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– Comparisons of different Nordic approaches for marine assessments

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Content

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Tribute to the memory of Britta Maria Petersen

The project was initiated by the marine scientist Britta Maria Petersen (Ph.D.), who passed away far too early 19th November 2003 at an age of 58, after more than 15 years of research in the marine environment. The scientific work she performed laid a solid ground for the knowledge of heavy metals and harmful substances in the Danish marine waters, both as a key person in the Danish monitoring programmes from the early start and up to the current NOVA and NOVANA programmes, that is designed to support both OSPAR, HELCOM and EU monitoring. She participated in the marine convention work, the ICES expert group on marine chemistry, as well as the QUASIMEME quality assurance programme, and acted as technical assessor for SWEDAC and was responsible for the implementation of accreditation in the NERI laboratory. On the scientific front, she was active in both EU-projects and NMR projects to the bitter end, and continued this work nearly to the end. All of these activities were supported by her good spirit and oversight, together with a warm and motherly feeling of responsibility for all who worked together with her. She is still missed.

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Abstract

Hazardous substances, both heavy metals and some man-made organic chemicals, are today widely distributed in the Baltic marine environment, and elevated levels of toxic contaminants are of concern, because they can pose a risk to sensitive organisms and the ecological structures and functions in the Baltic Sea.

The objective of this project was to compare and evaluate classification systems for assessing pollution with hazardous substances in the marine environment presently used by the countries surrounding the Baltic Sea and the Kattegat, with the aim to suggest a common strategy for classification. The intention is also to discuss an operational approach for the classification of our common sea areas, which can bring the current marine strategy using mainly biota and sediment more in line with the objectives of the status classes defined in the EU Water Framework Directive (WFD).

Classification of marine areas by statistical distribution approaches, as the five-class systems used in Sweden and Norway, are compared to ecotoxicological approaches used within OSPAR and WFD. In addition an alternative approach with five status classes derived from ecotoxicological threshold levels by integrating some of the OSPAR and/or WFD objectives and criteria is developed for evaluation of the environmental risks of hazardous substances in the Baltic Sea.

Since the Nordic countries have to comply with the WFD where the focus is on protection of biodiversity, it is our suggestion that an ecotoxicological approach should be included in the derivation of assessment criteria. It is furthermore suggested that the classification be based mainly on concentration levels in sediment and biota like the blue mussel *Mytilus edulis* and certain fish species and not on concentration levels in seawater, because water samples are generally regarded as a less suitable matrix for marine monitoring.

The comparisons and the evaluation of the different approaches are based on available data for tributyltin (TBT), cadmium (Cd) and polychlorinated bisphenyls (PCBs) as three examples of hazardous substances occurring in the Baltic Sea region.

The examples shows that especially the level of TBT contamination can be regarded as an environmental problem throughout the Baltic region, both based on data for TBT levels in sediment, mussels and TBT effects in gastropods. For Cd and PCB, elevated levels can in some examples be regarded as more local environmental problems, although it is also depending on which classification system used for assessing the risk of elevated contaminant levels.

1. Introduction

The environmental quality of our seas is generally evaluated on the basis of absence or presence of threats, which can affect the ecosystem structure and function, with the objective to achieve environmental conditions favouring the conservation of biodiversity. Contaminants are generally together with eutrophication-related problems and physical disturbance/exploitation recognised as the most important threats in assessments and evaluations of the environmental conditions in the marine environment.

Especially the high levels of persistent organic pollutants like DDT and PCB in the Baltic Sea during the 1970´ties and 1980´ties have been of concern. These substances were not only found to accumulate in the food web, but also related to high frequencies of reproductive disorders in seals, and eggshell thinning in fish-eating and predatory birds, thereby affecting the whole ecosystem. In the recent decades the DDT and PCB levels, as well as the observed effects in the marine top predators have declined (HELCOM, 2002). However, this does not necessarily imply that only insignificant levels of these substances occur locally in coastal waters today, or that other hazardous substances may not occur in levels which may be a threat to the ecosystem. Therefore there is an increasing interest from regional, national as well as international authorities to develop tools, which can be used for the evaluation of environmental risks posed by contaminants found in our seas today. Such tools are for instance necessary in relation to the implementation of the EU Water Frame Directive (WFD) (EU, 2000; 2001; 2006), since it includes binding obligations to assess the environmental quality.

This report includes comparisons between different kinds of national and international approaches for assessment and classification of the environmental conditions in the marine environment with respect to contaminants, which have been used in Nordic countries in the recent years. It is important to achieve a common understanding within the Nordic countries of the assessment and classification of the contamination levels in sea territories, because the coastal and open waters in the Baltic region are an interconnected system. Currently there are differences between the approaches used in the Nordic countries that could lead to differences in classification of the same areas.

The marine monitoring of contaminants has for several years been focussing on contaminant levels in sediment and biota samples (OSPAR, 2000; HELCOM, 2002), since these matrices are the most useful for spatial and temporal monitoring. Accumulation of contaminants in sediment and biota can provide stronger evidence of the general concentration levels in a restricted area, because function as a time-integrated measure of contaminant levels, whereas concentrations in seawater fluctuate more both within day-to-day and season-to-season. In addition, adequate detection limits for environmental relevant concentration levels can easier be achieved for hydrophobic compounds like organochlorines in sediment and biota because of their high affinity to particulate organic matter, and their high bioaccumulation potential.

Both Sweden (Swedish EPA 2000) and Norway (SFT, 1997) have developed national five-class systems of environmental assessment criteria based on the level of deviation from background levels, and with the focus on concentration levels in sediment and biota. In comparison, the international organisation OSPAR (Oslo-Paris convention) has suggested Ecotoxicologiocal Assessment Criteria (EAC) for contaminants in sediment and biota as a tool for assessing contaminant levels in the North Atlantic region (OSPAR, 1998). The EACs are derived from ecotoxicological threshold levels, which have been extrapolated from exposure levels in seawater to corresponding levels in sediment and biota. The intention was that the contaminant data for biota could be used in addition to water concentrations to assess the exposure level in situations where organisms at the lower trophic levels in the pelagic or benthic communities may be at risk. This was a first step to try and include the environmental risks of contaminant-induced effects on the marine ecosystem in the assessments of measured contaminant levels.

A similar, although not identical approach, has been introduced in relation to the EU Water Frame Directive (WFD) for classification of water bodies in Europe (EU, 2000). It also focuses at ecotoxicologically derived threshold levels, e.g. Environmental Quality Standards (EQSs) for priority substances in combination with biological quality elements. However, the WFD primarily focuses on contaminant concentrations in surface waters, and to lesser degree on concentration levels in sediment and biota, where only tentative QS-values are defined. QS-values for concentration levels in biota are derived if it assessed that the contaminants may pose a risk for top predators or humans due to secondary poisoning from intake of aquatic food sources (Lepper, 2002).

At a recent OSPAR workshop it has been suggested that also the Environmental Quality Standards in the WFD should be converted to corresponding accumulation levels in biota (OSPAR, 2004), as is the case for the EACs. One major argument for this conversion is that most contaminant data from the marine monitoring programmes in the Baltic Sea and the North Atlantic mainly consist of chemical analyses of concentration levels in sediment and biota samples (seaweed, molluscs and/or fish).

An extrapolation of assessment criteria from seawater to sediment and biota is therefore important and necessary, if this general strategy for marine monitoring of contaminants still should be operational in relation to the implementation of the WFD. If this is not done it could result in the destruction of existing time series, which first in the recent years have reached a length and thereby a statistical power for temporal trends to be assessed. An additional practical consideration is that it takes a much larger sampling regime to evaluate contamination levels using momentary sampling of water compared to integrated sampling of biota and sediments.

The different classification systems and their definition of protection levels, as well as the derivation of respective status classes are presented and compared in this report. The intention is to provide a basis for further discussions on harmonisation of assessment criteria between the Nordic countries around the Baltic Sea and the Kattegat, which may become relevant in relation to the future implementation of the WFD in marine systems.

The assessment criteria will be discussed using data for three types of hazardous substances (Cd, TBT and PCB) in biota and sediment from different regions in the Baltic Sea, the Kattegat and the Skagerrak, which have been extracted from national and regional monitoring programmes and from the scientific literature available.

1.1 Cadmium, TBT and PCB as model substances

Three different kinds of contaminants, cadmium (Cd), tributyltin (TBT) and polychlorinated biphenyls (PCB), have been chosen as model substances for this project. They represent three groups of substances with different physical-chemical characteristics, which will affect how the assessment criteria can be derived due to variation in fate and toxicity of these compounds in the marine environment. TBT and Cd are both on the priority lists of the WFD (EU, 2001;2006) and OSPAR (OSPAR, 1998), whereas PCBs only are categorised as a priority substance by OSPAR. However, these contaminants have all been identified as substances, which should be paid special attention to in the Baltic region, because of elevated levels and their possible environmental implications (HELCOM, 2002).

Cd belongs to the heavy metals, which also occurs naturally in the marine environment, but anthropogenic inputs of Cd have also contributed significantly to the elevated levels in the region (Szefer, 2002). Salinity seems to be one of the major factors, which affect the bioavailability and toxicity of Cd. Differences between the natural background levels in high and low saline areas of the Baltic region therefore have to be considered in the derivation of assessment criteria.

TBT is an organometallic compound, which is less hydrophobic ($K_{ow} \approx 3$) than PCB. TBT has been identified as a high-risk compound especially for invertebrates such as molluscs. Because TBT has been widely used as anti-fouling agent in ship paints, elevated TBT levels can be found in the Kattegat and the Baltic Sea due to the intense commercial ship traffic in this region. In addition, the data of ambient TBT concentrations can be combined with the occurrence of TBT-specific biological effects, e.g. imposex and intersex, in marine gastropods (Strand, 2003; 2006), which also is included in this analysis.

PCB is a group of persistent and highly hydrophobic organic pollutants (K_{ow} >5), which tend to accumulate through the food web, and where the risk of secondary poisoning has to be considered. The Baltic Sea has been identified as an area with elevated PCB levels, especially in the 1970´s and 1980's, where also the reproduction and health of top predators like Baltic seals and birds were highly affected (HELCOM, 2002). PCB congener CB153 is in this analysis regarded as an appropriate tracer of the total level of PCB contamination in mussels and fish in the marine environment. The sum of seven PCB congeners ($\Sigma PCB_{7;}$ CB28, 52, 101, 118, 138, 153 and 180) can as an alternative also be used.

1.2 Data and data handling

The assessment criteria will be discussed using available national and regional monitoring data and data from some surveys (mainly from Poland, Germany and Baltic states) published in the scientific literature

Data for concentration levels in biota, e.g. bladder wrack (*Fucus vesiculosus*), blue mussel (*Mytilus edulis*) and some fish species, are preferred in this project, since these kinds of data dominate the marine data available for the Baltic region, and because they are included in some of the presented assessment criteria. It is recommended not to transfer these assessment criteria to related taxonomic species, as there can be significant differences in uptake, even though they are at the same trophic level.

Concentrations in seawater are not considered as first priority in this analyses, because water samples are not the preferred sample type within Nordic monitoring programmes, due to the low concentrations in seawater, and because individual water samples only provide a short-term measure of the pollution level that can fluctuate significantly over time. Subsequently, only few relevant data on concentration levels in seawater exist. In contrast, biological samples and sediment dominate the existing data material, and it also regarded as a more time-integrated measure of the contaminant levels. A future alternative can be the inclusion of passive sampling devices on a routine level for marine monitoring.

1.3 Normalisation parameters

Normalisation of data is an important factor when comparing contaminant levels over time and geographic scale. Thereby the variability caused by matrix effects such as water content and lipid content are reduced and the data therefore become more comparable.

Average values of the relevant normalisation parameters (i.e. dry weight and lipid content) for mussels and fish have been extracted from Nordic monitoring data (Table 1.1.) and used to extrapolate/convert to normalised concentration levels in situations where the relevant normalisation parameters not are available.

Table 1.1. Conversion factors for normalisation from wet weight (ww) to dry weight and lipid content in blue mussel and various fish species. The values (average ± SD) are extracted from Nordic monitoring data (NERI, 2004; IVL, 2004; NIVA, 2000).

Normalisation parameter	Blue mussel	Fish liver	Fish muscle
Dry weight content (of ww)	16 ± 4.5%	25 ± 3.9% (cod: 55 ± 11%)	23 ± 2.3%
Lipid content (of ww)	1.1 ± 0.6%	15 ± 7.4% (cod: 45 ± 14%)	3.3 ± 4.5%

1.3.1 Normalisation of TBT data

TBT concentrations in tissue are generally normalised to wet weight (ww) or dry weight (dw) content depending on the study. Many studies list the TBT concentration based on wet weight since the chemical organotin analyses often are performed on wet tissue samples Concentrations in sediments are mainly normalised to the dry weight content. In this report we recommend to use the dry weight content as normalisation parameter for TBT concentrations in tissue as well as in sediments t (unit: µg Sn/kg dw).

Lipid content as normalisation parameter is not recommendable since TBT has a higher affinity for protein. Note that the unit is referring to Sn and the concentration has to be multiplied with factor 2.45 to be converted to µg TBT/kg dw.

1.3.2 Normalisation of Cd data

Cd concentrations in tissue as well as in sediment are generally normalised to the dry weight content (unit: mg Cd/kg dw) as analyses of Cd are generally performed on freeze-dried material.

The content of organic matter, measured as total organic carbon (TOC) or the content of lithium (Li) can also be relevant normalisators for Cd levels in sediment.

For other metals, the content of aluminium (Al) can be a more relevant normalisator than Li.

1.3.3 Normalisation of PCB data

The congener CB153 is in this project used as tracer for PCB levels in biota, and the lipid content is used as normalisation parameter (unit: mg CB153/kg lipid). PCB153 is the dominant congener and is therefore the most likely congener to be above the detection limit, which is not the case for some of the other PCB congeners. Alternatively, the sum of seven PCBs ("the Dutch seven") can be used. Average ratios between CB153 and ΣPCB_7 in Baltic mussels and fish are listed in Table 1.2.

Lipid content is often used as normalisation parameter, because of the strong correlation to the lipid content, especially in fish. However, normalising to lipid content in mussels may introduce an additional uncertainty/variability because of the relatively low lipid contents in mussels. Dry weight content may be a better normalisation parameter at low lipid contents.

In some situations it can also be relevant to normalise PCB concentrations to wet weight (ww), when secondary poisoning is regarded as the relevant endpoints to address in the derivation of assessment criteria.

Table 1.2. Ratios between CB153 and ΣPCB7 in mussels and fish from the Baltic Sea. The values (average ± SD) are extracted from Nordic monitoring data (NERI, 2004; IVL, 2004).

Ratio	Blue mussel	Fish
CB153/ ΣPCB_7 ratio	0.40 ± 0.10	0.34 ± 0.03

1.4 Detection limits

To evaluate "concentrations close to zero" for synthetic substances and "background levels" for metals, it is important for marine monitoring programmes to have access to sensitive analytical methods with appropriate detection limits. For comparison the detection limits for CB153, TBT and Cd used in Nordic marine monitoring programmes in Denmark, Sweden, Norway and Finland are listed in Table 1.3.

Table 1.3. Detection limits for PCB, TBT and Cd in seawater reported for Nordic marine monitoring programmes (NERI, 2004; IVL, 2004; NIVA, 2000; SYKE, unpubl.).

Substance	Seawater	Biota	Sediments
Cd	1–30 ng Cd/l	0.05–0.1 mg Cd/kg dw	0.001–0.03 mg Cd/kg dw
TBT	1 ng Sn/l	0.005–0.01 mg Sn/kg dw [1]	0.001–0.005 mg Sn/kg dw
CB153	0.1–1 ng CB/l [2]	0.01–0.05 mg CB/kg lipid [1]	0.05–0.2 µg CB/kg dw [1]

[1] Converted from µg/kg ww [2] It is estimated that for CB153 concentration levels in the range of pg/l can be achieved by using passive sampling devices. Otherwise detection limits about 1 ng/l are achievable.

