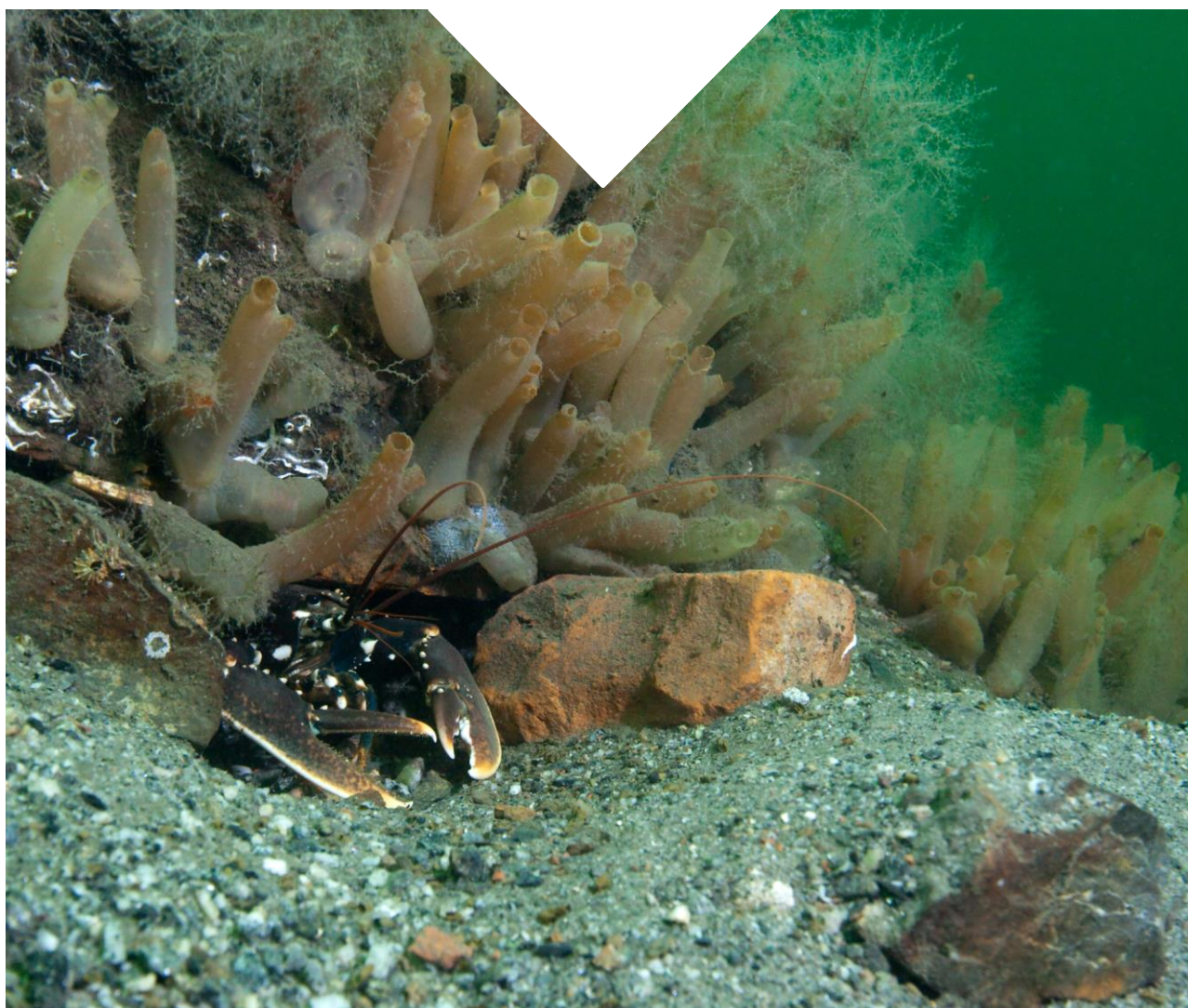


Risk assessment of contaminated sediments

Guidelines



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Veileder for risikovurdering av forurenset sediment; M-409
Guidelines for risk assessment of contaminated sediments

Sammendrag - summary

Miljødirektoratets veileder for vurdering av miljørisiko fra forurenset sediment fokuserer på risiko for spredning av hazardous substances fra sedimentene, virkninger på human helse og virkninger på økosystemet. Vurderingen foregår i tre trinn, der hvert trinn er mer lokalt forankret, mer arbeidskrevende og mindre konservativt enn foregående trinn. Veilederen er harmonisert med systemet for klassifisering av forurenset sediment. Veilederen, M-409, er i dette dokumentet oversatt til engelsk. The Norwegian environment agency guidelines for environmental risk assessment of contaminated sediments focus on the risk of release of hazardous substances from contaminated sediments, the impact on human health and the impact on the ecosystem. The assessment is carried out in a tiered approach at three levels, with increasing complexity and demand for local data. The greater the amount of local data, the less conservative the risk assessment. The guidelines are harmonised with the system for classification of contaminated sediments. This publication is a translation of the original Norwegian publication, M-409.

4 emneord

Risikovurdering, miljøgifter, sedimenter, tiltak, vurderinger

Keywords

Risk assessment, hazardous substances, sediments, remedial measures

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1. Introduction

These guidelines concern assessing the environmental risk posed by contaminated sediments in fjord and coastal waters, including harbours. They have been prepared for use by administrative personnel, problem owners, consultants and others in assessing the need for remedial measures in marine sediment areas. The Norwegian Environment Agency (NEA) has prepared several guidelines on sediment management (see *Figure 1*). For information regarding the basis of the threshold values included in the risk assessment, see Guidelines 02: 2013 *Klassifisering av miljøtilstand i vann* [Environmental quality classification of water bodies].

The guidelines for the environmental risk assessment of contaminated sediments provide guidance on the quantitative assessment of the risk of the spread of hazardous substances from contaminated sediments, as well as the risk of adverse effects on human health and on the ecosystem. The assessment is conducted in a tiered approach with three levels. Each level is increasingly based on local data, more labour-intensive and less conservative than the last. The guidelines are harmonised with the system for classification of contaminated sediments.

To aid the calculation of risk, a dedicated calculation tool has been prepared in Excel that includes all substance-related data and formulae featured in these guidelines (M-409). Hereafter, this is referred to simply as ‘the spreadsheet’.

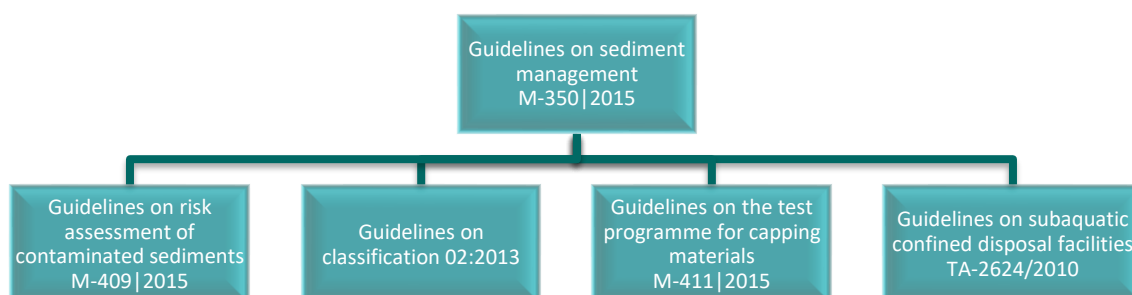


Figure 1 Relationship between the risk assessment guidelines and classification guidelines, background information documents and appendices

2. Background

2.1 What is risk?

The risk associated with an event is often presented as a product of the probability that the event will occur and the consequences of that event occurring (probability x consequences). An event is deemed high risk if the probability of the event is high or the consequences of the event would be severe, or both. The relationship between probability and consequences is illustrated schematically in Figure 2. In a risk assessment, each of the probability and consequence categories must be described unambiguously and as quantitatively as possible to enable the risk associated with the event to be classified.

