>4</sub> before the alkylation of the butyltin compounds with the Grignard reagent takes place. It is possible to alkylate with any chain length, however, for practical reasons pentyl- or hexylmagnesium bromides are commonly used for derivatization. The resulting compounds are sufficiently volatile for gas chromatography and, on the other hand, their boiling points are sufficiently high to allow concentration steps without excessive risk of losses.

Commercial Grignard reagents are usually dissolved in diethylether and are sealed under an inert atmosphere. Exposure to water or air can result in spontaneous reactions. For this reason, vessels containing Grignard reagents should be purged with nitrogen whenever open to the atmosphere.

Alkylmagnesium bromide concentrations in the reagent are typically 1–2 mol l<sup>-1</sup>. For a 5 g sample intake, 0.5 to 2 mmoles are added to the extract. After 10–60 minutes' reaction time, excess reagent is removed by shaking with 2–5 ml of 0.25 mol l<sup>-1</sup> aqueous sulphuric acid. The resulting tetraalkyltins are stable under appropriate storage conditions (dark, cool) for extended periods of time (at least 10 weeks).

### 6.2 Reaction with Sodium Tetraethylborate

The ethylation with sodium tetraethylborate can take place in an aqueous or (m)ethanolic environment. The reagent can be dissolved in water, methanol, or tetrahydrofuran. Although solutions in water have been shown to be stable for about one month at 4 °C, it is recommended to use only freshly prepared solutions. Solutions of the reagent in an organic solvent (e.g., tetrahydrofuran, methanol or ethanol) seem to be more stable. Typical concentrations for the reagent are 1–2 % sodium tetraethylborate and about 0.1 ml to 1 ml reagent is added for derivatization.

Because derivatization takes place in an aqueous phase, it can be used following almost all leaching or extraction techniques (Section 5.1). To apply ethylation with sodium tetraborate to an organic phase immiscible with water (Section 5.2), the solvent must be exchanged, for example, by ethanol. This can be performed by reducing the volume through evaporation to about 0.5 ml and dissolving the residue in a few millilitres of methanol. The derivatization is very rapid and takes place at a pH between 4 and 6. At a pH lower than 4, the sodium tetraethylborate decomposes more rapidly, as it reacts with the H<sup>+</sup> ions, and at a higher pH the butyltins turn to the hydroxide form (Weidenhaupt, 1995). Usually an acetate buffer is used to adjust the pH to around 5. Considering the high acid concentration in the leachates, care must be taken that the amount of buffer is sufficient. The ethylated butyltins can then be purged and trapped to a GC column with subsequent analysis (Cai *et al.*, 1993b) or extracted to an organic phase for off-line clean up followed by GC analysis.

### 6.3 Hydride Derivatization

The derivatization to hydrides, using sodium tetrahydroborate, is usually done to volatilize the butyltins for purge and trap analysis. As for ethylation, the pH should not be very low, so as to prevent rapid decomposition of the reagent by the acid. Within the pH range 4–11, no significant dependence in derivatization

efficiency is observed (Matthias, 1996). The method can be applied to leachates (Section 5.1) and extracts (Section 5.2) when transferred to an aqueous methanol solution. Sodium tetrahydrogenborate solutions have limited stability and are often stabilized with sodium hydroxide or used freshly prepared.

Extraction of hydrides to an organic phase for subsequent processing is possible, but loss of the monobutyltin hydride during the concentration steps is unavoidable because of its high volatility. With special care, the recovery of the other butyltin compounds might be under control but this will not lead to a robust procedure. Furthermore, it has been suggested that hydride derivatives may degrade more rapidly (Cai *et al.*, 1993b), which also does not favour the application of extraction with off-line analysis.

#### 6.4 Simultaneous Derivatization and Extraction

Using sodium tetraethylborate, the derivatization and extraction can be performed simultaneously. Water buffered at pH 4–5 and a non-polar solvent (e.g., hexane) are added to the wet or dry sediment. After the addition of sodium tetraethylborate (e.g., 0.1–1 ml of 2 % solution in water), the mixture is shaken vigorously (Wilken *et al.*, 1994). Although ethylation in the aqueous phase is very fast, the derivatization is limited by the desorption kinetics. Butyltins are strongly sorbed to the sediment and are not in solution at pH 4–5. Basically, the isolation is an on-going process of desorption and derivatization followed by extraction. Through the derivatization of butyltin ions, other butyltin ions will desorb to the aqueous phase to restore the equilibrium and will subsequently be derivatized and extracted. The transfer of tributyltin from the sediment to the aqueous phase can be improved by the addition of methanol. Mono- and dibutyltin are bound like trace metals and their release from the sediment will only be increased by lowering the pH. However, this is not an option, as the acid will degrade the reagent within seconds. Even at pH 5, the lifetime of the reagent is a matter of minutes. Considering that in five minutes only 70 % of the monobutyltin is desorbed with 6 M HBr (Gomez-Ariza *et al.*, 1995), it is clear that at pH 5 this process will take more time, and the recovery of monobutyltin will not be complete after a single addition of reagent due to its limited lifetime. Multiple additions have been applied, but a continuous addition of the reagent using a peristaltic pump supported by effective mixing conditions is more appropriate. In this way, the reagent is always present and every butyltin molecule desorbed from the sediment is immediately derivatized and extracted, which also makes the desorption process continuous. However, this very intensive derivatization may lead to the formation of boroxin, a six-angle ringed ethylborane. This compound is very reactive to the (bonded) phases used in gas chromatographic columns, affecting the column efficiency and mass spectrometric (MS) response. The boroxin is not removed by the normal phase column

clean-up procedure usually applied, but can be degraded by the addition of an alkaline aqueous solution with a pH above 12. Ethylated butyltin compounds will not be affected.

Although optimal conditions are still under investigation, continuous addition of 5 % sodium tetraethylborate for 15 minutes at 250  $\mu\text{l min}^{-1}$  (in total, 4 ml) under vigorous stirring seems appropriate for most sample types. Since organic matter also reacts with the sodium tetraethylborate, the amount of sample that can be used is limited. As a rule of thumb, the sample intake should represent about 20–50 mg organic carbon which is, in practice, 1 g fine material (dry weight). To the sample, 10–15 ml of a mixture containing an 0.2–0.5 M acetate buffer of pH 5 and 70 % methanol is added. Extraction takes place with about 10 ml of hexane. After the derivatization and while continually stirring, the pH is brought above 12 with a sodium hydroxide solution to decompose the boroxin. The extraction of the derivative itself is quantitative but to isolate the whole organic phase, a second extraction is necessary. Usually centrifugation is required to separate the phases.

### 7 CLEAN UP AND CONCENTRATION

#### 7.1 Clean Up

Whether a clean-up step must be applied depends on the sample type, separation (GC or LC), and detection method used. Furthermore, the nature of the extract determines whether a clean-up step is possible. In the literature, no clean-up procedures are reported for aqueous/methanol leachates (Section 5.1). Clean up is not necessary when the butyltin compounds are determined by purge and trap analysis, which acts as a superb clean up. However, extraction methods using an organic solvent (Section 5.2) will co-extract many kinds of other compounds from the sample, such as sulphur and sulphur-containing compounds, oil, and many other natural and anthropogenic compounds. In addition to co-extracted substances, the extract will contain by-products of the derivatization. Using sodium tetraethylborate for derivatization (Sections 6.2 and 6.4), compounds such as boroxin, diethylsulphide, and diethyltrisulphide can be formed in rather large quantities. If the basic wash, as described in Section 6.4, has not yet been conducted, it should be added here as a clean-up step. The ethylsulphides usually do not disturb the instrumental analysis. Also, co-extracted substances usually do not visually disturb the chromatogram because most detection methods are very selective. Nevertheless, a large amount of matrix in the sample can affect the chromatography when the loading capacity of the column is exceeded, and can influence the detector response (e.g., MS). A decrease in the amount of matrix is always favourable for instrumental analysis and therefore a clean up is recommended.

Generally, a simple SiO<sub>2</sub>, Al<sub>2</sub>O<sub>3</sub> or Florisil column clean up is sufficient. Alkylated tin compounds are as non-polar as PCBs and elute rapidly with hexane. Nevertheless, highly activated materials are not recommended, as the organotin compounds may degrade during elution. Using 2 g of SiO<sub>2</sub> deactivated with 1–5 % water or Al<sub>2</sub>O<sub>3</sub> with 5–10 % water in a glass column, butyltin compounds usually elute in 5–10 ml hexane or pentane. Elution patterns should always be checked for each batch of column material.

## 7.2 Concentration

Prior to and after clean up, extracts are usually concentrated by evaporation of the solvent. Ethylated butyltin compounds are more sensitive to losses than the pentyl or hexyl derivatives. Evaporation can be done either by Kuderna Danish or rotary evaporator. Although Kuderna Danish is the preferred method, a rotary evaporator can be used but it may be problematic with the rather volatile ethylated butyltin derivatives. When applied, care should be taken to stop the evaporation in time at about 5 ml. To reduce this volume further, a gentle stream of nitrogen should be applied. In order to avoid losses, the extract should never be evaporated to dryness. Since evaporation is very critical, the number of concentration steps should preferably be limited. For the extract to be transferred to the clean-up column, concentration to about 5 ml is safer with respect to losses than concentrating to a very small volume. Although applying a volume of 5 ml to the clean-up column seems in conflict with chromatographic principles, one should consider that only a simple removal of the most polar compounds has to be achieved, without any chromatographic aspiration. The possible small advantage of excellent chromatography in the clean up does not counterbalance the risk of losses through evaporation. Only in the final concentration step is the extract carefully concentrated to the required volume for instrumental analyses. By the addition of an amount of *iso*-octane as a keeper, the risk of evaporation to dryness is limited.

## 8 INSTRUMENTAL ANALYSIS

### 8.1 Gas Chromatography

Direct separation of organotin halides using GC is possible but not recommended, as the quantification is very poor due to sorption and unpredictable rearrangement reactions. For derivatized organotin compounds, gas chromatography has proved to be a much more robust technique. Derivatized extracts are mainly analysed using capillary columns, while purge and trap analysis using packed columns is commonly applied to leachates.

### 8.1.1 Injection techniques

Extracts containing derivatized organotins are injected by commonly used systems such as splitless injection, on-column injection, temperature-programmed or pressure-programmed injection. All techniques should be optimized thoroughly prior to use. A typical injection temperature for splitless injection is 250 °C, while the column temperature should be at or below the boiling point of the solvent to obtain trapping by solvent condensed in the column. For on-column injection, an initial oven temperature at or slightly above the boiling point of the solvent is appropriate. Considering that many detectors are mass sensitive to elemental tin, it is advantageous to use a non-discriminating injection technique such as on-column injection (Section 8.1.3). The peak areas of the different compounds are then directly proportional to the amount of tin, which thus favours quality assurance of the calibrants. To obtain reproducible injections, the use of an autosampler is essential. At the same time, constant equilibrium times are achieved that will decrease the variation in retention times.

### 8.1.2 Separation

For alkylated tin compounds that differ in carbon number, the chromatographic conditions to obtain separation are not critical. However, after derivatization equal carbon numbers can occur. For example, tetrapropyltin, a popular internal standard, and ethylated dibutyltin have the same number of carbon atoms. Nevertheless, normally available non-polar or semi-polar chromatographic capillary columns are able to separate these compounds. A typical column applied for separating organotin compounds has a length of 25 m to 30 m, an inner diameter of 0.25 mm, and a film thickness of 0.1 µm to 0.3 µm.

The carrier gas depends on the detector used. For GC/MS, helium must be applied, while both hydrogen and helium can be used for GC/AAS, GC/FPD, GC/PFPD and GC/AED. When using columns with very small inner diameters (< 0.15 mm), the use of hydrogen is essential.

The oven temperature programme has to be optimized for sufficient separation of organotin compounds. A reproducible temperature programme is important for constant retention times and, consequently, for accurate analysis.

### 8.1.3 Detection

After separation, many detectors can be used to quantify organotin compounds and, except for MS, all of them are

tin-specific as well as mass sensitive, which means that peak areas are proportional to the amount of tin. All detection methods have, in principle, sufficient sensitivity, differing by not much more than an order of magnitude. Detection limits are merely dependent on the concentration in the procedure and the sample capacity of the chromatographic system. The detectors used for organotin compounds are described below.

- a) The connection between the GC and the graphite- or quartz-furnace atomic absorption spectrometer (GFAAS and QFAAS, respectively) is usually custom-made from already existing equipment for trace metal analysis. Tin is quantified from absorption at 286.3 nm or 244.6 nm.
- b) A flame photometric detector (FPD) measures the fluorescence of tin atoms in a hydrogen-rich flame. Selectivity for tin is obtained by selection of the wavelength using a cut-off or interference filter of 610 nm. An FPD is a specially developed gas chromatographic detector, which is available from different suppliers. High concentrations of sulphur or sulphur-containing compounds may disturb the signal.
- c) Recently, a pulsed-FPD (PFPD) for GC has been introduced, that demonstrates one order of magnitude higher sensitivity (Jacobsen *et al.*, 1997). Also, a lower matrix dependence is claimed.
- d) The atomic emission detector (AED) is probably the most universal GC detector for the analysis of volatile organometals and is, at the same time, very specific. In a miniaturized helium plasma, the emission of the metals is determined as a measure of the concentration.
- e) Using mass spectrometry (MS) or ion trap detection (ITD), fragments of the organotin derivatives can be quantified. To confirm identity, usually 2–4 specific fragments are measured.
- f) With an inductively-coupled plasma as the ion source for a mass spectrometer (ICP-MS), tin can be determined; it is both tin-selective and very sensitive.

## 8.2 Purge and Trap Analysis

From aqueous leachates (see Section 5.1), the organotin compounds are usually transferred to the GC column after derivatization (hydration or ethylation) using a purge and trap technique. Before addition of the derivatization reagent, the pH should be adjusted to about 5 (see Section 6.2). Compounds are purged by helium and trapped on a packed (e.g., Chromosorb coated with OV-3 silicone oil) U-shaped column placed in liquid nitrogen. After trapping, the compounds are eluted in the order of their boiling points by heating the column. Detection can be performed by several detectors, but purge and trap systems usually apply graphite- or quartz-furnace atomic absorption spectrometry. Cai *et al.*

(1993a) encountered a critical interference for tributyltin using hydration and recommended the use of ethylation for the determination of di- and tributyltin.

## 8.3 High Performance Liquid Chromatography (HPLC)

### 8.3.1 Injection

Injection can be performed by the systems commonly used in HPLC. An autosampler is preferred, but in isocratic separations manual injection can be used as well. With gradient elution, an autosampler is essential to attain constant equilibrium times, ensuring equal initial conditions for each injection.

### 8.3.2 Separation

The advantage of HPLC is that it permits the separation of underivatized ionic organotin compounds. Separation techniques were reviewed by Attar (1996) and include normal and reverse-phase, ion-pair and ion-exchange chromatography.

Retention mechanisms based on adsorption, both normal and reversed phase, have encountered problems associated with the strong adsorption of the organotin compounds to residual silanol groups. To promote the transfer of the target compounds to the mobile phase, concurrent complexing agents such as tropolone and morin (2',3,4',5,7-pentahydroxyflavone) can be added. Nevertheless, normal-phase separations cannot be recommended because, especially for the monobutyl<sup>3+</sup> ions, very strong retention cannot always be prevented. In the reversed-phase mode, the addition of tropolone seems to be more successful provided that silanol groups are sufficiently end-capped. A typical eluent to separate organotin compounds on a reversed-phase column consists of methanol, water, acetic acid, and tropolone (80 %, 14 %, 6 %, and 0.1 %, respectively). Instead of the addition of complexing agents, ion-pair or dynamic ion-exchange chromatography has also been applied. This has mainly been used for the separation of methyltin compounds; for butyltins no applications have been reported thus far.

It should be realized, when using gradient elution in combination with complexing or ion-pair agents, that the distribution of those agents between the stationary and the mobile phases changes considerably during the gradient, and lengthy equilibration times may be necessary to achieve the same initial conditions again.

The ionic character of organotin species also permits the application of cation-exchange columns. For silica-based ion-exchange (SCX) columns, the eluent is usually an ammonium acetate or citric buffer at a pH of about 6 in a 30–60 % methanol-water mixture. After the elution of tri-

and dibutyltin, the pH is often decreased, e.g., to pH 4, to elute inorganic tin and monobutyltin.

In addition to SCX columns, the use of a resin-based cation-exchange (e.g., sulphonated polystyrene-divinylbenzene) column has been reported, applying an 0.05 M monolithium citrate buffer and 0.004 M oxalic acid in methanol (Schulze and Lehmann, 1994).

It is clear that the presence of complexing agents such as tropolone in extracts or leachates to be analysed with ion-exchange chromatography can seriously affect the chromatographic properties of organotin species.

### 8.3.3 Detection

The two most appropriate ways to determine tin compounds eluting from an HPLC system are post-column hydride generation followed by detection with GFAAS or QFAAS or interfacing the HPLC with ICP-MS. The use of ICP-AES is possible for method development, but the detection limit is generally not sufficient for use in marine environmental monitoring.

### 8.4 Identification

Most detectors used are tin-specific, which ensures that a signal represents a tin-containing compound. Further identification of organotin compounds can be performed by comparing the retention time of the substance in a sample with that of the respective compound in a standard.

Using GC-MS or ITD, confirmation of alkyl substitution can be obtained from mass spectra. In MS, the presence of tin is confirmed by its isotopic pattern. Some characteristic masses may be interfered with by column bleeding or co-eluting compounds; therefore, the total spectrum or, when applying the SIM mode, at least two (preferably four) masses in two characteristic and sensitive clusters must be observed.

### 8.5 Quantification

#### 8.5.1 Peak area or height

Using tin-specific detectors, chromatograms do not show 'compound noise' from co-eluting non-target compounds. In addition, all target compounds are usually baseline separated. Therefore, in general, no advantage of quantification on peak height exists. Furthermore, most tin-selective detectors are, or behave as, mass sensitive (i.e., peak area is independent of column flow), so the area response is proportional to the amount of tin. Peak area is therefore recommended for quantification. However, when the separation of the peaks appears to be incomplete or the peaks show tailing, the use of peak

heights may be preferred. Chromatograms processed by automated integrators should always be visually inspected to ensure that the baseline is set as required.

#### 8.5.2 Calibration curves

Although many detectors used to quantify butyltin compounds show an excellent linearity, this linearity always has to be checked because the injection and the chromatography can influence linearity. Therefore, multilevel calibration is essential. Depending on the linearity of the detection method and the required working range, the calibration levels are selected to be equidistant (e.g., 0, 10, 20, 30, etc.) or through a factorial increase (e.g., 0, 1, 2, 4, 8, etc.). Different mathematical fitting methods can be used. A simple point-to-point calibration may not be the most accurate method, but is certainly a very robust one. Whatever method is applied, the calibration curve obtained must always be checked by recalculating the standards as if they were samples and measuring solutions with concentrations just in the middle between the calibration points. The precision can be determined by comparing measured values with nominal values. When a deviation exceeds 5 %, more levels should be included in the calibration. This inspection can also identify a certain working range. Extracts of samples must then be diluted or concentrated to obtain a signal in this working range.

#### 8.5.3 Internal standards for quantification

Two types of internal standards can be identified:

- 1) underivatized standards, that are added prior to derivatization and extraction to determine the recovery of the whole procedure. These are defined here as recovery internal standards (RIS) and should simply be included in the calibration to be measured as if they were determinands. Details on the use of RIS are given in Section 9, below.
- 2) quantification internal standards (QIS).

In chromatography, signals of target compounds in standard solutions as well as samples are generally related to one or more quantification internal standard to compensate for variations in injection volume or extract volume in the final instrumental analysis. To accomplish this, a fixed concentration of QIS, either by volume or weight, should be added to all calibration solutions and the final extracts of samples.

For gas chromatographic analysis, appropriate QIS not present in the environment are tetrapropyltin or ethylated tri-*iso*-butyltin. Tetraethyltin, which has been detected in the environment, and tetra-alkyltin compounds, that are formed from inorganic tin by alkylation with Grignard reagent or sodium tetraethylborate, should not be used.

#### 8.5.4 Calibrant solutions

In general, two pathways for using standard solutions can be distinguished. They are described below.

##### (1) Alkyltin salts without prior derivatization

From commercially available liquid or crystalline alkyltin salts, standard solutions can be made in (m)ethanol or another solvent depending on the instrumental method used. Care must be taken with monoalkyltin compounds as they tend to decompose. These standard solutions can be used as such in HPLC methods in which alkyltins are separated without (prior) derivatization. In procedures based on the analysis of derivatives, the calibration solutions (at all levels) are treated as samples, either from the beginning or from the moment the derivatization is performed (*in situ* calibration). This is repeated each time a series of samples is analysed. For hydration or ethylation followed by purge and trap analysis there is no other way, but for methods in which the derivatives end in a solution it is also possible to calibrate the chromatographic system using derivatized tin compounds.

##### (2) Derivatized alkyltins

Standard solutions of derivatives can be prepared in the laboratory (see Sections 6.1 and 6.2, above), but many are available from QUASIMEME (pentyl- and ethylated derivatives). When the derivatives are purchased, however, it is always necessary to prepare solutions by derivatization to confirm the concentration of the underivatized standard solutions to be used for recovery tests. Furthermore, derivatives of organotin compounds used as recovery internal standards are not commercially available and have to be prepared from salts in any case. Provided that the initial concentration is high enough, the solution of derivatized compounds can be diluted to obtain the different calibration levels. Ethylated derivatives should be washed with an aqueous solution of 2 M NaOH to remove traces of boroxin.

#### 8.5.5 Measuring sequence

In spite of clean-up steps, the residual matrix may still have some influence on the chromatography when injected in capillary columns. The system will be restored when solutions without matrix are injected, e.g., standards and blanks. Matrix- and non-matrix-containing injections should alternate in the measuring sequence to level out this effect. For this reason also, prior to a series of samples and standards, the system should be equilibrated by injecting at least one sample extract which is ignored in the data processing. No knowledge of matrix influences is available for HPLC. For purge and trap analysis, the matrix may influence the derivatization and the transfer of derivatives to the column (see Section 9.1.2, below).

#### 8.5.6 Limit of detection

No simple ranking of the detection limits of the methods mentioned above can be given. The limit of detection of the method applied can be calculated following QUASIMEME guidelines (Topping *et al.*, 1992). Detection limits depend on the contamination in the blank, the sample matrix, concentrations of interfering compounds, and the sample intake size. In general, GC methods have a lower absolute detection limit than HPLC or purge and trap, but the sample capacity (i.e., injection volume) of the latter two is much higher. Due to the selectivity of most detectors and minimal blank values, low instrumental sensitivity may be compensated for by a larger concentration factor, either by a larger sample intake or further evaporation of the final extract.

To determine butyltin compounds in the fine fraction of marine sediments, the limit of detection for each compound should be about  $1 \text{ ng g}^{-1}$  (as tin on a dry weight basis) or better. For samples 'diluted' with sand, the sample intake size can be increased. For very sandy samples, isolation of the fine fraction by sieving will be required.

### 9 QUALITY ASSURANCE

The uncertainty of an analytical procedure is reflected by:

- the accuracy of measured concentrations of calibrants and internal standards;
- the recovery, and its variation, in extraction, derivatization, and clean up;
- long-term repeatability;
- the results of analyses of certified reference materials (CRMs).

#### 9.1 Recovery

In order to check the entire analytical procedure, it is essential to determine the yield of recovery internal standards (RIS) spiked to the sediment samples before leaching/extraction and/or derivatization. A complete recovery of the spiked amounts only indicates that the procedural steps between the sample preparation and instrumental analysis are adequate. It does not guarantee that extraction is complete for the more aged compounds already present in the sample, but nevertheless complete recovery is a minimum requirement for the assumption that extraction is complete. Another effect that can influence the recovery is decomposition of target compounds. Harsh extraction conditions that improve the recovery of monoalkylated tin compounds can lead to some degradation of higher alkylated tin compounds. Whether this applies to the procedure adopted can be checked by spiking a blank with trialkyltins only and



analysing giving special attention to possible decomposition products such as di- and monoalkyltin compounds, as well as inorganic tin or derivatives when applicable.

### 9.1.1 Measuring recovery using GC and HPLC

In principle, standard addition of butyltins to be analysed should be applied for recovery experiments, but in routine analysis it is more efficient to use tin compounds with other alkyl chains not present in the environment. Target compounds and recovery can then be measured in one extraction. Recoveries usually decrease from trialkyltin to monoalkyltin because binding to sediment is strong for monobutyltin, extraction is not very efficient and, when GC is applied, the derivatization requires more steps than for tributyltin. It is therefore essential to have at least a monoalkyltin as a 'worst case' recovery internal standard. As a counterpart for tributyltin, a trialkyltin compound should be added as a recovery standard. Alkyl chains such as ethyl, propyl, pentyl, hexyl, or heptyl can be applied, but with respect to derivatization the alkyl group should be different from the one used in the derivatization to avoid interference from tetra-alkylated inorganic tin. With regard to the possible evaporation of ethylated derivatives during concentration and considering the often low recovery for monobutyltin, monopropyltin would be the most appropriate recovery standard. When pentylation or hexylation is applied, triethyl- or tripropyltin may be indicators of possible losses through evaporation. When selecting recovery standards, the number of carbon atoms should be considered in relation to the target compounds to avoid possible co-elution.

### 9.1.2 Recovery measurement using purge and trap analysis

Purge and trap analysis is a special case. The hydration or ethylation is performed in the presence of the sample matrix, which can behave differently from a standard solution. Due to the low efficiency of the packed columns usually applied, the use of recovery standards other than the target compounds to check the recovery is not possible. Therefore, analysis standard addition is generally performed in purge and trap analysis. This implies that the leaching/extraction has to be performed twice. Using the excess response due to the addition, the derivatized amount can be read from the calibration curve obtained from standards only. Comparing this amount with the addition allows calculation of the recovery. The extraction and derivatization efficiencies can be distinguished by additions to the sample just before purge and trap analysis.

### 9.1.3 Requirements for the recovery

When derivatized calibrants (see Section 8.5.4(2)) are used for calibration, the recovery found for the spiked

compounds (recovery internal standards and blank standard solution) always applies to the entire analytical procedure. On the other hand, when quantification is carried out by underivatized standards which pass through the whole procedure including derivatization (*in situ* calibration, see Section 8.5.4(1)), the recovery only applies to the extraction step and losses occurring in subsequent steps will go unnoticed. It is therefore recommended that in the latter case a cross-check be made by comparing with already derivatized standards. The analytical procedure performs adequately when recoveries of the entire procedure are not significantly different from 100 % or are above a predefined value, e.g., 80 %. At the current state-of-the-art, a full recovery of monobutyltin often seems not feasible. In such cases, it is recommended to report the recovery with the results obtained to demonstrate its indicative character.

Correction for recovery is strongly advised against as it is most likely not representative of the actual recovery of aged compounds and is only a measure of how well the procedure has been performed. However, when it is local practice to correct for recoveries, three recovery standards (a mono-, di-, and trialkyltin) are required because of the different properties of the three butyltin compounds. The uncorrected values should be reported in brackets to show the elevation due to the recovery correction. Results of analyses that show recoveries lower than 50 % should be rejected or the samples should be re-analysed.

## 9.2 Calibrants and Calibration

Calibration solutions should be prepared from certified compounds, if possible. To do this, the laboratory should have the appropriate equipment and the expertise to handle hazardous substances. A cross-check should always be made, either by the preparation of two independent stock solutions, exchange with a colleague laboratory, or comparing derivatized standards with separately produced underivatized standards after derivatization. The mass sensitivity properties of some tin-specific detectors allow an internal check of calibration standards because the peak areas are directly proportional to the amount of tin, provided that no discrimination occurs during injection.

Underivatized tin compounds are usually dissolved in (m)ethanol or acetone and derivatized compounds in non-polar solvents, e.g., *iso*-octane. Calibration solutions should be stored in ampoules in a cool, dark place. Alkylated standards are stable, but for underivatized monoalkyltins decomposition may occur. This can be detected, first by a lower signal from the blank recovery in the procedure and, secondly, by an increase in the inorganic tin. Possible weight loss during storage should be recorded and, when the loss exceeds 5 %, the standards should be replaced. Alternatively, weight losses may be adjusted by addition of the same solvent

prior to use. For standards containing internal standards, some evaporation does not affect the final results. However, evaporation of recovery and internal standard solutions which have to be added to the samples is directly proportional to a negative shift in the result.

Using a calibration curve prepared with each series of samples (see Section 8.5.4), a reference solution of already derivatized compounds should be analysed within each series to confirm a complete derivatization. When already derivatized compounds are used for quantification in each series, a standard solution of butyltin salts should pass through the entire procedure for the same purpose.

### 9.3 System Performance

The performance of the HPLC or GC system should be monitored by regularly checking the resolution of two closely eluting compounds. To control the injection properties, the ratio of the area response of an early- and a late-eluting compound in the standards can be monitored. A further check is to compare the response (slope of the calibration curve) from series to series. For each series, this response can be compared with the noise level to register the signal-to-noise ratio.

### 9.4 Long-term Stability

One internal reference material (IRM) should be included in each series of samples. Such a laboratory reference sample should be well characterized in its content of target compounds and have a representative matrix composition. It should be stressed that the results for such a reference material are not a measure of recovery or accuracy. The repeated analysis of an IRM indicates whether, in time, comparable data are obtained for replicates. From the results, the long-term variation can be calculated. An additional check for the same reason, but with variable matrix, is duplicate analyses of a sample from another series within each series.

A certified reference material (CRM) should be analysed at least twice a year and each time the procedure is changed. Comparison of the results with the certified value may indicate the accuracy, however, it should be realized that even values of a CRM may be subject to discussion. But even when certified values are not accurate, a CRM serves as a reference point for laboratories to compare their methodology.

In order to assist the comparison of data on an international level, each laboratory analysing sediments should also participate in interlaboratory studies on the determination of butyltin compounds in sediments on a regular basis.

### 9.5 Check List

Given the complexity of the different measures to obtain and ensure adequate quality control, a summary of checkpoints is given below. It is suggested that a record by means of a control chart is kept for the points printed in bold type.

- 1) Possible contamination of storage containers (less than 1 % of the mass in a typical sample).
- 2) Contamination or losses during drying.
- 3) Cross-check for calibration solutions.
- 4) **Comparability with previous series of absolute response, response ratio of the first- and last-eluting compounds and signal-to-noise ratio.**
- 5) Adequate column performance, as shown by the resolution of two adjacent peaks.
- 6) Adequate calibration curve: when measuring standards (between calibration points), the deviation from the nominal values should be less than 5 %.
- 7) The blank level should be appropriately low.
- 8) **The recoveries of a standard solution (with underivatized compounds) passing through the entire analytical procedure should be within the required limits (e.g., 80–120 %).**
- 9) For GC/HPLC, the yield of the recovery internal standards added to each sample should be adequate.
- 10) For purge and trap analysis, the response of the standard addition should comply with that of the standard only for each sample.
- 11) **The results of the analysis of the IRM should fall within the required limits (e.g., two times the standard deviation).**
- 12) The result of the duplicate analysis should comply with that from the other series with respect to long-term variation.
- 13) The result of the analysis of the CRM should not give any reason to take actions concerning methodology.

### 9.6 Reporting of Results

Since the counter-ion of organotin ions is unknown in environmental samples, it is not possible to report results in sediments on a compound basis. Therefore, organotins should be clearly reported as their ions or as Sn in  $\text{ng g}^{-1}$  dry weight.

When the yield of recovery standards or the addition of standards is significantly different from 100 %, the recovery value representative of that compound should be reported between brackets. Furthermore, any other abnormalities should be reported.

## 10 ACKNOWLEDGEMENT

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## ANNEX 3

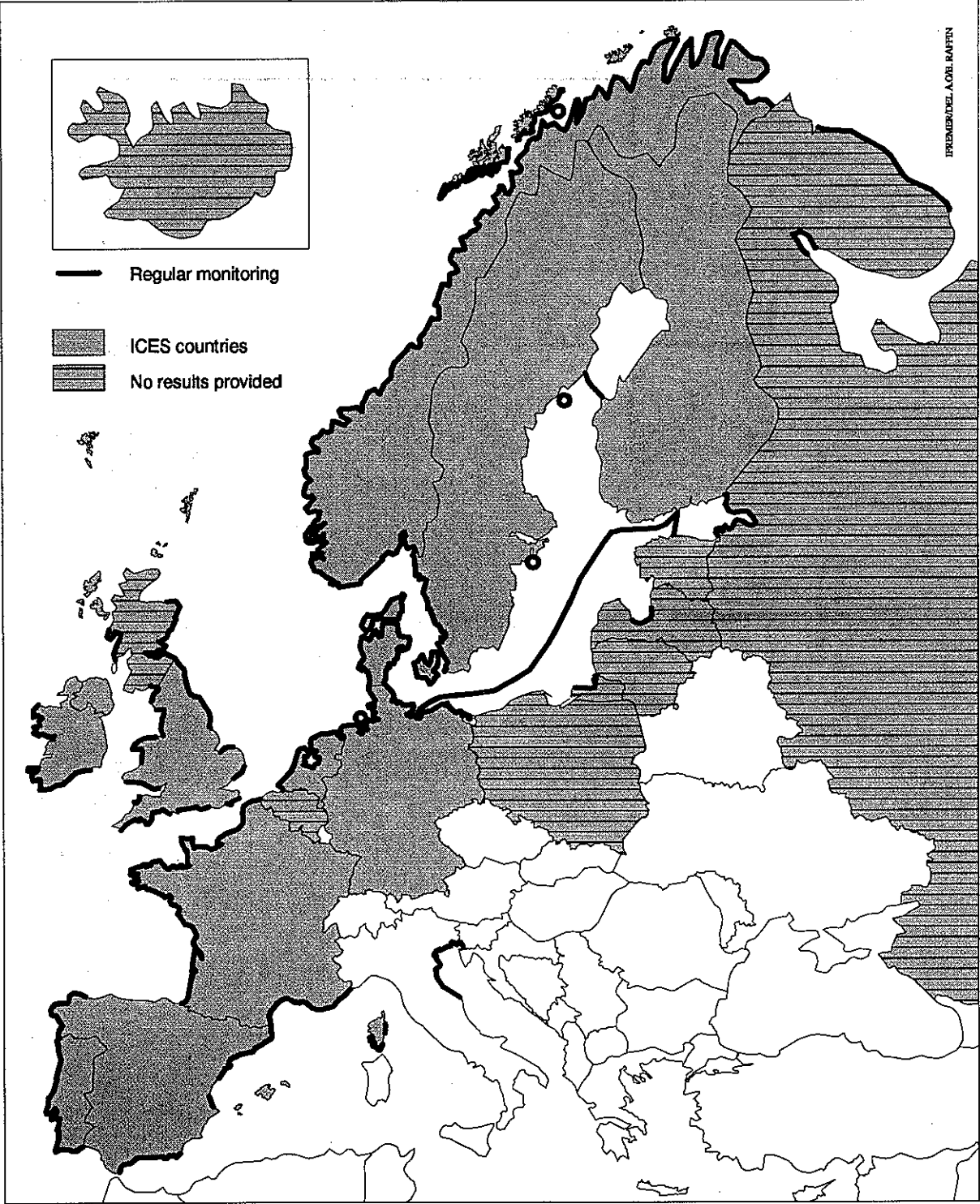
### MAPPING OF HARMFUL EVENTS RELATED TO PHYTOPLANKTON BLOOMS IN ICES MEMBER COUNTRIES

The purpose of this mapping exercise was to obtain a global, geographical overview of harmful events related to phytoplankton blooms in the ICES area for the ten-year period 1988–1997. The work was carried out through the ICES/IOC Working Group on Harmful Algal Bloom Dynamics (WGHABD), under the chairmanship of Dr P. Gentien (IFREMER, Brest, France). 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.

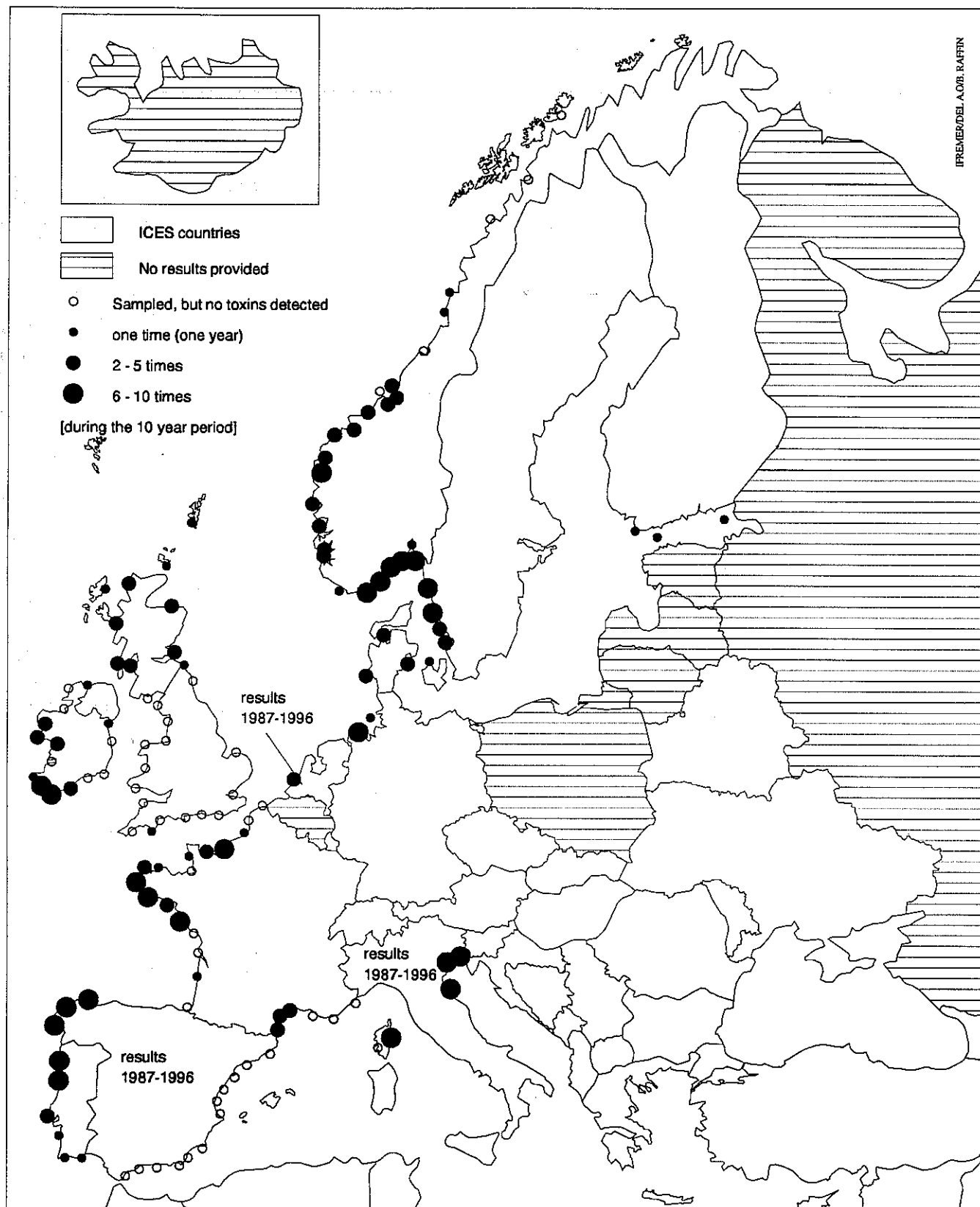
The maps indicate the presence of toxins or observations of animal/plant mortality if detected, regardless of the

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.

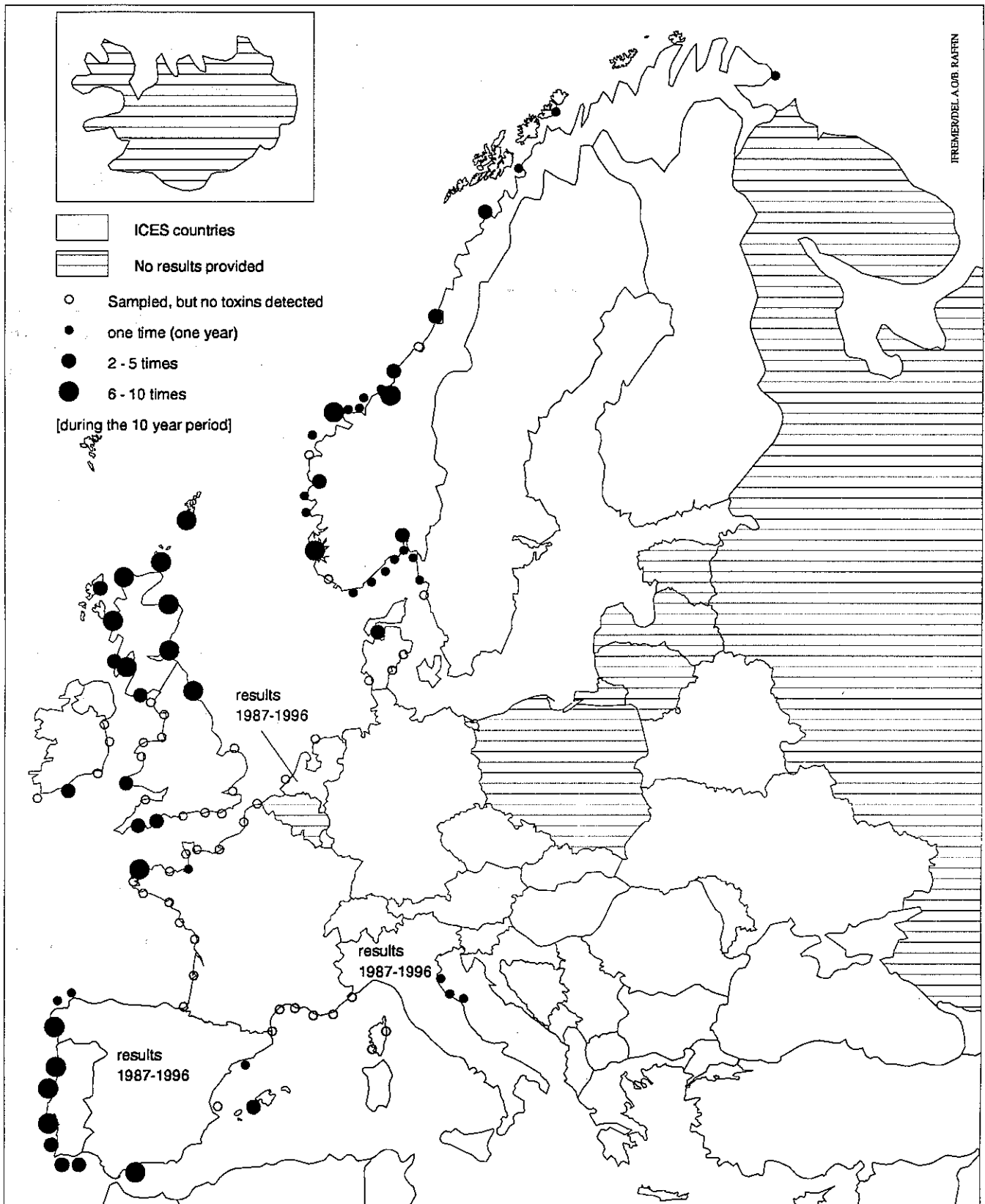
**Figure A3.1.** ICES Member Countries in Europe are indicated by gray shading. Regular monitoring of bloom events is carried out in the areas marked by a heavy black line.



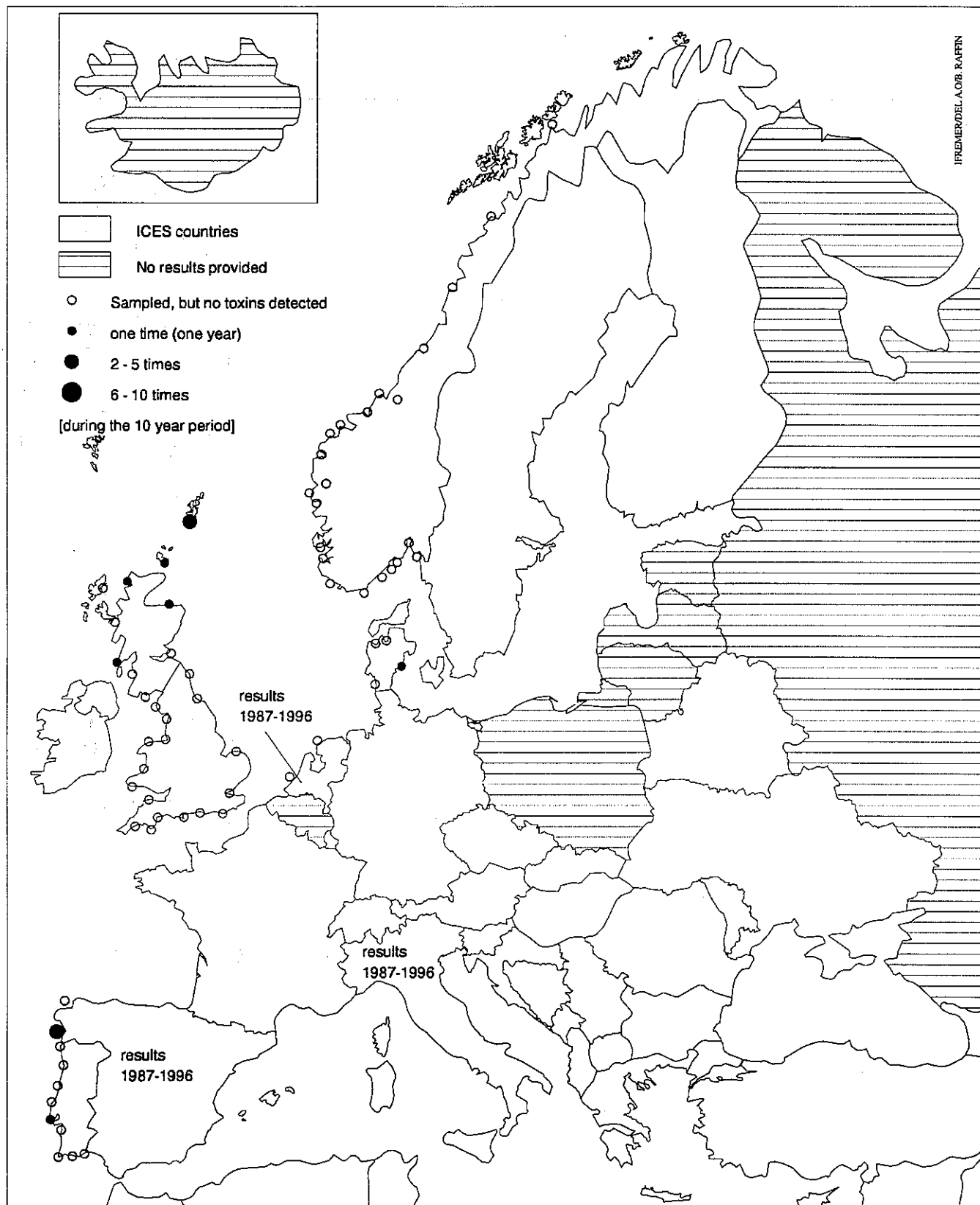
**Figure A3.2.** The occurrence and frequency of diarrhetic shellfish poisoning (DSP) events in Europe are indicated by the black circles.



**Figure A3.3.** The occurrence and frequency of paralytic shellfish poisoning (PSP) events in Europe are indicated by the black circles.

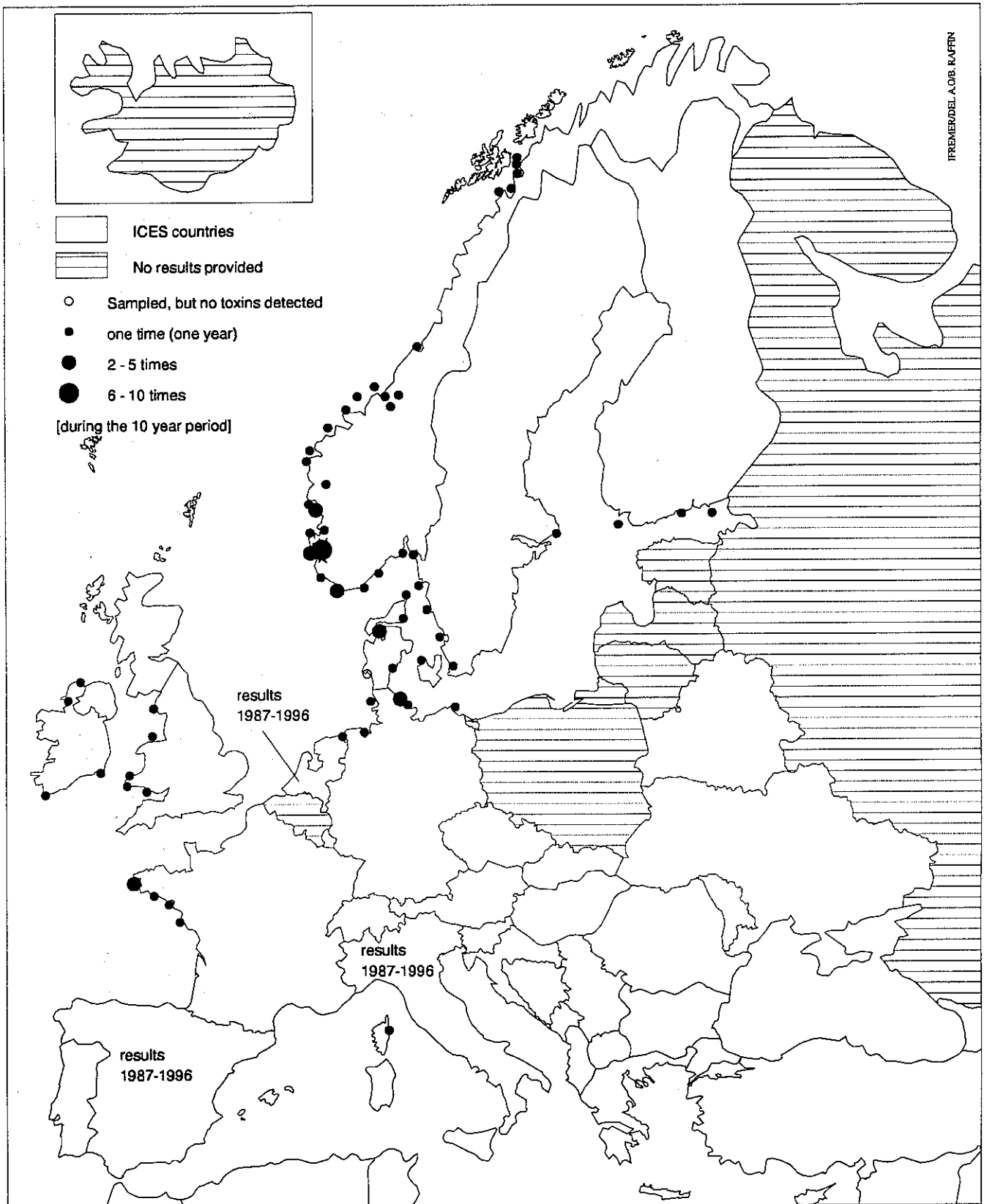


**Figure A3.4.** The occurrence and frequency of amnesic shellfish poisoning (ASP) events in Europe are indicated by the black circles.

