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Ecotoxicological Assessment Criteria for Trace Metals and Organic Microcontaminants in the North-East Atlantic

Contents

Preface

This document presents the Report of the OSPAR workshop ‘Assessment Criteria for Chemical Data of the Joint Monitoring Programme (JMP)’ held in Scheveningen (the Netherlands), 15-17 November 1993.

In the past the Joint Monitoring Group (JMG) had recognised the need to establish assessment criteria in order to improve the assessment of its chemical monitoring results. The aim of the workshop therefore was to establish ecotoxicological assessment criteria for chemical monitoring data from the North-East Atlantic Ocean, i.e. concentration levels below which no harm to the marine environment is expected.

The workshop reached agreement on ecotoxicological assessment criteria for seventeen compounds as best estimates, based on current knowledge. In view of the uncertainties involved in the derivation, a distinction was made between firm and provisional assessment criteria, which were presented as a range rather than a fixed number. As most of the available ecotoxicological data were for water most of the assessment criteria for water were considered firm. In contrast, sediment and biota data were limited or lacking and most of the available assessment criteria were considered provisional.

In applying the proposed ecotoxicological assessment criteria to specific field situations some of the assessment criteria for natural compounds may be lower than their background concentrations and should therefore be adjusted. However, because background concentrations vary from place to place and because an international consensus on background concentrations of natural compounds has not yet been reached, this was not done in the present report.

The Assessment and Monitoring Committee (ASMO) recommends the use of the ecotoxicological assessment criteria (after adjustment where appropriate, as outlined in the previous paragraph) for assessing monitoring data in future. The criteria are considered useful for preliminary assessments and should be considered as guidance for further work.

Caution, however, should always be exercised in using generic, particularly provisional, assessment criteria in specific situations. Their use does not preclude the use of common sense and expert judgement with regard to the natural concentrations of e.g. trace metals in assessing the (potential for) environmental effects. The assessment criteria should not be used as a trigger for source directed action without further evaluation.

Summary

In the strategy document of the Joint Monitoring Group concerning the future monitoring programme (Annex 5 of the summary record of the JMG meeting 1993; JMG 18/13/1) it was pointed out that the development of criteria for the assessment of monitoring data was considered to be essential for the success of the new monitoring programme. This document summarises the methodology and the outcome of the OSPAR workshop 'Assessment Criteria for Chemical Data of the Joint Monitoring Programme (JMP)', held in Scheveningen (the Netherlands), 15-17 November 1993.

In line with the aims of the new Convention for the Protection of the Marine Environment of the North-East Atlantic (Paris, 1992) "to reduce discharges and emissions of substances which are toxic, persistent and liable to bioaccumulate ......to levels that are not harmful to man or nature...", the purpose of the workshop was the establishment of ecotoxicological assessment criteria for chemical monitoring data of the North-East Atlantic Ocean i.e. concentration levels below which no harm to the marine environment is expected.

During the workshop the following compounds were considered: arsenic, cadmium, copper, chromium, lead, mercury, nickel, zinc, PAHs, PCBs, lindane, DDE, dieldrin and tributyltin (TBT). In principle, the goal of the workshop was to establish assessment criteria for all the matrices (water and/or sediment and/or biota (fish and mussel)) relevant for monitoring the compound under consideration.

The following rationale was adopted to derive ecotoxicological assessment criteria. Toxicity data for the different compounds that were available to the participants during the workshop were inventorised and evaluated. Generally accepted extrapolation rules were applied. According to these guidelines, the available toxicological data were extrapolated using an application factor of 10, 100 or 1000 depending on the (phylogenetic) coverage of the species and the amount of data available. Finally, the extrapolated concentration was "rounded" to a range, representing the nearest order of magnitude. The purpose of presenting a range rather than a single value was to express the uncertainty and variability of the data. This is in accordance with EC guidelines.

The workshop participants thought it appropriate to distinguish between 'firm' and 'provisional' values. Where a sufficient amount of marine toxicity data were available, the ecotoxicological guideline values were regarded as 'firm', in other cases they were regarded as 'provisional'. Table 1 gives an overview of the established ecotoxicological assessment criteria.

Sufficient aquatic toxicity data were available to derive firm ecotoxicological assessment criteria for trace metals and TBT in sea water. For PAHs and lindane provisional values were established because these were based mainly on freshwater data.

As the quantity and quality of spiked-sediment toxicity data is still relatively limited, the ecotoxicological assessment criteria for whole sediments were all considered to be provisional. The ecotoxicological assessment criteria that were derived are partly based on the North American Biological Effects Database for Sediment (BEDS). In some cases the Equilibrium Partitioning Approach was used to convert aquatic toxicity values to sediment values. For trace metals the values are close to the background level.

Table 1. Overview of ecotoxicological assessment criteria for trace metals, PCBs, PAHs, TBT and some organochlorine pesticides for the purpose of assessing monitoring data from the North-East Atlantic Ocean within the framework of the JMP. Concentrations in water refer to dissolved concentrations; in sediment to total sediment; in fish to muscle tissue, and in mussel to soft tissue.

These assessment criteria have no legal significance and should be used for preliminary assessments and as a guide for further work.

	WATER	SEDIMENT	FISH	MUSSEL
	µg/l	mg/kg dw (1% organic carbon)	mg/kg ww	mg/kg dw
Trace Metals				
As	1-10	1-10***	n.r.	n.r.
Cd	0,1-1	0,5-5*	f.c.	f.c.
Cr	1-10	5-50***	n.r.	n.r
Cu	0,1-1	5-50**	f.c.	f.c.
Hg	0,001-0,01	0,05-0,5**	0,05-0,5	f.c.
Ni	0,1-1	f.c.	n.r.	n.r.
Pb	0,1-1	5-50**	f.c.	f.c.
Zn	0,5-5	10-100***	n.r.	n.r.
Organochlorine pesticides				
DDE	n.r.	0,0005-0,005*	0,05-0,5*	f.c.
dieldrin	n.r.	0,005-0,05*	0,05-0,5	f.c.
lindane	0,005-0,05*	n.r.	f.c.	n.r.
PAHs				
naphthalene	1-10*	0,01-0,1*	n.r.	f.c.
anthracene	0,005-0,05*	0,001-0,01*	n.r.	f.c.
fluoranthene	0,05-0,5*	0,01-0,1*	n.r.	f.c.
benzo(a)pyrene	0,01-0,1*	0,05-0,5*	n.r.	f.c.
Σ CB$_7$	n.r.	0,001-0,01*	0,001-0,01	f.c.
TBT	0,0001-0,001	0,0001-0,001*	n.r.	0,05-0,5

* : provisional;
** : provisional, with lower boundary below background level;
*** : provisional, with lower and upper boundary below background level;
n.r. : not relevant in connection with current monitoring programme;
f.c. : future consideration (assessment criteria to be developed).

Toxicity data related to tissue residues are very scarce, and therefore it was difficult to come up with relevant toxicity data for biota that were directly applicable. Assessment criteria for fish were derived for compounds with a (strong) tendency to bioaccumulate (mercury, DDE, dieldrin, and PCBs) using an algorithm for the risk assessment of secondary poisoning of birds and mammals, based on the bioconcentration factors (BCFs) for fish and mussel and the No Observed Effect Concentrations (NOECs) for birds and mammals.

For the application of the ecotoxicological assessment criteria, the following guidance was given during the workshop. Levels below the range were regarded to be in a “green zone”, meaning that no harm to the marine environment was to be expected. Levels above the upper limit were regarded to be in a “red zone”, meaning that harmful effects were to be expected with the incidence and severity of adverse biological effects increasing with increasing contaminant concentration. The range itself was considered to represent a “grey zone”, which meant that the possibility of harmful effects could not be excluded. Comparisons with both national quality criteria for the marine environment and JMG monitoring data were carried out, to see if the ecotoxicological assessment criteria derived by the workshop were realistic. From this exercise it appears likely that the ecotoxicological assessment criteria can be an effective tool for assessing monitoring data from the North-East Atlantic Ocean to indicate areas of concern.

The workshop formulated the following recommendations :

Recommendations with respect to the JMP

1 The ecotoxicological assessment criteria should be used in future assessments of monitoring data.

2 The list of assessment criteria needs to be further refined and updated in order to establish a wider list of assessment criteria or to ensure consistency between values for different environmental compartments.

3 All ecotoxicological assessment criteria which are considered provisional should be revised as soon as the availability of new toxicological data makes this possible, in order to establish firm values.

4 Caution should always be exercised in using generic, particularly provisional, assessment criteria in specific situations. Their use does not preclude the use of common sense and expert judgement with regard to the natural concentrations of e.g. trace metals in assessing the (potential for) environmental effects.

General recommendations

5 In order to support the derivation of more reliable assessment criteria for marine monitoring data from the North-East Atlantic Ocean more toxicological (especially marine) data are required, covering a range of microcontaminants, species and conditions.

6 In order to support the process of developing and updating ecotoxicological assessment criteria for the marine environment it would be beneficial to determine which of the existing computerised toxicological databases, or combinations thereof, would be most useful.

For aquatic toxicity data the US-EPA “AQUIRE” database is an example of what is needed, although essential information on experimental conditions, data reliability, ecology of the species etc. is missing. An example of a database which does provide this kind of information is MARITOX, currently being compiled by the Dutch TNO. For marine sediment toxicity probably the most advanced database is BEDS. At present, BEDS is based largely on matching empirical data on effects with field concentrations, since very few spiked-sediment tests have been conducted. When available, spiked-sediment tests (preferably based on chronic endpoints) can be used in conjunction with field toxicity data to derive reliable sediment quality guidelines. BEDS is designed to be updated periodically and it should be investigated to determine the feasibility of developing a similar database for Contracting Parties to the Oslo and Paris Conventions. Furthermore, a database should be developed to compile data on biological effects and concentrations in biota as observed in the laboratory, mesocosm and in the field. This line of work based on tissue residues is considered to be promising and will provide additional information to establish more directly ecotoxicological assessment criteria for chemicals in biota.