Probability Consequences	Low	Medium	High
Limited	Low risk	Moderate risk	High risk
Moderate			
Major			

Figure 2 Risk matrix for the probability and and consequences of an event

The formal definition of risk is used most often in an analysis of events for which both probability and consequences are variable. These guidelines cover an assessment of the risk posed by sediments in their existing state. In cases such as this, processes such as the spreading of hazardous substances through diffusion and uptake by organisms will be occurring to a greater or lesser degree at all times, and the probability of the event is thus equal to 1. The probability of spreading via propeller erosion is dependent on the water depth. This probability is considered to be 1 in areas where there is ship traffic above sediments at a depth of less than 20 metres below normal sea level for sediments at greater depths. The risk assessment is thus first and foremost an impact analysis. Nevertheless, we use the term ‘risk assessment’ to refer to the assessment of the need for remedial measures to reduce risk to an acceptable level.

2.2 Aim of the risk assessment system –link with environmental objectives and the Norwegian Water Management Regulations

The aim of risk assessment is to describe the risk posed by contaminated sediments to the environment or to human health, so that one can then judge whether that risk is acceptable. As described in the NEA guidelines on sediment management (M-350/2015), such risk assessment is part of the procedure for the remediation of contaminated sediments.

All areas considered for remedial measures should have established environmental objectives and remediation objectives (should remedial measures be necessary) that describe the environmental quality and health status that one wishes to achieve in the area. The remediation objectives should be as realistic, valid and verifiable as possible to ensure that fulfillment of the objectives can be assessed. All sources of contaminants in an area contribute to a greater or lesser degree to the fact that a remediation objective has not already been achieved. Any remedial measures aimed at sediments must therefore be weighed against the gains that could be made through remedial measures targeting other sources of contaminants in the area.

The environmental objectives may have different levels of ambition and different weightings. The most important objectives relate to avoiding the spread of hazardous substances to new areas and avoiding adverse effects on human health (primarily because of seafood consumption) and on the marine ecosystem. This is reflected in the structure of the risk assessment system, which incorporates all these components.

The Norwegian regulations (the Water Management Regulation) came into force in 2007 and transposed the EU Water Framework Directive into Norwegian law. These regulations require to be an integrated system for the management of water from mountain top to fjord, and for river basin management plans to be prepared for all water bodies. These plans must describe how environmental objectives related to 'good ecological' and 'good chemical' status is to be achieved in all water bodies by 2021. Remedial measures targeting contaminated sediments (a Programme of Measures) may be appropriate for achieving these objectives, especially with respect to chemical status. The environmental objectives in the Water Management Regulations will provide the main guidelines for work concerned with the contaminated seabed because:

- Contaminated sediments are one of several sources from which hazardous substances may spread to water bodies, and remedial measures targeting the seabed may therefore be necessary to achieve the environmental objective.
- The environmental objective of good ecological status applies both to organisms living in seawater and freshwater and to organisms living on/in the sediment. The latter will be affected by the contamination level of the sediments.
- The environmental status of a water body will in most cases be monitored by means of sediment sampling.

NEA recommends using the threshold value for environmental quality Classes II in contaminated sediments (no chronic or acute effects on the biota) as an environmental target in areas where sources of contamination have been eliminated. This can be justified where analyses of discharges into the area show that this boundary can act as an appropriate target based on cost/benefit analyses, and where the target could be achieved using existing methods of remedial action.

The threshold value for environmental quality Classes III can be used as a remediation target if discharge into the area from land-based sources has not been stopped and industry and manufacturing are to be maintained. Opting for this remediation target could necessitate postponing the date at which the environmental objective for the waterbody will be achieved.

A lower ambition level (acceptance of a higher environmental quality class) or deferral of remedial measures may be acceptable if the following conditions are fulfilled:

- The risk assessment shows there to be low risk even with relatively high concentrations of hazardous substances in the sediments.
- A Level 3 risk assessment has been performed to verify the Level 2 risk assessment and reveals the effects of uptake of hazardous substances by the biota to be insignificant.
- The ecological status of the area is good according to the classification system of the Water Management Regulations.

When a lower ambition level is chosen, this should be entered in the spreadsheet used in the risk assessment.

2.3 Structure of the risk assessment system

The risk assessment is conducted at three levels as shown in *Figure 3*. The transition from one level to the next is characterised by:

- an increase in the complexity of the assessments
- greater reflection of local conditions
- reduced uncertainty and less conservative calculations and estimates

