Marine Habitat Committee

ICES CM 1998/E:4
Ref.: ACME

REPORT OF THE

WORKING GROUP ON ENVIRONMENTAL ASSESSMENT AND MONITORING STRATEGIES

ICES Headquarters, Copenhagen
23–27 March 1998

TABLE OF CONTENTS

Section		Page
1	OPENING OF THE MEETING	1
2	ADOPTION OF THE AGENDA	1
3	ARRANGEMENTS FOR THE PREPARATION OF THE REPORT	1
4	REPORTS OF ACTIVITIES IN OTHER FORA OF INTEREST TO THE MEETING	2
4.1	Advisory Committee on the Marine Environment (ACME) and Marine Habitat Committee (MHC)	2
4.2	Report on the Outcome of the OSPAR Ad Hoc Working Group on Monitoring (MON)	3
4.3	OSPAR Working Group on Concentrations, Trends and Effects of Substances in the Marine Environment (SIME)	4
4.4	Helsinki Commission (HELCOM)	4
4.5	Action List from 1997 Meeting	5
5	GENERAL FRAMEWORK FOR AN INTEGRATED APPROACH TO THE ASSESSMENT OF MARINE ECOLOGICAL QUALITY	5
5.1	Introduction	5
5.2	Ecological Quality	5
5.2.1	Assessing ecological quality	5
5.3	Indices of Ecosystem Health	5
6	IDENTIFICATION OF PRIORITY CONTAMINANTS	7
7	REVIEW OF THE COMPONENTS OF THE MONITORING GUIDELINES FOR PAHs IN BIOTA (BY MCWG) AND SEDIMENTS (BY WGMS), INCLUDING SAMPLING BY SGMPCS, FOR OVERALL COHERENCE AND COMPLETENESS [OSPAR 1998/1.1]	8
8	REVIEW INFORMATION AND DATA ON CONCENTRATIONS OF NON-*ORTHO* AND MONO-*ORTHO* CBs IN MARINE MAMMALS AND RELEVANT BIOLOGICAL EFFECTS AND PREPARE A REPORT ON THE FINDINGS AND IMPLICATIONS, AS A CONTRIBUTION TO THE OSPAR QSR 2000 (WITH MCWG, WGMMHA AND WGBEC) [OSPAR 1998/3.1]	9
9	REVIEW INFORMATION ON MONITORING STRATEGIES ASSESSING HAZARDS PRESENTED BY THE DISCHARGE OF PRODUCED WATER BY OFFSHORE OIL AND GAS INDUSTRIES	9
9.1	Produced Water	9
9.1.1	Composition of produced water	11
9.1.2	Distribution of the discharged produced water	14
9.1.3	Effects of produced water on the marine environment	16
9.1.4	References	17
9.2	Discussion	18
10	ROLE OF ICES IN MARINE ENVIRONMENTAL MONITORING AND ASSESSMENT IN RELATION TO THE ACTIVITIES OF THE EUROPEAN ENVIRONMENT AGENCY (EEA) AND THE EUROPEAN TOPIC CENTRE (ETC) ON MARINE AND COASTAL WATER	19
10.1	Provision of Data Bank Support and Data Products for EEA Assessments	19
10.2	Provision of Scientific Advice	19
10.3	Quality Assurance of Data	20
11	REVIEW INFORMATION COLLATED INTERSESSIONALLY ON PROCEDURES TO ASSESS THE COMBINED EFFECTS OF EXPOSURE OF ORGANISMS TO GROUPS OF CHEMICALLY SIMILAR, OR DISSIMILAR, CONTAMINANTS	20
12	PREPARE A SHORT REPORT ON WEBSITES PROVIDING INFORMATION OF RELEVANCE TO THE WORK OF WGEAMS	21
13	REVIEW OF THE FINAL REPORT OF THE BASELINE STUDY OF CONTAMINANTS IN BALTIC SEA SEDIMENTS	22
14	CONCLUDE THE PREPARATION OF PROPOSALS FOR A WORKSHOP ON RISK EVALUATION AND ENVIRONMENTAL MONITORING, IN COLLABORATION WITH WGSAEM	26
15	REVIEW OF THE OUTCOME OF SGMPCS (WITH WGMS AND WGSAEM)	26

TABLE OF CONTENTS

Section **Page**

16 ANY OTHER BUSINESS 27
16.1 New Chairman 27
16.2 GOOS 27
16.3 Global International Waters Assessment (GIWA) 29

17 CONSIDERATION AND APPROVAL OF ACTION LIST/RECOMMENDATIONS 31

18 PROPOSALS FOR A FURTHER MEETING 31

19 CONSIDERATION AND APPROVAL OF THE MEETING REPORT 31

20 CLOSURE OF THE MEETING 31

ANNEX 1: AGENDA 32

ANNEX 2: LIST OF PARTICIPANTS 33

ANNEX 3: LIST OF MEETING DOCUMENTS 34

ANNEX 4: ACTION LIST FOR 1997 35

ANNEX 5: ACTION LIST FOR 1998 36

ANNEX 6: RECOMMENDATIONS 37

APPENDIX 1 40

1 OPENING OF THE MEETING

The 1998 meeting of the Working Group on Environmental Assessment and Monitoring Strategies (WGEAMS) was opened by the Chairman, Dr Ian M. Davies, at 09.30 hrs on 23 March 1998 at ICES Headquarters in Copenhagen, Denmark. The list of participants is appended as Annex 1.

The terms of reference for the meeting are given below:

ICES C.Res.1997/2:12:3

The Working Group on Environmental Assessment and Monitoring Strategies [WGEAMS] (Chairman: Dr I. Davies, UK), will meet at ICES Headquarters from 23–27 March 1998 to:

a) start examing the general framework of an integrated approach to the assessment of marine ecological quality, including the formulation of some practical suggestions as to the possible implementation of such an approach and the potential tools to carry it out;
b) update of the review prepared at the WGEAMS meeting in Halifax (1991) on methods for the identification of priority contaminants, with particular reference to the Esbjerg Declaration regarding contaminants which are toxic, persistent, and liable to bioaccumulate;
c) review the components of the monitoring guidelines for PAHs in biota (by MCWG) and sediments (by WGMS), including sampling by SGMPCS, for overall coherence and completeness [OSPAR 1998/1.1];
d) review the information and data on concentrations of non-*ortho* and mono-*ortho* CBs in marine mammals and relevant biological effects and prepare a report on the findings and implications, as a contribution to the OSPAR Quality Status Reports (with MCWG, WGMMHA and WGBEC) [OSPAR 1998/3.1];
e) review information collated intersessionally on current national and international monitoring strategies which address the hazards presented by the discharge of produced water by the offshore oil and gas industries;
f) review the role of ICES in marine environmental monitoring and assessment in relation to the activities of the European Environment Agency and the European Thematic Centre on Marine and Coastal Water;
g) review information collated intersessionally on procedures to assess the combined effects of exposure of organisms to groups of chemically similar, or dissimilar, contaminants;
h) prepare a short report on websites providing information of relevance to the work of WGEAMS;
i) review the final report of the Baseline Study of Contaminants in Baltic Sea Sediments with a view to assessing implications for future monitoring strategies (with WGMS) [HELCOM 1998/4];
j) conclude the preparation of proposals for a Workshop on risk evaluation and environmental monitoring, in collaboration with WGSAEM;
k) review the outcome of SGMPCS (with WGMS and WGSAEM).

Through the auspices of the General Secretary, representatives of the EEA and UNEP should be invited to attend the meeting. WGEAMS will report to ACME before its June 1998 meeting and to the Marine Habitat Committee at the 1998 Annual Science Conference.

2 ADOPTION OF THE AGENDA

The draft Agenda [WGEAMS98/2/1] was adopted with the addition of discussion of the possible relations of ICES to GOOS. The revised draft Agenda is attached as Annex 2. A list of the papers considered at the meeting is contained in Annex 3.

3 ARRANGEMENTS FOR THE PREPARATION OF THE REPORT

The Chairman reminded the Working Group that the ICES Secretariat requires that the report of the meeting be drafted and approved by the end of the meeting, as is now usually the case. The deadline for receipt of the completed report at ICES Headquarters is 17 April 1997. Sections of the report were therefore drafted throughout the course of the meeting and time was set aside on the final day for approval of the drafts, including the recommendations. The Chairman undertook any final detailed editing of the text prior to submitting the final version of the report to ICES.

Only six members of WGEAMS, from five countries, attended the meeting. There was some discussion regarding the possible reasons for this low turn-out. As at the meeting in 1997, it was felt that, in some countries, funding for travel etc., to ICES Working Group meetings was increasingly difficult to secure. The Chairman undertook to send out a circular to WGEAMS members asking why they did not attend.

4 REPORTS OF ACTIVITIES IN OTHER FORA OF INTEREST TO THE MEETING

4.1 Advisory Committee on the Marine Environment (ACME) and Marine Habitat Committee (MHC)

A letter had been received from Mr Stig Carlberg, the Chairman of ACME, thanking WGEAMS for its 1997 contribution. Sections 5, 6, 7, 10, 11 and 12 of the 1997 WGEAMS Report had all been of direct relevance to ACME at its meeting in June 1997. Section 11 had been slightly amended by ACME to more clearly reflect the uncertainties in the analysis. Mr Carlberg had also noted the low attendances at recent WGEAMS meetings. He agreed with the 1997 WGEAMS report that the reason might be the 'philosophical' nature of some of the discussions. He suggested that WGEAMS could consider inviting experts outside the normal membership to address particular agenda items (with the approval from the ICES General Secretary). An alternative might be for members to approach prospective new members and to request their national Delegates that these experts be appointed as official members of WGEAMS.

Following recent restructuring within ICES, WGEAMS now reports to the Marine Habitat Committee (MHC), instead of to ACME, although the WGEAMS report will continue to be considered by ACME at its annual meeting. WGEAMS noted that many of the traditional environmental Working Groups (MCWG, WGMS, WGBEC, WGSAEM) had also been transferred to MHC parentage and welcomed that these groups had not been split between more than one parent Committee.

There was some discussion of the general remit of MHC, as given in the 1996/1997 ICES Annual Report. WGEAMS noted that the remit of this new Committee includes coastal zone management, habitat research and marine living resources, and marine biodiversity.

WGEAMS also noted that some of these subjects were not well covered by the current Working Group structure. The Working Group on Environmental Interactions of Mariculture (WGEIM) has made efforts in recent years to pioneer the subject of coastal zone management in ICES. WGEIM has also interpreted 'environmental interactions' as including a wide range of coastal resource allocation problems and the potential conflicts between different users of the coastal zone. WGEIM has expanded its expertise beyond mariculture scientists and now includes a mixture of experts comprising regulators and managers of coastal fisheries and aquaculture (which commonly involve balancing conflicting interests and interactions in the coastal zone), coastal zone management experts and economists.

In contrast, it is not clear which current Working Groups will consider the generality of habitat research or marine biodiversity. There are clear links with activities generated by the EU Habitats and Species Directive, including the designation and monitoring of Marine Protected Areas. However, it is also necessary to consider the consequences of economic pressure on the sea, for example, industrial and urban developments on the coast, fishing pressure on the shelf seas, and expansion of hydrocarbon exploration and extraction into deeper waters of the shelf edge and potentially to offshore areas such as Rockall Plateau. Working Groups addressing these matters will need to have a broad perspective.

WGEAMS was surprised to learn that the responsibilities of MHC do not include the word 'sustainable'. Sustainability is increasingly seen as an important test against which marine (and terrestrial) developments are assessed. WGEAMS feels that it is difficult to see how ICES can address these new topic areas without giving prominence to the concept of sustainability.

In addition, WGEAMS noted that ICES activities were coming under ever increasing financial pressures and that full funding for advisory work was a target for both of the Advisory Committees. While this may be achieved for ACFM activities, ACME is a considerable distance from such a goal. If, however, ACME were to be limited to a role of only providing advice in response to specific requests, primarily from regulatory commissions, this would severely limit the scope and coverage of the ICES environmental advisory function, which WGEAMS feels is of great value to ICES Member Countries as a whole, and not simply to the Commissions. If this limitation occurs, it will inevitably be reflected to some extent in the Terms of Reference for Working Groups with requests from ACME.

WGEAMS expressed concern that the success of ICES is largely based on the proactive role of Working Groups (and their parent Committees) in taking initiatives in response to ongoing scientific advances. Thus ICES has been able to place before the Commissions (and other fora) concepts and technical matters that have contributed greatly to the advancement of their activities.

If the environmental advisory functions are limited to responding only to paid requests for advice, this could result in restrictions of the core scientific activities of ICES and could greatly reduce the scope and impact of its work. The consequential loss of 'reputation' or 'standing' through concentration on reactive (rather than proactive) environmental advisory functions could (perversely) result in a reduction in the effectiveness of the advice and, potentially, reductions of customer interest and loyalty. It was noted that the types of requests received on environmental issues tend to be fairly narrow and focused, e.g., Guidelines for monitoring PAHs, or QA procedures for chemical and biological measurements, in contrast with ICES-initiated proactive advice, for example, the strategy for biological effects monitoring or the assessment of temporal trends in contaminants in biota, which have received broad appreciation and application in the marine scientific community and have had wide-reaching impacts on the design of present monitoring programmes.

WGEAMS noted that it would welcome more input to its activities from its parent Committee. WGEAMS felt it appropriate that this point should be made now during the transitional period. The previous involvement of ACME in WGEAMS activities, for example, through the 'shadow' system has been of great benefit in giving WGEAMS broader perspectives and grounding in the interests and needs of the parent Committee and, through ACME, of ICES itself.

WGEAMS also expressed concern that the transfer of pollution-related Working Groups such as WGEAMS, MCWG, WGBEC, and WGMS to the new Marine Habitat Committee might reduce the emphasis on pollution-related matters in ICES. WGEAMS feels that it is possible that members of MHC might have particular interests in 'habitat' matters from a dominantly biological perspective and would have rather less experience or interest in questions covered by the Working Groups mentioned above. WGEAMS hopes that ICES will ensure that appropriate balances between different areas of marine science be maintained under the new Committee structure.

4.2 Report on the Outcome of the OSPAR Ad Hoc Working Group on Monitoring (MON)

OSPAR97 had decided that the Ad Hoc Working Group on Monitoring (MON) should assess contaminant concentrations in biota for the QSR 2000 at its 1998 meeting. MON98 was held at the end of February at ICES Headquarters.

Prior to the meeting, a preparatory group invited by the Chairman of MON organized the selection of data to be assessed, the selection of statistical methods to be applied and clear assessment reference concentrations (Background Reference Concentrations (BRCs) and Ecotoxicological Assessment Criteria (EACs), both as agreed by OSPAR97). On a rough average, 50 % of the datasets submitted were eliminated due to poor, incomplete, and lacking QA or information on QA. For statistical treatment, the preparatory group decided to calculate means for time series of 3–4 years, to apply linear trend methods for time series of 5–6 years, and to run non-linear trend methods for time series of 7+ years. Calculations were made by the ICES Environmental Data Scientist and were accessible for the preparatory group on 'restricted access' ICES web pages.

BRCs were available for some metals and PCBs in blue mussels and for mercury in fish. Since BRCs were given as ranges, the group proposed to use the upper concentration of each range for comparison with datasets. EACs were available as whole-body concentrations; therefore, concentrations had to be converted to concentrations in muscle or fat tissue before application. For assessing datasets MON agreed to compare modelled averages of 3–4 years time series and upper confidence limits of 5+ years time series with the reference values. There is no classification for elevated concentrations.

MON recognized that the geographical distribution of datasets was rather uneven. By far the majority of datasets originated from the North Sea. Little information came from the the Iberian coast and none from the Irish coast. The assessment of datasets against BRCs demonstrated that, in general, the upper confidence limits of most of the time series exceed the reference concentrations. The largest deviations from the references occured in river mouths and close to identified point sources such as metal smelters. There were very few cases where increased concentrations could not be explained. BRCs proved to be a useful tool for assessing field data.

Trend analyses on the 5+ years time series revealed that about 90 % of the series are insignificant with respect to linear and nonlinear trends. Explanations could be found for only a small number of the significant trends, i.e., in river mouths or close to point sources where strong input reductions are well documented.

WGEAMS concluded after a short discussion that the outcome of MON should motivate ICES and OSPAR to look for the reasons of the results of the trend analyses, whether they are an artefact of the statistical methodology applied or a consequence of inappropriate sampling and/or monitoring strategies.

4.3 OSPAR Working Group on Concentrations, Trends and Effects of Substances in the Marine Environment (SIME)

Mr Andrew Franklin gave a brief outline of activities at the last OSPAR SIME meeting held in November 1997 in Bonn.

At that meeting considerable discussion took place on the progress that had been made on the products that were to be provided to the QSR Regional Task Teams (RTTs). It was noted that time was now short (the RTTs had to complete their assessments by the end of the third quarter of 1998) and the last product which could be made available was likely to be the assessment of the temporal trend data on contaminants in biota by MON98 in February. Additional products might be provided directly for the holistic QSR 2000, but this would be only in very unusual circumstances.

In relation to continuing with existing monitoring programmes, SIME discussed progress on TBT, PCBs (especially in marine mammals) and other synthetic organics, oil (including effects of discharges of produced water), nutrients, and chemicals used in mariculture which led to co-operation between ICES and OSPAR; considerable progress has been made in developing the biological effects component of the OSPAR programme. A project proposal on quality assurance of biological effects measurements, led by the Chairman of WGBEC, was submitted to the EC in November 1997.

The Netherlands updated its review of national monitoring programmes. It was agreed that this updating could usefully be undertaken each year, as this would facilitate co-ordination and maximise the likely national contribution to the Joint Assessment and Monitoring Programme (JAMP).

Work had also been undertaken to provide a SIME contribution to the selection and prioritisation mechanism for hazardous substances.

With regard to the future, the SIME Chairman and Vice-Chairman had developed a Co-ordinated Environmental Monitoring Programme, the components of which would come from outstanding JAMP issues and gaps in knowledge identified during the production of the regional QSRs and QSR 2000. Both temporal (continuous) and spatial (intermittent) monitoring would be undertaken of both chemical contaminants and biological effects. Development would take place, for example, by commencing temporal trend studies in areas of concern indicated by spatial studies and by undertaking spatial studies where a temporal trend changed dramatically. It was hoped that future assessments would be possible on integrated biological and chemical data.

The main tasks for ICES in the current work programme for OSPAR are the completion of Technical Annexes on PAHs and the preparation of a document on planar CBs and their effects in marine mammals.

In response to questions from WGEAMS, the ICES Environment Adviser indicated that it was possible that ICES would be involved in a review of the QSR 2000 in late 1999, but the nature of the ICES role in the review process was not yet clear.

4.4 Helsinki Commission (HELCOM)

Dr Eugene Andrulewicz reported that the HELCOM Environment Committee Working Group on Monitoring and Assessment would meet in May 1998 to discuss the future of monitoring in the Baltic area (COMBINE Programme). The Working Group will plan the future shape of monitoring and may include a biological effects component. Some Guidelines for future monitoring are in place but others have yet to be completed. There is an increasing interest in the role of nitrogen and phosphorus in relation to plankton blooms. It has been agreed that COMBINE should start in 1998, making use of existing Guidelines where possible.

Dr Andrulewicz informed WGEAMS that the Fourth Periodic Assessment (covering the period 1993–1998) is in the planning stage. The report is due to be completed by the year 2000. Sweden has offered to take a leading role in the preparation of the assessment.

There was some discussion on how comments from WGEAMS (and WGMS) on the report of the Baseline Study of Contaminants in Baltic Sediments could be conveyed to HELCOM in time for them to be of use in the planning of the future programme. The ICES Environment Adviser agreed to take the comments to HELCOM in May on an informal basis, in order to make HELCOM aware that ICES has significant advice to offer and that HELCOM would benefit by waiting for ICES to formulate official advice on sediment monitoring, as it has requested of ICES.

Other links between HELCOM and ICES seem to be working well. There are a series of joint groups concerned with QA matters. Technical Notes are being prepared for QA of chemical monitoring and a biological QA manual will follow. As with other areas of ICES work, financial concerns have assumed greater importance and it is possible that some revision will be made to the formal Memorandum of Understanding between ICES and HELCOM.

4.5 Action List from 1997 Meeting

The Action List from the 1997 meeting of WGEAMS, with an overview of intersessional work carried out, is attached as Annex 4.

5 GENERAL FRAMEWORK FOR AN INTEGRATED APPROACH TO THE ASSESSMENT OF MARINE ECOLOGICAL QUALITY

5.1 Introduction

The ACME has identified a need to examine the current status of and possible future methods for integrating multiple measurements of different environmental factors. The WGEAMS was asked to discuss whether indices of ecological health could be developed and used in assessments, or if summaries of large monitoring data sets could be made adequate enough to assess ecological quality. A number of such indices are currently used to monitor, e.g, the diversity of benthic fauna communities, but their usefulness in assessing ecological quality has not yet been fully evaluated. The ACME has also asked WGEAMS to initiate a discussion on the possibility to explore multivariate models for linking environmental factors and observed ecological effects. Monitoring programmes generally produce large amounts of data, but the interpretations are usually made on single variables or on restricted data sets.

5.2 Ecological Quality

Ecological quality must be related to the structure and function of the ecosystem under study. Productivity, biodiversity, supporting functions for other ecosystems, e.g., recruitment of open sea fish stocks in coastal areas, are all important parameters expressing the quality of an ecosystem. There are also other aspects of ecological quality which should be considered, for instance, the introduction of alien species, the quality of bathing water, contamination of seafood, and the occurrence of endangered species. Assessments of environmental quality are therefore related to the actual or desired uses of an area, or to the objectives (e.g., reference status) of the environmental management of an area.

5.2.1 Assessing ecological quality

WGEAMS discussed the strategy of an ecological assessment. In general, the first step in assessing ecological quality is to make appropriate biogeographic delimitations of the assessed areas. In the Baltic Sea, e.g., there are accepted subregions based on morphometric characteristics, oceanographic conditions, and substrate compositions. The Baltic lagoons, the archipelagos, and the open southeastern Baltic sandy shores are examples. Similar delimitations could also be made for the North Sea and other marine areas. Within each biogeographic region, representative ecosystems can be defined, e.g., the littoral ecosystem in archipelagos. After defining the characteristic ecosystem, priority variables for assessment can be identified, based on ecological criteria, but also other criteria related to socioeconomics, conservation purposes, and recreational activities may be set up for assessment.

In some countries the governments have set up environmental targets, e.g., on nutrient levels in different sea basins. In these cases, the definition of ecological quality should be influenced by defined environmental objectives, and variables related to these targets should be given priority in the assessment even if they are not otherwise of primary importance in the target area.

5.3 Indices of Ecosystem Health

Eutrophication, chemical contamination, climatic variability and the effects of fisheries are generally considered among the main factors affecting the health of an ecosystem. It is, however, not possible to assess ecological quality by analysing every aspect of ecosystem structure and function, so different indicator systems have been developed which are used to reduce the complexity of large data sets and to summarise the data in a brief manner. There are many examples in the literature in which eutrophication, climatic variability, and, in some cases, the impacts of fisheries have been explored through the investigation of changes in community structure and by combining sets of observations to an index. The diversity index is perhaps the most well known of these indices. Indices of diversity, evenness, etc., are commonly applied to benthic faunal data but can be applied to other communities, such as planktonic communities or benthic macro-algal communities. Careful selection of the most appropriate community can sometimes provide information about specific aspects of quality, for example, the changes in benthic faunal community structure in response to organic enrichment are well recognised and there are examples in North America of the use of fish communities to provide similar information.

A further development in the diversity concept is to include an evaluation of the capacity of different species to respond to different kinds of environmental change. Some species are sensitive to organic enrichment, while others are favoured by this change. The presence or absence of particular indicator species provides information about the quality of water bodies, most frequently in fresh water.

A more sophisticated use of indicator species is employed in schemes such as the UK RIVPACS scheme. This scheme is based on a concept of the composition of benthic fauna in 'good quality' streams. Species that would be expected to occur under such circumstances are given a high score. Species typical of 'degraded' streams are given lower scores. Data on the presence or absence of species are then used to calculate a total score for a stretch of river, and to calculate other indices such as the average score per taxa. These indices are used to give overall impressions of the quality of streams and can be used in comparing communities with respect to a defined stress and to indicate whether conditions are generally improving or deteriorating. As mentioned above, these indices have been developed mainly for benthic freshwater fauna, but similar indices of community structure have also been tested in North America on freshwater fish fauna.

Schemes are currently in use in the UK freshwater systems to provide indices of quality based on a greater number of factors than just those provided by benthic faunal analysis. A standard range of quality parameters, such as dissolved oxygen, nutrient levels, contaminant levels in water, sediment or biota, etc., has been selected for use. Each variable is given a maximum score and a range of scores for conditions which are deemed to be less desirable than those conditions which would attract the maximum score. A consistent set of measurements of these variables is made for the area under study. The total score for each stretch of river is calculated and used to make comparisons of quality between different stretches or different rivers. As with all scoring schemes, while the generalities of the conclusions may be quite robust, the details of the conclusions are rather dependent on the selection of variables and on the details of the scoring scheme (relative weighting of variables, scaling of scores within variables, etc.).

Ecological impacts of chemical contamination may be best analysed on selected indicator or sentinel species. Assessments of ecological quality cannot be made from contaminant analyses and biomarker responses alone; hence, data on individual organisms and preferably also on populations should be included. However, it must be realised that most biological effects techniques are also sensitive to the variability of natural factors like temperature and season. Such basic information thus should be available and the analysis should be made by integrating multiple measurements of different factors.

WGEAMS discussed principles underlying the selection of variables to include in multivariable assessments of environmental quality. Two fundamental approaches to the problem were identified. Firstly, it is possible to recognise a series of stresses on the environment, for example, fishing, contaminants, eutrophication, tourism, natural variability, etc. It is then possible to attempt to define variables which respond strongly to these stressors. For example, commercial fish stocks respond to fishing pressure, and depth zonation of benthic algae is a response to light penetration which in turn can be affected by increased turbidity due to eutrophication.

In some cases it may be possible to define variables which are totally, or almost totally, controlled by a single stress factor. However, it is more common that ecological variables respond to several factors, recognised and unrecognised. For example, benthic algal zonation will also be affected by turbidity from riverine discharges, or by water colour. This gives rise to the second fundamental approach in which the variables to be measured are selected on the basis of expert judgement in that they reflect aspects of the quality of the environment that have value in an ecological, economic, aesthetic, or other sense. For example, it is probably desirable to have seal populations in appropriate locations, although it may not be immediately clear what aspect (or aspects) of environmental quality their presence may reflect.

This approach gives rise to two series of measurements: one series on the intensity of stresses on the environment and one series on measurements of variables within the environment. The interpretation of the relationships between changes in the values recorded for items in the two series is not overly straightforward, and WGEAMS has been unable to simplify this situation. However, it may be possible to distinguish between environmental variables which are very closely linked with a single stress factor and others which are linked to several stress factors with different degrees of strength of linkage. The web of relationships, hopefully quantifiable relationships, created by such an analysis may be amenable to numerical analysis.

WGEAMS concluded that the use of multivariate models could be a possible way to improve assessments, but that it is important that all relevant variables involved in elucidating a biological effect should be included in the model, and not just the data available at the moment. For example, it is known that biomarker responses such as the activity of the liver detoxification system (EROD) cannot always be closely linked to only contaminant concentrations. It is known that EROD responds to a range of contaminants, including dioxins, some CBs, and some PAHs. Chemical analysis of the fish can help in the interpretation, but while CBs may largely be retained and available for measurement, PAHs may well be metabolized and lost. It is also known that EROD activity is influenced by water temperature, season, sex, maturity, lipid metabolizm, etc., as well as by post-capture handling and storage. The successful interpretation of EROD data is therefore dependent on the strict control (or measurement) of other factors known to influence the result. It is also necessary to have some understanding of the functional relationships between EROD and the various factors concerned. This problem is further compounded if higher order biological responses, e.g., reproduction or growth, are included in the assessment. The interpretation of the data therefore becomes a multivariate problem which may be amenable to numerical modelling approaches. However, reliable interpretation requires that supporting measurements are made of other variables known to influence the biological response so that an observed effect may be related to its cause, be it, for example, in response to habitat variations, natural or anthropogenic, or to exposure to a toxin.

6 IDENTIFICATION OF PRIORITY CONTAMINANTS

WGEAMS was aware that this topic has been the subject of considerable discussion at several recent OSPAR working group meetings; the summary records of the latest such meetings were checked before WGEAMS considered whether to commence any updating work on its 1991 review of methods for the identification of priority contaminants. Also available to the meeting was the latest version of the draft OSPAR strategy document on hazardous substances. This is a substantial document and it was noted that a very full list of candidate hazardous substances has been developed by collating lists from a number of different sources. The WGEAMS task had made specific reference to those contaminants listed in the Esbjerg Declaration; they are also included in the OSPAR strategy document list.