7 The Equilibrium Partitioning Approach appears to be an acceptable method for deriving assessment criteria for organic contaminants with log Kow < 5 in sediments from aquatic toxicity values. As regards organic contaminants with log Kow > 5 for which sediment toxicity data are missing, the Equilibrium Partitioning Approach can only be used to derive provisional assessment criteria at present.

8 Besides the general need for sediment toxicity data there is an urgent need for the development of standardised whole sediment bioassays.

9 The algorithm for the risk assessment of secondary poisoning was agreed upon by the workshop as an important approach for deriving assessment criteria for biota which aim to protect top predators such as fish-eating birds and mammals. More research of this type should be carried out, especially for lipophilic non-biodegradable contaminants such as PCBs.

10 To assess the ecotoxicological impact of PCB contamination in the marine environment planar (dioxin-like) PCB congeners should be monitored.

Introduction

Under the auspices of the Oslo and Paris Commissions a Joint Monitoring Programme (JMP) has been conducted by the Contracting Parties to the Oslo and Paris Conventions since 1979 to continuously review the pollution of the marine environment of the North-East Atlantic Ocean (Figure 1) and to study the effectiveness of the measures taken to improve the quality of the seas. To facilitate the evaluation of data on toxic substances in the marine environment, there is a clear need for ecotoxicological assessment criteria. These values should provide the basis for identification of areas of special concern.

The JMP has the objectives of assessing:

a. possible hazards to human health, i.e. estimating the levels of pollutants in edible (shell)fish;

b. harm to living resources and marine life, i.e. biological and biological-effect monitoring;

c. existing levels of marine pollution, i.e. determining the spatial distribution of pollution;

d. the effectiveness of measures taken for the reduction of marine pollution in the framework of the Oslo and Paris Conventions, i.e. temporal trend monitoring.

Over the years several assessments of the data on concentrations of contaminants in water, sediment or biota have been carried out to assess temporal trends or spatial distributions. For the purpose of reviewing the data from baseline studies, arbitrary ‘divisions’ were established often based on the numerical distribution of the data (e.g. quartiles or so called lower and upper levels). These ‘divisions’ were established purely for descriptive purposes, and were not intended to imply which levels should or should not be considered acceptable with respect to potential toxicity or bioaccumulation of a contaminant.

In September 1992 a new Convention for the Protection of the Marine Environment of the North-East Atlantic was signed at a Ministerial meeting in Paris. The Ministerial meeting stated that as a matter of principle, for the whole maritime area, “discharges and emissions of substances which are toxic, persistent and liable to bioaccumulate, in particular organohalogen substances, and which could reach the marine environment should, regardless of their anthropogenic source, be reduced by the year 2000 to levels that are not harmful to man or nature, with the aim of their elimination”.

The new Convention has consequences for the JMP. Ecotoxicological assessment criteria are considered to be an essential component of the success of implementation of the new monitoring programme, as has been pointed out in the strategy document of the Joint Monitoring Group (JMG) concerning the future monitoring programme (Annex 5 of the summary record of the JMG meeting 1993; JMG 18/13/1). To fulfil this need, the Dutch delegation to the JMG offered to organise a workshop in order to establish assessment criteria for a number of microcontaminants included in the spatial distribution and temporal trend monitoring programmes of the JMP, for use in future assessments. This document summarises the methodology and the outcome of the workshop, held 15-17 November 1993 in Scheveningen, the Netherlands. The participants are listed in Annex I.

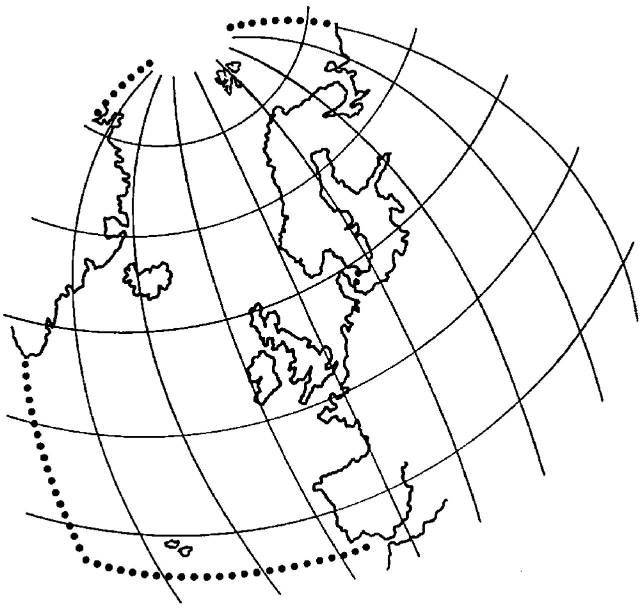

Figure 1. Maritime area of the Oslo and Paris Conventions

In line with the aims of the new Convention, the purpose of the workshop was to establish ecotoxicological assessment criteria for selected microcontaminants within the North-East Atlantic Ocean that represent ranges in concentration that are not harmful to the marine environment. The elements and compounds considered include: arsenic, cadmium, copper, chromium, lead, nickel, mercury, zinc, PAHs, PCBs, lindane, DDE, dieldrin and TBT. Annex II gives an overview of the matrices recommended by the JMG for sampling the contaminants listed above. With regard to the human consumption of marine fish and shellfish, an overview of standards had been prepared but these were not discussed as this topic fell outside the scope of the workshop.

The ecotoxicological assessment criteria established during the workshop were evaluated for their practicality based on a preliminary comparison with a selected set of monitoring data from the JMP.

Rationale followed during the workshop

General outline

General

This section describes the approach that was followed, especially with regard to:

- data requirements;
- the extrapolation of the toxicity data;
- the setting of ecotoxicological assessment criteria.

Two concepts that are used throughout this text are defined as follows:

- ‘extrapolated concentration’: the concentration which is derived by the division of a ‘safe or acceptable’ concentration, i.e. the most sensitive and relevant endpoint (being an NOEC or $E(L)C_{50}$ for growth, reproductive success or mortality rates) by an appropriate application factor;

- ‘ecotoxicological assessment criterion’: the order of magnitude concentration range set around the extrapolated concentration.

An overview of the approach followed during the workshop is shown in Table 2. Although the methodology described below was adopted as a general starting point, some ecotoxicological assessment criteria were derived using a slightly different approach. This is explained in the appropriate section.

Table 2. Scheme followed during the workshop to derive (‘firm’ or ‘provisional’) ecotoxicological assessment criteria for use in the JMP.

Step 1	**Evaluation of national guideline values for the marine environment** - variation in concentrations > factor of 3; go to Step 2 - variation in concentrations < factor of 3, calculate the mean; go to Step 3
Step 2	**Evaluation of available toxicity data** - Minimum data requirements (see Table 3) met ? YES: use an application factor (see Table 4) to derive a **firm** ‘extrapolated concentration’; go to Step 3. NO: evaluate freshwater toxicity data, values based on the Equilibrium Partitioning Approach etc. If sufficient information is available, use an application factor (see Table 4) to derive a **provisional** ‘extrapolated concentration’; go to Step 3
Step 3	**Setting the ecotoxicological assessment criterion** The ‘extrapolated concentration’ (or average guideline concentration) is rounded to the nearest order of magnitude interval to generate the ecotoxicological criterion.

Data requirements

Firstly (Step 1), national guidelines or criteria were evaluated for their applicability (i.e. whether they represented concentrations that are not harmful to the marine environment). The overview of quality guidelines presented in MacDonald *et al.* (1992) was used, as well as the papers summarising national guidelines prepared by P. Matthiessen (United Kingdom), D. Jonkers (the Netherlands), K. Hylland (Norway) and S. Smith (Canada). Draft sediment quality assessment values for the State of Florida were also presented by S. Smith[1]. These national approaches are summarised briefly at pages 14-18. With the exception of the EC water quality guidelines none of the other Contracting Parties have developed guidelines for their marine environment. Where there was little difference in the national assessment criteria (a variation of less than a factor of 3) an average concentration was calculated. However, this criterion was seldom met. In fact, the available guidelines were mainly used to make an *a posteriori* comparison with the ecotoxicological assessment criteria that were set by the workshop.

13 This information is currently being used in Canada to develop national sediment quality guidelines. Guidelines set by the State of Florida have been used as case examples throughout this document, and should not be construed as Canadian guidelines.

Secondly (Step 2), toxicity data for water, sediment and biota available during the workshop (i.e. the papers mentioned above and reports and books made available by participants) were evaluated. Minimum data requirements were established by workshop participants as described in Table 3. These data requirements had to be met in order to establish a 'firm' assessment criterion. Otherwise, the values were designated as 'provisional'.

Table 3. Minimum data requirements for the development of 'firm' ecotoxicological assessment criteria.

WATER	SEDIMENT	BIOTA
chronic toxicity data, or acute toxicity data if insufficient chronic toxicity data are available		
1 fish species 1 algal species 1 crustacean species (or other sensitive invertebrate)	log Kow > 5: 1 sensitive sediment-feeder log Kow < 5: Equilibrium Partitioning Approach	log Kow > 5: 1 mammal species 1 bird species log Kow < 5: BCF
bioconcentration		
-	-	BCF for: 1 fish species 1 mollusc species (for log Kow < 5)

At the workshop it was stated that:

- Marine data must be used to derive 'firm' assessment criteria. If insufficient marine data are available, they can be supplemented with freshwater data, but the resulting assessment criteria are defined as 'provisional'.
- For sediments and biota, the Equilibrium Partitioning Approach and BCFs can be used for organics with log Kow < 5, respectively.
- The data must have been produced by scientifically sound methods.
- If mesocosm or field experimental data are available, they should be compared with the predicted assessment criterion. However, the absence of such data would not downgrade the guideline to 'provisional'.