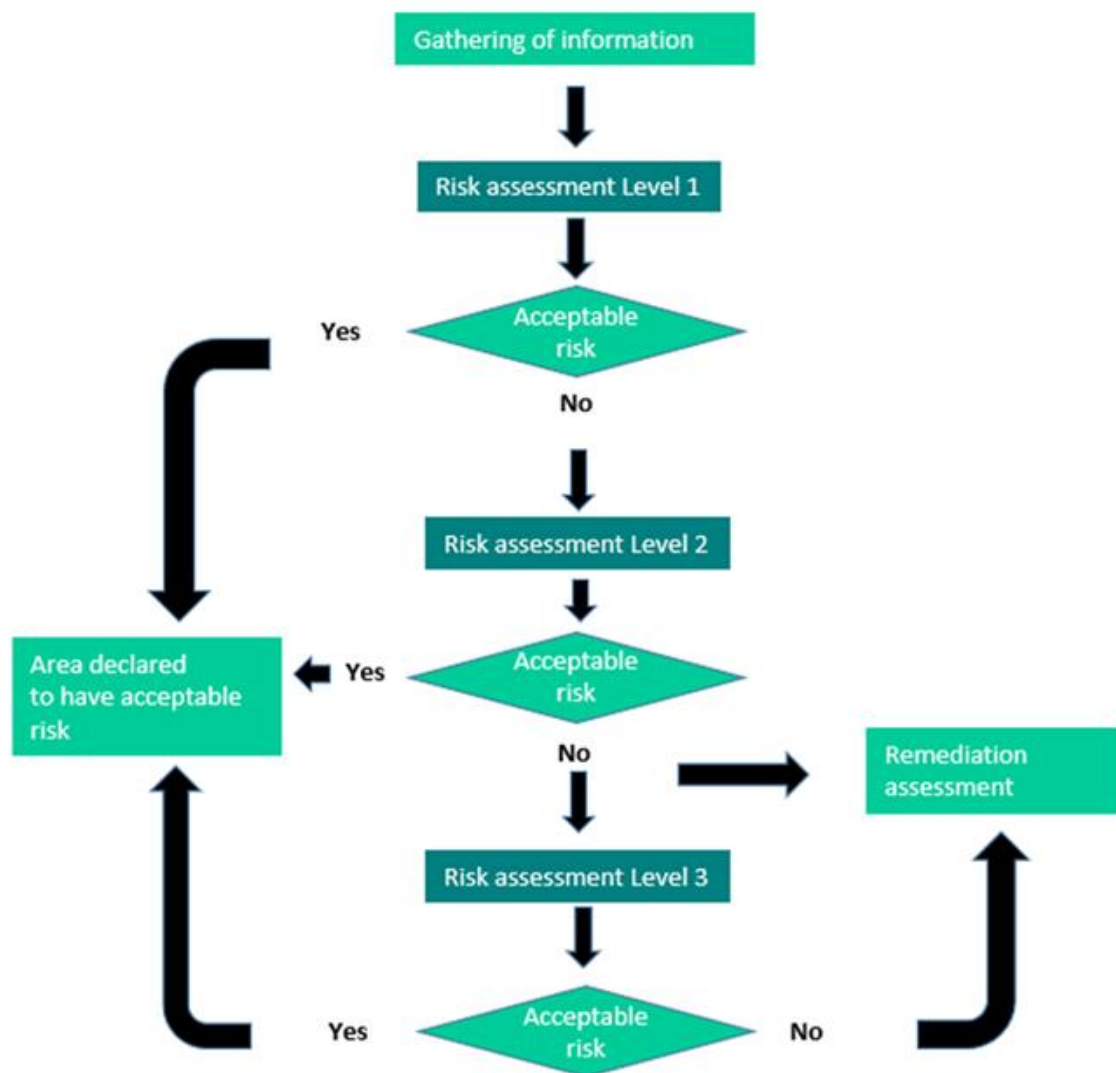


Figure 3 General structure of the risk assessment system for contaminated sediments

The risk assessment should initially be conservative to avoid assigning a clean bill of health to areas where remedial measures are in fact required. This means considering all sources of uncertainty in the evidence base. As one progresses through the three levels, the assessment will become increasingly based on local conditions, the uncertainty associated with the calculations will diminish, and the risk estimate will become more realistic, more precise and less conservative. This will ensure that remediation takes place only where necessary.

Level 1 of the risk assessment concerns only ecological risk. If a risk assessment related to human health is required, Level 2 must be performed.

[Appendix V](#) provides a brief checklist of the steps involved in carrying out the risk assessment.

2.4 Limitations to using the risk assessment tool

The assessment tool is to be used for risk assessment of seabed sediments based on a quantitative analysis of their hazardous substances content and their toxicity.

For rock and gravel seabeds, sampling for quantitative analysis may not be possible and the guidelines are therefore not applicable. However, coarse sediments will pose an environmental risk only in exceptional circumstances, since hazardous substances are normally bound to fine particles.

A risk assessment in accordance with these guidelines is a precursor to any planning of remedial measures, but it is not part of the actual planning. In some cases, however, the risk assessment may be used to help optimise interventions, for example by repeating a risk analysis taking account of the environmental quality that would be expected after the remedial action under consideration has been performed.

2.5 Definition of sediment areas included in the risk assessment

Prior to carrying out the risk assessment, an appropriate geographical demarcation of the total sediment area to be included in the assessment is required. These should be typical administrative definitions, such as areas that require specific dietary advice, or areas that local data show to stand out with respect to:

- Agreed/potential areas for remedial measures
- Presumed sources of contaminants,
- Ship traffic,
- Topography,
- Gradients in the concentrations of hazardous substances in the seabed

Dividing the area into smaller subareas may be appropriate if the total sediment area is large, or if indicated by local data (see above). The risk assessments are intended to form a basis for an analysis that involves evaluating the need for remedial measures in each subarea. The assessments should therefore provide a basis for ranking the various subareas in terms of the importance of remedial measures.

New sediment-related data generated as part of the risk assessment may sometimes lead to subareas being redefined, for example, upon discovery of a small and highly contaminated region (a 'hotspot'). The area of each individual subarea corresponds to area A_{sed} in *Figure 4*. If parts of the sediment area/subarea are affected by erosion because of shipping (propeller flow or water jets), this subarea must also be demarcated and treated separately in the risk assessment. Justification must be provided for the selection of sediment areas to be included in risk assessments.

The following areas are defined for an area/subarea that is to undergo risk assessment (*Figure 4*):

- A_{sed} : Area covered by the risk assessment.
- A_{ship} : Area exposed to sediment erosion due to propeller- and jet-induced resuspension. The area is defined as the area within A_{sed} that includes fairways and areas of ship manoeuvring, and where the depth is also 20 m or less. Studies indicate that even larger vessels do not erode sediment at depths greater than this. See also [section 4.2.2](#) and [Box 6](#) for discussion of substance transport and calculation of sediment spread for A_{ship} .
- $A_{sed} - A_{ship}$: Area unaffected by shipping

Each of the subareas of A_{sed} shown in *Figure 4* must undergo a separate risk assessment.

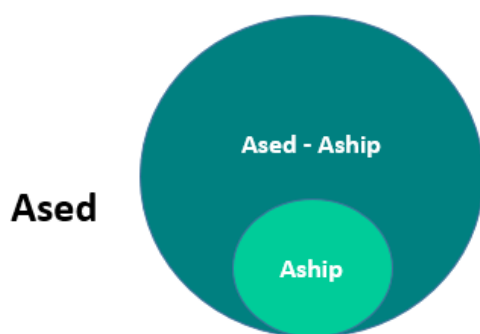


Figure 4. The area subjected to sediment resuspension by shipping traffic (A_{ship} = lighter coloured area) comprises part of the total sediment area (A_{sed} = entire area within the large circle) included in the risk assessment.

2.6 Risk assessment is dependent on seabed activities and area size

The guidelines on sediment management (M-350/2015) distinguish between areas on the basis of the size of the area and volume of sediment affected by a seabed activity (*Table 1*).

Table 1. Classification based on the area and volume affected by seabed activities

Category	Volume	Area
Minor activity	< 500 m ³	< 1 000 m ²
Moderate activity	> 500 m ³ and < 50 000 m ³	> 1 000 m ² and < 30 000 m ²
Major activity	> 50 000 m ³	> 30 000 m ²

These risk guidelines have primarily been developed for areas that fall within the scope of county-level remediation plans, i.e. medium-sized fjord and harbour areas. For all major activities affecting the sediment (area > 30 000 m² or comprises > 50 000 m³ sediment), the environmental

authorities recommend a Level 3 risk assessment to ensure that the assessment is tailored as closely as possible to the local conditions.

There may also be a need to assess sediments in marinas, outside private jetties and in other smaller and medium-sized areas (< 30 000 m²), where a full risk assessment may be excessive. In such cases, a degree of discretion should be exercised regarding the scope of the assessment, and the principles outlined in the guidelines may be considered advisory. *Table 2* provides an overview of the extent to which it is appropriate to carry out a risk assessment depending on the action and on area size.