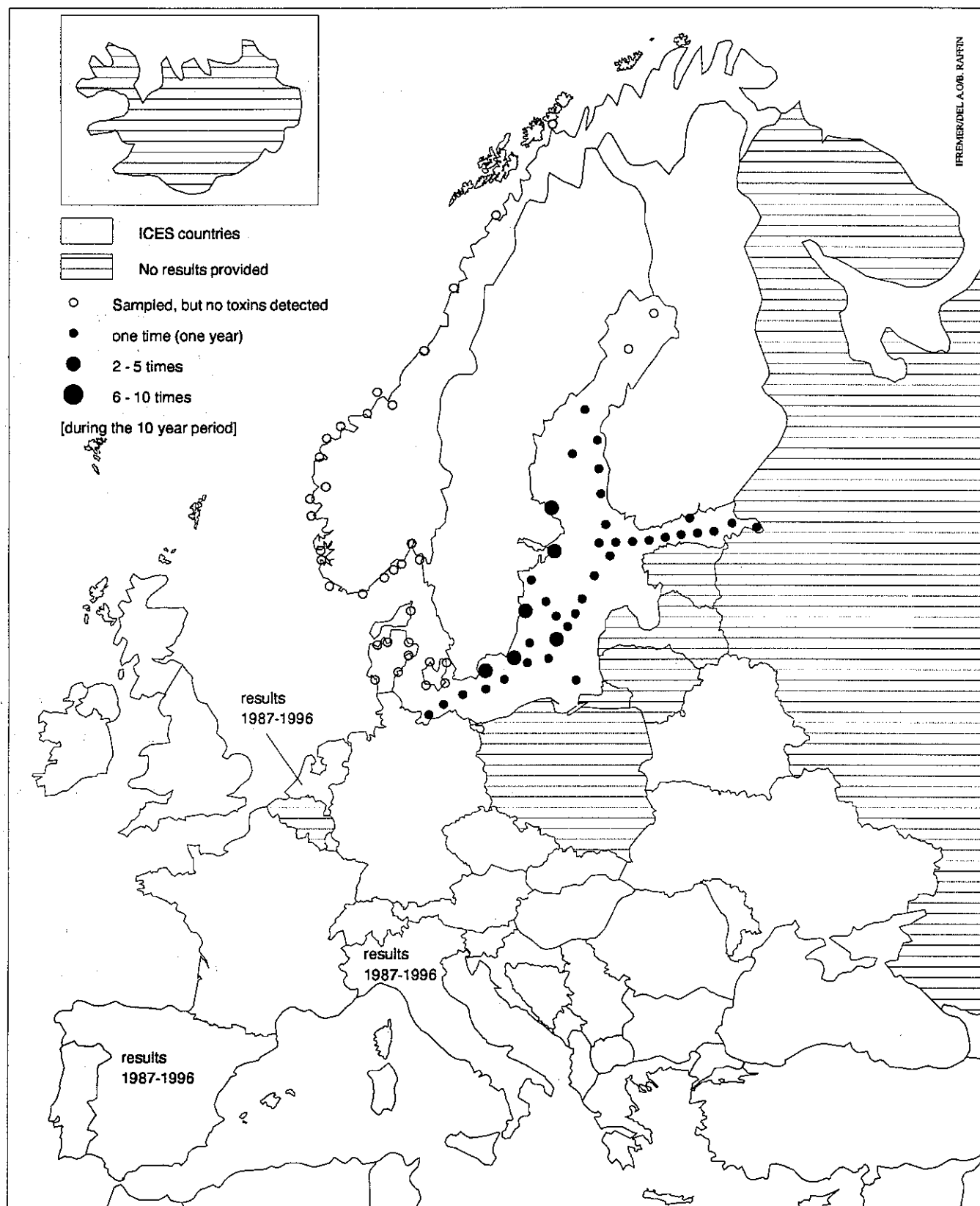




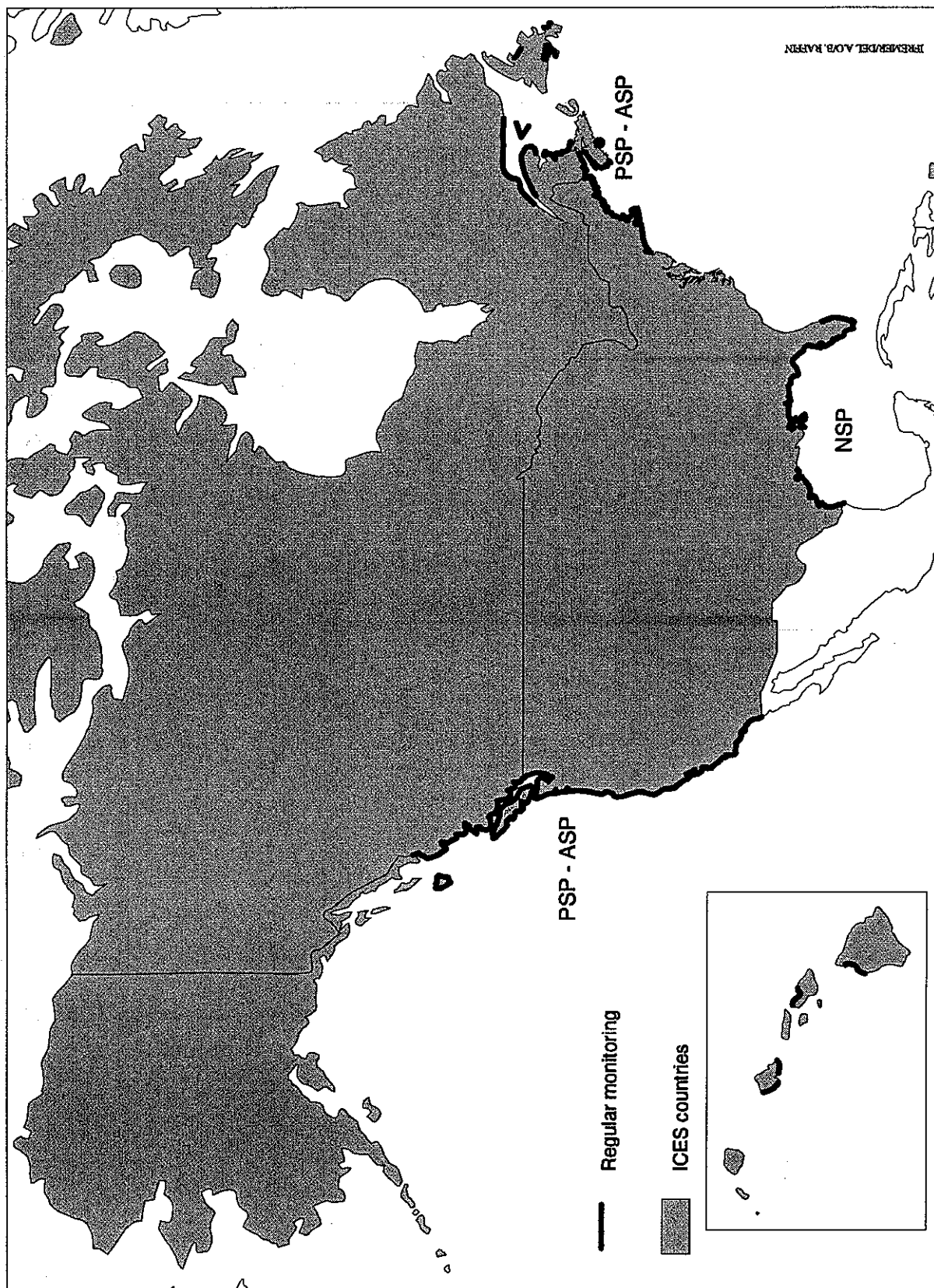
**Figure A3.5.** The occurrence and frequency of animal or plant mortalities in Europe are indicated by the black circles.



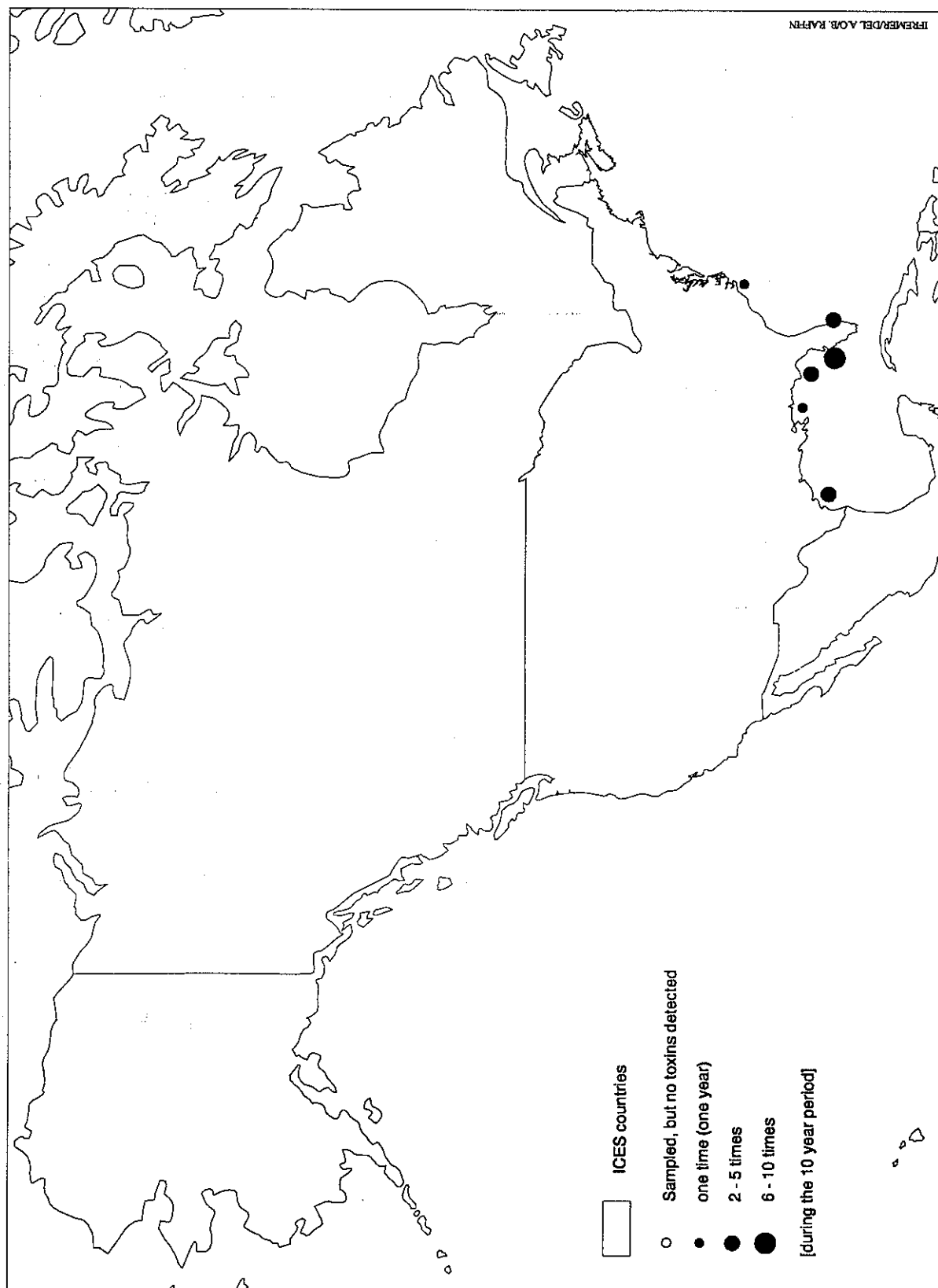
**Figure A3.6.** The occurrence and frequency of other toxic effects, such as cyanobacteria toxicity, in Europe are indicated by the black circles.



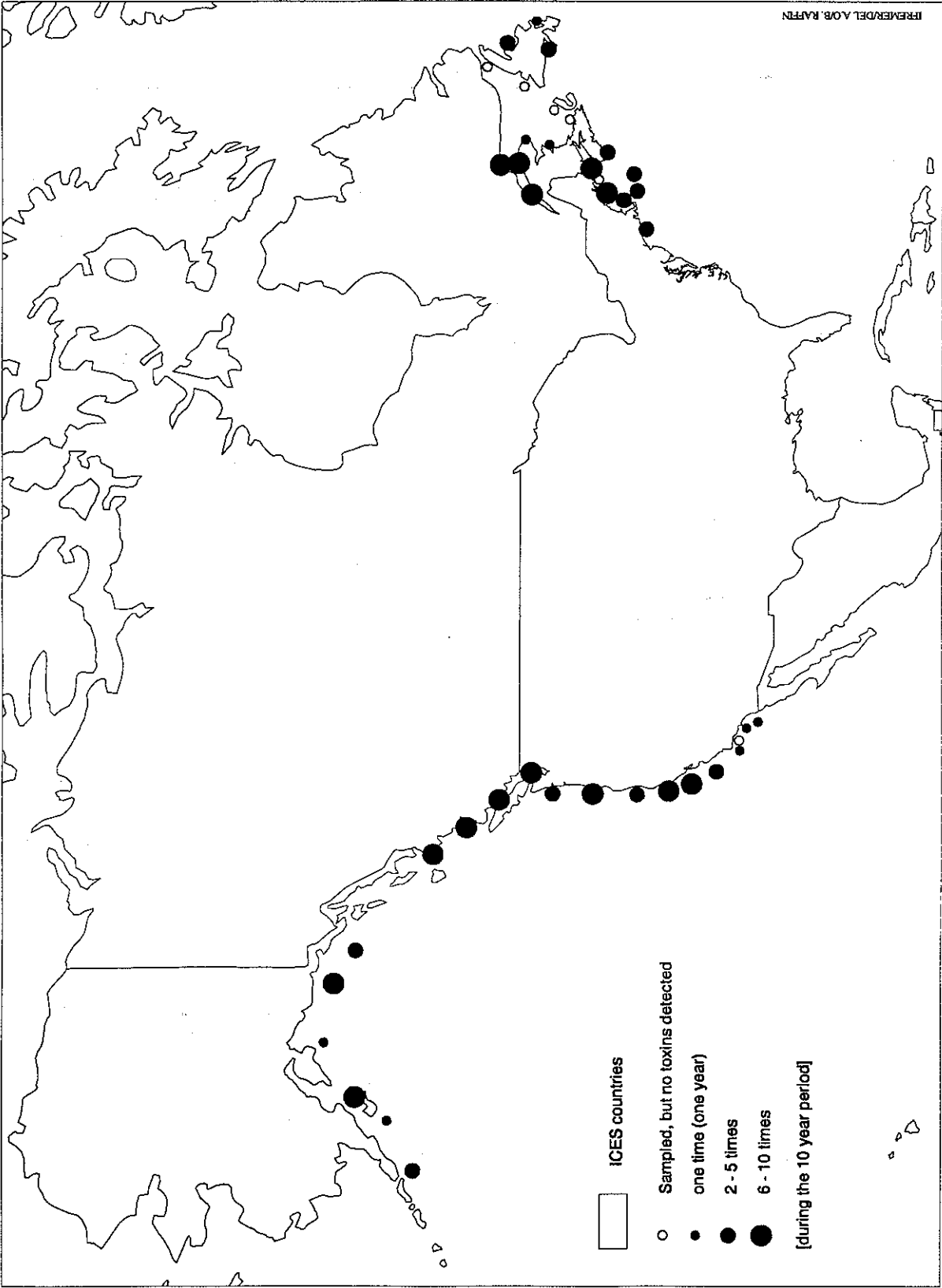
**Figure A3.7.** ICES Member Countries in North America are indicated by gray shading. Regular monitoring of bloom events is carried out in the areas marked by a heavy black line.



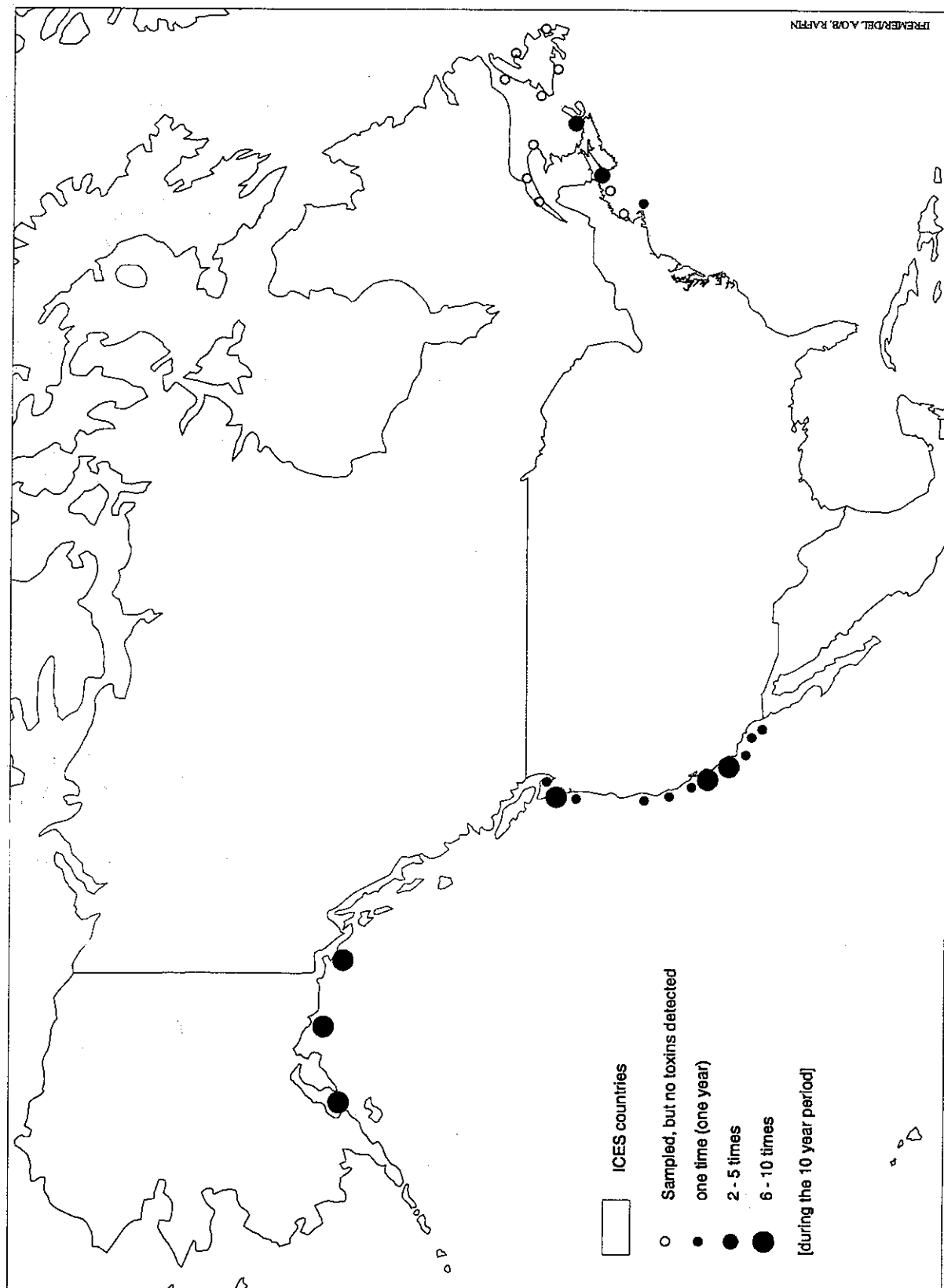
**Figure A3.8.** The occurrence and frequency of neurotoxic shellfish poisoning (NSP) events in North America are indicated by the black circles.



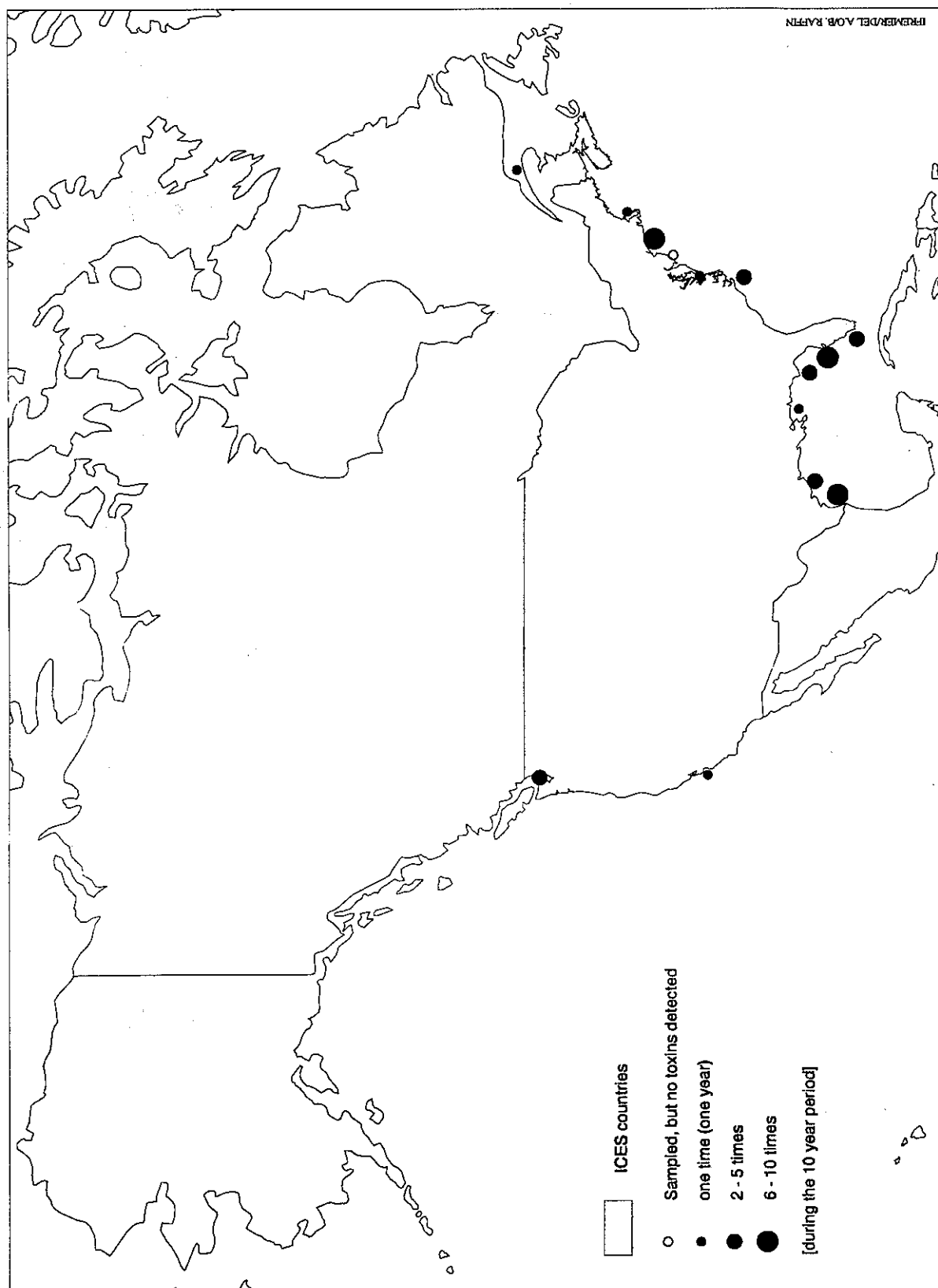
**Figure A3.9.** The occurrence and frequency of paralytic shellfish poisoning (PSP) events in North America are indicated by the black circles.



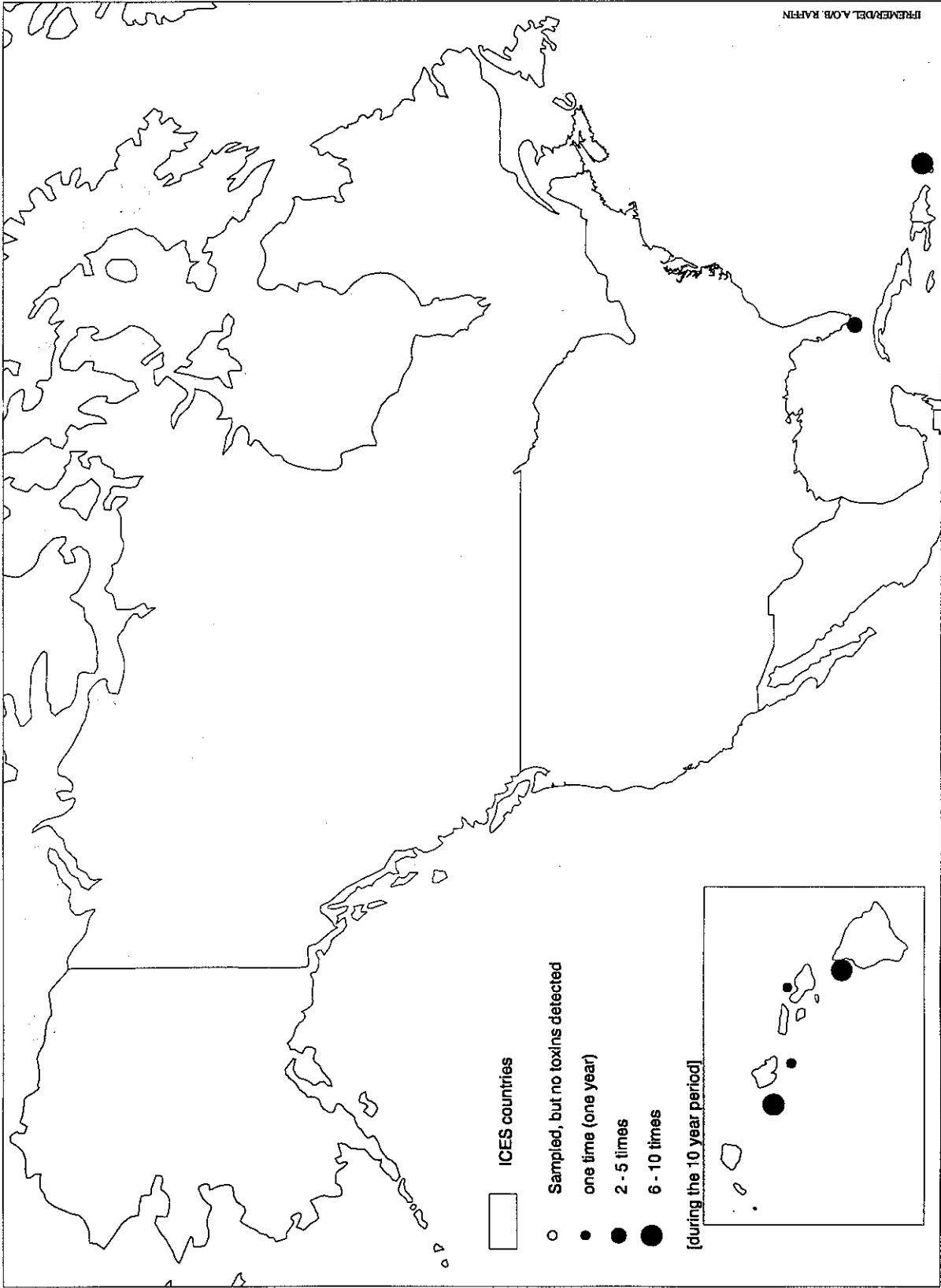
**Figure A3.10.** The occurrence and frequency of amnesic shellfish poisoning (ASP) events in North America are indicated by the black circles



**Figure A3.11.** The occurrence and frequency of animal or plant mortalities in North America are indicated by the black circles.



**Figure A3.12.** The occurrence and frequency of ciguatera fish poisoning (CFP) in North America are indicated by the black circles.





## REVIEW NOTE ON MERCURY IN THE MARINE ENVIRONMENT

## 1 INTRODUCTION

Among the toxic trace metals, mercury (Hg) is one of the most hazardous environmental pollutants, and therefore of major concern in ecotoxicology. Monomethylmercury (MMHg) accumulation in marine fish is an important human health concern as human exposure to MMHg occurs principally through the consumption of seafood and its products. Mercury exists in a large number of physical and chemical forms with a large variety of properties which determine its complex distribution, and its biological enrichment and toxicity. The most important chemical forms are elemental Hg ( $\text{Hg}^0$ ), inorganic Hg ( $\text{Hg}^{2+}$ ), monomethylmercury (MMHg,  $\text{CH}_3\text{Hg}^+$ ), and dimethylmercury (DMHg,  $\text{CH}_3\text{HgCH}_3$ ). In the biogeochemical cycle of Hg, these species may all interchange in atmospheric, aquatic, and terrestrial environments.

During the past decade, the introduction of contamination-free sampling and handling methodologies and sensitive and specific analytical equipment, as well as speciation- and reaction-oriented environmental Hg research, has considerably improved the knowledge on the Hg biogeochemical cycling. However, the majority of the environmentally related research on Hg performed in the past decade has been focused on terrestrial ecosystems, whereas the marine environment has received much less attention. This is apparent from the proceedings of the most recent conferences on mercury (Lindqvist, 1991; Watras and Huckabee, 1995; Porcella *et al.*, 1995). The database on Hg speciation in the marine environment is scarce and this results in uncertainties associated with the concentrations, stocks, and fluxes of the various Hg compounds to and from the marine environment.

## 2 GLOBAL MERCURY CYCLE: ANTHROPOGENIC INFLUENCES

Mercury enters the environment through a variety of natural and anthropogenic sources. Anthropogenic emissions to the atmosphere are estimated to be about 50 % to 75 % of the current total annual input (6000–7700 t; 30–38.5 Mmol) to the global atmosphere (Nriagu and Pacyna, 1988). As anthropogenic point sources, fuel combustion, waste incineration, industrial processes (chloralkali plants), and metal ore roasting, refining, and processing are the largest point source categories on a global scale. Together, these sources account for an annual emission of 3600–4500 t (18–22.5 Mmol) (Pacyna, 1996; Fitzgerald and Clarkson, 1991). Natural sources include ocean emission, degassing of the earth's crust, weathering, and emissions from volcanoes,

geothermal zones, and mercury-mineralized areas. Together, these emissions amount to approximately 3000 t (15 Mmol) per year, of which 1000 t (5 Mmol) are of terrestrial origin and 2000 t (10 Mmol) are of marine origin (Lindqvist *et al.*, 1991; Mason *et al.*, 1994). Net global emissions are probably increasing due to increased coal and gas combustion, metal mining and smelting, industrial emission processes, and waste incineration (Pacyna, 1996). Recycling of mercury at the earth's surface, especially from the oceans, extends the influence and active lifetime of anthropogenic Hg releases (re-emission) (Mason *et al.*, 1994). Approximately one third of the total current Hg emissions (2000 t; 10 Mmol) are thought to cycle from the oceans to the atmosphere and back again to the ocean, but a major fraction of the emissions from the oceans consists of recycled anthropogenic Hg (Figure A4.1). Natural (pre-industrial) Hg emissions from the oceans are estimated at 600 t (3 Mmol) (Fitzgerald and Mason, 1996).

Mass balance simulations of the present and pre-industrial global Hg cycle (Figure A4.1; Mason *et al.*, 1994) provide an assessment of the extent to which anthropogenic emissions may have perturbed the Hg cycle. These estimations show that the Hg concentration in the ocean mixed layer may have increased by a factor of three over the last 100 years.

## 3 MERCURY DISTRIBUTION IN THE MARINE ENVIRONMENT

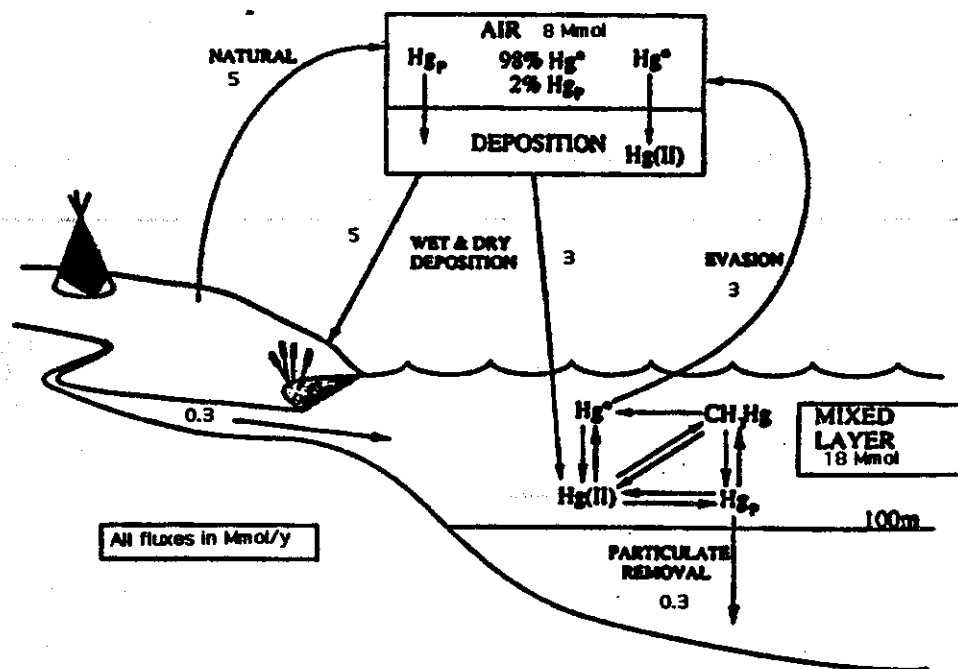
A compilation of data on Hg speciation in surface waters of rivers, estuaries, and coastal and open ocean waters obtained during the past decade using ultra-clean sampling and analytical techniques is presented in Table A4.1.

## 3.1 Oceanic Environment

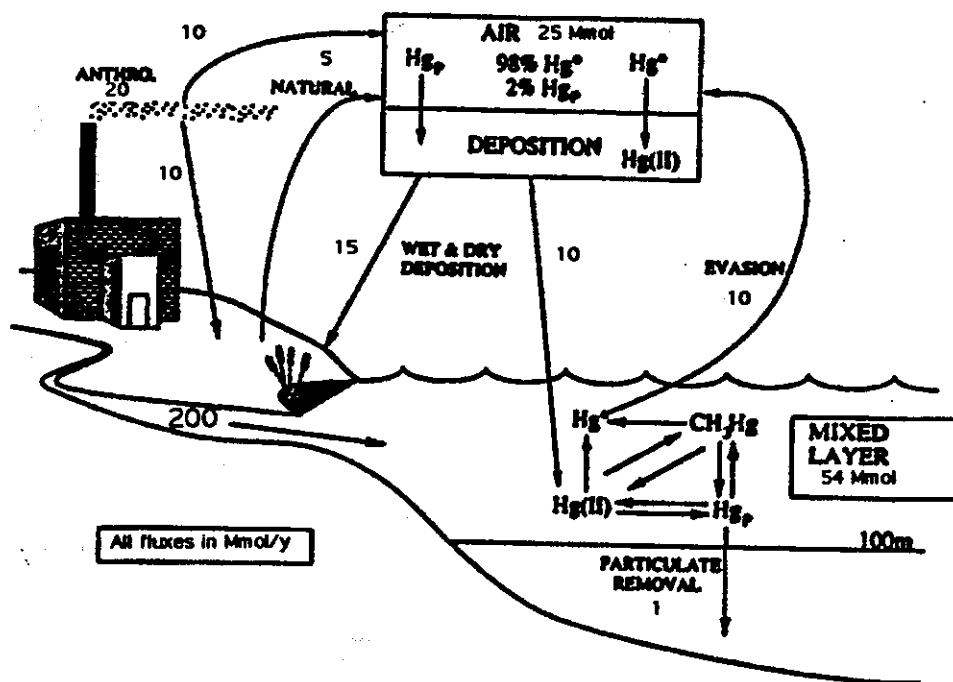
Total dissolved Hg concentrations in the open ocean range from 1 to 5 pM. Significantly higher concentrations are found (up to 10 pM) in coastal areas and in the depth region of the oxygen minimum where accumulation due to particle dissolution is enhanced. Particulate Hg concentrations are usually in the range of 0.1 to 0.5  $\mu\text{mol kg}^{-1}$ .

Methylated compounds have been detected in open ocean waters, with deep layers of the productive regions displaying the highest concentrations of MMHg and DMHg. The latter compound is often the major methylated species in oceanic waters. In general, the methylated species amount to 10 % of the total Hg.

Figure A4.1. Global mercury cycle (adapted from Mason *et al.*, 1994).



A pre-modern view of the global cycle.



The current global Hg cycle.

Table A4.1. Mercury concentrations and speciation in rivers and coastal and open ocean waters.

Location	Hg <sub>T</sub> (pmol l <sup>-1</sup> )	Hg <sub>TD</sub> (pmol l <sup>-1</sup> )	Hg <sub>R</sub> (pmol l <sup>-1</sup> )	Hg <sub>P</sub> (pmol l <sup>-1</sup> )	Hg <sub>P</sub> (nmol g <sup>-1</sup> )	MMHg <sub>T</sub> (pmol l <sup>-1</sup> )	MMHg <sub>D</sub> (pmol l <sup>-1</sup> )	MMHg <sub>P</sub> pmol g <sup>-1</sup>	MMHg <sub>P</sub> (pmol l <sup>-1</sup> )	Hg <sup>0</sup> (pmol l <sup>-1</sup> )	DMHg (pmol l <sup>-1</sup> )	Reference
<b>Rivers/estuaries</b>												
Scheldt estuary		3.5-14	1-10		1.9-8		0.065-3	10-50		0.1-0.65		Leermakers <i>et al.</i> , 1995
St. Lawrence (Canada)		0.4-4.5			0.2-2							Quemerais <i>et al.</i> , 1996
Pettaquamscutt (USA)		1-15	0.5-8	1-15			0.05-2		0.05-3			Mason <i>et al.</i> , 1993
Rhone (France)		1.4-16.5			0.4-7.8							Cossa <i>et al.</i> , 1996
Seine (France)		2.5-59.5			2.2-13.4							Cossa <i>et al.</i> , 1996
Loire (France)		2.1-10.1			0.45-2.45							Cossa <i>et al.</i> , 1996
Elbe (Germany)		3.8-16.4	0.8-4		1.5-7.05			11-46		0.27-0.6		Coquery and Cossa, 1995
Lena (Russia)		4.5-5.4			0.15-1.05							Coquery <i>et al.</i> , 1995
Ob (Russia)		2.4-3.2			0.2-0.3							Coquery <i>et al.</i> , 1995
Yenisei (Russia)		1.5-2.1			0.2-0.3							Coquery <i>et al.</i> , 1995
Framvaren fjord (Norway)	10.7-30.8					0.55-11.15						Parkman <i>et al.</i> , 1995
<b>Coastal and open sea waters</b>												
Alboran Sea											< dl-0.29	Cossa <i>et al.</i> , 1994a
Celtic Sea	1.8-13.7											Cossa <i>et al.</i> , 1996
North Sea	1.6-21.4	0.9-4.8	0.4-1.8	0.2-16.6	0.58-2.42		< dl	< dl-60	< dl-0.19	< dl-0.45	< dl	Coquery and Cossa, 1995
▪ Belgian coast	0.65-80.5	0.65-7.1			0.23-3.21		< dl-0.94	< dl-50		0.1-0.8	< dl	Leermakers, 1998
▪ Central North Sea	1.00									0.06	< dl	Baeyens and Leermakers, 1998
▪ Dogger Bank		0.95-2.1	0.8-1.9		0.2-1.05							Fileman <i>et al.</i> , 1991
▪ Northern North Sea		1-2.5										OSPAR, 1996
English Channel	4-20.5	1.5-4.2			2.55-8.85							Cossa and Fileman, 1991
▪ English Channel	0.75-4.35						0.075-0.33					Leermakers, 1998
▪ Dover Strait		0.6-6.7		0.3-26.8								Cossa <i>et al.</i> , 1994b
<b>Open Ocean</b>												
North Atlantic	1.55-4.25		0.75-1.05							0.07-0.9	< 0.01-0.2	Mason <i>et al.</i> , 1995c
			0.7-1.05									Dalziel, 1995
Equatorial Pacific			0.8-2							0.05-0.4	< 0.01-0.3	Mason and Fitzgerald, 1996

Hg<sub>T</sub> = total Hg    Hg<sub>TD</sub> = total dissolved Hg    Hg<sub>R</sub> = reactive Hg    Hg<sub>P</sub> = particulate Hg    MMHg<sub>T</sub> = total MMHg    MMHg<sub>D</sub> = dissolved MMHg

$\text{Hg}^0$  is found in the mixed layer and in the deeper waters of the ocean, with concentrations ranging from 0.01–0.5 pM. (Mason *et al.*, 1995c; Baeyens and Leermakers, 1998). In highly productive environments such as upwelling areas, concentrations as high as 1 pM are found in surface waters.

### 3.2 Estuarine and River Systems

In unimpacted rivers and estuaries, total dissolved Hg concentrations range from 1 to 6 pM, whereas particulate Hg concentrations range from 0.2 to 0.7  $\mu\text{mol kg}^{-1}$ . MMHg represents 1 % to 5 % of the total Hg concentration.

In polluted estuaries, particulate Hg concentrations up to 10  $\mu\text{mol kg}^{-1}$  and dissolved Hg concentrations up to 30 pM have been observed (Leermakers *et al.*, 1995; Ebinghaus and Wilken, 1996). The highest particulate Hg concentrations are found in industrialized and urbanized small rivers such as the Seine, the Scheldt, and the Elbe, whereas large industrialized rivers (the Rhone, the St. Lawrence) do not show the same elevated concentrations (Cossa *et al.*, 1996). Up to 50 % of the dissolved Hg concentrations could be attributed to MMHg in summer and autumn in the Scheldt, while  $\text{Hg}^0$  constitutes between 1 % and 10 % (Leermakers *et al.*, 1995). A large fraction of dissolved estuarine and riverine Hg is bound to organic complexes and/or colloidal matter (Mantoura *et al.*, 1978; Guentzel *et al.*, 1996).

MMHg concentrations in anoxic bottom waters can exceed surface water concentrations by a factor of 100. DMHg has not been detected in freshwater systems, but has been found in trace amounts (0.1 pM) in the high turbidity brackish water zone of the eutrophic Seine estuary (Cossa *et al.*, 1996). DMHg is more readily decomposed in fresh water and can easily escape from surface waters via gas evasion.

## 4 BIOGEOCHEMICAL BEHAVIOUR OF MERCURY

The present knowledge on Hg cycling in the marine environment has been summarized in a number of overview articles (Cossa *et al.*, 1996; Fitzgerald and Clarkson, 1991; Fitzgerald and Mason, 1996; Mason and Fitzgerald, 1996).

The main transformation pathways between the various Hg species in the different environmental compartments have been identified (Figure A4.2), although the reaction mechanisms and/or biological species involved in the interconversion of Hg species in the ocean remain uncertain. The *in situ* (bacterial) conversion of inorganic Hg species to MMHg is an important feature of the Hg cycle in aquatic systems as it is the first step in the bioaccumulation process. Methylation occurs both in the water column and in the sediments (its origin in the atmosphere is still unknown), and has been shown to be

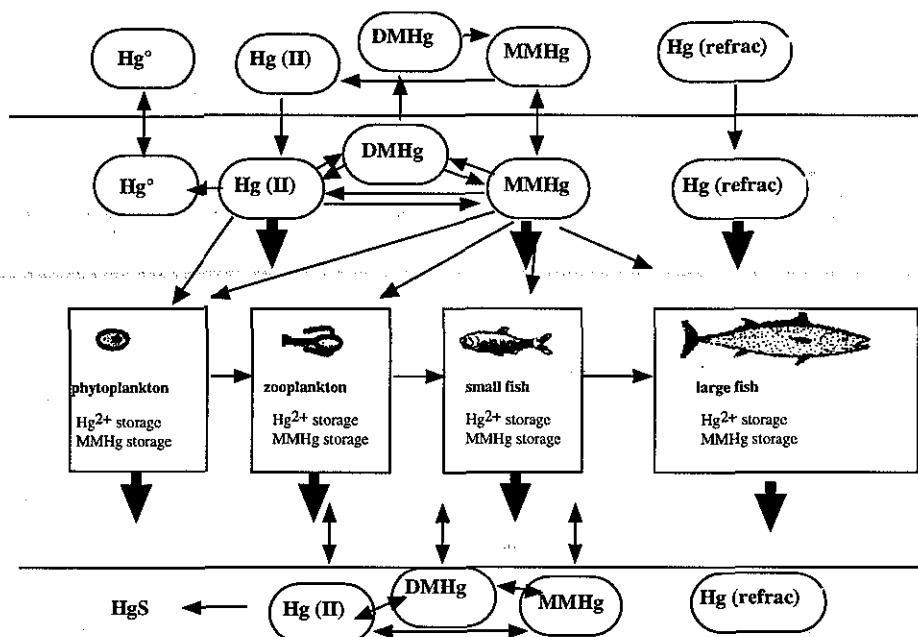
predominantly due to sulphate-reducing bacteria in freshwater and estuarine systems. In the ocean, DMHg is the main methylated compound, in contrast to freshwater systems where DMHg is not found. MMHg in the oceans is thought to derive from the decomposition of DMHg, suggesting that probably other species are responsible for the formation of MMHg in the oceans (Mason *et al.*, 1995c). Vertical profiles of Hg species in the Atlantic and Pacific Oceans (Mason and Fitzgerald, 1991; Mason *et al.*, 1995c) as well as in the Mediterranean Sea (Cossa *et al.*, 1994a, 1997) show similar patterns: low concentrations of  $\text{Hg}^0$ , reactive Hg ( $\text{Hg}_R$ ) and methylated species in the mixed layer, and increased concentrations of these species in sub-thermocline waters. Processes which govern the speciation of mercury in the oceans have been proposed (Mason and Fitzgerald, 1990, 1991, 1993; Mason *et al.*, 1995c; Cossa *et al.*, 1994a, 1997): in the surface layer,  $\text{Hg(II)}$  is reduced to  $\text{Hg}^0$  and recycled in the atmosphere or incorporated into particulate matter and subsequently released deeper in the water column. Low concentrations of  $\text{Hg}^0$  and DMHg in the mixed layer are the result of gas evasion to the atmosphere, and particulate scavenging removes MMHg from surface waters. Methylated species (DMHg and MMHg) show a maximum concentration below the thermocline. DMHg occurs mainly in the sub-thermocline regions where oxygen consumption is active, with the  $\text{Hg(II)}$  pool as a substrate for both methylation and reduction. Particulate dissolution in the deeper waters releases bound MMHg and  $\text{Hg(II)}$  into solution. The currently available data set suggests that there is a relationship between surface water productivity and deep water DMHg formation (Mason and Fitzgerald, 1996). Higher concentrations are found in the more productive eastern Equatorial Pacific than in the North Atlantic and western Equatorial Pacific. Formation of DMHg in deep water relies on the supply of  $\text{Hg(II)}$  to this zone via particle settling and remineralization, and this process is linked to surface water productivity. Deep water temperature may also influence DMHg formation. In the western Mediterranean, specific methylation rates were estimated to be six times higher than those in the North Atlantic (Cossa *et al.*, 1997).

The *in situ* production and air/water exchange of  $\text{Hg}^0$  in surface waters exert a major influence on the fate of Hg in the environment. Volatilization of  $\text{Hg}^0$  competes with MMHg formation for the available  $\text{Hg(II)}$ , the substrate for both reduction and methylation. There is a relationship between primary productivity and  $\text{Hg}^0$  in the surface mixed layer (Mason *et al.*, 1995c). The mechanisms by which Hg is reduced are still under study, but appear to be mainly biologically mediated and involve picoplankton (eukaryotic phytoplankton and bacteria) (Mason *et al.*, 1995a).

## 5 BIOACCUMULATION PATHWAYS

The factors controlling the accumulation of mercury in fish are not yet fully understood. The increasing concentrations of Hg (principally MMHg) in higher trophic levels of the food chain resemble those of

Figure A4.2. Mercury transformation in the aquatic environment (adapted from Fitzgerald and Mason, 1996).



hydrophobic trace pollutants. However, the lipid solubility of MMHg is an inadequate explanation because inorganic Hg complexes, which are not bioaccumulated, are as lipid soluble as their MMHg analogues and, unlike other hydrophilic compounds, MMHg in fish resides in protein rather than in fat tissue (Bloom, 1992). Mason *et al.* (1995b) have shown that the passive uptake of lipophilic neutral Hg compounds (such as  $\text{HgCl}_2$  and  $\text{CH}_3\text{HgCl}$ ) results in higher concentrations of both inorganic and MMHg in phytoplankton. The differences in partitioning within phytoplankton cells between inorganic Hg, which is principally membrane bound, and MMHg, which accumulates in the cytoplasm, lead to larger assimilation of MMHg during zooplankton grazing. Thus, the transfer efficiency of MMHg between plankton and planktivorous fish is approximately ten times greater than that for  $\text{Hg}^{2+}$ . Most of this discrimination between inorganic Hg and MMHg thus occurs during trophic transfer, while the major enrichment factor is between water and phytoplankton (ca.  $10^{5.5}$  between water and phytoplankton and  $10^{6.5}$  between water and fish). As a result, MMHg in fish is ultimately determined by water chemistry that controls MMHg speciation and uptake at the base of the food chain. In sea water,  $\text{HgCl}_4^{2-}$  is the principal inorganic Hg form and the neutral  $\text{HgCl}_2$  only accounts for 3%. MMHg, if present, is nearly 100%  $\text{CH}_3\text{HgCl}$ . Thus, despite the much lower concentration of MMHg compared to that of inorganic Hg in sea water (0.05 pM or less compared to 1 pM inorganic Hg), its overall bioaccumulation by planktivorous fish is expected to be sixteen times greater. In sea water, dimethylmercury (DMHg) is often the major methylated species. As a neutral complex, passive diffusion of DMHg followed by conversion to MMHg is likely to occur. In fresh water, the relative bioaccumulation depends on the water

chemistry (pH, DOC, oxygen, etc.) which governs the speciation and concentrations of  $\text{Hg}^{2+}$  and  $\text{CH}_3\text{Hg}^+$ . Although a larger fraction of Hg is found as  $\text{HgCl}_2$ , MMHg concentrations are usually considerably higher and can account for a large fraction of the total dissolved Hg (50% or more). In addition to accumulation through the food chain, passive uptake of DMHg and  $\text{CH}_3\text{HgCl}$  (as neutral compounds) through the gills may also be an important uptake route. The relative importance of direct versus food chain accumulation is not yet known.

Not all fish species follow the same bioaccumulation pattern. Three types of bioaccumulation patterns have been distinguished (Holsbeek *et al.*, 1997): Type I, which covers the majority of species, describes the normal pattern, with increasing levels of MMHg with age, combined with a low and constant inorganic level. This accumulation pattern leads to a relative increase of the organic mercury fraction with age, eventually reaching 90% to 100% of organic mercury in full-grown specimens. Type II is found in planktivorous fish and shows increasing levels of inorganic mercury combined with low and constant MMHg levels (leading to a relatively decreasing MMHg fraction with age). A third intermediate accumulation pattern, with increasing concentrations of both the organic and the inorganic Hg fractions with age, was found in one bottom-dwelling species only.

Juvenile fish and low food chain marine organisms such as mussels, shrimps, urchins, and anemones tend to have low methylmercury to total mercury ratios that are influenced by the degree of environmental contamination, with relatively lower methylmercury to total mercury

ratios in Hg-contaminated areas (Mikac *et al.*, 1985; Lasorsa and Allen-Gil, 1995).

At the higher trophic levels and more specifically in cetaceans, Hg is not only accumulated as a function of age, but this phenomenon is also linked to a change in the speciation of mercury and to a relocalization between different tissues resulting in a demethylation. Mercury, present as MMHg in the food of cetaceans, is readily assimilated under its organic form, but is slowly relocalized and demethylated leading to the formation of a Se-Hg compound (thiemanite). Particularly in liver tissue, thiemanite accumulates over time to extremely high but nonetheless non-toxic levels (Capelli *et al.*, 1989; Hansen *et al.*, 1990; Joiris *et al.*, 1991; Paludan-Muller *et al.*, 1993).

## 6 MERCURY IN FISH OF THE NORTH SEA AND THE NORTH ATLANTIC: CONCENTRATIONS AND TRENDS

Studies of contaminants in marine biota have mainly been conducted in the framework of joint monitoring programmes, of which the results for the time period 1978–1988 are reported in the North Sea Quality Status Report 1993 (NSTF, 1993). The main species analysed were cod (*Gadus morhua*), whiting (*Merlangius merlangus*), dab (*Limanda limanda*), flounder (*Platichthys flesus*), plaice (*Pleuronectes platessa*), and blue mussel (*Mytilus edulis*).

In these surveys, mercury concentrations in the muscle tissue of cod, whiting, dab, plaice, and sole ranged from 0.03 to 0.35 mg kg<sup>-1</sup> wet weight (ww). Relatively higher concentrations within this range were found in the coastal zones (German Bight, Southern Bight, Norwegian coast). Most concentrations fell within the 'lower' and 'medium' Joint Monitoring Programme (JMP) categories (< 0.1 mg kg<sup>-1</sup> ww and 0.1–0.3 mg kg<sup>-1</sup> ww respectively). In mussels (*Mytilus edulis*), Hg concentrations ranged from 0.002 to 0.17 mg kg<sup>-1</sup>, with concentrations at the higher end in the Southern Bight, the Wadden Sea, the Ems estuary, the Western Scheldt, and at a number of locations along the English coast as well as the Danish and Norwegian coasts. This is based on present-day background concentrations in the region of the OSPAR Convention Area of Hg in roundfish of 0.01–0.05 mg kg<sup>-1</sup> ww, in flatfish of 0.03–0.07 mg kg<sup>-1</sup> ww, and in mussels of 0.005–0.01 mg kg<sup>-1</sup> ww, found in areas remote from known point sources (OSPAR, 1996). Recently reported concentrations of Hg in fish from the North Sea and North Atlantic confirm the previously reported concentration ranges, of which the lower limits confirm the estimated background concentrations (Table A4.2).

Although the inputs of Hg to the North Sea have been significantly reduced in the past decades, temporal trends indicating a downward trend have only been reported in a limited number of areas. In Belgian coastal waters, Hg concentrations have decreased in flounder, sole, plaice,

and mussels by 50 % to 75 % between 1971 and 1993, but not in cod and shrimp (Vyncke *et al.*, 1996), and a downward trend has been observed in Hg concentrations in cod from the east coast of England between 1982 and 1988 (NSTF, 1993) and along the Dutch coast and the Wadden Sea between 1983 and 1991 (Oslo and Paris Commissions, 1994, cited in Pedersen, 1996). In all other subregions, no obvious temporal trend was found (NSTF, 1993).

The lack of clear time trends in biota is largely due to temporal fluctuations in the Hg concentrations, as have been shown for the Danish waters (Pedersen, 1996). These temporal variations may be due to biological factors, physicochemical conditions, and accumulation pathways. Important biological factors are:

- 1) the effect of age, length, fat content, and sex on Hg accumulation (Pedersen, 1996; Riget *et al.*, 1996);
- 2) seasonal differences in mercury accumulation resulting from seasonal differences in metabolic activity/growth rate and bioavailability of the Hg species (Cossa, 1989);
- 3) migratory behaviour of the fish species: for example, fish migrating to mesopelagic environments may accumulate more Hg due to enhanced concentrations of methylated compounds at these depths (Monteiro *et al.*, 1996).