It was obvious from the length of this list that the development of a prioritisation mechanism is essential. The summary record of the latest meeting (October 1997)of the OSPAR Working Group on Diffuse Sources (DIFF) indicated that this Working Group recommended the formation of an OSPAR ad hoc working group to specifically develop a dynamic selection and prioritisation mechanism (DYNAMEC) for hazardous substances; PRAM98 will consider this proposal. In addition, SIME has developed a series of action steps which could be taken to aid the selection mechanism process.

In view of the considerable activity already underway on the identification of priority contaminants, WGEAMS felt it would not be appropriate to spend time updating its earlier review on this topic. However, a number of points were made during discussions on the present draft OSPAR strategy document:

- The prioritisation scheme places considerable weight on acute toxicity effects—the SIME contribution, for example, emphasises that toxicity threshold levels should be used in accordance with the procedure for development of Ecotoxicological Assessment Criteria. Some concern was expressed that chemicals which are toxic only in the long term might be missed. It was noted that, in the OSPAR draft strategy document, 'toxicity' has a very broad definition which includes carcinogenicity, mutagenicity, and teratogenicity and a classification to encompass acute, sub-chronic, and chronic effects. Considerable steps have been made to include also endocrine disruptors in the list of hazardous substances. Assuming that these aspects were taken into account in any routine assessment of priority contaminants, the concerns expressed would hopefully be met in relation to other mechanisms of chronic toxicity.
- The definition of an 'endocrine disruptor' given in the glossary (page 21) of the document was broad, but the evaluation criteria in paragraph 4.6 were strongly directed to sex hormones. The development of tools to detect disruption of other hormone systems is required.
- It was noted that some of the terminology could usefully be updated as the language 'dates' some of the documents from which lists of substances have been drawn. For example, the generic term 'PAH' is not very useful.
- There were queries from WGEAMS members on why some of the substances, e.g., volatile solvents, had been included in a list of hazardous substances; it was assumed that they would be selected out by the prioritisation mechanism.

- There was considerable interest in how effects from combinations of possible hazardous substances could be dealt with. WGEAMS considered it unlikely that the proposed prioritisation system could take account of synergism, antagonism, and other similar interactions.

7 REVIEW OF THE COMPONENTS OF THE MONITORING GUIDELINES FOR PAHs IN BIOTA (BY MCWG) AND SEDIMENTS (BY WGMS), INCLUDING SAMPLING BY SGMPCS, FOR OVERALL COHERENCE AND COMPLETENESS [OSPAR 1998/1.1]

A draft version of the 'Guidelines for the determination of polycyclic aromatic hydrocarbons (PAHs) in biota' was presented to the organic sub-group of the Marine Chemistry Working Group (MCWG) at their meeting in Stockholm in March 1998. However, MCWG was unable to edit the draft during the meeting and had agreed to work on it intersessionally with a view to completing the draft in time for the 1998 ACME meeting.

WGEAMS noted that this task had been referred back to MCWG after it was decided that the original draft available in 1997 was unsuitable for its desired purpose, i.e., to be added to the series of Technical Annexes of the OSPAR JAMP Guidelines. The failure of MCWG 1998 to prepare a draft to meet this purpose meant that the system of review established by ACME for this paper, i.e., review by WGEAMS in association with papers on sampling and determination of PAHs in sediments, could not take place.

WGEAMS expressed concern that this agenda item could not be completed, as it was included in WGEAMS' Terms of Reference in direct response to an item on the OSPAR Work Programme. It was not possible for WGEAMS to recommend that the paper be included as an Annex to the ACME report (as recommended by MCWG) without having seen the paper. However, WGEAMS noted that the primary purpose of the paper is to serve as a Technical Annex to the JAMP Guidelines and that therefore it should be assessed on its suitability for that purpose, rather than on its suitability as an Annex to the MCWG or ACME reports.

As the meeting of WGEAMS was closing, a draft text prepared by members of MCWG on PAH determination in biota arrived at ICES HQ and was passed to WGEAMS. It was not possible to review the document during the meeting, but several members provided comments to the Chairman for inclusion in the report of the meeting.

WGEAMS members recognised that the current draft was a positive move forward from the version available to WGEAMS in 1997. ICES had been asked to provide monitoring guidelines for PAHs in biota, but the request did not specify which biota should be covered. According to the current JAMP documents, contaminant monitoring for PAHs should be carried out in shellfish, but there may be a need to analyse fish tissues in relation to biological effects work. It is therefore not a straightforward task to provide advice that precisely matches OSPAR needs, and it will be necessary for OSPAR to restructure the advice to suit details of OSPAR format and content, etc. In particular, it will be necessary for OSPAR to format the document in such a way that it does not duplicate or conflict with aspects of their Monitoring Guidelines already covered in other documents.

Additional comments are provided below.

1) The *Introduction* provides useful background information, provided it does not conflict with other JAMP documents. It would probably be helpful to state either in the Introduction, or in a covering note to the advice, that it has been aimed primarily at contaminant monitoring, and therefore does not cover fish.

2) *Appropriate species ...*

   Delete the first paragraph as far as 'in their food' and start with 'All teleost fish ...'.

   Delete the last sentence: 'Those fish and ...'.

3) *Sampling*

   It is the understanding of the Chairman that the JAMP Guidelines for contaminant monitoring only call for the analysis of mussels, not fish. Therefore, the parts of this section referring to fish should probably be deleted. The 1997 ACME report, Section 4.4.1 clearly states that MCWG should prepare a draft technical annex on the analysis of PAHs in mussels.

   It was not known whether JAMP papers have already defined the method of size stratification (if any) that should be used when sampling mussels. The authors or OSPAR should check that this section of the paper does not contradict pre-existing JAMP papers.

4) *Pretreatment and Storage*

   The sections on fish should be deleted.

5) *General*

WGEAMS noted that the draft raises the quaestion as to which PAHs should be included as monitoring targets. It will be necessary for OSPAR to define which PAHs should be determined (if this has not already been done).

The rest of the text should be carefully checked and references to fish removed.

The technical analytical matters are well covered in the '*Analysis*' and subsequent sections.

8 REVIEW INFORMATION AND DATA ON CONCENTRATIONS OF NON-*ORTHO* AND MONO-*ORTHO* CBs IN MARINE MAMMALS AND RELEVANT BIOLOGICAL EFFECTS AND PREPARE A REPORT ON THE FINDINGS AND IMPLICATIONS, AS A CONTRIBUTION TO THE OSPAR QSR 2000 (WITH MCWG, WGMMHA AND WGBEC) [OSPAR 1998/3.1]

ACME had arranged that the preparation of material for this item on the OSPAR Work Programme should be undertaken collaboratively by four Working Groups. Prior to consideration by WGEAMS 1998, contacts had been made with the other Working Group Chairmen involved and arrangements had been made to receive text from WGMMHA and from MCWG, and for work to be undertaken intersessionally with WGBEC immediately after the WGEAMS meeting in order to complete the task.

Text was received from WGMHHA, but MCWG had been unable to prepare any text. MCWG had experienced difficulty in obtaining information through a questionnaire issued by Norway (as lead country). ICES had provided information from the ICES Environmental Database, but MCWG felt that more data were available elsewhere. MCWG therefore put into place several initiatives to collate additional information on 'CBs in marine mammals' and to make the results of these initiatives available to the ICES Environmental Data Scientist in April 1998.

WGEAMS noted that this task had been structured to combine the expertise of four Working Groups. The failure of MCWG 1998 to prepare a draft of their contribution meant that it is unlikely that two of the other elements of the system, i.e., consideration by WGBEC and WGEAMS, can be carried out with optimal effectiveness.

WGEAMS expressed concern at this situation, as this task was included in WGEAMS' Terms of Reference in direct response to an item on the OSPAR Work Programme as a contribution to the QSR 2000, and that therefore completion of the task had direct relevance to the reputation of ICES with a major customer.

WGEAMS noted that the final Working Group to consider the paper should be WGBEC, which would be meeting the week after the WGEAMS meeting. WGEAMS therefore reviewed the paper prepared by WGMMHA and undertook the following tasks:

1) Editorial proposals were made concerning the elimination from the text of material not directly concerned with non-*ortho* or mono-*ortho* CBs.
2) Advice was obtained from C. McKenzie (Aberdeen, UK) on aspects of the scientific content of the paper and his recommendations (updating the text and including additional information) were added to the text.
3) Additional text and data tables from a recent publication by McKenzie *et al.* concerning planar CB concentrations, calculation and estimation of TEQ values, and the trans-generational transfer of contaminants were added to the draft.
4) Additional tables of analytical data supplied to ICES by laboratories in Member Countries following the 1998 MCWG meeting were added as an annex to the text.
5) The revised and expanded document was sent to the Chairman of WGBEC on 27 March 1998 for their consideration and amendment.

The information on CBs in marine mammals collated for consideration by WGBEC is attached as Appendix 1.

9 REVIEW INFORMATION ON MONITORING STRATEGIES ASSESSING HAZARDS PRESENTED BY THE DISCHARGE OF PRODUCED WATER BY OFFSHORE OIL AND GAS INDUSTRIES

9.1 Produced Water

A summary of amounts and composition of produced water from the offshore oil and gas industry was presented in a paper by Lars Føyn. The paper also briefly discussed the possible impacts of produced water as described below.

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	130×10^6 m^3
1990	146×10^6 m^3
1991	160×10^6 m^3
1998*	340×10^6 m^3

* = expected

A breakdown according to the different sectors of the North Sea is indicated below.

Year	Total Annual Discharge	Sector
1991	330×10^6 m^3	UK
1991	16.4×10^6 m^3	Norway
1998*	330×10^6 m^3	UK and Norway
1991	10.7×10^6 m^3	Netherlands
2000*	5.2×10^6 m^3	Netherlands
1991	1.5×10^6 m^3	Denmark
2000*	9.7×10^6 m^3	Denmark

* = expected

Typical water production rates from different types of installation (Anon., 1994) are:

Installation Type	Production Rate
Oil	2 400–40 000 m^3 day^{-1}
Gas	1.6–30 m^3 day^{-1}

Data for discharge into the Norwegian sector of the North Sea are summarized by Anon. (1998):

Year	Total Annual Discharge
1996	66×10^6 m^3
2000*	120×10^6 m^3

* = expected

Røe *et al.* (1996) estimated that discharges from the Norwegian sector of the North Sea were 26×10^6 m^3 in 1993 and will be as much as 90×10^6 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 Oslo and Paris Commissions (OSPAR) Secretariat in London, but this reporting has been insufficient in recent years (Anon., 1996). There are many reasons for this 'not so good' 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 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 x 10^6 m^3 will be used to calculate discharges of dissolved components to the Norwegian sector and, correspondingly, an annual total of 300 x 10^6 m^3 discharges into the North Sea.

9.1.1 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

Trace metals

The concentration of dissolved salts is usually higher than in sea water, but may range from about 3 g l^{-1} 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 comprise trace metals in considerably higher concentrations compared to the concentration in sea water.

Table 1. Examples of concentration ranges in produced water. [Source: Anon., 1994, 1995.]

Pb µg l^{-1}	Cd µg l^{-1}	Cu µg l^{-1}	Hg µg l^{-1}	Ni µg l^{-1}	Zn µg l^{-1}	As µg l^{-1}	Cr µg 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

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 contents 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 according to an OSPAR prescribed analytical method and mainly concerns aliphatic hydrocarbons as dispersed oil (Anon., 1994). WGEAMS is not aware of any national or international regulations for the dissolved components in produced water. This means that 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 the measuring of 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^{-1}; acetic acid comprises about 90 % (Anon., 1994, 1995). For further calculations an average concentration of 500 mg l^{-1} is 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.* (1996a) 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 2 concentration data on the so-called BTEX (benzene, toluene, xylenes and ethyl benzene) components from Stagg *et al.* (1996a) and Anon. (1994, 1995) are compiled.

Table 2. Concentrations (in mg l^{-1}) of BTEX components in produced water from oil and gas platforms in the North Sea.

Compound	Clyde[1]	Forties Charlie[1]	Brent Delta[1]	Brae Alpha[1]	Average oil[2]	Average gas[2]	Tampen[3]
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	
Σ BTEX	10.8	4.7	12.6	12.6	0.5–14	5–629	1–7.3

1 = Stagg *et al.* (1996a)   2 = Anon. (1994)   3 = Anon. (1995).

Phenols

The concentration of phenols in produced water from the North Sea varies between 1–23 mg l^{-1} (Anon., 1994), 1.3–8 mg l^{-1} (Anon., 1995) and 1.2–1.5 mg l^{-1} (Stagg *et al.*, 1996). Based on these data, an average concentration of phenols of 5 mg l^{-1} is assumed and is used in further calculations.

Polycyclic aromatic hydrocarbons (PAHs)

The discharge of PAH-containing compounds is considerable, ranging from 40–1600 µg l^{-1}, with naphthalene (41–1600 µg l^{-1}), phenanthrene (10.7–500 µg l^{-1}) and dibenzothiophene (10–170 µg l^{-1}) representing the majority (Anon., 1995). Variations in the reported concentrations, however, are considerable. Anon. (1994) reports values for naphthalene (66 µg l^{-1}), phenanthrene (< 2 µg l^{-1}), and dibenzothiophene (0.5 µg l^{-1}), while Stagg *et al.* (1996a), for example, report concentration values for methyl naphthalenes between 770–1700 µg l^{-1} and phenanthrene concentration values from 15–50 µg l^{-1}. Based on these figures, the assumed average PAH concentration of 300 µg l^{-1} in general use may be rather conservative.

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 3 summarizes the discharges of some potentially harmful major components into the Norwegian sector and into the whole North Sea.

Table 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 mg l^{-1}	BTEX 8 mg l^{-1}	Phenols 5 mg l^{-1}	PAHs 300 µg l^{-1}	Hg 5 µg l^{-1}	Cd 10 µg l^{-1}
1 x 10^8 m^3 (Norwegian Sector)	50,000 t	800 t	500 t	30 t	0.5 t	1 t
3 x 108 m^3 (Whole North Sea)	150,000 t	2400 t	1500 t	90 t	1.5 t	3 t

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 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 mg l^{-1}. Assuming an average concentration of 20 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 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) reports COD values for produced water in the North Sea from 100–15,800 mg O_2 l^{-1} and for oil platforms in USA waters from 100–3000 mg O_2 l^{-1}, with an average value for central North Sea oil platforms at 4160 mg O_2 l^{-1}. The reported BOD values vary between 28 mg O_2 l^{-1} to 6700 mg O_2 l^{-1}.

By using an average COD or BOD value of 4000 mg O_2 l^{-1} for produced water and assuming an oxygen content in North Sea water of 8 mg O_2 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 produced water annually discharged into the North Sea of 3 x 10^8 m^3 will then, based on these rough estimates, deplete the dissolved oxygen of approximately 15 x 10^{10} 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 model (Schobben et *al.*, 1996) 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 analyzed 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.

9.1.2 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 diffusor 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 for the actual field/platform are presented. Such presentations never consider 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 1 presents an example from the Tampen study (Anon, 1995) where oil and gas fields nearby are not shown on the figures describing the area influenced by the produced water.

Figure 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.* (1996b) present *in situ* measurements of oil hydrocarbons in the northern North Sea demonstrating a distinct burden of oil components on the whole area.

Figure 2, borrowed 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 what the distance from the discharge point in the vertical distributions is meant to be.

Reed *et al.* (1996) present 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 bioaccumulation and biomagnification of these two substances will be small.

Stagg et *al.* (1996b) 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 a relatively short run of the models.

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 note that the possible effects of multiple components need to be addressed.

Given the current patterns of the North Sea, contaminants entering into an area will sooner or later be transported out of the North Sea. The majority of this water transport will take place to the north 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.

Figure 1. Modelled horizontal distribution of concentration fields in percent of produced water in 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 are not included in the modelled distribution.

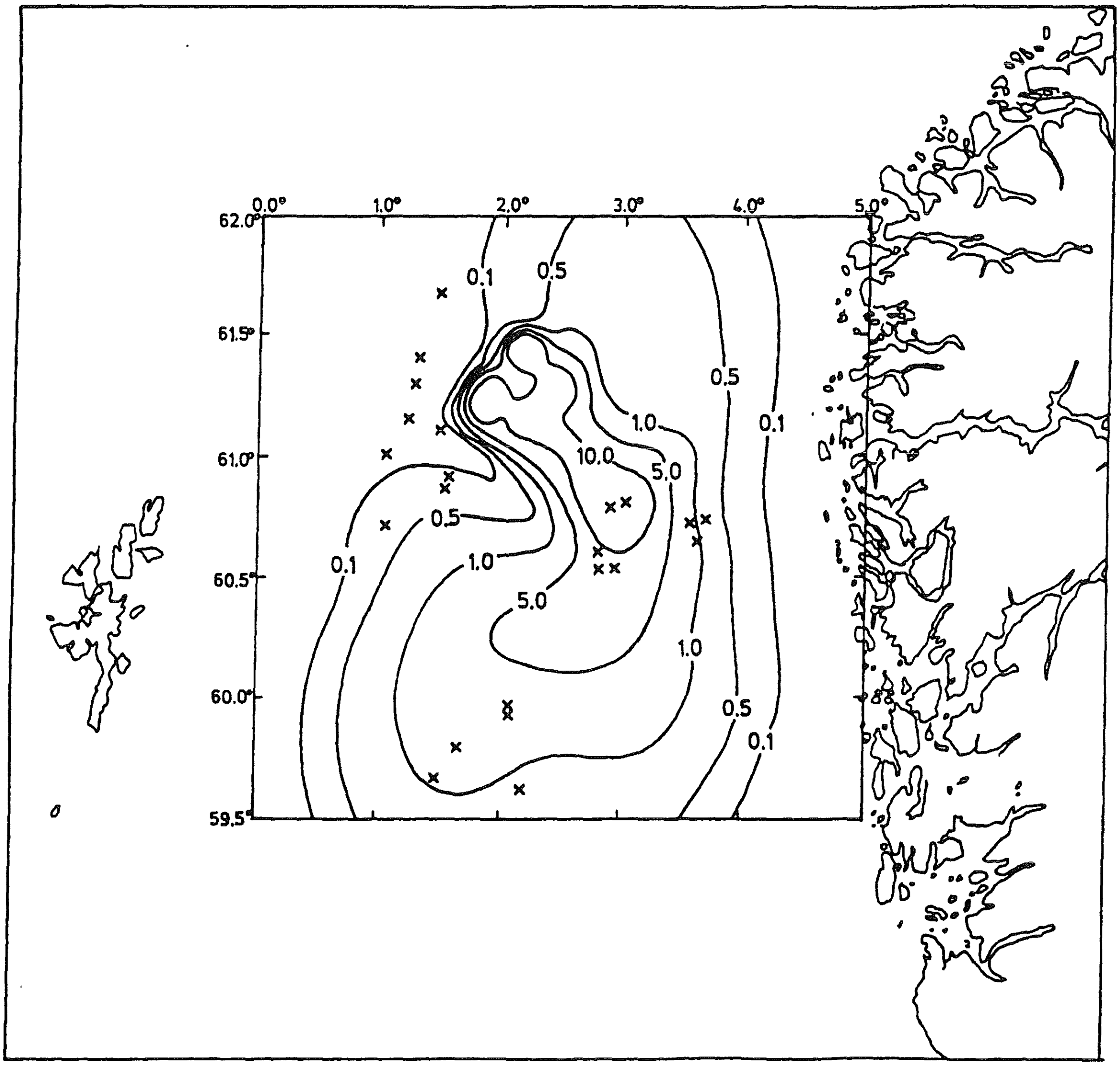
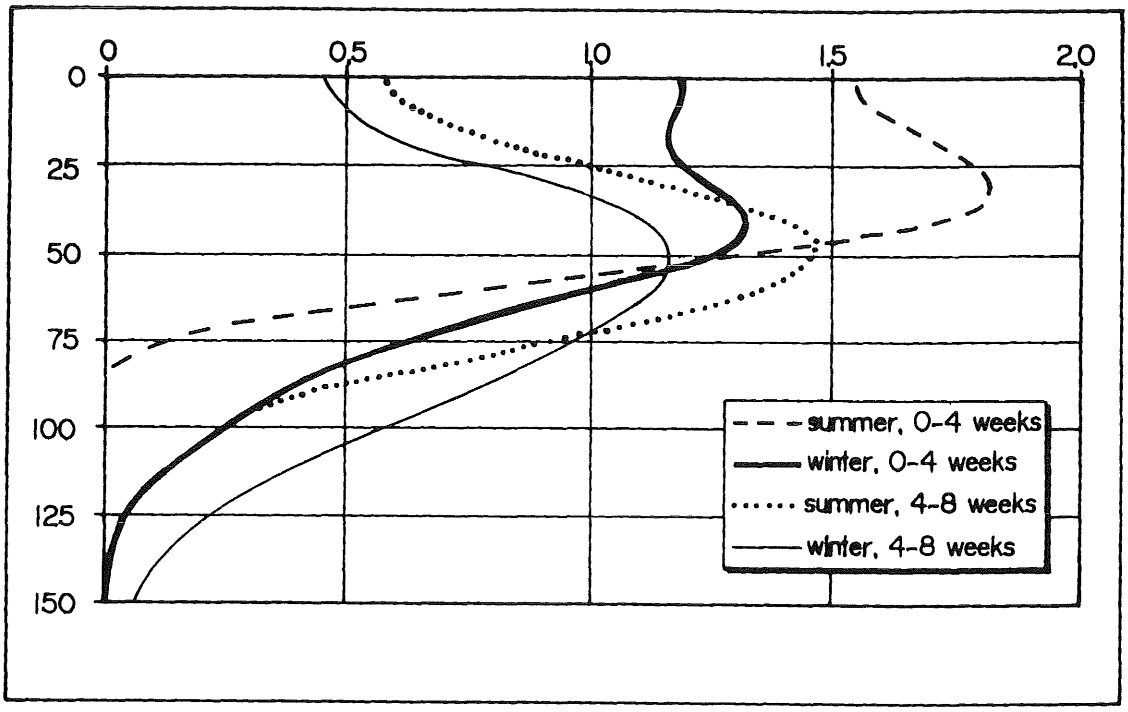


Figure 2. Modelled vertical distribution of produced water discharged during 8-week periods during summer and winter months. The distributions are given as percent share per depth meter (Anon., 1995).



As can be seen in Figure 2, the plume of produced water occurs between depths of 25 and 50 meters and this is the part of the water column where most planktonic organisms are found. There is obviously a need to run models that comprise all discharge points and of 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.

9.1.3 Effects of produced water on the marine environment

Produced water contains a wide variety of dissolved components, some organic and some inorganic. Each well has a different composition; both the composition and the quantity of 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 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 off-shore oil and gas production. Within OSPAR the discharge of produced water is not regulated.

Produced water is seldom tested as such. Reported tests are performed most often on single chemicals or groups of chemicals like 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 as those determined in their experiment are not likely to occur in the field given dilution factors of more than 1000.

Stagg *et al.* (1996b) observed significant gradients of EROD activity in gadoid and sand eel 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.*, 1996). 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.* (1996a) conclude 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.*, 1996a).

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 in 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, i.e., northern North Sea, basis.

The huge 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 finally 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. com.) 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.

9.1.4 References

Anon. 1998. Konkretisering og analyse av et fremtidstilpasset virkemiddelsystem for olje- og gassvirksomhet. Rapportutkast. Vedlegg A: Utslipp fra norsk petroleumvirksomhet under dagens virkemiddelbruk. (Realization and analysis of a future-adjusted (regulatory) system for oil and gas activity. Draft report. Appendix A: Discharge from Norwegian petroleum activity in the present system of measures.) Novatec A.S., SINTEF, Rogaland Research. In preparation. [In Norwegian.]

Anon. 1996. Oljeindustrien tar ansvar. Rapport fra MILJØSOK Styringsgruppen. (The oil industry take responsibility. Report from the steering group of MILJØSOK). Oslo. 139 pp. [In Norwegian.]

Anon. 1995. Regional konsekvensutredning for Tampenområdet. (Regional impact assessment for the Tampen area.) Presented by Norsk Hydro, Technology and Development Division, in cooperation with Statoil and Saga Petroleum. Oslo. 83 pp. [In Norwegian.]

Anon. 1994. North Sea produced water: fate and effects in the marine environment. E & P Forum Report No. 2. 62/204, London. 48 pp.

Booman, C., and Føyn, L. 1996. Effects of water soluble fraction of crude oil on marine fish larvae and crstaceans, pp. 149–162. *In* Produced water 2: Environmental issues and mitigation technologies. Ed. by M. Reed and S. Johnsen. Plenum Press, New York.

Krause, P.R., Osenberg, C.W., and Schmitt, J.R. 1992. Effects of produced water on early life stages of a sea urchin: stage-specific responses and delayed expression, pp. 431–444. *In* Produced water. Ed. by J.P. Ray and F.R. Engelhart. Plenum Press, New York.

Reed, M., Johnsen, S., Melbye, A., and Rye, H. 1996. PROVANN a model system for assessing potential chronic effects of produced water, pp. 317–330. *In* Produced water 2: Environmental issues and mitigation technologies. Ed. by M. Reed and S. Johnsen. Plenum Press, New York.

Røe, T.I., Johnsen, S., and The Norwegian Oil Industry Association (OLF). 1996. Discharges of produced water to the North Sea. Effects in the water column, pp. 13–25. *In* Produced water 2: Environmental issues and mitigation technologies. Ed. by M. Reed and S. Johnsen. Plenum Press, New York.

Schobben, H.P. M., Vik, E.A., Hutjes, G.G., Karman, C.C., and Øfjord, G.D. 1996. An introduction to the CHARM model, pp. 295–301. *In* Produced water 2: Environmental issues and mitigation technologies. Ed. by M. Reed and S. Johnsen. Plenum Press, New York.

Stagg, R., Gore, D.J., Whale, G.F., Kirby, M.F., Blackburn, M., Bifield, S., McIntosh, A.D., Vance, I., Flynn, S. A., and Foster, A. 1996a. Field evaluation of toxic effects and dispersion of produced water discharges from North Sea oil platforms, pp. 81–100. *In* Produced water 2: Environmental issues and mitigation technologies. Ed. by M. Reed and S. Johnsen. Plenum Press, New York.

Stagg,, R.M., Gillibrand, P.A., McIntosh, A. M., and Turrell, W.A. 1996b. The effects of produced water on hydrocarbon levels and P450 1A monooxygenase activity in fish larvae in the northern North Sea, pp. 195–210. *In* Produced water 2: Environmental issues and mitigation technologies. Ed. by M. Reed and S. Johnsen. Plenum Press, New York.

Stephenson, M.T. 1992. A survey of produced water studies, pp. 1–11. *In* Produced water. Ed. by J.P. Ray and F.R. Engelhart. Plenum Press, New York.

9.2 Discussion

In the discussion WGEAMS noted that there are currently no regulatory mechanisms within OSPAR or the national authorities for handling the discharge of produced water into the marine environment.

The amounts of produced water that are continuously discharged into the North Sea and the predicted increases with the ageing of the oil wells are alarming. The possibility of negative effects on marine organisms necessitates more efforts in determining the real impacts of these discharges. OSPAR has started work to include produced water in its regulations and monitoring efforts.

Regional studies should be undertaken to investigate effects on:

- pelagic communities from bacteria through phytoplankton to zooplankton;
- fish eggs and larvae;
- 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 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 impact of produced water:

1) verify modelling results by field surveys;
2) encourage development of more sensitive short-term tests;
3) develop tests for chronic effects on fish eggs/larvae;
4) encourage mesocosm experiments and field experiments;
5) experimental studies about the development of fish eggs/larvae;
6) investigations of the effects of produced water on the activity and compositon of bacterial communities;
7) field studies should take advantage of gradients and also take into account the variability of components found in produced water.

WGEAMS noted the critical importance of having oil companies be required to provide data on the composition and amounts of discharged components to national authorities on a regular basis in order to assess possible impacts both for contamination and for secondary effects such as oxygen reduction, bacterial growth, and so on.

10 ROLE OF ICES IN MARINE ENVIRONMENTAL MONITORING AND ASSESSMENT IN RELATION TO THE ACTIVITIES OF THE EUROPEAN ENVIRONMENT AGENCY (EEA) AND THE EUROPEAN TOPIC CENTRE (ETC) ON MARINE AND COASTAL WATER

WGEAMS invited Mr Evangelos Papathanassiou from the European Environment Agency (EEA) in Copenhagen to provide information about the work of the EEA and its concomitant European Topic Centre on Marine and Coastal Waters (ETC/MC) in La Spezia, Italy. The main goals of the EEA are to establish and coordinate the work of national environmental focal points and topic centres (EIONET), to develop and publish reports on the state of the environment on a regular basis (every five years in the DOBRIS+ reports), and to provide information for the development of environmental policy of the EU and its member states (Council of Ministers, European Parliament, the commissions, etc.). The ETC's main tasks are to develop methodologies for the identification and description of pressures, stresses on and states of the environment, and subsequent needs for action such as research or measures. It should facilitate exchange of information on monitoring between the regional conventions via the 'Inter-regional Forum' and further harmonisation of monitoring activities, assessments and related research between conventions. An important issue is to stimulate coastal zone management activities by developing methodologies and constructing a coastal zone database.