2. The different types of environmental assessment criteria available

This chapter presents an overview of the different kinds of national and international assessment criteria for hazardous substances, which have been used for assessing contaminant levels in the marine environment in the Nordic countries.

In addition the approach for derivation of ecotoxicological assessment criteria by OSPAR (1998), and Environmental Quality standards in the EU Water Frame Directive (WFD) derived by the Frauenhofer Institute and the Expert Advisory Forum on the WFD priority Substances (Lepper, 2002; EU, 2005a,b) will be presented, as well as the combined OSPAR/WFD approach recently suggested by OSPAR (2004), which intends to link the objectives in the WFD with assessment criteria based on concentration levels in biota. Finally, a new suggestion for assessment criteria for TBT and PCB will be proposed, the so-called Ecotoxicological approach. It combines the five-class approaches previously used in the Nordic countries and the principles in the combined OSPAR/WFD approach.

2.1 Sweden

In Sweden a five-class system of environmental quality criteria for coastal and open sea waters has been derived for various metals (As, Cd, Cr, Cu, Hg, Ni, Pb, Sn and Zn) as well as for organic pollutants, i.e. different organochlorines and PAHs (Table 2.1). These criteria cover concentration levels in sediment and in five different organisms belonging to macroalgae, mollusc and fish taxa (see also Appendix E).

Table 2.1. The statistical approach used to derive the five-class system of Swedish quality criteria for the conditions of organic pollutants and heavy metals (Swedish EPA, 2000). Px is the x-th percentile of the dataset, x is 5, 95 or 99.

	I	II	III	IV	V
Synthetic organic pollutants	0	< P_5 of reference data	$< P_5 \cdot \sqrt{\frac{P_{95}}{P_5}}$	III > and < V	> P_{95} of percentile of all data
Metals	< P_5 of reference/off-shore data	$< P_5 \cdot \sqrt[3]{\frac{P_{95}}{P_5}}$	$< P_5 \cdot \left(\sqrt[3]{\frac{P_{95}}{P_5}}\right)^2$	III > and < V	> P_{95} or P_{99} of all data

The concentration ranges related to the five status classes have been derived by a statistical distribution approach and the actual ranges are therefore dependent on the data material included in the statistical analyses. However, the concentration of man-made substances is defined as “nil” in status class I to reflect the objectives of OSPAR and HELCOM (se below). No ecotoxicological interpretations have been included in the derivation of the status classes.

For heavy metals including Cd, two sets of quality criteria have been defined both for the low-saline Baltic Sea and the high-saline Kattegat-Skagerrak area. The regional borders are defined to cross the Sound at Drogden, the Great Belt at Sprogø and the Little Belt at Middelfart (Swedish EPA, 2000), which reflect a threshold in salinity of about 15 psu.

2.2 Norway

In Norway a five-class system of assessment criteria have been derived as a practical tool useful for classification of the environmental quality in fiords and coastal waters (Table 2.2, SFT, 1997). The system was first derived in 1992 and updated in 1997, and they are scheduled to be updated again in 2008.

These quality criteria have been derived for heavy metals (As, Pb, Cd, Cu, Cr, Hg, Ni, Zn, Ag) and organic pollutants, i.e. TBT, organochlorines and PAHs in sediment and in seven different organisms belonging to macroalgae, invertebrates and fish. Assessment criteria for metals in seawater have also been derived (see also Appendix D).

Table 2.2. The Norwegian assessment criteria for classification of the quality due to contaminant levels in the marine environment (SFT, 1997).

	I	II	III	IV	V
Organic pollu-tants & metals	Insignificant contaminated Reference value in areas without inputs from local sources	Moderate contaminated due to input from local sources. Expert judge-ment	Marked contaminated Expert judge-ment	Severe contamina-ted Expert judgement	Extreme contaminated Expert judge-ment

Status class I is intended to reflect background levels of contaminants (metals as well as man-made substances) in areas with diffuse long-range input. The following status classes reflect increasing contaminant levels, which can be ascribed to input from local sources. Expert judgement with knowledge of Norwegian monitoring data has been used to define the concentration ranges related to the respective status classes.

It has been possible to make some coupling between the status classes related to elevated contamination level and effects on benthic communities and human health risks for TBT in mussels and mercury (Hg) in fish, respectively.

2.3 Denmark

No national assessment criteria have been derived in Denmark. However, the Ecotoxicological Assessment Criteria (EAC) derived by OSPAR have mainly been used in the regional and national assessments of monitoring data of organic pollutants in sediment and biota, whereas the levels of heavy metals mainly have been assessed by the Norwegian quality criteria (NERI, 2002).

2.4 Finland

There are no statutory assessment criteria for Cd, TBT or PCBs in Finland. However, a recent Finnish study used the Swedish quality standards for heavy metals in sediments (Swedish EPA, 2000) to assess the contaminant levels in the Gulf of Finland in the northern part of the Baltic Sea (Vallius & Leivuori, 2003).

Tentative thresholds, i.e. action levels, have been determined for dredging of sediments (Table 2.3, see also Appendix C). Action level 1 represents a level below which the sediments are considered clean. Action level 2 represents a level above which the sediments are considered contaminated and should not be dredged. Between levels 1 and 2 the sediments re considered possible contaminated and a case by case judgement should be done.

Table 2.3. Proposed action levels for Cd, TBT and PCB153 in dredged materials in Finland (HELCOM 2004).

	Action level 1	Level 2
Cadmium	0.5 mg/kg dw	2.5 mg/kg dw
TBT	3 µg/kg dw	200 µg/kg dw
PCB153	4 µg/kg dw	30 µg/kg dw

2.5 HELCOM

The Helsinki Commission (HELCOM) is covering the Baltic Sea and the Kattegat area. HELCOMs objective with regard to hazardous substances is identical with OSPARs objective, e.g. to prevent pollution of the convention area by continuously reducing discharges, emissions and losses of hazardous substances towards the target of their cessation by the year 2020. The ultimate aim is to achieve concentrations in the environment near background values for naturally occurring substances and close to zero for man-made synthetic substances.

HELCOM has not yet recommended any defined criteria for assessing the contaminant levels in the region, although the Finnish national quality criteria (level 1 and level 2) for dredged spoils has been presented (HELCOM, 2004) (see also Appendix C).

2.6 OSPAR

The Oslo-Paris Convention (OSPAR) covers the North Atlantic including the North Sea, the Skagerrak and the Kattegat. The objective of OSPAR is "making every endeavour to move towards the target of the cessation of discharges, emissions and losses of hazardous substances by the year 2020." This includes prevention of pollution in open seas. Therefore the target is to achieve concentrations of man-made priority substances close to zero, or at least below the limits of detection of the most advanced analytical techniques in general use. For naturally occurring substances such as many metals and PAHs the objective is to achieve concentration levels, which do not deviate from the background levels.

The approach taken to derive the Ecotoxicological Assessment Criteria (EACs) is mainly based on estimated Predicted No Effect Concentration (PNEC) from available ecotoxicological data (OSPAR, 1998). Generally the lowest NOEC or LC_{50} available have been used by applying an assessment factor between 10 and 1000 depending on the amount of toxicity data available and the type of endpoints used. The EACs are defined as a range; EAC (low) – EAC(high), which covers an order of magnitude in concentrations level around the lowest PNEC-value to account for the uncertainties in these kind of extrapolations. In a later discussion of the

EAC within OSPAR working groups the upper limit of EAC has been recommended to be used as basis for marine assessments. EAC for metals and some organic contaminants have been derived for concentration levels in seawater and sediment, and in some cases EACs for the organic pollutants in mussels and fish have also been derived (see also Appendix A).

Most sediment EACs are still based on equilibrium partitioning principle, due to insufficient ecotoxicological data. Such EAC are regarded as provisional and need validation with additional sediment toxicity tests and/ or co-occurrence data like in the Canadian TEL approach.

For biota, three different types of EAC can be derived. The first type is based on the derived EAC for water or sediment, and transferred to biota using an appropriate bioconcentration factor (BCF) or biomagnification factor (BMF). The second type takes secondary poisoning into account since fish or mussels are food for predators. Levels in mussel or fish can be derived in order to protect against this so-called secondary poisoning. The third type of EAC is derived by comparing critical body burdens to accumulated contaminant levels where significant effect have been observed in field or laboratory studies. However, data of critical body burdens are very seldom available. It is recommended to calculate the different types of biota EACs for comparison if possible. The EAC-values are recommended by OSPAR to be used as guidance for assessing contaminant levels but with no defined obligations.

At a recent OSPAR workshop (OSPAR, 2004) it has been suggested that the EAC from OSPAR (1998) should be harmonised with the quality standards derived within the framework of WFD (see paragraph 2.8).

2.7 EU. The Water Frame Directive (WFD)

In the European Community Water Framework Directive (2000/60/EC, WFD) both the chemical and biological status of the aquatic environment should be assessed by using both physical-chemical and biological quality elements (EU, 2000). As part of the WFD, 33 priority substances have been identified (EU, 2001), which partly overlap the OSPAR priority list, on the basis of their risk to the aquatic environment, or to human health via the aquatic environment.

The objective in the WFD is to prevent deterioration of surface waters and achieve good ecological and chemical status for freshwaters, estuaries and territorial waters. In addition also a high status class is defined for the chemical elements as concentrations of man-made substance close to zero (or at least below the detection limits) and concentrations which do not deviate from background levels for naturally occurring substances such as metals and PAHs. However, the high status class must not necessarily be achieved. In the upper part of the assessment, three other status classes, called moderate, poor and bad, are defined in the WFD, but these are entirely based on biological elements, which ascribed to changes in community structures and functions.

Table 2.4. Physical-chemical quality elements for specific synthetic pollutants for classification of water bodies. Definitions from WFD, annex V to the directive 2000/60/EC.

High status	Good status	Moderate status
Synthetic pollutants		
Concentrations close to zero or at least below the limits of detection of the most advanced analytical techniques in general use.	Concentrations not in excess of the standards set in accordance with the procedure detailed in section 1.2.6. of the WFD and described in Lepper (2002), e.g. concentrations below EQS and below MAC-QS.	Conditions consistent with the achievement of the values specified for the biological quality elements.
Natural occurring pollutants		
Concentrations close to background levels.	Concentrations not in excess of the standards set in accordance with the procedure detailed in section 1.2.6. of the WFD and described in Lepper (2002), e.g. concentrations below EQS = background + MPA	Conditions consistent with the achievement of the values specified for the biological quality elements.

For the priority substances good chemical status is achieved if the concentration is below the Environmental Quality Standards (EQS) (Table 2.4). The EQS is designed to protect all species against adverse effects caused by long-term exposure such as impaired growth, reproduction, and behaviour, or in other way affect their survival since such effects may result in alteration of ecosystem structure and function. The EQS is intended to reflect the annual average concentration in water, which must not be exceeded to achieve good status. The derivation of EQSs is based on an approach using ecotoxicological data for estimation of Predicted No Effect Concentration (PNEC) as is the case for the OSPAR EACs (OSPAR, 1998).

However, following recent developments in the methodology of risk assessment for the marine environment (TGD, 2003), other assessment factors were used, and also statistical methods has been adopted in the WFD for the derivation of Quality Standards. In the WFD, the ecotoxicological data material used for deriving PNEC-values may also not be identical with the data used in the OSPAR assessment.

In the WFD, a Maximum Acceptable Concentration Quality Standard (MAC-QS) is also included in the description of good ecological status, where MAC-QS should not to be exceeded in peaking episodic events, because of the risk of adverse effects caused by short-term exposure in the marine ecosystem. The MAC-QS is generally based on lowest acute ecotoxicological data like LC50-values and defined assessment factors designed to protect against short-term episodic events of contaminant exposure (Lepper, 2002; TGD, 2003).

In the WFD, EQSs are also derived for sediments and biota. EQS for sediments should protect benthic communities and is mainly derived based on the EQS for water and the equilibrium partitioning principles due to a generally lack of sediment toxicity data. These values are therefore mainly regarded as tentative values (see also Appendix B).

Quality standards (QS) for biota are only derived when secondary poisoning is assessed to be a risk for top predators in the marine food web such as birds and mammals ($QS_{toppred.}$) as well as for humans ($QS_{hum.}$). For persistence substances with a high potential for biomagnification (K_{ow} > 4.5), these QS-values are used to derive the corresponding EQS for seawater using BCF- and BMF-values from literature, or by using default factors defined in the TGD (2003). However, these BCF- and BMF-values are never used the other way around to derive quality standards for biota based on the EQS (and MAC-QS) for seawater derived for protection of the pelagic and benthic communities.

2.8 OSPAR-2004 approach.

A potential coupling of WFD and OSPAR strategies?

At a recent OSPAR workshop on evaluation update and use of background reference concentrations and ecotoxicological assessment criteria, (OSPAR, 2004), it was suggested that the EAC from OSPAR (1998) (see paragraph 2.5) should be harmonised with the quality standards derived within the framework of WFD (see paragraph 2.7). The intention is that the existing OSPAR and HELCOM strategies for marine monitoring of contaminants, mainly in biota and sediment, can be integrated in the future assessments and evaluations of the conditions in the marine environment. It has to be noticed that these new assessment criteria, called Environmental Assessment Criteria (also EAC) were not accepted by OSPAR in 2006, so they may still be revised.

These assessment criteria imply that the EQS and the MAC-QS for contaminants in seawater can be extrapolated to corresponding concentration levels in mussels and even fish using the same bioconcentration factors used for derivation of the previous EAC (OSPAR 1998). If such values not are available, as is the case for brominated flame retardants, default bioconcentration and biomagnification factors defined in TGD (2003) should be used. The status classes I, II and III refer to the chemical quality standards defined in the WFD, e.g. near zero concentration and EQS and the status classes IV is derived on basis of the MAC-QS-value.

However, it has to be emphasised that all thresholds in this scheme refer to mean concentrations and not exposure level, which can cause acute effects in short-term episodic events. Addressing acute effects by mean concentrations as in status class III and IV may therefore not be that consistent, but the definition of the status classes should rather be seen as a potential tool useful for the interpretation of monitoring data and the development of monitoring and assessment strategies.

2.9 Five-class ecotoxicological approach – a alternative suggestion

This approach is a modification of the suggestions made at the OSPAR workshop 2004 (see 2.8). In this approach a fifth status class (V) is also included, so that the derived assessment criteria can be compared with the five-class system of assessment criteria, which has been used in Sweden (paragraph 2.1) and Norway (paragraph 2.2). This fifth status class corresponds to concentration levels above the LC_{50}-value from which the MAC-QS-value was derived. For further definitions of the five status classes, see Table 2.5 and 2.6.

Table 2.5. Description of the principles for derivation of a five-class system of assessment criteria according to the Ecotox. approach (this study).