Physicochemical conditions such as temperature, oxygen, organic matter concentrations, and phytoplankton activity have an important influence on the methylation and reduction of Hg in the water column, and this has a direct influence on the bioavailability of Hg. In addition, a better understanding of the accumulation pathways (direct versus food-chain uptake) and the methylation processes in the marine environment is required.

## 7 SUMMARY OF UNCERTAINTIES AND GAPS IN INFORMATION

### 7.1 Inputs to the Marine Environment

#### Rivers

The available data on the concentrations of Hg in rivers may not be truly representative of the world's rivers. For example, there is no information on Hg in tropical rivers, including such large rivers as the Amazon, and impacted rivers of South America, Europe, Asia, and the former Soviet Union.

Future work should endeavour to understand the processes occurring in estuaries and coastal regions so that the flux estimates to the ocean can be further refined. Modelling the export of Hg from rivers needs to take these processes into account. Additionally, the fate of MMHg during estuarine mixing needs to be assessed.

**Table A4.2.** Concentrations of mercury in finfish and shellfish from the North Sea and North Atlantic.**Blue mussel (*Mytilus edulis*)**

Location	Period	Hg <sub>T</sub> (µg g <sup>-1</sup> ww)	Reference
Bergen Harbour, Norway	1993	0.01–0.06	Andersen <i>et al.</i> , 1996
Cork Harbour, Ireland	1990	0.28–1.5	Berrow, 1991
Ems Estuary, The Netherlands	1985–1990	0.02–0.06 (ave. 0.035)	Stronkhorst, 1992
Western Scheldt, The Netherlands	1985–1990	0.02–0.06 (ave. 0.038)	Stronkhorst, 1992
Belgian coast	1993	0.026*	Vyncke <i>et al.</i> , 1996
Irish coast	1994	0.02–0.09	Nixon <i>et al.</i> , 1995
Iceland	1978	0.010–0.026	Olafsson, 1986
Iceland	1995	0.002–0.009	G. Audunsson, pers. comm.
Baltic Sea	1989–1993	< 0.001–0.045	HELCOM, 1996
Greenland	1980–1982	0.057–0.097	Riget <i>et al.</i> , 1996
Present-day background concentration		0.005–0.010	OSPAR, 1996

\* = average concentration

**Plaice (*Pleuronectes platessa*)**

Location	Period	Hg <sub>T</sub> (µg g <sup>-1</sup> ww)	Reference
Liverpool Bay, UK	1994	0.13*	SIME, 1996
Morecambe Bay, UK	1994	0.09*	SIME, 1996
Southern Bight (UK waters)	1994	0.05*	SIME, 1996
Irish coast	1994	0.05–0.09	Nixon <i>et al.</i> , 1995
NE English coast (Tyne river)	1992	0.006–0.211	Dixon and Jones, 1994
Firth of Clyde, UK	1992	0.011–0.019	Mathieson and McLusky, 1995
Gulf of St. Lawrence, Canada	1992–1995	0.049 ± 0.020	Gobeil <i>et al.</i> , 1997
North Atlantic, French coast	1988	0.028–0.15	Cossa <i>et al.</i> , 1990
English Channel	1988	0.026–0.15	Cossa <i>et al.</i> , 1990
Present-day background concentration		0.03–0.07	OSPAR, 1996

\* = average concentration

**Cod (*Gadus morhua*)**

Location	Period	Hg <sub>T</sub> (µg g <sup>-1</sup> ww)	Reference
Liverpool Bay, UK	1994	0.10*	SIME, 1996
Southern Bight (UK waters)	1994	0.07*	SIME, 1996
Belgian coast	1993	0.09*	Vyncke <i>et al.</i> , 1996
Irish coast	1994	0.01–0.07	Nixon <i>et al.</i> , 1995
Iceland	1996	0.01–0.04	G. Audunsson, pers. comm.
Gulf of St. Lawrence, Canada	1992–1995	0.060 ± 0.023	Gobeil <i>et al.</i> , 1997
Northern North Atlantic	1994	0.01–0.21	Stange <i>et al.</i> , 1996
Baltic Sea	1989–1996	0.002–0.365	HELCOM, 1996
Present-day background concentration		0.01–0.05	OSPAR, 1996

\* = average concentration

Table A4.2. Continued.

**Whiting (*Merlangius merlangus*)**

Location	Period	Hg <sub>T</sub> (µg g <sup>-1</sup> ww)	Reference
Liverpool Bay, UK	1994	0.27 (n = 25)	SIME, 1996
Morecambe Bay, UK	1994	0.27 (n = 25)	SIME, 1996
NE English coast, Tyne River	1992	0.052–0.432	Dixon and Jones, 1994
Irish coast	1994	0.04–0.19	Nixon <i>et al.</i> , 1995
Present-day background concentration		0.01–0.05	OSPAR, 1996

**Dab (*Limanda limanda*)**

Location	Period	Hg <sub>T</sub> (µg g <sup>-1</sup> )	Reference
Liverpool Bay, UK	1994	0.20*	SIME, 1996
Morecambe Bay, UK	1994	0.15*	SIME, 1996
NE English coast, Tyne River	1992	0.042–0.255	Dixon and Jones, 1994
Firth of Clyde, UK	1992	0.017–0.046	Mathieson and McLusky, 1995
Iceland	1996	0.019–0.053	G. Audunsson, pers. comm.
Northern North Atlantic	1994	0.01–0.02	Stange <i>et al.</i> , 1996
Present-day background concentration		0.03–0.07	OSPAR, 1996

\* = average concentration

**Flounder (*Platichthys flesus*)**

Location	Period	Hg <sub>T</sub> (µg g <sup>-1</sup> ww)	Reference
Liverpool Bay, UK	1994	0.17*	SIME, 1996
Morecambe Bay, UK	1994	0.23*	SIME, 1996
Ems Estuary, The Netherlands	1985–1990	0.107*	Stronkhorst, 1992
Western Scheldt, The Netherlands	1985–1990	0.106*	Stronkhorst, 1992
Belgian coast	1993	0.15*	Vyncke <i>et al.</i> , 1996
Denmark, the Sound	1995	0.13*	B. Pedersen, pers. comm.
North Atlantic, French coast	1986	0.024–0.44	Cossa <i>et al.</i> , 1990
English Channel	1986	0.3–0.27	Cossa <i>et al.</i> , 1990
Present-day background concentration		0.03–0.07	OSPAR, 1996

\* = average concentration

**Sole (*Solea solea*)**

Location	Period	Hg <sub>T</sub> (µg g <sup>-1</sup> ww)	Reference
Liverpool Bay, UK	1994	0.14 (n = 40)	SIME, 1996
Morecambe Bay, UK	1994	0.17 (n = 50)	SIME, 1996
Southern Bight	1991	0.08*	De Clerck <i>et al.</i> , 1995
Irish Coast	1994	0.02–0.16	Nixon <i>et al.</i> , 1995
North Atlantic, French coast	1988	0.03–0.27	Cossa <i>et al.</i> , 1990
English Channel	1988	0.018–0.24	Cossa <i>et al.</i> , 1990
Present-day background concentration		0.03–0.07	OSPAR, 1996

\* = average concentration



## **Atmospheric deposition**

The database for Hg concentrations in oceanic precipitation is limited both temporally and spatially. The flux of MMHg from the atmosphere to the ocean is unknown. Coastal rain could be an important source of MMHg and this should be a goal of future atmospheric research. Again, the database for the Southern Hemisphere, for the Russian coastline and other parts of Asia, especially the Arctic coastlines, is non-existent. Data on rain in tropical regions are also needed to assess the impacts of current activities such as biomass burning and gold extraction on the transport of Hg to the tropical oceans.

## **Other potential sources of mercury to the ocean**

Evidence of high concentrations of Hg in terrestrial oil and gas deposits suggests that this could be a potentially important source in regions of the coastal ocean and shelf where these deposits have formed seeps into the ocean. Continental deposits have been shown to have elevated concentrations of Hg. The flux from such deposits to the ocean could be important on a regional and/or global scale. One area where this phenomenon could be studied is the North Sea, as this is a region where Hg geological belts and oil deposits coincide in the sediment.

### **7.2 Mercury Methylation and Bioaccumulation in the Ocean**

There is a need to try and assess the importance of direct uptake relative to food chain accumulation for open ocean fish, especially for long-living species, and also for marine mammals. An investigation of the mechanisms by which Hg is methylated in the ocean should be undertaken. It is not clear as to which organisms are producing DMHg and the presumption of biotic formation needs to be verified. To help assess the extent of Hg methylation in the ocean, we also need to quantify the fluxes of MMHg to and from the ocean, i.e., to produce an MMHg ocean budget.

## **8 ANALYTICAL IMPROVEMENTS**

Methods for the measurement of total and methylmercury compounds in biological and some environmental samples are relatively well developed. A number of comprehensive reviews evaluate the available analytical techniques for Hg analysis and speciation as well as highlight major analytical problems and critical steps in the analytical procedures (Horvat, 1996; Puk and Weber, 1994; Baeyens, 1992; Wilken, 1992; Lindqvist *et al.*, 1991). However, systematic errors have been made, for example, the use of the widespread distillation technique for the analysis of methylmercury in sediments produces artefacts (Bloom *et al.*, 1997). Therefore, there is a need for the development of independent, accurate analytical techniques in order not to propagate systematic errors. In addition, there is a need for the development of analytical

techniques for Hg speciation analysis that are applicable to ocean studies so that all speciation measurements can be performed on board research vessels, thereby alleviating the problems of potential sample contamination and/or speciation change during storage.

One way to control the accuracy of analytical data is by analysing certified reference materials (CRMs). CRMs are available for total mercury in biological, sediment, soil, and water samples. However, only a few biological samples and two sediment samples are certified for methylmercury compounds. CRMs covering various matrices and concentration ranges for total and methylmercury compounds are required. The use of CRMs can, however, only cover a limited number of environmental samples. Proficiency testing schemes such as QUASIMEME should include Hg at ambient levels in sea water as well as methylmercury. As the concentration levels of mercury in air and water are extremely low, the reliability of the results depends on the overall procedure, including sampling, storage and laboratory handling. To check the accuracy of the results, participation in field intercomparison exercises is required, as well as comparison of the results obtained by various methodologies.

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## STUDY OF THE BEHAVIOUR OF TRACE ELEMENTS IN THE LIVERS OF COD FROM ICELANDIC WATERS

### 1 INTRODUCTION

An important aspect of monitoring programmes concerns comparisons of concentrations of analytes in various compartments. In order to make comparisons, however, the laws governing the natural variation of the analytes in the compartment in question must be known. Cod livers are widely used for monitoring purposes, both for organic and inorganic analytes, where sampling is based on the length of the fish; usually 30–45 cm cod are selected for the monitoring of spatial trends and the sampling of 25 individuals per sample is performed prior to spawning. However, it is well known that levels of trace elements may vary considerably in cod livers, where factors of ten or more between the lowest and highest concentrations are not uncommon within a sample. This variability of trace element results has often been related to variation in the fat content of the livers (Grimås *et al.*, 1985; Nicholson *et al.*, 1991). The fat content of cod livers may vary dramatically with season. In the North Atlantic, an increase in liver lipids is observed from autumn until the end of the year. When gonads start to develop at the beginning of the year, the liver lipids decrease and reach a minimum shortly after spawning in the springtime. This cyclical variation may be between 10 % and 70 % fat in the liver. However, great variability is found between individual fish of similar sizes in the same area and at the same time of year. Differences are also observed in this cyclical behaviour from one year to another at the same fishing ground (affected by, for example, food availability and temperature). A higher fat content than 70–75 % is rarely found and a lower fat content than 10 % is uncommon. However, as low as 2 % fat has been found in the livers of 40–70 cm cod during extreme starvation in the laboratory (livers weighing 4–10 g) (Love, 1958). This cyclical change occurs both in mature and immature cod, although the variability may be greater in mature cod. Finally, the fatty acid composition of the fat varies with the amount of fat. In a study by Witt (1963) on cod liver oils from six fishing grounds (White Sea, Iceland, Bear Island, the Norwegian coast, Greenland, and the Faroe Islands), a distinct increase was found for the iodine value (a measure of unsaturation) of the oil from August 1961 to January 1962. In a study by Jangaard *et al.* (1967a, 1967b) on cod from Terence Bay, Nova Scotia, an increase was found in 20:1 and 22:1 (especially in female fish) with increased fat content of the livers, i.e., decreased unsaturation with fat content in contrast to the results for cod from the Northeast Atlantic. Thus, cod from different fishing grounds or different stocks of cod may show quite different behaviour.

The aim of the work presented here was to look into the data on trace elements in cod livers from Icelandic monitoring studies (1994–1996) to examine possible relations between element concentrations and various biological covariables. Possible ways to normalize trace element concentrations in cod livers have been considered by ICES Working Groups and date at least back to a paper by N.W. Green in 1987 (Green, 1987). As recently as March 1996, this was an issue for the ICES Working Group on Statistical Aspects of Environmental Monitoring (WGSAEM). WGSAEM referred to much of the earlier work by the Marine Chemistry Working Group (MCWG) on this subject and came to the following conclusion: 'Lean (wet – fat) is the most appropriate basis for expressing zinc concentrations. Further corrections cannot be recommended based on these (Swedish and Norwegian) data. In particular, there seems to be no gain in correcting for water and expressing metals on a dry lean basis, possibly due to major analytical errors in the measurement of total dry %. For other metals (Cd, Cu), no clear picture could be obtained.'

Concentrations of inorganic and organic contaminants are generally low in the livers of cod from Icelandic waters, and these waters may be considered unpolluted. However, direct comparison with data on cod from other waters, e.g., data appearing in ICES (1988) and ICES (1991), shows that cadmium levels are generally higher in Icelandic cod livers than in the livers of cod from other areas of the Northeast Atlantic, while the levels of copper and zinc are generally lower. These characteristics are most likely attributable to some natural processes, the nature of which is still not known.

### 2 MATERIALS

Sampling took place in the years 1994, 1995, and 1996. Most samples were taken off northwestern, northeastern, southeastern, southern, and southwestern Iceland in March or at the beginning of April each year. Each sample consisted of 25 individuals, except for the 30–45 cm cod off northeastern Iceland, where 50 individuals were collected. Additional samples of length classes 15–30 cm, 45–60 cm, 60–75 cm, and 75–90 cm were collected off northeastern Iceland in 1996 and of length classes 45–60 cm and 60–75 cm in 1994. Furthermore, cod samples of length class 30–45 cm were sampled off northeastern Iceland in July and October 1994, January 1995, and June 1996. Altogether seventeen samples were obtained, or 454 cod (eight samples in 1994, one sample in 1995, and eight samples in 1996).

After selection in a length class, each fish was weighed, gutted, and the sex determined. Each liver was placed in a pre-weighed and pre-cleaned glass jar. On arrival at the laboratory, the gutted fish was weighed, and the length and age were determined as well as the weight of the muscle tissue. The livers of each sample were pooled into five or more groups. The groups were chosen so as to have as homogeneous liver sizes as possible in each group. The number of livers in a group ranged from one to eight. When the subsamples were prepared, the livers were first homogenized by an Ultra-Turrax and then all the material from each glass jar was transferred and weighed into the pooled subsample. Losses upon this transfer were a more or less constant figure of about 0.1–0.25 g, independent of liver size. Subsamples from each group were taken after thorough homogenization of the pooled sample. A total of 110 groups were prepared from the seventeen samples.

### 3 PARAMETERS STUDIED

The parameters analysed in the grouped samples are listed in Table A5.1, together with the corresponding

ranges of values obtained. The higher ranges of the chemical constituents were most often found for cod livers of fish in the length class 30–45 cm that had been sampled in March.

## 4 RESULTS

### 4.1 Macroconstituents of Cod Livers

The weight fraction of fat,  $X_f$ , and the weight fraction of moisture,  $X_{aq}$ , are linearly related in the cod livers, as shown in Figure A5.1, i.e.,  $X_{aq} = a - bX_f$ . This relation prevails coherently for the entire range of data presented here. Additionally, several other studies at the Icelandic Fisheries Laboratory, dating back to 1965, show the same behaviour, and it has been found that  $a$  and  $b$  in this equation are insignificantly different (95 % confidence level). Thus, the relation may be written as

$$X_{aq} = b \times (1 - X_f) = (0.7736 \pm 0.0033) \times (1 - X_f)$$

( $n = 110$ ;  $r^2 = 0.994$ ; 95 % confidence interval)

**Table A5.1.** Parameters analysed along with the corresponding ranges of values obtained.

Parameter	Minimum for a group	Maximum for a group	Ratio
Length, cm	18.1	79.8	4.4
Weight, g	57.3	5239	91.4
Liver size, g	0.87	721.3	829
Age, +year	2	8	4
Moisture, mg g <sup>-1</sup>	193.2	710.7	3.7
Fat, mg g <sup>-1</sup>	91.0	749.6	8.2
Nitrogen, mg g <sup>-1</sup>	5.70	26.25	4.6
Phosphorus, mg g <sup>-1</sup>	0.88	3.16	3.6
Total ash, mg g <sup>-1</sup>	3.65	14.85	4.1
Ca*, µg g <sup>-1</sup>	20.7	403.2	19.5
Mg*, µg g <sup>-1</sup>	41.6	255.6	6.1
Na*, µg g <sup>-1</sup>	373	1876	5.0
K*, µg g <sup>-1</sup>	779	3733	4.8
Zn, µg g <sup>-1</sup>	6.10	66.8	11.0
Fe, µg g <sup>-1</sup>	7.3	86.9	11.9
Mn, µg g <sup>-1</sup>	0.34	2.67	7.8
Cu, µg g <sup>-1</sup>	1.6	22.2	13.9
As*, µg g <sup>-1</sup>	1.26	19.2	15.2
Se, µg g <sup>-1</sup>	0.24	2.80	11.7
Pb, ng g <sup>-1</sup>		< 50	
Cd, ng g <sup>-1</sup>	26	842	32.4

\*Only analysed for samples from 1996.

implying that on extrapolation to zero fat content ( $X_f = 0$ ), the moisture content becomes  $b$  (77.4 %), and on extrapolation to 100 % fat ( $X_f = 1$ ), the moisture content becomes zero. However, these extreme values of fat in cod liver will not be found in nature.

In this way, other macroconstituents can be evaluated in terms of fat or, for practical reasons, moisture, since moisture may be determined easily and accurately, e.g., weight fraction of dry matter:

$$X_{DM} = 1 - X_{aq} = 1 - b(1 - X_f)$$

weight fraction of fat-free dry matter:

$$\begin{aligned} X_{ffDM} &= X_{DM} - X_{fDM} \\ &= X_{DM} - X_f \\ &= (1 - b) \times (1 - X_f) \\ &= [(1 - b)/b] \times X_{aq} \end{aligned}$$

weight fraction of fat-free liver or lean fraction:

$$X_l = 1 - X_f = (1/b) \times X_{aq}$$

From the relation for  $X_{ffDM}$ , it is seen that the ratio of fat-free dry matter and moisture in cod liver is given by  $(1 - b)/b$  for all liver sizes, resulting in the well-known figure of about 1/4 ( $b = 0.8$ ).

Figure A5.2 shows how  $X_f$  is related to the liver size, whereby it may be seen that dramatic increases occur in fat content in livers of up to about 100 g, after which the fat levels off. The six points for 80–200 g livers that deviate from the rest, all derive from samples of 45–60 cm and 60–75 cm cod caught off northeastern Iceland in 1994. Similarly, Figure A5.3 shows how the lean fraction  $X_l = 1 - X_f$  decreases sharply for livers up to about 100 g, after which the lean fraction levels off at about 0.3.

Thus, all cod livers behave similarly with respect to fat content, i.e., independent of the length and age of the fish and independent of season or fishing ground. Liver sizes increase with length (age) of the fish, but the relationship between liver size and fish length may be quite different from one area to another and from one time to another. For example, cod samples (25 individuals each) taken from northwestern Iceland in the years 1990–1996 (one sample taken in March each year of cod with an average length between 35 cm and 40 cm) had livers with average weights ranging between 5 g and 50 g. Within a year, individual livers from four samples taken from different locations around Iceland varied between 2 g and 100 g. Thus, the recommended sampling procedure results in liver sizes in a range where the rate of change in composition of the livers is the greatest. Figures A5.2 and A5.3 imply that livers of sizes greater than about 100 g are of similar composition in macroconstituents, at least. Thus, liver sizes greater than 100 g need to be sampled to

obtain samples of homogeneous composition, a sampling procedure that is not possible for practical reasons.

The weight of fat increases more slowly than the lean mass for small cod livers (Figure A5.4). However, as the liver becomes larger in size, the rate of increase in fat content increases and for 15–20 g livers, the masses of lean liver and fat are equal. For livers larger than about 20 g, both lean mass and fat weight increase linearly with liver weight, and the rate of increase in fat weight is about 3.5 times faster than that for lean mass.

## 4.2 Microconstituents of Cod Livers

The behaviour of microconstituent concentrations in relation to liver size in cod generally resembles that of the lean fraction, as shown for cadmium in Figure A5.5 and phosphorus in Figure A5.6. This behaviour implies that the livers not only have a uniform composition of macroconstituents for liver sizes greater than 100 g, but also a uniform composition of most inorganic analytes as well. The similarity in behaviour of the lean fraction and other inorganic components indicates, not unexpectedly, that inorganic constituents are to a large extent contained in the lean fraction and that there might exist some simple relationships between lean fraction and trace element content. In general, the liver burdens of all inorganic constituents analysed in this study behave in a linear log-log fashion with the lean weight of the livers, i.e., usually a good correlation is obtained with an equation of the form:

$$\ln(B) = \alpha + \beta \times \ln(m_l) + \delta \ln(A)$$

where most often independence of age ( $A$ ) is observed, i.e.,  $\delta$  is zero. ( $B$  denotes the liver burden:  $B = C \times M_L$ , where  $C$  is the wet weight concentration of the analyte;  $M_L$  is the mass of the liver (g);  $m_l$  is the lean weight of the liver (g);  $A$  is the age of the fish in years;  $r^2$  ranges from 0.83 for iron up to 0.994 for phosphorus and nitrogen). Log-transformed data give better correlations than untransformed data since analytes and biological properties of cod liver are log-normally distributed.

However,  $m_l$  and  $M_L$  are dependent variables since they are related by  $m_l = I_l \times M_L$ . Therefore, it violates strict statistical methods to correlate  $B$  ( $= C \times M_L$ ) and  $m_l$  ( $= X_l \times M_L$ ). Using these relations, the equation  $\ln(B) = \alpha + \beta \times \ln(m_l) + \delta \ln(A)$  may be rewritten as

$$\begin{aligned} \ln(C) &= \alpha + (\beta - 1) \times \ln(M_L) + \beta \times \ln(X_l) + \delta \ln(A) \\ &= \alpha + \gamma \times \ln(M_L) + \beta \times \ln(X_l) + \delta \ln(A) \end{aligned}$$

Therefore, only if  $\gamma = \beta - 1$  would it be statistically justifiable to correlate  $\ln(B)$  and  $\ln(m_l)$ . Multiple regression applying this equation to the data gives the regressional parameters listed in Table A5.2.

Figure A5.1. Relationship between weight fractions of fat and moisture in cod livers.

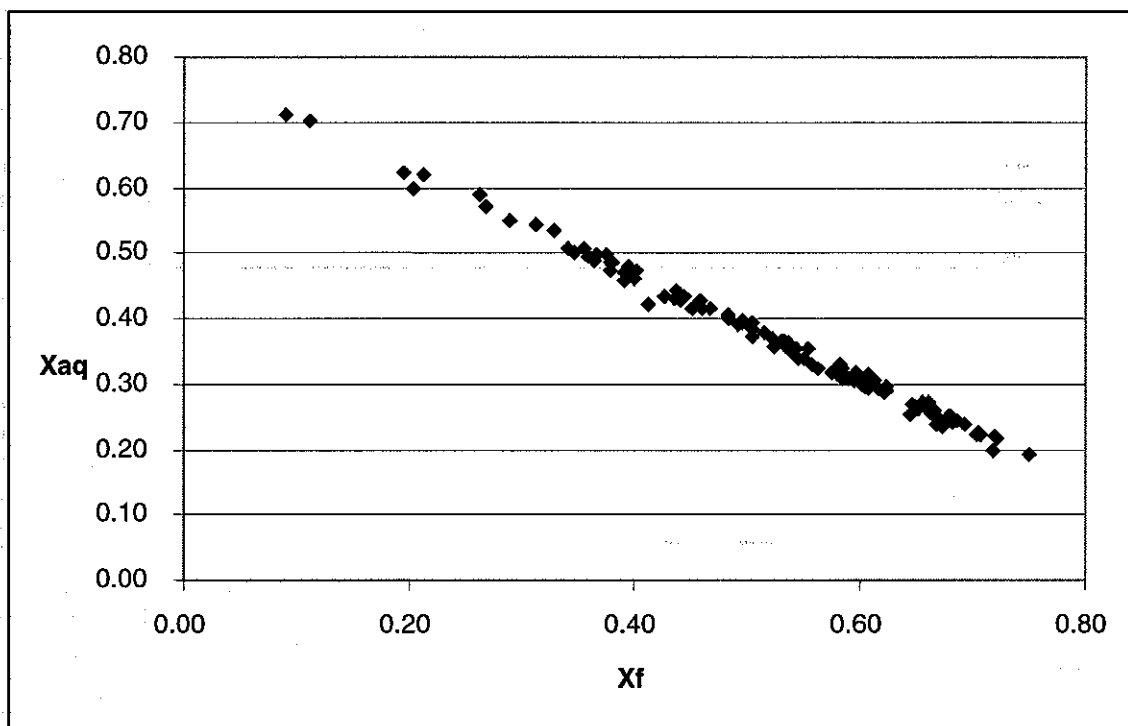
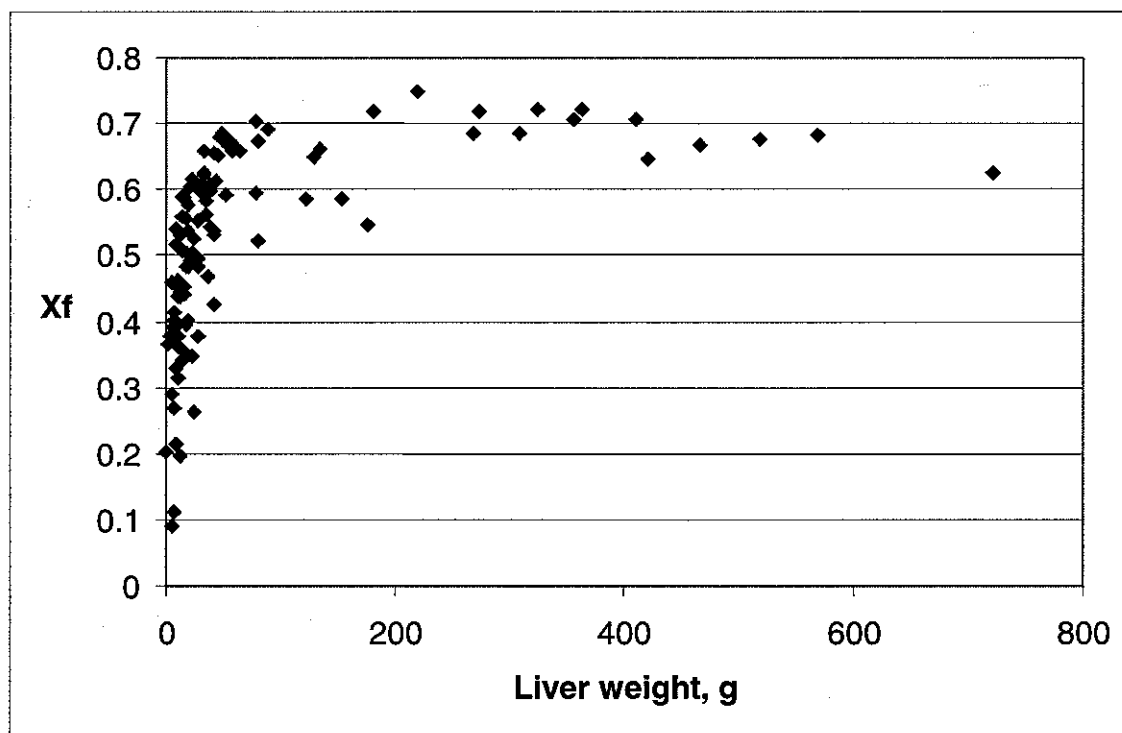
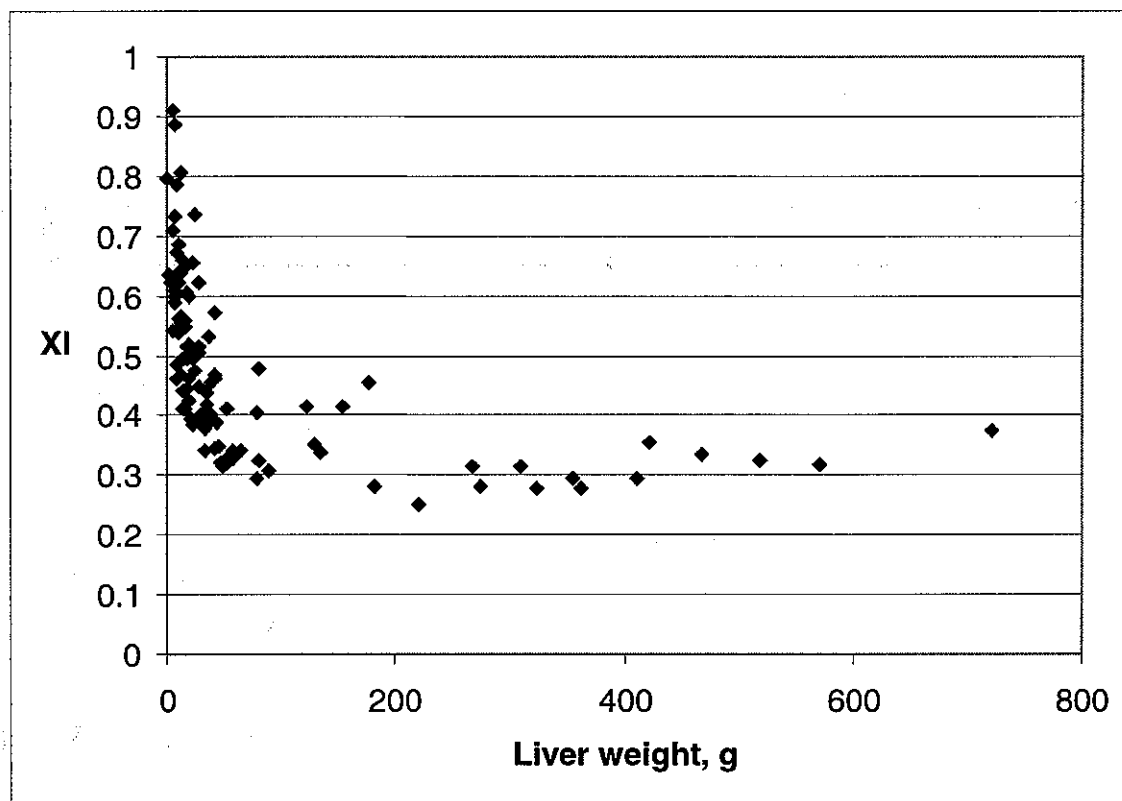


Figure A5.2. Relationship between the weight fraction of fat and the size of cod livers.





**Figure A5.3.** Relationship between the lean weight fraction and the size of cod livers.



**Figure A5.4.** Weights of lean mass and fat in cod livers in relation to total liver weight.

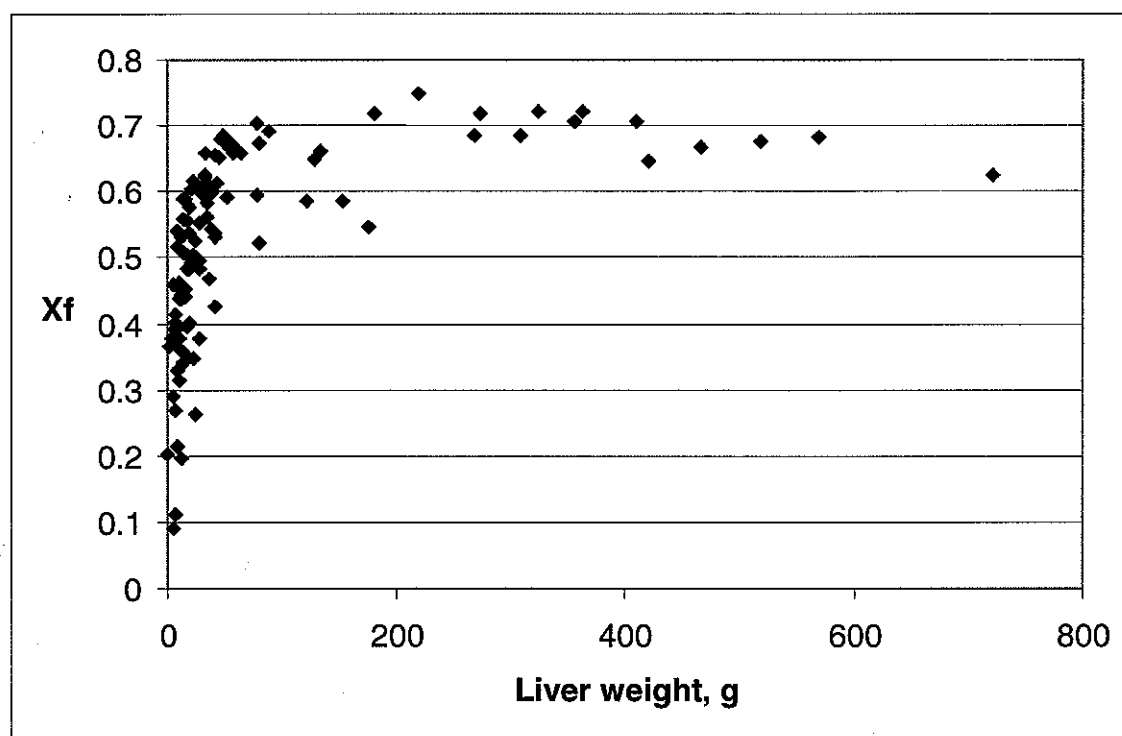


Figure A5.5. The relationship between cadmium concentrations and total liver weight in cod livers.

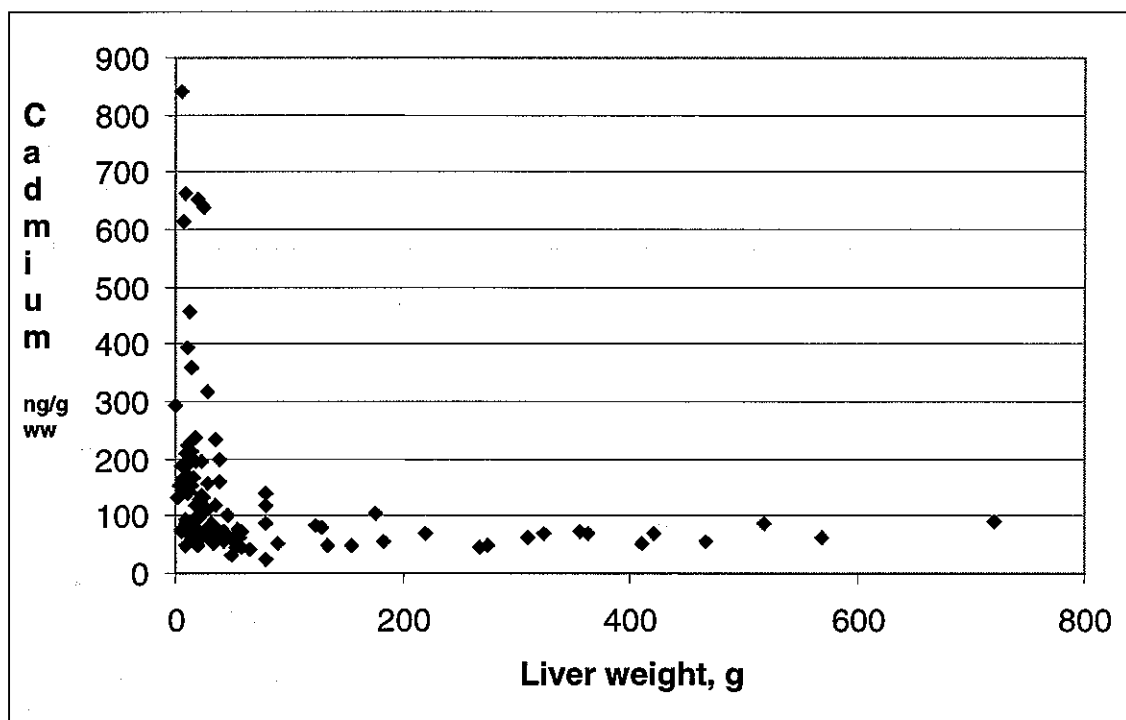
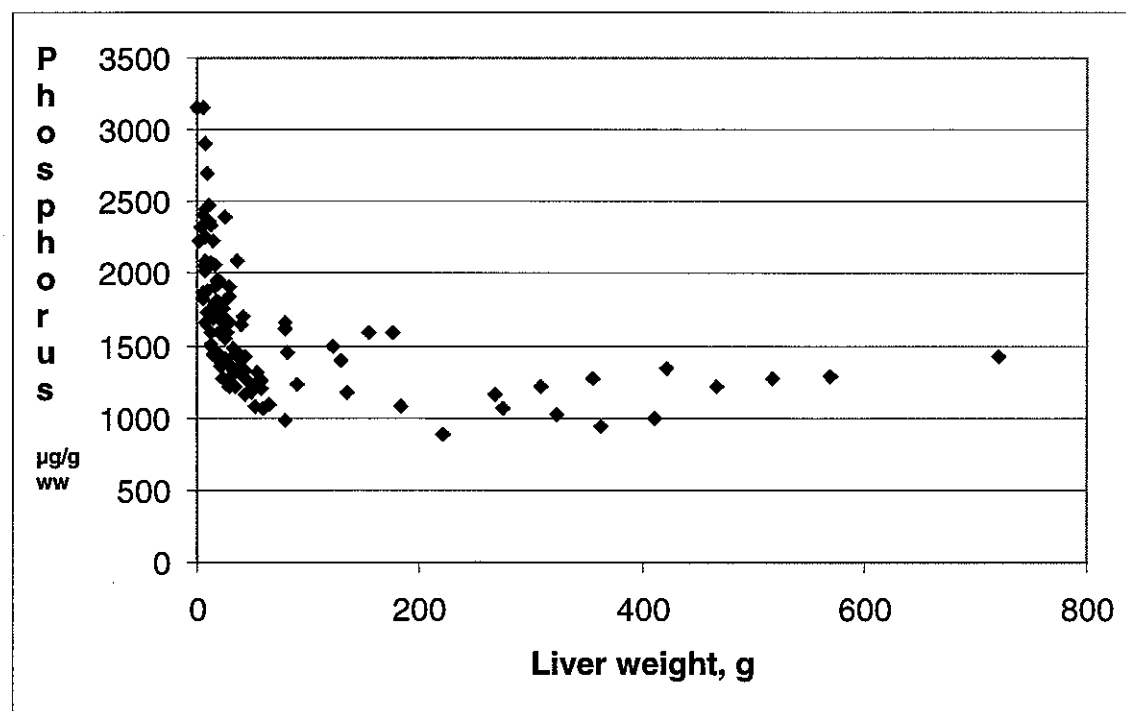


Figure A5.6. The relationship between phosphorus concentrations and total liver weight in cod livers.



**Table A5.2.** Estimates of  $\alpha$ ,  $\beta$ ,  $\gamma$ , and  $\delta$  in  $\text{Ln}(C) = \alpha + \beta \times \text{Ln}(X_l) + \gamma \times \text{Ln}(M_L) + \delta \times \text{Ln}(A)$  with 95 % confidence limits and the corresponding total correlation coefficients ( $n = 110$ , except for As and Zn where  $n = 59$ ).

Parameter	$\alpha$	$\beta$	$\gamma$	$\delta$	$r^2$
Moisture, $\text{mg g}^{-1}$	$6.664 \pm 0.013$	$1.022 \pm 0.015$	0	0	0.993
Nitrogen, $\text{mg g}^{-1}$	$3.25 \pm 0.06$	$0.971 \pm 0.084$	$-0.069 \pm 0.030$	$0.12 \pm 0.08$	0.95
Phosphorus, $\text{mg g}^{-1}$	$7.96 \pm 0.07$	$0.812 \pm 0.094$	$-0.046 \pm 0.032$	$0.16 \pm 0.09$	0.91
Total ash, $\text{mg g}^{-1}$	$2.64 \pm 0.07$	$0.951 \pm 0.093$	$-0.035 \pm 0.032$	$0.13 \pm 0.09$	0.93
Cd***, $\text{ng g}^{-1}$	$4.66 \pm 0.25$	$1.21 \pm 0.34$	$-0.40 \pm 0.11$	$1.78 \pm 0.30$	0.80
Cu, $\mu\text{g g}^{-1}$	$1.65 \pm 0.23$	$0.91 \pm 0.30$	$-0.14 \pm 0.10$	$0.67 \pm 0.29$	0.599
Mn, $\mu\text{g g}^{-1}$	$1.08 \pm 0.11$	$0.99 \pm 0.21$	$-0.089 \pm 0.048$	0	0.79
Se, $\mu\text{g g}^{-1}$	$1.19 \pm 0.12$	$1.44 \pm 0.22$	$-0.086 \pm 0.050$	0	0.86
Fe**, $\mu\text{g g}^{-1}$	$4.63 \pm 0.18$	$1.93 \pm 0.21$	0	0	0.78
As*, $\mu\text{g g}^{-1}$	$2.38 \pm 0.20$	$0.97 \pm 0.23$	0	0	0.56
Zn*, $\mu\text{g g}^{-1}$	$3.47 \pm 0.11$	$1.21 \pm 0.12$	0	0	0.877

\* Only analysed in samples from 1996.

\*\* For Fe, all subsamples from southwestern Iceland (both for 1994 and 1996) were excluded in the regression as they showed significantly higher concentrations than other samples when examined for covariables ( $n = 98$ ).

\*\*\* For Cd, all subsamples taken off northwestern Iceland were excluded since they showed significantly and consistently higher concentrations than other samples upon inspection for effects of covariables ( $n = 100$ ); these higher concentrations of cadmium off northwestern Iceland have also been noted in previous years.

The best correlation is obtained for the macro-constituents, i.e., moisture, nitrogen, phosphorus, and total ash. Of the trace elements, zinc, selenium, and cadmium show the best adherence to the equation, while arsenic and copper deviate the most.

On the basis of regression parameters shown in Table A5.2, the results may be summarized as follows:

### I. Neither age of fish nor total liver weight affects concentration ( $\gamma = \delta = 0$ ).

**Arsenic.** Arsenic shows the simplest behaviour since  $\beta = 1$ , i.e., the concentration of arsenic is constant in the lean fraction of the liver independent of the size of the lean fraction, liver size or age. Thus, the equation for arsenic simplifies to  $\text{Ln}(C_l) = \alpha$ , where  $C_l$  is the concentration based on lean weight.

**Moisture, zinc, iron.** For these analytes,  $\beta > 1$  and therefore their concentrations, whether expressed on a wet weight or lean weight basis, will increase with increased lean fraction of the liver.

For moisture,  $\beta$  is only slightly but significantly (at the 95 % level) higher than unity. Direct correlation of  $X_l$  and  $X_{aq}$  above gave a similar correlation ( $r^2 = 0.994$ ) as that here for the log-transformed data ( $r^2 = 0.993$ ).

### II. Total liver size influences the concentration ( $\gamma \neq 0$ ) while age does not ( $\delta = 0$ ).

**Manganese.** For manganese,  $\gamma$  is insignificantly different from  $\beta - 1$ , i.e., the use of the simplified equation for manganese is warranted:

$$\text{Ln}(B) = \alpha + \beta \times \text{Ln}(m_l).$$

This equation results in better estimates of the regression parameters due to better correlation:  $\alpha = 1.057 \pm 0.102$  and  $\beta = 0.896 \pm 0.036$  ( $r^2 = 0.95$ ). From this it is seen that  $\beta < 1$  and, therefore, the lean weight based concentration of manganese decreases with increased lean weight of the livers; this is also clearly seen by plotting the data [ $\text{Ln}(C_l)$  vs.  $\text{Ln}(m_l)$ ].

**Selenium.** For selenium,  $\gamma$  is significantly different from  $\beta - 1$  and  $\beta > 1$ . Increased liver sizes (and concomitant decreases in lean fraction up to 100 g livers) result in decreased concentrations of selenium, i.e., as for manganese both the lean fraction term [ $\beta \times \text{Ln}(X_l)$ ] and the term with total liver mass [ $\gamma \times \text{Ln}(M_L)$ ] result in decreased concentrations of selenium with increased liver size, both when the concentration is based on wet weight and when it is based on lean weight.

### III. Both age and total liver size affect concentration.

The effect of older age of the fish is always to increase the constituent concentration; this is similar for all the macroconstituents (total nitrogen, total phosphorus, and total ash), while copper and especially cadmium are much more affected.

**Copper, nitrogen, and ash.** For these constituents,  $\gamma + 1$  is insignificantly different from  $\beta$ . Therefore, a simplified equation is justified for them, i.e.,

$$\ln(B) = \alpha + \beta \times \ln(m_i) + \delta \times \ln(A)$$

resulting in better correlation ( $r^2$  from 0.941 (copper) to 0.994 (nitrogen and ash)), showing that  $\beta < 1$  at a 95 % confidence level. Thus, all these analytes on a lean weight basis decrease in concentration with increased lean weight of the liver. This is often counteracted by the positive effect of age, since lean weight generally increases with age.

**Cadmium.** This element has  $\gamma + 1$  significantly different from  $\beta$  and  $\beta$  is insignificantly different from unity. Thus, cadmium concentrations on a lean weight basis decrease with increased liver size, but this effect is somewhat counteracted by age. For the data here under study, normalization of the cadmium concentrations with the help of the relationship obtained reduces the relative standard deviation of the entire data set from about 90 % to 25 %.

**Phosphorus.** This element has  $\gamma + 1$  significantly different from  $\beta$  and  $\beta < 1$ . Phosphorus concentrations, on a wet weight basis, decrease with liver size, an effect strengthened by the concomitant decrease in the lean fraction. However, the variability decreases somewhat when phosphorus is expressed on a lean weight basis, since then the effect of the lean fraction:

$$(\beta - 1) \times \ln(X_i),$$

counteracts the effect of total liver weight.

### 5 GENERAL CONCLUSIONS

It is apparent that due to the great variability of sizes and thereby composition of cod livers and the complicated effects these variations have on trace elements, cod livers are not very well suited for monitoring studies. However, they usually contain higher levels of contaminants than other tissues of cod, especially muscle tissue, and thereby the analysis of trace elements becomes more reliable. Furthermore,

cod livers are believed to reflect concentrations in the marine environment. Therefore, large data sets on contaminants in cod livers have been collected, e.g., within the OSPAR Commission and the Arctic Monitoring and Assessment Programme (AMAP). The information obtainable from these data may be substantially increased if the effects of biological covariables are taken into account. The study presented here may be of help in interpreting data from Icelandic waters, but the models need not necessarily apply for cod from other waters. The effect of biological covariables on trace element concentrations in cod livers from other areas needs to be known if meaningful spatial comparisons are to be performed, and the effects of covariables must, of course, also be known when temporal trends in a given area are studied. A common basis for spatial and temporal comparisons might possibly be hypothetical cod livers of 1 g and 100 % lean fraction from cod of one year of age, where the effects of the lean fraction, liver size, and age cancel out, i.e.,  $\ln(C) = \alpha$  for all elements in this study.

### 6 ACKNOWLEDGEMENT

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## REVIEW NOTE ON PATHOLOGICAL ASPECTS OF ENDOCRINE-DISRUPTING CHEMICALS IN ESTUARINE AND MARINE ORGANISMS

### 1 INTRODUCTION

The endocrine system includes several feedback pathways between the central nervous system, the pituitary and the target organs. These pathways are concerned with the regulation of different metabolic functions and in the maintenance of homeostasis. They also have a role in growth regulation and reproduction. Reproduction in all vertebrates is regulated by a complex system of hormones, known as classical hormones. Hormones with similar functions, such as ecdysone, are found in several invertebrate groups. As well as for sexual development, hormones also play a critical role in brain development and immune system function. Thus, the effects of endocrine-disrupting chemicals (EDCs) on populations of estuarine and marine organisms are unlikely to be confined to reproduction. They may also include, for example, abnormal behaviour, altered immunomodulation, and increased risk of developing cancer.

In this review, EDCs are defined broadly, following the 1996 European workshop on the impact of endocrine disruptors on human health and wildlife, held in Weybridge, UK. They are defined as exogenous substances that cause adverse health effects at the whole-organism level as a result of changes in endocrine function. Thus all potential effects arising from endocrine disruption are considered. These effects include, but may not be restricted to:

- 1) effects via mediation of specific hormone receptors (oestrogen, androgen, progesterone, and thyroid hormone receptors);
- 2) effects through interference with hormone metabolism;
- 3) effects through interference with feedback mechanisms involving the hypothalamus or pituitary gland;
- 4) direct effects on endocrine organs.

Receptor-mediated effects, particularly those involving the oestrogen receptor, have received the most attention in the literature. However, other proposed mechanisms such as altered steroid metabolism are also important. An example of the latter is the inhibition of P450 aromatase by tributyltin oxide (TBTO) as a cause of imposex in marine gastropods.

The study of EDCs poses several problems. Firstly, although the majority of known EDCs are persistent and bioaccumulative, they are often excluded from routine monitoring programmes because of a lack of adequate

analytical methods. Secondly, most of the EDCs actually present in the environment seem to be unknown or unidentified compounds, concentrated in particular in the hydrophilic fraction of samples. The use of *in vitro* oestrogen-receptor assays indicates that the greater part of the response measured cannot be explained in terms of conventionally identifiable chemicals in samples (Legler *et al.*, 1997). Thirdly, EDCs with similar modes of action may have synergistic, cumulative effects. Fourthly, the critical period for exposure is often in the very early stages of the organism's life. Lastly, the effects of EDCs may be manifested at very low concentrations (the part-per-thousand level or lower).

Specific sources of EDCs in the marine environment include:

- estuarine discharges (e.g., oestrogenic discharges in UK and Dutch estuaries);
- offshore oil and gas installations (e.g., certain polycyclic aromatic hydrocarbons (PAHs), nonylphenol (NP));
- ships (e.g., TBT);
- atmospheric deposition (e.g., certain pesticides, toxaphenes).

### 2 LITERATURE REVIEW

An overview of pathological and non-pathological changes associated with exposure to EDCs is given in Table A6.1. Published work suggests an increase in the number of observed or suspected instances of endocrine disruption in recent years. There are recent reports of effects on a wide range of freshwater, estuarine and marine animals, including mammals, fish, and molluscs. Most instances have apparently been discovered by accident and appear to be local effects. An exception may be the world-wide occurrence of imposex in marine snails, caused by organotin compounds.

Observed effects associated with EDCs can be broadly categorized as follows:

- 1) decreased fertility in marine mammals, fish, and molluscs;
- 2) demasculinization and feminization in fish;
- 3) defeminization and masculinization in fish and gastropods;
- 4) decreased hatching success in fish;
- 5) abnormal thyroid function in marine fish and seals;

- 6) altered immune function in marine mammals and fish;
- 7) liver tumours and skin lesions in fish.

Most of the above effects involve associations or circumstantial evidence rather than proven causal relationships. A problem with the interpretation of field observations, in particular, is that causal relationships cannot be proved because other factors could be involved. These factors include other environmental contaminants, as well as environmental and host-related stress factors.

It should be noted that, strictly speaking, not all the conditions listed in Table A6.1 can be considered as pathological disorders. The population consequences of several observed effects are still unknown. For example, the significance of increased vitellogenin concentrations in the blood of male fish and of intersex in fish (see Table A6.1 for details) is not yet fully understood and requires further investigation. The background levels of intersex are not known, nor is the extent to which affected males remain functionally male. Recent studies, however, indicate that high vitellogenin levels may lead to kidney failure and death in rainbow trout (Nimrod and Benson, 1996). They may also lead to decreased testicular growth and inhibition of spermatogenesis (Jobling *et al.*, 1996).

The nature of the observed effects, together with the fact that they seem to be occurring with increasing frequency, underlines the potential risks posed by EDCs for marine ecosystems, fisheries, and human consumers.

### 3 FUTURE MONITORING

When considering monitoring requirements, it is useful to start with the recommendations of the Workshop on Endocrine Modulators and Wildlife: Assessment and Testing (EMWAT), held in Veldhoven, the Netherlands in April 1997 (Tattersfield *et al.*, 1997). The workshop proposed a general monitoring programme involving screening surveys to determine the general health status of wildlife populations (Vethaak *et al.*, 1997).

In accordance with the workshop recommendations, the current ICES monitoring programme on fish diseases should be continued and extended to include the following:

- 1) reproductive and developmental disorders;
- 2) relevant reproductive endpoints, such as the gonadosomatic index, and a general histological assessment of gonadal tissues. The latter would allow the occurrence and prevalence of phenomena such as intersex and hermaphroditism to be quantified in the target populations;

- 3) population-related parameters such as age structure, abundance, and sex ratios;
- 4) suitable biomarkers of EDC effects, such as plasma vitellogenin concentration as a measure of the effects of oestrogenic compounds;
- 5) immunological methods, in order to establish the immunological status of the target populations (e.g., Boonstra *et al.*, 1996; Vos *et al.*, 1996).