Extrapolation and setting of the ecotoxicological assessment criteria

From the available toxicological information, the concentration of a compound that was regarded as the most sensitive endpoint was divided by an application factor. Generally accepted application factors (10, 100, 1000) were applied to the lower end of the $L(E)C_{50}$ or NOEC data, depending on the (phylogenetic) coverage of species and according to the amount of data available (see Table 4). The concentrations derived in this way were denoted as 'extrapolated concentrations'.

Finally (Step 3), the extrapolated concentration was assigned to an order of magnitude-interval. In the light of all uncertainties within the process of setting the assessment criterion this rounding seemed more appropriate than setting a fixed concentration. The workshop decided to use one order of magnitude intervals, for instance 0,01-0,1, 0,1-1 etc. and also intermediate intervals of 0,05-0,5, 0,5-5 etc., which depended on the extrapolated concentration that was derived for each microcontaminant. For example, if from sufficient data a lowest NOEC of 20 µg/l was selected and divided by the appropriate factor of 10, the extrapolated concentration of 2 µg/l was rounded to the 'ecotoxicological assessment criterion' of 0,5-5 µg/l. The assignment to an order of magnitude interval is in accordance with the EC approach which was presented at the workshop by Professor G. Persoone, Vice President of the Scientific Advisory Committee on Toxicity and Ecotoxicity of Chemicals (CSTE) of the European Commission.

Table 4. Application factors applied to the lower bound of toxicity data to derive an 'extrapolated concentration' (*c.f.* Bro-Rasmussen *et al.*, in press).

Application Factor	
1000	Applied to the lower end of the range of acute $L(E)C_{50}$s when the data available are few, or the range of organisms is narrow, bearing in mind that outlier values may be due to error or experimental conditions which deviate too much from real environmental conditions.
100	Applied to the lower end of the range of acute $L(E)C_{50}$s when there is an extensive data base covering a (phylogenetically) wide range of test species, or to the lower end of the range of chronic $L(E)C_{50}$s or NOECs when few data are available.
10	Applied to the lower end of the range of (apparent) chronic NOEC data determined by a sufficient and representative number of tests.

Remarks with regard to the matrices

The approach outlined above was followed straightforwardly for water. The ecotoxicological assessment criteria refer to dissolved concentrations in sea water.

For sediments ecotoxicological assessment criteria were derived for whole sediments, expressed on a dry weight (dw) basis. No attempts were made during the workshop to standardise the concentrations. The ecotoxicological assessment criteria are for sediments with a 1% organic carbon content which is representative of the majority of sediments within the maritime area. The Equilibrium Partitioning Approach was used to derive assessment criteria for organic substances in sediments. The extrapolated concentration for aquatic toxicity was multiplied by a sediment/water partition coefficient, Kd, for sediments with 1% organic carbon. If not available from field studies or laboratory measurements, Kd's (l/Kg dw) for organic micropollutants were calculated according to the following equation:

$$Kd = 0{,}5 \bullet K_{ow} \bullet f_{oc}$$

where: K_{ow} = 1/octanol-water partition coefficient;

f_{oc} = organic carbon fraction in the sediment (0,01).

In cases where No Observed Effect Levels (NOELs) from BEDS (see Canada page 17) were used to derive an ecotoxicological assessment criterion, no application factor was used. Ecotoxicological assessment criteria for sediments with organic microcontaminants of log Kow > 5 derived by application of the Equilibrium Partitioning Approach were, by definition of the workshop participants, considered provisional. For trace metals the Equilibrium Partitioning Approach was considered inappropriate due to the large variability of reported partition coefficients and therefore was not used in deriving ecotoxicological assessment criteria for sediments.

With regard to ecotoxicological assessment criteria for biota, for those substances which have a tendency to biomagnify through the food chain, secondary poisoning was taken into account. For fish, assessment criteria were derived for muscle tissue, expressed on a wet weight (ww) basis. For mussels, the values were derived for soft tissue on a dw basis. If necessary, concentrations of tissue residues were transformed from ww to dw by assuming a dry to fresh weight ratio of 1:5 for both fish and shellfish.

Summary of national approaches

Prior to the workshop, papers were prepared by various participants that covered a spectrum of approaches describing national quality criteria or guidelines. These are summarised in this section. The guidelines presented by the United Kingdom can be referred to as guidelines for setting effluent discharge consents. The other guidelines are based on: 'anthropogenic' background levels (Norway); a weight of evidence approach (Canada); and extrapolation methods within an environmental risk-assessment framework (the Netherlands). Initially a short description is given of the method applied by the EC for setting water quality objectives (Bro-Rasmussen *et al.*, in press).

Setting water quality objectives in the European Community

In setting quality objectives for fresh and marine waters within the EC (Bro-Rasmussen *et al.*, in press), the lower end of selected $L(E)C_{50}$ or NOEC toxicity data is established and divided by commonly accepted application factors (see Table 4). The resulting concentration is called the 'extrapolated figure'. Taking into account the experimental uncertainties and the variables with respect to test conditions, the extrapolated concentration is subsequently rounded to the nearest order of magnitude. It should be clear however that the establishment of a quality objective is not merely a mathematical exercise, but always requires case-to-case consideration which calls for expert judgement. Indeed, due account must be taken not only of the quantity but also the quality of the available data. In practice, it is often a restricted toxicity data set, both in terms of the endpoints and the taxonomic coverage which is available. If justified in specific cases a more stringent extrapolation may be used. Also, in setting water quality objectives, attention must to be paid to any factors which can, directly or indirectly, contribute to the hazards of the compound, such as persistence, bioaccumulation potential, carcinogenic and mutagenic properties and, in specific cases, observations of avoidance reactions and sublethal effects on populations and communities.

Norway

In Norway a classification scheme for marine waters, sediments and biota is under development based on 'presumed high background levels within areas of diffuse loading only' (Knutzen, 1992). This is a rough estimate of the 'upper limit' of the concentration interval which is derived from measurements at sites remote from the influence of point sources. As an additional reference a consideration of 'natural background levels' is suggested. However, because for xenobiotic substances the latter values are zero and for most metals and PAHs concentrations are strongly related to human activities, the 'natural background approach' is not considered very useful (Knutzen, 1992). The measurements and other information forming the basis for the presumed high background levels have varying degrees of completeness and reliability. The basic material is derived from observations both in Norway and other countries over the period 1985-1989. Local and regional variations in natural concentrations of metals in water and the various degrees of diffuse loading, of mainly airborne metals and organic micropollutants, may warrant specific local/regional classifications.

United Kingdom

In the United Kingdom environmental quality criteria (Environmental Quality Standards - EQSs) are intended to provide a guide for setting effluent discharge consents, including marine discharges. In principle, EQS values should not be exceeded outside an agreed mixing zone of a given effluent in its receiving water. The EQS claims to define the maximum concentrations of a substance in water which will be protective to aquatic life and human uses of the water body in question. The intention is to protect 'all levels of organisms from the individual species up to the whole community'. If the respective data sets are sufficiently comprehensive, separate EQS values are set for the freshwater and marine environments. For the marine situation, these will usually cover the protection of fish, shellfish, and general marine life, as well as the protection of human consumers of fish and shellfish and human users of marine waters for bathing and contact sports etc. There may also be special EQS values set for such factors as the safe passage of migratory fish in estuaries.

As a general rule, EQS values are not set in the United Kingdom until a substantial body of data is available on the fate and effects of a given substance in the aquatic environment. In general a minimum data set would include (only reliable) acute toxicity data for eight species including fish, invertebrates and algae, as well as bioconcentration and biodegradation data. Ideally, the data set would also include chronic toxicity data for at least the more acutely sensitive species, especially for those substances expected to persist in, or be continuously discharged to, the aquatic environment. In general, mammalian toxicity data will not be reviewed in detail, but reference will be made to any Allowable Daily Intake (ADI) values which may have been set. If sufficient specifically marine data are unavailable, the use of freshwater data for deriving the EQS may be accepted, although due regard will be paid to the differential behaviour of the substance in the two media. A marine EQS derived in this way will be labelled 'provisional' until such time as more comprehensive marine data are available.

The methods used to derive predicted No-Effect Concentrations (NECs) are based on empirical safety factors or extrapolation factors which are applied to acute/chronic laboratory data and/or the results of field experiments. The safety factors are 100, 10 and 1 for the lowest acute, chronic or field value, although expert judgement may be used to modify these if reliable information on, for example, acute-chronic ratios is available. An additional safety factor of 10 is used if a substance has a high propensity to bioaccumulate (log BCF > 4) and is also persistent. The first 3 safety factors are similar to those recommended by the US Environmental Protection Agency (EPA, 1984), the UK Department of the Environment (DOE, 1993), the European Centre for Ecotoxicology and Toxicology of Chemicals (ECETOC, 1993) and the Organisation for Economic Co-operation and Development (OECD, 1992), although the latter document also discusses the use of extrapolation procedures based on statistical models of species sensitivity distribution.

To date, UK EQS values have all been set in terms of concentrations in water, including those for the protection of human consumers of fish and shellfish. In the latter case, the concentration is set such that bioconcentration and bioaccumulation will not result in unacceptable residue levels in edible flesh and are based on ADIs for the protection of human consumers. Apart from ‘action levels’ for certain metallic contaminants and a few organic compounds in the sediments at marine sewage sludge disposal sites, no general sediment quality standards are available for the UK.