Reporting of the EEA is mainly based on raw data supplied by national data centres from Member Countries (NODCs). There is no organized assessment procedure. Reports are written by the ETC/MC, sometimes supported by consultants. Reports undergo a peer review process via national environmental agencies. With regard to coastal zone management (CZM), the ETC/MC is in the process of applying for funding to install a GIS-type database on environmental and socio-economic information from a coastal band (10 km on both sides of the shore line).

WGEAMS discussed current and future links between ICES and the EEA.

10.1 Provision of Data Bank Support and Data Products for EEA Assessments

The EEA relies heavily on data collected for the Commissions and held at ICES HQ. There is currently no formal agreement between ICES and the EEA for payment for the work necessary to provide data to the EEA.

Possible funding mechanisms are:

a) the Topic Centre contracts ICES to provide data. This is currently unlikely as the existing contracts with the Topic Centre do not allow for this. Future contracts might have such an element. However, the size of the contract with the Topic Centre (6 man years per annum) was not large, and ICES' costs might not fit well with the budget.

   The contract will be re-negotiated soon and changes to accommodate ICES might be possible.

b) OSPAR might provide additional funds to ICES to undertake work on OSPAR data for the EEA.

   This was considered to be rather unlikely as OSPAR budgets are tightly controlled.

c) It might be possible for the EEA to contract ICES directly in order to support the work of the Topic Centre.

10.2 Provision of Scientific Advice

The EEA representative believes that this is unlikely, as they have contracts with the Topic Centres, who in turn could call upon their partners, or external consultants, for advice. Again, ICES costs might seem relatively high in comparison.

It seems appropriate that the EEA should in some way be able to have access to, or buy into, ICES' role as provider of scientific advice. WGEAMS was not clear how this could be achieved. ICES does not cover the full geographical area of interest of the EEA. ICES normally works on longer timescales than the EEA. WGEAMS considered that the way ahead might be through the active participation of ICES in the annual Inter-Regional Forum in Rome. This gathering of the EEA and its Topic Centres with the regional Commissions, regional seas groups, etc., could be an opportunity for ICES to demonstrate its potential role in various aspects of the EEA work, for example, the harmonisation of monitoring programmes and methods, standardisation of QA, development of statistical tools for data analysis, development of consistent methods for the presentation of data—tasks that are central to the EEA's aims and within the established expertise of ICES.

10.3 Quality Assurance of Data

The EEA took the view that all data submitted to the Commissions was 'approved' in some way and therefore of suitable quality for their purposes. This might be a fairly good assumption for recent OSPAR and HELCOM data, but QA is rather less well-developed in other areas of the EEA's interests. The EEA expected that data from the MEDPOL area would be banked within the MEDPOL area, and that the Monaco laboratory would provide QA support.

ICES has recognised that it is necessary to develop good and productive working arrangements with the EEA. The provision of data has caused some problems, for example, in relation to permission to release data and the timescale on which they are required. WGEAMS supported the suggestion that it would be more sensible for the EEA to be provided with partly-interpreted data (e.g., aggregated by station, or summarised in some other way), rather than raw data. As MON reports are fairly irregular, it is conceivable that ICES could create a regularly updated presentation of its information for use by the EEA. WGEAMS recommends that ICES should continue discussions with the EEA and the Commissions with a view to clarifying the needs of the EEA and developing a mechanism whereby these needs can be met, including appropriate funding arrangements.

11 REVIEW INFORMATION COLLATED INTERSESSIONALLY ON PROCEDURES TO ASSESS THE COMBINED EFFECTS OF EXPOSURE OF ORGANISMS TO GROUPS OF CHEMICALLY SIMILAR, OR DISSIMILAR, CONTAMINANTS

WGEAMS was unable to make a comprehensive response to this agenda item during the meeting, however, it was recognised that the interactions between contaminants with respect to biological responses remains a difficult and largely unresolved issue in toxicology and marine environmental science. 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 development 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.

WGEAMS did, however, consider a few examples of interactions between groups of contaminants. The first example is 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 P450 1A1 response to xenobiotics. It is known that dioxins effectively induce this enzyme system, as do some CB congeners. A system of Toxic Equivalent Factors (TEFs) has been developed, using 2,3,7,8-TCDD as the reference compound, to allow total effective toxicity of mixtures of dioxins and CBs to be estimated. However, it is also known that some PAH compounds can induce the same system. There is a system of factors linking PAH compounds, but these are based on benzo[*a*]pyrene as the reference compound and, therefore, are not directly linkable to the dioxin-based system. The following text has been taken from the 1998 Ph.D. thesis of Philip Hess (FRS Marine Laboratory, Aberdeen) who worked on samples from the Firth of Clyde in Scotland. The author has given WGEAMS permission to quote the following material:

> 'A common mechanism of action in the toxicity of planar aromatic compounds was their interaction with the P450 1A enzyme system. A model inducer of the P450 1A 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_{50}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^{-1} to several µg kg^{-1} 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 highest 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 by 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 (P450 1A1) 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 ß-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 (Forlin *et al.*, 1986).

The final example 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 noninfectious 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 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 did arise after sequential or simultaneous exposure to diesel and resin acids (Croce and Stagg, 1997; Croce *et al.*, 1997).

WGEAMS noted that the nature of the interactions between contaminants are by no means always predictable and that elucidation of interactions normally required a well-planned and careful experimental programme. These processes are generally not yet taken into account in the development of discharge control standards or international monitoring programmes. WGEAMS agreed to defer further discussion of this important topic to its 1998 meeting.

References

See Annex 3.

12 PREPARE A SHORT REPORT ON WEBSITES PROVIDING INFORMATION OF RELEVANCE TO THE WORK OF WGEAMS

WGEAMS reconsidered the need for this agenda item. It was concluded that the task is impractical (as any list would rapidly become out of date and it would be extremely difficult to cover the range of subjects that have been considered by WGEAMS at recent meetings) and unnecessary (as Web search engines and similar services are very effective and provide convenient up-to-date indexing of websites). WGEAMS therefore decided not to undertake any further work in this area at this time.

13 REVIEW OF THE FINAL REPORT OF THE BASELINE STUDY OF CONTAMINANTS IN BALTIC SEA SEDIMENTS

WGEAMS had reviewed an earlier draft of the report of the Baseline Study of Contaminants in Baltic Sea Sediments at its meeting in 1997. A further draft had been prepared since that meeting and was presented to WGEAMS for review. The current draft had also been reviewed by WGMS 1998, and their comments were available in a draft report.

General comments

WGEAMS reviewed the final report of the 1993 ICES/HELCOM Baseline Study of Contaminants in Baltic Sea Sediments and is of the opinion that the results of the Sediment Baseline Study should be published within the ICES system. Results should be published for at least two reasons:

- This study provides a lot of very good scientific material which leads to a better understanding of the nature of the Baltic Sea sediments as well as of processes within the sediments.
- This study is important for the HELCOM Monitoring Programme because it provides important information on how to run sediment monitoring in the Baltic Sea area.

While the scientific content of the report is in parts very good, the report has many flaws which have to be improved before publication:

- There are important discrepancies between the scientific outcome of the study and the recommendations for future monitoring. The 'Recommendations' should be reconsidered taking into account the outcome of the chapter on metals in sediment cores and the advice given by WGEAMS.
- Extensive editorial improvements have to be made before printing.

The following general remarks should be considered while preparing the report for publication:

WGEAMS proposes to change the title of the report from 'Contaminants in the Baltic Sea Sediments' to 'Baseline Study on Contaminants in Baltic Sea Sediments'

The report should include an abstract which summarizes the recommendations for future monitoring.

The structure of the different chapters in the report should follow generally accepted publication standards.

For a cohesive scientific overview, WGEAMS proposes the following order of chapters:

- General settings (bathymetry, hydrography geology);
- Selection of sample sites;
- Description of the sediment cores;
- Results on mineralogy;
- Results on isotopes;
- Results on carbon, nitrogen, phosphorus;
- Results on metals;
- Results on organic contaminants.

General comments applicable to all sections of the report

For presentation of maps and figures, authors should take into account the following:

- an introductory map describing all important features such as geographic locations, names of major rivers, Countries, etc., is necessary as background information–it will be unnecessary to repeat this information in subsequent maps;
- names should be checked in all existing maps for completeness and correctness (e.g., Latvia is always missing);
- the report presumably will not be published in colour;
- maps should be presented in a consistent format to improve the overall quality of the presentation;

- shading of map details might mask the scientific results presented in a geographical context and care should be taken to review the clarity of all maps and figures;
- there are difficulties in comparing scaled symbols in 3-D maps due to perspective shortening.

All references should be checked with respect to their completeness and consistency and to ensure that they are included in the overall list of references at the end of the report.

The names of samples, nomenclature of substances, etc. (e.g., ^{210}Pb, arsenic) should be correct and consistent throughout the whole report. Cores should be allocated standard names which are used without modification throughout the text of the report. It may be necessary to use a numbering system on some maps, but if possible this should be avoided.

Comments on specific chapters

Section (Page)	Comments on Specific Chapters of the *Baseline Study of Contaminants in Baltic Sea Sediments*
Chapter 1	▪ In the general introduction to the report, the purpose and specific objectives of the Baseline Study should be given—thus, requiring that the original text be rewritten.
Chapter 2	▪ The map in Figure 2.1 is not sufficient. The reader should be able to locate stations by LON and LAT. A sampling station should be indicated by a symbol and its name.
	▪ Since the hydrography is the ruling factor of the Baltic basins, this chapter should contain a short section on the hydrography of Baltic basins (including the most important references). Comments on the reasons for selecting basins for sampling should also be given.
(page 6)	▪ The map in Figure 2.2 is missing; check whether bottom variability is the correct title (better: bottom topography?).
	▪ Figure 3 is missing.
	▪ Different nomenclature for Pb 210 isotope on pages 7 and page 8.
	▪ Lack of references for chemical analysis methods and quality control.
	▪ Description of method for mineral and grain size analyses is missing.
(page 8)	▪ Title of subchapter 2.5 'Age determination' should be changed to something like 'Estimation of sedimentation rates'.
	▪ Most of the references on this page are missing: for Kizyrov, Niemistö, Vallius *et al.*, 1996; Bouma 1969; Krinitzky 1970, Axelson 1992c. This is a general problem in the report, and WGEAMS did not do any further detailed checking. However this is necessary prior to publication.
	▪ Subchapter 2.6 should be named 'Characterization of the sediment cores'.
(page 9)	▪ The mineral vivianite is not mentioned in mineralogical section.
	▪ Last sentence on this page is contradictory to Section 3.3.2 on 'Mineralogical composition of sediments'.
(page 10)	▪ The section entitled 'Station description' actually contains the core descriptions.
	▪ WGEAMS suggests that this section should contain a description of stations. Detailed descriptions of the cores could be put in an Appendix including good reproductions of all X radiographs (which seem to be missing from the current draft).
(page 21)	▪ WGEAMS advises that descriptions of methods should be moved to an Appendix.
	▪ What does 'KKJ datum' and 'WGS 94' mean?
(page 23)	▪ The Summary should follow scientific standards in both its structure and content.
	▪ The Summary should be a Joint Summary which combines all information on the material.
	▪ It is unclear whether the Recommendations are meant as Conclusions.
	▪ The recommendation of suitable stations is not always conclusive when compared with the graphs on ^{210}Pb.
	▪ WGEAMS feels that the Gotland Deep case is a bit over-emphasized with respect to the outcome of the whole report. Its importance is unclear and, furthermore, should be stressed in a more appropriate place, e.g., the Discussion suggested in the new structure given above.
	▪ Regarding the Recommendations, see also the general comments at the beginning and end of this evaluation.
	▪ The recommendation on how sediment monitoring should be carried out contradicts the conclusion of the chapter on metals in sediment cores.
	▪ The necessity of the extremely exact positioning of the vessel is unclear and questionable. Additionally, there are more appropriate sampling devices available on the market which fit the purpose, i.e., to get good quality surface sediment samples.
	▪ WGEAMS is of the opinion that more important organic pollutants have to be analysed.
Chapter 3 (page 26)	▪ A description of the method used for granulometry is missing. There should also be mention of the comparability with other granulometric methods.

Section (Page)	Comments on Specific Chapters of the *Baseline Study of Contaminants in Baltic Sea Sediments*
(28–29)	▪ Information on minerals unlikely in anoxic sediments such as goethite, gypsum, basanite and anhydrite should be reconsidered. It should be mentioned in this chapter (as is done elsewhere in the report) that these are most probably artefacts due to oxidation during sample storage. It is likely that exposure of reduced sediment to air has resulted in the oxidation of reduced iron forming goethite, and of pyrite forming sulphuric acid. In turn this acid will have reacted with calcium salts (e.g., calcium carbonate) and formed a range of calcium sulphate minerals such as gypsum and basanite. Thus an extensive discussion of these minerals is inappropriate, as is including maps of their geographical distribution. ▪ However, the detection of authigenic minerals typical of anoxic conditions in measurable quantities is a remarkable result which should be highlighted. For example, maps showing the distribution of manganese carbonate minerals could be included. ▪ Figures 1–6 should be readable after reproduction. Removing shading might help to improve the clarity of the figures. It is not clear from our copy whether the original was in colour. The authors should make every attempt to display their data in monochrome, as it may well be that the final publication will not include colour illustrations. ▪ Graphs on page 31 are not helpful for the interpretation, thus could be removed.
Chapter 4	▪ This chapter describes regional distributions of trace metals in the Baltic Sea using bar diagrams. Bar diagrams seem to be too rough and an insufficient presentation for describing the results. There are no scales and no information on concentrations. Other symbols should be used. ▪ The spatial trace metal survey shows certain areas with increased metal concentrations, in particular for As and Hg. The downcore profiles indicate that natural redox processes can have a marked influence on the accumulation of Mn and Cd at certain stations which makes it difficult to distinguish between natural and anthropogenic enrichments. ▪ WGEAMS is of the opinion that subchapters 4.3 ff (from p. 44 to p. 55) are well presented and of special importance for the baseline study.
Chapter 5	▪ This chapter contains the results of analyses of carbon, phosphorus and nitrogen in sediments; therefore, a better title would be 'Carbon, nitrogen and phosphorus'. ▪ Subchapter 5.2, 4th line from the top: 'Primordial bottom-near water data...' What is it? ▪ Names of the Baltic countries should be omitted or completed in Figures 5.1 and 5.4.
(page 59)	▪ Redox potential values seem to be not corrected to the standard electrode potential of hydrogen electrode (see line 20 from the top). ▪ Better use of the Figures could be made. For example, total carbon data alone is not very helpful. However, if both organic carbon and inorganic carbon were displayed as stacked bar charts on one map this would be much more useful. The same could be said for P species.
(page 60)	▪ The Key is missing in Figure 5.2.
(page 67)	▪ Provide subnumbers in Figure 5.9.
Chapter 6	▪ The sum parameters EOCl and EOX have attracted considerable attention in the report, presumably due to the large interest in persistent chlorinated organics during the 1980s, when elementary chlorine was used in pulp bleaching. Although the number of stations visited is too small for a geographical interpretation, the results indicate that EOCl concentrations are distributed in a pattern similar to that of 1986. A remarkable result is not discussed, i.e., that the situation in the sediments has not changed although discharges of EOX from pulp mills have decreased by 90 % since the 1980s. The implications of industrial discharges for sediment monitoring are not discussed in the report. ▪ This chapter describes EOCl, sDDT, sPCB and PAH distribution in sediments. It contains little information with respect to HELCOM guidelines (ICES Marine Chemistry Working Group, IUPAC No. of CBs). Measures are not harmonised with CB measurements within the Baltic Monitoring Programme. Consideration should be given to greater discussion of individual CB congeners, PAHs, etc., to allow comparisons with data from other programmes, e.g., OSPAR programmes, to be made more easily.
Chapter 7	▪ WGEAMS had insufficient expertise available at its meeting to make a full review of this chapter. WGEAMS strongly recommends that this chapter be reviewed by an independent expert.

Final remarks

WGEAMS is of the opinion that the Baseline Study has identified a large number of problems and uncertainties as far as application of sediments for monitoring of pollution. They apply to:

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

After examination of the sediment core data, WGEAMS agreed with the authors of the scientific chapters 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. The conclusions from the scientific chapters are not accurately reflected in the Conclusions section of the current draft (in Chapter 2), and the Recommendations in this draft are not a logical extension of the scientific conclusions. WGEAMS offers the following discussion of the interpretation of the results and of their significance for future monitoring activity.

Scientific interpretation of the overall results

From a review of the draft report, WGEAMS 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. WGEAMS 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 cores.

The Study has clearly shown that sediments are extremely active chemically. Post-depositional changes have resulted in the mobilisation 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 Study. The signal from changes in inputs is completely dominated 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 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 an interpretation of the analyses of sediment cores in terms of temporal (historical) patterns of contaminant inputs to the Baltic 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. As stated in the 1997 WGEAMS report, 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 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 organise 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, .

... WGEAMS 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.' [1997 WGEAMS Report]

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 investigation of 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, lines of stations up the sides of basins, crossing the boundary between the oxic and anoxic waters, and lines in relation to major inputs.

WGEAMS emphasizes again the importance of the scientific results obtained from this 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 monitoring of sediments within the forthcoming COMBINE programme.

WGEAMS also reiterates that it is essential to make all the data available for the scientific public on diskette with the final report, or perhaps via the Internet. The data should also be stored in the ICES Data Centre.

14 CONCLUDE THE PREPARATION OF PROPOSALS FOR A WORKSHOP ON RISK EVALUATION AND ENVIRONMENTAL MONITORING, IN COLLABORATION WITH WGSAEM

The Chairman reported that he had been unable to complete this task during the intersessional period. He recalled that WGEAMS 1997 had felt that they did not possess the necessary expertise in risk evaluation to be able to correctly formulate appropriate 'terms of reference' for a workshop or symposium on the subject and that they therefore required additional support from outside the Working Group. During the intersessional period, an expert in the UK Health and Safety Executive had been identified who was prepared to assist with the project, but it had not yet been possible to arrange a meeting with him.

WGEAMS discussed the topic once again and confirmed that they felt that the subject was important and that some kind of workshop should be organised. After discussing various options (e.g., ASC Theme Session or Mini-Symposium, independent Workshop, etc.), it was agreed to recommend to ACME that a three-day Workshop should be held in conjunction with (and preceding) the 1999 meeting of WGEAMS. The structure of the Workshop should allow experts in risk evaluation to present the bases of their procedures, scientists involved in environmental monitoring to present their perspectives, and an amalgamation of these two fields to take place. The outcome should subsequently be discussed by WGEAMS from an ICES perspective.

WGEAMS noted considerable advantages if some degree of co-sponsorship for the event could be organised, for example, with the Commissions, or with the oil industry.

15 REVIEW OF THE OUTCOME OF SGMPCS (WITH WGMS AND WGSAEM)

The meeting of SGMPCS planned for very early in 1998 was postponed until April or May 1998, hence, a report was not available to WGEAMS for review.

After a brief consideration of this item, the ICES Environment Adviser informed WGEAMS that the SGMPCS will not meet in April or May 1998 as earlier planned. The meeting has been cancelled.

WGEAMs noted that the task allocated to SGMPCS is important and recommends that ACME seek to ensure the completion of this work.

16 ANY OTHER BUSINESS

16.1 New Chairman

The Chairman informed WGEAMS that the ICES Delegates had agreed at the 1997 ASC that Working Group chairmen should serve for a maximum period of three years (or three meetings). As the current chairman has now served for 5 meetings, it is now necessary for WGEAMS to propose a new chairman.

During discussion, WGEAMS noted that the number of members attending WGEAMS meetings has tended to be rather small in recent years and that some previously active members are now less able to contribute and attend, for example, through pressures of other commitments such as new jobs, budget restrictions, etc. The increasing workload posed by the QSR 2000 during the next two years may create additional difficulties. It was also necessary to maintain an appropriate balance between work with particular relevance to specific parts of the ICES area and to have the breadth of experience and expertise required to address the Terms of Reference.

WGEAMS agreed to recommend that Dr Lars Føyn (Norway) be invited to act as chairman starting with the 1999 meeting.

16.2 GOOS

At the 1997 Annual Science Conference, ICES established a Steering Group on the Global Ocean Observing System (SGGOOS) to 'prepare an action plan for how ICES should take an active and leading role in the further development and implementation of GOOS at a North-Atlantic regional level with special emphasis on operational fisheries oceanography'. The Chairman of SGGOOS, Roald Sætre (IMR Bergen), had circulated a discussion paper in January 1998 to Chairmen of all Working Groups under the Oceanography, Marine Habitat and Living Resources Committees asking for their views and requesting that the documents also be discussed at WG meetings.

The ICES Oceanographer, Dr Harry Dooley, kindly agreed to make a short presentation on ICES and GOOS and to lead the subsequent discussion. He described the processes which have led to the formation of SGGOOS and recent discussions regarding the founding of a national NorGOOS for Norway.

WGEAMS noted that GOOS is an extremely broad concept which encompasses almost all activities in the marine and coastal environment. The original five modules (Climate, Health of the Oceans, Living Marine Resources, the Coastal Module, and the Services module) cover all aspects of marine science currently within ICES' expertise and others as well. The concept of linking all such projects into a single unit was attractive and offered considerable benefits over the normal more piecemeal approach. The possible involvement of ICES in GOOS will be greatly influenced by the outcome of an intergovernmental meeting later in 1998. It is anticipated that this meeting will result in the recognition of GOOS by national governments and some degree of commitment of resources to the programme. Should this occur, the status of GOOS will change dramatically. Whereas there is already considerable pressure for ICES to become involved in GOOS as an ocean science programme, if it becomes an intergovernmental venture it is likely that ICES will be required by Member Countries to participate. This should be seen as a positive move which will give ICES a more genuine and potentially significant position within the GOOS community. However, it could raise considerable difficulties for the resolution of the relationships between the range of international bodies involved in marine science, for example, ICES, OSPAR, HELCOM, EEA, EU, etc., in environmental fields. At the moment, the Commissions are customers for ICES advice, data storage, and data presentation. It would be logical for ICES to act in the same role towards GOOS. However, WGEAMS suspects that many of the scientific initiatives in GOOS have been, and will be, started by scientists outside the ICES community. These scientists may not immediately welcome ICES in this role, particularly if these scientists see the data products as having commercial value. Potential barriers to the progress of ICES within GOOS therefore exist in the reconciliation of current international bodies with the GOOS newcomer, conflicts in the area of commercial exploitation of the products generated within GOOS, and the resistance of non-ICES scientists to the arrival (and possible imposition) of the ICES newcomer.

WGEAMS noted the contrast between the broad base of GOOS and the much narrower project areas that are emerging from it. For example, EuroGOOS is heavily directed towards engineering support for coastal activities, shipping, meteorology, etc., and the current US element of GOOS seeks to unify aspects of US coastal zone management. There is considerable interest in the UK in using GOOS as a vehicle and justification for large oceanic monitoring programmes using automatic submarines (AUTOSUB) or smaller scale (but still primarily hydrographic) studies. This narrowing was also apparent in discussion documents circulated by Roald Saetre. The concept of operational fisheries oceanography was based on the combination of ICES expertise in the areas of fisheries science (mainly the analysis of data from fish stock surveys) and associated hydrography and nutrient measurements. This concept did not take account of the very considerable expertise within ICES in subjects falling within the Health of the Oceans module (pollution chemistry, biological effects, etc.) and developing within the Coastal Module (e.g., coastal zone management issues within WGEIM). WGEAMS felt that ICES should reflect on its unique position in having access to a wide range of expertise available in the ICES community and consider how this breadth might be exploited in relation to GOOS.

WGEAMS then considered the four alternative strategies outlined by Roald Saetre, Chairman of the ICES Steering Group on the Global Ocean Observing System, and listed below.

<u>Alternative A</u>

ICES is formally represented in all appropriate GOOS fora such as the new GOOS Steering Committee, I-GOOS, the relevant GOOS Modules Panels as well as in EuroGOOS. All operational activities are organised by the Member Countries themselves and there is no regional GOOS system within the ICES area. This alternative is only slightly above the present involvement level and may be characterised as 'business as usual'.

<u>Alternative B</u>

An official GOOS Pilot Project is established within the ICES area (e.g., Northeast Atlantic Ocean, North Sea, Baltic Sea) by other bodies. In addition to what is mentioned under Alternative A, ICES has a role as an advisory and service agency for the regional GOOS component.

Types of service could include:

- *data bases and data management;*
- *quality assurance—methods, manual, guidelines, intercalibration exercises;*
- *support for the Living Marine Resources Module—in particular concerning phytoplankton, zooplankton, and benthos.*

<u>Alternative C</u>

ICES takes responsibility to establish and run a centre for operational fisheries oceanography on non-meteorological time scales (i.e., more than two weeks) or on the time scale of fish stock assessment (some months) for the whole North Atlantic or parts thereof, i.e., the North Sea. The centre co-ordinates national and international data collections, the rapid transmission of data to computerised data assembly centres for processing through numerical and statistical models to produce regular

- *climatic predictions (time scale: season to some years);*
- *environmental status reports;*
- *time series for identifying trends or changes.*

<u>Alternative D</u>

In addition to the tasks mentioned under Alternative C, we also include processes of meteorological time scales, i.e., ICES establish a Centre for operational fisheries oceanography on times scales from days to years.

WGEAMS considered that *Alternative A* (the status quo, business as usual) is not a viable option. On the assumption that GOOS develops into a significant series of linked international programmes, the potential for GOOS projects to take ground from under ICES is large. ICES cannot afford to be a largely passive observer of GOOS, and as a member of GOOS Panels without direction of component programmes or providing significant support in cash or kind, it would have little claim on the attention of active participants.

Alternative D is unrealistic. The provision of Data Products on a daily time scale would, in the first instance, be seen as a direct trespass on the property of the meteorologists and related scientists. Secondly, WGEAMS feels that the provision of Products without any significant time for assessment would not fit well with the traditional role of ICES as provider of high quality, reliable, considered advice to national governments and international bodies. The strengths of ICES in these areas could be compromised by new tasks which by their nature could tend to dominate the day-to-day activity at ICES HQ.

WGEAMS believes that the role for ICES lies within *Alternatives B* and *C*. *Alternative B* remains rather narrow and represents ICES remaining firmly within its traditional roles. Database management and QA are routine activities for the Commissions. It is not clear what support in the fields of benthos and plankton might be offered. Is it intended to charter or co-ordinate research vessels, plan sampling programmes, supervise analysis of samples or data?

Alternative C is a new departure for ICES which would require intergovernmental support and new funding if the current workload is to be maintained in other areas. Time scales in the order of weeks would allow data to undergo appropriate QA screening within the originating organisations and perhaps also at ICES before they are used to prepare Data Products. In this way, the reputation of ICES would be protected and ICES traditional role enhanced. In times of emergency, such as shipping accidents or catastrophic toxic bloom incidents it could be envisaged that the data turn-around time could be shortened to meet the public demand for data and need for data on which to base (almost) real time environmental management.

WGEAMS therefore concluded that a combination of *Alternatives B* and *C* would be the most appropriate. This is contingent on GOOS receiving the necessary intergovernmental support and the GOOS management allocating the relevant tasks to ICES. ICES can enhance the likelihood of the necessary allocation of tasks by:

a) taking an active role in establishing a local GOOS project;
b) using ICES community expertise to offer a breadth of expertise not available elsewhere;
c) ensuring that the local GOOS project reflects the breadth of the GOOS concept;
d) ensuring that the project does not have dominant commercial aims;
e) ensuring that the dominant aims of the project lie within ICES expertise;
f) encouraging the recognition of GOOS by ICES Member Countries;
g) facilitating and working for the resolution of the new relationships that will be required between the many relevant international bodies once GOOS is recognised by national governments;
h) demonstrating that the ICES system can respond on the time scales necessary to design large programmes, seek funding for such programmes, manage programmes, provide technical advice during programme development and execution. This will require structures that can operate on a much shorter time scale than the annual ICES/Commissions cycle of meetings, and which can operate with appropriate and necessary levels of delegated authority.