Status class I: High status. Concentration of priority substances is close to zero and at least below the limits of detection of the most advanced analytical techniques in general in use. This reflects the objectives in the WFD and OSPAR strategy of hazardous substances for protection of the open waters.
An assessment of the achievement of the objective also requires that no biological effects at the individual as well as population level, which can be related specifically to exposure to the priority substances, can be detected. E.g. the response must not be significantly different from the natural background level.
Status class II: Good status. Concentration of priority substances is not in excess of the chemical quality standards, e.g. below the Environmental Quality standard (<EQS = <PNEC).
Adverse effects in the most sensitive species caused by long-term exposure are predicted to be unlikely to occur.
Status class III: Moderate status. Concentration of priority substances is not in excess of the so-called maximum admissible concentration quality standard ($<$MAC-QS = $<1/10 * LC_{50}$), but above the EQS.
Moderate deviations of biological communities may occur, because there is a risk of adverse effects caused by long-term exposure in the most sensitive species. However, adverse effects in the most sensitive species caused to short-term exposure are predicted to be unlikely to occur in the marine ecosystem.
Status class IV: Poor status. Concentration of priority substances is not in excess of lowest observed LC_{50}-value for the most sensitive species ($<LC_{50}$) but above $1/10 * LC_{50.}$
Substantial deviations of biological communities can occur, because there is evidence of adverse effects caused by long-term exposure in the more and maybe also in the less sensitive species. In addition, there is a risk of adverse effects caused by short-term exposure in the most sensitive species.
Status class V: Bad status. Concentration of priority substances is in excess of lowest observed LC_{50}-value for the most sensitive species ($> LC_{50}$).
Severe alterations of biological communities occur due to adverse effects caused by long-term exposure in the more and less sensitive species. In addition, there is a risk of adverse effects caused by short-term exposure in both the more and less sensitive species.

Table 2.6. Overview of the principles for derivation of a five-class system of assessment criteria according to the Ecotox. approach.

I	**Close to zero or background for metals and PAH**	
II	<EAC/EQS (<PNEC)	Adverse chronic effects are unlikely to occur
III	<MAC-QS (<LC50/10)	Risk of chronic effects, but acute effects are unlikely to occur
IV	<10*MAC-QS	Risk of acute effects
V	>10*MAC-QS	Acute effects are likely to occur

The ecotox. approach for deriving a five-class system of assessment criteria has in this analysis been applied for TBT and PCB only. For TBT it has also been possible to link the five-class system for TBT concentrations in different matrices with data of TBT-specific biological effects in marine gastropods. For PCB, it has been more difficult since these compounds are not yet included in the recent risk assessments, neither by EU (2006) or OSPAR (2004). In the derivation of status classes, we have decided to include the knowledge of environmental problematic PCB levels for marine top predators in the Baltic Sea in the 1970ties and early 1980ties as a valid evidence of the risk of chronic effects caused by long term exposure to PCB. It has not been possible to find adequate data on chronic and acute toxicity of Cd in marine species in the risk assessments by OSPAR (1998) and EU (EU, 2005b).

3. Comparison and discussion of different national and international assessment criteria for TBT, Cd and PCB

3.1 Assessment criteria for tributyltin (TBT)

TBT is a synthetic substance, which does not occur naturally in the marine environment. The general objective within OSPAR and HELCOM is therefore to achieve concentration close to zero, which can be linked to the definition of status class I.

TBT is highly toxic in the marine environment and molluscs are identified to belong to the most sensitive species with NOEC-values at 1 ng TBT/l, which has been used to derive both the EAC and the EQS at 0.1 and 0.2 ng TBT/l, respectively (OSPAR, 1998; EU, 2006). In addition the Expert Advisory Forum on Priority Substances under the WFD has derived a MAC-QS value of 1.5 ng TBT/l based on acute toxicity data for a pelagic crustacean (EU, 2005a). These values have in the approach by OSPAR (2004) provided the basis for the extrapolated assessment criteria suggested for TBT in mussels and in sediment.

3.1.1 Assessment criteria for TBT in blue mussel (Mytilus edulis)

The accumulation of TBT in *M. edulis* is relative high, although other species of bivalves may have even higher accumulation potential (Langston 1996, Strand et al. 2003). OSPAR has used a geometric mean value of BCF = 116.000 l/kg dw in *M. edulis* (OSPAR, 1998) to derive the EAC of 4 µg Sn/kg dw for TBT based on an extrapolation from EAC of 0.1 ng TBT/l in seawater.

In the derivation of the Norwegian assessment criteria, a BCF = 10.000 l/kg (SFT, 1997), i.e. an order of magnitude lower than in the OSPAR approach, has been used to extrapolate TBT-concentrations assigned to each status class. In addition a Norwegian quality standard of 1 ng/l has been used for derivation of the lowest status class. These are the main reasons for the large difference between Norwegian and the assessment criteria suggested by OSPAR (1998, 2004). Sweden has not derived any quality criteria for TBT.

In the work under the WFD no quality standards for TBT in mussels have been derived for protection of top predators. However, the WFD has found it relevant to derive a quality standard for TBT that refers to food uptake of (shell)fish by humans, QS_{human} of 15 µg TBT/kg ww (corresponds to 30 µg Sn/kg dw), which is based on Tolerable Daily Intake (TDI) of 0.25 µg TBT/kg bw/day (WHO, 1990; Belfroid et al., 2000). It is also assumed an average fish intake of EU citizens at 115 g/day, and that the average daily intake (ADI) is not exhausted for more than 10% by consumption of food originating from aquatic sources (EU, 2005a).

In the ecotoxicological approach (see Table 2.5) the BCF of 116.000 l/kg dw has been used in the derivation of thresholds for all the status classes II – V, which are extrapolated from EAC, e.g. EQS, and MAC-QS in the WFD for TBT in seawater.

Table 3.1. Comparison of assessment criteria for TBT in M. edulis.

Blue mussels µg Sn/kg dw	I	II	III	IV	V
Sweden	not derived				
Norway [1]	<40	40–200	200–800	800–2000	>2000
OSPAR (1998) [1]	close to zero	0.4–4			-
WFD	close to zero	<30 [2,3]			-
OSPAR (2004)	<0.4	0.4–<4	4–<60	60–600	>600 [4]

[1] Converted from µg TBT/kg dw to µg Sn/kg dw. [2] Converted from µg TBT/kg ww to µg Sn/kg dw
[3] Human health at risk. [4] Status class V derived according to the ecotox. approach, e.g. >10*MAC-QS*BCF

The assessment criteria by OSPAR (2004) in *M. edulis* have been used to classify the TBT contamination in the Baltic Sea, Kattegat and the Skagerrak in Figure 3.1 and 3.2. The figures illustrate that the level of TBT in *M. edulis* from the Baltic Sea, the Kattegat, and the Skagerrak indicates an environmental risks of the TBT contamination, and should be of high concern. All areas can be classified as status class III or IV, where chronic or even acute effects in sensitive organisms may occur. Many point sources like harbours can even be classified as status class V, where acute effects are likely to occur.

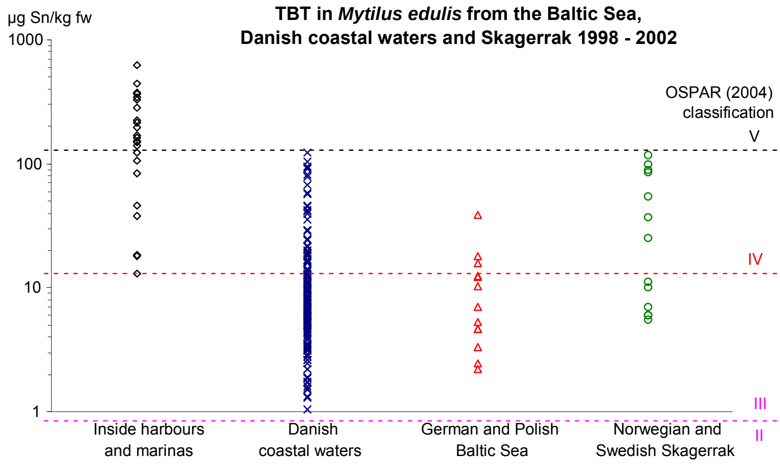

Figure 3.1. Comparison of TBT levels in blue mussels from different regions of the Baltic Sea, the Kattegat and the Skagerrak.

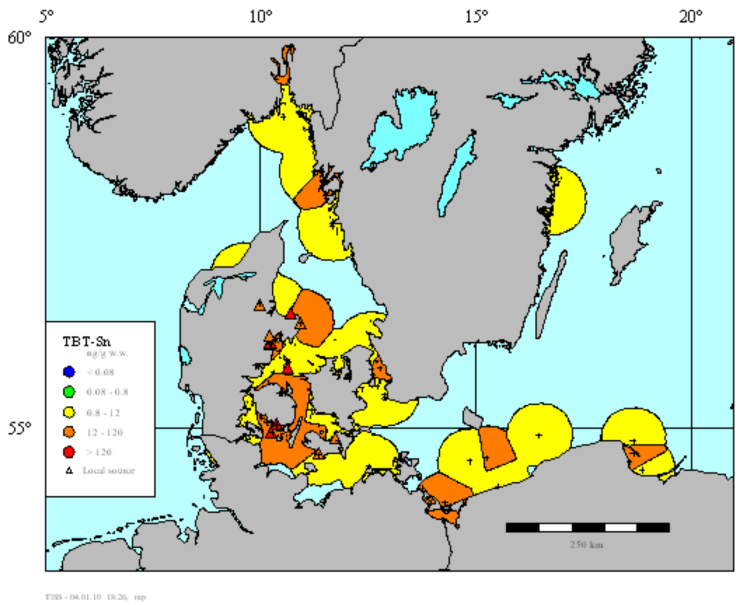

Figure 3.2. Classification of TBT-levels in the Baltic Sea, the Kattegat and the Skagerrak based on TBT levels in the blue mussel M. edulis according to the ecotox. approach by OSPAR (2004).

There exists only one recent study on TBT from Finland using the mussel *Macoma baltica* as a bioindicator that report TBT concentrations between 250–300 µg Sn/kg dw at reference sites and even 500–10000 µg Sn/kg close to harbours, which would place these areas to be in status class IV or even V. However, this comparison with assessment criteria based on TBT levels in *M. edulis* must be treated with a great deal of caution due to the high variability in accumulation potential between different species of mussels.

3.1.2 Assessment criteria for TBT in fish

Neither OSPAR nor the WFD have found it relevant to derive assessment criteria for TBT in fish because the risk for secondary poisoning of top predators in the food web is regarded as small, as TBT is relatively easily metabolised in most vertebrates. However recent studies have demonstrated relatively high exposure and accumulation of TBT (and breakdown products) particularly in coastal cetaceans (Tanabe, 1999; Strand, 2003). Therefore the risk for top predators should probably be reconsidered.

However, in the WFD it was found relevant for protection for human health to derive a quality standard for TBT referring to food uptake of (shell)fish by humans, QS_{human} = 15 µg TBT/kg ww corresponding to 30 µg Sn/kg dw (EU, 2005a). However, human risks of TBT and its breakdown products should also be considered (Belfroid et al., 2000).

3.1.3 Assessment criteria for TBT in sediment

In the approaches suggested by OSPAR (1998; 2004) and WFD (EU, 2005a), the assessment criteria for TBT in sediment can be derived by using the equilibrium partitioning coefficient for TBT, K_p. OSPAR has derived EAC-values by using K_p = 400 l/kg dw for sediment, whereas K_p = 1080 l/kg dw is used in the WFD. The difference in K_p is due to the different assumptions of the physical characters of the sediment used. K_p = 400 l/kg dw has also been used in OSPAR (2004) to derive the five status classes for TBT in sediment.

However, these values are only recognised as tentative values since they are not based on sediment toxicity data for sediment-dwelling organisms. In comparison such data has been used in Strand (2003) to derive a five-class system of assessment criteria from NOEC of 10 µg Sn/kg dw and a lowest observed LC50 of 200 µg Sn/kg dw. The TBT concentrations assigned to these assessment criteria are two orders of magnitude higher than the TBT concentrations derived using the equilibrium partitioning principle. Subsequently the assessment criteria based on the equilibrium partitioning principle are perhaps overprotective.

Table 3.2. Comparison of assessment criteria for TBT in sediment.

Sediment µg Sn/kg dw	**I**	**II**	**III**	**IV**	**V**
Sweden			not derived		
Norway [1]	<0.4	0.4–2	2–8	8–40	>40
OSPAR (1998) [1]	close to zero	0.002–0.02			-
WFD	close to zero	<0.004			-
OSPAR (2004)	close to zero	<0.004	0.004–0.06	0.06–0.6	>0.6
Strand (2003)	close to zero	<0.5	0.5–<20	20–<200	>200

[1] Converted from µg TBT/kg dw to µg Sn/kg dw.

It should also be noticed that the TBT concentrations in sediment assigned to status class II by OSPAR (1998; 2004) and in the WFD is far below the detection limit of TBT at 1 µg Sn/kg dw, which is achievable at present. It raises the question whether sediment is a suitable matrix to use in assessments of the environmental conditions for TBT.

The Norwegian assessment criteria are as for TBT in *M. edulis* orders of magnitude higher than the assessment criteria suggested by OSPAR (1998; 2004) or in the WFD. However, they are more in line with the assessment criteria based on sediment toxicity data suggested by Strand (2003).

The amount of data for TBT in sediment from coastal and open waters (i.e. outside harbours) in the Baltic region is limited. The majority of data is from the Danish waters, and only few data from the Swedish and Finnish coast have been found (Figure 3.3). The TBT levels in sediment from coastal waters can in many situations characterised as status class III or even IV, when using the assessment criteria suggested by Strand (2003). This is in line with the classification based on the OSPAR (2004) approach for assessment criteria for TBT in mussels (see Figure 3.1 and 3.2). The TBT concentration is only in sediment from open parts of the Kattegat and Skagerrak below the detection limit of 1 µg/kg dw, which may indicate that these areas can be classified below status class III.

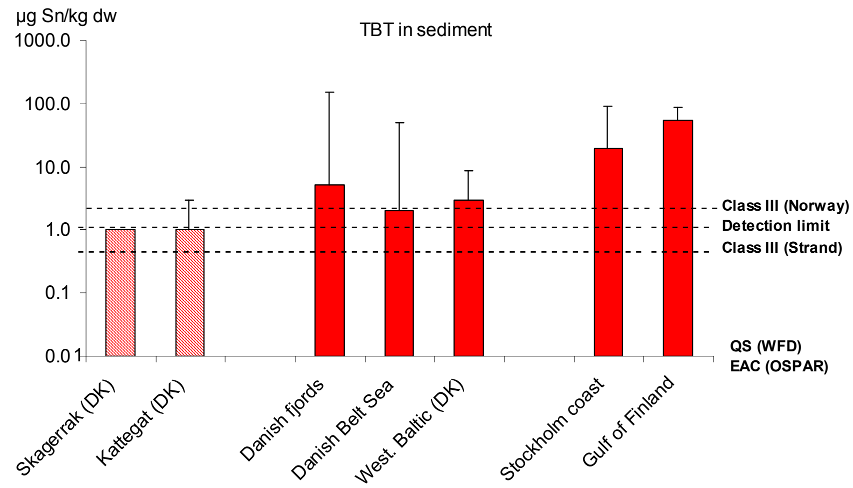

Figure 3.3. Comparison of TBT-levels (median + max.) in sediment from different regions of the Baltic Sea, the Kattegat and the Skagerrak (outside harbours). The TBT concentrations in Kattegat and Skagerrak are generally below the detection limit.

3.2 Combining levels of exposure and biomarker responses in the derivation of a five-class system of assessment criteria for TBT

A suggestion for a five-class system of assessment criteria for TBT-specific effects in five species of more or less TBT sensitive marine gastropods have also been suggested (OSPAR, 2003a). TBT is an endocrine disruptor, which induces imposex and intersex i.e. masculinisation of females, in prosobranch gastropods. These phenomena can be observed in several species inhabiting the Skagerrak, the Kattegat and the western Baltic Sea with salinity higher than 15 psu. *Buccinum undatum* and *Neptunea antiqua* can mainly be found at depths more than 10 m, whereas *Nucella lapillus*, *Littorina littorea* and *Hinia* (*Nassarius*) *reticulata* can be found in shallow waters or even in the tidal zone. *N. lapillus* has been used in the Norwegian monitoring programme, *H. reticulata* in Sweden and all five species in Denmark.