The Netherlands

For environmental risk assessment and standard setting in the Netherlands, two risk limits have been defined: an upper boundary concentration (or Maximum Tolerable Concentration - MTC) and a lower boundary concentration (or Negligible Risk Concentration - NRC). At the MTC the risk of a micropollutant causing ecosystem effects (at chronic exposure) is considered the maximum acceptable. This concentration should at least be fully protective of 95% of species including all priority species, in the sense that their chronic No Observed adverse Effect Concentration is not exceeded. This also means that secondary poisoning for top predators should be taken into account. Because MTCs are estimated for individual micropollutants, possible effects due to the combined toxicity (additivity, synergism) of different pollutants present in the environment at the same time are not taken into account. At the NRC the risk of effects on the ecosystem due to anthropogenic pollution is considered as to be negligible, taking into account the possibility of combination toxicity (possible synergistic and additive effects between different micropollutants).

Where possible, the calculation of the MTC should be based on (semi-)field tests or field observations. However, the current practice with regard to effects assessment is such that these data are seldom available and are often difficult to interpret. Therefore, in most cases acute and chronic laboratory studies are used and extrapolation procedures are applied to translate the results to environmentally relevant protection levels. When data on effects on certain species are lacking, estimation procedures (e.g. Quantitative Structure-Activity Relationships, QSARs) may serve as a last resource in the effects assessment procedure.

Depending on the quality and quantity of the data that are available, a series of methods can be used to calculate a concentration below which adverse effects on the aquatic ecosystem are not likely to be expected. Aquatic effects assessment is a sequential process that may comprise three stages: preliminary, refined and comprehensive effects assessment. In preliminary effects assessment the impact of the chemical is generally assessed against only one or two representatives of primary producers (algae), primary consumers (crustaceans, e.g. *Daphnia*) and predators (fish), by means of short-term toxicity tests or even QSARs. Refined effects assessments are based on chronic or semi-chronic tests, whereas (semi-)field studies provide the basis for comprehensive effects assessment. When only a few chronic NOEC values for different species are available, the safety factors applied on the lowest reliable data are 1000, 100 and 10 depending on the amount and character (chronic vs. acute, adult vs. early-life-stage tests etc.) of the available data.

In cases where numerous NOECs are available, the MTC is estimated using an extrapolation procedure based on a statistical log-logistic model of species sensitivity distribution (Aldenberg and Slob, 1991) based on the work of Kooijman (1987) and Van Straalen and Denneman (1989). As the process of effects assessment goes through stages of imprecise to precise as more data are generated, the reliability of the estimated ‘ecosystem safe concentration’ will increase. Based on uncertainty analysis and the precautionary principle the safety factors used in each approach are reduced as more information becomes available.

Based on an evaluation of available freshwater and marine toxicity data, and taking into account the limited availability of toxicity data in general, it has been concluded in the Netherlands that it is acceptable to combine freshwater and marine toxicity data sets in deriving MTCs for water and sediment.

For those substances which have a tendency to biomagnify through the food chain, a general algorithm for the risk-assessment of secondary poisoning in birds and mammals has been derived based on the water - fish/mussel - fish/mussel-eating bird or mammal pathway (Jonkers and Evers, 1992; Romijn *et al*., in press). Parameters used for the algorithm are the BCFs for fish and mussel and the NOECs for birds and mammals. For the derivation of reliable BCFs preference is given to the use of experimentally derived BCFs over QSAR estimates. NOECs for fish- and mussel-eaters are derived by extrapolating toxicity data on single species bird and mammalian laboratory toxicity tests. NOECs are corrected for differences in food composition (energetic value) and metabolism of species in the laboratory versus species in the field.

Because data on interstitial water quality and spiked sediment tests are very limited, and sediment toxicity tests using sediment-dwelling organisms are still under development, in the Netherlands MTCs for the sediment are, for the time being, derived by means of the Equilibrium Partitioning Approach (Shea, 1988; Di Torro, 1991). For organic substances partition coefficients are calculated from K_{OW} values, whereas for heavy metals partition coefficients are based on field measurements or derived experimentally and apply only to the aerobic top-layer of the sediment.

Canada

In Canada the protocol for deriving national Sediment Quality Guidelines builds on the strengths of two complementary approaches: the National Status and Trends Program Approach (NSTPA) and the Spiked-Sediment Toxicity Test Approach (SSTTA). The NSTPA involves the evaluation and compilation of data from a wide variety of sources to establish associations between concentrations of contaminants in sediments and adverse biological effects (i.e., cause and effect relationships cannot be inferred from this data; see Long and Morgan 1990; MacDonald, 1993 and Smith and MacDonald 1993 for supporting details of this approach). At the moment the protocol relies mainly on the NSTPA, with the use of the SSTTA scheduled for the near future once methodological concerns have been resolved.

To support the derivation of guidelines based on the NSTPA matching biological and chemical data are compiled into BEDS which is designed to be updated periodically as new information becomes available. Toxicological studies incorporated into BEDS include measures of altered benthic communities (e.g. reduced species richness or total abundance), significantly or relatively elevated sediment toxicity (field studies), histopathological disorders in demersal fish (field studies), sediment quality assessment values from other jurisdictions (e.g. Apparent Effects Thresholds, Screening Level Concentrations), results of spiked-sediment toxicity tests (EC_{50} or LC_{50} concentrations) and toxic concentrations predicted by equilibrium partitioning models.

The majority of information in BEDS has been derived from individual field (co-occurrence) studies conducted in North America for marine sediments (although the organic carbon content was not measured for the majority of studies in BEDS, it was estimated that the average organic carbon content was 1-2 %). The results of chemical analyses from these studies have been grouped according to the biological response observed (i.e. toxic vs. non-toxic sites). Concentrations of individual chemicals are considered to be associated with the observed toxic response if the mean concentration at sites at which significant adverse effects were observed was a factor of two or more greater than the mean concentration at the sites at which effects were not observed. These concentrations form the ‘effects’ data set. When chemical concentrations differed by a factor of less than 2 between the toxic and non-toxic groups, it is assumed that other factors (whether measured or not) were more important in the etiology of the observed effect than the contaminant under consideration. These concentrations are included in the ‘no-observed effect’ data set for that chemical.

To support the development of guidelines from BEDS, minimum toxicological data requirements have been set to ensure that sediment quality guidelines developed are supported by the weight of evidence that links contaminant concentrations to biological effects and that aquatic ecosystems are adequately protected (Smith and MacDonald 1993; MacDonald, 1993). Both the ‘effects’ data set and the ‘no-effects’ data set must contain at least twenty entries each in the chemical data table. Using the information in BEDS, guidelines are calculated to represent the No Observed Effect level (NOEL), below which no significant hazards to aquatic organisms are predicted. For each chemical, a Threshold Effects Level (TEL) is calculated as the square root of the product (i.e., the geometric mean) of the lower 15th percentile concentration of the ‘effects’ data set and the 50th percentile concentration of the ‘no-effects’ data set. The TEL is multiplied by a safety factor of 0,5 to define the $NOEL_{BEDS}$. In Canada, the $NOEL_{BEDS}$ may be recommended as an interim guideline. A second assessment value, the Probable Effects Level (PEL), is also calculated from BEDS to aid in the interpretation of the data compiled in BEDS. The PEL represents the concentration above which significant hazards to aquatic organisms are predicted.

The inherent strength of the protocol is the use of much evidence to support the development of guidelines. The approach is applicable to a wide variety of chemicals and virtually any sediment type that occurs in freshwater, estuarine and marine environments. The information is compiled from numerous geographic locations throughout North America, and the biological data compiled include a variety of different species and biological endpoints. Most of the information compiled consists of field data which considers complex mixtures of contaminants (and thus their interactive effects) and various sediment types (i.e. with different particle sizes and concentrations of substances), and thus, varying conditions of bioavailability. Therefore, the resultant guidelines are considered to be broadly applicable tools for sediment quality assessment. At present, however, sediment quality guidelines do not address the potential for the bioaccumulation of persistent substances.

Ecotoxicological assessment criteria

Each section of this chapter discusses a separate group of contaminants. Under the heading ‘general’ short remarks are made on the ecotoxicological properties and the relevance of the matrices water, sediment and biota as far as monitoring is concerned. Under the heading ‘ecotoxicological assessment criteria and expert judgement’ available toxicity data are evaluated and ecotoxicological assessment criteria are derived. Finally, under the heading ‘comparison with national guidelines’, the ecotoxicological assessment criteria are briefly examined in relation to existing (national) guidelines/criteria to establish whether they show major differences. These national guidelines should represent concentrations that are not harmful to the marine environment. The overview of quality guidelines presented in MacDonald *et al*.(1992) was used, as well as the papers summarising national guidelines that were prepared by P. Matthiessen (United Kingdom), D. Jonkers (the Netherlands) and K. Hylland (Norway).

PAHs

General

Polycyclic Aromatic Hydrocarbons (PAHs) such as benzo(a)pyrene have mutagenic and carcinogenic potential in (aquatic) organisms. PAHs may also affect reproductive success and growth etc. Under the influence of sunlight the toxic properties of certain PAHs (e.g. fluoranthene and benzo(a)pyrene) can increase very strongly (photo-toxicity). Probably the most important route of uptake of PAHs by aquatic organisms is through the water phase. However, PAH levels in sediments are much higher than in water which makes sediment a more convenient compartment to monitor. However, adsorption of the compounds onto the sediment particles may cause a substantial reduction in bioavailability. Most vertebrates (e.g. fish) and certain invertebrates (e.g. polychaetes) are able to rapidly metabolise PAHs which makes them poor indicators for assessing the potential exposure of aquatic organisms which lack this metabolic capacity. Bivalves such as mussel (*Mytilus edulis*) do show a strong tendency to bioaccumulate PAHs.