Table 2. Overview of seabed activities that typically trigger the need for testing and risk assessment. In each case, however, a specific assessment must be performed and a discretionary judgement made by the authorities (X = testing/risk assessment may be required, XX = testing/risk assessment must be performed)

Activity		Risk assessment
Dredging	Minor (< 1 000 m ²)	
	Moderate (> 1 000 m ² and < 30 000 m ²)	X
	Major (> 30 000 m ²)	XX
Dumping	Minor (< 1 000 m ²)	
	Moderate (> 1 000 m ² and < 30 000 m ²)	
	Major (> 30 000 m ²)	X
Covering	Minor (< 1 000 m ²)	
	Moderate (> 1 000 m ² and < 30 000 m ²)	X
	Major (> 30 000 m ²)	XX
Filling	Minor (< 1 000 m ²)	
	Moderate (> 1 000 m ² and < 30 000 m ²)	
	Major (> 30 000 m ²)	X

For areas < 30 000 m², the minimum requirement should be for data on the concentration of hazardous substances in the sediment to be obtained from three stations, and for these values to be compared with the threshold values for Level 1 in the guidelines. The hazardous substances selected for analysis should include at least those presented in *Table 3*, but omission of toxicity testing should be acceptable. This will in many cases be sufficient to obtain an understanding of risk and to provide grounds for planning remedial measures if required. The need for remedial measures must be decided based on the area's presumed environmental significance and type of usage. If the area is used for bathing, an assessment of the risk to human health should be performed with an emphasis on ingestion of and contact with contaminated sediments, particles and water. Such areas will often comprise a smaller subarea or a border zone of a larger basin, which may reduce their own significance with respect to risk.

2.7 Uncertainty in assessments

There will always be uncertainty related to the assessment of environmental risk and this uncertainty is difficult to quantify. The guidelines have taken this uncertainty into account by making the risk assessments deliberately conservative. Thus:

- When setting the threshold values for acceptable risk in Level 1, uncertainty in the toxicity data has been allowed for using application factors (a factor by which the threshold value is multiplied to account for uncertainty in the underlying data, see M-241/2014). The same

principles have been applied when setting the threshold values for ecological risk ([Appendix II](#)) and the risk to human health ([Appendix III](#)).

- The partition coefficients between sediment and water (K_d) and between water and organisms (BCF) for the individual hazardous substances have been chosen conservatively, i.e. they should ensure that the transport of hazardous substances from the sediment to other parts of the ecosystem, including seafood, is not underestimated. The guidelines also outline ways of establishing more realistic partition coefficients for a given situation ([Appendix VIII](#)).
- Other proposed default values and calculation tool variables ([Appendix VII](#)) are also conservative for the same reason but can be replaced by more realistic values (Level 3).
- When analysis results are below the limit of detection, it is recommended that half the limit of detection be used as the concentration in calculations.

3. Risk assessment Level 1

3.1 General

Level 1 is a simplified risk assessment in which the concentration of hazardous substances within the sediment, and the sediment toxicity, are compared with the threshold values for the ecological effects of contact with the sediment. Level 1 concerns only the risk of ecological impact, not the risk to human health.

The threshold values are based on conservative assumptions about exposure routes, bioavailability and the likelihood of spreading to other parts of the ecosystem. This is based to a large degree on the EU Technical Guidance Document on Environmental Risk Assessment (EU-TGD). Given compliance with the threshold values, the risk posed by the sediment is considered to be insignificant and remedial measures is not required. If the threshold values are exceeded, one must proceed to Level 2.

Level 1 does not entail assessment per se but is purely a classification of the sediments with respect to the threshold values. This is illustrated by the fact that the threshold values for almost all substances correspond to the boundary between environmental quality Classes II and III in NEA guidelines 02: 2013¹.

3.2 Information required for Level 1

Implementation of Level 1 sets a minimum requirement in terms of data on the hazardous substances content and toxicity of sediments. In some cases, these data will already be available in the form of results from previous sediment tests, but it will usually be necessary to conduct new analyses and toxicity tests. The data required will depend on the area's topography, the complexity of the sources of contaminants, the sediment type (fine or coarse grained), water depth and usage of the area. These factors also have a major impact on the approach to sediment sampling. Further guidelines and suggestions for sampling methods, quality of samples, sample treatment, and physical, chemical and toxicological analyses are presented in [Appendix VIII](#).

¹ One exception is TBT where the Level 1 threshold value of 35 µg/kg will be retained until further notice, whereas the boundary between Classes II and III is 5 µg/kg.

3.2.1 Area subdivision and sampling programme

In the following, the term ‘sediment area’ encompasses both the areas of seabed to be risk-assessed and the bodies of water above them. In areas shallower than 20 m, samples must be collected from at least five sediment stations, where each station can represent a maximum of 10 000 m² of seabed. Where the water depth is greater than 20 m, greater homogeneity in sediment structure can be expected. Each station can therefore represent up to 40 000 m² of seabed. It is important that the network of stations provides a representative picture of sediment contamination for both A_{ship} and $A_{\text{sed}} - A_{\text{ship}}$. If A_{sed} encompasses areas where contact with water and sediment may pose a risk to human health (bathing areas), the network of stations should be expanded to ensure that it is also representative of this area. Contamination of the land component of bathing beaches is excluded and is covered by the guidelines on health-based environmental quality classification of contaminated ground (TA-2553/2009).

In areas where the seabed is reasonably uniform, the most straightforward way to position the stations is in a grid. For areas with variable depth, complex topography (for example, multiple basins), varying sediment types, varying types of usage (recreation and industry in the same basin) or where subareas are exposed to resuspension by ship propellers, the stations should be positioned such that the various subareas are covered. The more variable the sediment area, the greater the number of stations that must be set up. All stations must be positioned using GPS coordinates.

The sample from each station should be a composite sample composed of four discrete samples collected in parallel from random positions within the area covered by the station. All analyses are performed on this composite sample. The sampling should cover the topmost, biologically active layer of the sediment, which in most cases will be within the upper 0-10 cm.

Analyses of sediment, water and biota must be carried out by laboratories accredited for the specific analyses performed.

General requirements for selection of sampling equipment and management of samples are outlined in NS-EN ISO 5667-19:2004 (which has replaced the earlier NS 9422). NS-EN ISO 5667-19:2004 also specifies which samplers are suitable for different tests and sediment conditions. Further advice on sampling is provided for different sediment conditions in [Appendix VIII](#). The same sampling methods and analyses apply for the purposes of risk assessment and classification. In some cases, it may be necessary to expand the analysis programme relative to what was planned at the beginning of the risk assessment process. One should therefore consider collecting a larger quantity of sediment than that required by the analyses when first in the field and storing the extra material in frozen form.

3.2.2 Parameter selection

Table 3 provides a minimum list of the physical, chemical and toxicological parameters that must be analysed/tested in the composite sample from each station to characterise the sediment. The parameter list should be adjusted and expanded based on local conditions if necessary, for example where knowledge about the source(s) of hazardous substances suggests that an alternative selection would be more appropriate². In such cases, justification must be provided for the selection. For further information see [Box 3](#), which provides an overview of all compounds for which threshold values are available.

² For further information about possible hazardous substances associated with industries and contaminant types, see NEA's Guidelines on sediment management, M-350/2015, Appendix X, Table X-1.