The different species not are equally sensitive to TBT. *N. lapillus* and *N. antiqua* for example have the highest likelihood to develop imposex. Subsequently the imposex levels (listed as VDSI-values) assigned to each status class must differentiate according to the species. These criteria presented in Table 4.2 are derived so that they are in line with the assessment criteria for TBT concentrations in seawater according to the ecotoxicological approach (see Table 2.5). They have thereby the potential to supplement each other in a combined assessment, which integrates concentration and effect levels of TBT.

It has to be stressed that significant chronic effects, e.g. impaired reproduction caused by sterile females, first are achieved in status class IV in this scheme. Imposex development should in Status classes I, II and III mainly be considered as a sensitive biological tool to assess the risk of TBT causing adverse effects in sensitive species in general. The level of imposex development in status class II indicates TBT level below the EQS = 0.1 ng TBT/l, whereas status class III indicates TBT concentrations between the EQS and MAC-QS.

Table 3.2. Assessment criteria for TBT combing TBT concentrations in water, mussel (Mytilus edulis) and sediment and TBT-specific biological effects, e.g imposex (as VDSI) and intersex (as ISI), in five species of prosobranch gastropods (Nucella lapillus, Littorina littorea, Hinia reticulata, Buccinum undatum and Neptunea antiqua) From Strand (2003), i.e. modification of OSPAR (2003a).

Status class	I	II	III	IV	V
TBT conc. (aq) in seawater	close to zero	< 0.2 (ng TBT/l)	0.1–< 1.5 (ng TBT/l)	1.5–15 (ng TBT/l)	>15 (ng TBT/l)
VDSI in N. lapillus	< 0.3	0.3–< 2	2–4	> 4–> 5 (sterile)	Disappeared
ISI in L. littorea			< 0.3	0.3–1.2	> 1.2
VDSI in N. antiqua	< 0.3	0.3–< 2	2–4	(4+)	(4+)
VDSI in B. undatum		< 0.3	0.3–< 2	2–4	(4+)
VDSI in H. reticulata		< 0.3	0.3–< 2	2–4	(4+)

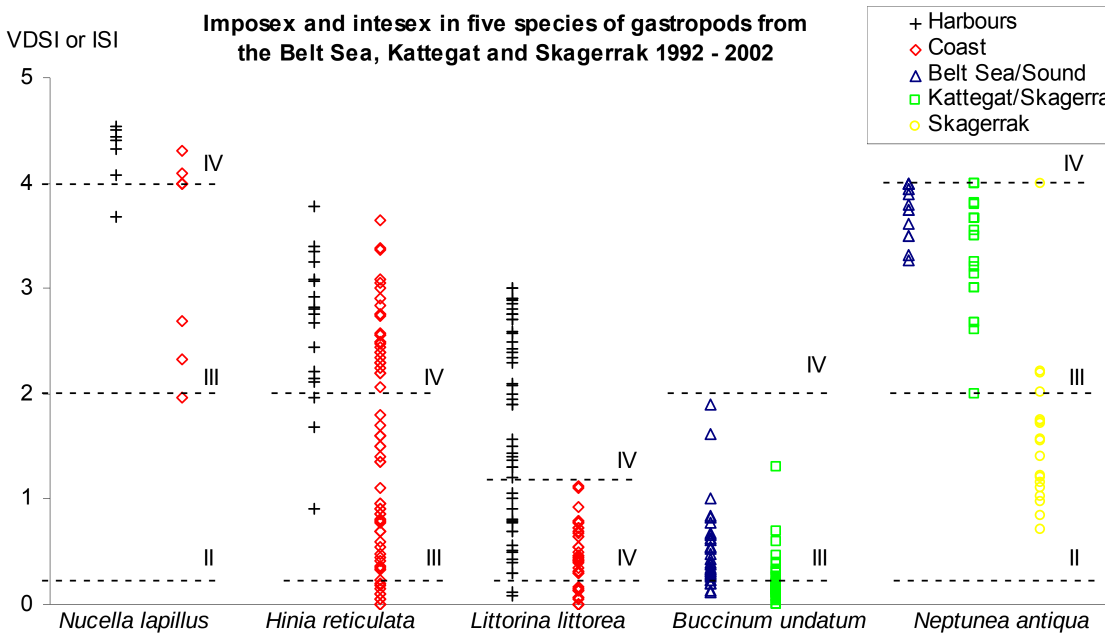

Figure 3.4. Comparison of imposex levels in five species of gastropods from the western Baltic Sea, the Kattegat and the Skagerrak.

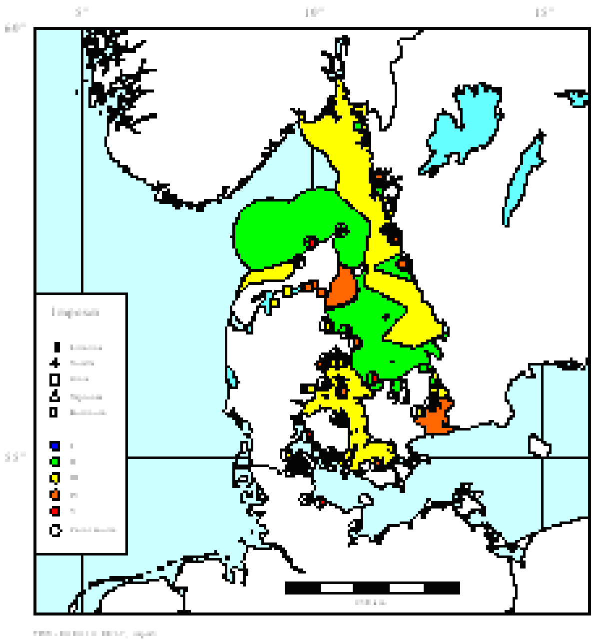

Figure 3.5. Classification of imposex levels in the western Baltic Sea, the Kattegat and the Skagerrak based on imposex levels in gastropods.

There seems to be a general high consistency between the assessment criteria for TBT in mussels and sediment and the imposex levels found in different species of prosobranch gastropods. The coastal areas where both blue mussels (*M. edulis*) and the gastropods can be found are generally assessed to be in status class III or even IV. Harbours can often even be assessed as class V. Only in deeper waters of the Kattegat and the Skagerrak (where *M. edulis* not can be found) indicate the imposex levels in *B. undatum* and *N. antiqua*, that these areas can be assessed as status class II. However, this is in line with the classification of TBT in sediments (see Figure 3.3) using the approach by Strand (2003). This supports that measured concentration levels of TBT actually can be linked to the presence of biological effects in the environment. A similar coupling has been made in a study by Strand et al. (2006).

3.3 Assessment criteria for cadmium (Cd)

For Cd it is not possible to use the same approach for derivation of assessment criteria as for TBT. Cd is as a metal naturally occurring and regional background concentrations have therefore to be defined. In addition the hardness of water (in freshwater) and the salinity affect the bioavailability and the comparability of thresholds found in different toxicity studies (EU, 2005b). It is therefore a major challenge to suggest assessment criteria, which can be applied for all regions, especially if they also should be based on an ecotoxicological approach

An added risk approach is suggested within the WFD where the EQS is derived by adding the Maximum Permitted Admission (MPA) to a regionally occurring background level (EQS = $C_{background}$ + MPA). A MPA of 0.21 µg Cd/l for seawater is derived (EU, 2005b).

In comparison OSPAR (1998) has derived an EAC between 0.1 µg Cd/l for seawater in the North Sea. However, the recent updated EAC of 0.02 µg Cd/l, which include the risk of secondary poisoning, seems to be due to an error of factor ten in the data sheets (OSPAR, 2004).

Table 3.2. Comparison of assessment criteria for Cd in seawater.

µg Cd/liter	I	II	III	IV	V
Swedish Baltic Sea	-	-	-	-	-
Swedish Kattegat	-	-	-	-	-
Norway (1997)	<0.03	0.03–0.07	0.07–0.2	0.2–0.5	>0.5
OSPAR (1998)	Background level 0.008–0.025[1]	0.01–0.1			-
WFD (2003)	Background level	Back-ground+ MPA (<0.21)			-
OSPAR (2004)	Background level	<0.16 (0.02[2])	-	-	-

[1] Background level of Cd in the North Sea (and not in the Baltic Sea)
[2] Risk of secondary poisoning for top predators included.

3.3.1 Assessment criteria for Cd in Mytilus edulis

The accumulation of Cd in *M. edulis* is highly dependent on the bioavailability of Cd in water and from food. In the Baltic the salinity is an important parameter for the accumulation level resulting in Cd concentration in soft tissue of *M. edulis* up to an order of magnitude higher in the low-saline northern part than in southern and western parts of the Baltic Sea (Phillips 1977; Bjerregaard & Depledge, 1994). Other factors, which naturally can affect the Cd levels in *M. edulis* locally, are food supply, other runoff than anthropogenic discharge, the geochemical composition of the sediment acting as substrate, and seasonal variations including the reproductive cycle. Therefore interpretation and comparison of Cd levels from data in areas with a salinity gradient should be paid special attention.

The above mentioned factors led to that OSPAR (1998) did not agreed on a EAC for Cd in mussels (and fish), however a recent updated EAC of 0.03 mg Cd/kg dw has been suggested by OSPAR (2004), which include the risk of secondary poisoning. Unfortunately this value seems to include a calculation mistake of factor ten in the data fact sheet. In this case it has also to be questioned whether it is feasible that concentrations at or even below background levels can give rise to effects. Background concentration of Cd in *M. edulis* from the North Sea have been assessed to be in the range of 0.35 – 0.55 mg Cd/kg dw (OSPAR 1998) and the background concentration is probably even higher in the low saline Baltic Sea.

In comparison no QS has been derived for Cd marine organisms in the WFD, mainly because the bioaccumulation patterns and toxicity of Cd in the marine environment could not be evaluated (EU, 2005b). The only QS which has be derived for Cd in molluscs is derived for aquatic food sources for human consumption, QS_{human} = 1.0 mg Cd/kg ww (corresponding to ~5 mg Cd/kg dw). However, for freshwater environment QS = 0.16 mg Cd/kg ww corresponding to ~0.8 mg Cd/kg dw in molluscs and fish has been derived for protection of top predators.

Table 3.2. Comparison of assessment criteria for Cd in M. edulis.

mg Cd/kg dw	I	II	III	IV	V
Swedish Baltic Sea	<4	4–4.8	4.8–6.4	6.4–8.0	>8
Swedish Kattegat	<1.3	1.3–1.7	1.7–2.2	2.2–3.0	>3
Norway (1997)	<2	2–5	5–20	20–40	>40
OSPAR (1998)[1]	Background level 0.35–0.55				-
WFD[1] (2003)	Background level	<5[2]			-
OSPAR (2004)	Background level	0.32 (0.03[3])	-	-	-

[1] (converted from mg Cd/kg ww to dw).
[2] Human health at risk.
[3] Risk of secondary poisoning for toppredators included.

Both in Sweden and Norway a five-class system of quality standards for Cd in molluscs (and fish) has been derived. The Swedish thresholds are considerable higher for the Baltic Sea than in the Kattegat and the Skagerrak. However, the Norwegian quality standards are even higher.

It could be argued that there seems to be a good consistency in the threshold related to status class II, since the criteria derived by the WFD, Norway and Sweden (Baltic Sea) since they all have come up with a threshold at ~5 mg Cd/kg dw. Only the EAC proposed by OSPAR are 1 – 2 orders of magnitude lower.

In areas with salinity above 20 psu the Cd concentration in blue mussels is generally close to the background levels defined in the Swedish assessment criteria (Swedish EPA, 2000) classifying these areas as status class I or in few cases as status class II. Elevated Cd levels, which can be classified as status class III or worse occur only in areas with salinity below 10 psu. There are indications that Cd concentrations in *M. edulis* from the northern parts of Baltic Sea can pose a significant environmental risk, especially in the study by Phillips (1977).

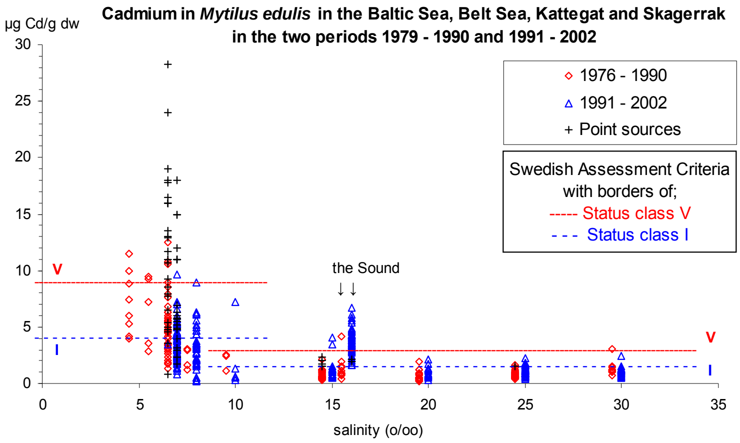

Figure 3.6. Comparison of Cd concentrations in blue mussels according to the salinity gradient in the region.

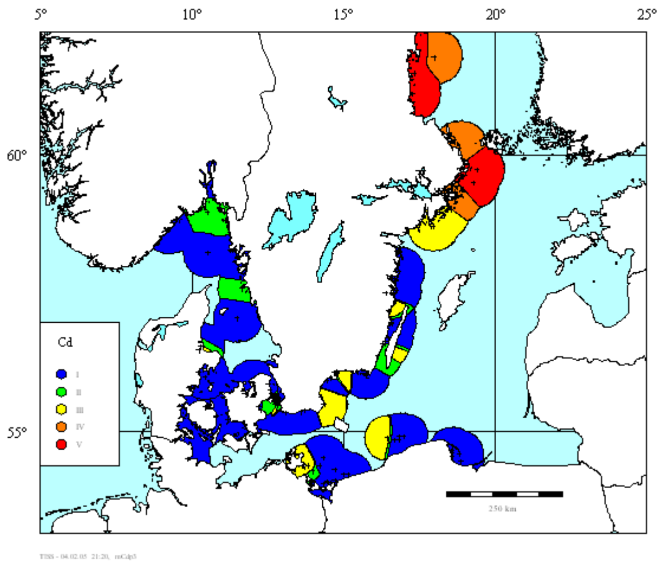

Figure 3.7. Classification of Cd levels in the Baltic Sea, the Kattegat and the Skagerrak based on Cd levels in the blue mussel M. edulis using the Swedish quality criteria for the Baltic Sea.Alternative bioindicators for Cd in the Baltic region.

3.3.2 Alternative bioindicators for Cd in the Baltic region.

Macroalges will effectively accumulate trace metals like cadmium, and for instance the bladder wrack (*Fucus vesicolusus*) is a bioindicator in Swedish and Norwegian monitoring programmes. The uptake of Cd in *F. vesicolusus* is probably in competition with other elements such as mangan (Mn) and zinc (Zn). Therefore the concentrations of Cd in *F. vesicolusus* do not necessarily reflect the Cd concentrations in the ambient water of the Baltic Sea (Szefer, 2001). Whether such factors is behind the high discrepancy between the Swedish and Norwegian assessment criteria (Table 3.8) has not been elucidated in this project.

Table 3.3. Swedish and Norwegian assessment criteria of Cd in annual growth of bladder wrack (Fucus vesiculosus) (Swedish EPA 2000; SFT 1997)

mg Cd/kg dw	I	II	III	IV	V
Sweden (2000)	<1	1.1	1.4	1.8	>1.8
Norway (1997)	<1.5	1.5–5	5–20	20–40	>40

Fish is another alternative bioindicator for Cd in the marine environment as fish accumulate trace elements from food and ambient water, but variations in accumulation levels can be expected, because choice of habitats and food items are important factors.