16.3 Global International Waters Assessment (GIWA)

WGEAMS was informed about the new Global International Waters Assessment (GIWA) project which has recently been approved by the United Nations Environment Programme (UNEP) Global Environmental Fund (GEF) for funding of US$ 7 million, subject to acquisition of co-financing for the remaining funds (about US$ 6 million), to be completed over a four-year period. The object of GIWA is to develop an assessment of international waters comparable to assessments under the Intergovernmental Panel on Climate Change, the Global Biodiversity Assessment, and the Stratospheric Ozone Assessment. This will serve as a comprehensive, strategic framework for the identification of priorities for remedial and mitigatory actions in international waters. The GIWA project will focus on five water-related issues: (1) freshwater scarcity, (2) pollution, (3) habitat modification and destruction, (4) over-exploitation of fisheries and other living aquatic resources, and (5) global changes. In addition to the conduct of a globally coherent assessment of the ecological status of transboundary waters, the GIWA assessment should review the root societal causes of the problems identified, and ultimately develop scenarios that consider implications of alternative management options for water use.

For GIWA, global waters have been divided into nine regions and 66 sub-regions; many of the sub-regions are Large Marine Ecosystems (LMEs). The North Atlantic region defined by GIWA is larger than the traditional area covered by ICES; it also includes the Gulf of Mexico, the Caribbean Sea, and the Mediterranean Sea including the Black Sea and the Caspian Sea. Seventeen LMEs (including the Baltic Sea) are included in the North Atlantic region. This region also includes the corresponding freshwater catchment areas. The sub-regions will be the basic units of assessment of GIWA.

The work of GIWA will be accomplished through the establishment of a network, consisting of national experts and institutions, regional and global collaborating bodies organized around the geographical units of assessment and grouped into nine major regions. Where possible, existing regional and thematic networks will be used. Overall coordination of the work of the participating individuals and institutions will take place through focal points for each of the sub-regions who will participate in the work of nine Regional Task Teams, which will be supported and assisted by a Core Team of four to six full-time specialists covering both regional and thematic concerns. The Core Team will be hosted by Kalmar University, Sweden, and Sweden is the host country for the GIWA project as a whole.

Informal inquiries have been made to ICES concerning the possibility for ICES to play a role in this project, although the nature and scope of this role has not been made clear. There appears to be a possibility for regional intergovernmental organizations who are willing to serve in a coordination role to receive funds to cover their expenses, but the amount of money available is not clear.

This activity would build on past activities that ICES has had in coordinating assessments of the marine environment, beginning with the first basic assessment of the Baltic marine environment in 1979, continuing with assessments of the Skagerrak/Kattegat area and the Irish Sea, and most recently the assessment of the North Sea environment under the North Sea Task Force. Thus, the GIWA work would build on the work that ICES has coordinated in the environmental area over the past two decades. However, it would also extend this work into the area of socioeconomic considerations, which have recently begun to be covered on the fisheries side of ICES. If ICES is interested in such issues, the GIWA platform could provide an excellent first opportunity for the handling of socioeconomic issues in an environmental context within ICES.

Following the presentation of the GIWA project by the ICES Environment Adviser, WGEAMS discussed how the strengths of ICES could fit into the GIWA structure. WGEAMS felt that the scale of the project and the similarities of some of the objectives of GIWA to the activities of the Commissions, ICES and the EEA, provide a strong basis for ICES involvement. ICES is in a unique position with regard to the combination of fisheries and environmental science on an international scale. ICES could therefore bring important balances to the project, particularly concerning the relationships between 'environment' and 'fisheries'. The ICES system should allow simultaneous complex interactions such as 'pollution as a hazard to the environment and fisheries' and 'fisheries as a hazard to the environment' to be placed in appropriate perspectives. The North Atlantic area is larger than the ICES area, but WGEAMS noted that ICES is already developing interests in wider fields, for example through links with the EEA and with global change through GLOBEC. WGEAMS identified a pattern of expansion of the scale of projects with ever-increasing internationalism, fostered by EU-funding patterns for science and other international activities. Individual scientists working in the ICES family are now much more likely to have scientific interests in areas far outside the ICES area than was the case ten or twenty years ago. WGEAMS considered that ICES as an organisation should acknowledge this pattern and accept the opportunities presented by possible new roles within and outside the traditional ICES area, bearing in mind, however, the North Atlantic basis of its constitution and core funding.

WGEAMS recommends that ICES should strongly consider preparing a proposal to act as the North Atlantic regional centre for GIWA. The proposal should be prepared in collaboration with a freshwater science organisation, preferably on the western side of the Atlantic. The International Joint Commission for the Great Lakes was suggested as a possibility, although a Caribbean organisation might provide additional elements of interest to the proposal. ICES has excellent links with the Commissions and other regional seas bodies throughout the North Atlantic GIWA area (although there may be some relative weaknesses in the Caribbean and Gulf of Mexico sub-regions). The scale of funding for the project (about 300,000 USD per region per year, including in kind contributions) is not huge and it will be necessary to make use of many pre-assessed data and reports rather than to initiate new works or undertake detailed new analyses of raw data. ICES contacts with other relevant organisations and national bodies in Member Countries could be very helpful for gaining access to the appropriate documents.

ICES should ensure that the proposal is fully costed so that ICES HQ is not financially disadvantaged by the new work. The ICES Secretariat workload is heavy and it is unlikely that GIWA could be accommodated within existing staffing levels and that additional staff or consultants would be needed. WGEAMS noted that only approximately half the project had been funded and that in kind contributions were expected to fill in the gaps. WGEAMS suggests, as it is anticipated that much of the work for this programme would be accomplished through the ICES WG system, that the costing of the time of WG members could be included as an in kind contribution. For example, if it was estimated that several WGs would spend one day of their annual meeting on GIWA, the costing of each WG member's time at a reasonable rate (e.g., around 5000 DKK per day) would quickly make up the necessary contribution.

As indicated above, WGEAMS anticipates that several ICES WGs would be involved in the project. This involvement could be in various forms, for example, advisory and review functions in relation to the development of unified data requirements and presentation and structure of reports. The WGs could refer the ICES GIWA office to appropriate national and international documents to fulfill aspects of the requirements of the reports and could, if necessary, prepare short overview documents where suitable material was not already available elsewhere. The WGs could review draft reports as they reached completion. WGEAMS envisaged that these tasks would be given to the WGs through the inclusion of appropriate items in their Terms of Reference.

It was noted that the global coordinating centre for GIWA would be at the University of Kalmar in Sweden.

17 CONSIDERATION AND APPROVAL OF ACTION LIST/RECOMMENDATIONS

WGEAMS noted the Action List (attached as Annex 5) for intersessional work to be carried out prior to the 1999 meeting and agreed the Recommendations which are attached as Annex 6 to this report.

18 PROPOSALS FOR A FURTHER MEETING

WGEAMS recommends that it meet again in March 1999 at ICES Headquarters in association with the proposed Workshop on the Application of Risk Evaluation to Monitoring.

19 CONSIDERATION AND APPROVAL OF THE MEETING REPORT

WGEAMS considered and approved the report of the meeting subject to completion of necessary editorial work to be carried out by the Chairman prior to the submission of the final report to the ICES Secretariat.

20 CLOSURE OF THE MEETING

The Chairman thanked participants for their efforts during both the current meeting and previous meetings that he had chaired. On behalf of WGEAMS, the Chairman thanked the staff of ICES for their hospitality, and closed the meeting at 13.00 hrs on Friday 27 March 1998. The Working Group requested that their appreciation of the contributions made to the work of WGEAMS by Dr Ian M. Davies during his period as Chairman be recorded in the WGEAMS report.

ANNEX 1

AGENDA

1 Opening of the meeting.

2 Adoption of the agenda.

3 Arrangements for the preparation of the report.

4 Reports of activities in other fora of interest to the meeting.

5 Initiate an examination of the general framework of an integrated approach to the assessment of marine ecological quality, including the formulation of some practical suggestions as to the possible implementation of such an approach and the potential tools to carry it out.

6 Update of the review prepared at the WGEAMS 1991 meeting on methods for the identification of priority contaminants, with particular reference to the Esbjerg Declaration regarding contaminants which are toxic, persistent, and liable to bioaccumulate.

7 Review of the components of the monitoring guidelines for PAHs in biota (by MCWG) and sediments (by WGMS), including sampling by SGMPCS, for overall coherence and completeness. OSPAR 1998/1/1

8 Review of the information and data on concentrations of non-*ortho* and mono-*ortho* CBs in marine mammals and relevant biological effects and prepare a report on the findings and implications, as a contribution to the OSPAR QSR (with MCWG, WGMMHA and WGBEC). OSPAR 1998/3.1

9 Review information collated intersessionally on current national and international monitoring strategies which address hazards presented by the discharge of produced water by the offshore oil and gas industries.

10 Review the role of ICES in marine environmental monitoring and assessment in relation to the activities of the European Environment Agency and the European Topic Centre on Marine and Coastal Water.

11 Review information collated intersessionally on procedures to assess the combined effects of exposure of organisms to groups of chemically similar, or dissimilar, contaminants.

12 Prepare a short report on websites providing information of relevance to the work of WGEAMS.

13 Review the final report of the Baseline Study of Contaminants in Baltic Sea Sediments with a view to assessing implications for future monitoring strategies (with WGMS). HELCOM 1998/4

14 Conclude the preparation of proposals for a Workshop on risk evaluation and environmental monitoring, in collaboration with WGSAEM.

15 Review the outcome of SGMPCS (with WGMS and WGSAEM).

16 Any other business.

16.1 Chairmanship of WGEAMS

16.2 GOOS

16.3 GIWA

17 Consideration and approval of recommendations.

18 Proposals for a further meeting.

19 Consideration and approval of the meeting report.

20 Closure of the meeting.

ANNEX 2

LIST OF PARTICIPANTS

Name	Address	Telephone No.	Fax No.	E-mail address
Eugeniusz Andrulewicz	Marine Environment Quality Laboratory Sea Fisheries Institute ul. Kollataja 1 81-372 Gdynia Poland	+48 58 6201728	+48 58 6202831	eugene@mir.gdynia.pl
Ian Davies (Chairman)	FRS Marine Laboratory P O Box 101 Victoria Road Aberdeen AB11 9DB UK	+44 1224 876544	+44 1224 295511	daviesim@marlab.ac.uk
Lars Føyn	Institute of Marine Research PO Box 1870 N-5024 Bergen-Nordnes Norway	+47 55 23 85 00 (direct) 85 01	+47 55 23 85 84	lars@imr.no
Andrew Franklin	CEFAS Burnham-on-Crouch Laboratory Remembrance Avenue Burnham-on-Crouch Essex CMO 8HA United Kingdom	+44 1621 787200	+44 1621 784989	a.franklin@cefas.co.uk
Hartmut Heinrich	Bundesamt f. Seeschiffahrt und Hydrographie Postfach 30 12 20 20305 Hamburg Germany	+49 60 31903513	+49 60 31905000	hartmut.heinrich@bsh.d400.de
Jan René Larsen	ICES Palægade 2–4 1261 Copenhagen K Denmark	+45 33 154225	+45 33 934215	jrl@ices.dk
Evangelos Papathanassiou (Guest/Part-time)	European Environment Agency Kongens Nytorv 6 DK-1050 Copenhagen K Denmark	+45 3336 7155 (direct)	+45 3336 7199	evangelos.papathanassiou@eea.dk
Janet Pawlak	ICES Palægade 2–4 1261 Copenhagen K Denmark	+45 33 154225	+45 33 934215	janet@ices.dk
Olof Sandström	National Board of Fisheries Institute of Coastal Research Gamla Slipvägen 19 740 71 Öregrund Sweden	+46 173 46460	+46 173 30949	olof.sandstrom@kust.fiskeriverket.se

ANNEX 3

LIST OF MEETING DOCUMENTS

WGEAMS98/1/1	1997/1998 Overview of ICES Committees and subsidiary groups
WGEAMS98/1/2	ICES ACME Report 1997
WGEAMS98/1/3	Arctic pollution issues
WGEAMS98/1/4	ICES Annual Report 1996
WGEAMS98/1/5	ICES ACME Report 1994
WGEAMS98/1/6	ICES ACME Report 1996
WGEAMS98/1/7	ICES Annual Report 1996/1997
WGEAMS98/2/1	Draft agenda
WGEAMS98/6/1	SIME contribution to the selection and prioritisation mechanism. SIME 1997, Annex 8.
WGEAMS98/6/2	Progress report on the development of EACs. LC/SG 20/7/2
WGEAMS98/6/3	OSPAR HOD Summary Record 1997.
WGEAMS98/6/4	Draft strategy to implement OSPAR's objective with regard to hazardous substances.
WGEAMS98/6/5	Report of ICES WGEAMS 1991
WGEAMS98/6/6	Esbjerg Declaration
WGEAMS98/6/7	Esbjerg Progress Report
WGEAMS98/7/1	OSPAR DIFF Summary Record 1997
WGEAMS98/7/2	Draft report of ICES WGMS 1998
WGEAMS98/7/3	Extract from ICES MCWG 1998 draft report
WGEAMS98/8/1	McKenzie *et al*, 1998. Chlorinated biphenyls and organochlorine pesticides in marine mammals stranded on the coasts of Scotland and Ireland. Fisheries Research Services Report 2/98
WGEAMS98/8/2	Draft report of the ICES WGMMHA 1998
WGEAMS98/8/3	Draft report of the ICES WGMMPD 1998
WGEAMS98/8/4	Extract from ICES MCWG 1998 draft report
WGEAMS98/8/5	Extracts from preprint of AMAP Arctic Environmental Assessment repoıt
WGEAMS98/8/6	Report of the IWC Workshop on chemical pollution and cetaceans (1995)
WGEAMS98/10/1	Section 5.6 of ACME minutes 1997.
WGEAMS98/10/2	Overheads used in presentation by Evangelos Papathanassiou (EEA)
WGEAMS98/11/1	Philip Hess, 1998. Extract from PhD thesis. Robert Gordon University, Aberdeen and Fisheries Research Services Marine Laboratory, Aberdeen.
WGEAMS98/11/2	Viarengo, A., *et al.* 1997. Heavy metal inhibition of EROD activity in liver microsomes from the bass *Dicentrarchus labrax* exposed to organic xenobiotics: Role of GSH in the reduction of heavy metal effects. Marine Environmental Research, 44: 1–11.
WGEAMS98/11/3	George, S.G. 1989. Cadmium effects on plaice liver xenobiotic and metal detoxification systems: dose-response. Aquatic Toxicology, 15: 303–310.
WGEAMS98/11/4	Croce, B., *et al.* 1997. Ecotoxicological determination of pigmented salmon syndrome. Ambio, 26: 505–510.
WGEAMS98/11/5	Croce, B., and Stagg, R.M. 1997. Exposure of Atlantic salmon parr (*Salmo salar*) to a combination of resin acids and a water soluble fraction of diesel fuel oil: a model to investigate the chemical causes of pigmented salmon syndrome. Environmental Toxicology and Chemistry, 16: 1921–1929.
WGEAMS98/13/1	Final draft report of the Baseline Study of Contaminants in Baltic Sea sediments 1993.
WGEAMS98/13/2	Draft report of ICES WGMS 1998
WGEAMS98/16.2/1	Internal ICES document 'ICES Science Function'
WGEAMS98/16.2/2	Package of papers concerning GOOS, including: a. Letter from Roald Saetre and enclosures b. Account of UK GOOS-related activities c. Note by W R Turrell on ICES involvement in GOOS.
WGEAMS98/16.3/1	GIWA Project Brief. 24 September 1997.
WGEAMS98/16.3/3	ICES CM 1997/A:14. The GEF GIWA Project.
WGEAMS98/16.3/3	Developments under the UNEP relevant to ICES. J Pawlak.

ANNEX 4

ACTION LIST FOR 1997

Action No.	Action required	Person(s) Responsible	Outcome
1	To develop firm proposals for a Workshop on Risk Evaluation intersessionally with a view to defining the scope and terms of reference for the Workshop and presenting these proposals to the next meeting of WGEAMS.	I. Davies	An appropriate expert had been identified in the UK to assist with the proposal, but this had not been completed. Work would continue intersessionally.
2	To write to QUASIMEME concerning the inclusion of sea bird egg analyses in the JAMP Guidelines and the need for Reference Materials and information on the within and between laboratory variances of the determination of organochlorine compounds and mercury in this matrix.	I. Davies	QUASIMEME Scientific Advisory Group had agreed to determine the level of interest in a Laboratory Performance Study for sea bird egg analysis. Dr Becker of Wilhelmshaven had indicated his willingness to assist.
3	To send copies of the section of the WGEAMS report on sea bird eggs, and the Annexed documents relating to variance components to the Chairman of WGSAEM.	I. Davies	Done. Noted that HELCOM will consider including bird eggs in monitoring programmes.
4	To send the final version of section 11 of the WGEAMS report to the Chairman of WGBEC for comment.	I. Davies	Done. Section 11 was subsequently slightly amended and included in the ACME report.
5	To prepare a review of endocrine disruptors, other than environmental oestrogens.	K Stange/C. de Wit	No progress
6	To ask the Chairman of WGBEC whether there are any actions in WGBEC on endocrine disruptors.	I. Davies	WGBEC included oestrogenic compounds in their discussions.
7	To collate information inter-sessionally on methods for the identification of priority contaminants in the contexts, among others, of a) Esbjerg Declaration	I. Davies/K. Stange	Done
	b) Arctic processes	K. Stange	Not available
	c) Long range transport atmospheric processes (LRTAP)	K. Stange	Not available
	d) OSPAR/SIME	M. Joanny	Information obtained from A. Franklin
	e) OSPAR/DIFF	I. Davies/C. de Wit	Done
8	To circulate the report from the WGEAMS meeting in Halifax to participants at WGEAMS97	I. Davies	Available at WG meeting
9	To prepare a discussion document on national and international approaches to monitoring the effects of produced water from oil and gas installations, incorporating information from:	L. Føyn	Completed at WG meeting
	Scotland	I. Davies	Provided
	England/Wales	A. Franklin	Provided
	Norway	L. Foyn	Provided
	E+P Forum	L. Foyn	Provided
	The Netherlands	WG member	No contribution
	USA	WG member	No contribution
	Canada	J. Piuze	Provided
	Denmark	WG member	No contribution
	JAMP/SIME	R. Stagg	Provided
	Approaches to persons named above but not present at WGEAMS97	I. Davies	Done
10	Compile information on procedures available to assess the combined effects of exposure to groups of chemically similar, or dissimilar, contaminants.	All	Some documents provided at WG meeting
11	Approach ICES Secretariat for a document outlining the current ICES links with EEA and ETC/CW	I. Davies	ICES informed that no document exists
12	Compile information on the activities of ETC/MC	M. Joanny	Not provided.
13	Compile information on the marine activities of EEA	I. Davies	Agreed with ICES this would be covered by 14 below.
14	Approach ICES Secretariat with a view to inviting a representative of EEA to the next WGEAMS meeting to present an account of the EEA activities in the marine environment.	I. Davies	EEA (and MEDPOL) invited to WGEAMS98. MEDPOL not able to attend.
15	Compile information on useful Websites.	All	Task not necessary
	Act as a postbox for the above information.	I. Davies	
16	Enquire of ICES Secretariat whether it is anticipated that WGEAMS will be involved in review of the QSR 2000, and if so, when.	I. Davies	Secretariat replied that it was likely that ICES would be involved, but it was not yet clear how this would occur.

ANNEX 5

ACTION LIST FOR 1998

Action No.	Action required	Person(s) responsible
1	To gather information on non-oestrogenic effects of endocrine disruptors.	All
2	To arrange for Lars Føyn to receive extracts of previous WGEAMS reports concerning the use of seaweeds in monitoring.	I. Davies
3	To distribute a circular to WGEAMS members requesting information on why a number of members were unable to attend WGEAMS 1998.	I. Davies
4	To gather information on spatial and temporal mismatches in monitoring programmes.	All
5	To gather information on the influence of Agenda 2000 on national monitoring and assessment programmes.	All
6	To gather information on procedures to assess the combined effects of groups of chemically similar or dissimilar contaminants.	All
7	To gather information on the value of long-time series data in monitoring and assessment.	All
8	To gather information on the temporal relationships between changes in contaminant inputs and environmental responses.	All

ANNEX 6

RECOMMENDATIONS

The following recommendations are numbered according to the relevant agenda item.

WGEAMS recommends that:

4.1 MHC consider explicitly the inclusion of the concept of sustainability in their terms of reference.

4.1 ICES maintains a balance between its role in providing specifically commissioned advice for OSPAR, HELCOM, etc., and the underpinning function of WGs in providing pro-active environmental advice to ICES Committees and Member Countries.

4.2 MHC include analysis of the implications of the outcome of MON98 in the Terms of Reference for WGEAMS and WGSAEM meetings in 1999.

4.3 ICES finalise agreements on its role in the review of the OSPAR QSR 2000 documents as soon as possible and ensure that the WG (or WGs) who may be involved are informed in good time.

6 ACME note to OSPAR that the proposed OSPAR approach to the prioritisation of hazardous substances contains weaknesses in relation to chemicals which exert chronic toxic effects (e.g., non-oestrogenic hormone disruptors, carcinogens, mutagens, teratogens).

7 ACME ensure that the draft Technical Annex on PAHs in biota being prepared by MCWG is given adequate review before it is considered for transmission to OSPAR as a completed task.

9 ACME note to OSPAR that large volumes of produced water and associated contaminants are discharged to the seas of the OSPAR Convention area and that the current OSPAR programmes do not include mechanisms for monitoring the effect of, or for the regulation of, produced water.

10 ICES continue to develop a mutually beneficial relationship with the EEA and that the current round of re-negotiations of subventions for Topic Centres might present an opportunity for ICES to formally establish a role.

10 ICES work with the EEA to try to identify subject areas in which ICES could act as a provider of scientific advice to the EEA.

13 ACME convey the comments of WGEAMS on the Baseline Study of Contaminants in Baltic Sea Sediments to HELCOM.

13 The report on Baseline Study of Contaminants in Baltic Sea Sediments be published in the *ICES Cooperative Research Report* series after revision according to the comments in Section 13 of the 1998 WGEAMS report.

WGEAMS encourages ACME to develop a mechanism to resolve the outstanding scientific questions concerning the report of the Baseline Study of Contaminants in Baltic Sea Sediments and that will handle the editorial and presentational aspects of the report and thereby expedite its completion and publication.

14 A Workshop on Environmental Risk Evaluation and Monitoring (Co-Chairmen: I. Davies and R. Fryer (UK)) should be held for three days in March 1999 at ICES Headquarters, immediately preceding the 1999 WGEAMS meeting, to develop a conceptual framework which encompasses:

a) risk assessment and evaluation prodecures in an international environmental context,

b) the interrelations of environmental risks in relation to such issues as conservation, sustainability, the precautionary principle, and aims declared through such fora as the North Sea Ministerial Conferences and international Commissions (e.g., OSPAR, HELCOM),

c) the objectives and effectiveness of current international marine monitoring activities,

d) the application of risk evaluation procedures to decisions on the prioritisation of monitoring targets and the appropriate scale of activity in international marine monitoring programmes.

Co-sponsorship should be sought from OSPAR, HELCOM, and AMAP.

15 ACME seek to ensure the completion of the work allocated to SGMPCS.

16.2 ICES should actively participate in GOOS. The ICES role should extend beyond its traditional role as databank and QA adviser and should encompass those matters included in Alternative C in the January 1998 circular from the Chairman of ICES SGGOOS. ICES should seek to take advantage of the very broad expertise available within the ICES community, taking into account the more detailed advice available in the main text of the 1998 WGEAMS report.

16.3 ICES develop a proposal to act as North Atlantic Regional Centre under GIWA, in association with an appropriate (preferably international) organisation with expertise in freshwater science and resources.

18 WGEAMS recommends that it meet for a period of five days in March 1999 at ICES Headquarters, Copenhagen, under the Chairmanship of L. Føyn (Norway) immediately following the Workshop on Environmental Risk Evaluation and Monitoring, to undertake, *inter alia*, the following tasks:

a) to continue to review information collated intersessionally on procedures to assess the combined effects of exposure of organisms to groups of chemically similar, or dissimilar, contaminants;

***Justification**: This is a continuation of an item from the 1998 agenda. Most national and international pollution control measures are based on the regulation of individual compounds, or groups of compounds. It has long been recognised that this may reflect a fragmented view of the mechanism of impact of marine pollutants, whereas in reality organisms are usually exposed to complex mixtures of similar and dissimilar substances. The purpose of this agenda item is to revisit the problems of synergism and antagonism between contaminants, and the possibilities of interactions between apparently unrelated substances, and hopefully to assess whether current monitoring and assessment procedures can take account of these processes.*

b) to review the outcome of the Workshop on Environmental Risk Evaluation and Monitoring and prepare recommendations;

***Justification:** This is to ensure that the results of the Workshop are reviewed and assessed from an ICES perspective, with particular regard to the application of risk evaluation procedures and concepts in the development of monitoring and assessment strategies.*

c) to review the developments in the relationships between ICES and the European Environment Agency and European Topic Centre on Marine and Coastal water in marine environmental monitoring and assessment and to invite representatives of EEA and UNEP to attend the meeting;

***Justification**: The growth of the European Environment Agency and its associated bodies is a relatively new phenomenon in marine environmental science. It is necessary that ICES develop mutually beneficial working relations with the EEA. WGEAMS anticipates receiving information from the ICES Secretariat and from the EEA to assist with this item.*

d) to consider how far the strategies adopted in the HELCOM COMBINE and OSPAR JAMP allow integrated environmental assessments to be made;

***Justification**: ACME has discussed the need to examine the current state and possible future of methods for integrating multiple measurements of different environmental factors. Basic data for such integrated assessments are produced by monitoring. A review of the recently revised HELCOM and OSPAR* *monitoring programmes is suggested, in order to examine the extent to which the strategies and designs in these programmes can assist in integrated assessments.*

e) to review the importance of long time series data for the interpretation of monitoring data and the preparation of assessments;

Justification: *Long time series of environmental data have revealed the importance of fundamental patterns of change in the environment, often on rather large scales. Monitoring programmes are undertaken against this background and yet it is not clear to what extent such factors are taken into account in developing monitoring strategies or assessing monitoring data.*

f) to review the report from the 1998 meeting of the OSPAR Ad Hoc Working Group on Monitoring (MON98) and prepare comments on the implications of the MON98 exercise for temporal trend monitoring (with WGSAEM);

Justification: *Members of MON98 were surprised and concerned at the high proportion of the temporal trend data series which failed to reveal statistically significant trends. There are a number of possible reasons for this, for example, inadequate quality control of sampling or analysis, high natural variability which could not be accommodated within the programme design, or overly demanding statistical procedures. Effective allocation of environmental science resources requires that the underlying reasons are understood and that the possible need for changes to monitoring programmes are discussed.*

g) to report on new opportunities for the application of microbiological measurements in monitoring programmes;

Justification: *Microbiological measurements are included on a voluntary basis in COMBINE, but are not included in JAMP. New developments in molecular techniques may open significant new opportunities for investigating the effects of contaminants in the marine environment on important marine microbiological processes.*

h) to prepare a report on the relationships between changes in contaminant input functions and consequential environmental responses, taking into account, *inter alia*, the outcome of Theme Session V at the 1998 ASC.

Justification: *It is generally accepted that there is normally a time lag between changes in contaminant inputs and consequential environmental responses. However, the length of the time lag and the form of the response is often difficult to predict. The objective of this item is to collate examples in which both input and response functions have been recorded and to develop a conceptual framework within which to view environmental responses to changes of stress of this type.*

APPENDIX 1

CONCENTRATIONS AND EFFECTS OF NON-*ORTHO* AND MONO-*ORTHO* CBs IN MARINE MAMMALS

The information in Appendix 1 was prepared by WGEAMS from text provided by the Working Group on Marine Mammal Habitats (WGMMHA). It does not represent a completed report, but is a contribution to the process of preparing such a paper. Portions of the text included here were sent to the Chairman of WGBEC on 27 March 1998 for consideration and elaboration by WGBEC. The main changes to the original WGMMHA document proposed by WGEAMS are described in Section 8 of the WGEAMS report and may be summarised as follows:

1) proposals for the elimination of text not directly concerned with planar CBs;

2) incorporation of new material from a recent publication on CBs in marine mammals (C. McKenzie *et al.*, Aberdeen, UK);

3) noting a number of other areas where further amendments are probably necessary.

As indicated above, the text was not finalised by WGEAMS, but contains proposals for deletions, additions, and amendments.

Accompanying the text and the raw data (which are not included in this report) sent to WGBEC were explanatory notes to the WGBEC Chairman from the Chairman of WGEAMS and the ICES Environmental Data Scientist. These notes are reprinted in Appendix 1.

Peter Matthiessen, ICES WGBEC

Dear Peter

Planar CBs in marine mammals

The situation on this task at the beginning of the WGEAMS meeting was that WGMMH had prepared some quite good text covering general aspects of the effects of contaminants on marine mammals, concentrating on planar CBs, and also covering the effects on marine mammals. However, MCWG had not provided any text or tabulations of data, but had put into place various mechanisms to gather the information. These mechanisms seemed unlikely to come to fruition beofre your meeting.