Table 3 Recommended minimum list of analytical parameters for characterisation of sediment samples in preliminary studies as part of Level 1 of the risk assessment.

Group	Parameter
Physical characterisation	Water content, content of silt (< 63 µm) and clay (< 2 µm)
Heavy metals	Hg, Cd, Pb, Cu, Cr, Zn, Ni, As
Non-chlorinated organic compounds	Single compounds in PAH ₁₆
Chlorinated organic compounds	Individual congeners in PCB ₇
Other analytical parameters	Total organic carbon (TOC), tributyl tin (TBT)
Toxicity tests	<i>Skeletonema</i> , <i>Tisbe</i> and <i>Crassostrea</i> (porewater) DR CALUX (extract)

3.2.3 Toxicity tests

General toxicity tests should be performed to detect possible toxic effects of substances not included in the chemical analyses, as well as interacting effects of substances. The tests should preferably be conducted on samples from each station, as for the chemical analyses, but for relatively homogeneous seabed areas, it will be sufficient to perform tests on one composite sample from the area shallower than 20 m and another composite sample from the area deeper than 20 m. This will provide an ‘average’ toxicity that is sufficient to determine whether the sediment in each subarea fulfils the criteria for acceptable risk in Level 1.

A minimum of 15 litres of sediment must be collected from each area to be tested. This will also be enough for a whole sediment toxicity test in Level 2 if required. The samples should be collected such that deeper anoxic layers are excluded. Normally the samples will span approximately the upper 10 cm of the sediment. It may also be practical (but is not mandatory) to collect the samples from the same stations as those for the chemical analyses. When treating the samples, it must be ensured that the porewater does not drain out (especially important for sandy sediments). The porewater should be extracted from the sediment within 1-2 weeks to avoid the formation of compounds such as ammonia. The porewater must be passed through a 0.2 µm filter prior to testing. If the tests cannot be performed immediately after extraction, the porewater must be frozen down.

The toxicity tests must be performed on at least two of a possible three different types of organism: microalgae (the marine alga *Skeletonema costatum*), bottom-dwelling crustaceans (e.g. *Tisbe battagliai*) and invertebrate larvae (e.g. the larva of the oyster *Crassostrea gigas*). The tests are all standardised screening tests that can be performed during a week.

A specific test (DR CALUX) for dioxins or dioxin-like hazardous substances is recommended if the presence of these substances in the sediments is suspected. The test is carried out on an organic extract of the sediment.

The tests are described briefly in [Box 1](#). A more in-depth description can be found in [Appendix VIII](#).

Box 1. Toxicity tests in Level 1 risk assessment

Porewater toxicity is assessed using three standardised tests covering three types of organisms:

1. Growth inhibition test with the marine diatom *Skeletonema costatum*. Algal growth rate is measured in a dilution series of porewater in a seawater-based growth medium. From a response curve showing growth rate as a function of porewater dilution, the concentration (in %) that produces 50% inhibition of algal growth (EC_{50}) can be calculated. From EC_{50} , $TU=100/EC_{50}$ is calculated to obtain a unit proportional to the toxicity.
2. Mortality test with the benthic marine copepod *Tisbe battagliai*. The test is conducted similarly to the algal test on a dilution series of porewater. The concentration (in %) that produces 50% mortality (LC_{50}) is calculated. TU is calculated as $100/LC_{50}$.
3. Mortality test with larvae of the American oyster *Crassostrea gigas*. This test is carried out in the same way as the copepod test.

At least two of these three tests must be performed.

Sediment extract in an organic solvent is assayed with the DR CALUX *in vitro* bioassay to measure the effects of dioxins and dioxin-like substances. The assay involves exposing cell cultures to different doses of the organic extract. The result is reported as toxicity equivalents of dioxin (TEQ ng/kg), i.e. as a concentration, as for the results of the chemical analyses. This test may be omitted if there is no suspicion that dioxins or dioxin-like substances are present.

3.3 Threshold values

The threshold values for insignificant risk in Level 1 are based on knowledge of the toxicity of the various substances and what exposure is acceptable to the environment. Such toxicity assessments are increasingly standardised and harmonised internationally. The threshold values in Level 1 correspond to the boundary between Class II and Class III of NEA guidelines 02: 2013. It is important to emphasise that these threshold values are based on the ecological impact of the substances, not their impact on human health. Assessment of the risk to human health is carried out in Level 2.

When establishing threshold values, strong emphasis has been placed on using the most up-to-date threshold values for toxicity developed by the EU wherever these are available. In the absence of such values, ecological threshold values have been developed based on an up-to-date review of the toxicity of the individual substances, and the guidelines for application factors specified by the EU to establish threshold values for insignificant ecological effects. The underlying principle is that the less one knows about the toxicity of substances in marine sediments, the larger the application factor that must be used to obtain safe threshold values for ecological effects. There are few toxicity data available for marine sediments. Hence the toxicity data from tests on aquatic organisms have been used - preferably marine organisms where sufficient data are available; otherwise, freshwater organisms. For substances lacking toxicity data, application factors will be high and threshold values very conservative.

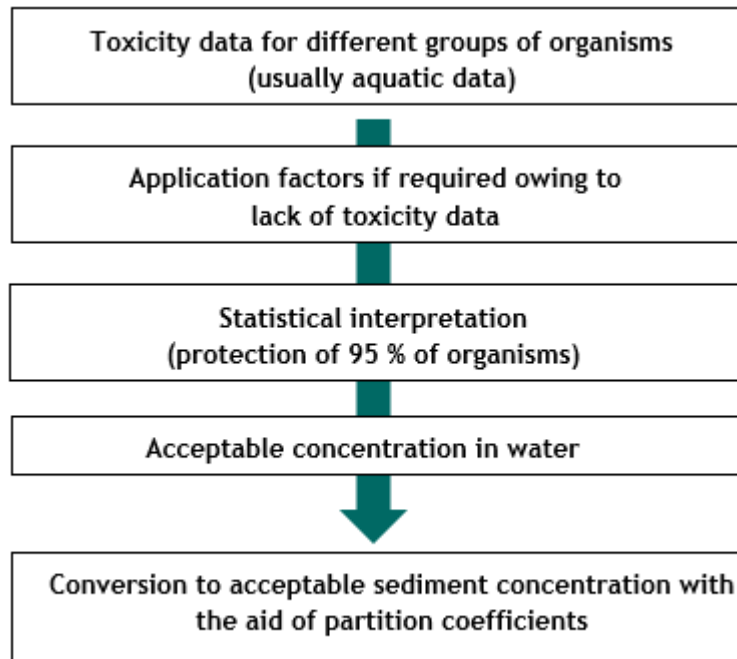
A formal fixing of ecological threshold values based on the EU TGD will therefore lead to unrealistically low values for many new and sparsely studied substances. Some values will be in the range of concentrations designated as background level in Norwegian coastal waters; others will also be below the limits of detection that can be achieved with today's analytical methods. For the risk assessment system to be practicable, it has therefore been necessary to abandon the principles of the EU TGD for several substances, by excluding application factors, setting limits directly with reference to similar substances or groups of substances that have been studied more thoroughly, or by adjusting limits upwards to levels that can be analysed in practice. For these substances, threshold values will be revised as the evidence base improves. NEA report M-241/2014 shows how threshold values have been calculated for individual substances and the adjustments that have been made in relation to the EU TGD.