The concentration of Cd in fish, especially in liver is often measured in many of the ongoing monitoring programmes in the Baltic Sea, the Kattegat and the Skagerrak. However, only Sweden has developed assessment criteria for Cd in liver from fish.

For protection of human health a quality standard of 50 µg Cd/kg ww (corresponding to 250 µg Cd/kg dw) in aquatic food sources like fish has been suggested in the WFD with the exception of eels where twice the value can be accepted (EU, 2005b).

Table 3.4. Swedish assessment criteria of Cd in liver of fish (Swedish EPA, 2000).

mg Cd/kg dw	I	II	III	IV	V
Perch, Baltic	<0,2	0.3	0.6	1.0	>1
Herring, all Sweden	<0.3	0.8	2.0	5.4	>5.4
Eelpout, Baltic	<0.35	0.6	1.0	1.6	>1.6

3.3.3 Cadmium in sediments.

The Swedish Environmental Quality Criteria for Cd in sediments is set at a reference level of 0.2 mg/kg dry weight for both total and Standard Swedish methods, taken as the 50th percentile of pre-industrial values (55 cm depth). (Swedish EPA, 2000)

For Danish sediments, sediment cores from the Bay of Aarhus indicate a background level of approximately 0.4 mg/kg dry weight around the 1900-century based on 20 cm depth of Pb^{210} dated cores (Figure 1). Cadmium is expected to be associated with both organic and clay particles, corresponding to Loss on Ignition (LOI) and to the Lithium content of the sediment, respectively.

For the national monitoring programme in Denmark (NOVA), 81 sediment samples were analysed for Cd in 2004, and 25 of these (31%) were below 0.2 mg/kg dw, and 51 (61%) below 0.4 mg/kg dry weight (NERI, 2004). The 5th percentile of the Danish NOVA data is 0.074 mg/kg dry weight. Normalisation of the NOVA datasets reduces the standard deviation for all results to 93% when using Li for normalisation and to 82% when using LOI, indicating that LOI (or organic carbon content) is the most efficient normaliser. In principle, TOC should be a better normalisation parameter for organic carbon content, but it was not possible to establish a TOC value for the NOVA sediments.

The interaction between cadmium and organic phases can either be through loosely bound organic chelates, or through a firm incorporation into insoluble organic matter (Nriagu, 1980). However, effects of salinity also play a role in the behaviour of Cd. In freshwater, Cd can be found as $CdCO_3$(s), which can be dissolved by complexing with Cl^- when mixing with seawater. In seawater, the level of free Cd^{++} ions is drastically reduced due to Cd-Chloride complexation, which may lower the toxicity. Chloride can also play a role in desorbing exchangeable Cd from sediment and organic particles, when exposed to the water column.

In the reduced sediment, cadmium sulphides are highly insoluble as most metal-sulphides are. Carbonate precipitation and co-precipitation of cadmium with iron and manganese hydrates removes Cd from the water column, but can be released again when the hydrates are reduced in the sediments. Cd is generally a weak competitor for adsorption processes, which makes it difficult to normalise the content of Cd in sediment.

For freshwater systems, it has been shown that it is necessary to include both pH (competition between Cd^{++} and H^+) and organic substance (scavenging Cd from the water phase) to describe the uptake of Cd in invertebrates (Hare & Tessier, 1996).

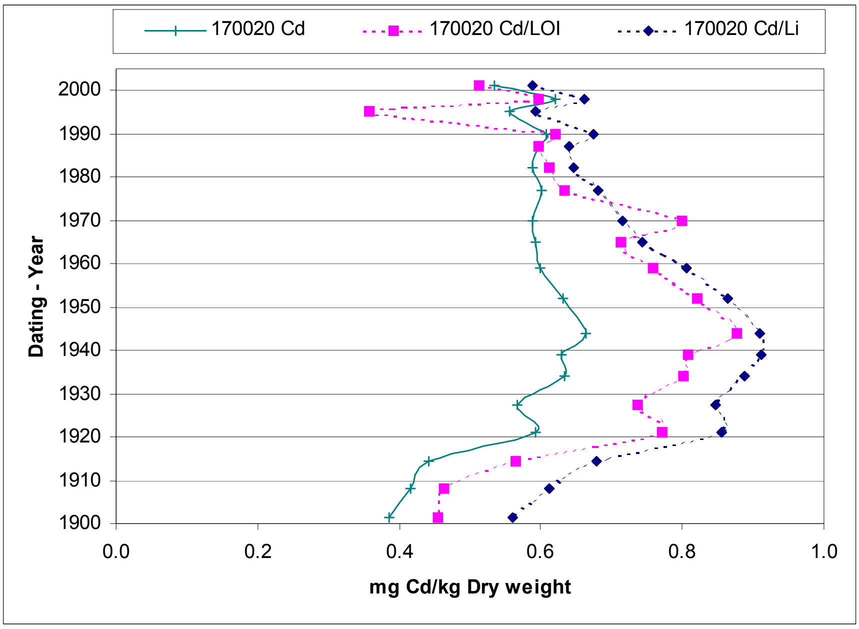

Figure 3.8. The 20 cm sediment columns from Bay of Aarhus, station 17002 (Larsen et al, in prep). The concentrations shown are based on dry weight and normalised to 30 mg Li/kg DM (Cd/Li) 10 % LOI (Cd/LOI), respectively, according to the OSPAR Sediment Monitoring guidelines annex on Normalisation (OSPAR, 2003b).

The suggested Swedish reference concentration of 0.2 mg/kg dw is reasonable, considering the open Kattegat surface sediments ranges from 0.07 to 0.27 (average 0.16/median 0.14) mg/kg dw. But for the Baltic Sea, with lower salinity and higher organic carbon content, 0.2 mg/kg might be to low, also when considering the difference in reference values for common mussel between the Baltic (4.0 mg/kg dw) and the Kattegat/Skager

Table 3.5. Comparison of assessment criteria for Cd in sediment.

mg Cd/kg dw	I	II	III	IV	V
Sweden (2000)	<0.2	0.2–0.5	0.5–1.2	1.2–3	>3
Norway (1997)	<0.25	0.25–1	1–5	5–10	>10
OSPAR (1998)	Background level	1–10			-
WFD	Background level	-			-
OSPAR (2004)	Background level	0.48 (0.06[1])			-

[1] Risk of secondary poisoning for toppredators included (converted from mg Cd/kg ww to dw).

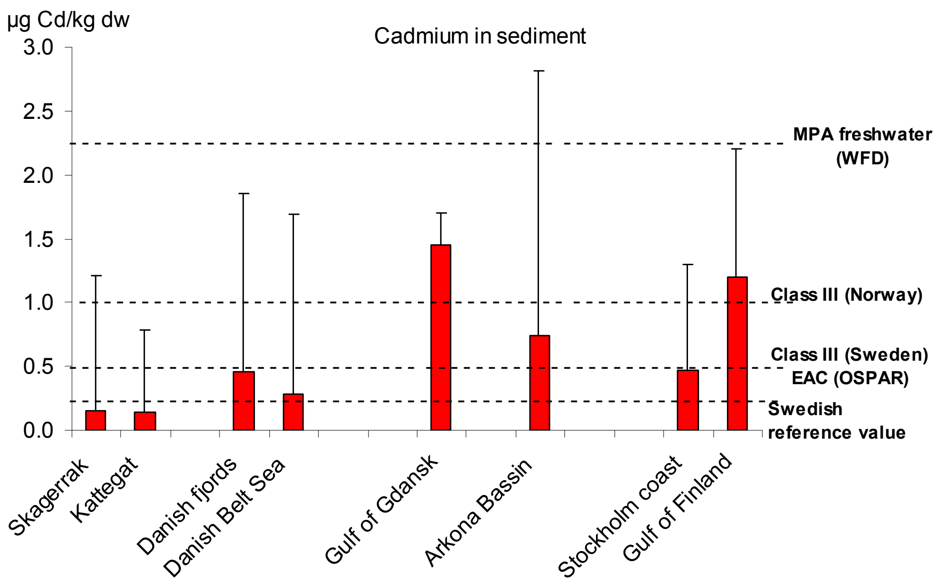

Figure 3.9. Comparison of Cd-levels (median + max.) in sediment from different regions of the Baltic Sea, the Kattegat and the Skagerrak. The Cd concentrations in Kattegat and Skagerrak are generally below the detection limit.

3.4 Assessment criteria for polychlorinated biphenyls (PCB)

PCB is a group of synthetic substances, which do not occur naturally in the marine environment. The general objective within OSPAR and HELCOM is therefore to achieve concentration close to zero, which can be linked to the definition of status class I.

3.4.1 Assessment criteria for PCB153 in fish

PCBs belong to the group of persistent organic pollutants that biomagnifies and causes sever effects for top predators. It is therefore suitable to give effect threshold values in fresh weight, whereas if biomagnification was the target, normalisation to lipid content would have been justified.

OSPAR have agreed on an EAC for ∑PCB7 in fish at 10 µg/kg ww, which has been derived from toxicological studies with mink that showed impaired reproduction after dietary exposure of PCB contaminated fish (OSPAR, 1998). This corresponds to 3.2 µg /kg ww of PCB153 when assuming that CB153 accounts for ~30%.

Assessment criteria for PCBs have not been derived within the WFD, since PCBs are not included in the priority list. However, we have in this project included assessment criteria based an ecotoxicological approach designed for protection of Baltic seals and for fish used for human consumption.

Table 3.6. Comparison of assessment criteria of PCB153 in fish (herring) muscle.

µg/kg ww	I	II	III	IV	V
Sweden (2000)	0	<0.6	0.6–2.4	2.4–9.0	>9.0
Norway (1997) [1]	<16	16–48	48–160	160–320	>320
OSPAR 1998 [1]	close to zero	<3.2	-	-	-
WFD	Not included as priority substance, i.e. no EQS derived				
Human tox [2]	0–0.7	0.7–6.7	>6.7	-	-
Ecotox approach	0–0.08	0.08–0.75	0.75–7.5	7.5–75	>75

[1] extrapolated from ΣPCB7.
[2] extrapolated from total PCB.

The human toxicological approach: The thresholds derived are based on a Swedish study on assessment of PCB in herring for human consumption. Assuming a Total Daily Intake of 20 ng PCB/kg per day, an average bodyweight of 60 kg and a daily intake of herring of 17 g per day, yields a Tolerable Average Residue Level (TARL) of 6.7 µg PCB153/kg ww. This value is used as the border between status classes III and IV. In order to derive the borders between status class II and III and I and II an application factor of 10 and 100, respectively, was used (Table 3.6).

The ecotoxicological approach: It must be emphasised that the availability of relevant data is very low, and in order to correctly perform these calculations PNEC/EQS, MAC-QS and LC_{50} values are needed.

However, for the sake of being able to compare with the statistical approaches we decided to use the mean level of total PCB in herring from the Baltic in 1978–1983 a LOEC. This was the period when there were PCB-induced effects reported such as reproductive impairment fish-eating Baltic seals. The mean level was as 2.5 mg/kg lipid (Bignert et al., 1998), which corresponds to 7.5 µg PCB153/kg ww, and defines the threshold between status class III and IV in the assessment criteria. Application factors of 10 and 100 are then used to set the remaining borders between status classes (Table 3.6).

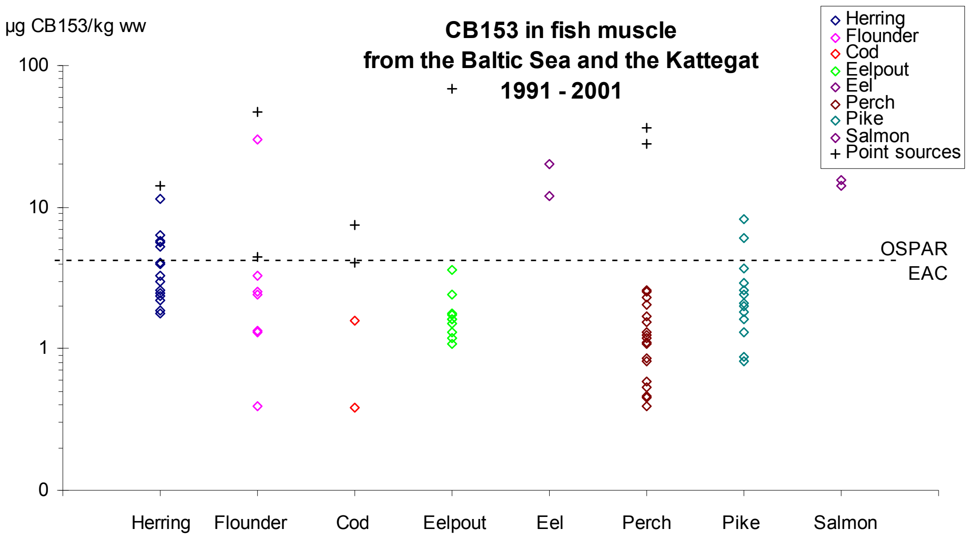

Figure 3.10. CB153 concentrations in muscle (at wet weight basis) of different fish species analysed in various studies in the Baltic Sea and the Kattegat.

Comparing the highly uncertain humans toxicological and ecotoxicological assessment criteria with the statistically well-defined Swedish assessment criteria, it can be concluded that the lie within the same concentration ranges. This is not the case when comparing to the Norwegian classification where the concentrations are at least an order of magnitude higher.

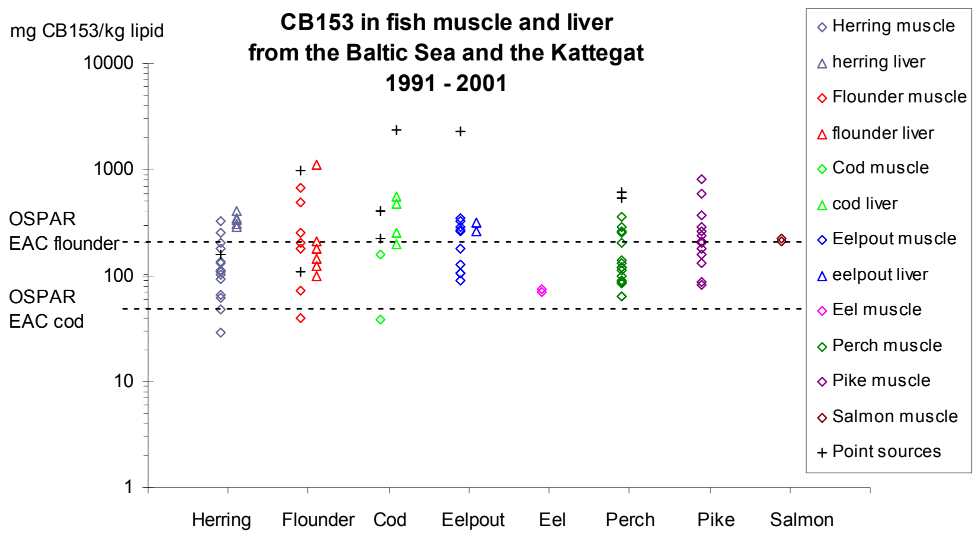

Figure 3.11. Normalisation of CB153 concentration to lipid content reduces the interspecies and organ variability.

However, it is generally difficult to use fish as bioindicators in spatial assessments within a region. First of all fish is not that stationary, secondly, there may be high variance in accumulation levels of PCB between different species as well as between individuals (OSPAR, 2004b).

This is highly influenced by the interspecies and organ variations in lipid content and normalisation to lipid content may therefore narrow the range (Figure 3.10). This will consequently also result in differences in lipid normalised EAC values between species with lower and higher lipid contents (Figure 3.11).