Ecotoxicological assessment criteria and expert judgement

Generally, few aquatic toxicity data and very few spiked-sediment toxicity data are available for PAHs. Therefore, no firm ecotoxicological assessment criteria were derived for water and sediment. However, provisional values were set for naphthalene, anthracene, fluoranthene and benzo(a)pyrene. These compounds are probably representative of other PAHs. Toxicity data that can be related to tissue residues are too sparse to enable even provisional values for biota (mussels) to be derived.

The following aquatic toxicity data were considered: for naphthalene the lowest NOEC is 40 µg/l and for anthracene, fluoranthene and benzo(a)pyrene the lower ends of available acute LC_{50} data are 1, 10 and 2 µg/l, respectively. For naphthalene an application factor of 10 was used, for the other compounds a factor of 100 was used. The extrapolated concentrations (4, 0,01, 0,1 and 0,02 µg/l) were rounded to the nearest order of magnitude as follows: naphthalene: 1-10 µg/l; anthracene: 0,005-0,05 µg/l; fluoranthene: 0,05-0,5 µg/l and benzo(a)pyrene: 0,01-0,1 µg/l.

Concentrations in sediment were calculated by applying the Equilibrium Partitioning Approach to the extrapolated concentrations for aquatic toxicity mentioned above. Estimated Kd values for naphthalene, anthracene, fluoranthene and benzo(a)pyrene were 16, 160, 630 and 5000 l/kg, respectively. In this way the following extrapolated concentrations for sediments were calculated for naphthalene, anthracene, fluoranthene and benzo(a)pyrene i.e. 64, 1,6, 63 and 100 µg/kg dw, respectively. In order to set assessment criteria for PAHs in sediment, a range of an order of magnitude was set around these extrapolated concentrations as follows: naphthalene: 10-100 µg/kg dw; anthracene: 1-10 µg/kg dw; fluoranthene: 10-100 µg/kg dw and benzo(a)pyrene: 50-500 µg/kg dw.

For sediments the NOELs from BEDS were considered. Entries in BEDS are mainly based on co-occurrence analysis and the apparent effect threshold approach. For naphthalene, anthracene, fluoranthene and benzo(a)pyrene the NOELs are 130, 85, 380 and 230 µg/kg dw, respectively. In general, these levels appear to be higher than the ecotoxicological assessment criteria which is partly explained by the fact that an application factor of only 2 was applied in the calculation of the NOELs (see section "Summary of National Approaches", starting on page 14) instead of the application factor of 10 applied to the aquatic toxicity data at the workshop. Compared to the upper limit of the ecotoxicological assessment criteria presented above, the NOELs for naphthalene and fluoranthene are comparable, in case of anthracene the $NOEL_{BEDS}$ is higher and in case of benzo(a)pyrene the $NOEL_{BEDS}$ is lower.

With regard to fluoranthene and benzo(a)pyrene, reported natural background levels are <30 µg/kg and <0,01-20 µg/kg dw (Laane, 1992) respectively, which are below or at the lower end of the ranges that were derived.

Comparison with national guidelines

In general, the ecotoxicological assessment criteria for PAHs in water are within the same range as the guidelines for the EC, Canada, US and the Netherlands. For PAHs in sediment, assessment criteria presented in this report are within the lower range of published guidelines.

PCBs

General

PCBs are highly persistent, have a high bioaccumulation potential and are regarded as widespread and threatening contaminants in the marine environment for top predators such as fish-eating birds and marine mammals. Immuno-incompetence and reproductive impairment are the most likely consequences. The most critical pathway is considered to be via the food chain.

Most JMP data concern congeners IUPAC Numbers 28, 52, 101, 118, 138, 153 and 180 (the sum of these seven congeners is represented by ΣCB_7). Except for CB 118, these congeners do not have a specific mode of action, but are of the narcotic type. Toxicity data for these individual congeners are very sparse. Most toxicity studies have been performed with technical mixtures where concentrations are expressed on a total PCBs basis, or with individual planar congeners having a dioxin-like mode of action. The latter group require a separate approach to derive ecotoxicological assessment criteria, but these are presently not included in the JMP. Based on a toxicity assessment using toxic equivalent factors, it appeared that the planar congeners CB 126 and CB 77 could be responsible for 90% of the toxic effect of the total PCB content in fish (De Boer *et al.*, 1993). However, for the purposes of the workshop, it was only feasible to treat the narcotic acting CBs as a single group. With respect to the toxicity data, it was assumed that the concentration of ΣCB_7 is equivalent to 50% of the concentration of total PCBs. Because of the technical difficulties of determining concentrations of PCBs in water this compartment was not included in the JMP and therefore ecotoxicological assessment criteria were derived for sediment and biota only.

Ecotoxicological assessment criteria and expert judgement

For PCBs in food items sufficient data on chronic NOECs are available to derive assessment criteria that indicate safe levels for fish-eating birds and mammals. In a semi-field experiment with harbour seals, one group was fed PCB-contaminated Wadden Sea fish as the sole food source while a control group was fed less-contaminated Atlantic fish (Reijnders, 1986; Boon *et al.*, 1987). After correcting the PCB concentrations in the fish to a standard fish with 5% lipid, the total PCBs (42 congeners) in the diets of the harbour seals were estimated to be 600 and 33 µg/kg ww, respectively. No effects were seen in the group with the Atlantic fish diet, whereas the experimental group suffered a 60% reduction in reproductive success. With regard to the reproductive success of mink, a no-effect level for total PCBs of 125 µg/kg food was assumed on the basis of a lowest effect concentration of 250 µg/kg ww; for birds a no-effect level of 2 500 µg/kg ww in food was assumed on the basis of a lowest effect concentration of 5 000 µg/kg for chick-embryo mortality (Safe, 1987). The NOEC for seals appears to be the most critical value. Extrapolated ΣCB_7 concentrations of 1,7 and 6,3 µg/kg ww were derived from the apparent NOEC for seals of 33 µg/kg ww ΣCB_{42} and the NOEC for mink of 125 µg/kg ww total PCBs, respectively (assuming ΣCB_7 is 50% of the total PCBs or ΣCB_{42}, and an application factor of 10). The assessment criterion was therefore set at 1-10 µg/kg ww.

There are very few spiked sediment toxicity data for PCBs. For the American amphipod *Rhepoxynius abronius* a "significant" acute toxicity was seen at 5 200 µg/kg whole sediment (Plesha *et al.,* 1988). Divided by an extrapolation factor of 1000 a value of 2,6 µg/kg was obtained, assuming ΣCB_7 is 50% of the total PCBs. The provisional ecotoxicological assessment criterion for ΣCB_7 was set at 1-10 µg/kg dw for sediments with a 1% organic carbon content. This is somewhat lower than the NOEL for ΣCB_7 from BEDS, 12 µg/kg dw, which is partly explained by the fact that an application factor of only 2 was applied in the calculation of the NOELs (see section "Summary of National Approaches", starting on page 14). It was argued that this range is not necessarily protective for non-sediment-dwellers because chronic effects related to biomagnification are not included in this approach. For instance, the feeding experiment with seals described above showed that seal reproduction was strongly impaired when the animals were fed with fish from the Wadden Sea, an area with an average concentration of ΣCB_7 of approximately 10 µg/kg in sediments. The upper value of the provisional assessment criterion for sediments may therefore not be protective for top predators. More work has to be done to incorporate the phenomenon of biomagnification and so set ecotoxicological assessment criteria that are protective for non-sediment dwellers (see also "intercompartmental tuning of assessment criteria", page 27).

Comparison with national guidelines

National guidelines for total PCBs or individual congeners had first to be converted to ΣCB_7 (see assumptions above) before a comparison could be made with the ecotoxicological assessment criteria for ΣCB_7. Values for sediments (Canada, the Netherlands, United Kingdom, United States) and fish (the Netherlands and United States) range between 7-35 µg/kg dw and 7-50 µg/kg ww, respectively. Thus, in the case of PCBs, it appears that the lowest national guidelines are of the same order of magnitude as the assessment criteria presented in this report.

TBT

General

It appears that marine species are more susceptible to TBT than freshwater organisms. The most critical endpoint is the occurrence of imposex in gastropods, a sub-lethal effect that causes reproductive impairment and which can eventually lead to a local extinction of the population. As TBT causes ecotoxicological effects at very low concentrations, analysis protocols have only recently become available for the determination of relevant levels in the marine environment. Concentrations are often expressed as tin in the TBT fraction. Concentrations are 2,5 times higher when expressed as TBT.

Ecotoxicological assessment criteria and expert judgement

Field observation of the occurrence of imposex in the dog whelk *Nucella lapillus* showed that the no-effect level for imposex is below a TBT concentration of 2,5 ng/l (Gibbs and Bryan, 1987). By applying a factor of 10 to this apparent NOEC an extrapolated concentration of 0,25 ng/l was derived. The ecotoxicological assessment criterion for TBT was therefore set at 0,1-1,0 ng/l.

A field experiment with artificially TBT-contaminated estuarine sediments (0,1-10 mg TBT/kg) showed a dose-response in several sediment-dwellers (Matthiessen and Thain, 1989). The burrowing activity of the polychaete *Arenicola marina* was reduced at 1 mg/kg but not at 0,1 mg/kg. An application factor of 100 to the apparent NOEC gave an extrapolated concentration for TBT in sediment of 1 µg/kg. Recovery of affected populations of sediment dwellers other than gastropods in British estuaries has been observed at concentrations of approximately 10-20 µg/kg dw (Waite *et al*., 1991). However, the extrapolated concentration that was derived might not be protective for species such as *N. lapillus*. By means of the Equilibrium Partitioning Approach the extrapolated concentration for water mentioned above (0,25 ng/l) resulted in an extrapolated concentration for sediment of 0,1 µg/kg dw. A rough estimate of the Kd of 400 for sediment with 1% organic carbon was applied in this calculation. Using the mean of both extrapolated concentrations for sediments (0,55 µg/kg) the ecotoxicological assessment criterion for sediment was set at 0,1-1 µg/kg dw.