It is important to emphasise that the threshold values for Level 1 in the guidelines only indicate the risk of the substances having an ecological impact, not an impact on human health. For certain substances, the threshold values in Level 1 are so high that sediments that do not exceed these values may nevertheless still pose an unacceptable risk to human health in Level 2. Level 2 must therefore be performed if the risk to human health is to be assessed.

[Box 2](#) shows the process by which threshold values for ecological risk are derived. For further details, see NEA report M-241/2014. [Box 3](#) shows the recommended threshold values for Level 1.

Box 2 Derivation of threshold values for ecological risk

Threshold values are based on a hazard assessment of substances and on the expected environmental exposure to these substances. Since there are still very few toxicity data available for contaminated sediments, toxicity data from aquatic environments are used as a starting point and are converted into sediment concentrations with the aid of partition coefficients. Where toxicological data are incomplete, application factors are used. The workflow is illustrated in the schematic below.



For some new substances with sparse knowledge, it has been necessary to deviate from this process to avoid unfeasibly low threshold values. This has been done either by removing application factors, setting the threshold value directly based on values for similar substances, or adjusting the limit upwards to a level where analysis becomes feasible (NEA report M-241/2014).

Most metals occur naturally in sediments, and the recommended threshold values indicate the maximum permitted concentration including the natural background concentration. The upper limit for the background concentration corresponds to the boundary between Classes I and II. If reliable values are available for local background concentrations, these may be used instead.

Little is known about the background levels of most organic hazardous substances in water and sediment, and background concentrations are assumed to be zero. Background levels are therefore not incorporated in the threshold values for organic hazardous substances. The threshold values for organic hazardous substances are calculated as total concentrations, using the equilibrium distribution between water and sediment (defined by the partition coefficient K_d), which is

dependent on the innate properties of the substances and the organic carbon content of the sediment. In the calculations of threshold values in Level 1, the organic carbon content is set conservatively at 1%. If the measured percentage of organic carbon in the sediment is found to differ from this value, K_d should be adjusted as part of the Level 2 and Level 3 risk assessment. The procedure for this normalisation is described in [Box 10](#).

Methylmercury

No specific threshold value has been established for methylmercury in sediment. In larger areas where high concentrations of mercury have been detected and remedial measures are to be implemented, a Level 3 risk assessment should be conducted to assess the risk posed by methylmercury. Methylmercury levels in the biota should then be quantified and the results compared with the threshold value in the Water Management Regulations. The remedial actions selected should also be evaluated for their potential to lead to the formation and spreading of methylmercury.

Total Hydrocarbon Content (THC)

THC is not a separate parameter since THC toxicity is covered by assessment of the risks posed by PAHs.

PCBs

Threshold values in Level 1 are given only for total PCB₇, and not for each individual congener. The same applies to classification. This is because toxicity data are available for only a minority of congeners.

PAHs

Threshold values are provided both for individual substances and for total PAH₁₆. Where a risk assessment finds that the threshold value has been exceeded, individual compounds must be evaluated and not total PAH₁₆. In larger areas where seabed activities are being studied and high PAH concentrations have been detected, a Level 3 risk assessment should be conducted to identify the risks posed by the individual compounds. This is important to ensure that local factors such as bioavailability are considered.

Dioxins and dioxin-like substances

Both the DR CALUX test and chemical analysis of dioxins and dioxin-like compounds yield results in the form of concentrations (as toxicity equivalents). Since the DR CALUX test covers all dioxin-like hazardous substances, not just those encompassed by a dioxin analysis, it is considered suitable for Level 1 assessment of areas where there is no reason to suspect dioxin contamination. The threshold value for direct chemical analysis of dioxins has therefore not been included among the threshold values for Level 1.

Tributyltin (TBT) and Triphenyltin (TFT)

TBT and TFT pose a specific challenge in the risk assessment. There are strong grounds to believe that TBT and TFT are highly toxic to several types of marine organisms, and the threshold values for ecological impact are therefore set as low as 0.002 and 0.036 µg/kg, respectively, in sediment (NEA guidelines 02:2013). Levels as low as these are almost impossible to analyse, and since the substances are only moderately degradable in sediment, threshold values will be exceeded almost everywhere. There is also a great deal of evidence to suggest that the sources of TBT and TFT in the marine environment are not yet under control, and in very many cases, there is therefore little to be gained from targeting sediment with remedial actions solely because of TBT or TFT. A threshold value of 35 µg TBT/kg will therefore be retained in Level 1 until further notice, even though this

deviates from the boundary between Classes II and III in the classification system (5 µg TBT/kg and 5 µg TFT/kg). The threshold value of 35 µg/kg nevertheless ensures that at least 75% of active and decommissioned shipyards in Norway will be obliged to proceed to Level 2 purely because of TBT or TFT (NEA guidelines 02:2013).

Estimated porewater concentration compared with PNEC

For certain substances, the spreadsheet may show that a sediment that just falls within the threshold values in Level 1 nevertheless yields theoretical porewater concentrations that exceed the threshold values in water in Level 2 ($PNEC_w$) of the same substances. In such cases, the results from Level 2, not Level 1, should take precedence in the risk assessment.

Box 3 Threshold values for Level 1 (ecological risk only)

All concentrations are on a dry weight basis.

Substance	CAS no.	Threshold value = Class II/III boundary
Metals		mg/kg
Arsenic	7440-38-2	18
Lead	7439-92-1	150
Cadmium	7440-43-9	2.5
Copper	7440-50-8	84
Chromium total (III + VI)	7440-47-3	660
Mercury	7439-97-6	0.52
Nickle	7440-02-0	42
Zinc	7440-66-6	139
PAHs		µg/kg
Naphthalene	91-20-3	27
Acenaphthylene	208-96-8	33
Acenaphthene	83-32-9	96
Fluorene	86-73-7	150
Phenanthrene	85-01-8	780
Anthracene	120-12-7	4.6
Fluoranthene	206-44-0	400
Pyrene	129-00-0	84
Benzo(a)anthracene	56-55-3	60
Chrysene	218-01-9	280
Benzo(b)fluoranthene	205-99-2	140
Benzo(k)fluoranthene	207-08-9	135
Benzo(a)pyrene	50-32-8	183
Indeno(1,2,3-cd)pyrene	193-39-5	63
Dibenzo(a,h)anthracene	53-70-3	27
Benzo(ghi)perylene	191-24-2	84
Total PAH ₁₆		2000
Other organic compounds		µg/kg
DDT	see footnote A	15
Tributyltin (TBT ion, footnote B)	688-73-3; 366643-28-4	35
Lindane	608-73-1	0.074

- A. DDT total is 1,1,1-trichloro-2,2 bis (p-chlorophenyl) ethane (CAS number 50-29-3; EU number 200-024-3); 1,1,1-trichloro-2 (o-chlorophenyl)-2-(p-chlorophenyl) ethane (CAS number 789-02-6; EU Number 212-332-5); 1,1-dichloro-2,2 bis (p-chlorophenyl) ethylene (CAS number 72-55-9; EU Number 200-784-6); and 1,1-dichloro-2,2 bis (p-chlorophenyl) ethane (CAS number 72-54-8; EU Number 200-783-0)
- B. Administrative boundary that does not correspond to the boundary between Classes II/III (5 µg/kg).