3.4.2 Assessment criteria for PCB153 in blue mussel (Mytilus edulis)

The lack of relevant data for mussels is the same as in the previous example using fish. Only OSPAR has agreed to an EAC for ΣPCB7 in blue mussel of 5–50 µg/kg dw. This corresponds to 3.0 µg/kg ww of PCB153 when assuming that CB153 accounts for ~40% of ΣPCB7.

The same approach designed for protecting Baltic Seals was used as in the fish example for calculating the thresholds for the ecotoxicological approach, with the only exception that PCB concentrations in mussels in the period 1978–1983 from the Baltic were used (Bignert el al., 1998).

In this case the highly uncertain ecotoxicolgical approach yields the lowest limits, but this time in higher agreement with the Norweigian assessment criteria. Unfortunately there are no Swedish classification to be compared with. The EAC from OSPAR falls within the range of the Norwegian status class II.

Table 1 Comparison of assessment criteria of PCB153 in blue mussel (Mytilus edulis).

µg/kg ww	I	II	III	IV	V
Sweden (2000)	-	-	-	-	-
Norway (1997) [1]	<1.6	1.6–6.0	6.0–16	16–40	>40
OSPAR (1998) [1]	near zero	<3.0			
WFD	Not included as priority substance, i.e. no EQS derived				
Ecotox approach	0–0.02	0.02–0.2	0.2–2.0	2.0–20	>20

[1] extrapolated from ΣPCB7

The use of PCB data from 1970 – 1980ties as toxicity data for lowest observed chronic effects and the principles behind ecotox approach (Table 2.5) imply that the Baltic Sea generally could be classified status class IV at that time. Today the PCB levels have declined with almost an order of magnitude (Bignert et al. 1998) and the environmental risks is thereby reduced. However, according to this approach further declines in PCB levels in *M. edulis* have to occur before the Baltic Sea can enter from staus class III and into status class II, where chronic effects will be unlikely to occur (Figure 3.12, 3.13).

In contrast, will the assessment criteria suggested by OSPAR (1998) and by Norway (SFT, 1997) classify the pesent PCB levels in *M. edulis* in the entire Baltic Sea as belonging to status class II. However these assessment criteria will also imply that the PCB levels did not pose an significant environmental risk even in the 1970–1980'ties, although at that time adverse effects on reproduction and health of marine toppredators like seals and birds clearly occurred in the Baltic Sea (HELCOM, 2002).

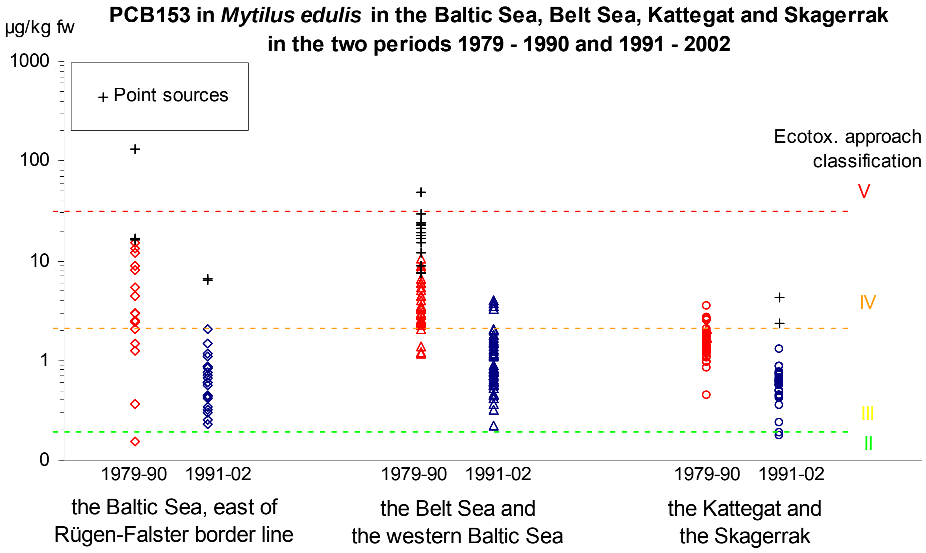

Figure 3.12 Comparison of CB153 concentrations in blue mussels from the Baltic Sea, the Belt Sea and the Kattegat and the Skagerrak in two periods 1979–1990 and 1991–2000.

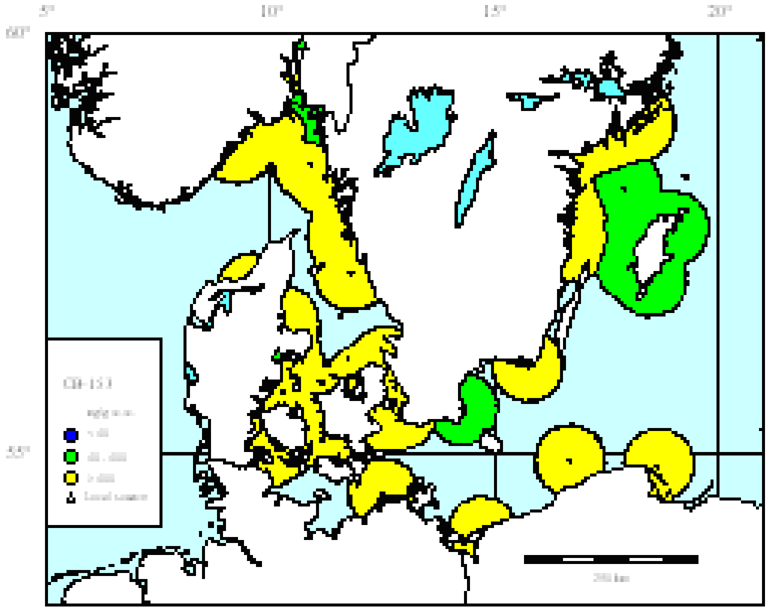

Figure 3.13 Classification of PCB levels in the Baltic Sea, the Kattegat and the Skagerrak based on CB153 levels in the blue mussel M. edulis based on Ecotox. approach.

4. Conclusions

The objective of this project was to compare and evaluate different types of classification systems for contaminants in the marine environment presently used in by the Nordic countries surrounding the Baltic and the Kattegat. The aim was to suggest a common strategy in order to secure that the same classification is made of the same area. The intention was also to suggest an operational approach for the classification of our seas, which can bring the current marine strategy using mainly biota and sediment in line with the objectives of the status classes defined in the Water Framework Directive.

It is evident that the different classification systems presently used leads to a very different classification of the Nordic waters. When the classification is based on TBT in the blue mussel *Mytilus edulis,* an area where the concentration is 20 µg TBT/kg dw, can belong either to class I, II or III depending on the approach used. Likewise for cadmium and PCB153 where a concentration of 2 mg/dw and 2 µg/kg ww in mussels, respectively, also spans between class I and III. The main reason for this discrepancy is that the Swedish and Norwegian systems are based on the statistical distribution of contaminant data available, whereas the other systems take the risk of adverse biological effects in to account. However, there are still discrepancies between the two distribution approaches, possibly caused by regional differences in data chosen to construct the distributions.

There are also discrepancies between the effect approaches caused by differences in evaluation of (eco)toxicological data. This is in turn related to lack of relevant ecotoxicological effect data and difficulties in extrapolating between water and biota/sediment concentrations.

The ecotoxicological approach suggested at the OSPAR workshop 2004 covers the objectives of the WFD, and provides clear guidelines on how to calculate the different limits. Relevant extrapolations between the different matrices can be made when sufficient data is available. Thereby the classification can potentially be harmonised in such a way that measurements directly in water, mussels or sediment, or indirectly through the use of concentration indicators, results in a uniform classification of areas, which is independent on the matrices used. TBT is an example where this approach works, as discussed in Chapter 3.1.

For the other substances the problem with this approach is to decide on how to calculate the background level, PNEC/EQS, MAC-QS and LC_{50} with insufficient ecotoxicological data, or when extrapolation between water and biota/sediment concentrations is difficult to make. An example of this is for PCBs where the WFD (Lepper, 2002) and OSPAR

(2004) have decided not to proceed with the ecotoxicological approach. The ecotoxicological approach can still be used as long as the derivation of PNEC/EQS, MAC-QS and LC_{50} is given, as is shown for PCB153. However, in such cases it is crucial to make a second validation of the calculated levels or the result can become nonsensical as in the case for OSPARs evaluation using the ecotoxicological approach for cadmium. Here the concentration given for *M. edilus* is 0.32 mg/kg dw, based on PNEC for water with an applied bioconcentration factor, a concentration that is below or in par with background levels. When the calculation is based on the concentration aimed at protecting top predators to secondary poisoning the result is the same. However due to a calculation mistake in the OSPAR report, the published concentration is ten times below the background (see chapter 3.2). In the cadmium case it has to be questioned whether it is feasible that concentrations at or even below background levels can give rise to effects.

Both the distribution and the ecotoxicological approaches have their respective advantages and drawbacks. The advantage of a distribution approach is that the only prerequisite is that of sufficient data in order to ensure a solid statistical distribution for the different areas and matrices chosen. A combination of the available data for the Nordic waters could form the bases for such an approach where consideration to salinity differences and matrices can be made. The major drawback is that this kind of classification is very difficult to link to the possible adverse effects caused by the contaminants on the biological world it is set to protect. Here lies the strength of the ecotoxicological approaches, in that it seeks to protect biodiversity. The major draw back lies in the amount of work and assumptions that have to be made before the environmental risks can be classified even for the top priority substances.

Since the Nordic countries have to comply with the WFD where the focus is on protection of biodiversity, it is our suggestion that an ecotoxicological approach should be included in the derivation of assessment criteria. When deciding on how to determine PNEC/EQS and MAC-QS, the parties can either chose to only use the data suggested by OSPAR, or decide to calculate such data themselves where OSPAR does not provide guidelines. For contaminants with no valid data available, a distribution approach can be used intermediately, but based on a combined data set for highest statistical power, and with agreement on which percentils that defines the different classes.

It is furthermore suggested that the classification should be based mainly on contaminant concentrations in sediment and biota like *M. edulis* or certain fish species and not based on concentrations in water. This is a practical consideration since it takes a much larger sampling regime to evaluate contamination levels using momentary sampling of water compared to integrated sampling of biota and sediments. In addition a change of the current monitoring strategy could result in destruction of important time series.

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Sammenfatning

Miljøfarlige stoffer, både tungmetaller og organiske miljøfremmede stoffer, er i dag udbredt forekommende i havmiljøet, og et øget niveau af visse toksiske stoffer udgør derfor en risiko for følsomme organismer og de marine økosystemers struktur og funktioner.

Det overordnede formål med dette projekt er at sammenligne og evaluere forskellige metoder til at vurdere og klassificere den kemiske tilstand for miljøfarlige stoffers i det marine miljø. Det er væsentligt såvel med en fælles nordisk forståelse for kriteriefastlæggelsen, som et sammenhængende vurderingssystem for de kystnære og åbne farvande i Østersøen og Kattegat.

Der findes i dag forskellige typer af klassificeringssystemer til at vurdere graden af forurening med miljøfarlige stoffer i det marine miljø. Disse metoder er blevet anvendt i forskelligt omfang i de enkelte nordiske lande, der omgiver Østersøen og Kattegat. Rapporten giver eksempler for brug af vurderingskriterier såvel i de kystnære som i de åbne havområder – baseret på eksisterende data indsamlet i de nationale overvågningsprogrammer og andre nationale undersøgelser.

Intentionen er også at foreslå en operationel metode til klassificering af miljøkvaliteten, der kan sammenbringe den hidtidige dominerende strategi i marine overvågning i Norden med målsætningen i EUs vandrammedirektiv om at opnå „god økologisk tilstand“. Vurderingskriterier kan bruges til at vurdere såvel den nuværende tilstand som en eventuel afvigelse fra en reference tilstand, der kan indikere en menneskeskabt påvirkning.

Hidtil har den marine overvågning i de nordiske lande primært været fokuseret på tidslige udviklinger og spatiale trends i koncentrationsniveauer af miljøfarlige stoffer i sediment og biota, og der foreligger derfor kun få relevant data for miljøfarlige stoffer i havvand. Derfor tages der i analysen udgangspunkt i koncentrationsmålinger i sediment og biologiske prøver som fisk og muslinger. Data er sammenstillet med henblik på at opstille 5 kvalitetsklasser for høj, god, moderat, ringe og dårlig kemisk kvalitet.

PCB, cadmium (Cd) og tributyltin (TBT) er valgt som teststoffer. Den sidste fase af projektet vil fokusere på udarbejdelsen af klassificeringsværdier for sediment og suspenderet materiale. De samme teststoffer som for biota vil så hvidt muligt blive bibeholdt. Data vedrørende effekter på grund af forekomsten af de miljøfarlige stoffer, f.eks. forekomsten af imposex/intersex i havsnegle vil også blive inddraget i analysen primært som en validering af klassificeringsmetoden for at herved sikre sammenlignelighed og harmonisering mellem klassificeringsmetoder baseret på koncentrationsdata og effektdata.

Eksemplerne viser at især TBT vurderes at udgøre et udbredt miljøproblem i Østersøen og Kattegat, både baseret på klassificering af koncentrationsniveauer i sediment og muslinger samt forekomst af hormonforstyrrende effekter af TBT i havsnegle. Mht. Cd og PCB synes øget koncentrationsniveauer mere at være lokale miljøproblemer, men i hvilket omfang afhænger i høj grad af hvilke vurderingskriterier der er anvendt.

Appendices

Appendix A. OSPARs EAC-values for seawater, sediment and biota

Table A.1. OSPAR. Agreed ecotoxicological assessment criteria for trace metals, PCBs, PAHs, TBT and some organochlorine pesticides (OSPAR, 1998).

Substance	Water (µg/l)	Sediment (mg/kg dw)	Fish (mg/kg ww)	Mussel (mg/kg dw)
Trace metals				
Arsenic (As)	1–10 (f)	1–10 (p)	n.r.	n.r.
Cadmium (Cd)	0.01–0.1 (f)	0.1–1 (p)	f.c.	f.c.
Chromium (Cr)	1–10 (f)	10–100 (p)	n.r.	n.r.
Copper (Cu)	0.005–0.05 (f)[1]	5–50 (p)	f.c.	f.c.
Mercury (Hg)	0.005–0.05 (f)	0.05–0.5 (p)	f.c.	f.c.
Nickel (Ni)	0.1–1 (p)	5–50 (p)	n.r.	n.r.
Lead (Pb)	0.5–5 (f)	5–50 (p)	f.c.	f.c.
Zinc (Zn)	0.5–5 (f)	50–500 (p)	n.r.	n.r.
Organochlorine pesticides				
DDE	n.r.	0.0005–0.005 (p)	0.005–0.05 (f)	0.005–0.05 (f)
Dieldrin	n.r.	0.0005–0.005 (p)	0.005–0.05 (f)	0.005–0.05 (f)
Lindane	0.0005–0.005 (f)	n.r.	0.0005–0.005 (f)	n.r.
PAHs				
Naphthalene	5–50 (f)	0.05–0.5 (f)	n.r.	0.5–5 (p)
Phenanthrene	0.5–5 (p)	0.1–1 (f)	n.r.	5–50 (p)
Anthracene	0.001–0.01 (p)	0.05–0.5 (f)	n.r.	0.005–0.05 (p)
Fluoranthene	0.01–0.1 (p)	0.5–5 (p)	n.r.	1–10 (p)
Pyrene	0.05–0.5 (p)	0.05–0.5 (p)	n.r.	1–10 (p)
Benzo[*a*]anthracene	n.d.	0.1–1 (p)	n.r.	n.d.
Chrysene	n.d.	0.1–1 (p)	n.r.	n.d.
Benzo[*k*]fluoranthene	n.d.	n.d.	n.r.	n.d.
Benzo[*a*]pyrene	0.01–0.1 (p)	0.1–1 (p)	n.r.	5–50 (p)
Benzo[*ghi*]perylene	n.d.	n.d.	n.d.	n.d.
Indeno[*123-c.d*]pyrene	n.d.	n.d.	n.r.	n.d.
ΣPCB_7	n.r.	0.001–0.01 (p)	0.001–0.01 (f)	0.005–0.05 (f)
Tributyltin (TBT)	0.00001–0.0001(f)	0.000005–0.00005 (p)	n.r.	0.001–0.01 (f)

f: firm, p: provisional, f.c.: for future consideration, n.r.: not relevant in relation to the current monitoring programme, n.d.: no data available or insufficient data available, 1: this range is within the background range for natural water. This value should be compared to the bioavailable fraction of Cu in seawater.