In an attempt to progress the overall task, WGEAMS has done the following:

1. We edited the WGMMH text and marked passages for removal (mainly because they were not closely directed at planar Cbs).

2. I got Craig McKenzie in Aberdeen to review the text. He made a number of very helpful comments concerning aspects that needed to be brought more up to date, bits that were incomplete etc.

3. I took a recent paper that Craig had written with Dave Wells and cut and pasted sections of it into the WGMMH report to provide sections which described the concentrations of planar Cbs in marine mammals, discussed Toxic Equivalents, and discussed trans-generational transfer of Cbs. Some associated Figures will come by fax.

4. Several labs had sent files of data to ICES as a result of the WGMC mechanisms. These have been edited down to reasonable size, and encompass data on planars plus data on CB153 (so that you could, if you wish, estimate total TEQs for the samples).

5. There are data in the ICES databank. I think this is included in the email that will be sent to you by ICES, but thre is a covering letter regarding the lack of prmission to use some of the data.

I aopolgise for the scruffy nature of the text etc that I am sending you. I felt it neessary to retain the original WGMMH text, at least at this stage. I suggest that if you are happy with the way we have edited things that you could chop it out and work towards a final document. There are a number of points raised by Craig which remain in the text as notes. You may be able to answer some or all of them. Please look towards the top of the front page, just below the title, where ther should be text explaining the significance of italicised text, bold text, text underlined, etc. With a bit of practice it should make sense!

I hope your meetings are going well.

Ian M Davies 27 3 98

4 CONCENTRATIONS OF NON-*ORTHO* AND MONO-*ORTHO* CBs IN MARINE MAMMALS AND RELEVANT BIOLOGICAL EFFECTS

Text in italics should be deleted. Underline means remove, **[bold]** indicates insertion, **bold** indicates a note [Editor's note: ~~strikethrough~~ indicates changes in the WGMMHA text after it was delivered to WGEAMS.]

Both pinnipeds and cetaceans have a long life span, are at or near the highest trophic levels in oceanic food webs, and probably summarise the average pollution load in the waters they inhabit, being less susceptible to primary sources of contamination than other components of the marine biota such as mussels, crabs and benthic fishes (Aguilar, 1984). Pollutant levels have been found to be positively correlated with increasing trophic level and ***[appear to be]*** *inversely correlated with* ***[species]*** *body size* ***[in a number of cases]*** *(Borrell, 1993).*

When environmental contaminants are discussed in relation to marine mammals, they may conveniently be classified into four major categories (Reijnders, 1994):

1) halogenated hydrocarbons (including polychlorinated biphenyls (PCBs), pesticides such as DDT, dieldrin, chlordanes (CHLs), toxaphenes, lindane (hexachlorocyclohexone (HCH) and hexachlorobenzene (HCB), etc.);
2) aromatic and non-aromatic hydrocarbons;
3) trace elements: metals, metalloids, bromide and iodide;
4) radioactive isotopes.

No population-level effects of radioactive isotopes on marine mammals have been described (Reijnders, 1994). Polycyclic aromatic hydrocarbons (PAHs) apparently do not accumulate in marine mammals (Hellou et al., 1990; Law and Whinnett, 1992; Reijnders, 1994). Although immediate effects of oil spills on marine mammals have occurred (Frost and Lowry, 1993; Geraci and St. Aubin, 1990; Hall et al., 1996), no adverse effects of aromatic and non-aromatic hydrocarbons on populations of marine mammals have so far been recorded with certainty (Geraci, 1990). For the purpose of this particular report, WGMMHA focused on the effects of halogenated hydrocarbons and trace elements. Emphasis was given to the non-ortho and mono-ortho chlorinated biphenyls [OSPAR 1998/3.1].

Global contamination of the marine environment by persistent, bioaccumulative 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 tissue is dependent on the sex, age, reproductive history, diet and habitat of individuals and species (Subramanian, 1987a; McKenzie *et al.*, 1997). 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 diet, concentrations of contaminants in dietary species, variations of diet and its contaminant burden in space and time, age, etc.) and loss mechanisms (e.g., condition of the animal, reproductive status, excretion rate, metabolic abilities and activity). 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 metabolise 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 (e.g., differences between old mature males and newly born young). The patterns of CB congeners found in marine mammals 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 localised 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.

4.1. Contaminants likely to induce effects on marine mammals

4.1.1. Halogenated hydrocarbons

The organochlorine compounds often measured are DDTs, ***[dieldrin]****, PCBs, HCHs, CHLs and HCB (Tanabe et al., 1997).* ***[The relatively more]*** *water soluble and biodegradable chemicals such as* ***[the]*** *HCHs ~~and HCB~~ are less accumulative in the* ***[marine]*** *ecosystem, whereas more lipophilic and less biodegradable contaminants like DDTs and PCBs are* ***[bioaccumulated by marine mammals to a greater extent]*** *retained in the animal bodies for a longer time (Tanabe and Tatsukawa 1991). ~~DDTs are retained in coastal and open ocean water bodies of point source areas while HCH is prone to disperse on a global scale.~~* ***[Location differences can be observed in marine mammal tissues due to the useage of different HCH mixtures, e.g., technical lindane comprises > 90 % γ-HCH and technical HCH contains predominantly the α-HCH]*** *(refs).*

DDE, an accumulative DDT derivative, is abundant in the environment because the chief metabolic pathway of DDTs ends in different forms of DDE (Aguilar 1984). ***[High proportions of DDD may be found when marine mammals are inhabiting or feed in areas where waste containing DDT has undergone anaerobic treatment prior to release or has undergone degradation in organic-rich sediments (Tanabe, 1997)].*** *Air transport is probably a major means of DDE movement, and will account for a greater proportion of DDE in the marine environment (Aguilar, 1984). Degradative dehydrochlorination of DDT takes place both in living organisms (Walker, 1975) and in non-living parts of environmental systems (Crosby, 1969). Consequently, the abundance of DDE increases progressively throughout time and through increasing steps in the trophic web. Aguilar (1984) proposed that DDE/DDTs ratio of 0.60 in marine mammals as a critical limit to DDT exposure* ***(Note - this does not mean anything uless it is explained!).***

Contaminant compounds are transferred from mothers to pups. **[For example,]** 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 grey seals (*Halichoerus grypus*) were about 30 % and 15 % of their total DDT and PCB burdens, respectively (Addison and Brodie, 1977).

A study on PCB residues in a marine food chain including i.a. copepods, squids and marine mammals in the western north Pacific revealed a bioconcentration factor of 10^7 in striped dolphins (Stenella coeruleoalba) (Tanabe et al., 1994). Wagemann and Muir (1984) suggested that PCB levels more than 50–200 $\mu g\ g^{-1}$ are harmful to the health of marine mammals. **(Note - This statement must be qualified, at ther moment it is completely abstract)**

Of the possible 209 polychlorinated biphenyl (PCB) congeners which exist, only a few have been demonstrated as causing **[a selected number of]** toxic effects. McFarland and Clark (1989) suggest 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).

[Major criticism is that when talking about 'toxicity' it only appears to 'dioxin-like' toxicity that is being discussed - there are 'non-dioxin like' CBs which show different toxic effects. Any estimation of CB toxicity based on 'dioxin-like' CBs only is highly likely to be an underestimate of the total potential toxicological impact]

It is this subset of congeners which can form a planar or coplanar **(NOTE ON NOMENCLATURE - 'The molecule is planar if the rings have the possibility to be in one plane. To be coplanar there would need to be a parallel ring system (Voogt *et al.*, 1990)** (flat) configuration that is 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* position on the rings (Figure 1). This coplanarity and the laterality of chlorine atoms are important structural features of PCBs which determine the specific binding behaviour with proteins and certain toxic responses in biological systems. From a structural point of view, there are two distinct classes of PCBs: the non-*ortho* **[CBs]**, 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).

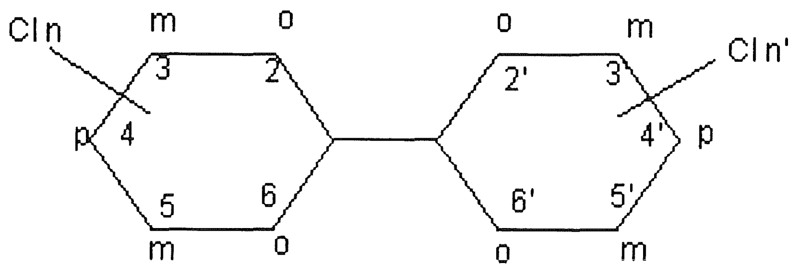


Figure 1. The generalized structure of polychlorinated biphenyls (Hall, 1998).

Once absorbed by an animal, a substance will enter the circulation and be distributed throughout the body. However, compounds are often segregated into non-sensitive tissues which become storage sites (in the case of PCBs in marine mammals, this is generally blubber, e.g., Addison and Brodie, 1987; Pomeroy *et al.,* 1996; Aguilar, 1987). Exchange between these storage sites and the bloodstream means that contaminants with access to actively metabolising tissues (particularly the liver, which is the main organ for the metabolism of xenobiotics) will be subjected to enzymatic attack and 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 metabolise 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 phase I enzymes responsible for this in mammalian species are those of the cytochrome P450 group. This biotransformation system (also known as the mono-oxygenase or mixed function oxidase, MFO), originally identified as the drug metabolising system, has evolved as an important biomarker response 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 **[in simplified terms - 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 distinct cytochrome P450 genes have been identified in various species, now classified into 8 major families (CYP1 to CYP8; Nerbert *et al.,* 1987; Nelson *et al.,* 1993). Subfamilies are identified by a capital letter. 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 dioxin or 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 coplanarity. 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 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]

4.1.1.1 Non-*ortho* congeners **[(CBs 77, 81, 126, 169)]**

One important distinguishing feature of the non-*ortho* group 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 PCBs 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, CB 77 (3,3′,4,4′-tetra), CB 81 (3,4,4′,5-tetra), CB 126 (3,3′,4,4′,5-penta) and CB 169 (3,3′,4,4′,5,5′-hexa) are approximate isostereomers of 2,3,7,8-tetrachloro-dibenzo-*p*-dioxin (TCDD) in their coplanar conformations. 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).

4.1.1.2 Mono-*ortho* congeners **[(CBs 105, 114, 118, 123, 156, 157, 167, 189)]**

A second set of potentially toxic PCB congeners, namely the mono-*ortho* coplanar analogues, are listed in Table 1. They **[generally]** occur in much 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 may **[have a tendancy to]** be more toxicologically important to these species than those which may elicit a greater response but which occur at much lower concentrations.

Table 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

[4.1.1.2a Di-*ortho* congeners [(CBs 170, 180)]

Two di-*ortho* substituted congeners, CBs 170 and 180, have also been shown to induce P450 1A enzymes and due to their high concentrations in marine mammals are thought to be of toxicological significance (Ahlborg et al., 1994).]

A classification of PCB 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 has been suggested by Boon *et al.* (1994) (Table 2). This type of classification may be useful in determining common structure-activity relationships and potential biological effects on marine mammals.

Table 2. Structural groups of PCB congeners.

CB Group	Substitution Pattern Position of vicinal H atoms	No. of *ortho*-chlorine atoms
Group I		
CB–180	-	2
CB–183	-	3
CB–187	-	3
CB–194	-	2
Group II		
CB-99	*o, m*	2
CB–128	*o, m* (2*)	2
CB–138/–163	*o, m* (both)	2 (both)
CB-170		
Group III		
CB-70	*o, m + m, p*	1
CB–105	*o, m*	1
CB–118	*o, m*	1
Group IV		
CB-44	*m, p* (2*)	2
CB-49	*m, p*	2
CB-52	*m, p* (2*)	2
CB–101	*m, p*	2

Table 2. Continued.

CB Group	Substitution Pattern Position of vicinal H atoms	No. of *ortho*-chlorine atoms
Group V		
CB–136	*m, p* (2*)	4
CB–149	*m, p*	3

Group I	=	congeners without any vicinal H atoms
Group II	=	congeners with vicinal H atoms in *ortho* and *meta* positions and > 2 *ortho*-Cl atoms
Group III	=	congeners with vicinal H atoms in *ortho* and *meta* positions and one *ortho*-Cl atom
Group IV	=	congeners with vicinal H atoms in *meta* and *para* positions and < 2 *ortho*-Cl atoms
Group V	=	congeners with vicinal H atoms in *meta* and *para* positions and > 3 *ortho*-Cl atoms
2*	=	2 pairs of vicinal hydrogen atoms in *ortho* and *meta* or *meta* and *para* positions

Concentrations of non-*ortho* and mono-*ortho* CBs in marine mammals

Data on the concentrations of non-*ortho* and mono-*ortho* CBs in marine mammalshave 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 blumber samples for CBs from twelve species of marine mammals, and 88 of these samples were analysed for planar CBs.

Chlorobiphenyls

The concentrations of CBs in marine mammals stranded on the Scottish coast range from 0.27–67.9 µg g^{-1}, 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 (CBs 138, 153, 180, 170) 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 the majority of animals from 11 of the 12 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 metabolise CBs in different species was found to be of the order pinnipeds > harbour porpoise > others > sperm whales (Wells *et al.*, 1996).

'Toxic' Congeners

CB congeners with one or no *ortho*-chlorines and which do not have vicinal hydrogen atoms in the *meta*- and *para*-positions (CBs 105, 118, 156 and 77, 126, 169, respectively) show a similar mode of toxicity to that of 2,3,7,8-tetrachloro dibenzo-*p*-dioxin (TCDD) but have lower potency. In addition to these CBs, two di-*ortho* chlorinated congeners (CBs 170 and 180) have also been shown to produce TCDD-like effects, including EROD 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 present at considerably lower concentrations (non-*ortho* CBs contributed < 0.5 % of total CB concentrations in all species analysed in this study). A summary of the non-*ortho* CB concentrations determined in the blubber of 88 marine mammal blubber samples is given in Table 3. 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 the in 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 and 28 ng g^{-1} 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

Table 3. Summary oganochlorine concentrations, median and (range) for marine mammals stranded on the Scottish coast (μg g^{-1} wet weight).

Species	Sex	N	ΣCB	HCB	Dieldrin	ΣDDT	ΣChlordane	ΣHCH	ΣHCH	CB 77(ng/g)	CB 126(ng/g)	CB 169(ng/g)
Harbour porpoise	M	7	3.03 (0.58-9.15)	0.15 (0.06-0.28)	0.24 (0.13-1.98)	1.96 (0.44-5.69)	0.23 (0.11-.224)	0.01 (0.004-0.02)	0.03 (0.004-0.04)	0.80 (0.22-1.22)	0.10 (0.06-0.14)	0.07 (0.01-0.13)
Harbour porpoise	F	15	2.42 (0.40-11.3)	0.21 (0.02-0.77)	0.61 (0.04-2.37)	1.58 (0.19-7.34)	0.49 (0.04-2.01)	0.01 (0.001-0.04)	0.005 (0.003-0.12)	0.19 (0.02-6.49)	0.05 (0.01-0.07)	0.08 (0.01-3.33)
Bottlenose dolphin	M	7	8.63 (0.48-30.9)	0.22 (0.03-0.35)	0.90 (0.10-2.68)	5.37 (0.40-12.5)	1.49 (0.14-5.25)	0.01 (0.01-0.02)	0.03 (0.01-0.04)	0.18 (0.10-0.36)	0.32 (0.07-0.91)	0.12 (0.08-0.67)
Bottlenose dolphin	F	6	4.12 (0.73-26.3)	0.14 (0.04-0.15)	0.12 (0.01-0.31)	3.31 (0.54-11.5)	0.78 (0.14-1.36)	0.007 (0.005-0.009)	0.01 (0.01-0.06)	0.18 (0.11-0.30)	0.29 (0.07-0.51)	0.20 (0.11-0.61)
White beaked dolphin	M	6	8.81 (4.19-27.4)	0.33 (0.05-0.50)	2.87 (1.60-4.61)	7.95 (4.52-11.0)	2.49 (1.94-7.28)	0.03 (0.01-0.06)	0.06 (0.04-0.13)	0.47 (0.02-1.81)	0.07 (0.03-0.58)	0.82 (0.01-2.43)
White beaked dolphin	F	4	9.00 (7.31-54.4)	0.32 (0.22-1.10)	2.00 (1.44-7.58)	7.17 (6.02-31.9)	2.71 (2.52-9.04)	0.04 (0.03-0.09)	0.06 (0.04-0.28)	0.48 (0.03-1.27)	0.17 (0.01-1.43)	0.49 (0.03-1.60)
White sided dolphin	M	12	25.3 (3.57-67.9)	0.23 (0.11-1.11)	2.44 (0.95-4.74)	13.8 (3.41-53.4)	3.61 (1.16-12.4)	0.03 (0.01-0.13)	0.06 (0.03-0.16)	0.44 (0.08-2.60)	0.09 (0.04-5.99)	0.14 (0.04-5.99)
White sided dolphin	F	11	2.89 (0.85-15.7)	0.18 (0.01-0.40)	0.69 (0.04-3.15)	1.92 (0.20-6.34)	0.76 (0.07-3.71)	0.02 (0.01-0.08)	0.04 (0.01-0.19)	0.12 (0.02-0.31)	0.04 (0.01-0.19)	0.10 (0.01-0.26)
Striped dolphin	M	6	6.16 (5.63-20.7)	0.21 (0.09-0.29)	0.68 (0.37-1.48)	6.31 (4.12-14.2)	1.75 (0.75-4.49)	0.02 (0.01-0.05)	0.02 (0.007-0.03)	0.28 (0.04-3.30)	0.03 (0.01-0.12)	0.79 (0.33-5.99)
Striped dolphin	F	1	2.24	0.15	0.28	2.49	0.69	0.02	0.02	0.37	0.21	0.34
Risso's dolphin	M	1	4.74	0.11	0.66	2.07	1.04	0.006	0.008	0.61	0.21	0.74
Risso's dolphin	F	2	(0.31-0.66)	(0.01-0.03)	(0.05-0.11)	(0.10-0.30)	(0.04-0.13)	(<0.002-0.005)	(<0.002-0.005)	(0.32-0.97)	(0.18-0.29)	(0.10-0.26)
Common dolphin	M	1	4.52	O.23	0.60	4.12	0.99	0.05	0.04	0.08	0.17	0.08
Common dolphin	F	2	(1.06-3.02)	(0.05-0.10)	(0.11-0.49)	(0.49-2.74)	(0.14-1.03)	(0.03-0.11)	(0.015-0.016)	0.08	0.05	0.12

Table 3. Continued.

Species	Sex	N	ΣCB	HCB	Dieldrin	ΣDDT	ΣChlordane	ΣHCH	ΣHCH	CB 77(ng/g)	CB 126(ng/g)	CB 169(ng/g)
Long-finned pilot whale	M	4	8.53 (6.16-10.3)	0.210 (0.16-0.26)	0.42 (0.37-0.12)	7.89 (7.09-14.1)	1.84 (1.45-4.51)	0.02 (0.02-0.06)	0.01 (0.01-0.04)	0.49 (0.28-0.49)	0.23 (0.22-0.31)	0.85 (0.61-1.04)
Killer whale	M	1	11.0	0.22	0.40	8.48	1.28	0.01	0.009	0.68	0.06	0.14
Killer whale	F	1	22.5	1.46	3.86	17.8	10.8	0.11	0.06	0.71	0.02	0.93
Minke whale	M	1	0.27	0.03	0.11	0.27	0.11	0.006	0.005	0.06	0.07	0.03
Sowerby's beaked whale	M	1	12.1	0.18	0.67	7.83	3.92	0.08	0.02	na	na	na
Sowerby's beaked whale	F	5	3.20 (3.10-3.33)	0.90 (0.07-0.11)	0.08 (0.02-0.09)	2.14 (1.89-2.82)	0.29 (0.12-0.54)	0.01 (0.003-0.01)	0.006 (0.002-0.01)	0.12 (0.09-0.21)	0.41 (0.18-0.45)	0.15 (0.12-0.23)
Grey seal	M	14	2.52 (1.65-10.8)	0.01 (0.002-0.02)	0.02 (0.006-0.17)	1.17 (0.82-3.60)	0.16 (0.01-0.96)	0.005 (0.002-0.02)	0.003 (0.001-0.01)	0.18 (0.07-0.32)	0.10 (0.08-0.14)	0.08 (0.08-0.08)
Grey seal	F	6	2.63 (1.46-8.87)	0.02 (0.01-0.06)	0.01 (0.01-0.12)	1.15 (0.79-2.62)	0.27 (0.03-0.73)	0.009 (0.002-0.02)	0.006 (0.001-0.01)	0.07 (0.03 0.07)	0.08 (0.06-0.08)	0.38 (0.01-0.86)
Common seal	M	2	(2.14-2.56)	(0.007-0.02)	(0.04-0.08)	(0.75-1.31)	(0.18-0.27)	(0.01-0.01)	(0.007-0.008)	0.06	0.05	0.03
Common seal	F	7	1.57 (0.37-4.26)	0.008 (0.002-0.01)	0.04 (0.02-0.10)	0.50 (0.23-1.60)	0.10 (0.08-0.32)	0.009 (0.006-0.02)	0.004 (0.003-0.004)	0.17 (0.04-1.10)	0.05 (0.01-0.40)	0.02 (0.01-0.06)

from the Mediterranean Sea ranged from 16–85 ng g^{-1} wet weight (Kannan *et al.*, 1993). The concentrations of CBs 77 and 169 measured in this study are, however, generally higher than those reported in the blubber of Hectors 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 Hectors dolphin samples to those determined in this study (Jones *et al.*, 1997).

Data on ΣCB and ΣDDT concentrations in cetacean and pinniped species sampled worldwide are given in Table 4.

4.1.1.3 Toxic Equivalence Factors (TEQs)

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.

Tables 5 and 6 (from Safe 1990) summarise the toxicity data used to determine the TEFs for the non and mono-*ortho* PCB congeners. **[This section needs to be updated, the Safe data was included in the assessment of toxicity data from which the WHO TEFs were derived - see Ahlborg et al. 1994.]** 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* PCBs appear to be the most toxic congeners, based on structure-activity and structure-binding relationships where PCB congeners exhibit Ah receptor agonist activity (Hall, 1998).

It is apparent from these data that there are considerable variations within this group of compounds (and that very little data are available for 3,4,4′,5-tetra CB 81) and that they depend on both the species and the response. The data show that 3,3′,4,4′,5-penta CB (CB 126) is clearly the most toxic coplanar PCB congener and the 2,3,7,8-TCDD/CB 126 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). However, both CB 77 and CB 169 congeners were considerably less toxic than CB 126 and their relative potencies were highly variable. It is apparent from *in vivo* studies in the rat that CB 77 is over 30 times less toxic than CB 169; 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 CB 77 in the rat may be due to rapid metabolism, but this congener has been detected in samples from marine mammals (Green *et al.*, 1996a).

Although the mono-*ortho* coplanar PCBs 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. CB 118 is the major mono-*ortho* coplanar CB and is routinely identified in marine mammal samples. Some of the higher chlorinated CB isomers are more toxic than CB 118 and the differences in potency may be related to different rates of *in vivo* metabolism.

The detection of toxic coplanar PCBs in cetaceans and seals has led to the assessment of the toxic potential of PCBs using the TEQ approach (Tanabe *et al.*, 1997; Kuehl *et al.*, 1991, 1994; Kannan *et al.*, 1988; [McKenzie *et al.*, 1998]). Green *et al.* (1996a) determined the concentrations of non-*ortho* CBs 77, 126 and 169 in grey seal milk. The relative concentrations were dominated by CB 126, followed by CB 169. They suggested that although there is some concern about the use of TEFs to estimate the potential toxicity of PCBs on the reproductive system of mammals (Battershill, 1994), they appear to be more reliable as an index of their potential effects on the immune system because PCB- and

Table 4. Comparative studies of organochlorine compounds in marine mammals.

Species	ΣCB ($\mu g\ g^{-1}$)	ΣDDT ($\mu g\ g^{-1}$)	Location	Sampling date	Reference
Harbour porpoise	3.0 (0.6–9.2)[M]	2.0 (0.44–5.7)[M]	Scotland	1992	This study
	2.4 (0.41–11.3)[F]	1.6 (0.19–7.3)[F]	Scotland	1992	This study
		41.6	Scotland	1966	FRS data
	23.1 (16.2–58.1)[A]	15.4 (5.9–20.4)	Scotland	1972	FRS data
	29.7 (11.3–47.2)[MA]	13.3 (6.5–25.3)[M]	Scotland	1974	FRS data
	21.7 (10.6–31.4)[FA]	8.4 (4.3–11.4)[F]	Scotland	1974	FRS data
	31.3 (± 7.9)		Baltic	1989–1990	Falandysz *et al.* (1994)
	55.6 (± 22.7)		Wales	1988	Morris *et al.* (1989)
	13.1 (0.47–36.2)[MA]	4.0 (0.64–10.4)[M]	Scotland	1988–1991	Wells *et al.* (1994)
	4.6 (1.9–22.4)[FA]	1.8 (0.64–7.3)[F]	Scotland	1988–1991	Wells *et al.* (1994)
	5.1 (0.12–13.0)	2.2 (0.04–5.1)[O]	Scotland (Shetland)	1989–1991	Kuiken *et al.* (1994)
	23.3 (3.7–65.3)[*]	16.4 (3.2–45.1)[*]	Scandinavia	1987–1991	Kleivane *et al.* (1995)
	13.4 (± 2.4)[*M]	5.6 (± 0.73)[*M]	Faroe Islands	1987–1988	Borrell (1993)
	8.8 (± 1.1)[F]	3.8 (± 0.38)[*F]	Faroe Islands	1988	Borrell (1993)
	22.7 (13–33)	9.1 (8.2–12)	NW Atlantic	1991	Stein *et al.* (1992)
	5.2 (1.8–10.6)[M]	4.1 (1.4–7.3)[M]	Newfoundland	1989–1991	Westgate *et al.* (1997)
	5.5 (1.4–14.2)[F]	3.1 (1.0–7.6)[F]	Newfoundland	1989–1991	Westgate *et al.* (1997)
	8.4 (4.4–16.0)	8.2 (3.1–22.0)	NW USA	1986–1989	Jarman *et al.* (1996)
	10.0 (3.3–31.0)		California	1986–1989	Jarman *et al.* (1996)
	16.0 (5.0–39.0)[M]	70.0 (25.0–180)[M]	Black Sea	1993	Tanabe *et al.* (1997)
	12.0 (1.6–29.0)[F]	50.0 (8.3–83.0)[F]	Black Sea	1993	Tanabe *et al.* (1997)
	6.5 (3.1–16)[M]	4.7 (1.9–12.0)[M]	Japan	1993	Tanabe *et al.* (1997)

Table 4. Continued.

Species	ΣCB (µg g⁻¹)	ΣDDT (µg g⁻¹)	Location	Sampling date	Reference
Bottlenose dolphin	8.6 (0.48–31)^{M}	5.4 (0.40–12.4)^{M}	Scotland	1992–1996	This study
	4.1 (0.72–26)^{F}	3.3 (0.54–11.5)^{F}	Scotland	1992–1996	This study
	13.6^{MA}	44.3^{M}	Scotland	1976	FRS data
	16.0 (6.0–67.2)^{MA}	38.0 (0.45–85.6)^{M}	South Africa	1986–1987	Cockroft (1989)
	2.6 (0.09–43.0)^{FA}	1.3 (0.04–49.7)^{F}	South Africa	1986–1987	Cockroft (1989)
	36.1 (4.1–149)*	15.3 (0.43–74.6)*	Gulf of Mexico	N/A	Salata *et al.* (1995)
	584 ± 456	170 (± 190)	Mediterranean	1992	Corsolini *et al.* (1995)
	310	145**	Wales	1988	Morris *et al.* (1989)
	12.1 (2.0–20.7)		Scotland	1988–1991	Wells *et al.* (1994)
	93 (64–187)^{M*}		Gulf of Mexico	1990	Keuhl and Haeber (1995)
	7.2(1.5–18)^{F*}		Gulf of Mexico	1990	Keuhl and Haeber (1995)
	0.52 (0.37–0.67)	7.25 (2.1–14.0)	India	1990–1991	Tanabe *et al.* (1993)
Striped dolphin	6.2 (5.6–20.7)^{M}	6.3 (4.1–14.2)^{M}	Scotland	1992–1994	This study
	2.2^{F}	2.5^{F}	Scotland	1992–1994	This study
	28 (17–38)	37 (23–51)	Japan	1986	Loganathan *et al.* (1990)
	393 ± 202	139 ± 84	Mediterranean	1990	Kannan *et al.* (1993)
	21.5	49.0	Wales	1988	Morris *et al.* (1989)
	46.9–86	23.6–63.5	Mediterranean	1990–1993	Marsili and Focardi (1996)
	16.8	15.4	Aegean Sea	1991	Georgakopoulou *et al.* (1995)
White-beaked dolphin	8.8 (4.2–27.3)^{M}	8.0 (4.5–11.0)^{M}	Scotland	1992–1994	This study
	9.0 (7.3–54.4)^{F}	7.1 (6.0–32.0)^{F}	Scotland	1992–1994	This study
	34.2 (13.4–87.0)^{M}	43.4 (4.5–86.8)^{M}	Canada	1980	Muir *et al.* (1988a)
	22.0 (9.6–39.7)^{F}	27.9 (0.8–88.6)^{F}	Canada	1980	Muir *et al.* (1988a)

Table 4. Continued.