Continued next page

Box 3 (continued) Threshold values for Level 1 (ecological risk only)

All concentrations are on a dry weight basis.

Substance	CAS no.	Threshold value = Class II/III boundary
Other organic compounds		µg/kg
Hexachlorobenzene	118-74-1	17
Pentachlorobenzene	608-93-5	400
Trichlorobenzene	12002-48-1	5.6
Hexachlorobutadiene	87-68-3	49
Pentachlorophenol	87-86-5	14
Octylphenol	140-66-9 (1806-26-4)	0.27
Nonylphenol	84852-15-3	16
Bisphenol A	80-05-7	1.1
Tetrabromobisphenol A	79-94-7	108
Pentabromodiphenylether (sum BDE 28, 47, 99, 100, 153 and 154)	32534-81-9	62
Hexabromocyclododecane	see footnote C	34
Perfluorooctane sulphonate (PFOS)	1763-23-1	0.23
Diuron	330-54-1	0.71
Irgarol	28159-98-0	0.036
PCB ₇ (sum PCB 28, 52, 101, 118, 138, 153 and 180)	1336-36-3	4.1
Triphenyltin	892-20-6, 900-95-8, 76-87-9, 639-58-7	35
Dodecylphenol with isomers	121158-58-5, 27193-86-8	4.4
Di(2-ethylhexyl)phthalate (DEHP)	117-81-7	10000
Perfluorooctanoic acid (PFOA)	3825-26-1 and similar substances	71
C10-13 chloralkanes	85535-84-8	800
Chlorinated paraffins (medium-chain)	85535-85-9	4600
Dioxins and dioxin-like compounds (Total TEQ)	see footnote D	0.00086
Decamethylcyclopentasiloxane (D5)	541-02-6	44
Tris(2-chlorethyl)phosphate (TCEP, phosphororganic flame retardants)	115-96-8	72
Diflubenzuron	35367-38-5	0,2
Teflubenzuron	83121-18-0	0.0004
Triclosan	3380-34-5	9.3
Alachlor	15972-60-8	0.3
Chlorfenvinphos	470-90-6	0.5
Chlorpyrifos	2921-88-2	1.3
Endosulfan	115-29-7	0.073
Trifluralin	1582-09-8	1600

Toxicity tests

Porewater	<i>Skeletonema</i>	TU < 1.0
	<i>Tisbe</i>	TU < 1.0
	<i>Crassostrea</i>	TU < 1.0
Org. extract	DR CALUX	TEQ < 50 ng/kg

- C. Total 1,3,5,7,9,11-Hexabromocyclododecane (CAS 25637-99-4), 1,2,5,6,9,10- Hexabromocyclododecane (CAS 3194-55-6), α-Hexabromocyclododecane (CAS 134237-50-6), 8-Hexabromocyclododecane (CAS 134237-51-7) and γ- Hexabromocyclododecane (CAS 134237-52-8)
- D. Dibenzodioxines, dibenzofurans and dioxin-like PCB compounds (DL-PCBs), see footnote 9 in Annex I in Directive 2013/39/EU priority substances in the field of water policy. TEQ: toxicity equivalents in accordance with World Health Organisation 2005 "Toxic Equivalence Factors".

3.4 Assessment of results and conclusions

3.4.1 Declaring an area to have acceptable risk

The results of the analyses should be compared with the threshold values given in Box 3. In this comparison, it is the average level of each hazardous substance that should be compared with the threshold value, not the level from the most polluted station (maximum level). This is because it is the overall risk posed by an area that is to be assessed, not just the risk from a single sampling point. To be on the safe side, when the results of the analysis are below the limit of detection, a value corresponding to 50% of this limit should be entered in the spreadsheet as the relevant concentration.

Sediments can be considered to pose an acceptable risk if:

- The average concentration of each hazardous substance across all samples (at least 5) is lower than the threshold value for Level 1, and no single concentration is greater than the larger of:
 - 2 x threshold value,
 - the boundary between Classes III and IV for the substance.
- The toxicity of the sediment lies below the threshold values for all tests.

If the results of the analysis show that exceedance of the threshold value is obviously related to only one or a few of the stations, one should consider whether it is appropriate to declare a demarcated zone within the area as contaminated (a 'hotspot'), while the remainder of the area can be considered to pose acceptable risk. This requires that the same substances have been analysed at all stations. With the number of stations required by the risk guidelines, only a rough geographical demarcation of such subareas will be possible. Additional sampling will often be required for the demarcation to be performed with a sufficient degree of certainty. If further sampling is required, the option of implementing remedial measures in only a demarcated subarea should be considered.

If the between-sample variation in concentrations is such that the ratio of the median value to the highest observed value is less than 2, this shows that the degree of contamination is reasonably homogeneous across all stations and is well represented by the average concentration. In such cases, the highest concentration provides no indication of any 'hotspots' in the sediments and expanded sampling of the same area would not notably change the outcome of the risk assessment.

It should be emphasised again that Level 1 concerns only ecological risk. If the environmental objectives for an area also relate to human health, or it is desirable for other reasons to carry out a risk assessment related to human health, then Level 2 must be performed, even if the area can be considered to pose acceptable ecological risk based on Level 1.

4. Risk assessment Level 2

4.1 General

The aim of Level 2 is to determine whether the sediments pose an acceptable or unacceptable risk regarding environmental and health-related adverse effects. In Level 2, the risk posed by the sediments is judged in relation to environmental objectives and the associated acceptance criteria for an area. The guidelines cover three independent assessments that correspond to NEA's three ambition levels for the environmental quality desired:

2A. Risk of spreading is assessed based on calculated transport of hazardous substances from the sediment to the water masses via diffusion and bioturbation, resuspension primarily because of ship traffic, and uptake by organisms and flow through the food web.

2B. Risk to human health is assessed since relevant transport routes to humans according to how a sediment area is used: harbour-related activities, recreation, harvesting of seafood, etc. The main route of exposure is through consumption of fish and shellfish, but ingestion of and contact with sediment and water are also included where they may be relevant in the context of recreation and bathing.

2C. Risk of impact on the ecosystem is assessed based on the calculated concentrations of hazardous substances to which organisms in water and sediment are exposed compared to the relevant threshold values for adverse effects. The results of toxicity tests from Level 1 and the whole sediment test in Level 2 are also considered.

The relevant transport pathways from the sediment are shown in simplified form in *Figure 5*. Simplified schematic showing routes for the spread of hazardous substances from the sediment to other parts of the ecosystem. In Level 2, calculations are performed to estimate the importance of these transport routes. The guidelines propose that typical variables, constants and coefficients should be used in the calculations (referred to as default values), but if reliable local values exist, these should be used instead. [Box 4](#) provides an overview of the data required to perform the calculations in Level 2 and other important information that can be used in interpreting the results from Level 2.