In addition, the use of computer-assisted sperm analysis (CASA), as recently developed for measuring pollution effects on sperm quality in freshwater fish (Kime *et al.*, 1996), should be investigated.

New target species should also be chosen. They should include both fish and invertebrate species, covering a variety of different life styles and levels in the food chain. Where possible, several different stages in the life cycle of each species should be investigated.

### 4 ACKNOWLEDGEMENT

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**Table A6.1.** Effects associated with exposure of estuarine and marine animals to endocrine-disrupting chemicals.

<b>Marine mammals</b>	<ul style="list-style-type: none"> <li>• High incidence of premature births, associated with unusually high concentrations of DDT-like compounds (sea lion; California) (DeLong <i>et al.</i>, 1973; Addison, 1989).</li> <li>• Low reproductive success and reproductive disorders, associated with high lipid concentrations of polychlorinated biphenyls (PCBs) (seal; Baltic Sea) (Helle <i>et al.</i>, 1976a, 1976b).</li> <li>• High degree of skull asymmetry indicative of disrupted development, attributed to pollution (grey seal in the Baltic Sea) (Zakharov and Yablokov, 1990).</li> <li>• Poor reproductive performance, associated with immunosuppression and presence of PCBs and polyhalogenated aromatic hydrocarbons (PHAHs) in body fat and food (common and grey seals; Baltic, North and Wadden Seas) (Reijnders, 1996; Brouwer <i>et al.</i>, 1989; Reijnders and Brasseur, 1992; De Swart <i>et al.</i>, 1992, 1994; Ross <i>et al.</i>, 1995).</li> <li>• Occurrence of thyroid and adrenal cortex lesions, hermaphroditism and reproductive disorders (beluga whale; Martineau <i>et al.</i>, 1988).</li> <li>• Contamination-related immunosuppression (striped dolphin; Mediterranean Sea) (Scott <i>et al.</i>, 1988).</li> <li>• High prevalence of neoplasms and frequent infection by mildly pathogenic bacteria, indicating contamination-related immunosuppression (beluga whale; Lahvis <i>et al.</i>, 1995; De Guise <i>et al.</i>, 1995).</li> </ul>
<b>Fish</b>	<ul style="list-style-type: none"> <li>• Elevated vitellogenin levels and testicular abnormalities, associated with pollution (flounder; UK estuaries) (Allen <i>et al.</i>, 1997).</li> <li>• Elevated blood vitellogenin in female fish, associated with pollution (flounder; Scottish estuaries) (Lye <i>et al.</i>, 1997).</li> <li>• Elevated blood vitellogenin in female fish, associated with estuarine pollution (winter flounder; Boston harbour) (Pereira <i>et al.</i>, 1992).</li> <li>• Premature vitellogenesis and reduced Vitamin A levels resulting from exposure to polluted dredged spoil in mesocosms (flounder; the Netherlands) (Janssen <i>et al.</i>, 1997; Besselink <i>et al.</i>, 1998).</li> <li>• Increased vitellogenin levels in the blood of caged male fish (rainbow trout; UK estuaries) (Harries <i>et al.</i>, 1996, 1997).</li> <li>• Increased vitellogenin levels in the blood of male fish (flounder; UK and Dutch estuaries) (Allen <i>et al.</i>, 1997; Allen, unpubl. results).</li> <li>• Occurrence of intersex at high prevalence in male fish (flounder; certain UK estuaries, but apparently absent from the Netherlands) (Allen <i>et al.</i>, 1997; Vethaak, unpubl. results).</li> <li>• Various reproductive effects (precocious maturation, reduced reproductive output, reduced hatching success, and reduced larval growth and survival), associated with increased concentrations of lipophilic xenobiotics (cod; Baltic Sea) (Petersen <i>et al.</i>, 1997).</li> <li>• Sterility, associated with high levels of plastic constituents and natural hormones in sewage outflows (trout; UK rivers) (Jobling <i>et al.</i>, 1996).</li> <li>• Reduced reproductive success and larval mortality (eelpout; Baltic Sea) (Draganik <i>et al.</i>, 1995).</li> <li>• Occurrence of thyroid disorders, reduced fertility and reduced embryonic survival (salmon; Great Lakes) (Moccia <i>et al.</i>, 1981; Leatherland, 1992).</li> <li>• Masculinization of appearance and behaviour of female fish, apparently caused by the concentrated mixture of phytosterols (from trees) and chlorine in pulp-mill effluent (mosquito fish; Howell <i>et al.</i>, 1980; Howell and Denton, 1989; Davis and Bortone, 1992; Bortone and Davis, 1994).</li> <li>• Altered serum steroid levels, associated with exposure to bleached kraft mill effluent (white sucker; Lake Superior) (Munkittrick <i>et al.</i>, 1991).</li> <li>• Disrupted ovarian development, associated with exposure to contaminants (English sole; Puget Sound, USA) (Johnson <i>et al.</i>, 1988).</li> <li>• Delayed gonadal maturation, associated with exposure to bleached kraft mill effluent and other anthropogenic influences (mummichog; Canada) (Leblanc <i>et al.</i>, 1997).</li> <li>• Inhibition of spawning in female fish exposed to diluted sewage sludge under laboratory conditions (sand goby; Waring <i>et al.</i>, 1996).</li> <li>• Increased occurrence of embryonic malformation in pelagic eggs (various fish species; North Sea coastal waters) (Cameron <i>et al.</i>, 1992).</li> <li>• Changes in the sex ratio of North Sea dab (<i>Limanda limanda</i>) in the period 1981–1995 (Lang <i>et al.</i>, 1995).</li> <li>• A decrease in the size and age at first maturation in North Sea plaice and sole between 1960 and 1995 (Rijnsdorp and Vethaak, 1997).</li> </ul>
<b>Molluscs</b>	<ul style="list-style-type: none"> <li>• Occurrence of imposex, caused by TBT/TPT (gastropods; North Sea) (Hallers-Tjabbes <i>et al.</i>, 1994; Mensink <i>et al.</i>, 1997; Matthiessen and Gibbs, 1998).</li> </ul>
<b>Crustaceans</b>	<ul style="list-style-type: none"> <li>• Occurrence of intersex, associated with exposure to sewage discharges (several species of harpacticoid copepod; UK) (Moore and Stevenson, 1991, 1994).</li> </ul>



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## ANNEX 7

### MONITORING STRATEGIES TO ASSESS HAZARDS PRESENTED BY THE DISCHARGE OF PRODUCED WATER BY OFFSHORE OIL AND GAS INDUSTRIES

#### 1 PRODUCED WATER

Produced water consists of water naturally present in the oil and gas reservoir (formation water), flood water previously injected into the formation and/or, in the case of some gas production, condensed water (Anon., 1994).

According to Stephenson (1992):

*'Produced water is the largest volume waste stream in the exploration and production process. Over the economic life of a producing field, the volume of produced water can exceed by ten times the volume of hydrocarbon produced. During the later stages of production, it is not uncommon to find that produced water can account for as much as 98 % of the extracted fluids.'*

The amount of produced water that is discharged into the sea varies over the lifetime of the oil field and with production type (oil or gas). Various numbers are given for the amount of produced water that is or will be discharged into the sea.

Anon. (1994) reports the following data for the annual discharges from oil and gas platforms into the North Sea:

Year	Total Annual Discharge
1989	$130 \times 10^6 \text{ m}^3$
1990	$146 \times 10^6 \text{ m}^3$
1991	$160 \times 10^6 \text{ m}^3$
1998*	$340 \times 10^6 \text{ m}^3$

\* = expected

A breakdown according to the different sectors of the North Sea is indicated below:

Year	Total Annual Discharge	Sector
1991	$130 \times 10^6 \text{ m}^3$	UK
1991	$16.4 \times 10^6 \text{ m}^3$	Norway
1998*	$330 \times 10^6 \text{ m}^3$	UK and Norway
1991	$10.7 \times 10^6 \text{ m}^3$	Netherlands
2000*	$5.2 \times 10^6 \text{ m}^3$	Netherlands
1991	$1.5 \times 10^6 \text{ m}^3$	Denmark
2000*	$9.7 \times 10^6 \text{ m}^3$	Denmark

\* = expected

Typical water production rates from different types of installation (Anon., 1994) are:

Installation Type	Production Rate
Oil	$2,400\text{--}40,000 \text{ m}^3 \text{ day}^{-1}$
Gas	$1.6\text{--}30 \text{ m}^3 \text{ day}^{-1}$

Data on discharges into the Norwegian sector of the North Sea are summarized by Anon. (1998) as follows:

Year	Total Annual Discharge
1996	$66 \times 10^6 \text{ m}^3$
2000*	$120 \times 10^6 \text{ m}^3$

\* = expected

Røe *et al.* (1996) estimated that discharges from the Norwegian sector of the North Sea were  $26 \times 10^6 \text{ m}^3$  in 1993 and will be as much as  $90 \times 10^6 \text{ m}^3$  in the year 2000. As these figures demonstrate, there is a significant increase in the amount of discharges from Norwegian sources.

The North Sea countries are supposed to report their discharges to the OSPAR Commission's Secretariat in London, but this reporting has been insufficient in recent years (Anon., 1996). There are many reasons for this inadequate reporting practice, some of which relate to the different ways national authorities implement regulations, which can influence discharge volumes and reporting; in addition, machinery and other tools used in oil and gas production have varying discharge ratios depending on their design and maintenance. The calculated discharge is typically based on theoretical formulas derived for the actual machinery in use.

In order to account for general uncertainty and the inaccuracy of theoretical calculations for total discharges of components with the produced water, an estimated annual amount of  $100 \times 10^6 \text{ m}^3$  will be used here to calculate discharges of dissolved components to the Norwegian sector and, correspondingly for the North Sea as a whole, an annual discharge total of  $300 \times 10^6 \text{ m}^3$  will be used.

#### 1.1 Composition of Produced Water

Produced water contains a variety of dissolved inorganic salts and organic compounds characteristic of the geological formation from which the water was produced.

## ***Inorganic Components***

### ***Salts and trace metals***

The concentration of dissolved salts in produced water is usually higher than in sea water, but may range from about 3 g l<sup>-1</sup> to near saturation (Anon., 1994). The salt components chloride and sodium, as in sea water, represent the major part of the salt content of the produced water. In addition to the normal salts, the inorganic components include trace metals in considerably higher concentrations compared to their concentrations in sea water. Table A7.1 shows examples of concentration ranges of several trace elements in produced water.

### ***Radionuclides***

Radionuclides, primarily radium, occur naturally in the formations of oil and gas wells and are transported to the surface with the well-stream in the water phase. Radium co-precipitates with barium, forming scales in pipes. Due to this concentration, radioactivity has in some cases been measured on the outside of the pipes. In the Norwegian sector, radioactive scales are brought to land for permanent storage. The discharge of naturally occurring radionuclides with produced water does not represent a serious contamination problem. However, it merits mention in order to demonstrate the complexity of handling produced water and its dissolved components.

## ***Organic Components***

The concentrations of dissolved organic components are not measured on a routine basis on oil and gas platforms in the North Sea. Only oil is measured routinely, and this is done according to an OSPAR-prescribed analytical method that mainly concerns aliphatic hydrocarbons as dispersed oil (Anon., 1994). The ACME is not aware of any national or international regulations covering the composition of the dissolved components in produced water. This means that the composition of the dissolved components and the magnitude of such discharges are more or less hidden as far as regulatory purposes are concerned.

The analytical difficulties involved in measuring dissolved organic and inorganic components in produced water do not allow for routine analysis on board the oil and gas platforms. Assessments of possible impacts of discharges from a single platform or field have to be based on assumptions.

As part of the production line, varying amounts of methanol are used to prevent hydratization and ice formation at the well-head. The injection of methanol in the well-stream may also increase the solubility of organic components in the water phase and, thereby, increase the discharge of the various components. The use of methanol and the amounts used are seldom

included in oil company presentations for regulatory purposes.

### ***Carboxylic acids***

Of the organic compounds dissolved in produced water, carboxylic acids represent the largest amount, with concentration ranges from 30–930 mg l<sup>-1</sup>; acetic acid comprises about 90 % (Anon., 1994, 1995). For further calculations, an average concentration of 500 mg l<sup>-1</sup> has been assumed.

### ***Volatile aromatic hydrocarbons***

The volatile aromatic hydrocarbons are relatively soluble in water and solubility increases with decreasing temperature; thus, this class of compounds in discharged produced water may have a greater impact in the North Sea and further north than, for example, in the Gulf of Mexico. Stagg *et al.* (1996b) have reported concentrations based on analytical measurements of heat-produced water from four installations discharging produced water into the UK sector of the North Sea, which again demonstrates the variation in concentrations from different oil fields.

In Table A7.2 concentration data on the so-called BTEX (benzene, toluene, ethyl benzene, and xylenes) components have been compiled from Stagg *et al.* (1996b) and Anon. (1994, 1995).

### ***Phenols***

The concentrations of phenols in produced water from the North Sea vary between 1–23 mg l<sup>-1</sup> (Anon., 1994), 1.3–8 mg l<sup>-1</sup> (Anon., 1995), and 1.2–1.5 mg l<sup>-1</sup> (Stagg *et al.*, 1996b). Based on these data, an average concentration of phenols of 5 mg l<sup>-1</sup> has been assumed and used in further calculations.

### ***Polycyclic aromatic hydrocarbons (PAHs)***

The discharge of PAH-containing compounds is considerable, ranging from 40–1600 µg l<sup>-1</sup>, with naphthalene (41–1600 µg l<sup>-1</sup>), phenanthrene (10.7–500 µg l<sup>-1</sup>), and dibenzothiophene (10–170 µg l<sup>-1</sup>) representing the majority (Anon., 1995). Variations in the reported concentrations, however, are considerable. Anon. (1994) reported values for naphthalene of 66 µg l<sup>-1</sup>, phenanthrene of <2 µg l<sup>-1</sup>, and dibenzothiophene of 0.5 µg l<sup>-1</sup>, while Stagg *et al.* (1996b), for example, reported concentrations of methyl naphthalenes between 770–1700 µg l<sup>-1</sup> and phenanthrene concentrations from 15–50 µg l<sup>-1</sup>. Based on these figures, the assumed average PAH concentration of 300 µg l<sup>-1</sup> in general use may be rather conservative.

**Table A7.1.** Examples of concentration ranges of trace elements in produced water. (Source: Anon., 1994, 1995.)

Pb $\mu\text{g l}^{-1}$	Cd $\mu\text{g l}^{-1}$	Cu $\mu\text{g l}^{-1}$	Hg $\mu\text{g l}^{-1}$	Ni $\mu\text{g l}^{-1}$	Zn $\mu\text{g l}^{-1}$	As $\mu\text{g l}^{-1}$	Cr $\mu\text{g l}^{-1}$
< 50	< 500	20–30	1.9–12	< 40	6–11	1–12	< 200
< 10	< 10	< 2	< 0.05	20–95	5–230	0.004	32–60
< 1	< 1	< 1	< 0.0001	< DL*–30	< DL*	< DL*	< 0.001

DL\* = detection limit

**Table A7.2.** Concentrations (in  $\text{mg l}^{-1}$ ) of BTEX components in produced water from oil and gas platforms in the North Sea.

Compound	Clyde <sup>1</sup>	Forties Charlie <sup>1</sup>	Brent Delta <sup>1</sup>	Brae Alpha <sup>1</sup>	Average oil <sup>2</sup>	Average gas <sup>2</sup>	Tampen <sup>3</sup>
Benzene	4.2	1.4	6.9	5.3	0.4–5	0.3–400	
Toluene	2.8	2.2	2.9	2.4	0.01–2	4–145	
Ethyl benzene	0.9	0.4	1.0	1.5			
Xylenes	2.9	0.7	1.8	3.4	0.1–7	0.8–84	
$\Sigma$ BTEX	10.8	4.7	12.6	12.6	0.5–14	5–629	1–7.3

1 = Stag *et al.* (1996b)

2 = Anon. (1994)

3 = Anon. (1995)

**Table A7.3.** Estimated annual discharges (in tonnes) of some dissolved components in the produced water from oil platforms in the Norwegian sector and the North Sea.

Quantity of produced water	Carboxylic acids 500 $\text{mg l}^{-1}$	BTEX 8 $\text{mg l}^{-1}$	Phenols 5 $\text{mg l}^{-1}$	PAHs 300 $\mu\text{g l}^{-1}$	Hg 5 $\mu\text{g l}^{-1}$	Cd 10 $\mu\text{g l}^{-1}$
$1 \times 10^8 \text{ m}^3$ (Norwegian sector)	50,000 t	800 t	500 t	30 t	0.5 t	1 t
$3 \times 10^8 \text{ m}^3$ (Whole North Sea)	150,000 t	2400 t	1500 t	90 t	1.5 t	3 t

Using known concentration data, assumed concentration values can be determined and then used to calculate the annual total discharge of dissolved components into the North Sea.

Table A7.3 summarizes the discharges of some potentially harmful major components into the Norwegian sector and into the whole North Sea.

### Oil

In addition to dissolved components, produced water contains oil as such, e.g., fine oil droplets and/or dispersed oil. This is rest oil that the oil/water separators have not been able to retain in the oil stream. OSPAR has set a limit of 40  $\text{mg oil l}^{-1}$  as the total allowable concentration in discharged water. Most oil companies claim to have achieved better results, i.e., oil concentrations between 10–30  $\text{mg l}^{-1}$ . Assuming an average concentration of 20  $\text{mg l}^{-1}$  of dispersed oil in the discharged produced water, this gives an annual input of 2000 tonnes of oil to the Norwegian sector and 6000 tonnes to the entire North Sea.

The assessment report for the Tampen area in the Norwegian sector (Anon., 1995) reports an annual

discharge of 405 tonnes of dispersed oil in 1993, increasing to an estimated 1774 tonnes in the year 2003. The annual discharge of oil with produced water represents only a small fraction of the total annual oil load into the North Sea, which totals about 130,000–260,000 tonnes (Anon., 1994).

### Total organic load

The organic load can be presented in terms of biodegradability, i.e., COD (chemical oxygen demand) and/or BOD (biological oxygen demand). Anon. (1994) reported COD values for produced water in the North Sea from 100–15,800  $\text{mg O}_2 \text{ l}^{-1}$  and for oil platforms in USA waters from 100–3000  $\text{mg O}_2 \text{ l}^{-1}$ , with an average value for central North Sea oil platforms at 4160  $\text{mg O}_2 \text{ l}^{-1}$ . The reported BOD values vary between 28  $\text{mg O}_2 \text{ l}^{-1}$  to 6700  $\text{mg O}_2 \text{ l}^{-1}$ .

By using an average COD or BOD value of 4000  $\text{mg O}_2 \text{ l}^{-1}$  for produced water and assuming an oxygen content in North Sea water of 8  $\text{mg O}_2 \text{ l}^{-1}$ , one litre of produced water needs the dissolved oxygen in 500 l of sea water for the biodegradation of its organic load. The oxygen consumption of the estimated  $3 \times 10^8 \text{ m}^3$  of produced water annually discharged into the North Sea will then,

based on these rough estimates, deplete the dissolved oxygen of approximately  $15 \times 10^{10} \text{ m}^3$  of North Sea water.

The produced water is also oxygen depleted and will, therefore, in addition to the oxygen consumption due to the biodegradation processes, require a certain amount of water for mixing in order to achieve the normal oxygen content of sea water. By measuring the COD in the discharged water this consumption is already catered for, while the BOD values refer to the oxygen consumption due to degradation of the actual organic content.

### **Other components**

In addition to the components that are dissolved in the produced water originating from the produced oil or gas itself and from the reservoir, a considerable amount of chemicals are used in the production processes and these compounds follow the produced water into the sea. The use and discharge of these chemicals are, however, regulated according to national and international regulations.

In order to assess the impact of chemicals, both toxicity tests and environmental conditions have to be considered. The CHARM (Chemical Hazard Assessment and Risk Management) model (Schobben *et al.*, 1996b) is a tool used by the oil industry and adopted by the regulatory authorities for hazard assessments, risk analysis, and risk management of the various chemicals in use. As with most standard laboratory toxicity tests, the organisms used are robust laboratory organisms which seldom reflect site-specific organisms.

### **Discharge regulations**

Of the various components/compounds discharged from oil and gas platforms, only a small fraction of the total amount is regulated by national and international authorities, i.e., chemicals used in the production processes and the oil content defined and analysed as dispersed oil. The majority, both in amount and number of components discharged from the offshore petroleum industry, are not regulated and the discharges are not under continuous control based on regular analysis of the effluents.

## **1.2 Distribution of the Discharged Produced Water**

The horizontal and vertical distribution of the discharged water depends on various factors such as the density of the produced water (temperature and salinity), the discharge point (above or below sea surface), whether the outlet is through a diffuser or a single point, varying hydrographic conditions of the water masses surrounding the platform, and water transport (tidal or permanent current direction), as well as actual wind conditions.

In the Norwegian sector, oil companies are required to file an impact assessment in conjunction with their presentation of plans for developing a new field. The oil companies are reluctant to produce information on what they actually discharge, such as tables with annual amounts of the various dissolved components. In these descriptions, the areas affected by the produced water from the actual field or platform are presented. Such presentations never consider a possible overlap of areas influenced by produced water from nearby fields. For some areas, such as the Tampen area (Anon, 1995), oil companies have joined forces to prepare regional impact assessments. But even regional assessments do not consider oil fields outside their defined area. Figure A7.1 presents an example from the Tampen study (Anon, 1995), where contributions from oil and gas fields nearby are not shown on the figures describing the area influenced by the produced water.

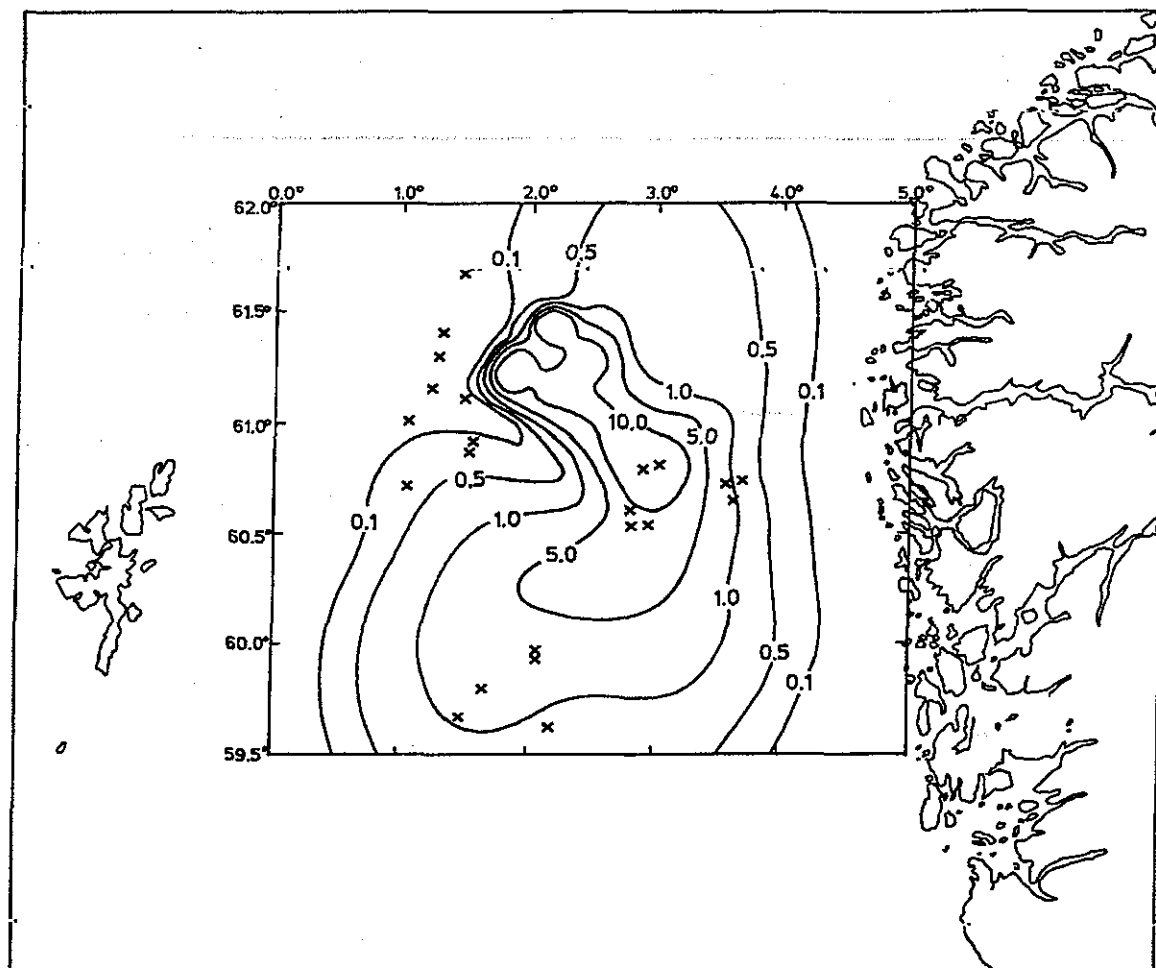
Figure A7.1 also demonstrates that the area where the model predicts a dilution of produced water to 1 % is quite large and will certainly connect with areas affected to the same degree from nearby fields. Stagg *et al.* (1996a) presented *in situ* measurements of oil hydrocarbons in the northern North Sea that demonstrate a distinct burden of oil components throughout the whole area.

Figure A7.2, taken from Anon. (1995), presents a model of the vertical distribution of produced water during summer and winter for an eight-week discharge period. The figure shows that the water column is affected to a depth of 100 m, with a peak from 25–50 m. Unfortunately, the Tampen assessment study (Anon., 1995) does not indicate the distance from the discharge point to the location at which the profiles of vertical distributions were modelled.

Reed *et al.* (1996) presented a model that includes a near-field release model, a far-field transport model, a biological exposure model, and a bioaccumulation and biomagnification model, meant to assess the potential for chronic effects from produced water. Reed *et al.* (1996) have used the model to simulate fish eggs and larvae as well as adult fish exposed to two individual components of produced water ( $C_7$  phenol and naphthalene) at various locations along the Norwegian continental shelf. The model indicates that the bioaccumulation and biomagnification of these two substances will be small.

Stagg *et al.* (1996a) and Reed *et al.* (1996) have modelled discharges of a certain duration, but they have not run the models for more than 50–60 days. The discharge of produced water is a continuous process throughout the year, and may therefore create a more complex (or steady state) picture than that shown by a relatively short run of the models.

**Figure A7.1.** Modelled horizontal distribution of concentration fields in percent of produced water at 0–25 m in a summer situation of discharges from oil platforms within the Tampen area (Anon., 1995). X indicates nearby oil and gas platforms which also discharge produced water, but which discharges are not included in the modelled distribution.



The produced water, however, contains a considerable amount of components which can create problems. This possibility is also one of the conclusions from Reed *et al.* (1996), who also noted that the possible effects of multiple components need to be addressed.

Given the water current patterns of the North Sea, contaminants will sooner or later be transported out of the North Sea. The majority of this water transport occurs in a northerly direction, ultimately ending up in the Norwegian coastal current. The transport northwards takes place through spawning grounds for some of the most important fish stocks of the Northeast Atlantic. However, taking into consideration the relatively huge water masses of the Norwegian coastal current (1–1.5 Sv), the potential for dilution is obvious. But a single organism such as a fish egg or larva is likely to be transported in the contaminated water mass during all of its critical developmental stages and may also pass

through multiple plumes of produced water introduced via northward-moving water masses.

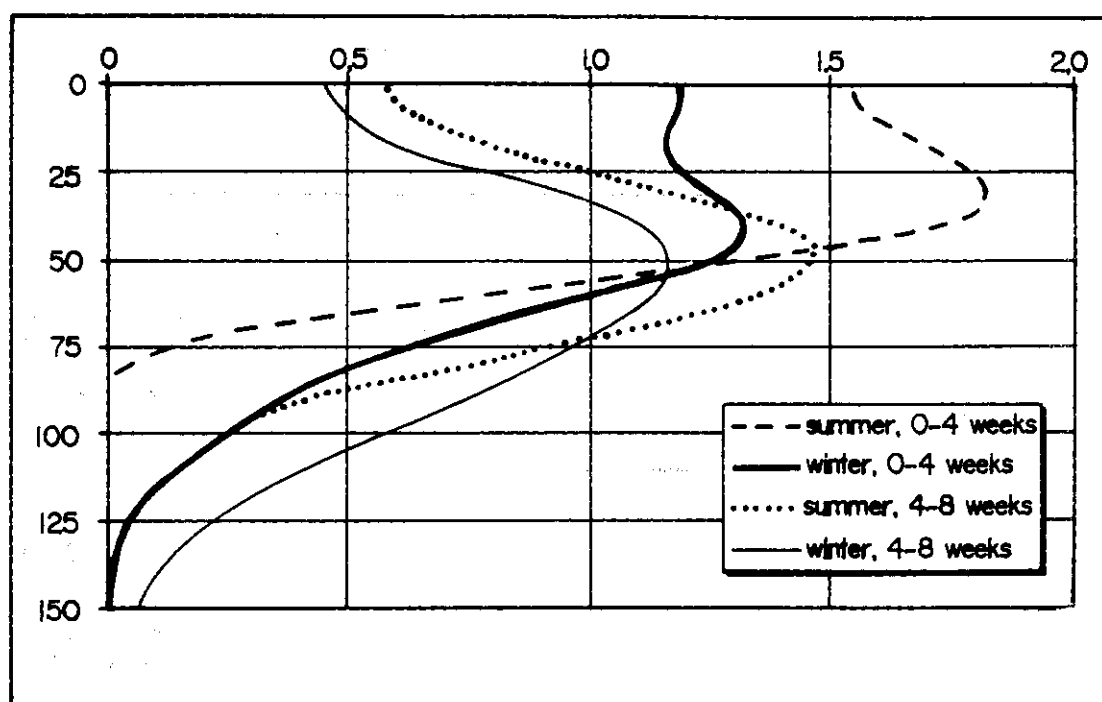
As can be seen in Figure A7.2, the plume of produced water occurs between depths of 25 m and 50 m and this is the part of the water column where most planktonic organisms are found. There is obviously a need to run models that include all discharge points and are run over a realistic duration according to common discharge practices. In addition, the models should be run as both a particle transport model and a salt model, i.e., a model that accounts for components completely dissolved in the water mass.

### 1.3 Effects of Produced Water on the Marine Environment

Produced water contains a wide variety of dissolved components, some organic and some inorganic. Each



**Figure A7.2.** Modelled vertical distributions of produced water discharged during eight-week periods during summer and winter months. The distributions are given as percent share per depth meter (Anon., 1995).



well has a different composition; both the composition and the quantity of the individual components change over the lifetime of the oil or gas well. At present, oil companies claim that the dilution effects will ensure that there will be no toxic effects of the produced water outside a radius of some few hundred meters from the platform.

The lack of evidence for acute toxicity in produced water plumes is one reason that both oil companies and the regulatory authorities are reluctant to establish monitoring strategies for produced water. This seems to be the case in Canada, the UK, and Norway and is most likely to be the rule in other countries with offshore oil and gas production. Within OSPAR, the discharge of produced water is not regulated.

Produced water is rarely tested as such. Reported tests are performed most often on single chemicals or groups of chemicals such as phenols, BTEX components, and PAHs or oil hydrocarbons. Booman and Føyn (1996), for example, tested the water-soluble fraction of crude oil, consisting mainly of BTEX and phenols, for effects on cod eggs and larvae, copepods and krill. They concluded that effects such as those determined in their experiment are not likely to occur in the field, given dilution factors of more than 1000.

Stagg *et al.* (1996a) observed significant gradients of EROD activity in gadoid and sandeel larvae along with

positive correlations using hydrocarbon fluorescence, indicating the presence of bioavailable aromatic hydrocarbons in the water column at sufficient levels to cause biological effects. The increase in *in situ* measured aromatic hydrocarbons correlated with the presence of oil fields in the northern North Sea (Stagg *et al.*, 1996a). Krause *et al.* (1992) tested produced water from an oil processing facility in California on gametes and early larval stages of the purple sea urchin, *Strongylocentrotus purpuratus*. They detected sublethal responses at produced water concentrations of 1 ppm (Krause *et al.*, 1992).

Stagg *et al.* (1996b) concluded that, despite the difficulties in collecting adequate water samples from the surroundings of platforms, the rapid dilution of the effluents is likely to reduce the concentrations to levels which are not acutely toxic to marine organisms. However, the possibility of chronic long-term effects cannot be excluded (Stagg *et al.*, 1996b).

There seems to be no observed evidence that produced water is acutely toxic to marine organisms outside a very limited radius from the platforms. The acute toxic effects that have been observed are most often referred to as effects of dissolved phenols and the light aromatics. However, sublethal or chronic effects over a far more extended area cannot be excluded. The fact that discharges of produced water are treated separately (i.e., each platform as being alone or the only one present) by

the regulatory authorities does not encourage monitoring on a regional, e.g., northern North Sea, basis.

The very large amount of carboxylic acids, mainly acetic acid, discharged with the produced water are not considered in the normal bio-tests performed on the produced water. Furthermore, data on the distribution and concentrations of carboxylic acids in the North Sea are non-existent. Carboxylic acids may act as a growth medium for bacteria and it is therefore necessary to address the question of whether such bacterial growth may change the abundance and composition of the marine bacteria communities in the North Sea. Such changes may ultimately lay the ground for changes in food webs that are important for the main living resources of the area.

The oxygen consumption owing to produced water may also have an impact over wide areas. This process will necessarily take some time and consequently will influence an area of some magnitude. Low oxygen values in the surface layers have been observed (Danielssen, pers. comm.) from time to time in recent years in the Skagerrak. To connect such an observation to the discharge of produced water is not obvious, but oxygen is seldom measured in connection with studies of produced water and there is, therefore, no data regarding such a possible influence.

## 2 ACKNOWLEDGEMENT

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## ANNEX 8

### STATISTICAL ANALYSIS OF FISH DISEASE PREVALENCE DATA FROM THE ICES ENVIRONMENTAL DATA CENTRE

#### 1 INTRODUCTION

Data on disease prevalence in wild marine fish stocks form a major component of the ICES Environmental Data Centre. ICES Member Countries conducting fish disease surveys as part of their national environmental monitoring programmes submit the data to ICES on a regular basis. The surveys are carried out according to standardized ICES methodologies established through the work of the ICES Working Group on Pathology and Diseases of Marine Organisms (WGPDMO) (ICES, 1989; Bucke *et al.*, 1996).

The ICES fish disease data comprise information from studies on the occurrence of externally visible diseases and macroscopic liver nodules/tumours in the common dab (*Limanda limanda*) and the European flounder (*Platichthys flesus*) from the North Sea and adjacent waters (including the Baltic Sea, Irish Sea, and the English Channel). In total, data on the length, sex, and health status of more than 420,000 individual specimens, partly dating back to 1981, have been submitted to ICES so far, as well as information on sampling characteristics (dates, locations, gear types, etc.). Therefore, the ICES fish disease data bank can be considered a unique set of data providing information on biological responses of marine organisms to, *inter alia*, environmental change.

In 1992, the Sub-Group on Statistical Analysis of Fish Disease Data in Marine Stocks (SGFDDS) was established. The Sub-Group existed until 1996 and was then replaced by a Study Group with the same name, which met once in 1997. The major tasks of the Sub-Group/Study Group were:

- to establish standardized procedures for the submission of fish disease data to the ICES Environmental Data Centre (the former ICES Environmental Databank) by ICES Member Countries; and
- to develop methods for and carry out a statistical analysis of the fish disease prevalence data submitted to the ICES Environmental Data Centre.

Disease prevalence is defined as the ratio between the number of diseased fish and the number of examined fish in a sample. This ratio and hence the prevalence is an empirical probability, which is thought to be a realization of an underlying stochastic process, whose probabilistic properties, including its relation to explaining quantities, must be estimated from observed data. Such an analysis needs specific statistical methods, which account for the particular distributional properties of prevalences.

At the 1997 meeting of WGPDMO, a progress report was presented on the activities of the SGFDDS, and its tasks were considered fulfilled. Through the work of the SGFDDS carried out in collaboration with the ICES Secretariat, the submission of fish disease data to ICES has been standardized via the implementation of the ICES Fish Disease Data Reporting Format (part of the ICES Environmental Data Reporting Formats), the ICES Fish Disease Data Entry Program, and procedures for validation of the data submitted. Statistical methods for analysing spatial and temporal trends in the disease prevalence have been elaborated and applied successfully using the data available prior to the 1997 meeting of the SGFDDS.

Recognizing that the data were still incomplete at that time, it was recommended that Member Countries submit additional fish disease data (historic and current data) to the ICES Environmental Data Centre and that a more comprehensive statistical analysis be carried out intersessionally prior to the 1998 WGPDMO meeting by selected experts using the complete data set. All data that were included in this comprehensive analysis had been collected following the ICES standard methodologies and were subjected to the same validation process before being entered into the ICES data bank.

The present report contains the main results of this statistical analysis. It highlights the methodologies applied and provides information on spatial and temporal trends with respect to the prevalence of the major diseases of dab and flounder in the North Sea and Baltic Sea.

The value of the ICES fish disease data as well as of the results of the analysis will increase in the future, since studies on externally visible diseases and liver nodules/tumours of dab are among the techniques designated for the biological effects component of the new OSPAR Joint Assessment and Monitoring Programme (JAMP) and since ICES serves as the data centre for the OSPAR Commission's monitoring data and will provide environmental data to be incorporated in the OSPAR Quality Status Report 2000. Furthermore, the status of the completed fish disease data bank and the establishment of methodologies for data submission, validation, and analysis will facilitate a more holistic approach for the future analysis of environmental data, combining fish disease data with other types of biological effects data, oceanographic data, and fisheries data held in the ICES data banks.

## DATA AVAILABLE FOR THE STATISTICAL ANALYSIS

The present analysis was carried out on data from the ICES fish disease data bank provided by the ICES Secretariat on 27 January 1998. At that time, data on a total of 424,998 specimens of fish were available, most of them on dab (*Limanda limanda*, n = 399,262) and some on flounder (*Platichthys flesus*, n = 25,736). The data had been submitted to ICES by various laboratories (see Table A8.1) and covered the time period from 1981 to 1997. The data were not uniformly distributed over time, neither over years nor over months or seasons within years. Table A8.2 shows the distribution of dab data over calendar years and months within the year. Table A8.3 contains the corresponding figures for flounder. Data on flounder refer in most cases to samples taken in the last quarter (October to December) of a year, while for dab usually more than one quarter of a year is covered by sampling.

**Table A8.1.** Number of dab (*Limanda limanda*) and flounder (*Platichthys flesus*) for which disease data were reported, by laboratory.

Reporting laboratory	Number of fish reported		
	Dab	Flounder	Total
ALUK	28,476	0	28,476
BFCG	274,951	7,465	282,416
DFHU	62,911	0	62,911
DGWN	10,631	0	10,631
DOUK	16,290	0	16,290
RIVO	6,003	18,271	24,274
Total	399,262	25,736	424,998

Reporting laboratories:

ALUK = Fisheries Research Services, the Marine Laboratory, Aberdeen, UK

BFCG = Federal Research Centre for Fishery, Institute of Fishery Ecology, Cuxhaven, Germany

DFHU = Danish Institute for Fisheries Research, Charlottenlund, Denmark

DGWN = National Institute for Coastal and Marine Management, Ecotoxicology Section, Middelburg, The Netherlands

DOUK = The Centre for Environment, Fisheries and Aquaculture Science, Fish Disease Laboratory, Weymouth, UK

RIVO = Netherlands Institute for Fishery Investigation, IJmuiden, The Netherlands

The geographical spread of sampling locations, as well as the frequency with which locations were visited, vary each year. Tables A8.4 and A8.5 summarize for dab and flounder, respectively, the geographical and temporal spread of sampling in terms of the number of visits per ICES statistical rectangle and year. Figure A8.1 provides an overview of rectangles for which disease reports for dab and flounder are available in the ICES

Environmental Data Centre. The number of rectangles which have been visited at least once is relatively high (123 for dab, 29 for flounder, five area designations were excluded because of implausible geographical details). However, not all rectangles were visited so frequently that the resulting time series of disease prevalence seems informative with respect to the investigation of temporal trends.

The data bank is organized in records, each of which refers in principle to an individual fish. Observations referring to fish specimens with the same host-specific attributes (species, gender, size in length class), the same location of observation (expressed as ICES rectangle), the same observation time (expressed by the sampling date), the same haul and the same reporting laboratory are summarized in one record, stating the number of specimens to which this record refers ('number examined').

The disease information in one record comprises, among others, the prevalence of lymphocystis, epidermal hyperplasia/papilloma, acute or healing skin ulcerations, skeletal deformities, and liver nodules/tumours. Most records also contain, depending on the reporting laboratory, information about additional diseases which were examined only occasionally. Prevalence information is expressed as the number of specimens examined (one variable per record) in relation to the number of fish infected. A missing value code for the number infected is used to express that the fish was not examined for the respective disease, while a zero indicates that no fish was affected. It is important that this distinction be maintained in future data reporting and administration within the ICES data bank.

Tables A8.6 and A8.7 contain an overview of the number of fish examined and affected; however, the number examined shown is the sum taken over all records referring to the respective ICES rectangle. As not all fish were inspected for all diseases, it is generally not possible to calculate the prevalence from these tables, though for the main diseases (lymphocystis, epidermal hyperplasia/papilloma, ulcerations), which were considered in nearly all samples, the error would be negligible.

### 3 AIM OF ANALYSIS

The aim of the present statistical analysis is the identification of location-specific temporal trends in disease prevalence and, as far as such exist, the comparison of such trends found at different geographical locations.

A temporal trend is generally understood to be a long-term change in a target quantity (here, the disease prevalence). Baggelaar *et al.* define a trend to be a '(semi-)permanent change in the location (mean or median) of a process over at least several years. It does

**Table A8.2.** Number of dab (*Limanda limanda*) examined, by year and month of sampling.

Year	January	February	March	April	May	June	July	September	October	December	Total
1981	4,809	0	0	0	7,401	0	0	0	13,380	0	25,590
1982	18,040	0	0	0	0	18,886	0	0	0	1,385	38,311
1983	5,579	0	0	0	15,716	0	0	0	0	10	21,305
1984	3,731	0	348	0	19,499	0	0	0	0	4,172	27,750
1985	6,408	0	0	0	12,100	0	0	0	0	144	18,652
1986	10,429	0	924	0	18,804	3,119	0	0	0	1,501	34,777
1987	8,183	0	1,321	0	4,888	7,607	0	0	0	119	22,118
1988	9,066	0	669	0	8,356	3,366	0	0	0	534	21,991
1989	4,145	0	0	959	9,379	2,043	0	83	0	0	16,609
1990	6,689	0	0	1,322	11,204	0	0	0	0	520	19,735
1991	9,408	837	0	1,145	6,584	8,770	1,383	0	387	673	29,187
1992	8,551	0	1,234	7,297	2,375	11,164	1,972	0	0	0	32,593
1993	5,088	879	1,057	0	4,911	8,161	1,943	0	0	1,237	23,276
1994	6,453	1,051	720	877	1,353	3,454	1,731	0	0	0	15,639
1995	3,595	5,399	0	0	0	7,084	1,665	0	0	729	18,472
1996	6,548	4,083	0	0	9,840	0	0	0	0	529	21,000
1997	0	4,322	0	0	5,889	2,046	0	0	0	0	12,257
Total	116,722	16,571	6,273	11,600	138,299	75,700	8,694	83	13,767	11,553	399,262

**Table A8.3.** Number of flounder (*Platichthys flesus*) examined, by year and month of sampling.

Year	Sep	Oct	Dec	Total
1983	1,336	0	121	1,457
1984	2,574	0	0	2,574
1985	1,089	0	97	1,186
1986	1,570	0	150	1,720
1987	1,188	0	615	1,803
1988	692	0	590	1,282
1989	1,265	0	0	1,265
1990	0	0	295	295
1991	0	1,517	1,365	2,882
1992	0	1,318	0	1,318
1993	0	1,305	1,568	2,873
1994	0	1,604	0	1,604
1995	0	1,402	1,472	2,874
1996	0	1,411	1,192	2,603
Total	9,714	8,557	7,465	25,736

not comprise changes related to seasonal cycles, or sudden and short-lived changes, caused by calamities' (ICES, 1997a). Though these authors discuss trend detection methods mainly for concentration or load measurements, not for prevalence, their definition can be

used here as well. The analysis methods must, however, be adapted for the present problem.

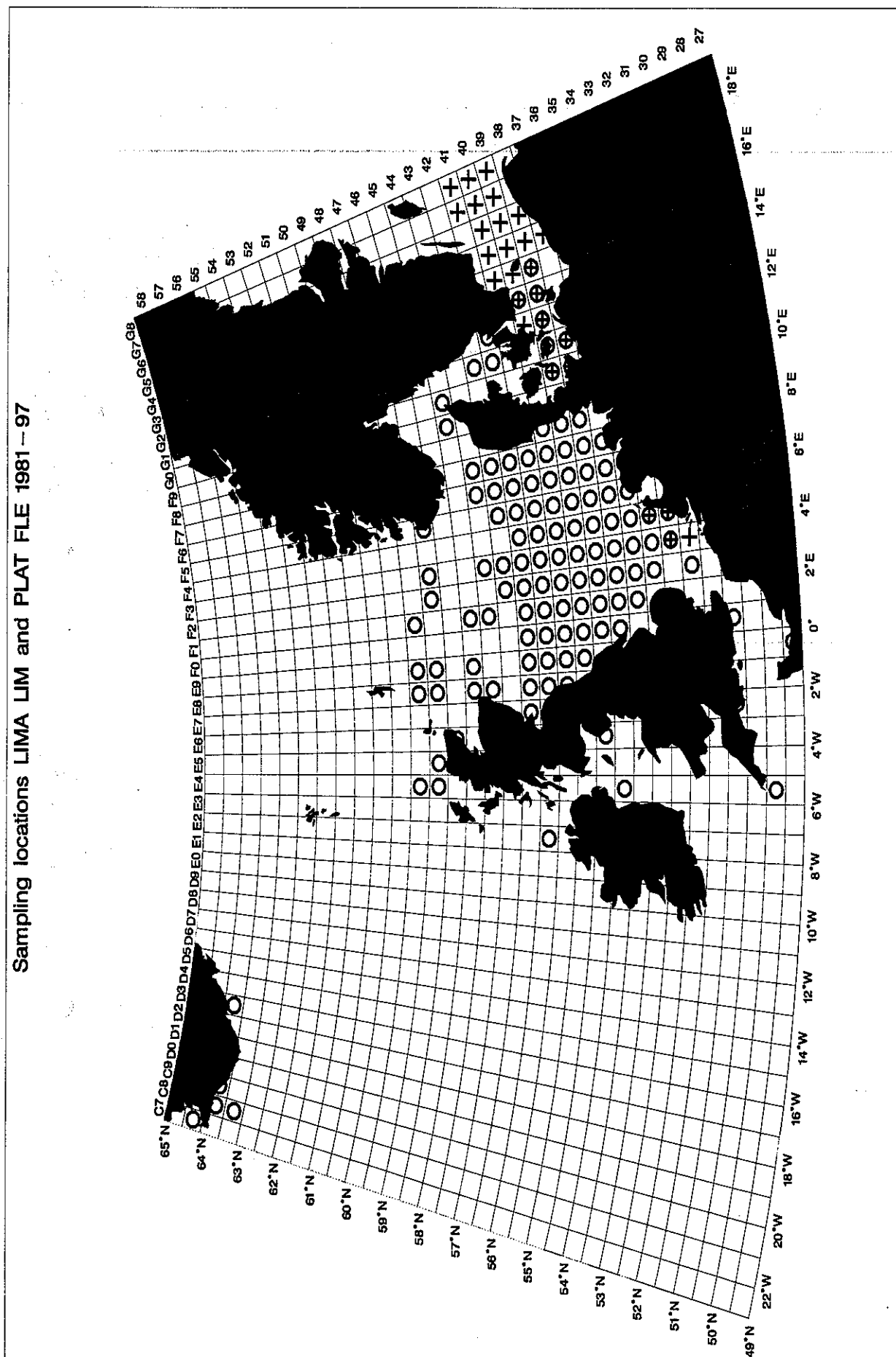
#### 4

#### DATA SUBSET FOR ANALYSIS

From the definition of a trend, it follows that only time series of a certain minimum length can be used for trend identification. In an earlier analysis, the Study Group on Statistical Analysis of Fish Disease Data in Marine Stocks (ICES, 1997b) decided, for dab data, to consider for analysis only those data series which contained on average at least one observation (sampling) within every two years over the reporting period (1981–1996) present in the 1997 database. Using a corresponding criterion for the selection of usable time series from the 1998 database leads to the requirement that a time series within one ICES rectangle should contain at least nine samplings to be used for statistical analysis. Table A8.4 shows that this criterion is fulfilled for the prevalence series in 42 ICES rectangles. However, the series for two of these rectangles (30F0 and 37F8) extend over only six years, which was considered as too short in view of the requirements of the trend detection technique (see below), so that these rectangles were excluded from further analysis. Furthermore, locations with fewer than four samples from the 20–24 cm size group were also excluded from the analysis in order to ensure a sufficiently accurate trend estimation. Combining all criteria resulted in 32 ICES rectangles for trend analysis. In Tables A8.4 and A8.5, samplings were counted on a

**Figure A8.1.** Map showing the ICES statistical rectangles from which fish samples were taken for disease investigation. Disease reports for dab (*Limanda limanda*) are indicated by circles; disease reports for flounder (*Platichthys flesus*) are denoted by plus signs.

Sampling locations LIMA LIM and PLAT FLE 1981-97



**Table A8.4.** Number of reports per ICES rectangle for dab (*Limanda limanda*). Hauls are combined; reports for different days are counted individually.

No.	ICES rect.	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	Total
1	37F7	12	5	3	8	4	5	3	6	4	4	3	3	2	2	2	2	1	69
2	38F7	6	3	2	7	5	3	4	3	1	1	1	1						37
3	40F7				2	2	3	3	3	4	4	2	4	2	2	2	2	1	36
4	41E7									4	2	4	5	5	3	4	4	3	34
5	38F2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	1	33
6	36F1			1	3	2	2	3	2			2	1	1	1	4	3	2	27
7	39F7	1	2	2	5	1	3	4	3	1	1	1	1						25
8	37F2	2	3			1	1		1	1	2	2	2	2	2	2	2	1	24
9	37F6	4	4	2	4	2	2	2	1		1	1	1						24
10	39F6	1	3	1	6	2	3	2	2	1	1	1	1						24
11	35F3		1		1		3	1	1	2	2	1	1		2	3	3	2	23
12	37F0	1	1	1	2	3		3	2			2		1	1	2	1	1	21
13	34F3	1	2				2	2	1	1	1	1	1	1	2	2	2	1	20
14	36F4				1		4	2	2	1	1	1	1	1		3	3		20
15	37F3					1	1	2	2	3	2	1	2	1	2	1	2		20
16	41F7				2	2	2	4	2	2	2	2	1	1					20
17	39E9				1		2		1	2	2	2	2	2		2	2	1	19
18	41E8							1	1	2	2	2	2	2	1	2	2	1	18
19	37F4						2	2	2	3	1	1	2	1	1		2		17
20	37F5	4	2	2	1	2	1					1	1			1	2		17
21	38G0		3	1		2	3	1	1		1	1		2		1	1		17
22	39F3		2	2	3	2	2	2	1			1	1				1		17
23	44E8									2	2	2	2	2	2	2	2	1	17
24	37F1	2	4	2	2	2	2	1	1										16
25	41F6		1	1	1	2	2	3	1	1	1	1	1	1					16
26	38F6	2	2	2	2	1	2	1			1								13
27	40F6			2	2	2	2	2	1					1	1				13
28	44F9				1	2	1	1	2		2	1	1	1					12
29	36F2		2	2	2	1	1	2	1										11
30	39F0		1	2	1	1			1	1	1	1				1	1		11
31	39F5	1	4	1	1	1	2	1											11
32	37F8	4	1	1	2	1	1												10
33	38F0	2		1		1	2	2	1			1							10
34	38F1	3	3				1					1	1					1	10
35	41G1				1	1	1	1	1	1	1	1	1	1					10
36	42F3									1	1	2	2	1	1		1	1	10
37	42G1				1	1	1	1	1	1	1	1	1	1					10
38	30F0												1	1	2	1	2	2	9
39	40F4		1		1	1	2	1			1	1	1						9
40	41G2					1	1	1	1	1	1	1	1	1					9
41	44G0				1	1	1	1	1		1	1	1	1					9
42	47E8									2	1		1	1	1	1	1	1	9
43	38F5	1	1	2	1		2				1								8
44	40F5				3	2	3												8
45	42F7					1	1	1	1	1	1	1	1						8
46	47F1								1	1	2		1	2			1		8
47	33F3						2	1	1	1	1	1							7
48	36F0		1	2	1		1					1						1	7
49	37G1		1					1	1			1		1		1	1		7
50	40E8				2	1	1	2	1										7
51	36F5			1			1	1				1	1		1				6
52	39F4		1			1	1				1	1	1						6
53	40E9				2	1		2	1										6
54	40F1		1	3	1		1												6
55	40F3		1	1	1		2	1											6
56	41F0			2			1	2		1									6
57	38F3		2		1	1		1											5
58	40F0		1	2	2														5
59	41F2			2	1			1					1						5
60	41F5		1	2	1	1													5
61	45E6												1	1	1	1	1		5
62	30E6														1	1	1	1	4
63	35F2		1	1	1		1												4
64	35F4						1	1								1		1	4
65	36F3		1	1					1						1				4

Table A8.4. Continued.