Species	ΣCB ($\mu g\ g^{-1}$)	ΣDDT ($\mu g\ g^{-1}$)	Location	Sampling date	Reference
White-sided dolphin	29.3 (3.6–29.8)[M]	33.2 (7.2–53.4)[M]	Scotland	1992–1994	This study
	6.2 (3.1 –9.2)[F]	5.5 (4.6–6.3)[F]	Scotland	1992–1994	This study
	0.77–63	0.16–54.6	West of Ireland	1994	McKenzie *et al.* (1997)
	42.7 (± 18.0)[*M]	22.5 (± 11.7)[*M]	Faroe Islands	1987	Borrell (1993)
	25.3 (± 21.2)[*F]	15.0 (± 14.4)[*F]	Faroe Islands	1987	Borrell (1993)
Common dolphin	4.5[M]	4.1[M]	Scotland	1993	This study
	(1.1–3.0)[F]	(0.49–2.7)[F]	Scotland	1992–1994	This study
	136[A]	44.3	Scotland	1976	FRS data
	36.5 (31.2–40.6)	14.4 (5.9–26)[O]	Eastern US	1987–1988	Keuhl *et al.* (1991)
Risso's dolphin	4.7[M]	2.1[M]	Scotland	1992	This study
	(0.31–6.6)[F]	(0.10–0.29)[F]	Scotland	1992	This study
	320 (20–610)	200 (5.2–400)	Mediterranean	1992	Corsolini *et al.* (1995)
	43 (7.4–120)[M]	29 (3.2–77)[M]	Japan	1991	Kim *et al.* (1996)
	7.0 (1.7–20)[F]	4.2 (0.45–19)[F]	Japan	1991	Kim *et al.* (1996)
	1.7[M]	5.0[M]	British Columbia	1988	Jarman *et al.* (1997)
Long-finned pilot whale	8.5 (6.2–10.3)[M]	7.9 (7.1–14.1)[M]	Scotland	1992	This study
	1.7 (± 0.22)	7.6 (± 1.0)	Eastern US		Varanasi *et al.* (1993)
	48.8 (± 23.1)[*M]	31.4 (± 19.2)[*M]	Faroe Islands	1987	Borrell (1993)
	26.3 (± 23.1)[*F]	26.3 (± 23.1)[*F]	Faroe Islands	1987	Borrell (1993)
	9.0 (± 3.8)	11.9 (± 6.1)	East Canada		Muir *et al.* (1988b)
	3.5 (± 3.3)	4.7 (± 5.3)	East Cananda		Muir *et al.* (1988b)
Minke whale	0.27[M]	0.27[M]	Scotland	1993	This study
	0.003–0.029	0.01–0.14	Antarctica		Tanabe *et al.* (1986)
	3.7	5.5	Eastern USA		Varanasi *et al.* (1993)
	1.85 (1.51–2.11)[*]	1.01 (0.99–1.07)[*]	Canada	1992	Gauthier *et al.* (1996)

Table 4. Continued.

Species	ΣCB (µg g^{-1})	ΣDDT (µg g^{-1})	Location	Sampling date	Reference
Killer whale	11[M]	8.5[M]	Scotland	1994	This study
	23[F]	17.8[F]	Scotland	1994	This study
	160	NA	Japan	1982	Tanabe *et al.* (1987)
	22 (8.6–56.0)	32 (11.0–93.0)	NW US	1997	Jarman *et al.* (1997)
	16.4[F]	24.5[F]	SE England	1997	Law *et al.* (1997)
Grey seal	2.5 (1.6–10.8)[M]	1.2 (0.82–3.6)[M]	Scotland	1988–1990	This study
	2.6 (1.5–8.7)[F]	1.1 (0.79–2.6)[F]	Scotland	1988–1990	This study
	16.2 (± 6.80)[F*]	2.39 (± 0.83)[F*]	E Canada	1984	Addison and Brodie (1987)
	30.3 (± 17.0)[F*]	3.71 (± 1.41)[F*]	E Canada	1985	Addison and Brodie (1987)
	46.79 (10.2–117)	1.20 (nd–4.16)[O]	NW England	1989	Simmonds *et al.* (1993)
	18 (5.7–28)	4.2 (0.99–6.21)	East England	1988	Law *et al.* (1989)
	77 (61–89)	30 (22–35)	Baltic	1988	Blomkvist *et al.* (1992)
	82 (32–110)	48 (19–91)	Baltic	1988	Blomkvist *et al.* (1992)
		1.1 (0.3–3.0)[†]	W Iceland	1988	Vetter *et al.* (1995)
	(5.0–11.0)		Scotland	1993	Green *et al.* (1996b)
Common seal	(2.1–2.6)[M]	(0.75–1.3)[M]	Scotland	1991–1994	This study
	1.6 (0.37–4.3)[F]	0.50 (0.23–1.6)[F]	Scotland	1991–1994	This study
	12 (5.0–19)[A]	4.5 (1.8–8.6)	Scotland (Mull)	1969	Holden (1978)
	15 (3.4–29)	N/A	South Norway	1988	Bernhoft *et al.* (1994)
	23 (6.7–33)	4.7 (1.02–8.00)	East England	1988	Law *et al.* (1988)
		1.2 (0.1–2.9)[*†]	W Iceland	1988	Vetter *et al.* (1995)
	0.14–38.0	0.10–8.80	Norway	1988	Skaare *et al.* (1990)
	26.0 (5.00–70.0)	1.71 (0.05–5.4)[O]	N Ireland	1988	Mitchell and Kennedy (1993)
	4.99 (0.80–11.5)[A]	1.36 (0.51–2.45)	Moray Firth	1988	Hall *et al.* (1992)
	6.44 (4.04–9.37)[A]	0.80 (0.41–1.45)	West Coast	1988	Hall *et al.* (1992)
	12.1 (2.70–24.8)[4]	1.23 (0.36–2.70)	Orkney	1988	Hall *et al.* (1992)

[*] Lipid weight; [**] ΣDDT = p,p′-DDE + p,p′-DDD + p,p′-DDT; [O] p,p′-DDE only; [†] p,p′-DDE + p,p′-DDT; [A] Arochlor 1254 equivalent; [M] males only; [F] females only; ( ) range or ± standard deviation

Table 5. Comparative toxic and biochemical potencies of coplanar PCBs (Safe, 1990) [to be revised].

		ED50/EC50 values			
Response	Target cell/species	3,3′,4,4′-tetra CB (CB 77)	3,3′,4,4′,5-penta CB (CB 126)	3,3′,4,4′,5,5′-hexa CB (CB 169)	2,3,7,8-TCDD
Subchronic toxicities					
Body weight loss	Rat (mol/kg)	>500	3.3	15	0.05
Thymic atrophy	Rat (mol/kg)	>500	0.95	8.9	0.09
Bursal lymphoid development	Chick embryo (g/kg)	50	4	300	
Thymic lymphoid development	Mouse foetuses	3×10^{-7}	2×10^{-9}	2×10^{-7}	2×10^{-10}
Immunotoxicity	Mice mol/kg				0.0024
Teratogenicity	Mice mol/kg			0.055-0.110	0.011
AHH induction	Rat mol/kg	500	0.50	1.10	0.004
AHH induction	H-4 II E cells	3.5×10^{-8}	2.4×10^{-10}	6.0×10^{-8}	7.2×10^{-11}
AHH induction	Chick embryo hepatocytes	2.2×10^{-9}	2.0×10^{-9}		2.0×10^{-11}
Receptor binding	Rat cytosol	4.3×10^{-7}	1.2×10^{-7}	Insoluble	1.0×10^{-8}

Table 6. Comparative toxic and biochemical potencies of mono-*ortho* coplanar PCBs [to be revised].

		ED50/EC50 values							
Response	Target species/cell	2,3′,4,4′,5-Penta CB 118	2,3,3′,4,4′-Penta CB 105	2′,3,4,4′,5-Penta CB 123	2,3,4,4′-Penta CB 114	2,3,3′,4,4′,5-Hexa CB 156	2,3′,4,4′,5,5′-Hexa CB 167	2,3,3′,4,4′,5′-Hexa CB 157	2,3,7,8-TCDD
Subchronic studies									
Body weight loss	Rat (mol/kg)	1450	750	370	180	180		220	0.05
Thymic atrophy	Rat (mol/kg)	1550	1030	2790	200	180		225	0.09
Bursal lymphoid development	Chick embryo (g/kg)	>6000	<6000			<6000	>6000	>6000	
Immuno-toxicity	Mice mol/kg					2			0.0024
Teratogenicity	Mice mol/kg					328			0.011
AHH induction	Rat mol/kg	165	65	130	30	7		6	0.004
AHH induction	Rat hepatoma cells	1.2×10^{-5}	8.8×10^{-8}	3.9×10^{-6}	9.7×10^{-7}	2.1×10^{-4}	1.3×10^{-5}	1.1×10^{-5}	7.2×10^{-11}
AHH induction	Chick embryo hepatocytes	5.0×10^{-8}			1.0×10^{-7}	1.4×10^{-5}			2.0×10^{-11}
Receptor binding	Rat cytosol	9.1×10^{-5}	4.3×10^{-6}	1.4×10^{-6}	4.7×10^{-6}	7.1×10^{-6}	1.6×10^{-5}	5.0×10^{-6}	1.6×10^{-5}

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 TEQs for the di-, mono-, and non-*ortho* substituted PCBs (ΣCB-TEQ) are given for each species in Table 7 and range from 8.8–1154 pg g^{-1} lipid weight (8.7–894 pg g^{-1} 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 ΣTEQ-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 7. A less significant correlation was observed between ΣCB and ΣTEQ-CB as noted previously by de Boer *et al.* (1993). The linear regression for all species is given in Figure 2 and shows an R^2 of 0.912 ($p < 0.0005$). The 95 % prediction intervals show that ΣTEQ-CB may be predicted from CB153 concentrations within a factor of two at lower concentrations (2.5 μg g^{-1} lipid CB153) and within a factor of 1.2 at higher concentrations (20.0 μg g^{-1} lipid CB153).

The contribution of each congener to ΣTEQ-CB is shown graphically in Figure 3. It can be seen that CBs 126, 118 and 156 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 4a–4c show Atlantic white-sided dolphin data, for which 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 ΣTEQ-CB as a result of increasing CB burden (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. The animals were found to be lactating females, previously found to have CB patterns that differed from those of the other individuals due to preferential transfer of lower chlorinated congeners during lactation (McKenzie *et al.*, 1997). 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 metabolisable CB118 to logCB153 and a significant positive relationship of the unmetabolisable 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 metabolisable congener. However, as for the unmetabolisable congeners, the proportion of ΣTEQ-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 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 CBs 156, 170 and 180 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 metabolising 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]**. Members of the same class of compound 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

Figure 2. ΣTEQ-CB as a function of CB153 concentration in marine mammals from Scottish and Irish waters.

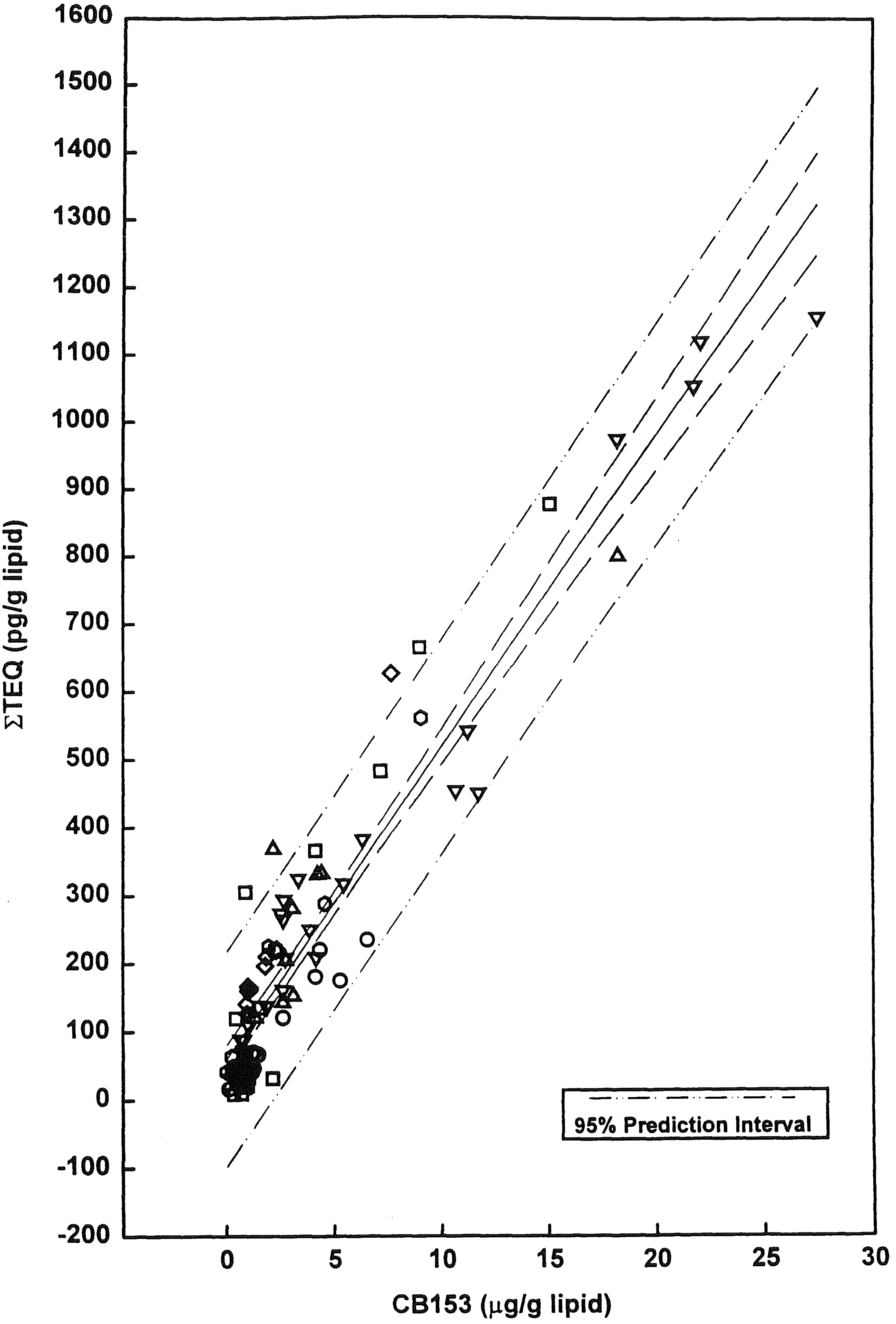


Figure 3. Percent contribution of individual CBs to ΣTEQ-CB in marine mammals from Scottish and Irish waters.

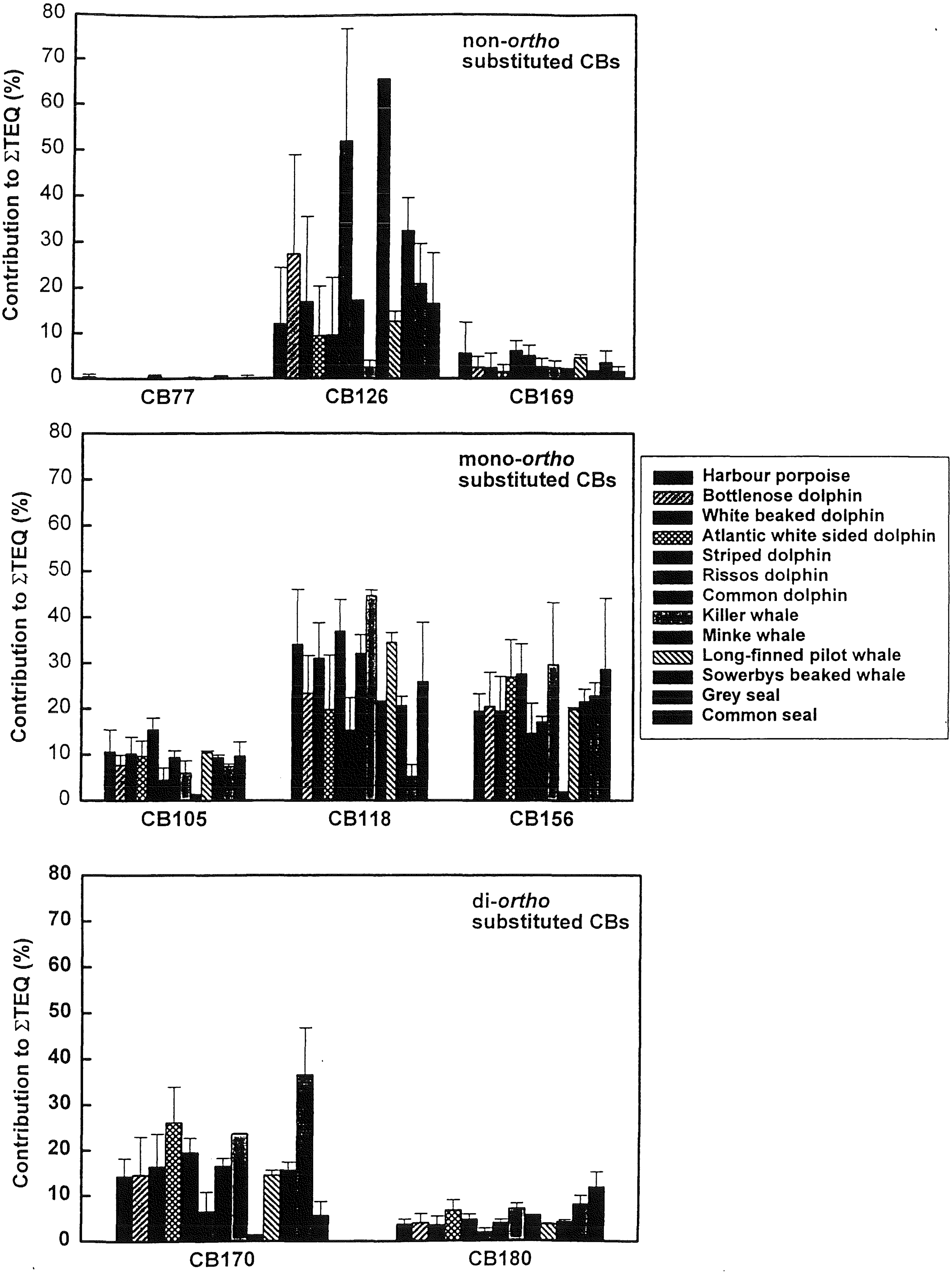


Figure 4a. Changes in the contribution of mono-*ortho* CB118 to ΣTEQ-CB in relation to contaminant burden.

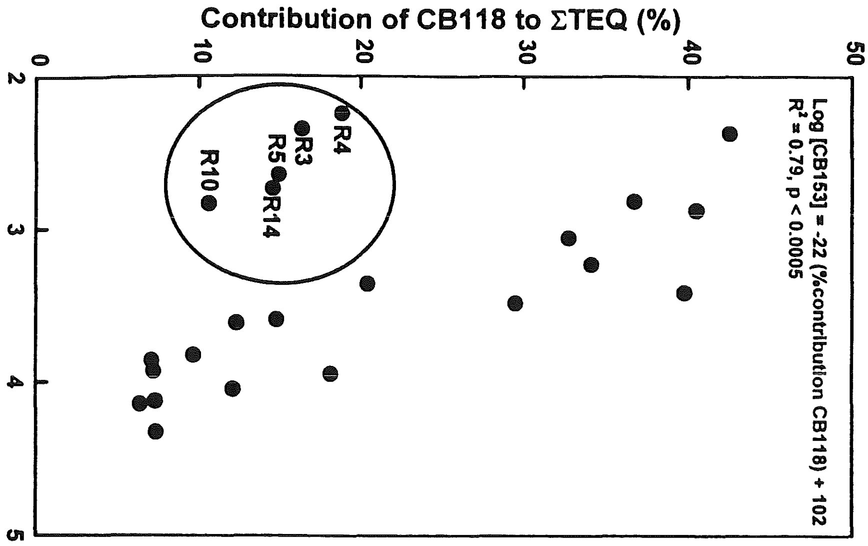


Figure 4b. Changes in the contribution of mono-*ortho* CB156 to ΣTEQ-CB in relation to contaminant burden.

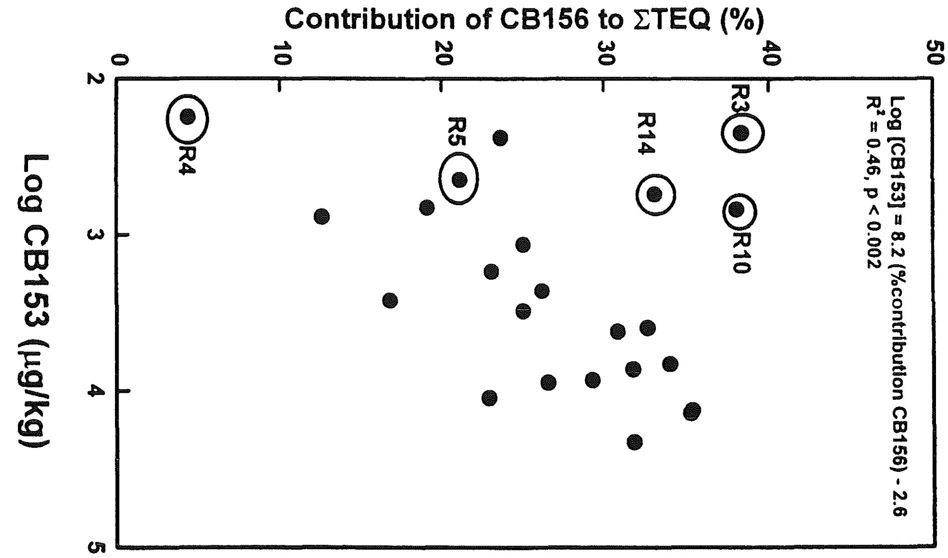


Figure 4c. Changes in the contribution of mono-*ortho* CB170 to ΣTEQ-CB in relation to contaminant burden.

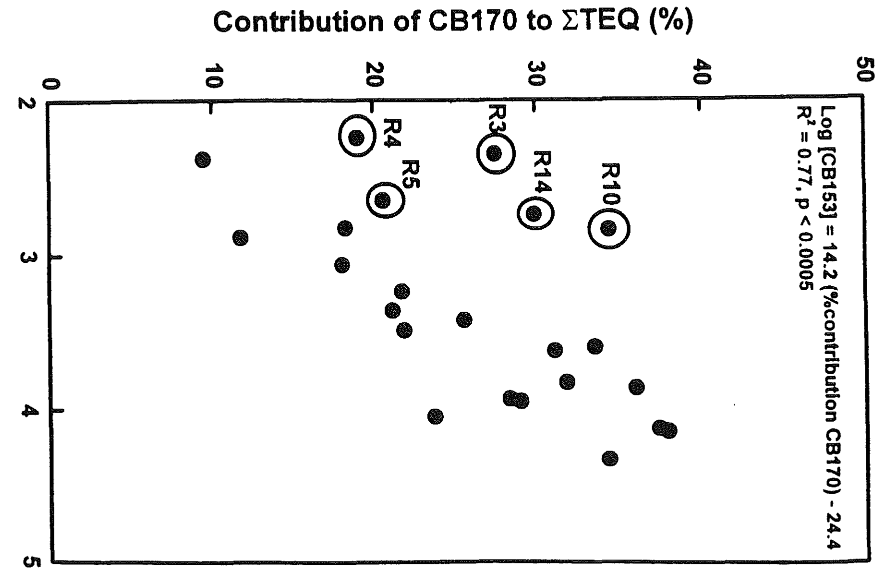


Table 7. Median (and range) of ΣEQ-CB concentrations in marine mammals from Scottish and Irish waters.

Species	**n**	**% extractable lipid in blubber**	**pg g^{-1} lipid**			
			ΣTEQ-CB^A	**ΣEQ-CB^B**	**R^2**	**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

ΣTEQ-CB^A = (TEQCB77 + TEQCB105 + TEQCB118 + TEQCB126 + TEQCB156 + TEQCB169)
ΣTEQ-CB^B = (TEQCB77 + TEQCB105 + TEQCB118 + TEQCB126 + TEQCB156 + TEQCB169 + TEQCB170 + TEQCB180)
R^2: Pearson correlation coefficient for the equation ΣTEQ-CB^B = gradient [CB153 (μg g^{-1} lipid)] + constant

may, however, be indicative of where the emphasis for studying the biological effects of non-*ortho* and mono-*ortho* PCBs for marine mammals should be directed (Hall, 1998).

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 metabolised. 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 PCB congeners. By comparing the ratio of each congener to a persistent congener (such as CB 153 or CB 180; which is found at consistently high levels in marine biota), patterns and concentrations can be standardised 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; **[Wells *et al.*, 1996; Boon *et al.*, 1997]**).

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 metabolise all congeners with *m,p*-vicinal H atoms (Groups IV and V in Table 2) **[although there is direct evidence that beluga whales (*Delphinapterus leucas*) possess CYP2B enzymes which are capable of metabolising these types of compounds (White *et al.*, 1994) and indirect evidence from CB patterns that harbour porpoise are able to metabolise some (CBs 44 and 101) (Bruhn *et al.*, 1996; 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 (CBs 105, 118 and 156) with a mixed CYP1A/CYP2B induction pattern were among the metabolisable 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 total TEQ and CB153; however, the proportion of toxicity derived from at least two of the mono-*ortho* congeners decreases with increasing CB153 concentrations whilst the toxicity derived from CB156 and the persistent di-*ortho* congeners increases (McKenzie *et al.*, 1998; van Scheppeningen *et al.*, 1996).]**

4.1.2 Trace elements and metals

With the exception of zinc, concentrations of heavy metals increase with age in both sexes of marine mammal species. But, in contrast to lipophilic pollutants, concentrations in females are similar to or higher than in males. The significance of these variables should be taken into account when designing sampling, comparing sample groups, or evaluating toxicological impact (Aguilar et al., in press).

Mercury, copper, zinc and other heavy metals accumulate mainly in the liver, while cadmium does so in the kidney, and lead in bone (Honda et al., 1983; André et al., 1990a, 1990b). Wagemann and Muir (1984) found the tolerance limit of marine mammals to mercury to be 100–400 µg Hg/g liver fresh weight.

Frank et al. (1992) determined the concentrations of different metals in liver and kidney cortex in seals from Swedish waters. The metals studied were Al, Ca, Cd, Co, Cr, Cu, Fe, Mg, Mn, Ni, Pb, V, W and Zn. No spatial difference in concentrations between the Swedish west coast and the Baltic Sea was found, except for Cr which was higher in Baltic harbour seals, indicating possible local contamination. Metal concentrations, except for Cr, among the Baltic juveniles were generally lower in the harbour seals than those in both grey and ringed seals (Phoca hispida botnica). Grey seals displayed the highest Pb, Hg and Se concentrations compared to the other two seal species studied. Age-dependent increases in concentrations were only ascertained for Cd, Hg and Se.

4.2 Effect of contaminants

4.2.1. Introduction

The most commonly reported responses to PCB exposure, largely based on data from laboratory studies, indicate that a range of body systems may be affected (Hall, 1998). These include body weight loss, thymic atrophy, impairment of immune responses, hepatotoxicity and prophyria, chloracne and related dermal lesions, tissue-specific hypo- 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 metabolising enzymes, and it is the induction of the phase I, P450 enzymes in particular, (recently reported in various cetaceans and seal species) in conjunction with higher PCB uptake which increases the concern about the toxicological importance of the non- and mono-*ortho* PCB 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 PCB 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 PCB congeners in mammals and these are shown in Table 8. 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 metabolise 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 and coplanar PCBs 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- and mono-*ortho* PCB congeners.

Table 8. Target organ toxicity of non- and mono-*ortho* PCBs in mammals (from Battershill, 1994).