TBT tissue residues in *N. lapillus* exposed to <0,5 ng/l that were not suffering from imposex contained 195 µg/kg dw (Gibbs and Bryan, 1987). No extrapolation factor was applied, as *N. lapillus* is probably one of the most sensitive species. The ecotoxicological assessment criterion was set at 50-500 µg TBT/kg dw. Because concentrations in mussels and dog whelks from the same location are comparable, the range that was set can be applied to mussels.

Comparison with national guidelines

Existing national guidelines for TBT are available for water only and range between <1-5 ng/l, being approximately one order of magnitude less stringent than the ecotoxicological assessment criterion derived here.

Organochlorine Pesticides

General

Organochlorine pesticides such as DDE and dieldrin are very persistent in the aquatic environment and have a strong tendency to bioaccumulate. Therefore, ecotoxicological assessment criteria were set for biota and sediment. For lindane, no ecotoxicological assessment criteria were derived for biota and sediment because bioaccumulation and adsorption to sediments is relatively limited.

Ecotoxicological assessment criteria and expert judgement

DDE

Data on marine aquatic toxicity of p,p'-DDE were not available. For sediments, national guidelines were taken into account. For p,p'-DDE in sediments the $NOEL_{BEDS}$ is 1.7 µg/kg dw. The NRC for the Netherlands is 3 µg/kg dw for sediments with 1% organic carbon. The provisional ecotoxicological assessment criterion was set at 0,5-5 µg/kg dw.

With regard to secondary poisoning, few toxicity data on DDE levels in food are available, the lower end of the NOEC data for birds being 1 mg/kg ww in food. Using an application factor of 10 the provisional ecotoxicological assessment criteria was set at 0,05-0,5 mg/kg ww in fish.

Dieldrin

No marine sediment toxicity data for dieldrin were available. Therefore the Equilibrium Partitioning method was used. The lower end of the NOECs for marine organisms is 0,1 µg/l. An application factor of 10 and a Kd of approximately 1 260 gave an extrapolated concentration of 13 µg/kg dw in sediment. The provisional ecotoxicological assessment criterion was set at 5-50 µg/kg dw.

With regard to secondary poisoning by dieldrin, many toxicity data are available: the lower end of the NOEC data for birds is 1 mg/kg ww in food. To derive an assessment criterion for fish an application factor of 10 was applied to the NOEC for birds' food and rounded to 0,05-0,5 mg/kg ww.

Lindane

There are very few marine toxicity data for lindane in water. However, sufficient freshwater toxicity data were available to derive a provisional assessment criterion. The lower end of the NOEC data in freshwater is 2,2 µg/l. An application factor of 100 gave an extrapolated concentration of 0,02 µg/l which was rounded to a provisional ecotoxicological assessment criterion of 0,005-0,05 µg/l.

Comparison with national guidelines

National guideline values for lindane are available for water only and vary between 0,004 and 0,6 µg/l. The ecotoxicological assessment criterion is within this range.

National guideline values for DDE in sediment vary between 3-300 µg/kg dw and for dieldrin in sediment between 1-40 µg/kg dw. The ecotoxicological assessment criterion is comparable for dieldrin but more conservative for DDE.

For DDE, the specific guideline value for wildlife in the State of New York (Newell *et al.*, 1987) is 0,2-0,3 mg/kg ww fish flesh. This is comparable to the MTC of 0,15 mg/kg ww in fish derived according to the Dutch secondary poisoning method. For dieldrin, the specific guideline value for wildlife in the State of New York (Newell *et al.*, 1987) is 0,02-0,12 kg ww fish flesh. This is also comparable to the MTC of 0,11 mg/kg ww in fish derived according to the Dutch secondary poisoning approach. For both DDE and dieldrin the ecotoxicological assessment criteria for fish are comparable.

Trace Metals

General

When present in the environment in high concentrations due to pollution, trace metals may disrupt normal cellular processes. This may be particularly the case for non-essential elements (e.g. lead, mercury, cadmium) but also for essential elements (e.g. zinc, copper, chromium, nickel), although the latter are required in trace amounts by most living organisms if they are to grow and reproduce normally. The adverse effects of a specific metal and species will depend on the bioavailability of the metals ('exposure') and the presence/absence of additional stress factors such as food availability and the presence of other metals etc.

Based on a sufficient amount of marine and freshwater toxicity data all ecotoxicological assessment criteria for water are considered to be ‘firm’.

Due to a lack of standardised spiked-sediment toxicity tests for sediment dwelling marine organisms, there was insufficient information to derive ‘firm’ values for trace metals in sediments. Moreover, the Equilibrium Partitioning Approach was considered inappropriate due to large uncertainties in partition coefficients. For example, the partition coefficients for lead proposed by the United Kingdom and the Netherlands differed by two orders of magnitude (4 and 600 l/g dry sediment respectively). Therefore, provisional ecotoxicological assessment criteria were derived based on the NOEL as calculated from BEDS. $NOEL_{BEDS}$ are themselves very close to the presumed high background levels. When the NOEL was rounded to the nearest order of magnitude, it appeared that for several trace metals the ecotoxicological assessment criteria were below the background concentrations.

Little information was available during the workshop to evaluate the toxic effects of metals for fish and mussels on the basis of their tissue residues. Considering that ecotoxicological assessment criteria for trace metals in water and sediment are probably close to or within the range of natural background concentrations and because of regulation mechanisms for metals within the organisms, it was considered very difficult to derive ecotoxicological assessment criteria which differ from background concentrations. However, the workshop participants considered it inappropriate to use the Norwegian presumed high background levels as estimates of ecotoxicological assessment criteria.

In the case of mercury, it was considered necessary to take into account secondary poisoning of top predators through the consumption of fish. Cadmium was evaluated for secondary poisoning but this appeared not to be a critical pathway.

Ecotoxicological assessment criteria and expert judgement

Arsenic

For sea water sufficient NOECs are available, the lowest NOEC being 631 µg/l (Lussier *et al.*, 1985). With respect to freshwater organisms, numerous NOECs are available for species from different taxonomic groups with a lower limit of 10 µg/l. An extrapolated concentration of 6 µg/l was derived using an extrapolation factor of 100 applied to the lowest NOEC for marine species, and an extrapolated concentration of 1 µg/l was derived by applying a factor of 10 to the lower end of the freshwater NOEC data. Consequently, the ecotoxicological assessment criterion was set at 1-10 µg/l.

According to BEDS the NOEL for arsenic in sediment is 8 mg/kg dw. Consequently, the ecotoxicological assessment criterion was set at 1-10 mg/kg dw. This range appears unrealistic as it is well below the Norwegian presumed high background concentration of 20 mg/kg dw.

Cadmium

For marine organisms numerous NOECs are available for species from different taxonomic groups. The lower range of the NOECs is 5 µg/l. An extrapolated concentration of 0,5 µg/l was derived using an extrapolation factor of 10 applied to the lower end of the NOEC and consequently the range was set at 0,1-1 µg/l.

According to BEDS the NOEL for cadmium in sediment is 1 mg/kg dw. The ecotoxicological assessment criterion for sediment was set at 0,5-5 mg/kg dw. The lower limit is below the background concentration of approximately 0,25 mg/kg dw.

Chromium

For marine organisms sufficient aquatic NOECs are available. An extrapolated concentration of 5 µg/l was derived using an extrapolation factor of 10 applied to a lowest NOEC of 50 µg/l. With respect to freshwater organisms numerous NOECs are available for species from different taxonomic groups with a lower end of 10 µg/l. An extrapolated concentration of 1 µg/l was derived by application of an extrapolation factor of 10 to the lower end. The ecotoxicological assessment criterion was set at 1-10 µg/l.

According to BEDS the NOEL for chromium in sediment is 33 mg/kg dw and consequently the ecotoxicological assessment criterion was set at 5-50 mg/kg dw for sediment. This range appears unrealistic as it is well below the Norwegian presumed high background concentration of 70 mg/kg dw.

Copper

For marine aquatic organisms few NOECs are available with a lower end of 20 µg/l. However, numerous acute and (semi-)chronic EC_{50} values are available with a lower end of 6 µg/l. Application of an extrapolation factor of 100 gave 0,06 µg/l as an extrapolated concentration, being below the natural background concentration of approximately 0,2 µg/l. With respect to freshwater numerous toxicity data (NOECs) are available for different taxonomic groups, the lower end being 5 µg/l. Application of an extrapolation factor of 10 to the lower end gave 0,5 µg/l as an extrapolated concentration. The ecotoxicological assessment criterion for copper was set at 0,1-1 µg/l.

According to BEDS the NOEL for copper in sediment is 28 mg/kg dw and consequently the ecotoxicological assessment criterion was set at 5-50 mg/kg dw for sediment. The lower end of this range appears unrealistic as it is below the Norwegian presumed high background concentration of 35 mg/kg dw.

Lead

For marine aquatic organisms few NOECs are available with a lower end of 20 µg/l. Because no NOECs are available for algae and fish an extrapolation factor of 100 was applied to the lower end giving a extrapolated concentration of 0,2 µg/l. With respect to acute EC_{50} values numerous data are available, also with a lower end of 20 µg/l. Application of an extrapolation factor of 100 to the lower end also gives a extrapolated concentration of 0,2 µg/l. With respect to freshwater toxicity data numerous NOECs are available for different taxonomic groups showing a lower end of 5 µg/l. Application of an extrapolation factor of 10 to the lower end gave 0,5 µg/l. The ecotoxicological assessment criterion for lead was set at 0,1-1 µg/l.