No.	ICES rect.	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	Total
66	36F7		1	1	1		1												4
67	37G0		1			1	1	1											4
68	40E2														1	2	1		4
69	41F1						1	2			1								4
70	41F3			2	1		1												4
71	41F4			1			2	1											4
72	44E9								1	1					1	1			4
73	30E7														1		1	1	3
74	34E3														1	1	1		3
75	37E6														1	1	1		3
76	38F8	1	1		1														3
77	38G3											1		1		1			3
78	39G3		1											1		1			3
79	40F2				1					1	1								3
80	42F5		1						1							1			3
81	46E4														1	1	1		3
82	46E5														1	1	1		3
83	46F3									1	1			1					3
84	33F4																1	1	2
85	34F2		1										1						2
86	34F4																1	1	2
87	36E4															1	1		2
88	36F6	1						1											2
89	38E9				1		1												2
90	38F4						2												2
91	38G2							1	1										2
92	38G4						1							1					2
93	39F8				1			1											2
94	41E9						1	1											2
95	42F2							1					1						2
96	46E8									1	1								2
97	56D3												1	1					2
98	27E9																	1	1
99	28E4															1			1
100	30F2														1				1
101	32F2												1						1
102	35F1			1															1
103	35F5														1				1
104	38G1							1											1
105	39E8											1							1
106	39F1																	1	1
107	39F2		1																1
108	42F6		1																1
109	43E8													1					1
110	43F1																1		1
111	43F3							1											1
112	43F6							1											1
113	43F7							1											1
114	44F1															1			1
115	46E9										1								1
116	46F2									1									1
117	46F5															1			1
118	47E4										1								1
119	47E9									1									1
120	55C8												1						1
121	56C8												1						1
122	56C9												1						1
123	57C7												1						1
Total		51	72	57	89	61	97	88	61	54	58	57	63	48	41	57	57	28	1,039



**Table A8.5.** Number of reports per ICES rectangle for flounder (*Platichthys flesus*). Hauls are combined; reports for different days are counted individually.

No.	ICES rect.	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	Total
1	38G4			3		1	2	2	2		1	2		2		1	1		17
2	38G5			2		1	1	3	2		1	1		2		1	1		15
3	32F3			1	1	1	1	1	1	1		1	1	1	1	1	1		13
4	38G3			1			1	2	1		1	2		1		2	2		13
5	33F4					1	1	1	1	1		1	1	1	1	1	1		11
6	39G6						2	1	2		1	1		2		1	1		11
7	38G0			1		1	3		1					1		1	1		9
8	39G8			1				1	2		1	1		1		1	1		9
9	39G3							1	1		1	1		1		1	1		7
10	31F3											1	1	1	1	1	1		6
11	34F4											1	1	1	1	1	1		6
12	35F6											1	1	1	1	1	1		6
13	37G1			1				1				1		1		1	1		6
14	40G7						1	2	3										6
15	39G7						1	2	2										5
16	40G6			1			1		1							1	1		5
17	38G2			1		1		1	1										4
18	40G5							1				1		1		1			4
19	37G5			1				1				1							3
20	38G6						1	1	1										3
21	40G8							1	1					1					3
22	33F3			1	1														2
23	39G4			1		1													2
24	41G7						1	1											2
25	41G8											1					1		2
26	37G0						1												1
27	39G2			1															1
28	39G5													1					1
29	40G4													1					1
Total		0	0	16	2	7	17	23	22	2	6	17	5	20	5	16	16	0	174

daily basis: all hauls reported in one rectangle for one day were combined, but hauls from different days were counted as different samples. In previous studies, samples were combined within a whole month, which led to a smaller number of samples per ICES rectangle and year.

Carrying over the selection criteria for dab data to the flounder data would have left the time series from only five ICES rectangles for further analysis (32F3, 33F4, 38G3, 38G4, 38G5). In order to provide at least a partial impression about temporal developments in flounder disease prevalence, the original criterion was relaxed by

performing an exploratory analysis also for time series with six to eight sampling dates. This resulted in the performance of an analysis for a total of twelve rectangles.

## 5 METHOD OF STATISTICAL ANALYSIS

For all diseases under study, the target quantity is a prevalence. This makes it natural to use a logistic model for the analysis of relationships between disease prevalence and (potentially) explaining variables. The standard form of a (linear) logistic model is

**Table A8.6.** Number of dab (*Limanda limanda*) examined and numbers found with specific diseases. Only rectangles with at least nine reports are shown. Note: While all fish were examined for the presence of lymphocystis, epidermal hyperplasia/papilloma, and skin ulcers, only some were examined for the presence of skeletal deformities and liver nodules.

Area	Number of specimens examined	Lymphocystis	Epidermal hyperplasia/papilloma	Skin ulcers	Skeletal deformities	Liver nodules
30F0	3,773	122	63	110	44	16
34F3	6,744	332	203	28	13	25
35F3	7,164	607	302	166	9	15
3651	13,265	1,474	584	354	68	34
36F2	3,484	274	62	92	6	.
36F4	8,603	372	201	50	6	49
37F0	7,337	677	218	196	5	132
37F1	7,701	985	190	148	9	.
37F2	6,586	822	283	391	18	.
37F3	8,074	1,340	298	78	8	1
37F4	6,810	860	256	54	12	4
37F5	6,368	357	83	81	3	2
37F6	9,247	505	126	101	7	.
37F7	33,097	3,207	1,625	485	36	.
37F8	3,700	91	14	48	2	.
38F0	2,939	391	67	18	1	4
38F1	7,292	862	118	173	6	58
38F2	19,072	2,356	706	1,643	37	.
38F6	3,769	256	72	20	1	.
38F7	17,525	1,332	563	164	4	.
38G0	5,188	304	10	65	21	.
39E9	8,822	2,096	257	63	28	.
39F0	3,848	548	46	39	3	.
39F3	8,150	1,360	227	335	10	5
39F5	3,245	323	31	42	3	.
39F6	7,868	921	174	59	8	.
39F7	8,493	778	288	89	7	.
40F4	1,609	375	58	34	1	.
40F6	4,420	605	104	53	8	.
40F7	18,366	2,124	791	214	24	.
41E7	24,991	3,816	644	928	178	15
41E8	4,812	1,413	244	351	114	.
41F6	8,010	846	174	69	3	.
41F7	6,757	576	200	62	1	.
41G1	4,224	329	64	11	.	.
41G2	3,725	264	65	3	.	.
42F3	6,878	1,786	118	212	15	.
42G1	3,395	206	60	7	.	.
44E8	6,643	1,579	316	152	172	.
44F9	5,817	57	11	45	.	.
44G0	2,196	10	3	33	.	.
47E8	3,987	349	62	45	0	2
Total	333,994	37,887	9,981	7,311	891	362

**Table A8.7.** Number of flounder (*Platichthys flesus*) examined and numbers found with specific diseases. Only rectangles with at least nine reports are shown. Note: While all fish were examined for the presence of lymphocystis, epidermal hyperplasia/papilloma, and skin ulcers, only some were examined for the presence of skeletal deformities and liver nodules.

Area	Number of specimens examined	Lymphocystis	Epidermal hyperplasia/papilloma	Skin ulcers	Skeletal deformities	Liver nodules
31F3	1,774	14	.	13	.	7
32F3	5,300	177	.	60	.	11
33F4	5,339	760	.	137	.	55
34F4	1,800	64	.	134	.	6
35F6	1,320	22	.	14	.	1
37G1	369	76	.	5	.	.
38G0	50	13	.	0	.	.
38G3	2,512	571	.	58	.	.
38G4	1,421	261	.	25	.	.
38G5	1,192	249	.	31	.	.
39G3	508	185	.	11	.	.
38G6	408	103	.	16	.	.
39G8	250	30	.	14	.	.
40G7	34	1	.	0	.	.
<b>Total</b>	<b>22,277</b>	<b>2,526</b>	.	<b>518</b>	.	<b>80</b>

$$\text{Prob}(\text{disease present}) = f(a_0 + a_1x_1 + a_2x_2 + \dots + a_qx_q) \quad (1)$$

where  $x_1, x_2, \dots, x_q$  denote the explaining variables,  $a_0, a_1, \dots, a_q$  are unknown coefficients that must be estimated from empirical data, and  $f(\cdot)$  is the logistic function

$$f(z) = 1 / (1 + \exp(-z)). \quad (2)$$

A logistic model (1) is a member of the class of Generalized Linear Models (GLMs; cf. McCullagh and Nelder, 1989).

The explaining variables (the  $x_i$ ) in the present problem are:

- gender, technically a factor to be coded by a dummy variable for female/male;
- size (length), a factor with groupings  $\leq 14$  cm, 15–19 cm, 20–24 cm,  $\geq 25$  cm for dab, and with groupings 15–19 cm, 20–24 cm, 25–29 cm,  $\geq 30$  cm for flounder (these size group definitions allow the use of early data which were reported only for size groups);
- quarter of the year, a factor expressing seasonal variation and biological phase, with levels: Q1 (for Jan, Feb, Mar), Q2 (for Apr, May, Jun), Q3 (for Jul, Aug, Sep), Q4 (for Oct, Nov, Dec);
- calendar time (to describe the temporal trend).

While gender, size, and quarter can easily be modelled within the framework of the linear logistic model (1), this is not true for the temporal trend, the quantity of central interest. There is no reason to assume that a trend is linear over a long time or could be represented by simple elementary functions such as low-order polynomials, logarithms, or exponentials of time. Instead, earlier analyses suggest that trends of a more general non-linear shape are present (ICES, 1997b). Consequently, a generalization of (1) is needed to allow the incorporation (and estimation) of general non-linear trend functions. Hence, an extended model of the form

$$\text{Prob}(\text{disease present at time } t) = f(a_0 + a_1x_1 + a_2x_2 + \dots + a_qx_q + s_1(t) + s_2(t) + \dots + s_p(t)) \quad (3)$$

was used, which contains smooth functions  $s_j(t)$  to describe temporal trends. These functions are estimated as a whole, i.e., also their shape is estimated by a fitting procedure. The only restriction is that the shape is to exhibit a certain degree of smoothness. This is not a real restriction here, as a certain smoothness is by definition a required property of a trend. The degree of smoothness can be expressed in terms of 'equivalent degrees of freedom' (df), which must be specified by the investigator as a prerequisite to use the Generalized Additive Model (GAM, cf. Hastie and Tibshirani, 1990) in (3). High df values indicate high variability or, equivalently, non-smooth curves.

For the dab data, the optimal smoothing parameter was determined for the longest time series (reported for rectangle 37F7) by cross validation within the GAM model. The optimal number of degrees of freedom there was found to be  $df = 7$ . This value was then used uniformly for all dab time series. For the flounder data, a similar calculation on the basis of the data from rectangle 32F3 (13 samples in 13 years, values in quarters Q3 and Q4) recommended a smoothing with 9 df, while a cross validation for the short-term series for rectangle 31F3 (6 samples in six years, only values in Q4) led to a value of  $df = 2$ . Though this difference seems enormous at first sight, it does not for the present problem go along with a severe change in the prevalence that is predicted by (3). The reason seems to be that a major component of the temporal variation is the seasonal fluctuation, which, with a very smooth trend  $s_i(t)$ , is accounted for by large coefficients of the seasonal components, while, in the case of a non-smooth trend (high df), the seasonal changes are incorporated in the trend, not in the model term for season. This shift of weights within the mathematical model could generate various interpretations of the model components, but leads to nearly identical estimates for  $\text{Prob}(\text{disease present})$ . As a result of these considerations, a smoothing with 2 df was generally used for all flounder data in order to separate trend and seasonal fluctuations clearly also for the shorter data series.

It should be noted that the choice of a smoothing parameter in general is still a field of ongoing research. The procedure applied here follows general recommendations as given by Hastie and Tibshirani (1990). It seems to cover the actual practical needs and is at the same time driven by some economic considerations, as the choice of an optimal smoothing parameter is a computer- and time-consuming exercise, for which, at present, execution times must be measured in hours. But before establishing a general protocol for the application of a GAM approach to fish disease data, one might want to review the strategy for choosing a smoothing parameter, particularly if future data collection requires assessing longer time series.

For each combination of gender and size class, an individual trend  $s_i(t)$  was estimated. This is equivalent to introducing a threefold interaction term for  $\text{gender} \times \text{size} \times \text{time}$ . All of these trends were estimated with the same degree of smoothness.

All model fitting was done individually for each ICES rectangle. Confidence intervals for the predicted prevalence (the trends) were determined by a parametric bootstrap simulation.

All calculations within the GAM approach were carried out using the function `gam()` of the S-Plus software (MathSoft, 1997).

## 6

## RESULTS

The primary result of fitting a Generalized Additive Model is a set of fitted coefficients  $a_0, a_1, \dots, a_q$  and the fitted trends  $s_1(t), s_2(t), \dots, s_p(t)$ . Approximate confidence intervals for all of these terms are also supplied directly by the software, however, not for the estimated prevalence, for which confidence intervals were estimated by simulation. The aim of the analysis, namely the identification of temporal trends, requires the presentation of estimated prevalence trends together with confidence bounds. These trends are shown for each of the ICES rectangles with sufficiently long time series, where the operational definition of 'sufficiently long' was described above. Only the trends for female specimens with a length of 20–24 cm for dab and a length of 25–29 cm for flounder are shown. These subgroups were selected because they are covered with the highest frequency. Members of these length groups are relatively homogeneous in length, due to the grouping, but can also be assumed to be relatively homogeneous with respect to age, as individuals in these length classes are neither extremely young nor likely to be extremely old, but instead somewhere in the middle of their growth phase.

Trends are shown in Figures A8.2 to A8.6 with full detail, i.e., including seasonal variation (for the locations of the ICES rectangles covered, see Figures A8.7 and A8.8). Solid black lines in the figures refer to trends in fish disease prevalence and dotted lines to the 90 % confidence bounds for the time series at the observed points. These can lie in different quarters of a year and can thus be influenced by seasonal variation. Frequently a data series contains varying quarters of years, which causes the (erroneous) impression of erratic fluctuation over the years. To remove this problem and to facilitate the interpretation of the temporal trends, an additional season-adjusted trend estimation is added to the trend at observed time points. For dab, this additional trend shows the prevalence estimate valid for 15 February of each year, for the time period that is covered by the observed series. For flounder, the estimate valid for 15 November is shown. It refers as before to females in the length class 20–24 cm (dab) or 25–29 cm (flounder) and is shown as a dashed line. As this trend is calculated only for dates in one quarter of a year (dab: Q1; flounder: Q4), no seasonal effects can show up, so this curve is easier to use for an interpretation.

The season-adjusted trend can also be used for a first assessment of recent temporal developments. Also for this trend, a 90 % confidence bound was estimated by simulation. By comparing the prevalence estimates for 1992 with those for 1997 (or 1996, if a series does not extend to 1997), a rough statement about the general temporal development of the trend (constant, downward, upward) can be derived. The comparison can be performed by checking the sign of the prevalence difference and by checking whether the two confidence intervals (for 1992 and 1997) overlap. Non-overlapping

**Figure A8.2.** Estimated temporal trend (solid line) with 90 % confidence intervals (dotted lines) of lymphocystis prevalence in common dab (*Limanda limanda*) for females in the 20–24 cm length class, by statistical rectangle. This trend is based on the empirical sampling dates. The season-adjusted estimated prevalence is shown as a dashed line.

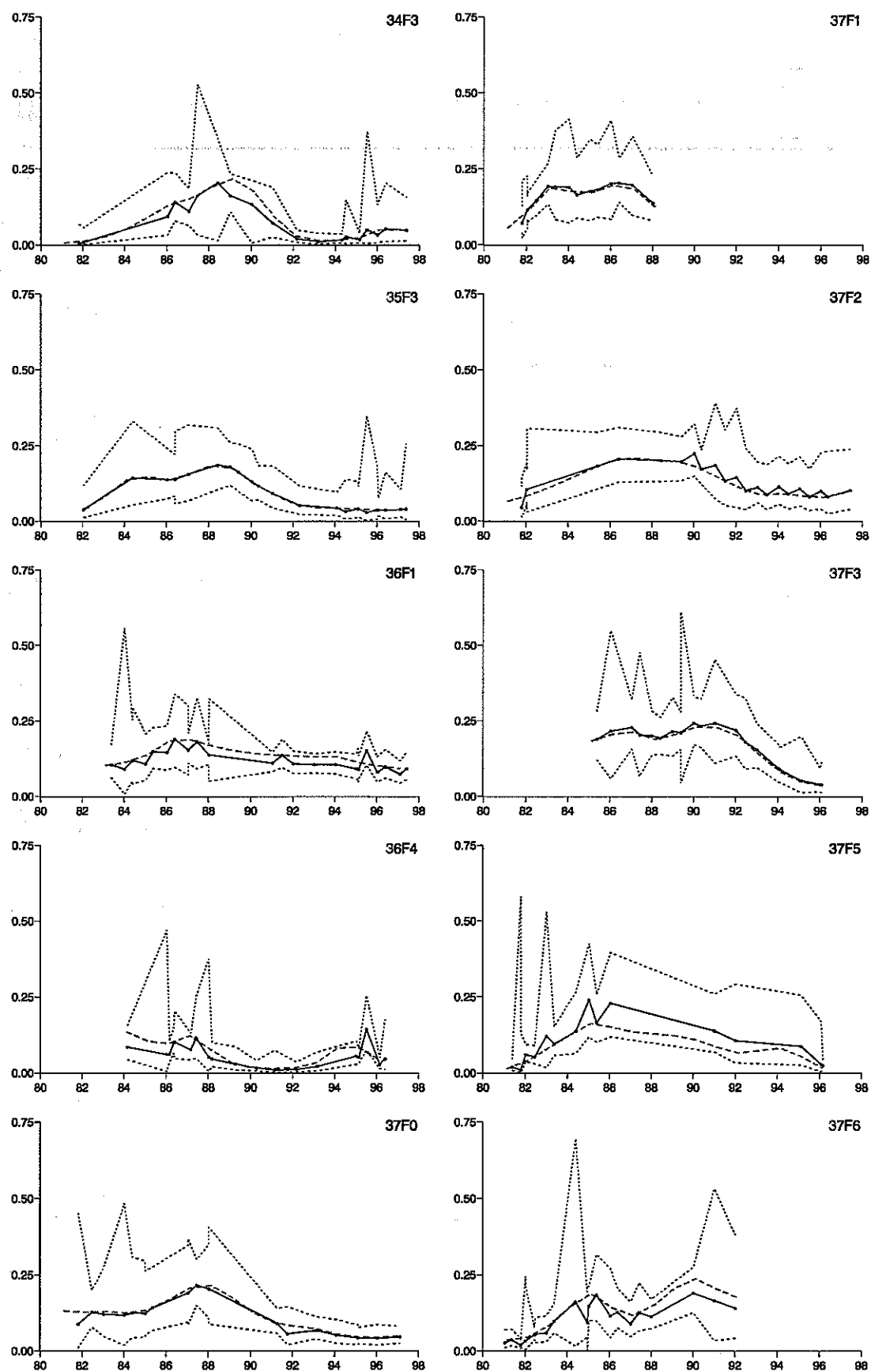


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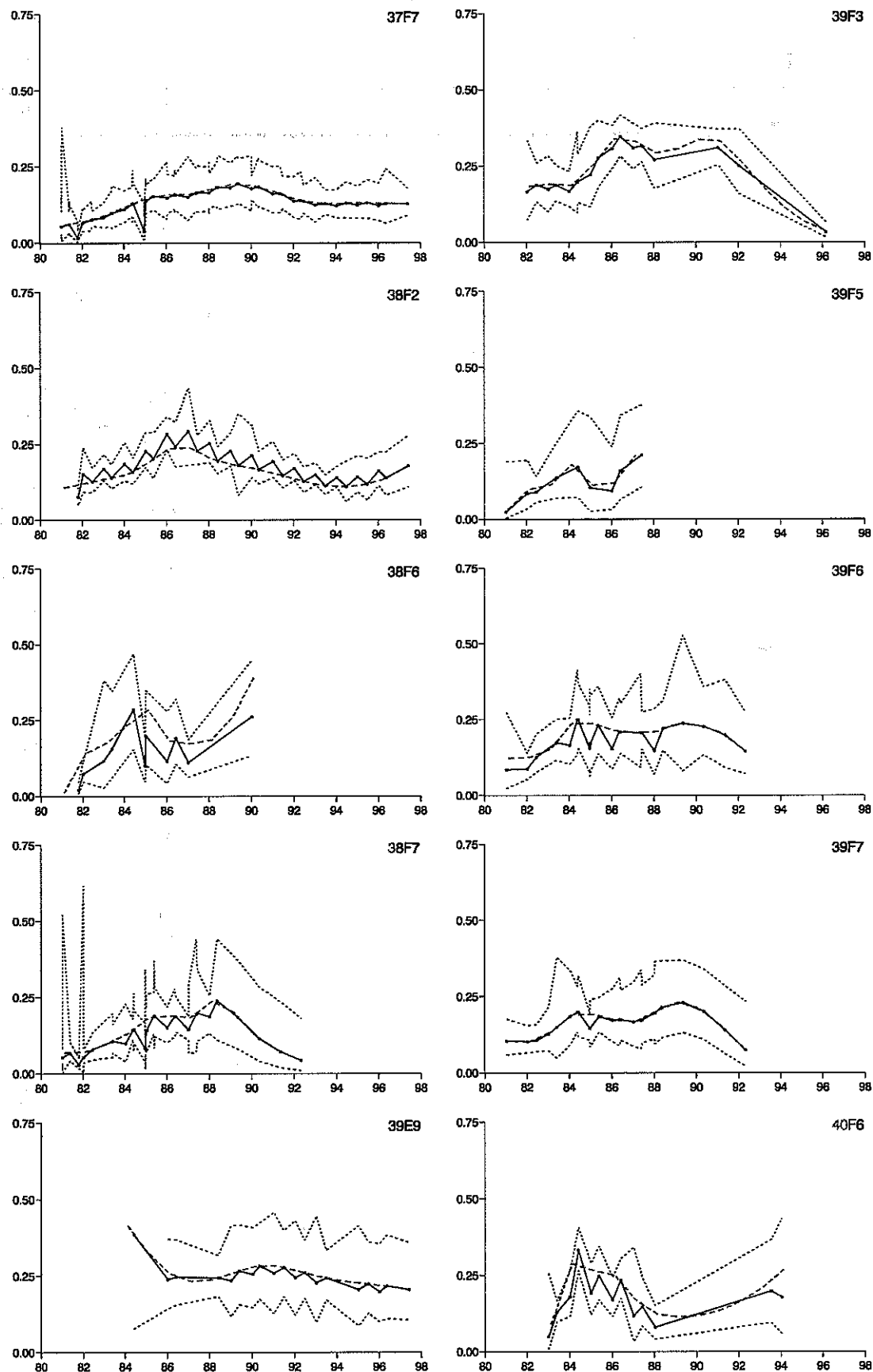


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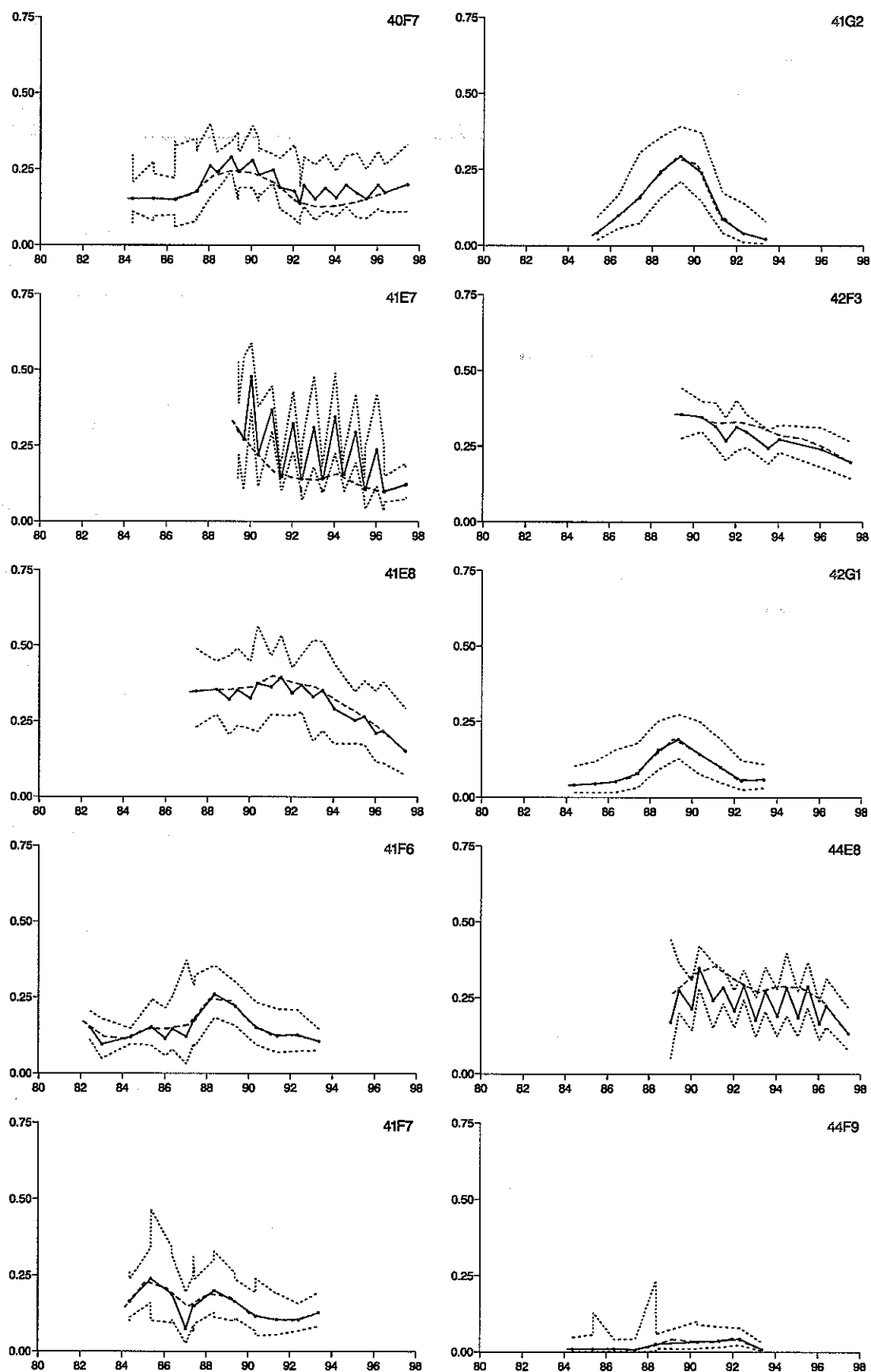
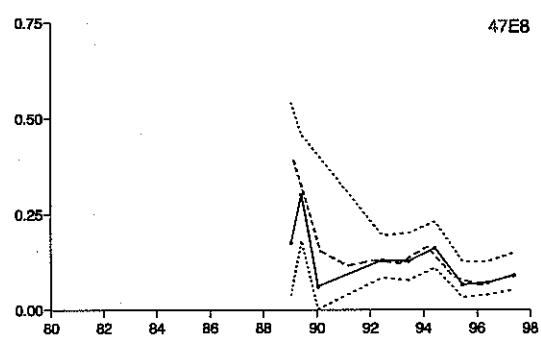
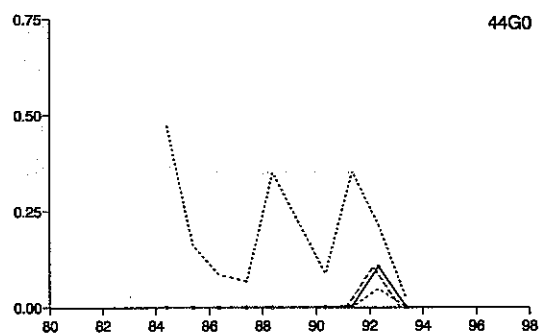


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**Figure A8.3.** Estimated temporal trend (solid line) with 90 % confidence intervals (dotted lines) of epidermal hyperplasia/papilloma prevalence in common dab (*Limanda limanda*) for females in the 20–24 cm length class, by statistical rectangle. This trend is based on the empirical sampling dates. The season-adjusted estimated prevalence is shown as a dashed line.

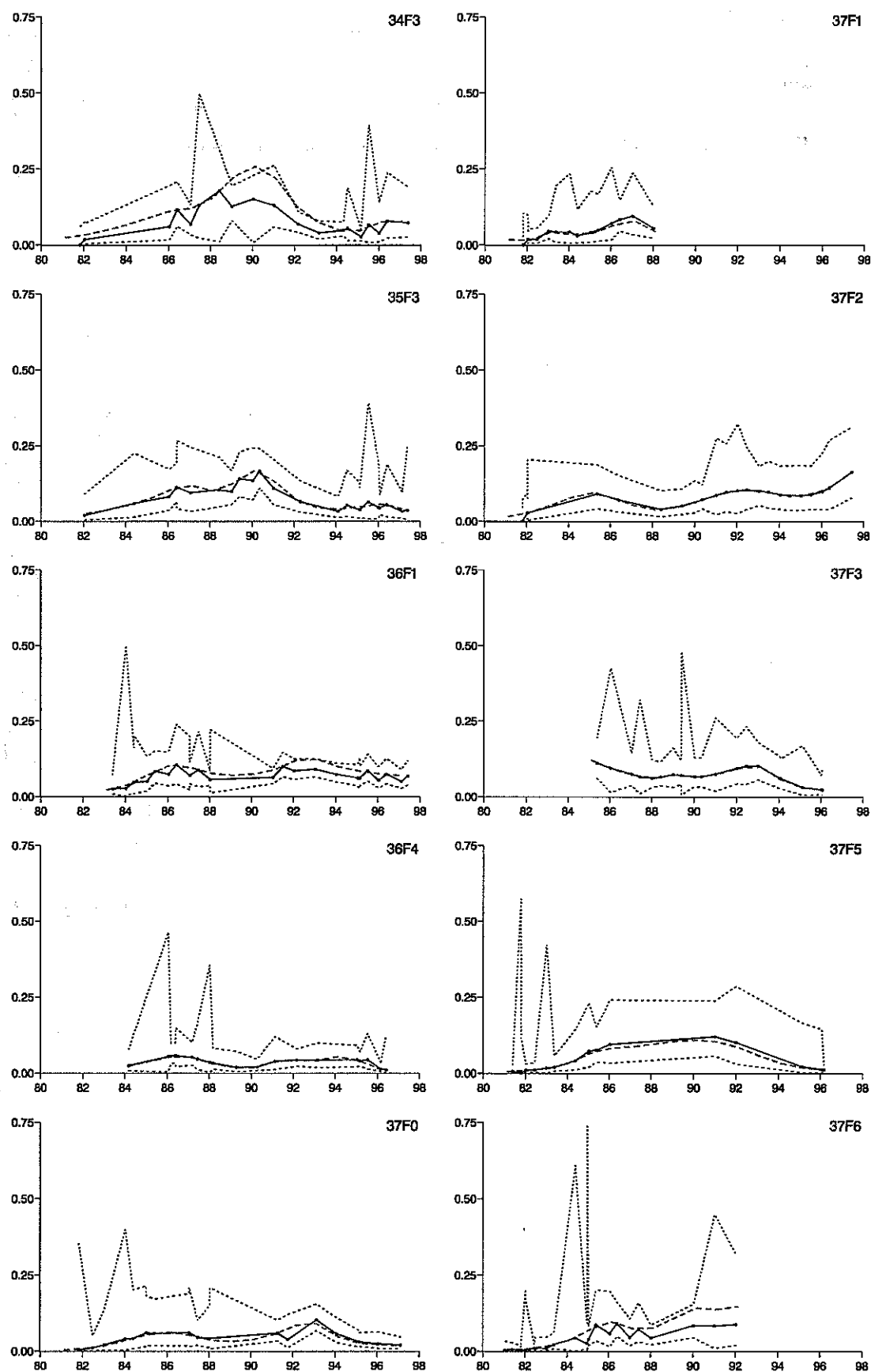


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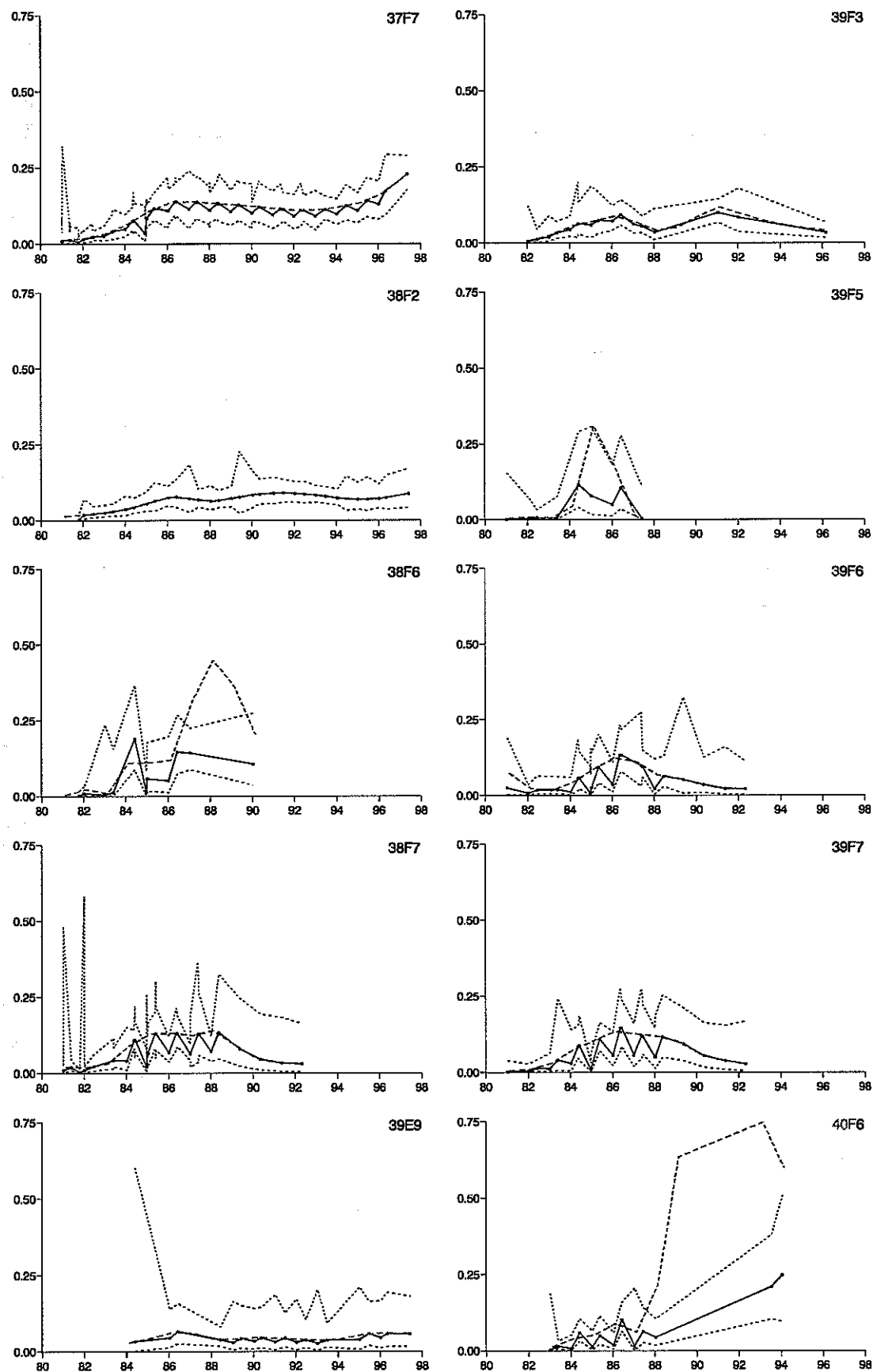


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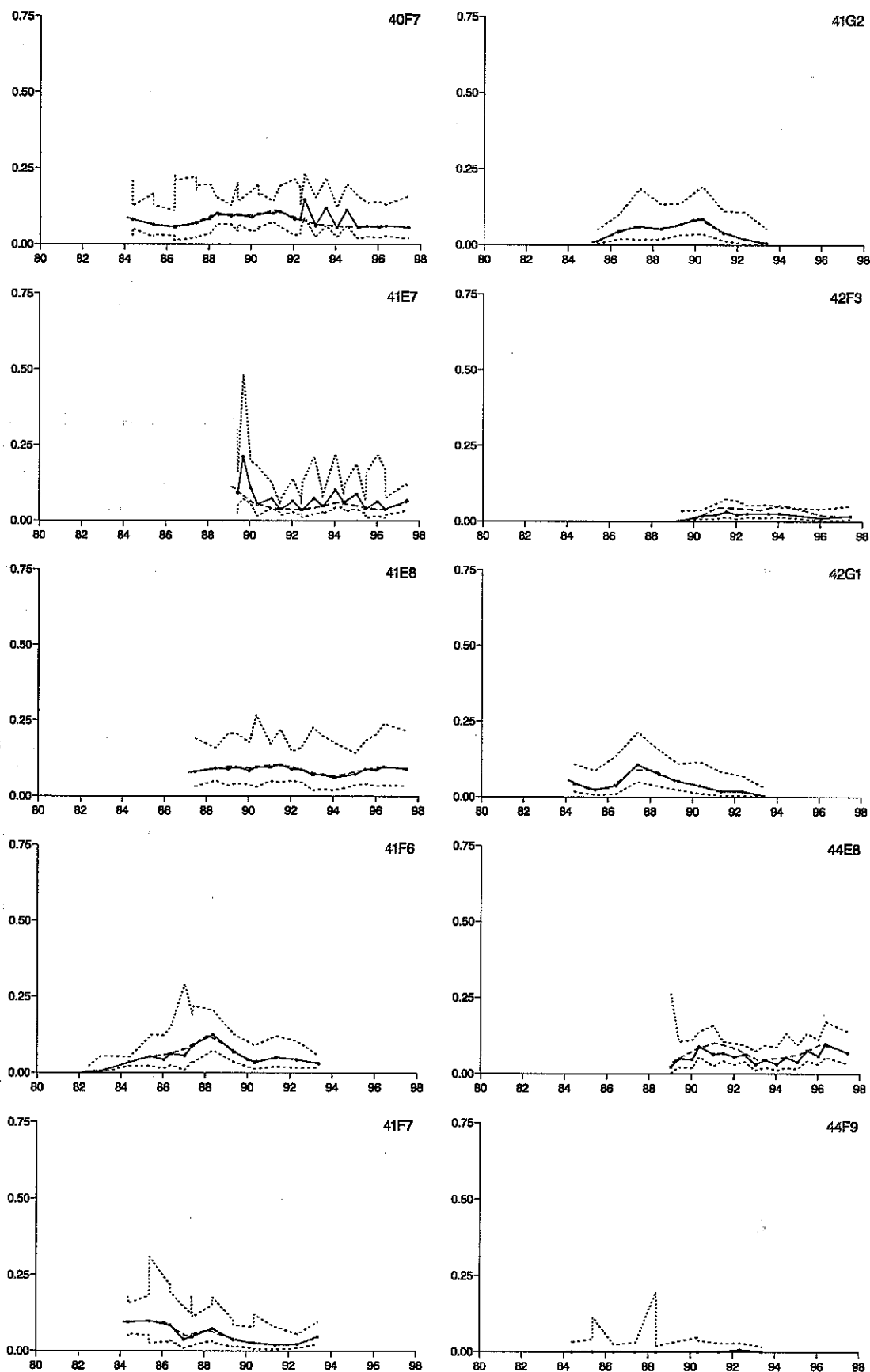
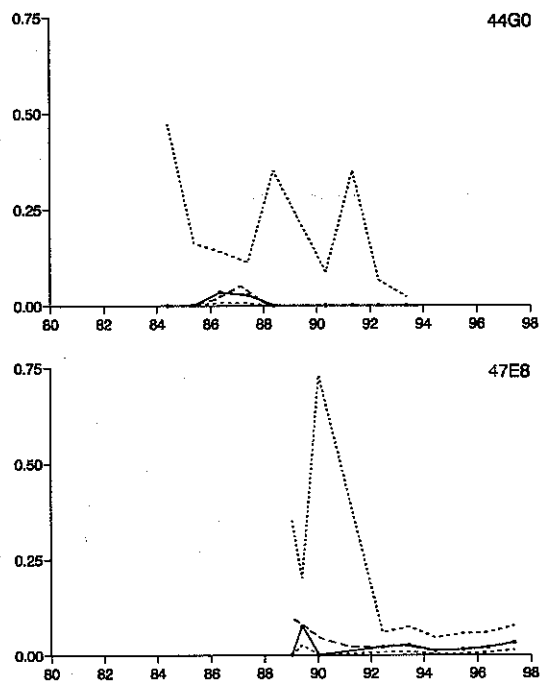


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**Figure A8.4.** Estimated temporal trend (solid line) with 90 % confidence intervals (dotted lines) of acute/healing skin ulcer prevalence in common dab (*Limanda limanda*) for females in the 20–24 cm length class, by statistical rectangle. This trend is based on the empirical sampling dates. The season-adjusted estimated prevalence is shown as a dashed line.

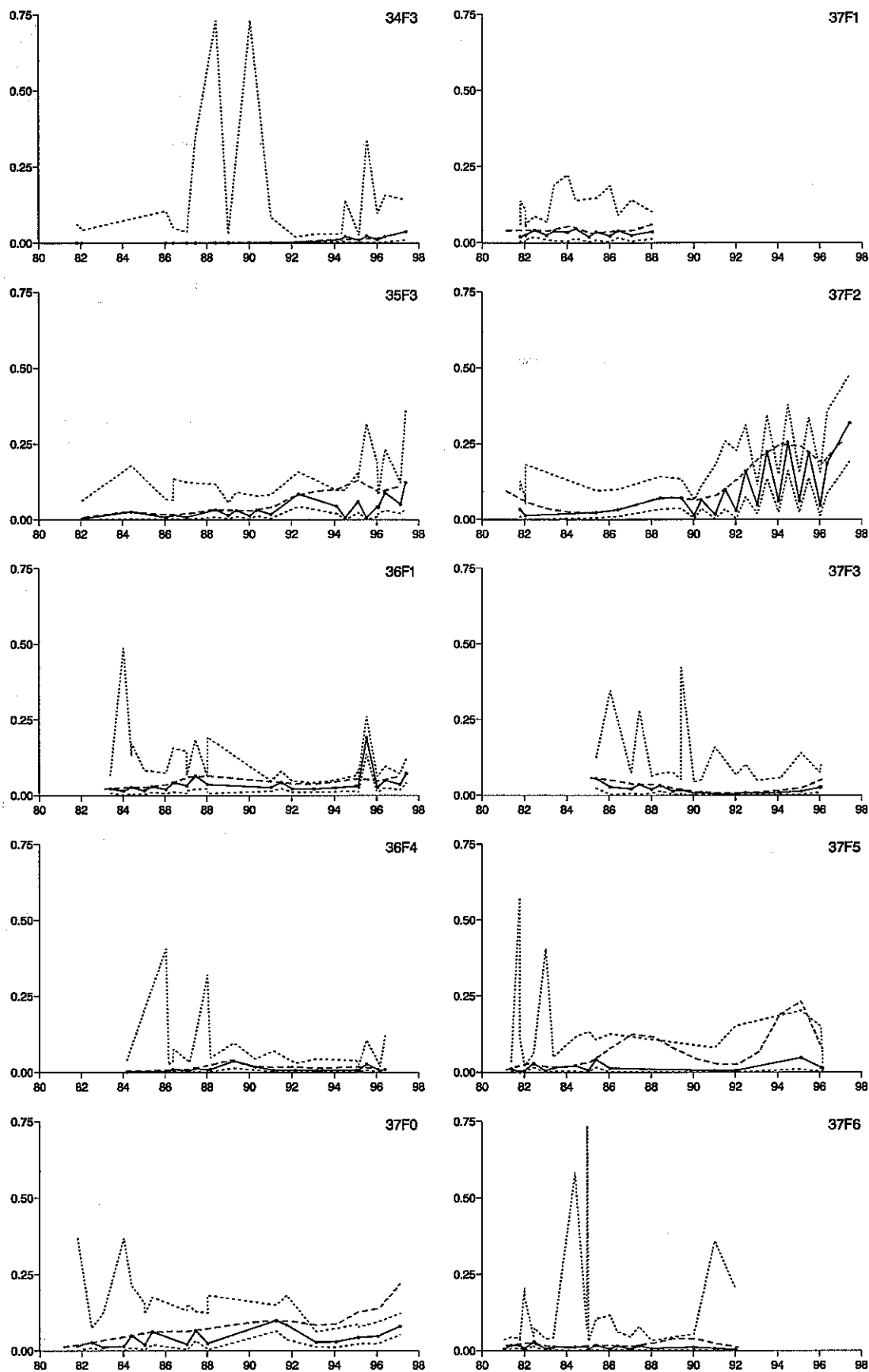


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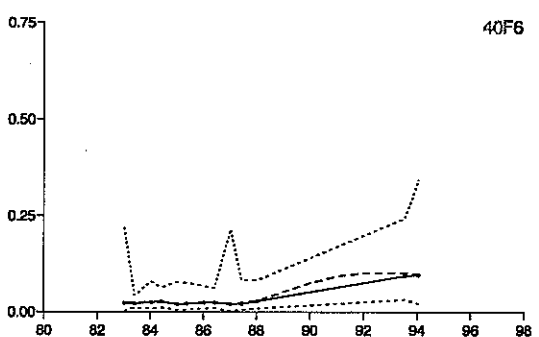
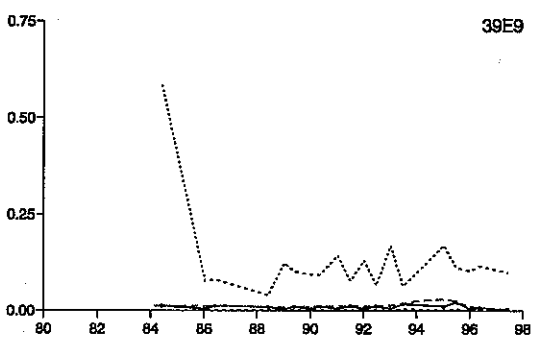
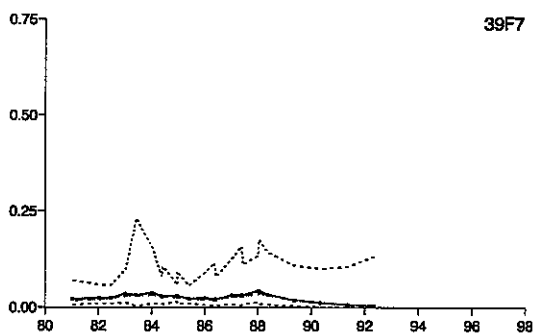
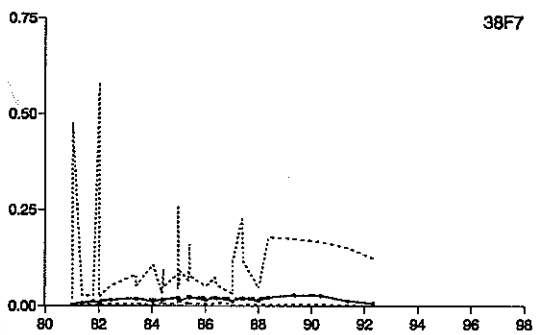
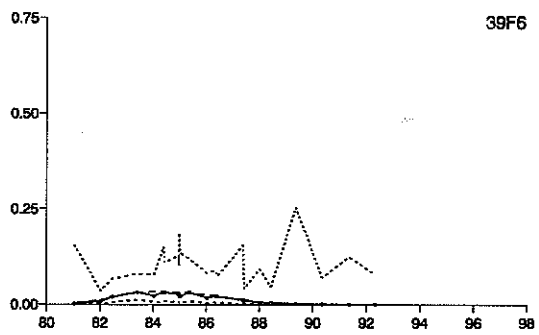
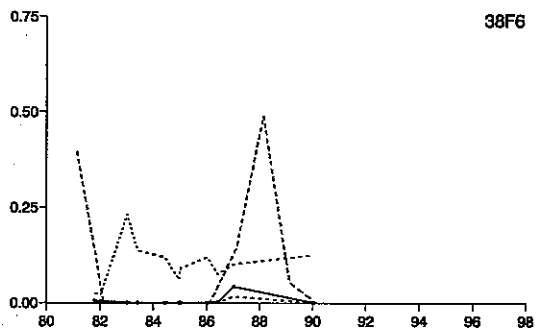
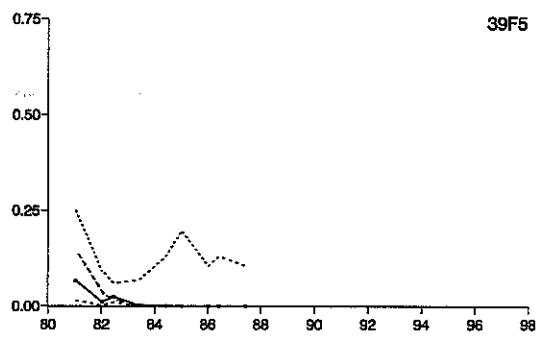
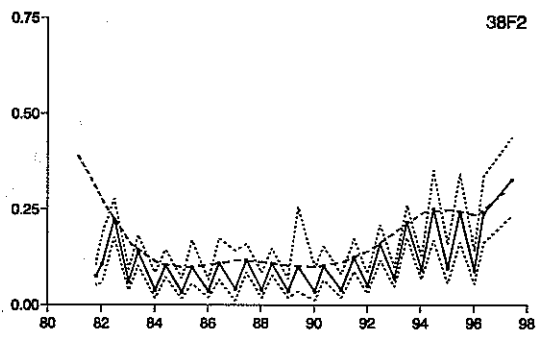
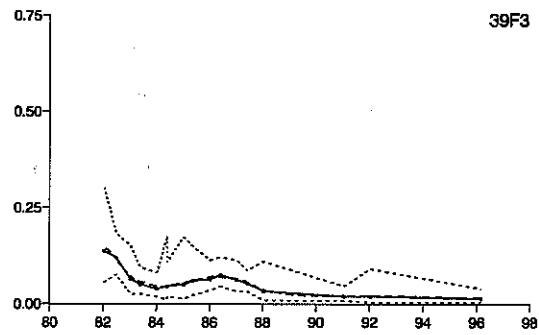
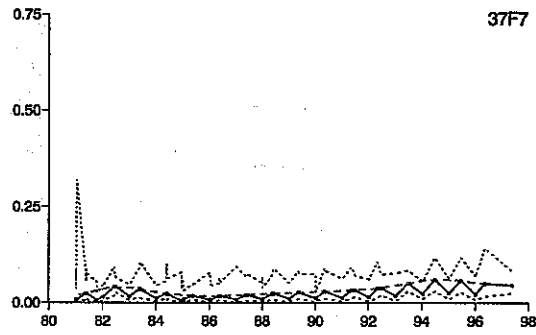


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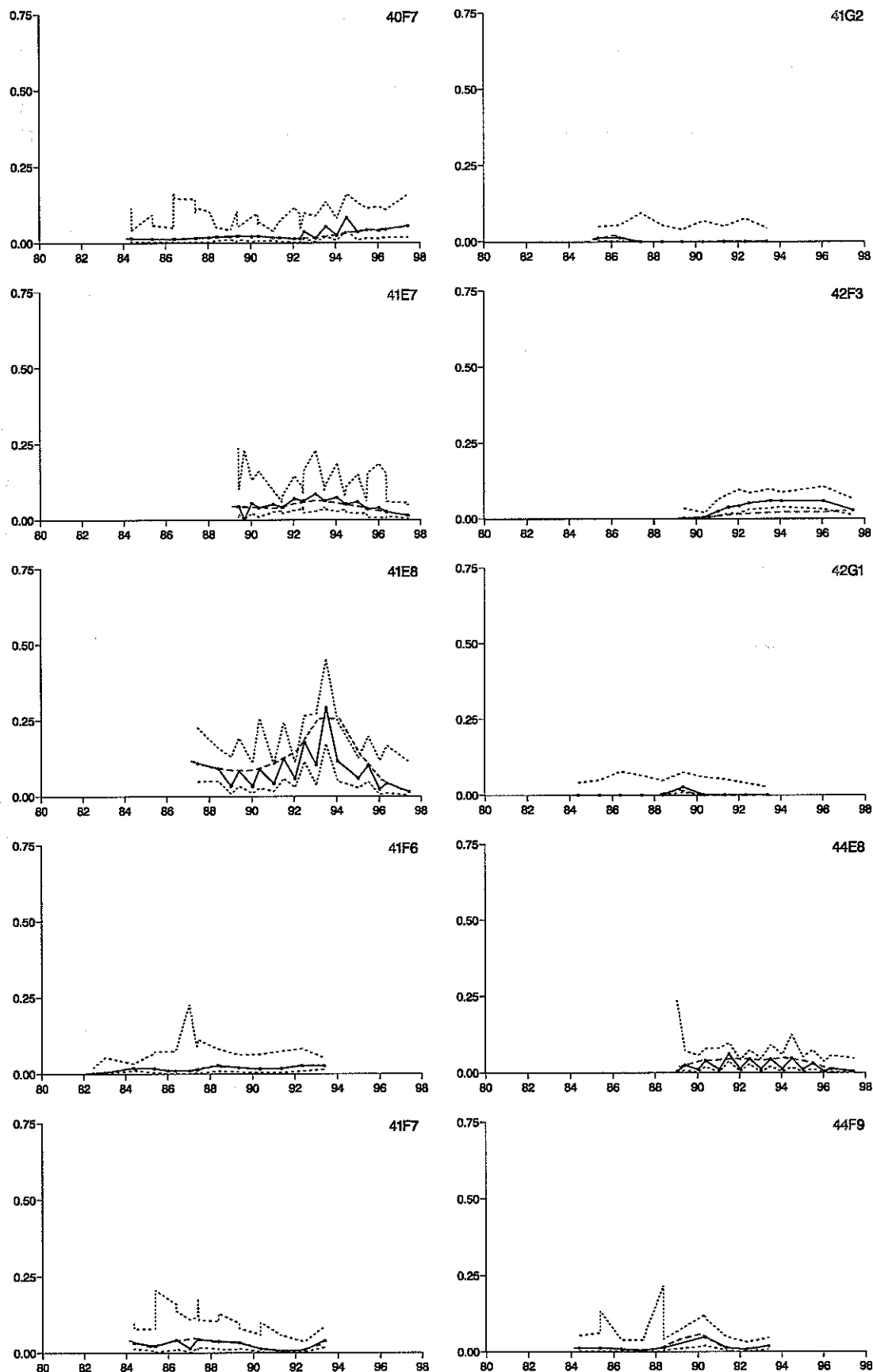
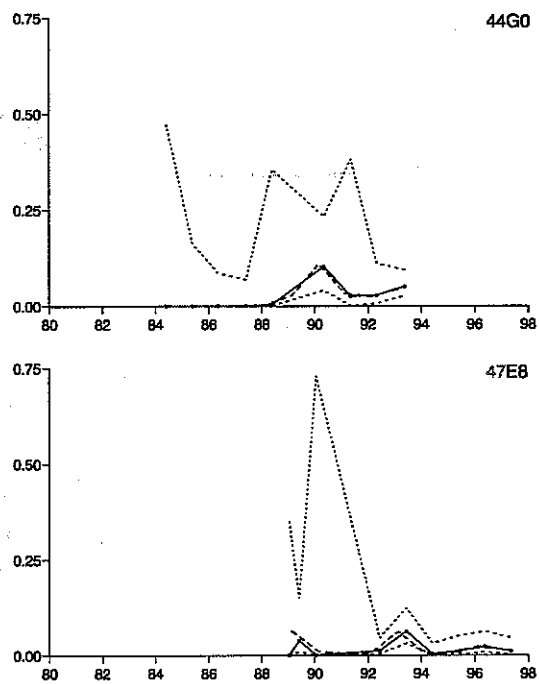


Figure A8.4. Continued.