Target organ/system	Comments
Skin (Chloracne in monkeys)	Non-*ortho* coplanar PCBs, CB 169 and CB 77 most potent. Di-*orthos* gave a weak response No effects seen with tetra-*orthos*. No clear gradation in potency across groups of PCBs evident.
Liver (Histology, rats, mice, monkey)	A clear gradation between different groups of PCBs: non-*ortho* > mono-*ortho* > di-*ortho* > others. However, potency of CB 77 andCB 105 less than expected on this gradation, possibly due to greater metabolism of these compared with other non and mono-*ortho* CBs.
Immunotoxicity (rat, mouse) (plaque forming response)	A clear gradation between different groups of PCBs; non-*ortho* > mono-*ortho* > di-*ortho* > others. ED50 for thymic atrophy in rats correlated with AHH induction except for CB 77 and CB 105. Effects different in 2,3,7,8-TCDD responsive and non-responsive strains of mice.
Reduced serum thyrosine (mink)	Evidence of effects with CB 169.
Reduced plasma corticosteroid (mice)	Evidence of effects with CB 77.
Follicular hyperplasia (monkey) Gastrointestinal tract (mink, monkey)	Evidence of effects with CB 77; no effects seen with di-*ortho* coplanars. No effects seen in mice fed CB 169 or di-*ortho* coplanars
Decreased liver/serum retinol levels (rat)	Reported potency CB 126> CB 77 > CB 124. Available data consistent with an Ah-related effect.

Many studies have measured the concentrations of mono- and non-*ortho* PCBs 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, in press).

4.2.2 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).

4.2.3 Sub-lethal effects

4.2.3.1 Effects on reproduction and early development

Trans-Generational Transfer of CBs

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; McKenzie *et al.*, 1997). 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). 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 summarized in Table 9. The harbour porpoise foetus contained the highest contaminant concentrations, 770 and 309 $\mu g\ kg^{-1}$ 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 concentration present was lower than observed in the harbour porpoise. The ΣTEQ from the non-, mono-, and di-*ortho* CBs (CBs 77, 105, 118, 126, 156, 169, 170 and 180) was measured in the Risso's dolphin foetus (39 $pg\ g^{-1}$ lipid) which represented 39 % of the mothers total ΣTEQ 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 seal (1 %) than in harbour porpoise (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 may be 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 were not 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.

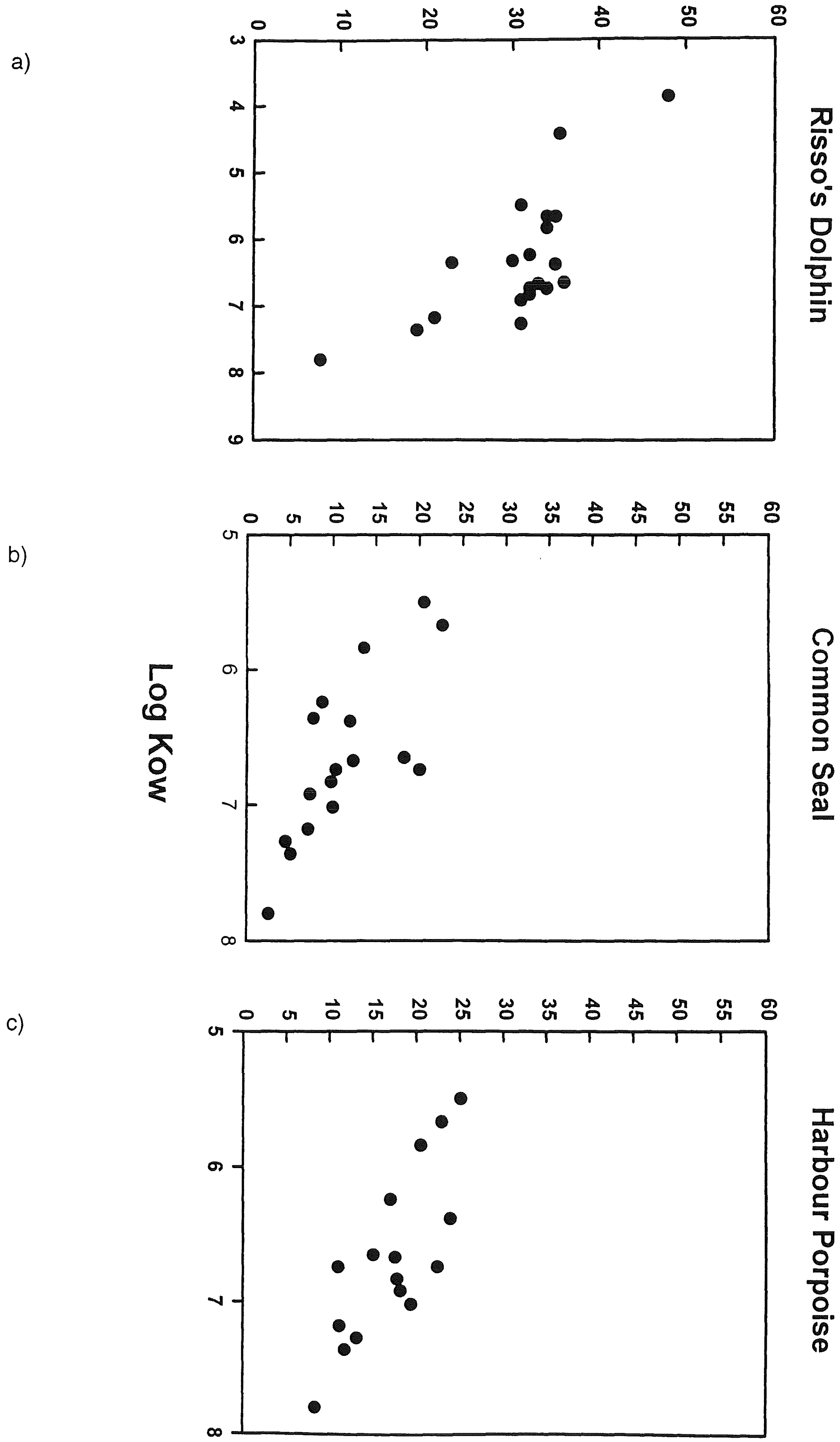
During gestation and lactation, organochlorines are selectively transferred to the foetus or calf on the basis of their lipophilicity (Tanabe *et al.*, 1981; Green *et al.*, 1996b; McKenzie *et al.*, 1997). Figure 5 shows the preferential gestational transfer to the foetus of less lipophilic compounds, which have lower octanol/water

Table 9. Percentage of mother's burden and concentration (µg/kg) of ΣCB and ΣDDT passed to the foetus during transplacental transfer.

Species	% ΣCB* transferred	ΣCB transferred	% ΣDDT transferred	ΣDDT transferred	% Σchlordane transferred	Σchlordane transferred	%HCB transferred	HCB transferred	% ΣTEQ transferred	ΣTEQ (pg g^{-1}) transferred
Risso's dolphin	29 (40)	270 (614)	30 (40)	123 (279)	32 (43)	60 (135)	26 (36)	11 (21)	28 (39)	17 (39)
Common seal	7.7 (8.9)	164 (382)	8.9 (10.3)	58 (135)	12 (14)	35 (43)	21 (23)	3.3 (7.8)	na	na
Harbour porpoise	17 (17)	770 (1030)	17 (17)	309 (412)**	na	na	25 (25)	53 (71)	na	na

na = not available; values in parentheses are or have been calculated from lipid weight concentrations
*%X transferred = (foetus burden/(foetus + mother burden))*100
**Σp,p'-DDE only
ΣTEQ = (TEQCB77+TEQCB105+TEQCB118+TEQCB126+TEQCB156+TEQCB169+TEQCB170+TEQCB180) calculated using TEFs derived by Ahlborg *et al.* (1994)

Figure 5. Dependency of transplacental transfer on lipophilicity (log K_{ow}) in (a) Risso's dolphin, (b) common seal, and (c) harbour porpoise.



coefficients (log K_{ow}), such as γ-HCH, 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.*, 1997) Log K_{ow} values for CBs were taken from Hawker and Connel (1991) 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 humans and animals have only become known in the past years. Effects have also been observed in the reproduction 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, such as stenosis and occlusion of the uterus, have been described for seals from British, Swedish and Finnish waters (Helle *et al.*, 1976a, 1976b, 1980; Baker, 1989; Olsson *et al.*, 1994; Bergman, 1997). These lesions have been attributed to high PCB and DDT levels in these animals. No such lesions have been found in cetaceans. Between 1977–1986 about 42 % of 4-year-old female Baltic grey seals that were examined showed stenosis 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 in the second period it increased up to 60 % (Bergman, 1997). It has been suggested that an improvement of the seal population is due to a declining contaminant burden of the Baltic biota (ICES, 1997) [**although this could not be categorically assigned to CBs as there are likely to have been a great number of other unknown contaminants present**].

Little is known about the pathology of sexual development during pregnancy in marine mammals. True hermaphroditism 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, pseudohermaphroditism 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 reproduction system and early development appear to have a strong biological effect on marine mammal populations. Further systematic investigations are **recommended**, as for example on the effects of transplacental transfer of toxins to the foetus.

In vitro effects of methyl-mercury on steroid synthesis in the adrenal glands and testes of harp seals (*Phoca groenlandica*) and grey seals have been measured (Freeman *et al.*, 1975; Freeman and Sangaland, 1977).

4.2.3.2 Immune suppression, diseases and epizootics

Evidence of the effects of CBs on the immune system has been demonstrated in both humans and other animals [**Brouwer *et al.*, 1989—lower blood retinol leveles in seals fed contaminanted fish (this is indicative of vitamin A deficiency)**]. In the Netherlands, seals experimentally fed on contaminated fish from the Baltic Sea showed functional failure of 'Natural Killer' (NK) cell activity and a decrease of T-cell function with increasing 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 resistance to cancer development (De Guise *et al.*, 1995a) and to morbillivirus infections (Dietz *et al.*, 1989; Kannan *et al.*, 1993; Aguilar and Borrell, 1994). Other experimental studies could not reproduce such effects on mortality (Hall *et al.*, 1992; Harder *et al.*, 1992). However, morbilliviruses are highly virulent agents acting immunosuppressively, so it is therefore very difficult to investigate the potential relationship between CBs and epizootics on this virus. No experiments have been done on cetaceans, but some [**very limited**] *in vitro* experiments, responses such as mitogen-induced lymphocyte proliferation response in bottlenose dolphins (*Tursiops truncatus*) were studied (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 $HgCl_2$. Decreased splenocyte and thymocyte proliferation was observed with the highest concentration of $HgCl_2$ and $CdCl_2$ (10^{-5} M). De Guise *et al.* (1995c) altered 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.

4.2.3.3 Effects on skeletal and endocrine systems

Several authors have described loss of bone substance in the skull and asymmetry of the skull in several seal species from the North and Baltic Seas (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).

4.2.3.4 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 cancers, 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 worldwide 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.*, in press). 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 recognised as promoters for the induction of tumours in initiated cells (De Guise *et al.*, 1995a).

A higher prevalence of tumours such as leiomyomas 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 benign 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, despite the high prevalence of tumors, recent surveys show recovery of the beluga population in the St. Lawrence estuary after hunting was banned in 1979 (Kingsley, in press).*

4.2.3.5 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 co-ordination failures in a harbour seal with a burden of 200 µg Hg g^{-1} liver fresh weight (Law *et al.*, 1991).

4.2.3.6 Other 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 lipofuszin, 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 picture of blood status indicating toxic hepatitis, uremia, and failure of kidneys (Holden, 1978).

4.2.4 Indirect effects

The IWC Workshop focused on the role of fish in pollutant transmission to cetaceans (IWC, 1995). Fish are unable to metabolise 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 abundance of marine resources and therefore also cetacean food availability.

4.3 Summary and Conclusions

Chlorobiphenyls 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 with regard to contaminants and cetaceans. *(WGMMHA, however, found it most)* **It is** likely that population-level effects of contaminants on marine mammals, in particular caused by non- and mono-*ortho* CBs, do occur.

With particular reference to the recommendations from the IWC Workshop (IWC, 1995, 1997), WGMMHA therefore ***recommended*** *that a research programme aimed at understanding and describing cause-effect relationships between environmental contaminants and population-level effects in marine mammals be carried out (cf. Sections 6, 7 and 8 of this report).* The effects of chronic exposure and the combined effects of several toxins in the natural environment need to be understood.

[Methods need to be developed and knowledge needs to be gained by investigating live and dead animals for effects of contaminants. Therefore, long-term biological effects monitoring is urgently required.]

References

Addison, R.F., and Brodie, P.F. 1977. Organochlorine residues in maternal blubber, milk, and pup blubber from grey seals (*Halichoerus grypus*) from Sable Island, Nova Scotia. Journal of the Fisheries Research Board of Canada, 34: 937–341.

Addison, R.F., and Brodie, P.F. 1987. Transfer of organochlorine residues from blubber through the circulatory system to milk in the lactating grey seal, *Halichoerus grypus*. Canadian Journal of Fisheries and Aquatic Sciences, 44: 782–786.

Aguilar, A. 1984. Relationship of DDE/ΣDDT in marine mammals to the chronology of DDT input into the ecosystem. Canadian Journal of Fisheries and Aquatic Sciences, 41: 840–844.

Aguilar, A. 1987. Using organochlorine pollutants to discriminate marine mammal populations: a review and critique of the methods. Marine Mammal Science, 3: 242–262.

Aguilar, A., and Borrell, A. 1994. Abnormally high polychlorinated biphenyl levels in striped dolphins (*Stenella coeruleoalba*) affected by the 1990–1992 Mediterranean epizootic. Science of the Total Environment, 154(2ñ3): 237–247.

[Aguilar, A., Bloch, D., and Desportes, G. 1995. Age trends and reproductive transfer of organochlorine compounds in long-finned pilot whales from the Faroe Islands. Environmental Pollution, 88: 283–292.]

Ahlborg, U.G., Becking, G.C., Birnbaum, L.S., Brouwer, A., Derks, H.J.G.M., Feeley, M., Golor, G., Hanberg, A., Larsen, J.C., Liem, A.K.D., Safe, S.H., Schlatter, C., Waern, F., Younes, M., and Yrjänheikki, E. 1994. Toxic equivalency factors for dioxin-like PCBs. Chemosphere, 28: 1049–1067.

Amdur, M.O., Doull, J., and Klaassen, C. (eds) 1991. Casarett and Doull's Toxicology. The Basic Science of Poisons. Pergamon Press, New York.

André, J.M., Amiard, J.C., Amiard-Triquet, C., Boudou, A., and Ribeyre, F. 1990a. Cadmium contamination of tissues and organs of delphinid species (*Stenella attenuata*) - influence of biological and ecological factors. Ecotoxicology and Environmental Safety, 20: 290–306.

André, J.M., Ribeyre, F., and Boudou, A. 1990b. Mercury contamination levels and distribution in tissues and organs of Delphinids (*Stenella attenuata*) from Eastern Tropical Pacific, in relation to biological and ecological factors. Marine Environmental Research, 30: 43–72.

Baker, J.R. 1989. Pollution-associated uterine lesion in grey seals from the Liverpool Bay of the Irish Sea. The Veterinary Record, 125: 303.

Bannister, J. 1962. An intersexual fin whale *Balaenoptera physalus* (L.) from South Georgia. Proceedings of the Zoological Society, London, 141: 811–812.

Battershill, J.M. 1994. Review of the safety assessment of polychlorinated biphenyls (PCBs) with particular reference to reproductive toxicology. Human and Experimental Toxicology, 13: 581–597.

[Beck, H., Breuer, E.M., Dross, A., and Mathar, W. 1990. Residues of PCDDs, PCDFs, PCBs and other organochlorine compounds in harbour seals and harbour porpoise. Chemosphere, 20: 1027–1034.]

Béland, P., De Guise, S., Girard, C., Lagacé, A., Martineau, D., Michaud, R., Muir, D.C.G., Norstrom, R.J., Pellitier, E., Ray, S., and Shugart, L.R. 1993. Toxic compounds and health and reproductive effects in St. Lawrence beluga whales. Journal of the Great Lakes Research, 19: 766–775.

Bergman, A. 1997. Trends of disease complex in Baltic grey seals (*Halichoerus grypus*) from 1977 to 1996: improved gynecological health but still high prevalence of fatal intestinal wounds. WP19 presented at the Working Group on Seals and Small Cetaceans in European Seas, Stockholm, Sweden. [Mimeo.]

Bergman, A., and Olsson, M. 1986. Pathology of Baltic grey seal and ringed seal females with special reference to adrenocortical hyperplasia: Is environmental pollution the cause of a widely distributed disease syndrome? Proceedings from the Symposium on the seals in Eurasian lakes. Finnish Game Research, 44: 47–62.

Bergman, A., Olsson, M., and Reiland, S. 1992. Skull-bone lesions in the Baltic grey seal (*Halichoerus grypus*). Ambio, 21(8): 517–519.

[Bernhoft, A., and Skaare, J.U. 1994. Levels of selected individual polychlorinated biphenyls in different tissues of harbour seals (*Phoca vitulina*) from the southern coast of Norway. Environmental Pollution, 86: 99–107.]

[Blomkvist, G., Roos, A., Jenson, S., Bignert, A., and Olsson, M. 1992. Concentrations of sDDT and PCB in seals from Swedish and Scottish waters. Ambio, 21(8): 539–545.]

Boon, J.P., and Eijgenraam, F. 1988. The possible role of metabolism in determining patterns of PCB congeners in species from the Dutch Wadden Sea. Marine Environmental Research, 24: 3–8.

Boon, J.P., Oostingh, I., van der Meer, J., and Hillebrand, M.T.J. 1994. A model for the bioaccumulation of chlorobiphenyl congeners in marine mammals. European Journal of Pharmacology, 270: 237–251.

Boon, J.P., Reijnders, P.J.H., Dols, J., Wensvoort, P., and Hillebrand, M.T.J. 1987. The kinetics of individual polychlorinated biphenyl congeners in female harbour seals (*Phoca vitulina*), with evidence for structure-related metabolism. Aquatic Toxicology, 10: 307–324.

Boon, J.P., van Arnhem, E., Jansen, S., Kannan, N., Petrick, G., Duinker, J.C., Reijnders, P.J.H., and Goksøyr, A. 1992. The toxicokinetics of PCBs in marine mammals with special reference to possible interactions of individual congeners with the cytochrome P450-dependent monooxygenase system: an overview. *In* Persistent pollutants in the marine ecosystem, pp. 119–159. Ed. by C.H. Walker and D.R. Livingstone. Pergammon, Oxford.

[Boon, J.P., van der Meer, J., Allchin, C.R., Law, R.J., Klungsøyr, J., Leonards, P.E.G., Spliid, H., Storr-Hansen, McKenzie, C., and Wells, D.E. 1997. Concentration-dependent changes of PCB patterns in fish-eating

mammals: structural evidence for induction of cytochrome P450. Arch. Environ. Contam. Toxicol., 33: 298–311.]

Borrell, A. 1993. PCB and DDTs in blubber of cetaceans from the northeastern North Atlantic. Marine Pollution Bulletin, 26(3): 146–151.

Borrell, A., and Reijnders, P.J.H. In press. Summary of temporal trends in pollutant levels observed in marine mammals. Reports of the International Whaling Commission, (Special Issue) 17.

[Brouwer, A., Klasson-Wehler, E., Bokdam, M., Morse, D.C., and Traag, W.A. 1990. Competative inhibition of thyroxine binding to transthyretin by monohydroxy metabolites of 3,4,3',4'-tetrachlorobiphenyl. Chemosphere, 20: 1257.]

Brouwer, A., Reijnders, P.J.H., and Koeman, J.H. 1989. Polychlorinated biphenyls (PCB)-contaminated fish induces vitamin A and thyroid hormone deficiency in the common seal (*Phoca vitulina*). Aquatic Toxicology, 15: 99–106.

Bruhn, R., Kannan, N., Petrick, G., Shulz-Bull, D.E. and Duinker, J.C. 1995. CB pattern in the harbour porpoise:bioaccumulation, metabolism and evidence for cytochrome IIB activity. *Chemosphere*, 3721-3732.

[Cockcroft, V.G., de Cock, A.C., Lord, D.A., and Ross, J.G.B. 1989. Organochlorines in bottlenose dolphins (*Tursiops truncatus*) from the east coast of South Africa. South African Journal of Marine Science, 8: 207–217.]

Corsolini, S., Focardi, S., Kannan, K., Tanabe, S., and Borrell, A. 1995. Congener profile and toxicity assessment of polychlorinated biphenyls in dolphins, sharks and tuna collected from Italian coastal waters. Marine Environmental Research, 40: 33–53.

Crosby, D.G. 1969. The non-metabolic decomposition of pesticides. *In* Biological effects of pesticides in mammalian systems, pp. 82-96. Ed. by H.F. Kraybill. Annals of the New York Academy of Sciences, Vol. 160.

[de Boer, J., Stronck, C.J.N., Traag, W.A., and van der Meer, J. 1993. Non-*ortho* and mono-*ortho* substituted chlorobiphenyls and chlorinated dibenzo-*p*-dioxins and dibenzofurans in marine and freshwater fish and shellfish from the Netherlands. Chemosphere, 26: 1823–1843.]

de Boer, J., Stronck. C.J.N., Traag, W.A., and van der Meer, J. 1993. Non-*ortho* and mono-*ortho* substituted chlorobiphenyls and chlorinated dibenzo-*p*-dioxins and dibenzofurans in marine and freshwater fish and shellfish from the Netherlands. Chemosphere, 26: 1823–1842.

De Bruin, A. 1980. Biochemical toxicology of environmental agents. Elsevier, Amsterdam.

De Guise, S., Bernier, J., Defresue, M.M., Martineau, D., Béland, P., and Fournier, M. 1995b. Immune functions in beluga whales (*D. leucas*): evaluation of mitogen-induced blastic transformation of lymphocytes from peripheral blood, spleen and thymus. Veterinary Immunology and Immunopathology, 50: 117–126.

De Guise, S., Flipo, D., Boehm, J.R., Martineau, D., Béland, P., and Fournier, M. 1995c. Immune functions in beluga whales (*D. leucas*): evaluation of phagocytosis and respiratory burst with peripheral blood leucocytes using flow cytometry. Veterinary Immunology and Immunopathology, 47: 351–362

De Guise, S., Lagacé, A., and Béland, P. 1994a. Tumors in St. Lawrence beluga whales (*Delphinapterus leucas*). Veterinary Pathology, 31(4): 444–449.

De Guise, S., Lagacé, A., and Béland, P. 1994b. Gastric papillomas in eight St. Lawrence beluga whales (*Delphinapterus leucas*). Journal of Veterinary Diagnostic Investigations, 6: 385–388.

De Guise, S., Lagacé, A., and Béland, P. 1994c. True hermaphroditism in a St. Lawrence beluga whale (*Delphinapterus leucas*). Journal of Wildlife Diseases, 30(2): 287–290.

De Guise, S., Martineau, D., Béland, P., and Fournier, M. 1995a. Possible mechanisms of action of environmental contaminants on St. Lawrence beluga whales (*Delphinapterus leucas*). Environmental Health Perspectives, 103 (Suppl 4): 73–77.

De Swart, R.L., Ross, P.S., Vedder, E.J., Timmerman, H.H., de Heisterkamp, S.H., van Loveren, H., Vos, J.S., Reijnders, P.J.H., and Osterhaus, A.D.M.E. 1994. Impairment of immune function in harbor seals (*Phoca vitulina*) feeding on fish from polluted waters. Ambio, 23: 155–159.

De Swart, R.L., Ross, P.S., Vos, J.G., and Osterhaus, A.D.M.E. 1996. Impaired immunity in harbour seals (*Phoca vitulina*) exposed to bioaccumulated environmental contaminants: review of a long-term feeding study. Environmental Health Perspectives, 104 (Suppl. 4): 823–828.

DeLong, R.L., Gilmartin, W.G., and Simpson, J.G. 1973. Premature birth in California sea lions: association with high organochlorine pollution residue levels. Science, 181: 1168–1169.

Dietz, R., Heide-Jørgensen, M.P., and Härkönen, T. 1989. Mass death of harbour seals (*Phoca vitulina*) in Europe. Ambio, 18(5): 258–264.

[Donkin, P., Mann, S.V., and Hamilton, E.I. 1981. Polychlorinated biphenyl, DDT and dieldrin residues in grey seal (*Halichoerus grypus*) males, females and mother-foetus pairs sampled at the Farne Islands, England, during the breeding season. Science of the Total Environment, 19: 121–142.]

[Duinker, J.C., and Hillebrand, M.T.H. 1979. Mobilisation of organochlorines from female lipid tissue and transplacental transfer to fetus in a harbour porpoise (*Phocoena phocoena*) in a contaminated area. Bulletin of Environmental Contamination and Toxicology, 23: 728–732.]

Duinker, J.C., Hillebrand, M.T.J., Zeinstra, T., and Boon, J.P. 1989. Individual chlorinated biphenyls and pesticides in tissues of some cetacean species from the North Sea and the Atlantic Ocean; tissue distribution and biotransformation. Aquatic Mammals, 15: 95–124. Fossi, M.C., Marsil, L., Leonzio, C., Notarbartolo Di Sciara, G., Zanardelli, M., and Focardi, S. 1992. The use of non-destructive biomarker in Mediterranean cetaceans: preliminary data on MFO activity in skin biopsy. Marine Pollution Bulletin, 24: 459–461.

[Falandysz, J., Yamashita, N., Tanabe, S., Tatsukawa, R., Rucinska, L., and Skora, K. 1994. Congener-specific data on polychlorinated biphenyls in tissues of common porpoise from Puck Bay, Baltic Sea. Arch. Environ. Contam. Toxicol., 26: 267–272.]

[Finizio, A., Vighi, M., and Sandroni, D. 1997. Determination of n-octanol/water partition coefficient (K_{ow}) of pesticides: Critical review and comparison of methods. Chemosphere, 34: 131–161.]

Frank, A., Galgan, V., Roos, A., Olsson, M., Petersson, L.R., and Bignert, A. 1992. Metal concentrations in seals from Swedish waters. Ambio, 21: 529–538.

Freeman, H.C., and Sangaland, G.B. 1977. A study of the effects of methyl mercury, cadmium, arsenic, selenium and a PCB (Arochlor 1254) on adrenal testicular steroidogenesis *in vitro*, by the grey seal *Halichoerus grypus*. Archives of Environmental Contamination and Toxiciology, 5: 369–383.

Freeman, H.C., Sangaland, G.B., Uthe, J.F., and Ronald, K. 1975. Steroidogenesis *in vitro* in the harp seal (*Pagophilus greonlandicus*) without and with methylmercury treatment *in vivo*. Environmental Physiology and Biochemistry, 5: 428–439.

Frost, K.J., and Lowry, L.F. 1993. Marine Mammal Study Number 5: Assessment of injury to harbor seals in Prince William Sound, Alaska, and adjacent areas following the Exxon Valdez oil spill. State-federal natural resources damage assessment, April through September 1991.

Geraci, J.R. 1990. Physiologic and toxic effects on cetaceans. *In* Sea mammals and oil: confronting the risks, pp. 167–198. Ed. by J.R. Geraci and D.J. St. Aubin. Academic Press Inc., San Diego.

Geraci, J.R., and St. Aubin, D.J. 1990. Sea mammals and oil: confronting the risks. Academic Press Inc., San Diego.

Geraci, J.R., Palmer, N.C., and St. Aubin, D.J. 1987. Tumors in cetaceans: analysis and new findings. Canadian Journal of Fisheries and Aquatic Sciences, 4: 1289–1300.

[Gauthier, J.M., Metcalfe, C.D., and Sears, R. 1996. Validation of the blubber biopsy technique for monitoring of organochlorine contamination Balaenopterid whales. Marine Environmental Research, 43(3): 157–179.]

[Georgakopoulou-Gregoriadou, E., Psyllidou-Giouranovits, R., Voutsinou-Taliadouri, F., and Catsiki, V.A. 1995. Organochlorine residues in marine mammals from the Greek waters. Fresenius Environmental Bulletin, 4: 375–380.]

[Gladen, B.C., Rogan, W.J., Hardy, P., Thullen, J., Tingelstad, J. and Tully, M. 1988. Developement after exposure to polychlorinated biphenyls and dichlorodiphenyl dichloroethene transplacentally and through human milk. Journal of Pediatrics, 113: 991–995.]

Goksøyr, A., Beyer, J., Larsen, H.E., Andersson, T., and Forlin, L. 1992. Cytochrome P450 in seals: monooxygenase activities, immunochemical cross-reactions and response to phenobarbital treatment. Marine Environmental Research, 34: 113–116.

Goksøyr, A., Solbakken, J., Tarlebo, J., and Klungsøyr, J. 1986. Initial characterisation of the hepatic microsomal cytochrome P-450 system of the piked (minke) whale *Balaenoptera acutorostrata*. Marine Environmental Research, 19: 185–187.

Green, N.J.L., Jones, K.C., and Harwood, J. 1996a. Contribution of coplanar and non-coplanar polychlorinated biphenyls to the toxic equivalence of grey seal, *Halichoerus grypus,* milk. Chemosphere, 33: 1273–1281.

[Green, N., van Raat, P., Jones, K.C., and de Voogt, P. 1996b. PCBs and their methyl sulphone metabolites in the maternal blubber, milk, pup blubber and faeces of grey seals. Organohalogen Compounds, 29: 453–457.]