The $NOEL_{BEDS}$ for lead in sediment is 21 mg/kg dw and consequently the ecotoxicological assessment criteria was set at 5-50 mg/kg dw. The lower end of the range appears unrealistic as it is below the Norwegian presumed high background level of 30 mg/kg dw.

Nickel

For marine aquatic organisms very few NOECs are available. An extrapolated concentration of 0,6 µg/l was derived using an extrapolation factor of 100 applied to the lowest NOEC of 61 µg/l (Lussier *et al.*, 1985). With respect to EC_{50} values numerous data are available for different taxonomic groups, with a lower range of 30-100 µg/l. Application of an extrapolation factor of 100 to this range gave 0,3-1 µg/l as a extrapolated concentration. With respect to freshwater organisms numerous NOECs are available for species from different taxonomic groups with a lower end of 2 µg/l. An extrapolated concentration of 0,2 µg/l was derived by application of an extrapolation factor of 10 to the lower end. The ecotoxicological assessment criterion for nickel in water was set at 0,1-1 µg/l.

For nickel no NOEL has been established in BEDS.

Mercury

For marine organisms NOECs are available for different taxonomic groups with a lowest value of 0,25 µg/l (Thain, 1984). An application factor of 10 was used resulting in an extrapolated concentration of 0,025 µg/l. To set an ecotoxicological assessment criterion for mercury in water that is also 'safe' for top predators, an evaluation of secondary poisoning was made. The extrapolated concentration for mercury in fish of 0,09 mg/kg ww (see below) was transferred to an extrapolated concentration in water by applying a BCF of 14 000. The extrapolated concentration in water was 0,006 µg/l. Because biomagnification is expected to be the most critical 'route' of exposure, the ecotoxicological assessment criterion was consequently set at 0,001-0,01 µg/l. The Norwegian presumed high background value of 0,005 µg/l, which is approximately 10 times the natural background value (Laane, 1992), is within this range.

Based on a sufficient number of laboratory dietary bird studies a Tolerable Residue Level (TRL) for methyl-mercury was estimated at 0,09 mg/kg food. In fish, mercury is predominantly present in the methyl-mercury form. The ecotoxicological assessment criterion was set at 0,05-0,5 mg/kg ww.

According to BEDS the NOEL for mercury in sediment is 0,1 mg/kg dw. The assessment criterion was consequently set at 0,05-0,5 mg/kg dw. The lower end of this range is below the Norwegian presumed high background level of 0,15 mg/kg dw.

Zinc

For marine organisms numerous NOECs are available for different taxonomic groups with a lower end of 20 µg/l. Application of an extrapolation factor of 10 to this lower end gave 2 µg/l as a extrapolated concentration. The ecotoxicological assessment criterion for dissolved zinc was set at 0,5-5 µg/l.

The $NOEL_{BEDS}$ for zinc in sediment is 68 mg/kg dw. Consequently, the ecotoxicological assessment criterion was set at 10-100 mg/kg dw. This range appears unrealistic as it is below the Norwegian presumed high background concentration of 150 mg/kg dw.

Comparison with national guidelines

For trace metals hardly any guidelines exist for tissue residues, except for human consumption standards. The latter however, have not been taken into account.

As apparent from the previous section, it did not seem useful to make a comparison between the ecotoxicological assessment criteria and national guidelines because the former are very close to background levels.

For water the ranges in national guidelines/criteria that represent 'safe' concentrations for trace metals are as follows: arsenic 8-25 µg/l; cadmium 0,1-2,5 µg/l; chromium 5-10 µg/l; copper 2-5 µg/l; lead 2-10 µg/l; nickel 1,5-15 µg/l; mercury 0,006-0,3 µg/l; zinc 2,6-10 µg/l. The upper limit of the ecotoxicological assessment criteria lay within the range of the existing guideline values, except for copper, nickel and lead in which cases the upper limits were substantially lower than national guidelines.

Discussion

Combination toxicity

The assessment criteria presented in this document were derived for individual substances separately, without specifically addressing the possible interactions with other substances. In the field, however, organisms are exposed to numerous substances at the same time. Especially in estuaries and in coastal areas where numerous chemicals may be present in detectable concentrations. Interactions between different (groups of) substances may lead to antagonistic, synergistic or additive effects. For chemicals with a specific toxic mechanism (such as PCBs, dioxins and dibenzofurans) or a non-specific (narcotic) mode of action the effects of mixtures of chemicals can often be adequately described by concentration addition.

With respect to aquatic organisms research on the toxicity of mixtures of substances within a group of chemicals with the same mode of action (e.g. for chlorobenzenes, chlorophenols, anilines, aromates, chlorinated solvents, pesticides, PAHs, heavy metals) shows that the effects of the individual substances are almost always additive. As an example recent research showed almost complete concentration addition for a mixture of PAHs at concentrations below the individual NOEC for fish (Hooftman *et al.*, 1993). The combined toxicity of a mixture of substances with different toxic mechanisms however, is extremely difficult to estimate. Besides the fact that the mixture of chemicals in the field changes from place to place, present scientific knowledge is too limited to establish a scientifically based (additional) safety factor for combination toxicity. Therefore, in deriving ecotoxicological assessment criteria, the workshop decided, for the time being, not to take into account possible effects of combination toxicity. An exception to this may be the use of BEDS which does indirectly account for mixtures and bioavailability. However, it was concluded by the workshop that it is of great importance that a scientifically based approach, which includes possible effects of combination toxicity, should become available soon.

Intercompartmental tuning of assessment criteria

During the workshop ecotoxicological assessment criteria were derived, where possible, for water, sediment and biota, based on available toxicological data for organisms within those compartments. Except for PAHs, for which spiked-sediment toxicity data were lacking and for which the Equilibrium Partitioning Approach was used to convert aquatic toxicity values to sediment values, no attempt was made by the workshop to link the ecotoxicological assessment criteria in one compartment with assessment criteria in other compartments. In the environment however, concentrations of substances in water, sediment and biota are related, e.g. through processes such as adsorption, desorption and bioaccumulation. Therefore it was argued that assessment criteria for water, sediment and biota should also be tuned to each other (e.g. by means of Equilibrium Partitioning and/or the application of BCFs). As the goal of the assessment criteria was to set criteria for a sustainable ecosystem, the assessment criteria for the different compartments should be determined for the most critical compartment. This might be especially relevant for compounds with a (strong) tendency to bioaccumulate (mercury, DDE, dieldrin, PCBs) when setting safe levels for top predators such as fish-eating birds and mammals. In these cases the assessment criteria for water and sediment should be based on toxicity data on the basis of tissue residues in fish because this will be the most critical pathway. Although the necessity for intercompartmental tuning of assessment criteria was raised during the workshop, no specific attempts were made to do so.

It was recommended that attention should be paid to the intercompartmental tuning of assessment criteria when the values are updated or evaluated.

Extrapolation factors and natural background concentrations in sediments

Because in sediments the toxic levels for many trace metals are close to background levels, it appeared during the workshop that application of an extrapolation factor of 10, 100 or 1000 (depending on the (phylogenetical) coverage of species and the amount of data available) to the lower end of available whole sediment toxicity values often lead to figures which are below the natural background levels. Therefore the workshop agreed to base the assessment criteria for trace metals in sediments directly on the $NOEL_{BEDS}$ without the application of an additional safety factor.

A possible explanation for this phenomenon might be that when applying safety factors to whole sediment concentrations, not only the (potential) bioavailable sediment fraction is taken into account and divided by the safety factor but also the non-bioavailable (inert) sediment fraction. This lead to the suggestion that both sediment toxicity data (e.g. LC_{50}s and NOECs from toxicity experiments) and quality objectives (standards, guidelines, assessment criteria) should refer to the (potential) bioavailable fraction instead of the total sediment. More research is needed to develop methods to define and identify ‘the’ (potential) bioavailable fraction of chemicals in sediments and to formulate an appropriate set of extrapolation factors.

Development of standardised sediment toxicity tests

Sediment toxicity assessment is a complex discipline with many variables. Especially the interpretation of test results and extrapolation of results to sediments which differ for example in granulometry, organic carbon and sulphide content and redox conditions. As sediment toxicity tests and bioassays are becoming instruments of increasing importance to regulators and scientists, there is an urgent need for international standardisation in this area. Over the last few years much progress has been made towards a standardised methodology. Besides recommended procedures which have recently become available (ASTM, 1991,1993; Environment Canada, 1992 a,b), the Organisation for Economic Co-operation and Development (OECD, 1992), the US-Environmental Protection Agency (US-EPA, 1992) and SETAC-Europe (Hill et al., 1993) held workshops on sediment toxicity testing and bioassay procedures. However, with respect to standardisation of the methodology (materials, organisms, chemistry and design) and the interpretation and extrapolation of results, much work has still to be done.

Normalisation of chemical concentrations in sediments

With respect to organic chemicals the assessment criteria for sediment, as defined during the workshop, refer to a ‘typical whole marine sediment’ with 1% organic carbon. In order to compare measured concentrations of organic chemicals in sediments with the assessment criteria, measured concentrations (both in whole sediment samples and in separated sediment fractions), should be normalised to organic carbon by linear extrapolation and expressed for a carbon content of 1%.

With respect to metals, assessment criteria for whole sediment are assumed to represent an average organic carbon content of 1-2%, regardless of the granulometry. However, no methodology was put forward at the workshop to compare measured concentrations of metals in whole sediment samples or separated sediment fractions with significantly deviating granulometry and/or organic carbon content, with the ecotoxicological assessment criteria. Therefore, it was recommended by the workshop that a methodology is developed with respect to the normalisation of metals in marine sediments.