**Figure A8.5.** Estimated temporal trend (solid line) with 90 % confidence intervals (dotted lines) of lymphocystis prevalence in flounder (*Platichthys flesus*) for females in the 25–29 cm length class, by statistical rectangle. This trend is based on the empirical sampling dates. The season-adjusted estimated prevalence is shown as a dashed line.

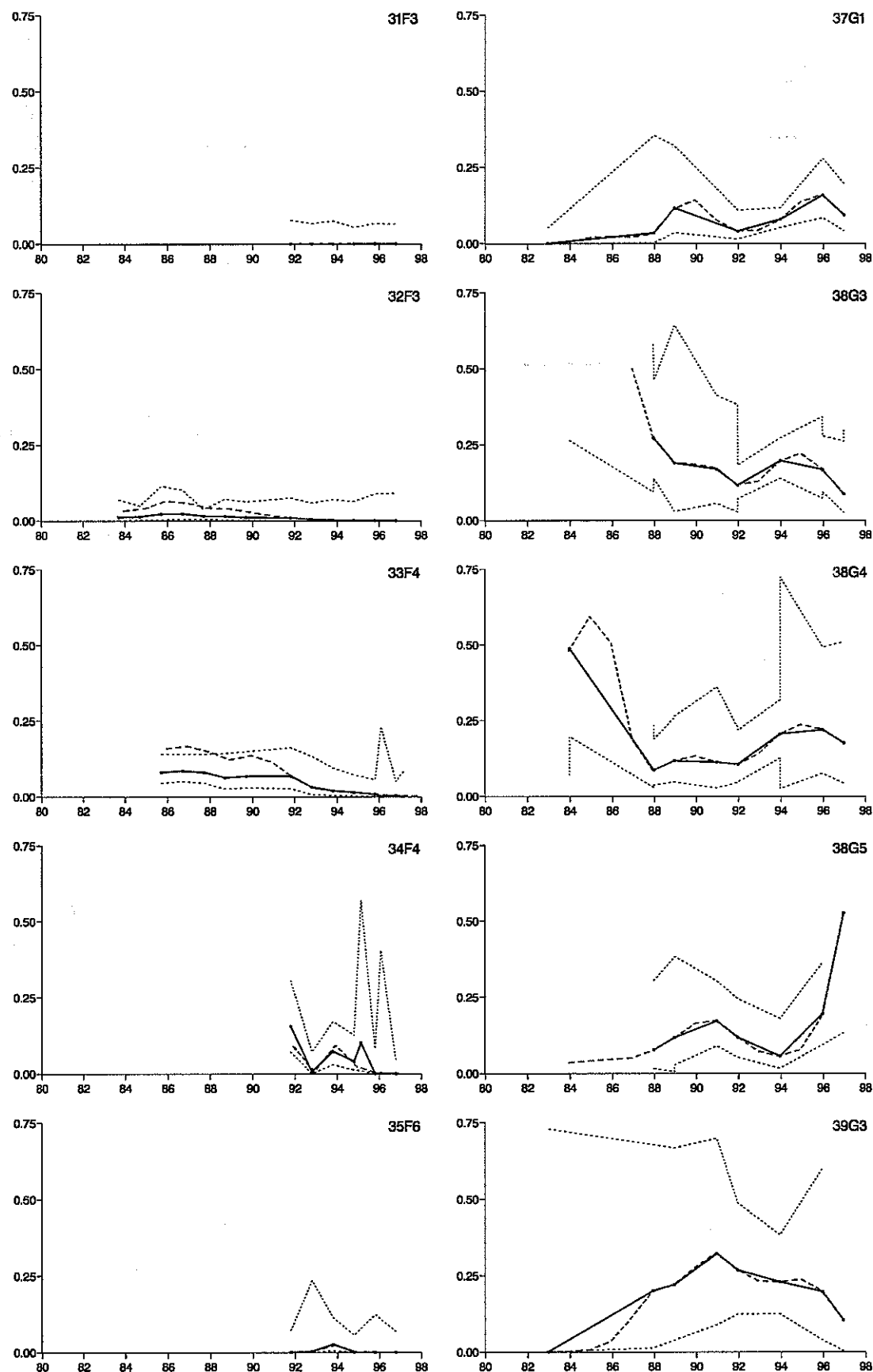
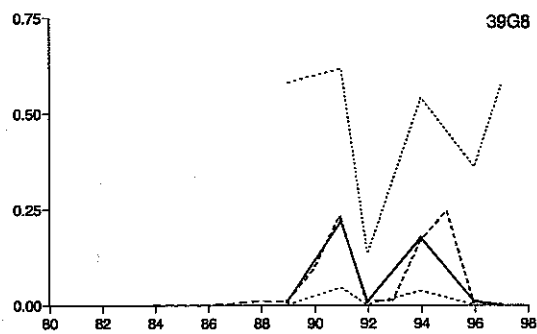
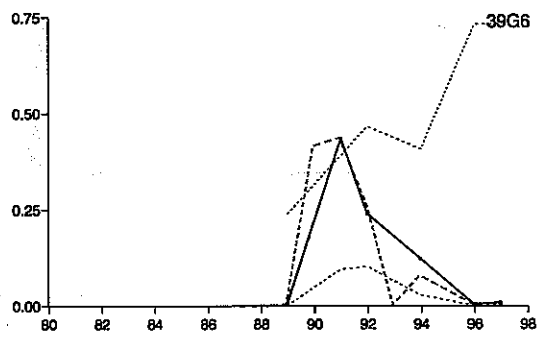


Figure A8.5. Continued.



**Figure A8.6.** Estimated temporal trend (solid line) with 90 % confidence intervals (dotted lines) of skin ulcer prevalence in flounder (*Platichthys flesus*) for females in the 25–29 cm length class, by statistical rectangle. This trend is based on the empirical sampling dates. The season-adjusted estimated prevalence is shown as a dashed line.

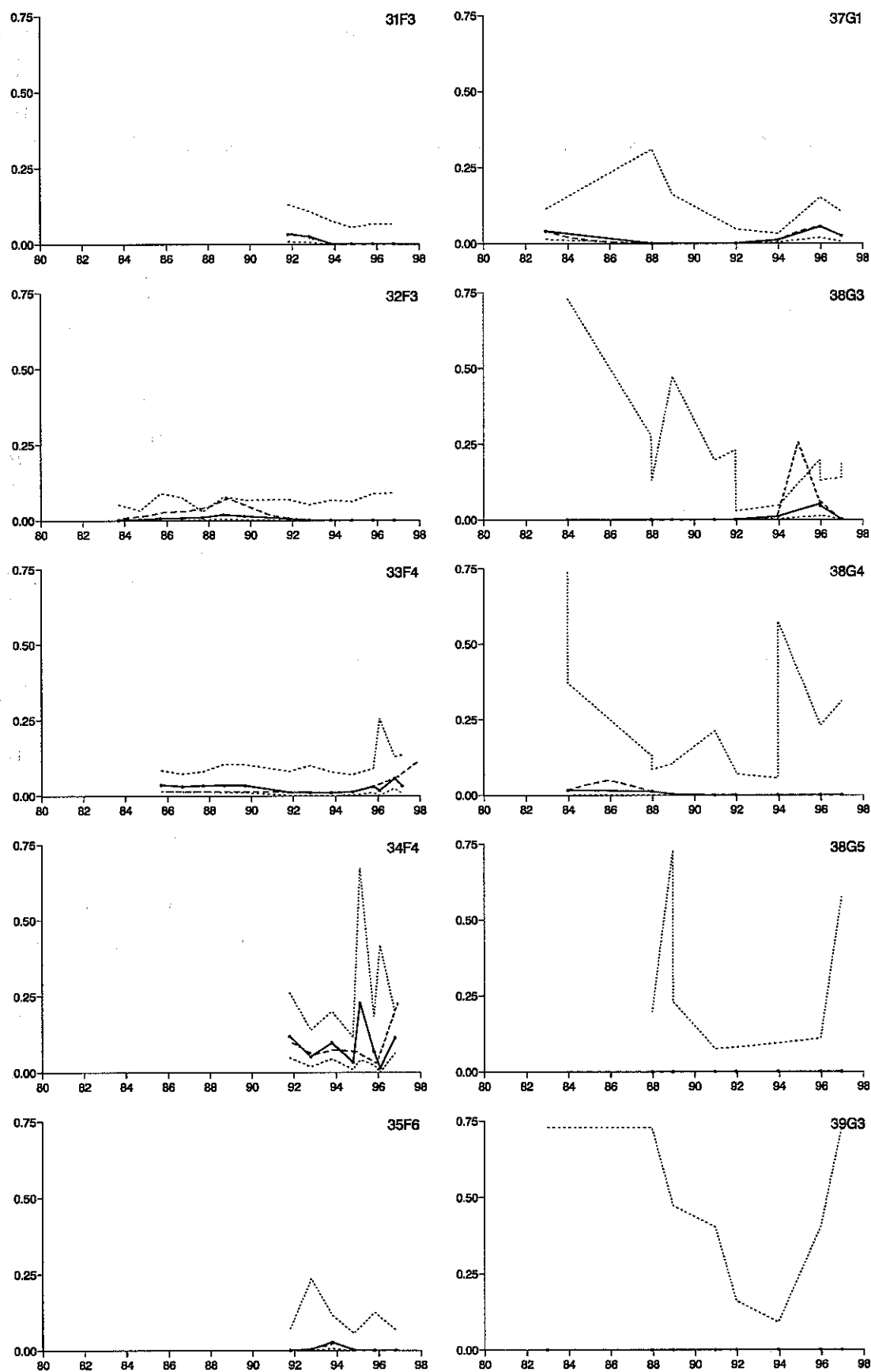
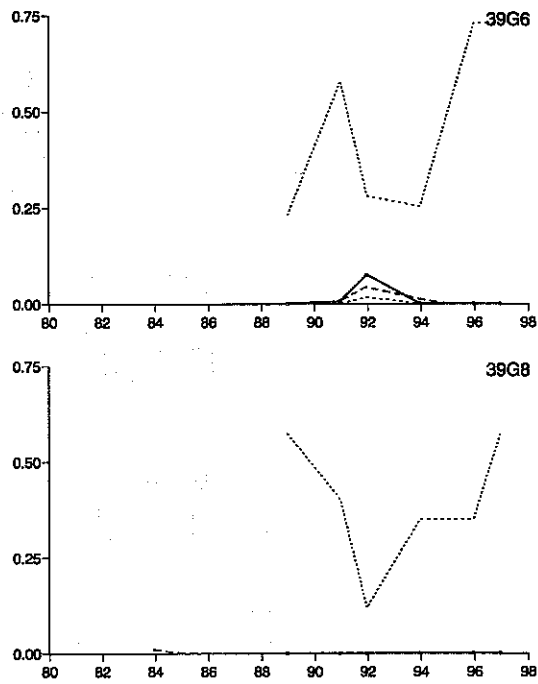
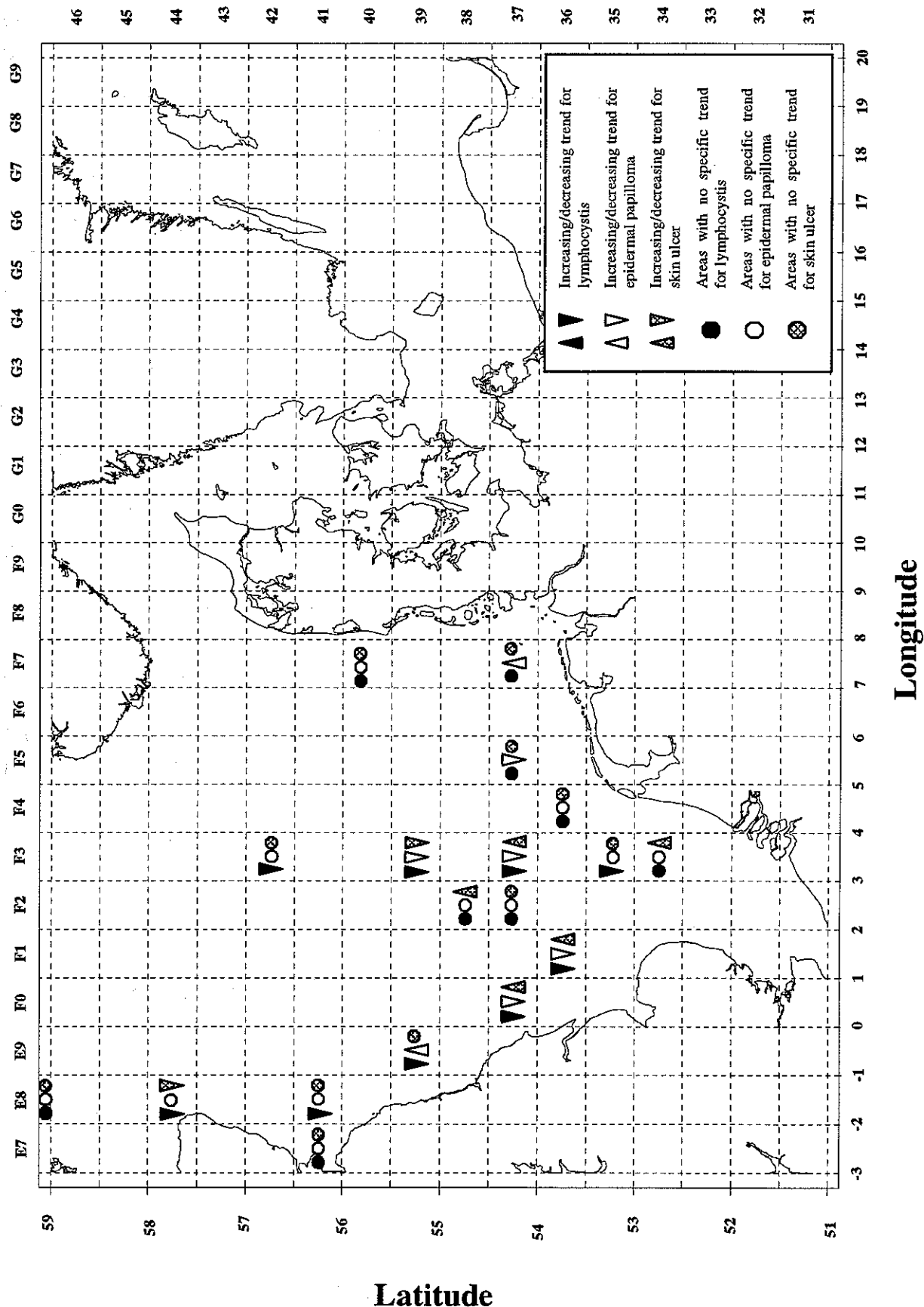


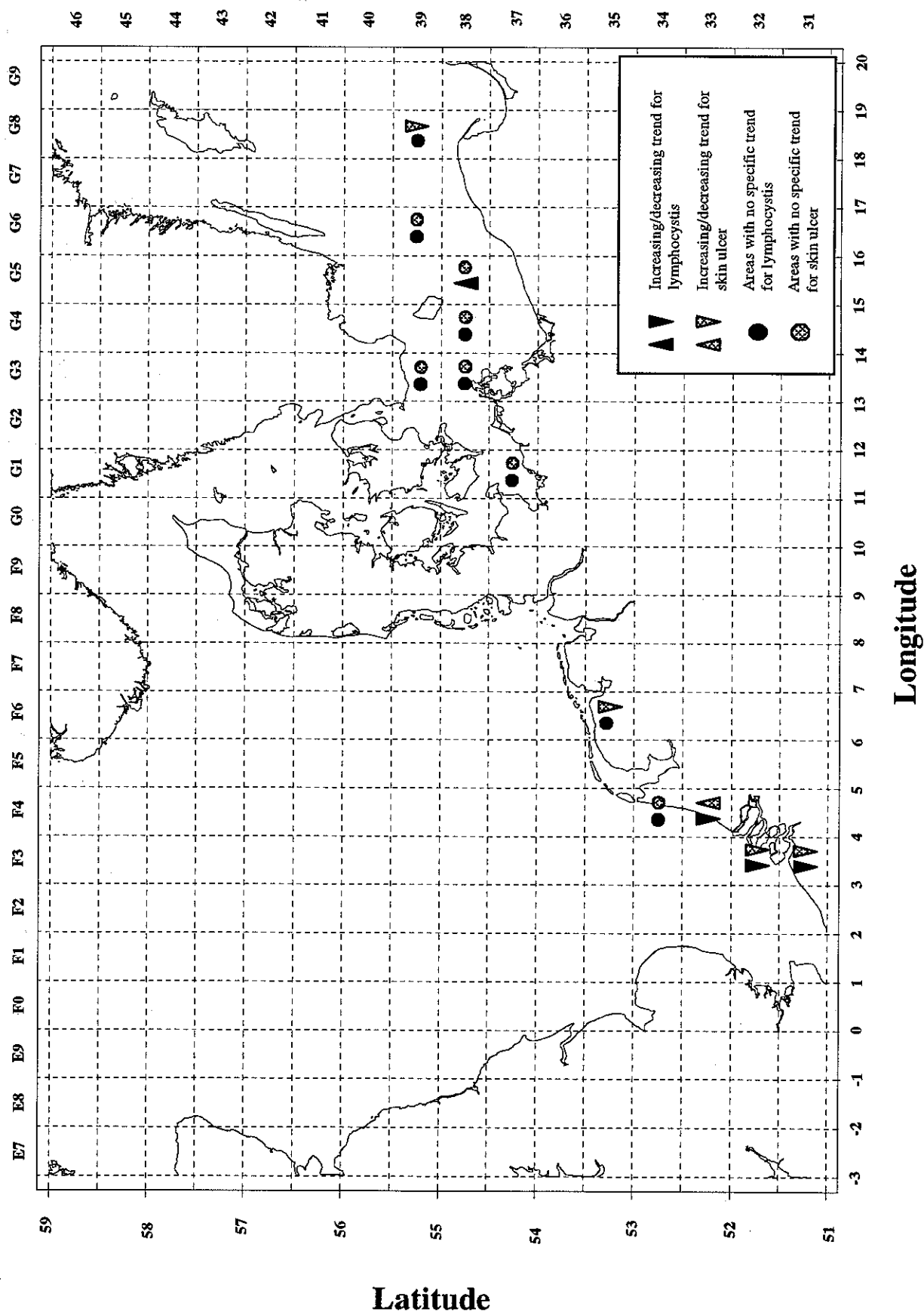
Figure A8.6. Continued.



**Figure A8.7.** Recent developments in the prevalence of lymphocystis, epidermal hyperplasia/papilloma, and acute/healing skin ulcers in common dab (*Limanda limanda*). Only trends from 1992 onwards until the last observation in a rectangle are considered. The trend is checked for significance ( $p=10\%$ ) by the Mann-Kendall test for non-parametric trends. Significant increasing/decreasing trends are marked with arrows (up/down) and areas with no changes are marked with circles.



**Figure A8.8.** Recent developments in the prevalence of lymphocystis, epidermal hyperplasia/papilloma, and acute/healing skin ulcers in flounder (*Platichthys flesus*). Only trends from 1992 onwards until the last observation in a rectangle are considered. The trend is checked for significance ( $p = 10\%$ ) by the Mann-Kendall test for non-parametric trends. Significant increasing/decreasing trends are marked with arrows (up/down) and areas with no changes are marked with circles.



confidence intervals indicate a clear increase or decrease in the trend.

However, even if confidence intervals overlap, a significant trend can exist. The reason is that a confidence band as a graphical tool can only give point-wise information, i.e., it does not say that an arbitrary shape of curve within the band would be compatible with the structure of the trend. Hence, an additional test is needed to derive a more concise statement about recent trends. The Mann-Kendall test for the existence of a non-parametric trend is an appropriate tool for this purpose (see ICES, 1997b, p. 121). Only prevalence data from 1992 onwards was used to consider recent trends. Locations for which the observation series had no or too few data in that time window could clearly provide no information in this respect. The Mann-Kendall test needs at least four observations to be able to detect a trend with a two-sided error level of 10 %. The two-sided test level of 10 % is certainly only an approximate value, as there is no adjustment for the fact that the test is performed on quantities which were already estimates based on the observational data. The predicted prevalences for 15 February of each year were used for the trend tests, thus providing a common time pattern for all rectangles and avoiding quarter effects. Figures A8.7 and A8.8 summarize the results of the tests for dab and flounder, respectively, with the same sex/length properties as in Figures A8.2–A8.6. Rectangles and diseases with significant trends over the period 1992–1996/1997 are marked with upward or downward arrows; rectangles and diseases showing no trends are marked with circles.

In the following paragraphs, the main results of the analyses carried out are summarized for dab and flounder, separately.

### 6.1 Dab (*Limanda limanda*)

- There is a sufficient spatial coverage of disease prevalence data in the southern and central North Sea, whilst data for areas in the northern North Sea and outside the North Sea are relatively scarce (exception: areas 41G2, 42G1, 44F9, and 44G0 in the Kattegat and Skagerrak). Disease data covering the whole period from 1981–1997 are only available from the southern part of the North Sea (south of 55 °N), whilst data series from the northern parts (north of 55 °N) and the Skagerrak/Kattegat area either started later or were terminated earlier.
- For all diseases considered, the absolute levels of the estimated disease prevalence differ considerably between ICES rectangles. Some areas are characterized by a consistently high prevalence of a particular disease, such as areas 37F2 and 38F2 (Dogger Bank) for acute/healing skin ulcers and area 37F7 (German Bight) for epidermal hyperplasia/papilloma. Others are characterized by a consistently low prevalence, such as areas 44F9 and 44G0

(Skagerrak) for lymphocystis and epidermal hyperplasia/papilloma.

- In some areas, the prevalence over time is rather uniform with a relatively low degree of variation (e.g., 37F7 and 39E9 for lymphocystis and 41E8 for epidermal hyperplasia/papilloma), whilst prevalence shows strong variation in other areas (e.g., 40F6 for lymphocystis and 37F2 for acute/healing skin ulcers). Sometimes, even sudden and marked changes occur within a relatively short period of time (e.g., 41G2 for lymphocystis and 37F2 for acute/healing skin ulcers).
- The prevalence undergoes significant and consistent seasonal change in some diseases and areas (first quarter of the year vs. second quarter of the year, e.g., areas 37F7 and 38F2 for acute/healing skin ulcers, areas 38F2 and 41E7 for lymphocystis).
- The prevalence of lymphocystis was either stable or decreasing in all rectangles with sufficient data for the period 1992–1996/1997. In none of the areas was a significant increase detected.
- The prevalence of epidermal hyperplasia/papilloma was more variable. Although stable or decreasing trends dominated, there are two areas (37F7, 40F6) which showed a significant upward trend. Especially the marked increase in rectangle 37F7 situated in the German Bight requires further attention.
- Acute/healing skin ulcers are the only diseases which increased in prevalence in a considerable number of rectangles, particularly in the southern and central North Sea.

### 6.2 Flounder (*Platichthys flesus*)

- There are many fewer data for diseases of flounder compared to dab, resulting in a lack of spatial and temporal coverage. In the North Sea, data are restricted to Dutch coastal areas and in the Baltic Sea to the southwestern part.
- The prevalence of lymphocystis is considerably higher in the Baltic Sea areas than in the North Sea areas analysed. In most areas in Dutch coastal waters, there was a significant downward trend in the prevalence of lymphocystis since 1992. No significant trends were found in the Baltic Sea, except in area 38G5, which was characterized by a significant increase.
- For acute/healing skin ulcers, no marked differences seem to exist in the prevalence between the North Sea and Baltic Sea areas considered. Both the North Sea and Baltic Sea data do not reveal any consistent temporal trends.

## 7 DISCUSSION AND CONCLUSIONS

The present status of the fish disease section of the ICES Environmental Data Centre database can be considered

sufficiently complete to allow a statistical analysis of spatial and temporal characteristics of disease prevalences. However, most data available refer to diseases of dab and only limited information exists so far on flounder diseases. This is mainly due to the fact that the submission of flounder data has only started recently. Up to now, only two laboratories have provided information, but there are a number of others (particularly in the Baltic Sea countries) which, in the future, will contribute to the data bank by submitting historic and current data. Once the flounder data bank has been completed, a more comprehensive data analysis similar to the one carried out with the dab data will be possible, providing better insight into spatial and temporal trends.

Furthermore, it has to be taken into account when assessing the results of the analyses that the disease data analysed only represent a part of the whole set available in the ICES Environmental Data Centre, since data from male fish and from female fish outside the size groups 20–24 cm (dab) and 25–29 cm (flounder) were not included in the analysis. More complete information on spatial and temporal trends will be available once these data have been considered in the analysis as well.

The results of the analysis indicate that there are marked spatial differences with respect to the absolute levels and the temporal changes in the prevalences of the diseases under study (see Figures A8.2 to A8.6). For dab, with only one exception (39F3), there are no rectangles where the diseases considered followed the same temporal trend (either an increase or decrease in prevalence) over the period 1992–1996/1997 (see Figure A8.7). The situation is different for flounder, for which in rectangles located in the southern North Sea a corresponding decrease in both lymphocystis and skin ulcer disease was detected (see Figure A8.8). For dab, there is no clear indication of consistent temporal trends in larger areas comprising a number of neighbouring rectangles. However, it seems that the southeastern part of the North Sea is characterized by a more stable situation with respect to possible changes in the disease prevalence than the other areas.

However, the dab data also reveal the occurrence of some common temporal features. For instance, the prevalences of lymphocystis and epidermal hyperplasia/papilloma in dab seem to follow a similar temporal pattern in some areas (e.g., areas 34F3, 35F3, and 36F1) (see Figures A8.2 and A8.3). Furthermore, there are some common peaks in disease prevalence in the majority of areas in the period 1988–1992 for both dab and flounder (see Figures A8.2 to A8.6), indicating the presence of underlying mechanisms driving the development of all diseases in the same way, possibly linked to general ecosystem change.

In general, the analysis for temporal trends for the period 1992–1996/1997 showed that, when combining the disease data, stable to decreasing trends in the prevalence dominate (see Figures A8.7 and A8.8). The only disease which increased in a considerable number of rectangles is acute/healing skin ulcers of dab. This disease, however, occurs at a low prevalence in most areas.

In addition to the establishment of these general trends, the results of the analysis help to identify areas of concern which differ from other areas with respect to both the absolute disease prevalence levels and the temporal changes in disease prevalence and, therefore, deserve particular attention (examples: exceptionally high prevalence levels of acute/healing skin ulcers in dab from areas 37F2 and 38F2 since 1990, the increasing prevalence of epidermal hyperplasia/papilloma in dab from area 37F7 in the German Bight, and the increase in the prevalence of lymphocystis in Baltic flounder from area 38G5).

It must be emphasized, however, that the results of the analyses do not provide information on possible natural and/or anthropogenic causes of the observed spatial and temporal trends. This will only be possible when environmental parameters known or suspected to be involved in the disease aetiology (e.g., oceanographic, contaminant, fish stock assessment and fishery intensity data) are included in a holistic data analysis. Since these kinds of data are available in the different ICES data banks, it will be a desirable future task to assess the suitability of these data and, if considered appropriate and promising, to undertake a holistic data analysis providing information on cause/effect relationships between fish diseases and environmental factors.

## 8 ACKNOWLEDGEMENT

This material was prepared by W. Wosniok (Institute for Statistics, University of Bremen, Bremen, Germany), T. Lang (Federal Research Centre for Fishery, Institute of Fishery Ecology, Cuxhaven, Germany), A.D. Vethaak (RIKZ, Ecotoxicology Section, Middelburg, The Netherlands), S. des Clers (Ecosystem Analysis and Management Group, Biological Sciences, University of Warwick, Coventry, United Kingdom), and S. Møllergaard (Danish Institute for Fisheries Research, Charlottenlund, Denmark).

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## MARINE MAMMAL BY-CATCH IN THE NORTHEAST ATLANTIC

## 1 INTRODUCTION

The only direct way to obtain reliable quantitative estimates of total marine mammal by-catch in a fishery is via an independent observer scheme covering a representative sample of the fishery (Northridge, 1996; IWC, 1996). Other studies (such as voluntary reporting schemes, examination of stranded animals for net-marks, etc.) can indicate areas where there may be significant by-catch, and where closer monitoring may be needed, but do not provide trustworthy numerical estimates. Small-boat fisheries, where it is difficult or impossible to find space on board for an observer, pose a particular problem for by-catch monitoring. In fisheries where no direct observer scheme is feasible, indirect estimates based on by-catch rates for similar gears may provide a useful starting point for estimates. Although observer schemes give the best available estimates, it should be noted that there is always some negative bias associated with the estimates, because of unobserved by-catch (e.g., animals that sink without being seen); while there is a general expectation that such bias should be small, there are no quantitative estimates of its magnitude.

There have been few published studies of marine mammal by-catches in the regions of the OSPAR maritime area (see Figure A9.1). These studies are summarized below along with an assessment of which additional fisheries are also likely (based on information from elsewhere) to have a marine mammal by-catch. The summaries are organized by OSPAR region (Figure A9.1) and by gear type. Gear types often correspond to the species caught: certain species tend to get caught in certain gear types. It is important to note that the lack of a by-catch estimate for a fishery does not necessarily imply that the by-catch is negligible. Also, the absolute numbers of animals in a by-catch may not be a good guide to its importance. Thus, there may only be in the order of 10,000 white-beaked dolphins (*Lagenorhynchus albirostris*) in the North Sea (Hammond *et al.*, 1995); a by-catch of 100 animals would have a greater population effect than a by-catch of the same number of harbour porpoises (*Phocoena phocoena*), where there is likely to be in excess of 250,000 in the North Sea.

The Working Group on Marine Mammal Population Dynamics and Trophic Interactions (WGMMPD) has reviewed and evaluated available information for each OSPAR region. This evaluation indicates which known by-catch rates in each region are believed to be non-sustainable and which fisheries merit further immediate study. In some cases, some of these studies have started, but have yet to report. It follows from the statements above that many of the data do not have the desired quality. However, the information reviewed here is probably the best currently available.

The issue of by-catch in the North Sea and Baltic Sea has been the subject of study by parties to the Agreement on the Conservation of Small Cetaceans in the Baltic and North Seas (ASCOBANS). A report was written for and accepted at the 1997 Meeting of Parties to this Agreement (ASCOBANS, 1997). This report took as an interim rule (which may be tightened in future in the light of further studies) that any by-catch rate above 2 % of the estimated abundance of the population is likely to lead to a population size below about half of its potential, and carries a significant risk of not being sustainable at all and of driving the population towards extinction. It should be noted that this is very much a minimum standard, and a precautionary approach to uncertainty would give a lower threshold. See ASCOBANS (1997) for a full discussion of this issue.

## 2 OSPAR REGION I: ARCTIC WATERS

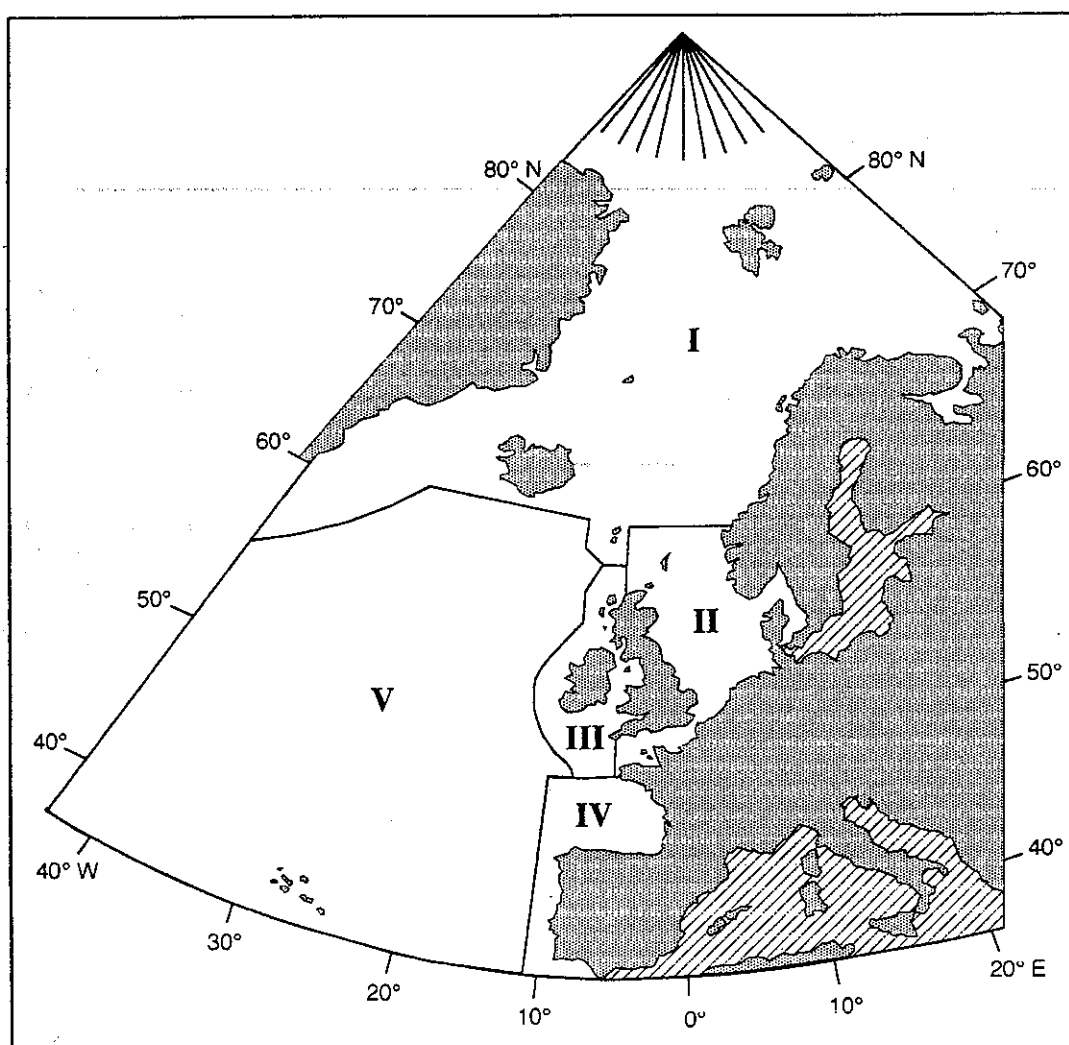
## 2.1 Fixed Bottom-set Nets

Material collected during dietary studies in Icelandic coastal waters over five years in the early 1990s indicated that a minimum of two hundred harbour porpoises were by-caught per year in nearshore bottom-set gillnet fisheries, mostly between March and May. This timing coincides with the capelin spawning migration in these waters (Víkingsson and Sigurjónsson, 1996). This fishery also catches small numbers of white-beaked dolphins.

Norway has conducted marking programmes with conventional external tags on ice-breeding and coastal seals to study migration, ageing methods and abundance. Recoveries of these tagged seals have also been made from fishing gear associated with coastal shelf bottom fisheries using bottom gillnets and fyke nets. They include grey seals (*Halichoerus grypus*) and harbour seals (*Phoca vitulina*) along the Norwegian coastline and harp seals (*Phoca groenlandica*) along the Norwegian coast and the Icelandic coasts. The vast majority of these catches are taken within OSPAR Region I, while a few are taken in OSPAR Region II. The recapture information indicates that by-catches occur, but not their size (although 4.6 % of all tagged animals were recovered from gillnets (Henriksen *et al.*, 1996) nor whether this constitutes a problem. Most of the recaptures in fishing gear are of young seals during their first year of life.

In addition, there were some very exceptional seal invasions on the Norwegian coast during the years 1986–1988. These were mostly harp seals thought to originate from the White Sea/Barents Sea population. The number of by-caught animals peaked in 1987 at 56,000 animals. A further invasion occurred in winter

**Figure A9.1.** Regions of the OSPAR maritime area (OSPAR, 1995).



1994/1995, when about 10,600 harp seals were by-caught in bottom-set cod nets (Nilssen *et al.*, 1996).

Hooded seals (*Cystophora cristata*) breed on the ice to the east of Greenland and off Newfoundland. Both populations range widely to feed in deep-water areas of Region I; however, no by-catches of this species in deep water have been recorded.

Hauksson and Bogason (1995a, 1995b, 1995c, 1995d, 1995e) reported seal by-catches in the Icelandic set net fishery. Most of the by-catch was in northern Icelandic waters and was caused by lumpsucker gillnets. By-catch species included harp, ringed (*Phoca hispida*), bearded (*Erignathus barbatus*) and hooded seals.

A large number of vessels participate in the Norwegian fisheries, most of which are small vessels that operate in coastal waters. In 1996, a total of 5561 vessels used

gillnets. Of these, 1381 operated in 1997 in waters south of 62°N (mostly in OSPAR Region II), while the remainder were in OSPAR Region I (in literature of the Norwegian Directorate of Fisheries). In 1989 and 1990, bottom-set gillnet fisheries were surveyed using similar methods to those used to survey drift nets (see below), and significantly lower catch rates and actual catch figures were recorded (Bjørge *et al.*, 1991).

Larsen (1995) reported, on the basis of interviews with fishermen, that there were only a few by-catches of harbour porpoises in the offshore (beyond 30 n.m. from the coast) net fishery for cod.

## 2.2 Drift Nets

In 1988, the drift net fishery for salmon in Norwegian waters was surveyed for cetacean by-catches. During a six-week period, incidental catches of 96 porpoises were

observed. The catch rate in this fishery was relatively high and averaged 0.8 porpoises per 1000 net km hours (this is a unit of fishing effort expressed as the number of nets used, multiplied by their length in kilometres and the number of hours they were deployed). After the 1988 salmon fishing season, all use of large-mesh drift nets was prohibited in Norwegian waters due to the inability of this fishery to discriminate between salmon from different populations.

### 2.3 Longlines

Larsen (1995) reported that harbour porpoises were very occasionally taken on longlines off the Faroe Islands.

### 2.4 Evaluation

With the exception of the recording of the seal by-catch off Norway, there have been no direct assessments of by-catches of marine mammals in this region during the 1990s. On the basis of the information available, bottom-set fixed nets appear to offer the greatest potential for by-catch in this region. The fisheries off Iceland, and by small vessels off Norway, deserve further investigation. No information was available on by-catches in pelagic trawl or deep-water trawl fisheries in the region; these types of fisheries are known to catch marine mammals elsewhere (Donoghue, 1997; Lens, 1997; Pemberton *et al.*, 1994).

## 3 OSPAR REGION II: GREATER NORTH SEA

Much of this section is based on a report compiled for the second Meeting of Parties to the ASCOBANS Agreement (ASCOBANS, 1997). The contribution of persons who helped to write that report who are not members of WGMMPD is gratefully acknowledged.

### 3.1 Fixed Bottom-set Nets

This covers a variety of gears: trammel nets, tangle nets, and gillnets set at different heights and with different mesh sizes. Evidence from the North Sea and elsewhere indicates that any nets standing off the seabed are liable to catch harbour porpoises (if present in the area), regardless of net type or attachment methods (Frady *et al.*, 1994). However, some types of bottom-setting seem to cause higher by-catch rates than others in the same area.

An observer scheme monitored Swedish cod gillnetters in the marine part of a single ICES statistical rectangle covering about 1500 km<sup>2</sup> lying off Gothenburg, Sweden, in 1995 and 1996 (Carlström and Berggren, 1996). By-catch rates were very similar in the two years, at around 32 porpoises per 10,000 net km hours, giving an annual by-catch estimate of 53 porpoises in this single rectangle for 1995. Further observations in the cod and pollock

fishery in the Swedish Skagerrak were made from March 1996 to February 1997 (Carlström and Berggren, 1996). By-catch rates were 40 porpoises per 10,000 net km hours in spring, 39 in autumn, and 0 in winter. This produced an annual estimated by-catch in this fishery of 113 porpoises. However, several Swedish and Danish set-net fisheries also operate in this area targeting cod, plaice, spiny dogfish, and lump sucker, so there is the potential for a high total by-catch. A Danish discard-recording project is now active in this area.

About 1380 small vessels operated with gillnets in the Norwegian fisheries in 1997 in waters south of 62°N, mostly in OSPAR Region II (in literature of the Norwegian Directorate of Fisheries). When last monitored in 1989 and 1990, this fishery had a significantly lower catch rate than the now-ceased salmon drift net fishery.

A number of UK gillnetters operate around and between Orkney and Shetland. Porpoises are numerous in the northern North Sea and in neighbouring waters, and there appears to be considerable gillnet effort in this region, so there is the potential for substantial by-catch.

The Danish bottom-set gillnet fleet is presently the largest in the European Community (Lowry and Teilmann, 1994), and the fisheries for cod, sole, and turbot in the eastern central North Sea were studied in 1992–1994 (Vinther, 1995, 1996). Extensive observer coverage revealed an estimated annual porpoise by-catch of 4450 (95 % confidence interval: 2580 to 6320) in the cod and turbot fisheries. This was based on an observed by-catch of 161 harbour porpoises in 2106 km of net on 61 trips. One *Lagenorhynchus* dolphin was also by-caught. Almost all of the by-catch was between 55°30'N and 57°30'N, and no by-catch was seen in the sole fishery, which mostly sets closer to shore. The study and the by-catch estimate excluded smaller Danish boats (ca. 20 % of the landings) and the fisheries for plaice, lump sucker, and hake (the latter are associated elsewhere with a high porpoise by-catch rate (Tregenza *et al.*, 1997)). The total Danish by-catch is therefore likely to be substantially higher. Further studies off Denmark are re-examining these fisheries as well as those directed at other species. There is also a recreational inshore gillnet (not all bottom-set) fishery off Denmark which has not been investigated.

There are several UK set gillnet fisheries in this area of the central North Sea, with substantial overall effort. The largest component, the English wreck net fishery (about twelve boats working out of Grimsby), is being studied at present in the BY-CARE project, and results are expected at the end of 1998. Most of the English fishery is for cod. The mode of operation is similar to that in the Danish gillnet fishery and there is a partial overlap in the areas fished. There is a variety of inshore gillnet fisheries along the east coast of the UK, with the target species including cod, sole, turbot and salmon, and with most of

the effort off the Yorkshire coast. Some by-catches have previously been reported along most of the coast (Northridge, 1988). A small fishery off the east coast of Scotland, which has been in decline in recent years, was reported to be taking from 1–20 animals per year in the 1960s and 1970s (Rae, 1965, 1973).

Very little set-netting is prosecuted off the Netherlands or Belgium. However, since 1988, at least 24 harbour porpoises have stranded dead in Belgium. The cause of death of at least six animals was most probably by-catch (not all animals undergo necropsy) (Coignoul and Jauniaux, 1995; Van Gompel, 1991, 1996; J. Haelters, pers. comm.).

In recent years, there have been small German set-net fisheries for cod and sole in the North Sea (Kock and Benke, 1996). Of the 565 porpoises found dead on beaches or reported as by-catch, only 23 could with certainty be ascribed to by-catch, with another 38 having skin lesions consistent with by-catch. Most of the by-caught animals were less than two years old.

Considerable quantities of gillnets and trammel nets are deployed off France and England. By-catch has not been studied systematically, but two porpoises were recorded in the 1980s (Martin *et al.*, 1990). This area has very low cetacean densities and inevitably by-catch rates will be low, so that very high percentage coverage would be required to obtain reliable estimates from a conventional observer scheme in this area.

### 3.2 Pelagic Trawls

By-catch in the German pelagic trawl fisheries for herring and mackerel in the North Sea was investigated in 1996. Observers were on board during five fishing trips (out of 33). Four pilot whales (*Globicephala* sp.) were caught in August while fishing for mackerel, and an additional four were caught in December while fishing for herring (Kock, 1997).

### 3.3 Drift Nets

There are few drift net fisheries left in the North Sea and overall cetacean by-catch is therefore probably low compared with other fisheries. Although large mesh nets have been prohibited in Norwegian waters since 1988, small mesh (mesh size 3.5 cm) nets (e.g., for mackerel) may still be used. However, in order to avoid by-catches it is mandatory to submerge these nets at depths of at least 3 m below the surface. In 1996, a total of 149 vessels used nets for mackerel, and 147 operated in the ASCOBANS area. Most of these vessels were small and 125 vessels were under 11 m in length (in literature of the Norwegian Directorate of Fisheries). A similar, but smaller (27 vessels in 1997), fishery operates off Sweden.

A salmon fishery off the northeast coast of England uses relatively short nets, with the fisherman always in attendance. By-catch has been reported, but most animals are reputedly released alive. A small (circa 50 small boats) UK inshore drift net fishery for herring operates off the East Anglia coast as far north as the Wash and there is a commercial herring drift net fishery of around fifteen boats in the Blackwater estuary. Porpoises are rare in this area.

### 3.4 Fixed Gear apart from Set Nets

Harbour porpoises are caught in pound nets around Denmark, but many are released alive (Lowry and Teilmann, 1994). A variety of similar gears are used in England and Scotland for catching salmon, and porpoises have been reported caught; however, there is no estimate of numbers. The small scale of the fisheries means that by-catch is likely to be very low compared to that caused by set nets.

### 3.5 Other Fishing Methods

The other common fishing methods in this OSPAR region are bottom trawling, beam trawling, seining, and longlining. There are records and several anecdotal reports of cetacean by-catches from some of these fisheries (Northridge 1988, 1991; Martin *et al.*, 1990; Kock and Benke, 1996). By-catch rates appear to be very low and at present it seems likely that any by-catch rates from these unrecorded fisheries are small compared to those from set nets and pelagic trawls. However, the effort from bottom trawls in particular is very high, so that even a low by-catch rate could potentially cause significant total by-catch.

### 3.6 Evaluation

The harbour porpoise by-catch in the central and southern North Sea by a component of the bottom-set gillnet fishery is estimated at 4450 annually, which comprises more than 2.6 % of the number of harbour porpoises inhabiting this area. This level of by-catch could very likely lead to a decline in the population size. Estimates are required of the level of by-catch in other similar fisheries in the central and southern North Sea. There are no by-catch estimates available in the northern North Sea, although observer schemes are now running in some fisheries. Understanding of this by-catch and of the harbour porpoise population that is being impacted is required. The by-catch of harbour porpoises in the Swedish Skagerrak is likely to exceed 4 % of the population; this, coupled with evidence of declining numbers of harbour porpoises in the area (Carlström and Berggren, 1996), would indicate that action is now needed to reduce the by-catch. By-catches in the pelagic trawl (other than German) and demersal trawl fisheries in the region have not been systematically monitored.

## 4 OSPAR REGION III: THE CELTIC SEAS

### 4.1 Fixed Bottom-set Nets

From August 1992 to March 1994, there was an observer scheme for the English and Irish hake gillnet, tanglenet, and wreck net fisheries in the seas to the west of England and south of Ireland, originally to monitor the by-catch of common dolphins (*Delphinus delphis*) (Tregenza *et al.*, 1997). Observers were present for the hauling of over 2500 km of net which caught 43 harbour porpoises and four common dolphins. Nearly all of the porpoises caught were in the hake fisheries, with only one in a tangle net and none in wreck nets. Although the common dolphin by-catch was small, the harbour porpoise by-catch was estimated to be 2200 (95 % confidence interval: 900 to 3500). This figure represents 6.2 % of the estimated number of porpoises in this area and there is a serious cause for concern about the ability of this population to sustain this level of by-catch. The scheme did not cover trammel netters or smaller boats, which may contribute substantially to the overall by-catch. In the southern Celtic Shelf, where porpoise densities may be lower, there are large French set-net fisheries (Morizur *et al.*, 1992).

During a study of predator damage to net-caught anglerfish off the south coast of Ireland, Collins *et al.* (1993) examined four young grey seals which had been by-caught.

A fishery on the shelf edge to the west of Scotland uses bottom-set nets targeting anglerfish. There have been no reports of by-catch in this fishery and it also appears to operate to the west of the main part of the range of the harbour porpoise. In recent years, vessels have been licensed to use these nets closer to the coast, within the main part of the harbour porpoise range. An observer scheme for this fishery is presently being implemented.

### 4.2 Pelagic Trawls

There is evidence from the Celtic Shelf that pelagic trawling catches substantial numbers of dolphins (Morizur *et al.*, 1997a, 1997b); thirteen common dolphins, five white-sided dolphins (*Lagenorhynchus acutus*), and four grey seals were observed as by-catch in 1788 hours of pelagic trawling. These fisheries were targeting tuna, hake, sea bass, horse mackerel, Atlantic mackerel, and herring. Forty-seven white-sided dolphins were reported as by-catch in Dutch trawl fisheries in 1993 and 1994 off southwest Ireland (Addinck *et al.*, 1996). Kuiken *et al.* (1994) reported a mass stranding of common dolphins in southwest England bearing characteristic markings of by-catch. The high overall effort from pelagic trawls in these waters means that there is the potential for significant by-catch.

## 4.3 Drift Nets

The albacore drift net fishery to the west of Britain and France and southwest of Ireland has a by-catch of common and striped dolphins (*Stenella coeruleoalba*) (Goujon *et al.*, 1993b; Antoine *et al.*, 1997). The fishery straddles the boundaries of OSPAR Regions III, IV, and V. The by-catch of striped dolphins is likely to exceed 2 % of the number of animals in the area.

### 4.4 Evaluation

The by-catch in fixed bottom-set nets in the seas to the south of Ireland is likely to place the population of harbour porpoises in this area at risk. The by-catch in the pelagic trawl fisheries in the same area may be placing populations of other delphinids at risk. There is presently no information available on by-catches in waters to the west and north of Ireland. Monitoring schemes for the fixed bottom-set net fisheries in these areas are needed. The low populations of cetaceans in the Irish Sea would probably mean that any by-catch rate there would be low (no information is available), but equally any by-catch might place any localized population in this area at risk.

## 5 OSPAR REGION IV: BAY OF BISCAY AND IBERIAN COAST

### 5.1 Gillnets

By-catches, principally in gillnets (type unspecified), have been reported for several decades in the Bay of Biscay and in Atlantic waters off the Iberian coast (Reiner, 1980; Garcia-Castrillo *et al.*, 1990, 1993, 1994; Cendrero and Garcia-Castrillo, 1992; Lens *et al.*, 1995; Pérez *et al.*, 1997) based on reports by fishermen and also from the examination of stranded animals.

Pérez *et al.* (1997) reported seven by-catches and twelve strandings of common dolphins with evidence of having been caught in gillnets.