Appendix B. EU proposals for quality standards in seawater, sediment and biota

Table B.1. EU WFD Quality Standards for 33 priority substances in other surface waters than inland waters. AA-EQS and MAC-EQS are from (EU 2006): Proposal for a Directive of the European Parliament and of the Council on environmental quality standards in the field of water policy and amending Directive 2000/60/EC.

	Substance	AA-EQS (µg/l)	MAC-EQS (µg/l)	QS Marine sediment (µg/kg dw)	QS Biota, predators / humans (µg/kg ww)
1	Alachlor	0.3	0.7	-	- / 304
2	Anthracene	0.1	0.4	310	33.3 / -
3	Atrazine	0.6	2.0	5.2	-
4	Benzene	8	50	-	-
5	Pentabromodiphenylether	0.0002	n.a.	310	1000 / 274
6	Cadmium (Cd)	0.2	-	-	160 / 100–1000
7	C10-C13 chloroalkanes	0.4	1.4	998	16600 / 60870
8	Chlorfenvinphos	0.1	0.1	-	33 / 304
9	Chlorpyrifos	0.03	0.1	-	67 / 608
10	1,2-dichlorethane	10	n.a.	-	-
11	Dichlormethane	20	n.a.	-	- / 420
12	Di(2-ethylhexyl)phthalate (DEHP)	1.3	n.a.	-	3200 / 2920
13	Diuron	0.2	1.8	-	- / 426
14	Endosulfan	0.0005	0.004	-	1000 / 365
15	Fluoranthene	0.1	1	173	11530 / -
16	Hexachlorbenzene	0.01	0.05	16.9	16.7 / 9.74
17	Hexachlorbutadiene	0.1	0.6	493	55.3 / 12.2
18	Hexachlorcyklohexane (lindan, HCH)	0.002	0.02	1.1	33 / 61
19	Isoproturon	0.3	1.0	-	- / 910
20	Lead (Pb)	7.2	n.a.	MPA: 53.4	1000 / 200–1000
21	Mercury (Hg) and species	0.05	0.07	MPA: 470	22 / 500
22	Naphthalene	1.2	n.a.	-	12270 / 80900
23	Nickel (Ni)	20	n.a.	(2900 as ww)	730 / 670
24	Nonylphenol	0.3	2.0	180	10000 / 8700
25	Octylphenol	0.01	n.a.	3.4	10000 / 8700
26	Pentachlorbenzene	0.0007	n.a.	400	367 / 30–49
27	Pentachlorphenol	0.4	1	119	1830 / 18300
28	Polyaromatic hydrocarbons (PAH)	n.a.	n.a.	-	-
-	Benzo(a)pyrene	0.05	0.1	543	- / 365
-	Benzo(b,k)fluoranthene	0.03	n.a.	1743	- / -
-	Benzo(g,h,i)perylene Ideno(1,2,3-cd)pyrene	0.002	n.a.	- / -	- /-
29	Simazine	1	4	15.5	- / 304
30	Tributyltin compounds	0.0002	0.0015	0.02	230 / 15.2
31	Trichlorbenzenes	0.4	n.a.	(90, as ww)	4000 / 3650
32	Trichlormethane	2.5	n.a.	(12, as ww)	- / -
33	Trifluraline	0.03	n.a.	3140	6700 / 1460

The proposals for specific quality standards for benthic community (i.e. QS Marine sediment) and for secondary poisoning of toppredators or uptake by human of mussels, fish or other seafood (i.e. QS Biota) are extracted from the respective substance fact sheets (EU 2005a,b).
n.a.: not applicable, -: no relevant value derived.
Note: Many of the specific quality standards for marine sediments and biota are only tentative and not firm values.

Appendix C. Finnish proposal for quality criteria for dredged materials in the Baltic Sea

Table C.1. Quality criteria (action levels) for trace metals and organic contaminants in dredged materials proposed by Finland for the Baltic Sea (HELCOM, 2004).

Contaminant	Action level 1 (µg/kg dw)	Action level 2 (µg/kg dw)
Trace metals		
Hg	100	1000
Cd	500	2500
Cr	65000	270000
Cu	50000	90000
Pb	40000	200000
Ni	45000	60000
Zn	170000	500000
As	15000	60000
PAHs		
naphthalene	10	100
anthracene	10	100
phenanthrene	50	500
fluoranthene	300	3000
benzo(a)anthracene	30	400
chrysene	1100	11000
benzo(k)fluoranthene	200	2000
benzo(a)pyrene	300	3000
benzo(ghi)perylene	800	8000
indeno(123-cd)pyrene	600	6000
mineral oil	50000	1500000
Organochlorines		
DDT + DDE +DDD	10	30
CB28	1	30
CB52	1	30
CB101	4	30
CB118	4	30
CB138	4	30
CB153	4	30
CB180	4	30
PCDDs and PCDFs	0.02 (as WHO-TEQ)	0.5 (as WHO-TEQ)
Tributyltin (TBT)	3	200

Appendix D. Norwegian assessment criteria for seawater, sediment and biota

Table D.1. Norwegian assessment criteria for trace metals and organic contaminants in seawater and sediment (copied from SFT, 1997).

*Tabell 7. Klassifisering av tilstand ut fra innhold av metaller og klororganiske forbindelser i vann og sedimenter. * ved verdien i kl. I markerer forandring fra første utgave av veiledningen (ledsagende justeringer i de øvrige klasser ikke avmerket). Nye parametre er merket **.*

	Parametre	Tilstandsklasser: I Ubetydelig–Lite forurenset	II Moderat forurenset	III Markert forurenset	IV Sterkt forurenset	V Meget sterkt forurenset
Metaller m.m. i vann	Arsen (µg As/l)	<2	2-5	5-10	10-20	>20
	Bly (µg Pb/l)	<0.05	0.05-0.15	0.15-0.5	0.5-1	>1
	Fluorid (µg F/l)	<1300	1300-4000	4000-6000	6000-10000	>10000
	Kadmium (µg Cd/l)	<0.03	0.03-0.07	0.07-0.2	0.2-0.5	>0.5
	Kobber (µg Cu/l)	<0.3	0.3-0.7	0.7-1.5	1.5-3	>3
	Krom (µg Cr/l)	<0.2	0.2-0.5	0.5-1.5	1.5-3	>3
	Kvikksølv (µg Hg/l)	<0.001*	0.001-0.005	0.005-0.015	0.015-0.03	>0.03
	Nikkel (µg Ni/l)	<0.5	0.5-2	2-5	5-10	>10
	Sink (µg Zn/l)	<1.5	1.5-5	5-10	10-20	>20
	Sølv (µg/ Ag/l)	<0.01	0.01-0.03	0.03-0.1	0.1-0.2	>0.2
Metaller m.m. i sedimenter (tørrvekt)	Arsen (mg As/kg)	<20	20-80	80-400	400-1000	>1000
	Bly (mg Pb/kg)	<30	30-120	120-600	600-1500	>1500
	Fluorid (mg F/kg)	<800	800-3000	3000-8000	8000-20000	>20000
	Kadmium (mg Cd/kg)	<0.25	0.25-1	1-5	5-10	>10
	Kobber (mg Cu/kg)	<35	35-150	150-700	700-1500	>1500
	Krom (mg Cr/kg)	<70	70-300	300-1500	1500-5000	>5000
	Kvikksølv (mg Hg/kg)	<0.15	0.15-0.6	0.6-3	3-5	>5
	Nikkel (mg Ni/kg)	<30	30-130	130-600	600-1500	>1500
	Sink (mg Zn/kg)	<150	150-700	700-3000	3000-10000	>10000
	Sølv (mg Ag/kg)	<0.3	0.3-1.3	1.3-5	5-10	>10
	TBT ** [1] (µg/kg)	<1	1-5	5-20	20-100	>100
Organiske miljøgifter i sedimenter (tørrvekt)	Σ PAH [2] (µg/kg)	<300	300-2000	2000-6000	6000-20000	>20000
	B(a)P [3] (µg/kg)	<10	10-50	50-200	200-500	>500
	HCB [4] (µg/kg)	<0.5	0.5-2.5	2.5-10	10-50	>50
	Σ PCB_7** [5] (µg/kg)	<5	5-25	25-100	100-300	>300
	EPOCl [6] (µg/kg)	<100	100-500	500-2000	2000-15000	>15000
	$TE_{PCDF/D}$ [7] (ng/kg)	<0.01*	0.01-0.03	0.03-0.10	0.10-0.5	>0.5
	Σ DDT** [8] (µg/kg)	<0.5	0.5-2.5	2.5-10	10-50	>50

1) TBT: Tributyltinn (antibegroingsmiddel i skipsmaling).
2) PAH: Polysykliske aromatiske hydrokarboner. Gruppe tjærestoffer der en del forbindelser er potensielt kreftfremkallende (KPAH, deriblant benzo(a)pyren B(a)P). ΣPAH: sum av tri- til heksasykliske forbindelser bestemt ved gasskromatografi med glasskapillarkolonne. Inkluderer de 16 i EPA protokoll 8310 minus naftalen (disyklisk). Omfatter dessuten alle KPAH (gr. 2A og gr. 2B i IARC, 1987).
3) Se under PAH.
4) HCB: Heksaklorbenzen.
5) PCB: Polyklorerte bifenyler. Gruppe forbindelser (ulike kommersielle blandinger). Σ PCB_7= sum av de 7 enkeltforbindelsene nr. 28, 52, 101, 118, 138, 153 og 180. I den tidligere utgave av veiledningen er PCB angitt som total PCB ut fra likhet med kommersielle blandinger. Enkelte PCB har dioksinlignende egenskaper (se note 2 til tabell 9).
6) EPOCL: Ekstraherbart persistent organisk bundet klor.
7) Toksisitetsekvivalenter, se note 2 til tabell 8.
8) DDT: Diklordifenyltrikloretan. ΣDDT betegner sum av DDT og nedbrytningsproduktene DDE og DDD

Table D.2. Norwegian assessment criteria for trace metals in macroalge (Fucus vesiculosus and Ascophyllum nodosum), blue mussels (Mytilus edulis), periwinkle (Littorina littorea) and cod (Gadus Morhua) (copied from SFT, 1997).

22 *Klassifisering av miljøkvalitet*

*Tabell 8. Klassifisering av tilstand ut fra organismers innhold av metaller, arsen og fluorid. * ved verdien i kl. I markerer forandring fra tidligere (justeringer i de øvrige klasser ikke avmerket). Ny parameter er merket **.*

Arter/vev:	Parametre:	Tilstandsklasser: I Ubetydelig- Lite forurenset	II Moderat forurenset	III Markert forurenset	IV Sterkt forurenset	V Meget sterkt forurenset
Blæretang	Arsen (mg/kg)	< 50	50 - 150	150 - 350	350 - 700	> 700
og grisetang	Bly (mg/kg)	< 1*	1-3	3-10	10 - 30	> 30
øvre 10 cm	Fluorid (mg/kg)	< 15	15 - 50	50 - 100	100 - 300	> 300
(tørrvektsbasis)	Kadmium (mg/kg)	< 1.5	1.5 - 5	5 - 20	20 - 40	> 40
	Kobber (mg/kg)	< 5*	5 - 15	15 - 50	50 - 150	> 150
	Krom (mg/kg)	< 1	1 - 5	5 - 15	15 - 50	> 50
	Kvikksølv (mg/kg)	< 0.05	0.05 - 0.15	0.15 - 0.5	0.5 - 1	> 1
	Nikkel (mg/kg)	< 5	5 - 25	25 - 50	50 - 100	> 100
	Sink (mg/kg)	< 150 *	150 - 400	400 - 1000	1000 - 2500	> 2500
	Sølv (mg/kg)	< 0.5	0.5 - 1.5	1.5 - 5	5 - 10	> 10
Blåskjell	Arsen (mg/kg)	< 10	10 - 30	30 - 100	100 - 200	> 200
bløtdeler minus	Bly (mg/kg)	< 3*	3 - 15	15 - 40	40 - 100	> 100
lukkemuskler	Fluorid (mg/kg)	< 15	15 - 50	50 - 150	150 - 300	> 300
(tørrvektsbasis)	Kadmium (mg/kg)	< 2	2 - 5	5 - 20	20 - 40	> 40
	Kobber[1)] (mg/kg)	< 10	10 - 30	30 - 100	100 - 200	> 200
	Krom (mg/kg)	< 3	3 - 10	10 - 30	30 - 60	> 60
	Kvikksølv (mg/kg)	< 0.2	0.2 - 0.5	0.5 - 1.5	1.5 - 4	> 4
	Nikkel (mg/kg)	< 5	5 - 20	20 - 50	50 - 100	> 100
	Sink[1)] (mg/kg)	< 200	200 - 400	400 - 1000	1000 - 2500	> 2500
	Sølv (mg/kg)	< 0.3	0.3 - 1	1 - 2	2 - 5	> 5
	TBT [2)] ** (mg/kg)	< 0.1	0.1 - 0.5	0.5 - 2	2 - 5	> 5
Vanlig	Arsen (mg/kg)	< 30	30 - 75	75 - 300	300 - 600	> 600
strandsnegl	Bly (mg/kg)	< 10	10- 25	25 - 75	75 - 150	> 150
bløtdeler	Kadmium (mg/kg)	< 2	2 - 8	8 - 25	25 - 50	> 50
(tørrvektsbasis)	Kobber (mg/kg)	< 150	150 - 300	300 - 750	750 - 1500	> 1500
	Krom (mg/kg)	< 3	3 - 10	10 - 30	30 - 60	> 60
	Kvikksølv (mg/kg)	< 0.5	0.5 - 2	2 - 5	5 - 10	> 10
	Nikkel (mg/kg)	< 10	10 - 30	30 - 100	100 - 200	> 200
	Sink (mg/kg)	< 100	100 - 300	300 - 1000	1000 - 2000	> 2000
	Sølv (mg/kg)	< 3	3 - 10	10 - 20	20 - 40	> 40
Torsk filét (friskvektsbasis)	Kvikksølv (mg/kg)	< 0.1	0.1 - 0.3	0.3 - 0.5	0.5 - 1	> 1

1) Blåskjell har evne til å regulere opptak , særlig ved moderate konsentrasjoner. Tang er bedre som indikator.

2) Tributyltinn. Grensen for kl. I er beregnet ut fra vannkvalitetskriterium på 1 ng/l (kfr. Zabel et al. 1988, Moore et al. 1992) og et forhold mellom konsentrasjonene i blåskjell (våtvektsbasis) og vann på ca. 10000. Forholdet skjell : vann varierer fra ca. 5000 til over 50000, og øker med avtagende TBT-innhold i vannet (Knutzen et al.1995 m.ref.). Ved svak belastning (1 ng/l og mindre) kan det derfor antas at bruk av et forholdstall på 10000:1 gir en sikkerhetsmargin (0,1 mg/kg tørrvekt i blåskjell tilsvarer < 1 ng/l i vann).

Table D.3. Norwegian assessment criteria for organic contaminants inblue mussels (Mytilus edulis), cod (Gadus Morhua), flounder (Platichthys flesus), herring (Clupea sp) (copied from SFT, 1997).