Use of marine and freshwater toxicity data

In a Dutch study carried out to derive environmental quality objectives for micropollutants in the marine environment (Jonkers and Everts, 1992) the question was raised as to whether marine organisms might be more sensitive to chemical pollution than freshwater organisms. Based on available (and for the marine environment often limited) toxicological information for seventeen substances, and taking into account that a theoretical framework for possible differences in sensitivity (other than differences in bioavailability and thus exposure) was missing, it was concluded that (with the exception of TBT) marine organisms appear to be approximately as sensitive to chemical pollutants as freshwater species. After confirming this conclusion by a comparison of marine and freshwater toxicity data for metals at the level of taxonomic groups (Van der Plassche *et al.*, 1992), the Dutch Government decided to combine freshwater and marine data sets for deriving environmental quality objectives for chemicals. Also, within the methodology of the Scientific Advisory Committee on Toxicity and Ecotoxicity of Chemicals of the European Commission, toxicity data for freshwater and marine organisms are brought together and evaluated in coherence to derive environmental quality objectives for chemicals.

However, the workshop decided to give preference to marine toxicity data and, with respect to the algorithm for the risk assessment of secondary poisoning, to terrestrial dietary toxicity data and measured BCFs for marine fish. In cases where the assessment criteria were primarily determined by freshwater toxicity data or based on the algorithm for the risk assessment of secondary poisoning using calculated BCFs, the assessment criteria were regarded as ‘provisional’.

Application and evaluation of the ecotoxicological assessment criteria

For the application of the assessment criteria the following interpretation was suggested by the workshop. Levels below the range were regarded to be in a “green zone”, meaning that no harm to the marine environment was to be expected. Levels above the upper limit were regarded to be in a “red zone”, meaning that harmful effects are to be expected with the incidence and severity of adverse biological effects increasing with increasing contaminant concentration. The range itself was considered to represent a “grey zone”, which meant that the possibility of harmful effects could not be excluded. With a view to an effective monitoring programme the following recommendations were made with respect to monitoring:

a. in those areas where concentrations of certain substances are above the upper boundary concentration, monitoring of those substances and within those areas has highest priority with a view to:

   (i) spatial monitoring to identify the major sources and/or pathways of pollution;

   (ii) temporal trend monitoring to assess environmental quality changes and to evaluate the effect of (e.g. source directed) environmental measures;

b. in those areas where concentrations of certain substances are below the lower boundary concentration, monitoring of those substances and within those areas should be minimised and, if still considered necessary (e.g. for political reasons), should be focused on long-term trends.

c. For those substances and areas where concentrations are within the grey zone, monitoring should be carried out according to a standard programme, and focused on both spatial distributions and temporal trends.

The workshop recommended that the ecotoxicological assessment criteria are re-evaluated, added to and updated on a regular basis e.g. using new scientific information and/or existing information which was not available to the workshop at the time. Besides the derivation of additional values, assessment criteria which were considered provisional should be revised as soon as new toxicological data makes it possible to derive firm values.

Comparison with field data

To test the practical use of the ecotoxicological assessment criteria as assessment tools for screening monitoring data in the future a comparison was made during the workshop between the ecotoxicological assessment criteria and monitoring data. A selection of the monitoring data collected under the JMP were made available to the workshop by ICES and comprised a large data set of concentrations in water, sediment and biota collected in the North-East Atlantic Ocean over the last 10 years. The coverage of the data however, was not representative of the maritime area as a whole. In general it was concluded that the ecotoxicological assessment criteria were realistic, considering the monitoring data. Although the available monitoring data had not been assessed in detail, taking into account the geographical coverage of the data, the ecotoxicological assessment criteria were considered useful as a tool for identifying areas of special concern.

Recommendations

Recommendations with respect to the JMP

1 The ecotoxicological assessment criteria should be used in future assessments of monitoring data.

2 The list of assessment criteria needs to be further refined and updated in order to establish a wider list of assessment criteria or to ensure consistency between values for different environmental compartments.

3 All ecotoxicological assessment criteria which are considered provisional should be revised as soon as the availability of new toxicological data makes this possible, in order to establish firm values.

4 Caution should always be exercised in using generic, particularly provisional, assessment criteria in specific situations. Their use does not preclude the use of common sense and expert judgement with regard to the natural concentrations of e.g. trace metals in assessing the (potential for) environmental effects.

General recommendations

5 In order to support the derivation of more reliable assessment criteria for marine monitoring data for the North-East Atlantic Ocean more toxicological (especially marine) data are required, covering a range of microcontaminants, species and conditions.

6 In order to support the process of developing and updating ecotoxicological assessment criteria for the marine environment it would be beneficial to determine which of the existing computerised toxicological databases, or combinations thereof, would be most useful.

For aquatic toxicity data the US-EPA “AQUIRE” database is an example of what is needed, although essential information on experimental conditions, data reliability, ecology of the species etc. is missing. An example of a database which does provide this type of information is MARITOX, currently being compiled by the Dutch TNO. For marine sediment toxicity probably the most advanced database is BEDS. At present, BEDS is based largely on matching empirical data on effects with field concentrations, since very few spiked-sediment tests have been conducted. When available, spiked-sediment tests (preferably based on chronic endpoints) can be used in conjunction with toxicity data in the field to derive reliable sediment quality guidelines. BEDS is designed to be updated periodically and it should be investigated to determine the feasibility of developing a similar database for Contracting Parties to the Oslo and Paris Conventions. Furthermore, a database should be developed to compile data on biological effects and concentrations in biota as observed in the laboratory, mesocosm and in the field. This approach based on tissue residues is considered to be promising and will provide additional information to establish more directly ecotoxicological assessment criteria for chemicals in biota.

7 The Equilibrium Partitioning Approach appears to be an acceptable method for deriving assessment criteria for organic contaminants with log Kow < 5 in sediments from aquatic toxicity values. As regards organic contaminants with log Kow > 5 for which sediment toxicity data are missing, the Equilibrium Partitioning Approach can only be used to derive provisional assessment criteria at present.

8 Besides the general need for sediment toxicity data there is an urgent need for the development of standardised whole sediment bioassays.

9 The algorithm for the risk assessment of secondary poisoning was agreed upon by the workshop as an important approach to derive assessment criteria for biota which aim to protect top predators such as fish-eating birds and mammals. More research of this type should be carried out especially for lipophilic non-biodegradable contaminants (e.g. PCBs).

10 To assess the ecotoxicological impact of PCB contamination in the marine environment planar (dioxin-like) PCB congeners should be monitored.

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Annex I List of workshop participants

During the workshop, the plenary sessions were chaired by Dr. J. Seager. The three working-groups that were established (trace metals, PCBs & PAHs, organochlorine pesticides & TBT) were chaired by Dr. G. Persoone, Dr. A. Opperhuizen and Dr. A. Künitzer. Rapporteurs of the working-group sessions were Dr. K. Hylland, Dr. P. Matthiessen and Dr A. Granmo. The workshop has been organized by Mrs. C. van Zwol, Mr. D. Jonkers and Mr. J. Stronkhorst.

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Annex II Overview of determinands and the matrix recommended by JMG/NSTF for sampling and analysis. The recommended matrix may differ for geographical distribution studies and temporal trend studies.

Determinand	Water	Sediment	Fish liver	Fish muscle	Shellfish
As		V			
Cd	M	M	M		M
Cr		V			
Cu	M	M	M		M
Hg	M	M		M	M
Ni		V			
Pb	M	M	M		M
Zn	M	M			
DDE		V,P P	S 3) P		S 2)
dieldrin		V,P P	S 3) P		S 2)
lindane	M				
PAHs		V,P			S 2) P
PCBs	V	V,P	S 3) M P		S 2) M
TBT	S 1) P	V,P			V,P P

M = mandatory; V = voluntary; S = secondary matrix; P = primary matrix

1) near-shore water
2) on opportunistic basis : may provide additional info.
3) sedentary species only

Annex III Abbrevations

ADI	=	Allowable Daily intake
ASTM	=	American Society for Testing and Materials
AQUIRE	=	Aquatic Toxicity Information Retrieval Data Base of US-EPA
BCFs	=	Bioconcentration Factors
BEDS	=	Biological Effect Database for Sediments
CSTE	=	Scientific Advisory Committee on Toxicity and Ecotoxicity of Chemicals
DDE	=	2,2-bis (p-chlorophenyl)-1,1-dichloroethylene
DOE	=	Department of the Environment (UK)
dw	=	dry weight
EC	=	European Community
EC_{50}	=	Effect Concentration on 50 % individuals
EPA	=	Environmental Protection Agency
EQS	=	Environmental Quality Standards
JMG	=	Joint Monitoring Group
JMP	=	Joint Monitoring Programme
Kd	=	sediment/water partition coefficient
Kow	=	1-octanol/water partition coefficient
LC_{50}	=	Lethal Concentration on 50% individuals
MARITOX	=	Marine Toxicity Database of TNO
MTCs	=	Maximum Tolerable Concentrations
NECs	=	No-Effect Concentrations
NOECs	=	No Observed Effect Concentrations
NOEL	=	No Obseved Effect Level (information from BEDS)
NRC	=	Negligible Risk Concentration
NSTF	=	North Sea Task Force
NSTPA	=	National Status and Trends Program Approach
OC	=	Organic Carbon
OECD	=	Organisation for Economic Cooperation and Development
OSPAR	=	Oslo and Paris Commissions
PAHs	=	Polycyclic Aromatic Hydrocarbons
ΣCBs	=	sum of PCB congeners 28,52,101,118,138,153,170
PCBs	=	Polychlorinated bi-phenyls
PEL	=	Probable Effects Level
QSAR	=	Quantitative Structure-Activity Relationship
SETAC	=	Society of Environmental Toxicity and Chemistry
SSTA	=	Spiked-Sediment Toxicity Test Approach
TBT	=	TriButylTin
TEL	=	Threshold Effect Level
TNO	=	Dutch Insitute of Environmental Sciences
TRL	=	Tolerable Residue Level
ww	=	wet weight