Ten cetacean species (pygmy sperm whale (*Kogia breviceps*), goose-beaked whale (*Ziphius cavirostris*), fin whale (*Balaenoptera physalus*), pilot whale, Risso's dolphin (*Grampus griseus*), common dolphin, bottlenose dolphin (*Tursiops truncatus*), spotted dolphin (*Stenella* sp.), striped dolphin, and harbour porpoise), and two Phocid seal species (grey and common seals) have been recorded by-caught in fishing gear. Cetacean by-catches are typically higher in waters off the northwest coast of Spain compared with the Bay of Biscay. In all regions, the frequency and scale of strandings (many net-marked animals) coincide with available by-catch data. Numerically, common dolphins are the species most frequently taken (10–80 animals annually since the early 1980s). Strandings data, however, are likely to be negatively biased, particularly during the early phases (1980s) of monitoring programmes.

Sequeira and Ferriera (1994) considered that there were few reliable data on marine mammal by-catches on the Portuguese coast, but that common dolphin and harbour porpoise dominate in the marine mammal by-catch of gillnets.

## 5.2 Pelagic Trawls

Morizur *et al.* (1997a, 1997b) investigated several pelagic trawl fisheries in the Bay of Biscay. Cetaceans were observed as by-catches in the French tuna, hake, and sea bass trawl fisheries. The tuna fishery by-catch observed consists of bottlenose and common dolphins, while the other two fisheries recorded only common dolphins. The amount of observation undertaken was low, but catch rates were comparatively high. The authors considered that by-catch was not insignificant and that it required continued monitoring.

A comparatively small (i.e., when compared with gillnets) by-catch of common dolphins in trawls (type unspecified, but likely to be pelagic trawls) in this region was reported by Pérez *et al.* (1997).

## 5.3 Fish Traps

Sequeira and Ferriera (1994) noted that minke whales (*Balaenoptera acutorostrata*) occasionally become entangled in fishing trap leader-lines.

## 5.4 Evaluation

By-catches in this region have been briefly assessed for some pelagic trawl fisheries. These assessments indicate that some trawl fisheries are catching delphinids. These fisheries, and others presently not assessed, should receive further investigation. There is some evidence of by-catch in gillnets; these fisheries could also usefully be formally monitored.

# 6 OSPAR REGION V: THE WIDER ATLANTIC

## 6.1 Drift Nets

Following two years of experimental fishing, French fishermen initiated a summer/autumn albacore tuna drift net fishery in 1987. The fishery primarily occurs between 10°W to 21°W longitude and 51°N to 53°N latitude (Goujon *et al.*, 1993a, 1993b, 1996; Antoine *et al.*, 1997). An observer programme to estimate cetacean by-catch was conducted during the 1992 and 1993 fishing seasons. Fifty-eight and 63 vessels (trips) were covered in 1992 and 1993, respectively. Annually this represented about 25 % of the total number of trips undertaken. Ten cetacean species were by-caught (Table A9.1). The total incidental mortality of the common dolphin was estimated at 330–400 and that of the striped dolphin,

1135–1160. These two species accounted for 90 % of the by-catch.

**Table A9.1.** Numbers of cetaceans observed as by-catch in the French Atlantic tuna fishery in 1992 and 1993 (Goujon *et al.*, 1996).

Species	1992	1993
Striped dolphin	330	243
Common dolphin	114	90
Long-finned pilot whale	13	16
Bottlenose dolphin	10	8
Sperm whale	1	6
Fin whale	2	0
Minke whale	0	1
Risso's dolphin	1	7
Pygmy sperm whale	0	1
Unidentified cetaceans	4	5

## 6.2 Evaluation

WGMMPD found information on by-catch in only one fishery in this area. No information concerning fisheries around the Azores was found. No information was available on by-catch in pelagic trawl or deep-water trawl fisheries in this region; however, these types of fisheries are known to catch delphinids elsewhere.

Overall, ICES is concerned regarding the number of fisheries in the OSPAR regions that do not have adequate by-catch monitoring programmes. Similarly, the lack of information on marine mammal distribution and abundance, particularly in areas with known or suspected high levels of by-catch (based on intermittent observer programmes and/or stranding data), makes it difficult to evaluate the impact of mortalities on populations. ICES wishes to draw the attention of fisheries managers to those fisheries where marine mammal mortality exceeds levels likely to cause population decline.

## 7 ACKNOWLEDGEMENT

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## CONCENTRATIONS AND RELEVANT EFFECTS OF NON-*ORTHO* AND MONO-*ORTHO* CHLOROBIPHENYLS IN MARINE MAMMALS

### 1 INTRODUCTION

Global contamination of the marine environment by persistent organochlorine compounds such as chlorobiphenyls (CBs) and organochlorine pesticides (OCPs) including the DDTs, chlordanes and drins is well documented. As a result of their lipophilicity and persistence, these compounds bioaccumulate in the food chain resulting in high concentrations in top predators such as cetaceans and pinnipeds (Kawai *et al.*, 1988). The concentrations of CBs and OCPs found in marine mammal tissues is dependent on the sex, age, reproductive history, diet and habitat of individuals and species (Subramanian *et al.*, 1987a; McKenzie *et al.*, 1997a, 1997b). The adverse effects of high concentrations of CBs and OCPs include reproductive dysfunctions such as sterility and implantation failure (Reijnders, 1988), immune dysfunctions such as suppression of natural killer cell activity in seals (Ross *et al.*, 1996), lower thyroid hormone and vitamin A concentrations (Brouwer *et al.*, 1990), and decreased lymphocyte response in bottlenose dolphins (*Tursiops truncatus*) (Lahvis *et al.*, 1995).

The primary uptake route for these compounds is through the diet. The resultant concentrations and patterns of compounds in the predators, including cetaceans, are therefore a function of intake (species composition of the diet, concentrations of contaminants in dietary species, variations of diet and its contaminant burden in space and time, etc.) and loss (e.g., condition of the animal, reproductive status, excretion rate, metabolic abilities and activity) mechanisms. For example, the loss of lipophilic contaminants during lactation has been shown to lead to considerable differences in contaminant concentrations between mature males and females of some species. Different species show different abilities to metabolize or degrade organic contaminants.

Consequently, there are often very large differences (e.g., factors of 1000) in the concentrations of organic contaminants among individuals of the same species. The patterns of CB congeners found in marine mammals, however, show greater consistency than the absolute concentrations, and species-specific patterns can be recognized which have resulted from factors indicated above, for example, differences in diet and metabolic capability.

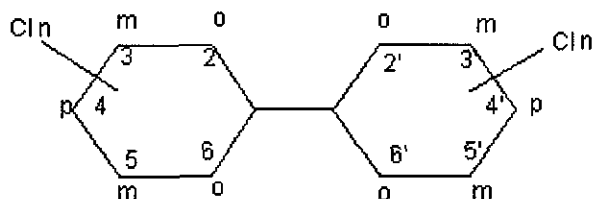
Wells *et al.* (1996) demonstrated species-specific patterns of CBs in marine mammals from northern European waters. Marine mammals feeding on similar food in localized areas show similar congener patterns, but species feeding at different trophic levels (i.e., on different prey species) are likely to have distinctly different CB patterns, reflecting different uptake and metabolic processes. Broad divisions can be drawn between fish feeders, mixed feeders, and cephalopod feeders.

### 2 COMPOUNDS LIKELY TO INDUCE EFFECTS ON MARINE MAMMALS

Of the 209 polychlorinated biphenyl (PCB) congeners which exist, only a few have been demonstrated as causing toxic effects. McFarland and Clark (1989) suggested that if potential toxicity, environmental prevalence, and relative abundance in animal tissues are used as criteria, only 36 congeners are environmentally relevant. Of these, approximately nine occur frequently in environmental samples and exhibit the greatest potential toxicity to marine organisms (Hall, 1998).

It is the congeners that can form a planar (flat) configuration that are of most concern. Chlorobiphenyls can rotate and align around the single bond between the phenyl rings if there are fewer than two chlorine atoms at the *ortho* positions on the rings (Figure A10.1). This planarity and the laterality of chlorine atoms are important structural features of CBs which determine their specific binding behaviour with proteins and certain toxic responses in biological systems. From a structural point of view, there are two distinct classes of CBs: the non-*ortho* CBs, i.e., those with no chlorine atoms at the *ortho* position, and the *ortho*-substituted CBs. Of the *ortho*-substituted group, based on the likelihood that they will achieve planarity, the mono-*ortho* CBs (those with a single chlorine at the *ortho* position) are potentially the most important toxicologically (Hall, 1998).

**Figure A10.1.** The generalized structure of polychlorinated biphenyls (Hall, 1998).



## 2.1 Non-ortho congeners (CB77, CB81, CB126, CB169)

One important distinguishing feature of the non-ortho group of CBs is the ability of the compounds to undergo stacking-type molecular interactions with other planar aromatic ring systems (McKinney and Waller, 1994). The important criteria for receptor-mediated responses are the stereo-selective ligand-receptor interactions and the corresponding structure-activity relationships (SARs). The structure-binding relationships for the CBs have shown that the most active compounds are the non-orthos that are substituted on both *para* and at least two *meta* positions. These congeners, for example, CB77 (3,3',4,4'-tetra), CB81 (3,4,4',5-tetra), CB126 (3,3',4,4',5-penta) and CB169 (3,3',4,4',5,5'-hexa), are approximate isostereomers of 2,3,7,8-tetrachloro-dibenzo-*p*-dioxin (TCDD) in their planar conformations and are strong inducers of the cytochrome P450 detoxification system. Since 2,3,7,8-TCDD is the most potent member of the organochlorine family of chemicals, compounds which mimic the structure of this highly toxic contaminant are often seen as accounting for most of the toxicity exerted by PCBs in the environment (Lemesh, 1992).

## 2.2 Mono-ortho congeners (CB105, CB114, CB118, CB123, CB156, CB157, CB167, CB189)

A second set of potentially toxic CB congeners, namely the mono-ortho planar analogues, are listed in Table A10.1. They generally occur in greater concentrations in marine mammal tissues than the non-ortho congeners (Boon and Eijgenraam, 1988; Boon *et al.*, 1987; Kawano *et al.*, 1988; Oehme *et al.*, 1995a, 1995b) and as such have a tendency to be more toxicologically important to these species than CB congeners which may elicit a greater response but which occur at much lower concentrations.

Table A10.1. Mono-ortho chlorobiphenyl congeners.

Configuration	IUPAC No.
2',3,4,4',5	123
2,3,4,4',5	114
2,3,3',4,4'	105
2,3',4,4',5	118
2,3,3',4,4',5	156
2,3',4,4',5,5'	167
2,3,3',4,4',5	157
2,3,3',4,4',5,5'	189

## 2.3 Di-ortho congeners (CB170, CB180)

Two di-ortho-substituted congeners, CB170 and CB180, have also been shown to induce P4501A enzymes and, due to their high concentrations in marine mammals, are

thought to be of toxicological significance (Ahlborg *et al.*, 1994).

## 2.4 Metabolism

Once absorbed by an animal, a substance will enter the circulatory system and be distributed throughout the body. However, compounds are often segregated into non-sensitive tissues which become storage sites (in the case of CBs in marine mammals, this is generally blubber, e.g., Addison and Brodie, 1987; Aguilar, 1987; Pomeroy *et al.*, 1996). Exchange between these storage sites and the bloodstream means that contaminants may gain access to actively metabolizing tissues (particularly the liver, which is the main organ for the metabolism of xenobiotics) and hence will be subjected to enzymatic attack and may undergo biotransformation (Boon *et al.*, 1992). This process may involve a number of different enzymes and many factors can affect the ability of animals to metabolize or detoxify xenobiotics, such as age, sex, species, nutritional and disease status, and even time of day or year (De Bruin, 1980). In quantitative terms, extrahepatic metabolism of xenobiotics is less important than hepatic metabolism, although it does occur, for example, in the skin and blood phagocytes (Griem *et al.*, 1998).

The dynamics and toxicity of contaminants in marine organisms can be investigated by studying the activity of these catalytic enzymes. The major class of metabolizing enzymes responsible for this in mammalian species are those of the cytochrome P450 group (see also Section 5.1, below). This biotransformation system (also known as the mono-oxygenase, mixed-function oxidase, MFO, or the phase I detoxification enzyme system), originally identified as the drug metabolizing system, has evolved as an important biomarker response that can be used for investigating contaminants in marine organisms (Skaare *et al.*, 1991; Stegeman and Hahn 1994; Fossi *et al.*, 1992; Goksøyr *et al.*, 1986, 1992). Groups of enzymes are also classified as phenobarbital (PB) inducible (i.e., globular xenobiotics), 3-methylcolanthrene (3-MC) (compounds with a planar, or which may attain a planar, configuration) inducible or mixed inducible (inducing both PB and 3-MC). Reaction rates involving these enzymes are altered by inducing or inhibiting agents and chemicals, such as PCBs. Over 70 (according to recent information this number may exceed 150) distinct cytochrome P450 genes have been identified in various species, and they are now classified into eight major families (CYP1 to CYP8; Nebert *et al.*, 1987; Nelson *et al.*, 1993). Subfamilies are identified by a capital letter, that is often combined with a number. CYP1A1, CYP1A2 (the 3-MC group) and CYP2B (the PB group) are enzymes all inducible by foreign compounds.

A further structural feature central to the likely toxicity of organochlorine compounds is their ability to bind to the cytosolic aryl hydrocarbon (Ah) receptor (Hahn *et al.*, 1992; Landers and Bunce, 1991; Timbrell, 1991). For the non-ortho-substituted congeners, most, if not all, of the

important effects known appear to be mediated by binding to this receptor. For the *ortho*-substituted congeners, the link to specific toxic endpoints is not well established and it is likely that various mechanisms are involved (McKinney and Waller, 1994). To some extent, differences appear to be due to variations in the degree of planarity. Agonist and antagonist effects may also depend on variability in the important reactivity properties; as *ortho* substitution appears to lower the dioxin-like binding properties in biological systems, differences in toxicokinetic properties should correlate with the degree of *ortho* substitution (Hall, 1998).

There may be non-additive interactions between non-dioxin-like CBs (e.g., PB-type inducers) and dioxin-like compounds. These non-dioxin-like CBs also appear to have their own independent toxic effects (tumour promotion, neurotoxicity) which may be as important as those caused by dioxin-like compounds.

### 3 TOXICITY EQUIVALENCE

CB congeners with one or no *ortho* chlorines and which do not have vicinal hydrogen atoms in the *meta* and *para* positions (CB105, CB118, CB156 and CB77, CB126, CB169, respectively) show a similar mode of toxicity to that of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) but have lower potency. In addition to these CBs, two di-*ortho*-chlorinated congeners (CB170 and CB180) have also been shown to produce TCDD-like effects, including cytochrome P450A1 induction, although only limited data on the toxicity of these compounds are available in the literature (Ahlborg *et al.*, 1994). The non-*ortho*-chlorinated congeners are generally more toxic than the mono-*ortho* congeners but are often present at considerably lower concentrations in marine mammals. Table A10.2 contains a classification of the CB congeners based on their structure, degree of chlorine substitution (i.e., the number and position of chlorine atoms), and the position of neighbouring or vicinal hydrogen atoms, as suggested by Boon *et al.* (1994). This type of classification may be useful in determining common structure-activity relationships and potential biological effects on marine mammals.

Results of several studies have demonstrated a relationship between the structure-binding and structure-activity (biochemical and toxic) relationships for several classes of halogenated aromatics. This observation forms the basis for the development of toxic equivalence factors (TEFs) for individual halogenated aromatic compounds (Safe, 1990; Ahlborg *et al.*, 1994). The relative toxicity of different compounds is compared to the most potent member of this family of chemicals, namely 2,3,7,8-TCDD. It has been suggested, from a human toxicology perspective, that when data are available from more than one response, the TEF values should be derived from the following effects in descending order of priority:

- 1) results obtained from long-term carcinogenicity studies;
- 2) data from reproductive studies;
- 3) results of subchronic experiments to measure the Ah receptor-mediated responses such as thymic atrophy, body weight loss, and immunotoxicity;
- 4) acute toxicity studies;
- 5) *in vivo*-or *in vitro* biochemical responses such as enzyme induction, receptor binding, etc.

**Table A10.2.** Classification of CB congeners based on their structure, degree of chlorine substitution (i.e., the number and position of chlorine atoms), and the position of neighbouring or vicinal hydrogen atoms as proposed by Boon *et al.* (1994).

CB Group	No. of <i>ortho</i> -chlorine atoms	Position of vicinal H atoms
<b>Group I</b>		
CB180	2	-
CB183	3	-
CB187	3	-
CB194	2	-
<b>Group II</b>		
CB99	2	<i>o, m</i>
CB128	2	<i>o, m</i> (2*)
CB138/CB163	2 (both)	<i>o, m</i> (both)
CB170	2	<i>o, m</i>
<b>Group III</b>		
CB70	1	<i>o, m + m, p</i>
CB105	1	<i>o, m</i> (2*)
CB118	1	<i>o, m</i>
<b>Group IV</b>		
CB44	2	<i>m, p</i> (2*)
CB49	2	<i>m, p</i>
CB52	2	<i>m, p</i> (2*)
CB101	2	<i>m, p</i>
<b>Group V</b>		
CB136	4	<i>m, p</i> (2*)
CB149	3	<i>m, p</i>

- Group I = congeners without any vicinal H atoms  
 Group II = congeners with vicinal H atoms in *ortho* and *meta* positions and  $\geq 2$  *ortho*-Cl atoms  
 Group III = congeners with vicinal H atoms in *ortho* and *meta* positions and  $\leq 1$  *ortho*-Cl atom  
 Group IV = congeners with vicinal H atoms in *meta* and *para* positions and  $\leq 2$  *ortho*-Cl atoms  
 Group V = congeners with vicinal H atoms in *meta* and *para* positions and  $\geq 3$  *ortho*-Cl atoms  
 2\* = 2 pairs of vicinal hydrogen atoms in *ortho* and *meta* or *meta* and *para* positions

These TEFs are then used as multipliers for each congener identified in a sample, the summation of which gives an overall toxicity equivalence (TEQ). The non-*ortho* CBs appear to be the most toxic congeners, based on structure-activity and structure-binding relationships where CB congeners exhibit Ah receptor agonist activity (Hall, 1998).

It is apparent from the data available that there are considerable variations within this group of compounds (and that very few data are available for 3,4,4',5-tetra CB (CB81)) and that they depend on both the species and the response. The data show that 3,3',4,4',5-penta CB (CB126) is clearly the most toxic planar CB congener, and the 2,3,7,8-TCDD/CB126 potency ratios were 66/1 (body weight loss, rat), 8.1/1 (thymic atrophy, rat), 10/1 (mouse foetal thymic lymphoid development), 125/1 (aryl hydrocarbon hydroxylase (AHH) induction, rat), 3.3/1 (AHH induction, rat hepatoma H4-II E cells), and 100/1 (chick embryo hepatocytes). Care should be taken when extrapolating the data, such as TEQs, derived from toxicity tests (or others) using cell lines because these cells usually develop specific characteristics that differ from the parent cells. This is one of the main drawbacks when using cell lines. However, both CB77 and CB169 congeners were considerably less toxic than CB126 and their relative potencies were highly variable. It is apparent from *in vivo* studies in the rat that CB77 is over 30 times less toxic than CB169, whereas in most of the *in vitro* assays these compounds exhibit similar potencies, or the reverse order of potency is observed. The relatively low *in vivo* toxicity of CB77 in the rat may be due to metabolism, but this congener has been detected in samples from marine mammals (Green *et al.*, 1996a).

Although the mono-*ortho* planar CBs all exhibit Ah receptor agonist activity, only limited quantitative structure-activity relationships (QSARs) are available (Safe, 1990). Again, the relative potencies for this subgroup are highly variable. CB118 is the major mono-*ortho* planar CB and is routinely identified in marine mammal samples. Some of the higher chlorinated CB congeners are more toxic than CB118 and the differences in potency may be related to different rates of *in vivo* metabolism.

The detection of toxic planar CBs in cetaceans and seals has led to the assessment of the toxic potential of CBs using the TEQ approach (Tanabe *et al.*, 1997a; Kuehl *et al.*, 1991, 1994; Kannan *et al.*, 1988; McKenzie *et al.*, 1998). Green *et al.* (1996a) determined the concentrations of the non-*ortho* CBs, CB77, CB126 and CB169, in grey seal milk. The relative concentrations were dominated by CB126, followed by CB169. These authors suggested that although there is some concern about the use of TEFs to estimate the potential toxicity of CBs on the reproductive system of mammals (Battershill, 1994), they appear to be more reliable as an index of potential effects on the immune system because PCB- and PCDD/F-induced immunotoxicity is largely mediated by

the Ah receptor. The non-*ortho* congeners made up half of the total PCB-TEQ in the four samples for which there were also data on the concentrations of mono- and di-*ortho* congeners. The strong correlation between TEQ values estimated from the non-*ortho* congeners, and those from mono- and di-*ortho* congeners suggested that the concentrations of either group could provide a reliable index of the toxicity of PCBs in grey seal milk. This measure is valid as a comparative guide within a colony but is not directly transferable to other seal populations (Hall, 1998).

In McKenzie *et al.* (1998), the toxicities of the di-, mono- and non-*ortho*-substituted CBs, relative to TCDD, were calculated by multiplying the concentration of each congener by its toxic equivalency factor (TEF) based on *in vivo* and *in vitro* studies, as given by Ahlborg *et al.* (1994). The sum of the TEFs for the di-, mono-, and non-*ortho*-substituted CBs (TEQ-CB) are given for several species of marine mammals in Table A10.3 and range from 8.8–1150 pg g<sup>-1</sup> lipid weight (8.7–894 pg g<sup>-1</sup> wet weight), with the highest individual concentrations being measured in the blubber of Atlantic white-sided dolphins (*Lagenorhynchus acutus*) from the Scottish and Irish coasts.

In all species (with the exception of common seals), when all data are expressed on a lipid-weight basis, there was a significant linear relationship between TEQ-CB and the recalcitrant congener CB153, which is used as an indicator of the degree of contamination in each individual. The results of the regression are given in Table A10.3. A less significant correlation was observed between ΣCB and TEQ-CB, as noted previously by de Boer *et al.* (1993). The linear regression for all species is given in Figure A10.2 and shows an R<sup>2</sup> of 0.912 (p < 0.0005). The 95 % prediction intervals show that TEQ-CB may be predicted from CB153 concentrations within a factor of two at lower concentrations (2.5 µg g<sup>-1</sup> lipid CB153) and within a factor of 1.2 at higher concentrations (20.0 µg g<sup>-1</sup> lipid CB153).

The contribution of each congener to TEQ-CB shows that CB126, CB118, and CB156 are the most important toxic CB components in most species and that there is also a considerable amount of within-species variance for many of the species. Figures A10.3.a–A10.3.c show Atlantic white-sided dolphin data, for which data on all of the TCDD-like congeners are available for the largest number of individuals of one species. It can be observed that the within-species variance can be explained by changes in the relative contribution of the di-*ortho* congeners (CB170 and CB180) and two of the mono-*ortho* congeners (CB118 and CB156) to the TEQ-CB as a result of increasing CB burden (as described by logCB153). The contributions of the non-*ortho* CBs and the mono-*ortho* congener CB105 do not appear to change over the concentration range in this data set.

Before considering any possible log-linear relationships, it was noted that for CB118 in particular there appeared to be a sub-population within the data set. These animals were found to be lactating females, previously found to have CB patterns that differed from those of the other individuals due to the preferential transfer of lower chlorinated congeners during lactation (McKenzie *et al.*, 1997b). The data from these animals are circled in all the figures and they can be seen to increase the scatter of the data.

When these animals are removed from the regression calculations, there is a highly significant negative relationship between the contribution of the metabolizable CB118 to logCB153 and a significant positive relationship of the unmetabolizable congener CB170 (and CB180, not shown). On the basis of the structural requirements for metabolism proposed by Boon *et al.* (1997), CB156 would be expected, like CB118, to behave like a metabolizable congener. However, as for the unmetabolizable congeners, the proportion of TEQ-CB increases with increasing CB burden. This behaviour has been noted previously, with CB156 being reported to accumulate as a stable congener in another delphinid species, the common dolphin (Boon *et al.*, 1997). The changes in the importance of the mono- and di-*ortho* CBs can be explained by previous observations that the ratio of CB118/CB153 decreases with increasing burden and that the ratio of CB170/CB153 remains constant. As in most cases CB153 has been observed to be well correlated with age (in males), the relative toxic significance of CB118 will decrease over time while that of congeners such as CB156, CB170 and CB180 will increase.

However, the relevance of using the TEQ approach for determining the responses and assessing the risks for marine mammals has yet to be determined. Given the differences in metabolizing capabilities between species, the sensitivity of this approach should perhaps be viewed with some caution. This technique does not take into account the potential toxicity, dioxin-like or not, of hydroxylated or methylsulphonated metabolites. Some members of this class of compounds can elicit different responses via common or different mechanisms and it is evident that if each individual congener or class of compounds causes different responses and acts via independent mechanisms, the relative toxicities of every congener must be determined. With this in mind, the responses studied in laboratory animals and their cell lines may, however, be indicative of where the emphasis for studying the biological effects of non-*ortho* and mono-*ortho* CBs for marine mammals should be directed (Hall, 1998). However, it should again be mentioned that cell lines often develop specific characteristics.

Comparisons between, and changes in, the patterns of individual CB congeners in tissues such as blubber and muscle in species at various levels of the food chain have been used to infer the metabolic capacity of animals at higher trophic levels. Contaminants present in the prey, but not found in the predators, are assumed to be metabolized. This approach identifies the capabilities of different species and genera to deal with xenobiotics and allows the potential toxic effects to be assessed based on the different abilities of species to biotransform different CB congeners. By comparing the ratio of each congener to a persistent congener (such as CB153 or CB180, which are found at consistently high levels in marine

**Table A10.3.** Median (and range) of TEQ-CB concentrations in marine mammals from Scottish and Irish waters.

Species	n	% extractable lipid in blubber	TEQ-CB <sup>A</sup> (pg g <sup>-1</sup> lipid)	TEQ-CB <sup>B</sup> (pg g <sup>-1</sup> lipid)	R <sup>2</sup>	p
Harbour porpoise	12	73.4 (38.6–79.5)	53.6 (15.6–197)	62.7 (16.1–235)	0.94	< 0.0005
Bottlenose dolphin	9	54.8 (42.8–69.9)	296 (36.3–605)	305 (40.4–878)	0.93	< 0.0005
White-beaked dolphin	8	79.3 (23.0–85.1)	212 (94.1–569)	306 (143–801)	0.85	< 0.001
Atlantic white-sided dolphin	22	67.4 (33.5–86.5)	150 (20.5–690)	209 (31.7–1150)	0.98	< 0.0005
Striped dolphin	6	62.2 (50.2–69.7)	158 (115–434)	215 (140–627)	0.99	< 0.0005
Risso's dolphin	3	64.0 (57.5–69.8)	60.8 (40.3–182)	63.6 (41.2–225)	na	na
Common dolphin	2	(74.8–83.9)	(28.7–98.8)	(37.3–121)	na	na
Killer whale	2	(53.0–63.0)	(202–382)	(289–562)	na	na
Long-finned pilot whale	4	70.3 (58.2–77.4)	233 (220–259)	283 (265–324)	na	na
Sowerby's beaked whale	4	70.9 (69.5–72.8)	129 (96.5–137)	161 (126–166)	na	na
Grey seal	9	78.0 (68.6–80.7)	26.6 (14.5–64.1)	67.9 (34.8–273)	0.98	< 0.0005
Common seal	7	84.0 (67.7–98.5)	16.2 (7.54–28.3)	22.7 (8.62–32.8)	0.45	< 0.107

TEQ-CB<sup>A</sup> = (TEFCB77 + TEFCB105 + TEFCB118 + TEFCB126 + TEFCB156 + TEFCB169)

TEQ-CB<sup>B</sup> = (TEFCB77 + TEFCB105 + TEFCB118 + TEFCB126 + TEFCB156 + TEFCB169 + TEFCB170 + TEFCB180)

R<sup>2</sup>: Pearson correlation coefficient for the equation TEQ-CB<sup>B</sup> = gradient [CB153 (µg g<sup>-1</sup> lipid)] + constant

na: not available.

Figure A10.2. TEQ-CB as a function of CB153 concentrations in marine mammals from Scottish and Irish waters.

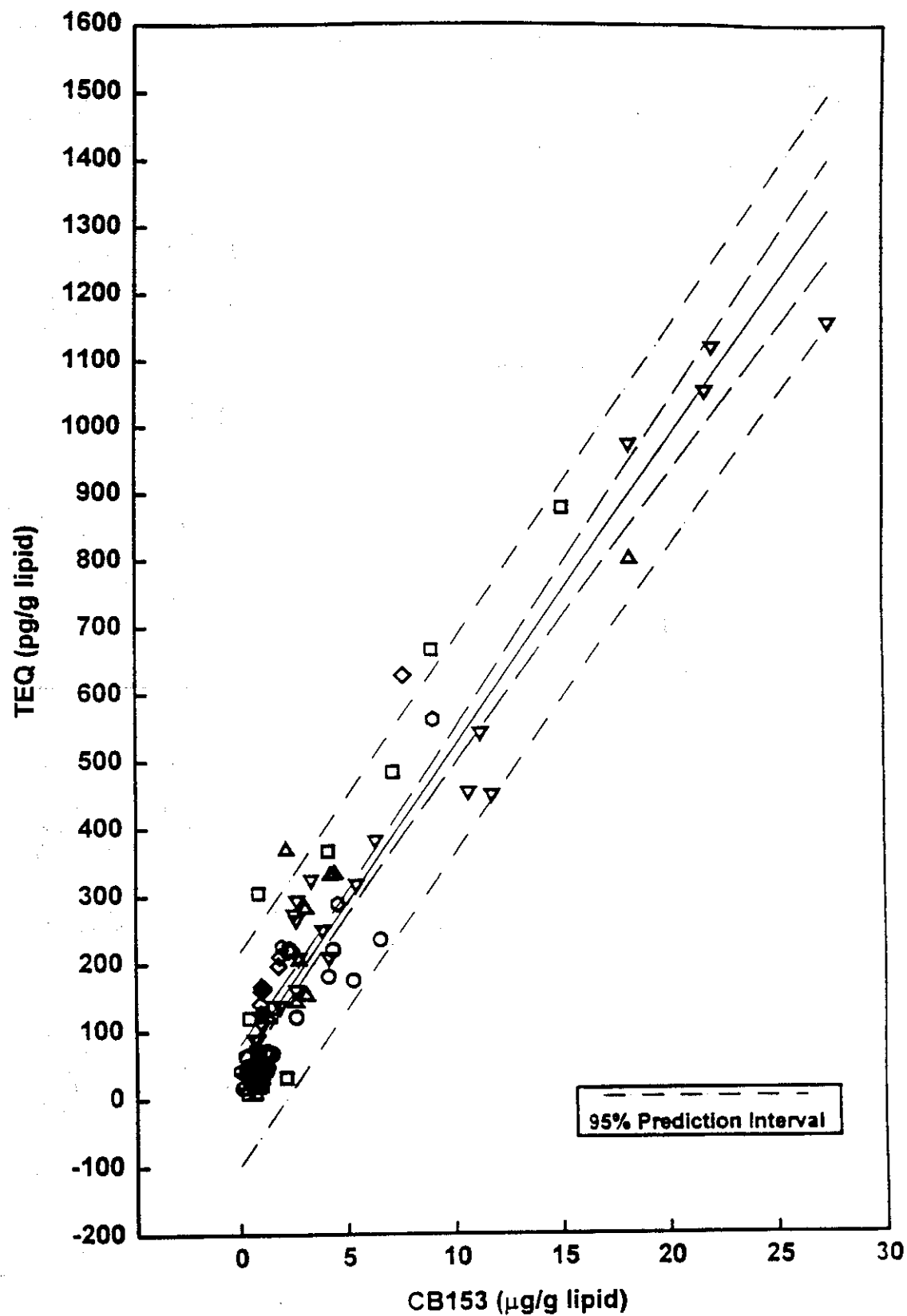




Figure A10.3.a. Changes in the contribution of mono-*ortho* CB118 to TEQ-CB in relation to contaminant burden.

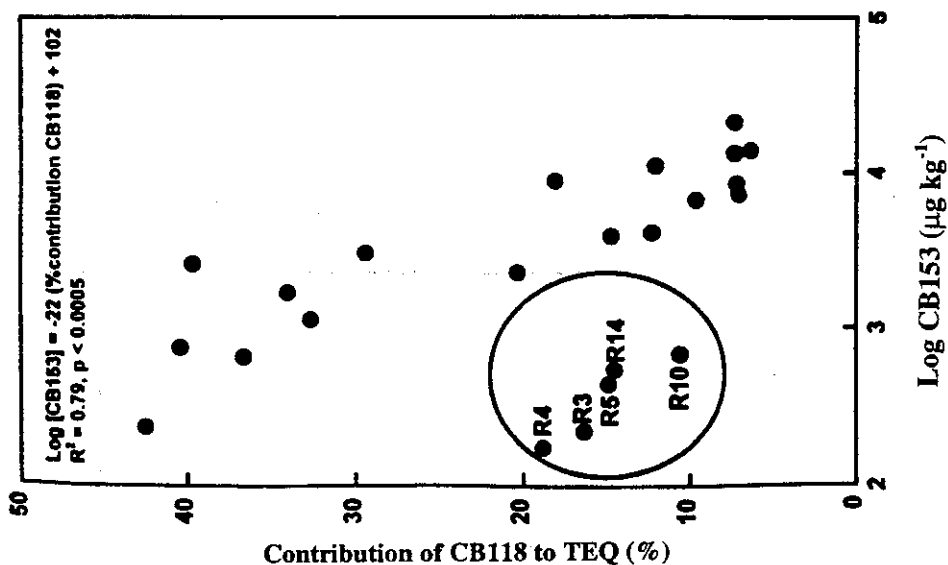


Figure A10.3.b. Changes in the contribution of mono-*ortho* CB156 to TEQ-CB in relation to contaminant burden.

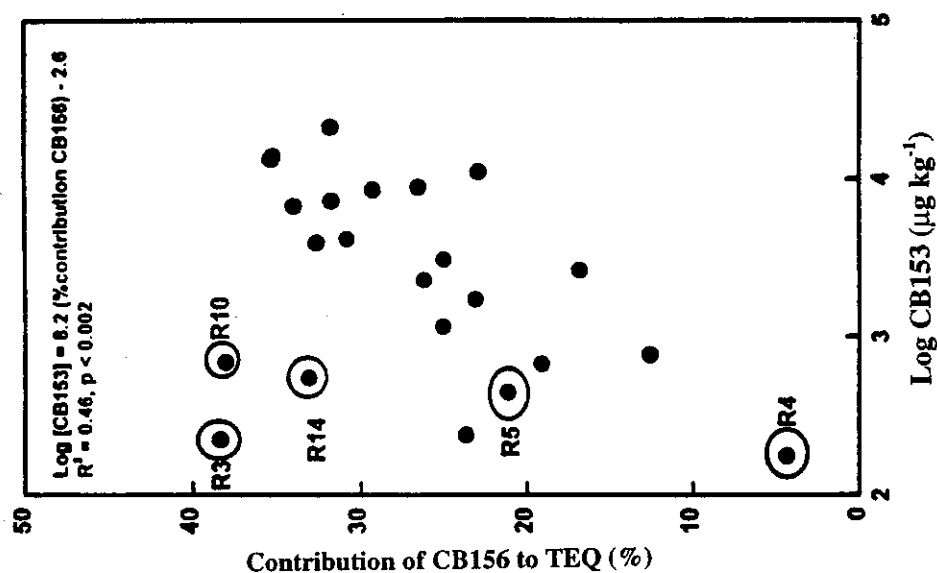
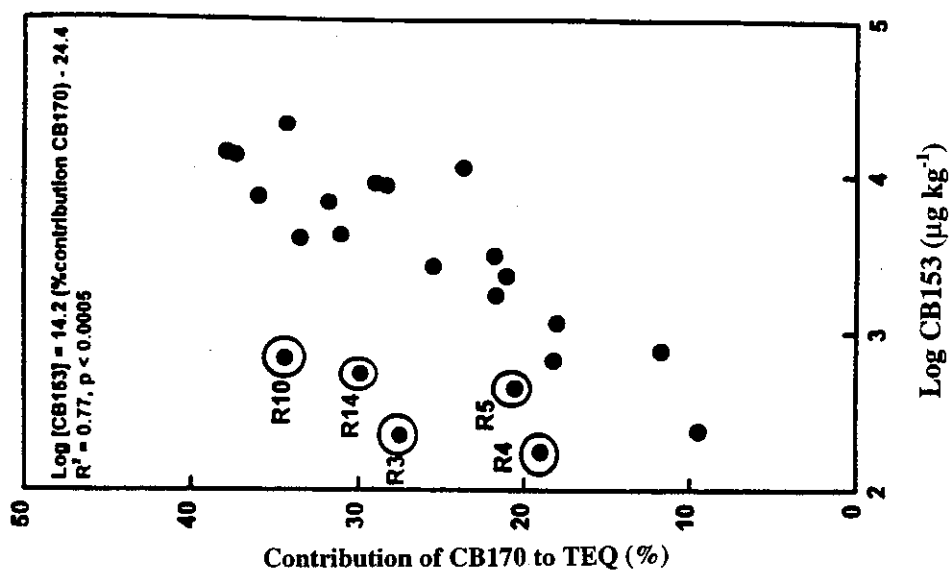


Figure A10.3.c. Changes in the contribution of mono-*ortho* CB170 to TEQ-CB in relation to contaminant burden.



biota), patterns and concentrations can be standardized making them directly comparable. It is from these indirect studies that some of the conclusions about the metabolic capabilities of different marine mammal species have come (Tanabe *et al.*, 1987; Boon *et al.*, 1994, 1997; Wells *et al.*, 1996). However, metabolism may transform CBs into intermediate, but more toxic compounds than the parent compounds.

Using this technique, various authors have found that biotransformation occurs generally in the order: seals > small cetaceans > large cetaceans > fish (Duinker *et al.*, 1989). Tanabe *et al.* (1988) found that, in contrast to seals, which possess both P4501A and P4502B genes, cetaceans were unable to metabolize all congeners with *m,p*-vicinal H atoms (Groups IV and V in Table A10.2), although there is direct evidence that beluga whales (*Delphinapterus leucas*) possess CYP2B enzymes which are capable of metabolizing these types of compounds (White *et al.*, 1994) and indirect evidence from CB patterns that harbour porpoise are able to metabolize some (CB44 and CB101) (Bruhn *et al.*, 1995; Wells *et al.*, 1996; Boon *et al.*, 1997). However, this approach is of limited use in determining the potential effects of the more important mono-*ortho* and non-*ortho* congeners. Boon *et al.* (1994) noted that since the three important mono-*ortho*-substituted chlorobiphenyl congeners (CB105, CB118, and CB156) with a mixed CYP1A/CYP2B induction pattern were among the metabolizable congeners in harbour porpoise samples and the *in vitro* phase I biotransformation of CB77, a pure CYP1A inducer, to hydroxylated metabolites was demonstrated in microsomal samples of a harbour seal and a porpoise, it did not appear feasible to derive dioxin-type 'toxic equivalents' in marine mammals by assuming constant concentration ratios between chlorobiphenyl congeners with a dioxin-type toxicity and persistent congeners occurring in much higher concentrations (such as CB153), as proposed by de Boer *et al.* (1993) for fish and shellfish. It should be noted, however, that CB156 behaves as a persistent congener in common dolphin and white-sided dolphin (Boon *et al.*, 1997; McKenzie *et al.*, 1998). There is a strong positive correlation between TEQ and CB153; however, the proportion of toxicity derived from at least two of the mono-*ortho* congeners decreases with increasing CB153 concentrations, while the toxicity derived from CB156 and the persistent di-*ortho* congeners increases (McKenzie *et al.*, 1998; van Scheppeningen *et al.*, 1996).

#### 4 CONCENTRATIONS OF NON-ORTHO AND MONO-ORTHO CBs IN MARINE MAMMALS

Data on the concentrations of non-*ortho* and mono-*ortho* CBs have been collated from a number of laboratories in the ICES area, from the ICES data bank, from McKenzie *et al.* (1998), and from other published sources. The following presentation of data is taken

from McKenzie *et al.* (1998), who analysed 131 blubber samples from twelve species of marine mammals for CBs, with 88 of these samples also analysed for planar CBs.

A summary of the non-*ortho* CB concentrations determined in 88 marine mammal blubber samples (McKenzie *et al.*, 1998) is given in Table A10.4. In general, the concentrations measured are of the same order as those measured in harbour porpoise (*Phocoena phocoena*) and common seals (*Phoca vitulina*) stranded on the Dutch and German coasts and in the Bay of Gdansk on the Polish coast (Beck *et al.*, 1990; Falandysz *et al.*, 1994; van Scheppeningen *et al.*, 1996) and previously sampled animals from the Scottish coast (Wells and Echarri, 1992). Considerably higher concentrations of non-*ortho* CBs have previously been determined in marine mammals from the Dutch coast and the Mediterranean Sea. De Boer *et al.* (1993) have reported levels, for example, of CB77 of 10 ng g<sup>-1</sup> and 28 ng g<sup>-1</sup> wet weight in the blubber of a harbour porpoise and a white-beaked dolphin (*Lagenorhynchus albirostris*), respectively, while concentrations of the same congener in striped dolphins (*Stenella coeruleoalba*) from the Mediterranean Sea ranged from 16–85 ng g<sup>-1</sup> wet weight (Kannan *et al.*, 1993). The concentrations of CB77 and CB169 measured by McKenzie *et al.* (1998) are, however, generally higher than those reported in the blubber of Hector's dolphin (*Cephalorhynchus hectori*), a species endemic to the coastal waters of New Zealand, and the more pelagic common dolphin (*Delphinus delphis*), although the concentrations of CB126, the potentially most toxic congener, are similar in the Hector's dolphin samples to those determined in this study (Jones *et al.*, 1999).

The concentrations of ΣCBs in individual marine mammals stranded on the Scottish coast ranged from 0.27–67.9 µg g<sup>-1</sup>, a wide range reflecting the diverse sample set. In the majority of species, where both male and female samples were available, males had higher median concentrations than females. As observed in nearly all marine mammal species analysed, chlorobiphenyl compounds without vicinal hydrogen atoms or with vicinal hydrogen atoms in the *ortho* and *meta* positions in combination with two or more *ortho*-chlorine atoms (CB138, CB153, CB170, CB180) are present in the highest concentrations, due to their metabolic stability and their high proportions in a number of industrial mixtures (Schulz *et al.*, 1989). An extensive study has previously been carried out using multivariate techniques to study chlorobiphenyl patterns (ratios of individual congeners to the recalcitrant CB153) in animals from eleven of the twelve species reported here. The majority of species could be differentiated from one another, with the patterns being influenced by the total contaminant burden, diet, and metabolic capacity. The ability to metabolize CBs in different species was found to be of the order pinnipeds > harbour porpoise > others > sperm whales (Wells *et al.*, 1996).

**Table A10.4.** Concentrations of non-*ortho* CBs,  $\Sigma$ CB, and  $\Sigma$ DDT in marine mammals stranded on the Scottish coast, given as the median, with the range in parentheses, on a wet weight basis (from McKenzie *et al.* (1998)).

Species	Sex	N	$\Sigma$ CB ( $\mu\text{g g}^{-1}$ )	CB77 ( $\text{ng g}^{-1}$ )	CB126 ( $\text{ng g}^{-1}$ )	CB169 ( $\text{ng g}^{-1}$ )	$\Sigma$ DDT ( $\mu\text{g g}^{-1}$ )
Harbour porpoise	M	7	3.03 (0.58–9.15)	0.80 (0.22–1.22)	0.10 (0.06–0.14)	0.07 (0.01–0.13)	1.96 (0.44–5.69)
Harbour porpoise	F	15	2.42 (0.40–11.3)	0.19 (0.02–6.49)	0.05 (0.01–0.07)	0.08 (0.01–3.33)	1.58 (0.19–7.34)
Bottlenose dolphin	M	7	8.63 (0.48–30.9)	0.18 (0.10–0.36)	0.32 (0.07–0.91)	0.12 (0.08–0.67)	5.37 (0.40–12.5)
Bottlenose dolphin	F	6	4.12 (0.73–26.3)	0.18 (0.11–0.30)	0.29 (0.07–0.51)	0.20 (0.11–0.61)	3.31 (0.54–11.5)
White-beaked dolphin	M	6	8.81 (4.19–27.4)	0.47 (0.02–1.81)	0.07 (0.03–0.58)	0.82 (0.01–2.43)	7.95 (4.52–11.0)
White-beaked dolphin	F	4	9.00 (7.31–54.4)	0.48 (0.03–1.27)	0.17 (0.01–1.43)	0.49 (0.03–1.60)	7.17 (6.02–31.9)
White-sided dolphin	M	12	25.3 (3.57–67.9)	0.44 (0.08–2.60)	0.09 (0.04–5.99)	0.14 (0.04–5.99)	13.8 (3.41–53.4)
White-sided dolphin	F	11	2.89 (0.85–15.7)	0.12 (0.02–0.31)	0.04 (0.01–0.19)	0.10 (0.01–0.26)	1.92 (0.20–6.34)
Striped dolphin	M	6	6.16 (5.63–20.7)	0.28 (0.04–3.30)	0.03 (0.01–0.12)	0.79 (0.33–5.99)	6.31 (4.12–14.2)
Striped dolphin	F	1	2.24	0.37	0.21	0.34	2.49
Risso's dolphin	M	1	4.74	0.61	0.21	0.74	2.07
Risso's dolphin	F	2	(0.31–0.66)	(0.32–0.97)	(0.18–0.29)	(0.10–0.26)	(0.10–0.30)
Common dolphin	M	1	4.52	0.08	0.17	0.08	4.12
Common dolphin	F	2	(1.06–3.02)	0.08	0.05	0.12	(0.49–2.74)
Long-finned pilot whale	M	4	8.53 (6.16–10.3)	0.49 (0.28–0.49)	0.23 (0.22–0.31)	0.85 (0.61–1.04)	7.89 (7.09–14.1)
Killer whale	M	1	11.0	0.68	0.06	0.14	8.48
Killer whale	F	1	22.5	0.71	0.02	0.93	17.8
Minke whale	M	1	0.27	0.06	0.07	0.03	0.27
Sowerby's beaked whale	M	1	12.1	na	na	na	7.83
Sowerby's beaked whale	F	5	3.20 (3.10–3.33)	0.12 (0.09–0.21)	0.41 (0.18–0.45)	0.15 (0.12–0.23)	2.14 (1.89–2.82)
Grey seal	M	14	2.52 (1.65–10.8)	0.18 (0.07–0.32)	0.10 (0.08–0.14)	0.08 (0.08–0.08)	1.17 (0.82–3.60)
Grey seal	F	6	2.63 (1.46–8.87)	0.07 (0.03–0.07)	0.08 (0.06–0.08)	0.38 (0.01–0.86)	1.15 (0.79–2.62)
Common seal	M	2	(2.14–2.56)	0.06	0.05	0.03	(0.75–1.31)
Common seal	F	7	1.57 (0.37–4.26)	0.17 (0.04–1.10)	0.05 (0.01–0.40)	0.02 (0.01–0.06)	0.50 (0.23–1.60)

In addition to the above information, data on  $\Sigma$ CB and  $\Sigma$ DDT concentrations in cetacean and pinniped species sampled world wide are given in Table A10.5.

## 5 EFFECTS OF CONTAMINANTS ON MARINE MAMMALS

### 5.1 General information

The most commonly reported responses to CB exposure, largely based on data from laboratory studies, indicate that a range of body systems may be affected (Hall, 1998). Observed effects include body weight loss, thymic atrophy, impairment of immune responses, hepatotoxicity and porphyria (defect in haemoglobin synthesis), chloracne and related dermal lesions, tissue-specific hypoplastic and hyperplastic responses, carcinogenesis, teratogenicity, and reproductive toxicity (Amdur *et al.*, 1991). Hallmarks of exposure are the induction of both phase I and phase II drug metabolizing enzymes, and it is the induction of the phase I P450 enzymes in particular (recently reported in various cetacean and seal species) in conjunction with higher CB uptake which increases the concern about the toxicological importance of the non-*ortho* and mono-*ortho* CB congeners for vulnerable and threatened marine mammal species (Norstrom *et al.*, 1992; Stegeman and Hahn, 1994; White *et al.*, 1994).

Genetic studies have shown that in-bred mice and their back-crosses exhibit a variety of toxic responses to CB exposure, including immunotoxicity, teratogenicity, body weight loss, hepatotoxicity and porphyria. These responses appear to segregate with the Ah locus (Poland and Glover, 1986), that is, the responsiveness of a particular organ or cell to the effects of 2,3,7,8-TCDD and related compounds depends on the presence of a functional Ah receptor. Cells with low levels of this receptor are not induced into increasing the activity of cytochrome P450s, although the presence of the Ah receptor alone is not sufficient for the induction of a biochemical response. Several groups have investigated the molecular biology of CYP1A1 gene expression and their results confirm the proposed receptor-mediated mechanism of action of 2,3,7,8-TCDD and related compounds (Whitlock, 1987; Poland and Knutson, 1982). Initial binding of the toxin to the cytosolic Ah receptor is followed by an activation or transformation step and the subsequent accumulation of occupied nuclear receptor complexes. These nuclear complexes then interact with specific DNA sequences which are located in the 5'-upstream region from the CYP1A1 gene (Safe, 1990). These interactions lead to the enhancement of CYP1A1 gene expression. It is assumed that many of the toxic effects elicited by halogenated aryl hydrocarbons are also the result of altered receptor-mediated gene expression. However, the molecular mechanisms of these responses are currently unknown (Stegeman and Hahn, 1994). There is, therefore, a close relationship between the actions of PCBs on mammalian systems and the physicochemical properties inherent in

the molecular structure of PCBs. These properties determine the molecular reactivities of PCBs and are responsible for their recognition at biological acceptors and receptors, as well as for triggering molecular mechanisms that lead to tissue response and possible damage.

Battershill (1994) reviewed the toxicity data for individual CB congeners in mammals and these are shown in Table A10.6. Generally, the non-*ortho* congeners were more toxic to most systems investigated than the mono-*ortho* congeners, with variability between congeners and between species exposed.

It has been suggested that marine mammals have a smaller capacity of either PB-inducible or MC-inducible enzymes (CYP1A and CYP2B) than terrestrial mammals (Tanabe *et al.*, 1988; Kannan *et al.*, 1989). Because their ability to metabolize the contaminants may be reduced as a result, they are potentially more vulnerable to the toxic effects of PCBs in general, and to the planar CBs in particular. It is clear that further genetic studies of the Ah locus and cytochrome gene expression in seals and cetaceans are required to establish the true nature and variation in the ability of marine mammals to deal with non-*ortho* and mono-*ortho* CB congeners.

Many studies have measured the concentrations of mono- and non-*ortho* CBs in marine mammal tissues (Tanabe *et al.*, 1987; Green *et al.*, 1996a; Oehme *et al.*, 1995a, 1995b; Nakata *et al.*, 1995; Koistinen *et al.*, 1997; Kuehl *et al.*, 1994; Lake *et al.*, 1995; Corsolini *et al.*, 1995; McKenzie *et al.*, 1997, 1998), but little information exists on their specific effects. Little is also known about the effects of heavy metals, although increasing levels of heavy metals have been recorded in some areas (Wagemann *et al.*, 1990; Borrell and Reijnders, 1999).

### 5.2 Acute Lethal Effects

No acute lethal effects of contaminants on marine mammals have been reported, except for seals being acutely poisoned by an accidental discharge of mercury-contaminated disinfectant (Koeman *et al.*, 1971), and immediate intoxication and thermal imbalance of marine mammals following oil spills (Frost and Lowry, 1993; Geraci and St. Aubin, 1990; IWC, 1995).

### 5.3 Sub-lethal Effects

#### 5.3.1 Effects on reproduction and early development

Female marine mammals are able to eliminate lipophilic contaminants via reproductive transfer. Lactational transfer is the most significant route in bottlenose, Atlantic white-sided and striped dolphins, with 4–7 % of the mother's burden transferred to the calf during weaning (Tanabe *et al.*, 1981; Salata *et al.*, 1995;

**Table A10.5.** Data on  $\Sigma$ CB and  $\Sigma$ DDT concentrations in cetacean and pinniped species sampled world-wide (from McKenzie *et al.*, 1998).