*Tabell 9. Klassifisering av tilstand ut fra organiske miljøgifter i organismer. * ved verdien i kl. I markerer forandring fra tidligere. Revisjoner i øvrige klasser er ikke avmerket. Nye parametre er merket **. Forklaring til forkortelser er gitt under tabell 7 og som fotnote til denne tabellen.*

Arter/vev:	Parametre:	Tilstandsklasser: I Ubetydelig-Lite forurenset	II Moderat forurenset	III Markert forurenset	IV Sterkt forurenset	V Meget sterkt forurenset
Blåskjell bløtdeler minus lukkemuskler (friskvektsbasis)	Σ PAH (µg/kg)	< 50 *	50 - 200	200 - 2000	2000 - 5000	> 5000
	Σ KPAH ** (µg/kg)	< 10	10 - 30	30 - 100	100 - 300	> 300
	B(a)P (µg/kg)	< 1	1 - 3	3 - 10	10 - 30	> 30
	Σ DDT (µg/kg)	< 2	2 - 5	5 - 10	10 - 30	> 30
	HCB (µg/kg)	< 0.1 *	0.1 - 0.3	0.3 - 1	1 - 5	> 5
	Σ HCH[1)] (µg/kg)	< 1 *	1 - 3	3 - 10	10 - 30	> 30
	Σ PCB_7 (µg/kg)	< 4	4 - 15	15 .- 40	40 - 100	> 100
	$TE_{PCDF/D}$ [2)] (ng/kg)	< 0.2 *	0.2 - 0.5	0.5 - 1.5	1.5 - 3	> 3
Torsk lever (friskvekts-basis)	Σ DDT (µg/kg)	<200	200 - 500	500 - 1500	1500 - 3000	> 3000
	HCB (µg/kg)	<20	20 - 50	50 - 200	200 - 400	> 400
	Σ HCH (µg/kg)	<50	50 - 200	200 - 500	500 - 1000	> 1000
	Σ PCB_7 ** (µg/kg)	<500	500 - 1500	1500 - 4000	4000 - 10000	> 10000
	$TE_{PCDF/D}$ (ng/kg)	<15 *	15 - 40	40 - 100	100 - 300	> 300
Torsk filét (friskvekts-basis)	Σ DDT (µg/kg)	<1 *	1 - 3	3 - 10	10 - 25	> 25
	HCB (µg/kg)	<0.2	0.2 - 0.5	0.5 - 2	2 - 5	> 5
	Σ HCH (µg/kg)	<0.5 *	0.5 - 2	2 - 5	5 - 15	> 15
	Σ PCB_7 ** (µg/kg)	<5	5 - 20	20 - 50	50 - 150	> 150
	$TE_{PCDF/D}$ (ng/kg)	<0.1 *	0.1 - 0.3	0.3 - 1	1 - 2	> 2
Skrubbe filét (friskvekts-basis)	Σ DDT (µg/kg)	<2 *	2 - 4	4 - 15	15 - 40	> 40
	HCB (µg/kg)	<0.2 *	0.2 - 0.5	0.5 - 2	2 - 5	> 5
	Σ HCH (µg/kg)	<1 *	1 - 3	3 - 10	10 - 30	> 30
	Σ PCB_7 ** (µg/kg)	<5	5 - 20	20 - 50	50 - 150	> 150
	$TE_{PCDF/D}$ (ng/kg)	<0.1 *	0.1 - 0.3	0.3 - 1	1 - 3	> 3
Sild filét (friskvekts-basis)	Σ DDT (µg/kg)	< 20	20 - 50	50 - 150	150 - 300	> 300
	HCB (µg/kg)	< 2	2 - 5	5 - 20	20 - 50	> 50
	Σ HCH (µg/kg)	< 10	10 - 30	30 - 100	100 - 250	> 250
	Σ PCB_7 ** (µg/kg)	< 50	50 - 150	150 - 500	500 - 1000	> 1000
	$TE_{PCDF/D}$ (ng/kg)	< 1.5 *	1.5 - 3	3 - 10	10 - 30	> 30
Taskekrabbe hepatopancreas (friskvektsbasis)	$TE_{PCDF/D}$ (ng/kg)	<10 *	10 -30	30 - 100	100 - 250	> 250

[1)] HCH: Heksaklorsykloheksaner, bl. a. lindan. Med ΣHCH forstås minimum sum av alfa-, beta- og gammaisomerene.
[2)] PCDF/PCDD: Polyklorerte dibenzofuraner/dibenzo-p-dioksiner ("dioksiner"). Innen PCDF/PCDD er det en mindre gruppe forbindelser som er sterkt til ekstremt giftige. Konsentrasjonen av disse stoffene angis her som sum toksisitetsekvivalenter (TE), dvs. ekvivalenter av den giftigste dioksinforbindelsen (2,3,7,8-TCDD). TE er innført istedenfor TCDD-ekvivalenter (som ble brukt i I. utgave av klassifiserings-systemet) fordi også en del andre stoffer (særlig non- og mono-orto PCB) har samme virkningsmekanisme som dioksinene og har fått beregnet toksisitetsekvivalentfaktorer. I klassifiseringstabellene er det imidlertid bare angitt TE-bidraget fra PCDF/PCDD, dvs. at verdiene er sammenlignbare med tidligere angivelser for TCDD-ekv. (Foreløpig er det ikke data nok til å anslå "bakgrunns"bidraget fra andre stoffer til TE).

Appendix E. Swedish assessment criteria for marine sediment and biota

Table E.1. Swedish status classes for trace metals (total analyses, mg/kg dw) and organic contaminants (µg/kg dw) in marine surface sediments (Swedish EPA 2000).

Substance	Class I	Class II	Class III	Class IV	Class V
HCB **(µg/kg dw)**	0	0–0.04	0.04–0.2	0.2–1	>1
CB28	0	0–0.06	0.06–0.2	0.2–0.6	>0.6
CB52	0	0–0.06	0.06–0.2	0.2–0.8	>0.8
CB101	0	0–0.16	0.16–0.6	0.6–2	>2
CB118	0	0–0.15	0.15–0.6	0.6–2	>2
CB153	0	0–0.03	0.03–0.3	0.3–3.5	>3.5
CB138	0	0–0.3	0.3–1.2	1.2–4.1	>4.1
CB180	0	0–0.1	0.1–0.4	0.4–1.9	>1.9
sum PCB7	0	0–1.3	1.3–4	4–15	>15
Total PCB	0	0–5	5–20	20–75	>75
sum HCH	0	0–0.03	0.03–0.3	0.3–3	>3
sum chlordane	0	0–0.02	0.02–0.08	0.08–0.3	>0.3
sum DDT	0	0–0.2	0.2–1	1–6	>6
Phenanthrene	0	0–10	10–30	30–100	>100
Antracene	0	0–2	2–8	8–30	>30
Fluoranthene	0	0–20	20–80	80–270	>270
Pyrene	0	0–12	12–50	50–200	>200
Benzo(a)anthracene	0	0–10	10–35	35–110	>110
Crysene	0	0–13	13–50	50–180	>180
Benzo(b)fluoranthene	0	0–50	50–150	150–400	>400
Benzo(k)fluoranthene	0	0–20	20–50	50–160	>160
Benzo(a)pyrene	0	0–20	20–60	60–180	>180
Benzo(g,h,i)perylene	0	0–30	30–100	100–350	>350
Ideno(cd)pyren	0	0–50	50–170	170–600	>600
Sum PAH11	0	0–280	280–800	800–2500	>2500
As **(mg/kg dw)**	<10	10–16	16–26	26–40	>40
Cd	<0.2	0.2–0.5	0.5–1.2	1.2–3	>3
Co	<12	12–17	17–24	24–34	>34
Cr	<80	80–112	112–160	160–224	>224
Cu	<15	15–30	30–60	60–120	>120
Hg	<0.04	0.04–0.10	0.1–0.27	0.27–0.72	>0.72
Ni	<33	33–43	43–56	56–79	>79
Pb	<31	31–47	47–68	68–102	>102
Zn	<85	85–128	128–196	196–298	>298

Table E.2. Swedish status classes for trace metals (mg/kg dw) in bladder wrack (Fucus vesiculosus) from all Sweden (Swedish EPA, 2000).

Substance	Class I	Class II	Class III	Class IV	Class V
As	<20	20–24	24–30	30–40	>40
Cd	<0.9	0.8–1.1	1.1–1.3	1.3–1.6	>1.6
Cr	<0.2	0.2–0.4	0.4–0.8	0.8–1.6	>1.6
Cu	<2.5	2.5–3.5	3.5–5.5	5.5–8	>8
Ni	<3.5	3.5–4.6	4.6–6.0	6.0–8.1	>8.1
Pb	<0.3	0.3–0.6	0.6–1.2	1.2–3	>3
Zn	<40	40–80	80–152	152–300	>300

Table E.3. Swedish status classes for trace metals (mg/kg dw) in blue mussel (Mytilus edulis) from the Baltic Sea (Swedish EPA, 2000).

Substance	Class I	Class II	Class III	Class IV	Class V
Cd	<4	4–4.8	4.8–6.4	6.4–8	>8
Cr	<2	2–3	3–4	4–6	>6
Cu	<10	10–15	15–20	20–30	>30
Hg	<0.2	0.2–0.4	0.4–0.7	0.7–1.2	>1.2
Ni	<4	4–5.6	5.6–7.6	7.6–10	>10
Pb	<2	2–5	5–12	12–30	>30
Zn	<120	120–204	204–300	300–504	>504
Sn	<1	1–1.5	1.5–2.5	2.5–4.0	>4.0

Table E.4. Swedish status classes for trace metals (mg/kg dw) in blue mussel (Mytilus edulis) from the Kattegat and Belt Sea (Swedish EPA, 2000).

Substance	Class I	Class II	Class III	Class IV	Class V
Cd	<1.3	1.3–1.7	1.7–2.2	2.2–3.0	>3.0
Cu	<8	8–10	10–14	14–16	>16
Hg	<0.5	0.5–0.7	0.7–0.9	0.9–1.2	>1.2
Ni	<1	1–1.5	1.5–2.0	2.0–3.0	>3.0
Pb	<0.9	0.9–1.8	1.8–3.2	3.2–6.0	>6.0
Sn	<0.2	0.2–0.26	0.26–0.34	0.34–0.4	>0.4

Table E.5. Swedish status classes for trace metals (mg/kg dw) in Baltic clam (Macoma baltica) from the Baltic Sea (Swedish EPA, 2000).

Substance	Class I	Class II	Class III	Class IV	Class V
Cd	<0.4	0.4–1	1–2	2–4	>4
Cr	<2	2–3	3–5	5–8	>8
Cu	<20	20–40	40–90	90–200	>200
Hg	<0.2	0.2–0.3	0.3–0.6	0.6–1	>1
Ni	<4	4–5.6	5.6–7.6	7.6–10	>10
Pb	<2	2–6	6–15	15–40	>40
Zn	<300	300–390	390–600	600–810	>810

Table E.6. Swedish status classes for trace metals (mg/kg dw (muscle, as ww)) and organic contaminants (µg/kg lipid) in liver from perch (Perca fluviatilis) from the Baltic Sea (Swedish EPA, 2000)

Substance	Class I	Class II	Class III	Class IV	Class V
Cd	<0.2	0.2–0.34	0.34–0.6	0.6–1	>1
Cr	<0.1	0.1–0.14	0.14–0.21	0.21–0.31	>0.31
Cu	<7	7–11	11–17	17–26	>26
Hg* (muscle, as ww)	<0.04	0.04–0.10	0.10–0.23	0.23–0.56	>0.56
Ni	<0.06	0.06–0.12	0.12–0.24	0.24–0.48	>0.48
Pb	<0.04	0.04–0.07	0.07–0.11	0.11–0.18	>0.18
Zn	<65	65–91	91–124	124–176	>176
p,p-DDE	0	0–10	10–90	90–300	>300
p,p-DDD	0	0–4	4–20	20–60	>60
p,p-DDT	0	0–5	5–30	30–70	>70
sum DDT	0	0–30	30–100	100–400	>400
a-HCH	0	0–4	4–8	8–10	>10
g-HCH	0	0–4	4–7	7–10	>10
HCB	0	0–4	4–7	7–10	>10
CB153	0	0–20	20–60	60–200	>200
CB118	0	0–8	8–20	20–70	>70

Table E.7. Swedish status classes for trace metals (mg/kg dw (* muscle, as ww)) and organic contaminants (µg/kg lipid) in liver from herring (Clupea sp.) from all Sweden (Swedish EPA, 2000)

Substance	Class I	Class II	Class III	Class IV	Class V
Cd	<0.3	0.3–0.8	0.8–2	2–5.4	>5.4
Cr	<0.2	0.2–0.3	0.3–0.46	0.46–0.7	>0.7
Cu	<7	7–11	11–15	15–23	>23
Hg* (muscle, as ww)	<0.01	0.01–0.02	0.02–0.04	0.04–0.09	>0.09
Ni	<0.1	0.1–0.18	0.18–0.33	0.33–0.62	>0.62
Pb	<0.05	0.05–0.09	0.09–0.16	0.16–0.3	>0.3
Zn	<63	63–88	88–120	120–158	>158
p,p-DDE	0	0–10	10–80	80–600	>600
p,p-DDD	0	0–3	3–30	30–300	>300
p,p-DDT	0	0–5	5–20	20–100	>100
sum DDT	0	0–30	30–200	200–1000	>1000
a-HCH	0	0–8	8–20	20–40	>40
g-HCH	0	0–7	7–20	20–30	>30
HCB	0	0–6	6–20	20–50	>50
CB153	0	0–20	20–80	80–300	>300
CB118	0	0–8	8–30	30–100	>100

Table E.8. Swedish status classes for trace metals (mg/kg dw) and organic contaminants (µg/kg lipid) in liver (or *muscle, as ww) from eelpout (Zoarces viviparus) (Swedish EPA, 2000)

Substance	Class I	Class II	Class III	Class IV	Class V
Cd	<0.35	0.35–0.60	0.60–0.98	0.98–1.6	>1.6
Cr	<0.15	0.15–0.27	0.27–0.51	0.51–0.89	>0.89
Cu	<7.6	7.6–14	14–28	28–54	>54
Hg* (muscle, as ww)	<0.03	0.03–0.05	0.05–0.08	0.08–0.13	>0.13
Ni	<0.2	0.2–0.34	0.34–0.56	0.56–0.9	>0.9
Pb	<0.03	0.03–0.05	0.05–0.10	0.10–0.17	>0.17
Zn	<97	97–136	136–184	184–243	>243
As* (muscle, as ww)	<0.15	0.15–0.20	0.20–0.30	0.30–0.41	>0.41
Cd* (muscle, as ww)	<0.0005	0.0005–0.001	0.001–0.0015	0.0015–0.0025	>0.0025
Cu* (muscle, as ww)	<0.7	0.7–1.4	1.4–2.8	2.8–5.6	>5.6
Ni* (muscle, as ww)	<0.35	0.35–0.60	0.60–0.98	0.98–1.7	>1.7
Pb* (muscle, as ww)	<0.05	0.05–0.09	0.09–0.15	0.15–0.25	>0.25
p,p-DDE	0	0–40	40–300	300–2000	>2000
p,p-DDD	0	0–6	6–40	40–200	>200
p,p-DDT	0	0–20	20–70	70–200	>200
sum DDT	0	0–60	60–400	400–2000	>2000
a-HCH	0	0–10	10–40	40–200	>200
g-HCH	0	0–10	10–40	40–200	>200
HCB	0	0–10	10–30	30–100	>100
CB153	0	0–40	40–200	200–1000	>1000
CB118	0	0–6	6–40	40–300	>300