Griem, P., Wulferink, M., Sachs, B., Gonzalea, J., and Gleichmann, E. 1998. Allergic and autoimmune reactions to xenobiotics: how do they arise? Immunology Today, 19: 133–141.

Hahn, M.E., Poland, A., Glover, E., and Stegeman, J.J. 1992. The Ah receptor in marine animals: phylogenetic distribution and relationship to P450IA inducibility. Marine Environmental Research, 34: 87–90.

Hall, A.J. 1998. The possible biological effects of non-*ortho* and mono-*ortho* chlorobiphenyls, with particular reference to marine mammals. Working paper at the 1998 meeting of the Working Group on Marine Mammal Habitats.

Hall, A.J., Law, R.J., Wells, D.E., Harwood, J., Ross, H., Kennedy, S., Allchin, C.R., Campbell, L.A., and Pomeroy, P.P. 1992. Organochlorine levels in common seals (*Phoca vitulina*) which were victims and survivors of the 1988 phocine distemper epizootic. Science of the Total Environment, 115: 145–162.

Hall, A.J., Watkins, J., and Hiby, L. 1996. The impact of the 1993 Braer oil spill on grey seals in Shetland. Science of the Total Environment, 186: 119–125.

Harder, T.C., Willhaus, T., Leibold, W., and Liess, B. 1992. Investigations on course and outcome of phocine distemper infection in harbour seals (*Phoca vitulina*) exposed to polychlorinated biphenyls. Journal of Veterinary Medical Sciences, 39: 19–31.

[Hawker, D.W. and Connel, D.W. 1988. Octanol water partition coefficients of polychlorinated biphenyl congeners. Environmental Science and Technology, 22: 382–387.]

Helle, E. 1980. Lowered reproductive capacity in female ringed seals (*Pusa hispida*) in the Bothnian Bay, northern Baltic Sea, with special reference to uterine occlusions. Annales Zool. Fennici, 17: 147–158.

Helle, E., Olsson, M., and Jensen, S. 1976a. PCB levels correlated with pathological changes in seal uteri. Ambio, 5(4): 261–263.

Helle, E., Olsson, M., and Jensen, S. 1976b. DDT and PCB levels and reproduction in ringed seal from the Bothnian Bay. Ambio, 5(4): 188–189.

Hellou, J., Stenson, G., Ni, I.-H., and Payne, J.F. 1990. Polycyclic aromatic hydrocarbons in muscle tissue of marine mammals from the northwest Atlantic. Marine Pollution Bulletin, 21: 469-473.

Holden, A.V. 1978. Pollutants and seals - A review. Mammal Review, 8(1+2): 53–66.

Honda, K., Tatsukawa, R., Itano, K., Miyazaki, N., and Fujiyama, T. 1983. Heavy metal concentrations in muscle, liver and kidney tissue of striped dolphins, *Stenella coeruleoalba*, and their variations with body length, weight, age and sex. Agric. Biol. Chem., 47(6): 1219–1228.

ICES. 1997. Report of the Working Group on Seals and Small Cetaceans in European Seas. ICES CM 1997/N:1.

IWC. 1995. Report of the Workshop on Chemical Pollution and Cetaceans. Bergen, Norway, 27–29 March 1995. SC/47/Rep2.

IWC. 1997. Proposal to the IWC on furthering the recommendations of the Pollution Workshop. SC/49/Rep6.

[Jacobson, J.L., Jacobson, S.W., and Humphrey, H.E.B. 1990. Effects of *in utero* exposure to polychlorinated biphenyls and related contaminants on cognitive functioning in young children. Journal of Pediatrics, 113: 38–45.]

Jepson, P.D., Bennett, P.M., Allchin, C.R., Baker, J.R., Kuiken, T., Rogan, E., Lockyer, C., Law, R.J., Walton, M.J., and Kirkwood, J.K. 1998. Chronic PCB exposure is associated with infectious disease mortality in harbour porpoises stranded in England and Wales 1990–1996. The World Marine Mammal Science Conference, Monaco, 20–24 January 1998. [ABSTRACT]

Kannan, K., Tanabe, S., Borrell, A., Aguilar, A., Focardi, S., and Tatsukawa, R. 1993. Isomer-specific analysis and toxic evaluation of polychlorinated biphenyls in striped dolphins affected by an epizootic in the western Mediterranean Sea. Archives of Environmental Contamination and Toxicology, 25(2): 227–233.

Kannan, N., Tanabe, S., and Tatsukawa, R. 1988. Toxic potential of non-*ortho* and mono-*ortho* coplanar PCBs in commercial PCB preparations—2,3,7,8-TCDD toxicity equivalence factors approach. Bulletin of Environmental Contamination and Toxicology, 41: 267–276.

[Kannan, N., Tanabe, S., Borrell, A., Focardi, S., and Tatsukawa, R. 1993. Isomer-specific analysis and toxic evaluation of polychlorinated biphenyls in striped dolphins affected by an epizootic in the western Mediterranean Sea. Archives of Environmental and Contamination Toxicology, 25: 227–233.]

Kannan, N., Tatsukawa, R., Tanabe, S., and Ono, M. 1989. Critical evaluation of polychlorinated biphenyl toxicity in terrestrial and marine mammals - increasing impact of non-*ortho* and mono-*ortho* coplanar polychlorinated biphenyls from land to ocean. Archives of Environmental Contamination and Toxicology, 18: 850–857.

[Kawai, S., Fukushima, M., Miyazaki, N. and Tatsukawa, R. 1988. Relationship between lipid composition and organochlorine levels in the tissues of striped dolphin. Marine Pollution Bulletin, 19: 129–133.]

Kawano, M., Inoue, T., Wada, T., Hidaka, H., and Tatsukawa, R. 1988. Bioconcentration and residue patterns of chlordane compounds in marine animals: invertebrates, fish, mammals and seabirds. Environmental Science and Technology, 22: 792–797.

[Kim, G.B., Tanabe, S., Iwakiri, B., Tatsukawa, R., Amano, M., Miyazaki, N., and Tanaka, H. 1996. Accumulation of butyltin compounds in rissos dolphin (*Grampus griseus*) from the pacific coast of Japan. Environmental Science Teqniques, 30: 2620–2625.]

Kingsley, M.C.S. In press. Population index estimates for the St. Lawrence belugas 1973–1995. Marine Mammal Science, 14(3).

[Kleivane, L., Skaare, J.U., Bjorge, A., de Ruiter., E., and Reijnders, P.J.H. 1995. Organochlorine pesticide residue and PCBs in harbour porpoise (*Phocoena phocoena*) incidentally caught in Scandinavian waters. Environmnetal Pollution, 89(2): 137–146.]

Koeman, J.H.Y., Wousha, A., Peters, W.H.M., de Goeij, J.J.M., Zegers, C., and van Haafen, J.J. 1971. Mercury in the Dutch littoral environment. TNO Nieuws, 26: 402–409.

Koistinen, J., Stenman, O., Haahti, H., Suonpera, M., and Paasivirta, J. 1997. Polychlorinated diphenyl ethers, dibenzo-*p*-dioxins, dibenzofurans and biphenyls in seals and sediment from the Gulf of Finland. Chemosphere, 35: 1249–1269.

Kuehl, D.W., Haebler, R., and Potter, C. 1991. Chemical residues in dolphins from the US Atlantic coast including Atlantic bottlenose obtained during the 1987/88 mass mortality. Chemosphere, 22: 1071–1084.

Kuehl, D.W., Haebler, R., and Potter, C. 1994. Coplanar PCB and metal residues in dolphins from the United States Atlantic coast including Atlantic bottlenose obtained during the 1987/8 mass mortality. Chemosphere, 28: 1245–1253.

[Kuiken, T., Bennet, P.M., Allchin, C.R., Kirkwood, J.K., Baker, J.R., Lockyer, C.H., Walton, M.J., and Sheldrick, M.C. 1994. PCBs, cause of death and body condition in harbour porpoises (*Phocoena phocoena*) from British waters. Aquatic Toxicology, 28: 13–28.]

Lahvia, G.P., Wells, R.S., Casper, D., and Via, C.S. 1993. *In-vitro* lymphocyte response of bottlenose dolphins (*Tursiops truncatus*): mitogen-induced proliferation. Marine Environmental Research, 35(1-2): 115–119.

[Lahvis, G.P., Wells, R.S., Kuehl, D.W., Stewart, J.L., Rhinehart, H.L., and Via, C.S. 1995. Decreased lymphocyte responses in free ranging bottlenose dolphins (*Tursiops fruncatus*) are associated with increased concentrations of PCB's and DDT in peripheral blood. Environmental Health Perspectives, 103: 67–72.]

Lake, C.A., Lake, J.L., Haebler, R., McKinney, R., Boothman, W.S., and Sadove, S.S. 1995. Contaminant levels in harbor seals from the Northeastern United States. Archives of Environmental Contamination and Toxicology, 29: 128–134.

Landers, J.P., and Bunce, N.J. 1991. The Ah receptor and the mechanism of dioxin toxicity. Biochemical Journal, 276: 349–353.

[Law, R.J., Allchin, C.R., and Harwood, J. 1989. Concentrations of organochlorine compounds in the blubber of seals from eastern and north-eastern England, 1988. Marine Pollution Bulletin, 20(3): 110–115.]

Law, R.J., and Whinnett, J.A. 1992. Polycyclic aromatic hydrocarbons in muscle tissue of harbour porpoises, *Phocoena phocoena*, from UK waters. Marine Pollution Bulletin, 24: 550–553.

Law, R.J., Fileman, C.F., Hopkins, A.D., Baker, J.R., Harwood, J., Jackson, D.B., Kennedy, S., Martin, A.R., and Morris, R.J. 1991. Concentrations of trace metals in the liver of marine mammals (seals, porpoises and dolphins) from waters around the British Isles. Marine Pollution Bulletin, 22(4): 183–191.

Lemesh, R.A. 1992. Polychlorinated biphenyls: An overview of metabolic, toxicologic and health consequences. Veterinary and Human Toxicology, 34: 256–260.

Lohman, S., Folkow, L.P., Blix, A.S., Olsson, M., and Seljelid, R. 1998. Physiological and immunological effects of PCB-exposure in harp seals (*Phoca groenlandica*). The World Marine Mammal Science Conference, Monaco, 20–24 January 1998. [ABSTRACT]

[Marsili, L., and Focardi, S. 1996. Organochlorine levels in subcutaneous blubber biopsies of fin whales (*Balaenoptera physalus*) and striped dolphins (*Stenella coeruleoalba*) from the Mediterranean Sea.]

Martineau, D., Béland, P., Desjardins, C., and Lagacé, A. 1987. Levels of organochlorine chemicals in tissues of beluga whales (*Delphinapterus leucas*) from the St. Lawrence Estuary, Quebec, Canada. Archives of Environmental Contamination and Toxicology, 16: 137–147.

Martineau, D., Lagacé, A., Massé, R., Morin, M., and Béland, P. 1985. Transitional cell carcinoma of the urinary bladder in a beluga whale (*Delphinapterus leucas*). Canadian Veterinary Journal, 26: 297–302.

Martineau, D., Lair, S., De Guise, S., Lipscomb, T.P., and Béland, P. In press. Cancer in beluga whales from the St. Lawrence estuary, Quebec, Canada: a potential biomarker of environmental contamination. Reports of the International Whaling Commission (Special Issue 17).

McFarland, V.A., and Clarke, J.U. 1989. Environmental occurrence, abundance and potential toxicity of polychlorinated biphenyl congeners: considerations for congener specific analysis. Environmental Health Perspectives, 81: 225–239.

[McKenzie, C., Reid, R.J. and Wells, D.E. 1997. Organochlorine contaminants and trace metals in bottlenose dolphins *(Tursiops truncatus)* stranded in the Moray Firth, Scotland. *In* The Influence of Environmental Contaminants on Skin Disease in Bottlenose Dolphins. Project MF 0516. Final report to the Ministry of Agriculture, Fisheries and Food and the Department of the Environment, Transport and the Regions.]

McKenzie, C., Reid, R.J. and Wells, D.E. 1998. Chlorinated biphenyls and organochlorine pesticides in marine mammals stranded on the coasts of Scotland and Ireland. *FRS Report* No. 2/98, FRS Marine Laboratory, Aberdeen, UK

[Mckenzie, C., Rogan, E., Reid, R.J., and Wells, D.E. 1997. Patterns and concentrations of organic contaminants from Atlantic white-sided dolphins (*Lagenorhynchus acutus*) from Irish and Scottish coastal waters. Environ. Poll., 98: 15–27.]

McKinney, J.D., and Waller, C.L. 1994. Polychlorinated biphenyls as hormonally active structural analogues. Environmental Health Perspectives, 102: 290–297.

Mortensen, P., Bergman, A., Bignert, A., Hansen, H.J., Härkönen, T., and Olsson, M. 1992. Prevalence of skull lesions in harbour seals (*Phoca vitulina*) in Swedish and Danish museum collections: 1835–1988. Ambio, 21(8): 520–524.

[Muir, D.G.C., Norstrom, R.J., and Simon, M. 1988a. Organochlorine contaminants in arctic food chains. Accumulation of specific polychlorinated biphenyls and chlordane related compounds. Environmental Science Techniques, 22: 1071–1079.]

[Muir, D.G.C., Wagemann, R., Grift, N.P., Norstrom, R.J., Simon, M., and Lien, J. 1988b. Organochlorine chemical and heavy metal contaminants in white-beaked dolphins (*Lagenorhynchus albirostris*) and pilot whales (*Globicephala melaena*) from the coast of Newfoundland, Canada. Arch. Environ. Contam. Toxicology, 17: 613–629.]

Nakata, H., Tanabe, S., Tatsukawa, R., Amano, M., Miyazaki, N., and Petrov, E. 1995. Persistent organochlorine residues and their accumulation kinetics in Baikal seal (*Phoca sibirica*) from Lake Baikal, Russia. Environmental Science and Technology, 29: 2877–2885.

Nebert, D.W., Adesnik, M., Coon, M.J., Estabrook, R.W., Gonzalez, F.J., Guengerion, F.P., Gunsalus, I.C., Johnson, E.F., Kemper, B., Levin, W., Phillips, I.R., Sato, R., and Waterman, M.R. 1987. The P450 gene superfamily: recommended nomenclature. DNA and Cell Biology, 6: 1–8.

Nelson, D.R., Kamataki, T., Waxman, D.J., Guengerich, F.P., Estabrook, R.W., Feyereisen, R., Gonzalez, F.J., Coon, M.J., Gunsalus, I.C., Gotoho, O., Okuda, K., and Nebert, D.W. 1993. The P450 superfamily - update on new sequences, gene mapping, accession numbers, early trivial names of enzymes and nomenclature. DNA and Cell Biology, 12: 1–12.

Nishiwaki, M. 1953. Hermaphroditism in a dolphin (*Prodelphinus caeruleoalbus*). Sci. Rep. Whales Res. Inst. Tokyo, 8: 215-218.

Norstrom, R.J., Muir, D.C.G., Ford, C.A., Simon, M., MacDonald, C.R., and Béland, P. 1992. Indications of P450 monooxygenase activities in beluga *Delphinapterus leucas* and narwhal *Monodon monoceros* from patterns of PCB, PCDD and PCDF accumulation. Marine Environmental Research, 34: 267–272.

Oehme, M., Schlabach, M., and Boyd, I. 1995a. Polychlorinated dibenzo-*p*-dioxins, dibenzofurans and coplanar biphenyls in Antarctic fur seal blubber. Ambio, 24: 41–46.

Oehme, M., Schlabach, M., Hummert, K., Luckas, B., and Nordoy, E.S. 1995b Determination of levels of polychlorinated dibenzo-*p*-dioxins, dibenzofurans, biphenyls and pesticides in harp seals from the Greenland Sea. Science of the Total Environment, 162: 75-91.

Olsson, M., Karlsson, B., and Ahnland, E. 1994. Diseases and environmental contaminants in seals from the Baltic and the Swedish west coast. Science of the Total Environment, 154: 217–227.

Poland, A., and Knutson, J.C. 1982. 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin and related halogenated aromatic hydrocarbons: examinations of the mechanism of toxicity. Annual Review of Pharmacology and Toxicology, 22: 517–520.

Poland, A.P., and Glover, E. 1986. 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin: segregation of toxicity with the Ah locus. Molecular Pharmacology, 17: 86.

Pomeroy, P.P., Green, N., Hall, A.J., Walton, M., Jones, K., and Harwood, J. 1996. Congener specific exposure of grey seal (*Halichoerus grypus*) pups to chlorinated biphenyls during lactation. Canadian Journal of Fisheries and Aquatic Sciences, 53: 1526–1534.

Rawson, A.J., Patton, G.W., Hofmann, S., Pietra, G.G., and Johns, L. 1993. Liver abnormalities associated with chronic mercury accumulation in stranded Atlantic bottlenose dolphins. Ecotoxicology and Environmental Safety, 25: 41–47.

Reijnders, P.J.H. 1986. Reproductive failure in common seals feeding on fish from polluted coastal waters. Nature, 324: 456–457.

Reijnders, P.J.H. 1988. Environmental impact of PCBs in the marine environment. In: Environmental protection of the North Sea, Newman, P.J.H., and Agg, A.R. (eds). Heinemann, pp85-98.

Reijnders, P.J.H. 1994. Contaminants and cetaceans: Reasons for concern? IWC SC/46/O8.

Ross, P.S., De Swart, R.L., Reijnders, P.J.H., van Loveren, H., Vos, J.G., and Osterhaus, A.D.M.E. 1995. Contaminant related suppression of delayed-type hypersensitivity and antibody responses in harbour seals fed herring from the Baltic Sea. Environmental Health Perspectives, 103(2): 162–167.

Ross, P.S., De Swart, R.L., Timmerman, H.H., Reijnders, P.J.H., Vos, J.G., van Loveren, H., and Osterhaus, A.D.M.E. 1996. Suppression of natural killer cell activity in harbour seals (*Phoca vitulina*) fed Baltic Sea herring. Aquatic Toxicology, 34: 71–84.

[Ross, P.S., de Swart, R.L., Timmerman, H.H., Reijnders, P.J.H., Vos, J.G., van Loveren, H., and Osterhaus, A.D.M.E. 1996. Suppression of natural killer cell activity in harbour seals (*Phoca vitulina*) fed Baltic Sea herring. Aquatic Toxicology, 34 71–84.]

Safe, S. 1990. Polychlorinated biphenyls (PCBs), dibenzo-*p*-dioxins (PCDDs), dibenzofurans (PCDFs) and related compounds: environmental and mechanistic considerations which support the development of toxic equivalency factors (TEFs). Critical Reviews in Toxicology, 21: 51–88.

[Salata, G.G., Wade, T.A., Sericano, J.L., Davis, J.W., and Brooks, J.M. 1995. Analysis of Gulf of Mexico bottlenose dolphins for organochlorine pesticides and PCBs. Environmental Pollution, 88: 167–175.]

Schulz, D.E., Petrick, G., Duinker, J.C. 1989. Complete characterisation of PCB congeners in commercial Aroclor and Clophen mixtures by MDGC/ECD. Environ. Sci. Technol., 23, 852-859.

Schumacher, Z., Zahler, S., Horny, H.-P., Heidemann, G., Skrinisson, K., and Welsch, U. 1993. Histological investigation on the thyroid glands of marine mammals (*Phoca vitulina, Phocoena phocoena*) and the possible implications of marine pollution. Journal of Wildlife Diseases, 29: 103–108.

Siebert, 1995. Analyse des Quecksilbergehaltes in Organen von Kleinwalen aus deutschen Gewässern der Nord- und Ostsee und seine möglichen Beziehungen zu Erkrankungs- und Todesursachen. Thesis, University of Giessen, Germany, 1995.

Skaare, J.U., Jensen, E.G., Goksøyr, A., and Egaas, E. 1991. Response of xenobiotic metabolising enzymes of rainbow trout *Oncorhynchus mykiss* to the mono-*ortho* substituted polychlorinated PCB congener 2,3′,4,4′,5-pentachlorobiphenyl, PCB–118, detected by enzyme activities and immunochemical methods. Archives of Environmental Contamination and Toxicology, 20: 349–352.

Stede, G., and Stede, M. 1990. Orientierende Untersuchungen von Seehundschädeln auf pathologische Knochenveränderungen. *In* Zoologischen und ethologische Untersuchungen zum Robbensterben, pp. 31–53. Institut für Haustierkunde, Kiel, Germany.

Stegeman, J.J., and Hahn, M.E. 1994. Biochemistry and molecular biology of monooxygenases: Current perspectives on forms, functions, and regulation of cytochrome P450 in aquatic species. *In* Aquatic toxicology: molecular, biochemical and cellular aspects, pp. 87–206. Ed. by D.C. Malins and G.K. Ostrander. Lewis Publishers, Boca Raton, FL, USA.

[Stein, J.E., Tilbury, K.L., Brown, D.W., Wigren, C.A., Meador, J.P., Robisch, P.A., Chan, S., and Varanasi, U. 1992. Intraorgan distribution of chemical contaminants in tissues of harbour porpoises *(Phocoena phoceona)* from the northwest Atlantic. National Marine Fisheries Service. Technical memorandum NMFS-NWFSC-3.]

[Subramanian, A.N., Tanabe, S., and Tatsukawa, R. 1987. Use of organochlorines as chemical tracers in determining some reproductive parameters in Dalli-type Dall's porpoise (*Phocoenides dalli*). Mar. Environ. Res., 25: 161–174.]

Subramanian, A.N., Tanabe, S., Tatsukawa, R., Saito, S., and Miyazaki, N. 1987b. Reduction in the testosterone levels by PCBs and DDE in Dall's porpoises of the northwestern North Pacific. Marine Pollution Bulletin, 18 (12): 643–646.

Tanabe, S., and Tatsukawa, R. 1991. Persistent organochlorines in marine mammals. *In* Organic contaminants in the environment, pp. 276–289. Ed. by K.C. Jones. Elsevier Applied Science, London.

Tanabe, S., Iwata, H., and Tatsukawa, R. 1994. Global contamination by persistent organochlorines and their ecotoxicological impact on marine mammals. Science of the Total Environment, 154: 163–177.

Tanabe, S., Kannan, N., Subramanian, A., Watanabe, S., and Tatsukawa, R. 1987. Highly toxic coplanar PCBs: occurrence, source, persistency and toxic implications to wildlife and humans. Environmental Pollution, 47: 147–163.

Tanabe, S., Madhusree, B., Ozturk, A.A., Tatsukawa, R., Miyazaki, N., Ozdamar, E., Aral, O., Samsun, O., and Ozturk, B. 1997. Isomer-specific analysis of polychlorinated biphenyls in harbour porpoise (*Phocoena phocoena*) from the Black Sea. Marine Pollution Bulletin, 34: 712–720.

Tanabe, S., Madhusree, B., Ozturk, A.A., Tatsukawa, R., Miyazaki, N., Ozdamar, E., Aral. O. and Samsun, O. 1997. Persistant organochlorine residues in harbour porpoise (*Phocoena phocoena*) from the Black Sea. *Mar. Poll. Bull.*, 34, 338-347.

[Tanabe, S., Subramanian, A.N., Ramesh, A., Kumaran, P.L., Miyazaki, N., and Tatsukawa, R. 1993. Persistant organochlorine residues in dolphins from the Bay of Bengal, south India. Marine Pollution Bulletin, 26(6): 311–316.]

[Tanabe, S., Tatsukawa, R., Tanaka, H., Maruyama, K., Miyazaki, N., and Fujiyama, T. 1981. Distribution and total burdens of chlorinated hydrocarbons in bodies of striped dolphins (*Stenella coereoalba*). Agric. Biol. Chem., 45: 2569–2578.]

Tanabe, S., Watanabe, S., Hiroshi, K., and Tatsukawa, R. 1988. Capacity and mode of PCB metabolism in small cetaceans. Marine Mammal Science, 4: 103–124.

Timbrell, J.A. 1991. Principles of biochemical toxicology. Taylor and Francis, London.

[van Scheppeningen, W.B.,Verhoeven, A.J.I.M., Mulder, P., Addink, M.J. and Smeenk, C. 1996. Polychlorinated biphenyls, dibenzo-*p*-dioxins and dibenzofurans in harbour porpoise (*Phocoena phocoena*) stranded on the Dutch coast between 1990 and 1993. Archives of Contaminant and Environmental Toxicology, 30: 492–502.]

[Varanasi, U., Stein, J.E., Tilbury, K.L., Brown, D.W., Meador, J.P., Krahn, M.M., and Chan, S-L. 1993. Contaminant monitoring for NMFS Marine mammal Health and Stranding Response Programme. *In* Proceedings of the 8th Symposium on Coastal and Ocean Management. Coastal Zone 1993. Eds. Magoon, O.T., Wilson, W.S., Converse, H., Tobin, L.T. American Society of Civil Engineers, New York, 2517–2530.]

Wagemann, R., and Muir, D.C.G. 1984. Concentrations of heavy metals and organochlorines in marine mammals of northern waters: overview and evaluation. Can. Tech. Rep. Fish. Aquat. Sci., 1279.

Wagemann, R., Stewart, R.E.A., Béland, P., and Desjardins, C. 1990. Heavy metals and selenium in tissues of beluga whales, *Delphinapterus leucas*, from the Canadian Arctic and the St. Lawrence esturary. Canadian Bulletin of Fisheries and Aquatic Sciences, 224: 191–206.

Walker, C.H. 1975. Variations in the intake and elimination of pollution. *In* Organochlorine insecticides: persistent organic pollutants, pp. 73–130. Ed. by F. Moriarty. Academic Press, New York.

[Wells, D.E., and Campbell, L.A. 1990. Organochlorine residues in blubber and milk of seals from Scottish coastal waters, 1988–1989. Report of the chemical analysis prepared for the Sea Mammal Research Unit, Cambridge. Marine Laboratory, Aberdeen.]

[Wells, D.E., and Echarri, I. 1992. Determination of individual chlorobiphenyls (CBs), including non-*ortho*, and mono-*ortho* substituted CBs in marine mammals from Scottish waters. Int. J. Environ. Anal. Chem., 47: 75–97.]

[Wells, D.E., Campbell, L.A., Ross, H.M., Thompson, P.M., and Lockyer, C.H. 1994. Organochlorine residues in harbour porpoise and bottlenose dolphins stranded on the coast of Scotland, 1988–1991. Science of the Total Environment, 151:77–99.]

[Wells, D.E., Mckenzie, C., and Ross, H. 1996. Chlorobiphenyl patterns in marine mammals from northern European waters. Scottish Fisheries Working Paper No. 1/96. Marine Laboratory, Aberdeen.]

White, R.D., Hahn, M.E., Lockhart, W.L. and Stegeman, J.J. 1994. Catalytic and immunochemical characterisation of hepatic microsomal cytochromes P450 in Beluga whale (Delphinapterus leacas). *Toxicol. Appl. Pharmacol.*, 126, 45-57.

White, R.D., Hahn, M.E., Lockhart, W.L., and Stegeman, J.J. 1994. Catalytic and immunochemical characterization of hepatic microsomal cytochromes P450 in beluga whale (*Delphinapterus leucas*). Toxicology and Applied Pharmacology, 126: 45–57.

Whitlock, J.P. 1987. The regulation of gene-expression by 2,3,7,8-tetrachlorodibenzo-*para*-dioxin. Pharmacological Reviews, 39:147–161.

Zakharov, V.M., and Yablokov, A.V. 1990. Skull asymmetry in the Baltic grey seal: Effects of environmental pollution. Ambio, 19: 266–169.

INTERNATIONAL COUNCIL FOR THE EXPLORATION OF THE SEA
CONSEIL INTERNATIONAL POUR L'EXPLORATION DE LA MER

General Secretary: Professor Christopher C. E. Hopkins

ICES Working Group on Biological Effects of Contaminants

att Peter Matthissen

Our Ref: P1

27 March 1998

Dear Peter,

As a part of the data package on CBs in marine mammals from WGEAMS, you will find extracts of data from the ICES Environmental Data Centre. These data have been submitted to the Data Centre by

- National Environmental Research Institute/Arctic Department, Denmark (GERI)
- Marine Research Institute, Iceland (MRII)
- Veterinary Institute, Oslo, Norway (VETN)
- Norwegian Polar Institute (NPIN)

The Data Centre can only provide data to third parties, if agreed by the data originators. Given the short notice for the request, I have only managed to get positive reply from GERI and MRII. I hope to be able to contact VETN and NPIN next week.

The implication for the work of your group is that there is a risk (although minimal) that data from VETN and NPIN must be taken out of the final data set.

I will contact you on Monday. I will communicate with you through Jean Piuze's fax number and email address. Please let me know, if there are more direct channels.

Regards,

Jan Rene Larsen
Environmental Data Scientist