Species	$\Sigma$ CB ( $\mu\text{g g}^{-1}$ )	$\Sigma$ DDT ( $\mu\text{g g}^{-1}$ )	Location	Sampling date	Reference
Harbour porpoise	3.0 (0.6–9.2) <sup>M</sup>	2.0 (0.44–5.7) <sup>M</sup>	Scotland	1992	McKenzie <i>et al.</i> (1998)
	2.4 (0.41–11.3) <sup>F</sup>	1.6 (0.19–7.3) <sup>F</sup>	Scotland	1992	McKenzie <i>et al.</i> (1998)
	23.1 (16.2–58.1) <sup>A</sup>	15.4 (5.9–20.4)	Scotland	1972	FRS <sup>1</sup> data
	29.7 (11.3–47.2) <sup>MA</sup>	13.3 (6.5–25.3) <sup>M</sup>	Scotland	1974	FRS <sup>1</sup> data
	21.7 (10.6–31.4) <sup>FA</sup>	8.4 (4.3–11.4) <sup>F</sup>	Scotland	1974	FRS <sup>1</sup> data
	31.3 ( $\pm 7.9$ )		Baltic Sea	1989–1990	Falandysz <i>et al.</i> (1994)
	55.6 ( $\pm 22.7$ )		Wales	1988	Morris <i>et al.</i> (1989)
	13.1 (0.47–36.2) <sup>MA</sup>	4.0 (0.64–10.4) <sup>M</sup>	Scotland	1988–1991	Wells <i>et al.</i> (1994)
	4.6 (1.9–22.4) <sup>FA</sup>	1.8 (0.64–7.3) <sup>F</sup>	Scotland	1988–1991	Wells <i>et al.</i> (1994)
	5.1 (0.12–13.0)	2.2 (0.04–5.1) <sup>O</sup>	Scotland (Shetland)	1989–1991	Kuiken <i>et al.</i> (1994)
	23.3 (3.7–65.3) <sup>*</sup>	16.4 (3.2–45.1) <sup>*</sup>	Scandinavia	1987–1991	Kleivane <i>et al.</i> (1995)
	13.4 ( $\pm 2.4$ ) <sup>*M</sup>	5.6 ( $\pm 0.73$ ) <sup>*M</sup>	Faroe Islands	1987–1988	Borrell (1993)
	8.8 ( $\pm 1.1$ ) <sup>*F</sup>	3.8 ( $\pm 0.38$ ) <sup>*F</sup>	Faroe Islands	1988	Borrell (1993)
	22.7 (13–33)	9.1 (8.2–12)	NW Atlantic	1991	Stein <i>et al.</i> (1992)
	5.2 (1.8–10.6) <sup>M</sup>	4.1 (1.4–7.3) <sup>M</sup>	Newfoundland	1989–1991	Westgate <i>et al.</i> (1997)
	5.5 (1.4–14.2) <sup>F</sup>	3.1 (1.0–7.6) <sup>F</sup>	Newfoundland	1989–1991	Westgate <i>et al.</i> (1997)
	8.4 (4.4–16.0)	8.2 (3.1–22.0)	NW USA	1986–1989	Jarman <i>et al.</i> (1996)
	10.0 (3.3–31.0)		California	1986–1989	Jarman <i>et al.</i> (1996)
	16.0 (5.0–39.0) <sup>M</sup>	70.0 (25.0–180) <sup>M</sup>	Black Sea	1993	Tanabe <i>et al.</i> (1997b)
	12.0 (1.6–29.0) <sup>F</sup>	50.0 (8.3–83.0) <sup>F</sup>	Black Sea	1993	Tanabe <i>et al.</i> (1997b)
	6.5 (3.1–16) <sup>M</sup>	4.7 (1.9–12.0) <sup>M</sup>	Japan	1993	Tanabe <i>et al.</i> (1997b)

<sup>1</sup>Fisheries Research Services, The Marine Laboratory, Aberdeen, UK. \*Lipid weight; \*\* $\Sigma$ DDT =  $p,p'$ -DDE +  $p,p'$ -DDD +  $p,p'$ -DDT; <sup>O</sup> $p,p'$ -DDE only; <sup>A</sup>Arochlor 1254 equivalent; <sup>M</sup>males only; <sup>F</sup>females only; ( ) = range or  $\pm$  standard deviation.

Table A10.5. Continued.

Species	$\Sigma\text{CB}$ ( $\mu\text{g g}^{-1}$ )	$\Sigma\text{DDT}$ ( $\mu\text{g g}^{-1}$ )	Location	Sampling date	Reference
Bottlenose dolphin	8.6 (0.48–31) <sup>M</sup>	5.4 (0.40–12.4) <sup>M</sup>	Scotland	1992–1996	McKenzie <i>et al.</i> (1998)
	4.1 (0.72–26) <sup>F</sup>	3.3 (0.54–11.5) <sup>F</sup>	Scotland	1992–1996	McKenzie <i>et al.</i> (1998)
	13.6 <sup>MA</sup>	44.3 <sup>M</sup>	Scotland	1976	FRS <sup>1</sup> data
	16.0 (6.0–67.2) <sup>MA</sup>	38.0 (0.45–85.6) <sup>M</sup>	South Africa	1986–1987	Cockroft (1989)
	2.6 (0.09–43.0) <sup>FA</sup>	1.3 (0.04–49.7) <sup>F</sup>	South Africa	1986–1987	Cockroft (1989)
	36.1 (4.1–149) <sup>*</sup>	15.3 (0.43–74.6) <sup>*</sup>	Gulf of Mexico	N/A	Salata <i>et al.</i> (1995)
	584 ± 456	170 (± 190)	Mediterranean Sea	1992	Corsolini <i>et al.</i> (1995)
	310	145 <sup>**</sup>	Wales	1988	Morris <i>et al.</i> (1989)
	12.1 (2.0–20.7)		Scotland	1988–1991	Wells <i>et al.</i> (1994)
	93 (64–187) <sup>M*</sup>		Gulf of Mexico	1990	Kuehl and Haebler (1995)
	7.2 (1.5–18) <sup>F*</sup>		Gulf of Mexico	1990	Kuehl and Haebler (1995)
	0.52 (0.37–0.67)	7.25 (2.1–14.0)	India	1990–1991	Tanabe <i>et al.</i> (1993)
Striped dolphin	6.2 (5.6–20.7) <sup>M</sup>	6.3 (4.1–14.2) <sup>M</sup>	Scotland	1992–1994	McKenzie <i>et al.</i> (1998)
	2.2 <sup>F</sup>	2.5 <sup>F</sup>	Scotland	1992–1994	McKenzie <i>et al.</i> (1998)
	28 (17–38)	37 (23–51)	Japan	1986	Loganathan <i>et al.</i> (1990)
	393 ± 202	139 ± 84	Mediterranean Sea	1990	Kannan <i>et al.</i> (1993)
	21.5	49.0	Wales	1988	Morris <i>et al.</i> (1989)
	46.9–86	23.6–63.5	Mediterranean Sea	1990–1993	Marsili and Focardi (1996)
	16.8	15.4	Aegean Sea	1991	Georgakopoulou <i>et al.</i> (1995)
White-beaked dolphin	8.8 (4.2–27.3) <sup>M</sup>	8.0 (4.5–11.0) <sup>M</sup>	Scotland	1992–1994	McKenzie <i>et al.</i> (1998)
	9.0 (7.3–54.4) <sup>F</sup>	7.1 (6.0–32.0) <sup>F</sup>	Scotland	1992–1994	McKenzie <i>et al.</i> (1998)
	34.2 (13.4–87.0) <sup>M</sup>	43.4 (4.5–86.8) <sup>M</sup>	Canada	1980	Muir <i>et al.</i> (1988a)
	22.0 (9.6–39.7) <sup>F</sup>	27.9 (0.8–88.6) <sup>F</sup>	Canada	1980	Muir <i>et al.</i> (1988a)

<sup>1</sup>Fisheries Research Services, The Marine Laboratory, Aberdeen, UK. \*Lipid weight; \*\* $\Sigma\text{DDT} = p,p'\text{-DDE} + p,p'\text{-DDD} + p,p'\text{-DDT}$ ; <sup>o</sup> $p,p'\text{-DDE}$  only; <sup>A</sup>Arochlor 1254 equivalent; <sup>M</sup>males only; <sup>F</sup>females only; ( ) = range or ± standard deviation.

Table A10.5. Continued.

Species	$\Sigma\text{CB}$ ( $\mu\text{g g}^{-1}$ )	$\Sigma\text{DDT}$ ( $\mu\text{g g}^{-1}$ )	Location	Sampling date	Reference
White-sided dolphin	29.3 (3.6–29.8) <sup>M</sup>	33.2 (7.2–53.4) <sup>M</sup>	Scotland	1992–1994	McKenzie <i>et al.</i> (1998)
	6.2 (3.1–9.2) <sup>F</sup>	5.5 (4.6–6.3) <sup>F</sup>	Scotland	1992–1994	McKenzie <i>et al.</i> (1998)
	0.77–63	0.16–54.6	West of Ireland	1994	McKenzie <i>et al.</i> (1997)
	42.7 ( $\pm 18.0$ ) <sup>*M</sup>	22.5 ( $\pm 11.7$ ) <sup>*M</sup>	Faroe Islands	1987	Borrell (1993)
	25.3 ( $\pm 21.2$ ) <sup>*F</sup>	15.0 ( $\pm 14.4$ ) <sup>*F</sup>	Faroe Islands	1987	Borrell (1993)
Common dolphin	4.5 <sup>M</sup>	4.1 <sup>M</sup>	Scotland	1993	McKenzie <i>et al.</i> (1998)
	(1.1–3.0) <sup>F</sup>	(0.49–2.7) <sup>F</sup>	Scotland	1992–1994	McKenzie <i>et al.</i> (1998)
	136 <sup>A</sup>	44.3	Scotland	1976	FRS <sup>1</sup> data
	36.5 (31.2–40.6)	14.4 (5.9–26) <sup>O</sup>	Eastern USA	1987–1988	Kuehl <i>et al.</i> (1991)
Risso's dolphin	4.7 <sup>M</sup>	2.1 <sup>M</sup>	Scotland	1992	McKenzie <i>et al.</i> (1998)
	(0.31–6.6) <sup>F</sup>	(0.10–0.29) <sup>F</sup>	Scotland	1992	McKenzie <i>et al.</i> (1998)
	320 (20–610)	200 (5.2–400)	Mediterranean Sea	1992	Corsolini <i>et al.</i> (1995)
	43 (7.4–120) <sup>M</sup>	29 (3.2–77) <sup>M</sup>	Japan	1991	Kim <i>et al.</i> (1996)
	7.0 (1.7–20) <sup>F</sup>	4.2 (0.45–19) <sup>F</sup>	Japan	1991	Kim <i>et al.</i> (1996)
	1.7 <sup>M</sup>	5.0 <sup>M</sup>	British Columbia	1988	Jarman <i>et al.</i> (1996)
Long-finned pilot whale	8.5 (6.2–10.3) <sup>M</sup>	7.9 (7.1–14.1) <sup>M</sup>	Scotland	1992	McKenzie <i>et al.</i> (1998)
	1.7 ( $\pm 0.22$ )	7.6 ( $\pm 1.0$ )	Eastern USA		Varanasi <i>et al.</i> (1993)
	48.8 ( $\pm 23.1$ ) <sup>*M</sup>	31.4 ( $\pm 19.2$ ) <sup>*M</sup>	Faroe Islands	1987	Borrell (1993)
	26.3 ( $\pm 23.1$ ) <sup>*F</sup>	26.3 ( $\pm 23.1$ ) <sup>*F</sup>	Faroe Islands	1987	Borrell (1993)
	9.0 ( $\pm 3.8$ )	11.9 ( $\pm 6.1$ )	Eastern Canada		Muir <i>et al.</i> (1988b)
	3.5 ( $\pm 3.3$ )	4.7 ( $\pm 5.3$ )	Eastern Canada		Muir <i>et al.</i> (1988b)
Minke whale	0.27 <sup>M</sup>	0.27 <sup>M</sup>	Scotland	1993	McKenzie <i>et al.</i> (1998)
	0.003–0.029	0.01–0.14	Antarctica		Tanabe <i>et al.</i> (1986)
	3.7	5.5	Eastern USA		Varanasi <i>et al.</i> (1993)
	1.85 (1.51–2.11) <sup>*</sup>	1.01 (0.99–1.07) <sup>*</sup>	Canada	1992	Gauthier <i>et al.</i> (1996)

<sup>1</sup>Fisheries Research Services, The Marine Laboratory, Aberdeen, UK. \*Lipid weight; \*\* $\Sigma\text{DDT} = p,p'\text{-DDE} + p,p'\text{-DDD} + p,p'\text{-DDT}$ ; <sup>O</sup> $p,p'\text{-DDE}$  only;

<sup>A</sup>Arochlor 1254 equivalent; <sup>M</sup>males only; <sup>F</sup>females only; ( ) = range or  $\pm$  standard deviation.

Table A10.5. Continued.

Species	$\Sigma\text{CB}$ ( $\mu\text{g g}^{-1}$ )	$\Sigma\text{DDT}$ ( $\mu\text{g g}^{-1}$ )	Location	Sampling date	Reference
Killer whale	11 <sup>M</sup>	8.5 <sup>M</sup>	Scotland	1994	McKenzie <i>et al.</i> (1998)
	23 <sup>F</sup>	17.8 <sup>F</sup>	Scotland	1994	McKenzie <i>et al.</i> (1998)
	160		Japan	1982	Tanabe <i>et al.</i> (1987)
	22 (8.6–56.0)	32 (11.0–93.0)	NW USA	1997	Jarman <i>et al.</i> (1996)
	16.4 <sup>F</sup>	24.5 <sup>F</sup>	SE England	1997	Law <i>et al.</i> (1997)
Grey seal	2.5 (1.6–10.8) <sup>M</sup>	1.2 (0.82–3.6) <sup>M</sup>	Scotland	1988–1990	McKenzie <i>et al.</i> (1998)
	2.6 (1.5–8.7) <sup>F</sup>	1.1 (0.79–2.6) <sup>F</sup>	Scotland	1988–1990	McKenzie <i>et al.</i> (1998)
	16.2 ( $\pm 6.80$ ) <sup>F*</sup>	2.39 ( $\pm 0.83$ ) <sup>F*</sup>	E Canada	1984	Addison and Brodie (1987)
	30.3 ( $\pm 17.0$ ) <sup>F*</sup>	3.71 ( $\pm 1.41$ ) <sup>F*</sup>	E Canada	1985	Addison and Brodie (1987)
	18 (5.7–28)	4.2 (0.99–6.21)	East England	1988	Law <i>et al.</i> (1989)
	77 (61–89)	30 (22–35)	Baltic Sea	1988	Blomkvist <i>et al.</i> (1992)
	82 (32–110)	48 (19–91)	Baltic Sea	1988	Blomkvist <i>et al.</i> (1992)
	(5.0–11.0)		Scotland	1993	Green <i>et al.</i> (1996b)
Common seal	(2.1–2.6) <sup>M</sup>	(0.75–1.3) <sup>M</sup>	Scotland	1991–1994	McKenzie <i>et al.</i> (1998)
	1.6 (0.37–4.3) <sup>F</sup>	0.50 (0.23–1.6) <sup>F</sup>	Scotland	1991–1994	McKenzie <i>et al.</i> (1998)
	12 (5.0–19) <sup>A</sup>	4.5 (1.8–8.6)	Scotland (Mull)	1969	Holden (1978)
	15 (3.4–29)		South Norway	1988	Bernhoft and Skaare (1994)
	23 (6.7–33)	4.7 (1.02–8.00)	East England	1988	Law <i>et al.</i> (1989)
	0.14–38.0	0.10–8.80	Norway	1988	Skaare <i>et al.</i> (1990)
	4.99 (0.80–11.5) <sup>A</sup>	1.36 (0.51–2.45)	Moray Firth	1988	Hall <i>et al.</i> (1992)
	6.44 (4.04–9.37) <sup>A</sup>	0.80 (0.41–1.45)	West Coast	1988	Hall <i>et al.</i> (1992)
	12.1 (2.70–24.8) <sup>4</sup>	1.23 (0.36–2.70)	Orkney	1988	Hall <i>et al.</i> (1992)

\*Lipid weight; \*\* $\Sigma\text{DDT} = p,p'\text{-DDE} + p,p'\text{-DDD} + p,p'\text{-DDT}$ ; † $p,p'\text{-DDE} + p,p'\text{-DDT}$ ; <sup>A</sup>Arochlor 1254 equivalent; <sup>M</sup>males only; <sup>F</sup>females only; ( ) = range or  $\pm$  standard deviation.



**Table A10.6.** Target organ toxicity of non-*ortho* and mono-*ortho* CBs in mammals (from Battershill, 1994).

Target organ/system	Comments
Skin (chloracne in monkeys)	Non- <i>ortho</i> planar CBs, CB169 and CB77, were most potent. Di- <i>ortho</i> CBs gave a weak response. No effects seen with tetra- <i>ortho</i> CBs. No clear gradation in potency across groups of CBs is evident.
Liver (histology, rats, mice, monkeys)	A clear gradation between different groups of CBs: non- <i>ortho</i> > mono- <i>ortho</i> > di- <i>ortho</i> > others. However, potency of CB77 and CB105 was less than expected on this gradation, possibly due to greater metabolism of these CBs compared with other non- <i>ortho</i> and mono- <i>ortho</i> CBs.
Immunotoxicity (rats, mice) (plaque-forming response)	A clear gradation between different groups of CBs: non- <i>ortho</i> > mono- <i>ortho</i> > di- <i>ortho</i> > others. ED <sub>50</sub> for thymic atrophy in rats correlated with AHH induction, except for CB77 and CB105.
Reduced serum thyrosine (mink, mice)	Effects different in 2,3,7,8-TCDD responsive and non-responsive strains of mice.
Reduced plasma corticosteroid (mice)	Evidence of effects with CB169.
Follicular hyperplasia (monkeys)	Evidence of effects with CB77.
Gastrointestinal tract (mink, monkeys)	Evidence of effects with CB77; no effects seen with di- <i>ortho</i> planar CBs. No effects seen in mice fed CB169 or di- <i>ortho</i> planar CBs.
Decreased liver/serum retinol levels (rats)	Reported potency CB126 > CB77 > CB124. Available data consistent with an Ah-related effect.

McKenzie *et al.*, 1997b). A higher degree of lactational transfer was observed in female long-finned pilot whales (*Globicephala melas*), with 60–100 % of the mother's contaminant burden being transferred to progeny (Aguilar *et al.*, 1995). Nakata *et al.* (1995) estimated that the Baikal seal (*Phoca sibirica*) transferred about 20 % of its total DDTs and 14 % of its total PCBs to the pup during the reproductive process. The values for Canadian grey seals (*Halichoerus grypus*) were about 30 % and 15 % of their total DDT and PCB burdens, respectively (Addison and Brodie, 1977). High lactational transfer of organochlorines was also reported in grey seals from Scottish waters, with the transfer of selected CBs, ΣDDT, dieldrin, and HCB being 15 %, 45 %, 75 %, and 80 %, respectively, of the maternal blubber burden (Wells and Campbell, 1990).

The second route for the excretion of environmental contaminants for female marine mammals is via transplacental transfer. Three pregnant females, two cetaceans and one pinniped, were sampled and both maternal and foetal tissues were taken for analysis. Concentrations of organochlorine contaminants in the blubber of the mother/foetus pairs of a Risso's dolphin (*Grampus griseus*), common seal and harbour porpoise were compared and the results are summarized in Table A10.7. The harbour porpoise foetus contained the highest contaminant concentrations, 770 and 309 mg kg<sup>-1</sup> wet weight ΣCB and *p,p'*-DDE, respectively, representing 17 % of the mother's burden. The common seal samples showed that 7.7 % and 8.8 % of ΣCB and ΣDDT, respectively, were transferred to foetal tissue. The Risso's dolphin foetus had a larger proportion, 29 % on a wet weight basis, of the mother's CB burden, although the absolute concentrations present were lower than those observed in the harbour porpoise. The TEQ from the non-, mono-, and di-*ortho* CBs (CB77, CB105, CB118,

CB126, CB156, CB169, CB170 and CB180) were measured in the Risso's dolphin foetus (39 pg g<sup>-1</sup> lipid), which represented 39 % of the mother's total TEQ value. The porpoise and seal data are in agreement with previous results which indicated that the proportion of the mother's contaminant burden transferred during gestation is lower in common seals (1 %) than in harbour porpoises (15 %) (Duinker and Hillebrand, 1979; Donkin *et al.*, 1981). With such few data, however, little is known about the within-species variance on the transfer of contaminants from mother to progeny. The contaminant concentrations transferred to the foetus is dependent, as with lactation, on whether the female is primo gravid or, if not, the number of progeny produced previously.

Although the levels transferred to the foetus can be considerably lower than those transferred through lactation, the presence of organochlorine compounds and particularly 'dioxin-like' CBs during key stages of foetal development has been associated in many mammalian species with reduced birth weight, behavioural anomalies and poor recognition memory. In humans, higher transplacental CB burdens have been correlated with lower psychomotor skills, but such effects have not been observed following lactational transfer only. Similar responses have been observed in rats, mice, and monkeys (Gladen *et al.*, 1988; Jacobson *et al.*, 1990). Although it is unwise to extrapolate such effects in other species to marine mammals without a great degree of caution, they must be taken into account when investigating possible toxic effects of organochlorine compounds in pinnipeds and cetaceans.

During gestation and lactation, organochlorines are selectively transferred to the foetus or calf on the basis of

**Table A10.7.** Percentage of the mother's burden and concentration of  $\Sigma$ CB and  $\Sigma$ DDT passed to the foetus during transplacental transfer.

Species	% $\Sigma$ CB* transferred	$\Sigma$ CB transferred ( $\mu\text{g kg}^{-1}$ )	% $\Sigma$ DDT transferred	$\Sigma$ DDT transferred ( $\mu\text{g kg}^{-1}$ )	% $\Sigma$ TEQ transferred	$\Sigma$ TEQ transferred ( $\text{pg g}^{-1}$ )
Risso's dolphin	29 (40)	270 (614)	30 (40)	123 (279)	28 (39)	17 (39)
Common seal	7.7 (8.9)	164 (382)	8.9 (10.3)	58 (135)	na	na
Harbour porpoise	17 (17)	770 (1030)	17 (17)	309 (412)**	na	na

na = not available; values in parentheses are, or have been calculated from, lipid weight concentrations

\*%X transferred = (foetus burden/(foetus + mother burden))  $\times$  100

\*\* $\Sigma p,p'$ -DDE only

$\Sigma$ TEQ = (TEQCB77+TEQCB105+TEQCB118+TEQCB126+TEQCB156+TEQCB169+TEQCB170+TEQCB180) calculated using TEFs derived by Ahlborg *et al.* (1994)

their lipophilicity (Tanabe *et al.*, 1981; Green *et al.*, 1996b; McKenzie *et al.*, 1997b). Figure A10.4 shows the preferential gestational transfer to the foetus of less lipophilic compounds, which have lower octanol/water coefficients (log  $k_{ow}$ ), such as  $\gamma$ -hexachlorocyclohexane ( $\gamma$ -HCH), hexachlorobenzene (HCB), and the lower chlorinated CBs. This preferential transfer is also observed during lactational transfer in both cetaceans and pinnipeds (Green *et al.*, 1996b; McKenzie *et al.*, 1997b). Log  $k_{ow}$  values for CBs were taken from Hawker and Connel (1988) and for the pesticides were calculated using the equation from Finizio *et al.* (1997).

The potential abilities of CBs to disrupt the endocrine systems of vertebrates have only become known in recent years. Effects have also been observed in the reproductive system of marine mammals, but exact causes are not yet known (Siebert and Bruhn, 1998). Experimental reproductive failure in harbour seals was induced by feeding seals on fish from a polluted area of the Wadden Sea (Reijnders, 1986).

Lesions of the reproductive system have been described for seals from British, Swedish, and Finnish waters (Helle *et al.*, 1976a, 1976b; Helle, 1980; Baker, 1989; Olsson *et al.*, 1994; Bergman, 1997). From 1977–1986, about 42 % of four-year-old female Baltic grey seals examined showed stenosis (narrowing) or occlusions of the uterus, whereas between 1987–1996 only 11 % had those lesions. The pregnancy rate during the first period was only 17 %, whereas it increased up to 60 % in the second period (Bergman, 1997). It has been suggested that the lesions may have been attributed to high PCB and DDT levels in these animals, and that improvement of the seal population is due to a declining contaminant burden in the Baltic biota (ICES, 1997). These observations could not be categorically assigned to CBs as there are a great number of other unknown contaminants present. No such lesions of the reproductive system have been found in cetaceans.

Little is known about the pathology of sexual development during pregnancy in marine mammals. True intersex has only been described once for a beluga

(*Delphinapterus leucas*) from the St. Lawrence estuary, possibly due to pollutants mimicking oestrogenic activity (De Guise *et al.*, 1994c, 1995a). However, pseudo-intersex has been found several times in different species of cetaceans (e.g., Nishiwaki, 1953; Bannister, 1962). The reproductive performance of belugas from the St. Lawrence estuary is lower than that of the Arctic populations (Béland *et al.*, 1993; De Guise *et al.*, 1995a). A connection between reduced testosterone levels and high PCB and DDE concentrations has been suggested in Dall's porpoises (*Phocoenoides dalli*) from the northwestern North Pacific (Subramanian *et al.*, 1987b). Furthermore, in California sea lions (*Zalophus californianus*), premature birth has been related to high levels of organochlorines (DeLong *et al.*, 1973).

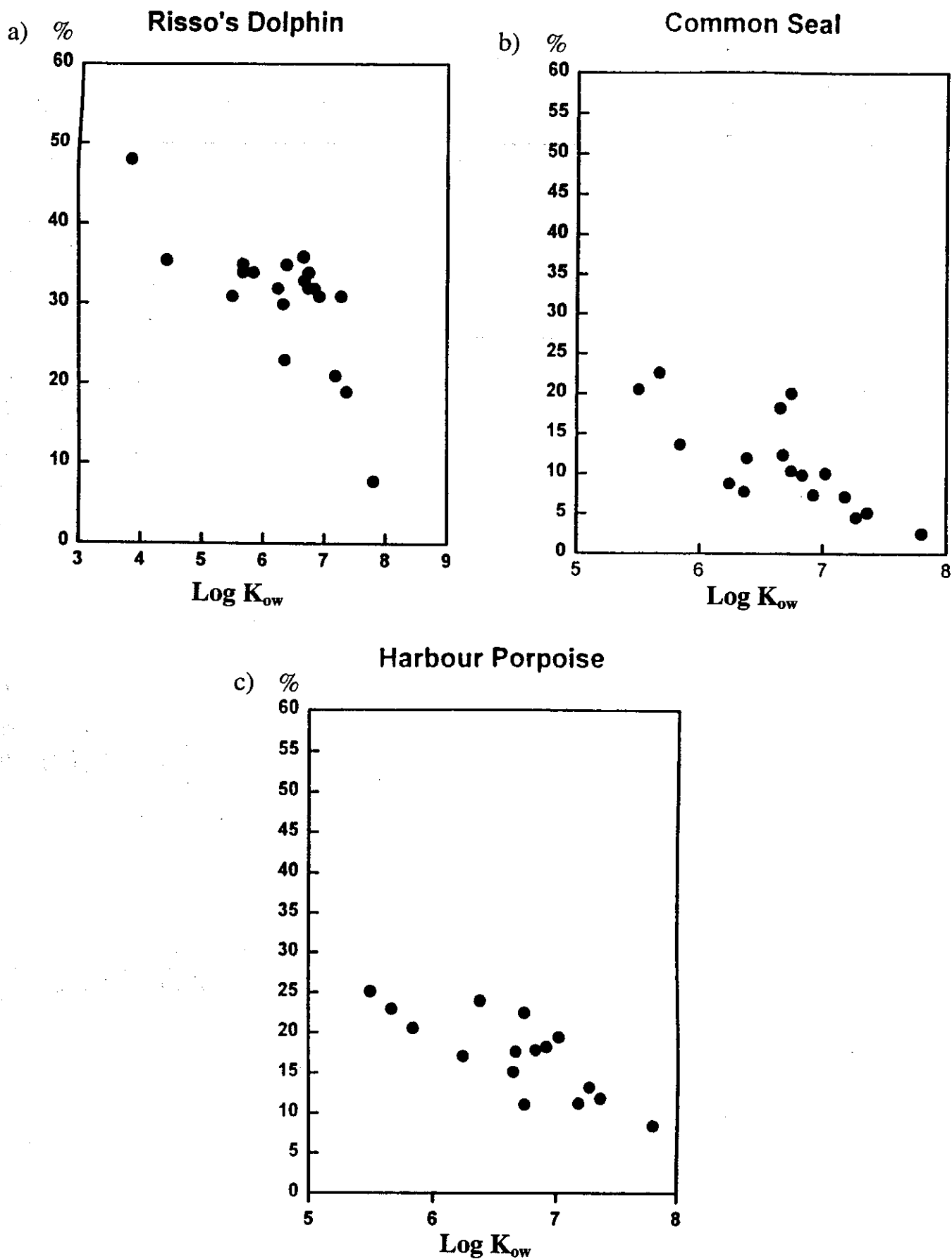
The effects of non-ortho CBs and mono-ortho CBs on the reproductive system and early development appear to have a strong biological effect on marine mammal populations. Further systematic investigations are required, for example, on the effects of transplacental transfer of toxins to the foetus.

### 5.3.2 Effects on the immune system

Effects of CBs on the immune system have been demonstrated in humans and other animals. Brouwer *et al.* (1989) observed lower blood retinol levels in seals fed contaminated fish (which is indicative of vitamin A deficiency). In the Netherlands, seals experimentally fed contaminated fish from the Baltic Sea showed functional failure of 'Natural Killer' (NK) cell activity and a decrease in T-cell function with increasing levels of polyhalogenated hydrocarbons compared with seals fed on less contaminated fish from the Atlantic (De Swart *et al.*, 1994, 1996; Ross *et al.*, 1995, 1996).

These investigations support suggestions that CBs reduce disease resistance, e.g., with respect to cancer agents acting immunosuppressively, and it is, therefore, difficult to investigate the potential relationship between CBs and epizootics of this virus. Only limited experiments have been done with cetaceans. However, in

**Figure A10.4.** Dependency of transplacental transfer (in percent) on lipophilicity ( $\log K_{ow}$ ) in (a) Risso's dolphin, (b) common seal, and (c) harbour porpoise.



*vitro* experiments have revealed responses such as mitogen-induced lymphocyte proliferation response in bottlenose dolphins (*Tursiops truncatus*) (Lahvia *et al.*, 1993). *In vitro* experiments on blood, thymus, and spleen cells of belugas showed increasing cytotoxicity with increasing PCB and PAH burdens (De Guise *et al.*, 1995b). In a study of stranded harbour porpoises from England and Wales, those animals diagnosed as having infectious diseases had significantly higher PCB concentrations than a previously healthy group. This provides support for the hypothesis that chronic PCB exposure negatively influences the health of cetaceans (Jepson *et al.*, 1998).

Harbour porpoises stranded on the German North Sea and Baltic Sea coasts showed a decreased nutritional condition and increased pathology of the respiratory system with increased mercury levels (Siebert, 1995). In belugas, an increased percentage of cell deaths was observed in Con-A-stimulated thymocytes cultured with HgCl<sub>2</sub>. Decreased splenocyte and thymocyte proliferation was observed with the highest concentration of HgCl<sub>2</sub> and CdCl<sub>2</sub> (10<sup>-5</sup> M). De Guise *et al.* (1995c) described an alteration in the ability of lymphocytes to proliferate, which might in turn lead to a reduced ability to mount an adequate immune response by the belugas of the St. Lawrence estuary. This might explain the high prevalence of severe diseases observed in that population. However, further studies and new techniques are needed to understand the effects of CBs and heavy metals on the immune system.

### 5.3.3 Effects on skeletal and endocrine systems

Several authors have described loss of bone substance in the skull and an asymmetry of the skull in several seal species from the North Sea and Baltic Sea (Stede and Stede, 1990; Zakharov and Yablokov, 1990; Bergman *et al.*, 1992; Mortensen *et al.*, 1992; Olsson *et al.*, 1994). Similar lesions have not been described in cetaceans. Swedish investigations suggest that lesions of the skeletal system were related to hyperplasia of the cortex of the adrenal gland, caused by a high load of organochlorines (Bergman and Olsson, 1986; Bergman, 1997). This is supported by a new study on harp seal pups. Those animals fed on increasing doses of specific PCB congeners showed higher cortisol levels which may indicate an adrenal hyperplasia in response to PCB exposure (Lohman *et al.*, 1998). Further changes of the endocrine system associated with the CB levels are described for the thyroid gland. Harbour seals fed on high concentrations of PCBs showed decreased concentrations of plasma retinol (vitamin A) and thyroid hormones in comparison to another group fed on less polluted fish (Brouwer *et al.*, 1989). Morphological changes of the thyroid gland, such as colloid depletion and interstitial fibroses, were found in harbour seals and harbour porpoises from the North Sea which were carrying high levels of PCBs (Schumacher *et al.*, 1993).

### 5.3.4 Tumours in marine mammals

A systematic investigation of belugas from the St. Lawrence estuary has shown a high prevalence of tumours. These include several malignant tumours, such as scirrhous gastric adenocarcinoma, hepatocellular carcinoma, mammary and intestinal adenocarcinoma, and transitional cell carcinoma of the urinary bladder (Martineau *et al.*, 1985; De Guise *et al.*, 1994a). In total, 28 of the 75 cases of tumours reported world-wide for cetaceans and 13 of 27 cases of cancer (47 %) have been found in this genetically closed beluga population (Geraci *et al.*, 1987; De Guise *et al.*, 1994a; IWC, 1995; Martineau *et al.*, 1999). Except for gastric papillomas found in eight belugas caused by papillomaviruses (De Guise *et al.*, 1994b), the etiology of the tumours remains unclear. It has been suggested that this high prevalence was due to carcinogenic compounds and decreased resistance to the development of tumours (De Guise *et al.*, 1995a). High levels of toxic compounds, such as PCBs and polycyclic aromatic hydrocarbons (PAHs), are found in the belugas (Martineau *et al.*, 1987). Some PAHs, such as BaP (benzo[a]pyrene), are among the most potent carcinogens, acting as initiators, whereas others such as PCBs are recognized as promoters for the induction of tumours in initiated cells (De Guise *et al.*, 1995a).

A higher prevalence of tumours, such as leiomyomas (benign tumour of the muscle tissue), is also described for the Baltic grey seal (Bergman, 1997). About 53 % of the females investigated between 1977–1986 showed leiomyomas, whereas between 1987–1996 only 44 % carried these tumours, which in some cases may hamper reproductive success.

With the possible exception of the belugas from the St. Lawrence estuary, no population-level effects have been described for marine mammals from tumours that may have been induced by PAHs and PCBs. However, recent surveys show recovery of the beluga population in the St. Lawrence estuary after hunting was banned in 1979 (Kingsley, 1998).

### 5.3.5 Other pathological effects

In bottlenose dolphins from the Atlantic coast of the USA, histopathological changes in the liver, such as central portal necroses, fatty-liver lymphocyte infiltration and lipofuscin accumulation, have been correlated with high mercury levels in the liver (Rawson *et al.*, 1993). In an experimental study, grey seals fed on mercury-contaminated fish showed changes in the blood picture indicating toxic hepatitis, and failure of the kidneys associated with uremia (failure of the renal function associated with an accumulation of nitrogenous compounds in the blood) (Holden, 1978).

### 5.3.6 Effects on behaviour

There are obvious difficulties in measuring behavioural changes in free-ranging cetaceans that might result from contamination (IWC, 1995), even though such changes are not precluded. In pinnipeds, one study describes coordination failures in a harbour seal with a burden of 200 µg Hg g<sup>-1</sup> liver fresh weight (Law *et al.*, 1991).

### 5.4 Indirect Effects

The IWC Workshop on Chemical Pollution and Cetaceans focused on the role of fish in pollutant transmission to cetaceans (IWC, 1995). Fish are unable to metabolize many pollutants, such as the organochlorine compounds, and thus are carriers of these persistent chemicals. Pollutants affect fish and other prey of cetaceans at many levels, and concerns were expressed that, in the long term, these effects have the potential for reducing the abundance of marine resources and, therefore, also cetacean food availability.

### 5.5 Summary and Conclusions

Chlorobiphenyls, in particular non-*ortho* and mono-*ortho* CBs, are likely to affect the health of certain marine mammal populations, but the extent of this effect is unclear, despite some experiments linking contaminants to their sub-cellular, cellular or systemic level effects (De Guise *et al.*, 1995b, 1995c; De Swart *et al.*, 1994, 1996; Ross *et al.*, 1995, 1996). Although suppression of population growth and fecundity rates have been reported for marine mammal populations resident in contaminated areas (e.g., grey and ringed seals in the Baltic Sea, harbour seals in the Wadden Sea), there is no well-defined cause-effect relationship linking specific contaminants to population-level effects. This is in accordance with the findings of the IWC Workshop on Chemical Pollution and Cetaceans (IWC, 1995).

With particular reference to the initiative taken by the IWC Scientific Committee to describe cause-effect relationships between environmental contaminants and three species of odontocetes (IWC, 1995, 1997), an ICES research programme aimed at understanding and describing cause-effect relationships between environmental contaminants and population-level effects in marine mammals (with the emphasis on seals) should be carried out in collaboration with ongoing research in other organizations. Methods need to be developed and knowledge needs to be gained by investigating live and dead animals for effects of contaminants. The effects of chronic exposure and the combined effects of several toxins in the natural environment need to be understood. Therefore, long-term biological effects monitoring is urgently required.

## 6 ACKNOWLEDGEMENT

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# ANNEX 11

## ACME/ACMP ADVICE BY TOPIC FOR THE YEARS 1987-1998

Numbers in the table refer to sections of the present report and of the ACMP or ACME reports from 1987 to 1998, in reverse chronological order.  
\*Signifies major advice on that topic.

Topic	Sub-topic	1988	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998
Monitoring	Strategy			*4				5.1	5	*4; *Ann. 1	5.1			
	Programme evaluation									4.2				
	Multi-purpose					6.2								
	Benthos			*7.1	*Ann. 1	9	8.1	8; *Ann. 6				6.1.2; 11.1; *Ann. 8		
	NSTF/MMP				4	9.3			5.2					
	Sediments/guidelines				*14				6.1; *Ann. 1	5.5		5.5; *Ann. 4	4.5; *Ann. 1	4.6; *Ann. 2
	Sediment data normalization									5.5	5.6	5.5.1	4.5.2	
	Sediment sensitivity, variance factors		15.2	14.6; Ann. 2										
	Metals/sediments			12.6						5.5	5.6	9.5		
	Matrix tables													
	•general (JMP)			4	*6.1	*Ann. 1								
	•organic					6.1								
	•NSTF					6.1								
	Substances that can be monitored													
	•organic										6.6	5.4	4.2	4.5
	•inorganic													4.5
	Use of seaweeds				6.3	6.8				5.1				
	Use of seabird eggs													4.7.5
	Spatial monitoring									5.1	5.3		*4.7.2	
	JAMP/JMP guidelines							13.3			5.4	5.2.5.4	4.1	
	BMP guidelines		*12						5.3		5.4	5.1.2		
	AMAP									5.4		5.1.3		4.4
	Effects of nutrient enrichment										5.8	9.1		
	Monitoring PAHs												4.4.1; 4.5; *Ann. 1	4.2; *Ann. 1

Topic	Sub-topic	1998	1997	1996	1995	1994	1993	1992	1991	1990	1989	1988	1987
Assessment	Statistical methods	4.7.1											
	Combined effects of contaminants	11.2											
Temporal trend monitoring	Strategy/objectives			*4; Ann. 1		*4; Ann. 1							
	Guidelines			4.4		4.5.2						5.2	6.2; 6.3
	Data analysis	4.7.4	15.2			5.2	6.2	5.2					
	Nutrients			5.7			6.3					11.1	
	Fish/IMP			5.6					5.1	6.2			
	Fish/CMP			5.6					5.1	6.5	*6.4	5.1	6.1
	Biota/BMP				7.3								
	Biological effects	4.1											
	Mussels								5.1	*Ann. 3			6.3
	Pooling									6.7; Ann. 5	6.4.3		6.2.1
	Precision							*Ann. 1			6.4.4		
	Sediment storage					5.2						14.2	
	Sea water												6.5
	Sediments			4.3; 5.5.3								5.3; Ann. 2	6.5
Integration of biological/ chemical measurements	Statistical requirements	4.7			4.3								
	Sediment quality			5.2.2	4.2; Ann. 2	5.4	6.4; *Ann. 2						
	Monitoring strategy			*5.3	4.1; *Ann. 1								
	Concepts										*7.1		
	Statistical design	4.1*	4.3.1										
	Methods		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	
	Fish egg bioassays												
	Data analysis												
	•general							6.3					
	•EROD							*Ann. 2					
	•oyster bioassay							*Ann. 2					





Topic	Sub-topic	1988	1987	1989	1988	1987
Intercomparison exercises	Status				Ann. 3	
	Nutrients/sea water			Ann. 2	Ann. 3	
	Specific hydrocarbons			13.4	11.3	16.2
	Hydrocarbons in					13.3
	• biota			13.7		
	• sediments			13.7		
	• sea water			13.7		
	PAHs/standards			13.2	12.1	
	PCBs/CBs in biota				12.3	
	Organochlorines in biota					
	CBs/standards			14.3	12.2	
	CBs in sediments					14
	Metals in					
	• sea water	5.5	6.5			13.1
	• sediments				12.5	
	• biological tissue					13.2
	• SPM				14.4; Ann. 1	
	Dissolved oxygen in sea water					
	Methyl Hg in biological tissue			12.5	13.4	*14.5
Methods	Primary production					12.4
	Oyster embryo bioassay				10.2	
	EROD					
	Nutrients in sea water					
	DO in sea water	*Ann. 3			*Ann. 4	13.6
	Sediment normalization		5.5			*14.1
	Organic carbon measurements	4.6; Ann. 2				
	Analysis of total OCs					
	Primary production methods			6.5	11	12.1
	Initiating factors				*11.3	12.2
Algal blooms	Dynamics	10.2		8	10	
	Exceptional blooms	Ann. 3				
	Phycotoxins/measurements				11.2	
					11.4	12.3
	<i>C. polyplepis</i> bloom			*11.1	10.3	



Topic	Sub-topic	1998	1997	1996	1995	1994	1993	1992	1991	1990	1989	1988	1987
Overviews (cont.)	Tris(4-chlorophenyl) methanol/methane			10 *Ann. 7									
	Octachlorostyrene								20.1				
	Toxaphene		8.1; *Ann. 5			12.3							
	Atrazine					12.1							
Classification/ assessment tools	Irgarol 1051		8.1; *Ann. 4										
	PCDEs		8.1; *Ann. 6										
	Human health									19			
	Hazardous substances	11.1			12.3			*15					
Sand/gravel extraction	Background concentrations		15.1		12.1								
	Ecotoxicological reference values				12.2								
	Code of Practice									16			
	Effects	6.1	*6.3						15	16	15	15	
Modelling	Environmental impact assessment				*15	*15	13						
	Radioactive contaminants/Baltic Sea							*17.1					
	Use in monitoring and assessment					16		17.2					
	Nutrients	15.2	17.2	16.1.2									
Data banks and management	Contaminants	15.1	17.1	16.1.1; 16.3	17	2.2	2.2						
	NSTF							20	*21	22			
	ICES format			16.6									
	ICES databases						14						
Ecosystem effects of fishing	Biological database	15.1.3	17.3; 17.4			11.2; Ann. 4							
	AMAP		17.1.1	16.2									
	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		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											



## ANNEX 12

### TITLES OF RECENTLY PUBLISHED ICES COOPERATIVE RESEARCH REPORTS

No.	Title
210	Report of the ICES Advisory Committee on Fishery Management, 1994 (Part I and Part 2)
211	Intercalibration Exercise on the Qualitative and Quantitative Analysis of Fatty Acids used in <i>Artemia</i> and Marine Samples used in Mariculture
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

## ACRONYMS

ACFM	Advisory Committee on Fishery Management	CRIMP	Centre for Research on Introduced Marine Pests (Australia)
ACHe	acetylcholinesterase	CRMs	certified reference materials
ACME	Advisory Committee on the Marine Environment	CTD	conductivity-temperature-density
ACMP	Advisory Committee on Marine Pollution	CUSUM	<b>Cumulative Sum</b>
ADI	acceptable daily intake	CV	coefficient of variation
AGDATA	Ad Hoc Group of Database/GIS Practitioners	DBT	dibutyltin
AHF	altered hepato fico	DCM	dichloromethane
AHH	aryl hydrocarbon hydroxylase	DDE	dichlorodiphenylethylene
ALA-D	D-aminolevulinic acid dehydratase	DDT	dichlorodiphenyltrichloroethane
AMAP	Arctic Monitoring and Assessment Programme	ΣDDT	total DDT
ASE	accelerated solvent extraction	DIFFCHEM	OSPAR Working Group on Diffuse Sources
ASMO	Environmental Assessment and Monitoring Committee (OSPAR)	DMA	dimethylarsenic
ASP	amnesic shellfish poisoning	DNA	deoxyribonucleic acid
BCF	bioconcentration factor	DO	dissolved oxygen
BCPS	bis- <i>p</i> -chlorophenyl sulfone	DOC	dissolved organic carbon
BCR	European Commission Bureau of Community References	DSP	diarrhetic shellfish poisoning
BEWG	Benthos Ecology Working Group	DST	diarrhetic shellfish toxin
BMB	Baltic Marine Biologists	EC	European Commission
BMP	Baltic Monitoring Programme (HELCOM)	ECD	electron capture detection
BSPAs	Baltic Sea Protected Areas	EI	electron impact ionization
CBs	chlorobiphenyls	EIA	environmental impact assessment
CDEs	chlorodiphenylethers	EMD	evaporative mass detector
CD-ROM	compact disc: read-only memory	ENDS	Environmental Data Services (UK)
CEFAS	Centre for Environment, Fisheries and Aquaculture Science (UK)	EOC	elemental organic carbon
CFP	ciguatera fish poisoning	EOCI	extractable organic chlorine
CHBs	chlorinated bornanes	EPA	Environmental Protection Agency (USA)
CIEM	Conseil International pour l'Exploration de la Mer	EQG	environmental quality guidelines
CMDGC	comprehensive multidimensional gas chromatography	EQS	environmental quality standard
CMP	Cooperative ICES Monitoring Studies Programme	EROD	ethoxyresorufin- <i>O</i> -deethylase
COMBINE	Cooperative Monitoring in the Baltic Marine Environment (HELCOM)	ESE	enhanced solvent extraction
		EU	European Union
		EUT	Ad Hoc Working Group on Eutrophication (OSPAR)
		FDE	Fish Disease Data Entry Program
		FID	flame ionization detection
		GABA	gamma-aminobutyric acid
		GC	gas chromatography

GC/ECD	gas chromatography/electron capture detection	JGOFS	Joint Global Ocean Flux Study (IGBP)
GC/MS	gas chromatography/mass spectrometry	JMP	OSPAR Joint Monitoring Programme
GESAMP	Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection	JNCC	Joint Nature Conservation Committee (UK)
GPC	gel permeation chromatography	LD	lethal dose
GLOBEC	Global Ocean Ecosystem Dynamics Programme	LOI	loss-on-ignition
GOOS	Global Ocean Observing System	LRMs	laboratory reference materials
GSI	gonadosomatic index	LRMS	low resolution mass spectrometry
GST	glutathion-S-transferase(s)	MAST	Marine Science and Technology Programme (EC)
HAB	harmful algal bloom	MBT	monobutyltin
HCB	hexachlorobenzene	MCWG	Marine Chemistry Working Group
HCH	hexachlorocyclohexane	MDGC	multidimensional gas chromatography
H&E	haematoxylin and eosin	MDR	multidrug resistance
HELCOM	Helsinki Commission (Baltic Marine Environment Protection Commission)	MEDPOL	Monitoring and Research Programme of the Mediterranean Action Plan
HPLC	high performance liquid chromatography	MFO	mixed-function oxidase
HRMS	high resolution mass spectrometry	MMA	monomethylarsenic
HTCO	high temperature catalytic oxidation	MMHg	monomethylmercury
IAEA	International Atomic Energy Agency	MMP	Monitoring Master Plan (NSTF)
ICES	International Council for the Exploration of the Sea	MON	Ad Hoc Working Group on Monitoring (OSPAR)
ICS	International Chamber of Shipping	MS	mass spectrometry
IEH	Institute for Environment and Health (UK)	MSD	mass selective detector
IFREMER	Institut Français de Recherche pour l'Exploitation de la Mer	mtDNA	mitochondrial DNA
IJC	International Joint Commission (Canada and the USA)	MXR	multixenobiotic resistance
IMO	International Maritime Organization	NCI	negative chemical ionization
IMPACT	Working Group on Impacts on the Marine Environment (OSPAR)	NIHS	National Institute of Health Science (Japan)
IOC	Intergovernmental Oceanographic Commission	NIST	U.S. National Institute of Standards and Testing
ISO	International Organization for Standardization	NIVA	Norsk institutt for vannforskning [Norwegian Institute for Water Research]
ITIS	Interagency Taxonomic Information System (USA)	NMR	nuclear magnetic resonance
IUCN	International Union for the Conservation of Nature and Natural Resources	NOAA	National Oceanic and Atmospheric Administration (USA)
IWC	International Whaling Commission	NOAEC	no observable adverse effects concentration
JAMP	OSPAR Joint Assessment and Monitoring Programme	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)	QUASH	Quality Assurance of Sampling and Sample Handling (EC)
NRC	National Research Council (Canada)	RIA	radioimmunoassay
NSP	neurotoxic shellfish poisoning	RIKZ	Rijksinstituut voor Kust en Zee [National Institute for Coastal and Marine Management]
NSTF	North Sea Task Force		
OCs	organochlorines	RM	reference materials
OIE	Office International des Epizooties	RUBIN	Rutin för Biologiska Inventeringar
OM	oxidizable matter	SFE	supercritical fluid extraction
OPs	organophosphates	SFG	scope for growth
OSPAR	OSPAR Commission	SG	Study Group
PAHs	polycyclic aromatic hydrocarbons	SGBSC	Steering Group for the Coordination of the Baseline Study of Contaminants in Baltic Sea Sediments
PAR	photosynthetic available radiation		
PBDEs	polybrominated diphenylethers	SGBWS	ICES/IOC/IMO Study Group on Ballast Water and Sediments
PBTs	persistent, bioaccumulative, toxic compounds	SGFDDS	Study Group on Statistical Analysis of Fish Disease Data in Marine Fish Stocks
PCA	principal component analysis		
PCBs	polychlorinated biphenyls	SGBIS	Study Group on Marine Biocontrol of Invasive Species
PCDDs	polychlorinated dibenzo- <i>p</i> -dioxins	SGMPCS	Study Group on Monitoring Programmes for Contaminants in Sediments
PCDEs	polychlorinated diphenylethers		
PCDFs	polychlorinated dibenzofurans	SGQAB	ICES/HELCOM Steering Group on Quality Assurance of Biological Measurements in the Baltic Sea
PCNA	proliferating cell nuclear antigen	SGQAC	ICES/HELCOM Steering Group on Quality Assurance of Chemical Measurements in the Baltic Sea
PCNs	polychlorinated naphthalenes	SGQAE	ICES/OSPAR Steering Group on Quality Assurance of Biological Measurements related to Eutrophication Effects
PCP	pentachlorophenol		
PCTs	polychlorinated terphenyls	SIM	selected ion monitoring
PEEK	polyetheretherketone	SIME	Working Group on Concentrations, Trends and Effects of Substances in the Marine Environment (OSPAR)
PFC	plaque-forming cell		
PICT	pollution-induced community tolerance	SMLIPA	ICES Special Meeting on the Use of Liver Pathology in Flatfish for Monitoring Biological Effects of Contaminants
POC	particulate organic carbon		
POPs	persistent organic pollutants	SOAEFD	Scottish Office Agriculture, Environment and Fisheries Department
PROD	pentoxoresorufin- <i>O</i> -deethylase	SOPs	Standard Operating Procedures
PSP	paralytic shellfish poisoning	SPM	suspended particulate material
PSU	practical salinity unit	SRMs	standard reference materials
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		



TALs	total annual loads	WGECO	Working Group on Ecosystem Effects of Fishing Activities
TBA	tetrabutylammonium	WGEIM	Working Group on Environmental Interactions of Mariculture
TBPS	tertiary butylbicyclophosphorothionate	WGEXT	Working Group on the Effects of Extraction of Marine Sediments on the Marine Environment
TBT	tributyltin	WGHABD	ICES/IOC Working Group on Harmful Algal Bloom Dynamics
TCDD	tetrachlorodibenzo- <i>p</i> -dioxin	WGITMO	Working Group on Introductions and Transfers of Marine Organisms
TEF	toxic equivalency factor	WGMDM	Working Group on Marine Data Management
TIE	toxicity identification evaluation	WGMMHA	Working Group on Marine Mammal Habitats
TIMES	<i>ICES Techniques in Marine Environmental Sciences</i>	WGMMPD	Working Group on Marine Mammal
TIP	training and intercalibration programme	WGMS	Working Group on Marine Sediments in Relation to Pollution
TMA	trimethylarsenic	WGNAS	Working Group on North Atlantic Salmon
TOC	total organic carbon	WGOH	Working Group on Oceanic Hydrography
TPT	triphenyltin	WGPDMO	Working Group on Pathology and Diseases of Marine Organisms
UK	United Kingdom	WGPE	Working Group on Phytoplankton Ecology
UNEP	United Nations Environment Programme	WGSDEM	Working Group on the Statistical Aspects of Environmental Monitoring
Unesco	United Nations Educational, Scientific, and Cultural Organization	WGSE	Working Group on Seabird Ecology
U.S.	United States	WGSSO	Working Group on Shelf Seas Oceanography
USA	United States of America	WGZE	Working Group on Zooplankton Ecology
USEPA	United States Environmental Protection Agency	WKSMTD	ICES/OSPAR Workshop on the Identification of Statistical Methods for Trend Detection
UV	ultraviolet	WOCE	World Ocean Circulation Experiment
VIC	Voluntary International Contaminant Monitoring in Temporal Trends	WWW	world wide web
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		
WGEAMS	Working Group on Environmental Assessment and Monitoring Strategies		