Level 2 also requires testing of the toxicity of the sediment to benthic infauna (whole sediment test). One can choose to test either the lugworm *Arenicola marina* or the crustacean *Corophium volutator*. The test records both behaviour and survival of experimental animals following exposure. A mortality rate of more than 20% is considered significant and is set as a threshold for unacceptable risk.

It will usually be sufficient to carry out the tests on a composite sample of sediment from the entire sediment area (giving an average toxicity and bioaccumulation). In larger areas, a local differentiation of sediment toxicity may provide a useful basis for demarcating subareas for remediation. The test is described in [Appendix VIII,4](#).

For sediment areas where only TBT or TFT and no other substances exceed the threshold values in Level 1, a Level 2 assessment should be performed. However, when interpreting the results,

emphasis must be placed on potential health risks, particularly in areas with environmental objectives related to human health. This is because the actual threshold value for ecological impact of TBT and TFT in water and sediment is so low that it will be known *a priori* that the risk of adverse ecological effects is unacceptable for all sediment areas that continue to Level 2. An ecological risk assessment based on concentrations of TBT or TFT alone will therefore not be a suitable tool in practice for distinguishing between areas that should proceed to the planning of remediation. However, this should not preclude areas with especially high levels of TBT or TFT from being considered for remediation.

The interpretation of results from Level 2 must allow sediment areas that pose an acceptable risk (remediation is unnecessary) to be distinguished from those for which action is required. Unacceptable risk does not necessarily mean that remediation must be taken, but rather that it must at least be considered. In this assessment, it will also be necessary to consider the risk posed by the sediments against the risks from other potential sources of contaminants.

It may be helpful to improve the local relevance of the data used in the calculations in Level 2 prior to remediation planning. The use of variables, constants and coefficients specific to an area in place of the proposed default values will result in a more realistic (and probably less conservative) risk assessment. Such improvements to the evidence base for the calculations constitute Level 3 of the risk assessment (see section 5).

The Excel calculator tool covers the implementation of Levels 1 and 2. The spreadsheet includes the formulae, constants and default values given in the Boxes in the guidelines, as well as threshold values and data for hazardous substances. The structure of the spreadsheet is shown in the text box on the following page.

Overview of spreadsheet structure (will be translated together with til spreadsheet)

The spreadsheet has been prepared to enable calculations to be performed in accordance with the risk guidelines. The spreadsheet also makes comparisons with threshold values stipulated in the guidelines and ensures that the basis for all calculations is transparent.

Below is a brief explanation of the function of the various worksheets. Green worksheets require the user to input data, yellow worksheets show the calculations that are performed, and blue worksheets state the results. Grey worksheets contain substance-related data.

1 a. Stedsspesifikke data	<p>Dette arket gir brukeren muligheten til å legge inn stedsspesifikke data for lokaliteten som skal risikovurderes. Det foreligger sjablongverdier for de fleste parametrene, men i kolonnen "anvendt verdi" kan man legge inn andre verdier. Dette må i tilfelle begrunnes i kommentarfeltet. Nbl Korreksjonsfaktoren (celle C15), kan ikke endres. Denne sørger for riktig benevnning for spredning som følge av bio-diffusion.</p>
1 b. Konsentrasjoner sediment	<p>I dette arket skal målte sedimentkonsentrasjoner legges inn. Legg også inn navn på prøvene. Basert på det som legges inn, beregnes antall prøver, snittkonsentrasjon og maksimumskonsentrasjon for hvert stoff. Disse verdiene benyttes videre av regnearket. I tillegg beregnes forholdet mellom høyeste verdi og medianverdi for hver av stoffene det er lagt inn konsentrasjoner for. Dersom forholdet er lavere enn 2, tyder det på at datasettet gir en god beskrivelse av området og at det ikke skiller seg ut en prøve som kunne indikere en "hotspot".</p>
1 c. Konsentrasjoner sjøvann	<p>I dette arket skal målte sjøvannskonsentrasjoner legges inn. Legg også inn navn på prøvene. Basert på det som legges inn, beregnes antall prøver, snittkonsentrasjon og maksimumskonsentrasjon for hvert stoff. Verdiene benyttes videre av regnearket. Dersom det ikke er målt sjøvannskonsentrasjoner, vil regnearket benytte beregnet sjøvannskonsentrasjon jmfør Faktaboks 9 i veilederen (dersom de nødvendige inngangsparametrene finnes) eller målte eller beregnede porevannskonsentrasjoner.</p>
1 d. Konsentrasjoner porevann	<p>I dette arket skal målte porevannskonsentrasjoner legges inn. Legg også inn navn på prøvene. Basert på det som legges inn, beregnes antall prøver, snittkonsentrasjon, maksimumskonsentrasjon og fordelingskoeffisient (K_d) for hvert stoff. Verdiene benyttes videre av regnearket. Dersom det ikke er målt porevannskonsentrasjoner, beregner regnearket verdier ut i fra sedimentkonsentrasjon og fordelingskoeffisienter.</p>
1 e. Vevskonsentrasjon i bunnfauna	<p>I dette arket skal målte vevskonsentrasjoner i bunnfauna (C_{bio}) legges inn. Legg også inn navn på prøvene. Basert på det som legges inn, beregnes antall prøver, snittkonsentrasjon og maksimumskonsentrasjon for hvert stoff. Verdiene benyttes videre av regnearket. Dersom C_{bio} ikke er målt, beregner regnearket konsentrasjonen ut i fra sedimentkonsentrasjon, biokonsentrasjonsfaktor vann/fisk og fordelingskoeffisienter.</p>
1 f. Vevskonsentrasjon i fisk	<p>I dette arket skal målte vevskonsentrasjoner i fisk (C_{fisk}) legges inn. Legg også inn navn på prøvene. Basert på det som legges inn, beregnes antall prøver, snittkonsentrasjon og maksimumskonsentrasjon for hvert stoff. Verdiene benyttes videre av regnearket. Dersom C_{fisk} ikke er målt, benytter regnearket målt eller beregnet vevskonsentrasjon i bunnfauna (C_{bio}), se forklaring til ark 1e.</p>
1g . Økotoksisitet	<p>I dette arket skal resultater fra økotokstestene legges inn. Navn på prøvene kan også legges inn. Basert på det som legges inn, genereres en resultattabell i ark 4 som sammenligner resultater med grenseverdier for trinn 1.</p>
2a. Mellomberegning	<p>Dette arket inneholder diverse mellomberegninger som må til for å beregne spredning og human helseisiko. Her kan du blant annet finne hva sjøvannskonsentrasjonen basert på Faktaboks 9 i veilederen blir, hvilken porevannskonsentrasjon som benyttes og hvordan innholdet av organisk karbon påvirker fordelingskoeffisientene. <i>Det skal ikke legges inn tall i dette arket.</i></p>
2b. Beregnet stedsspesifikk K_d	<p>Dette arket beregner stedsspesifikke fordelingskoeffisienter (K_d) dersom målt porevannskonsentrasjon er lagt inn i regnearket. Dersom det ikke er målt porevannskonsentrasjoner, benyttes K_d lik standard eventuelt justert for innholdet av TOC (må legges inn i ark 1a). <i>Det skal ikke legges inn tall i dette arket.</i></p>
2c. Beregnet tillatt spredning	<p>Dette arket beregner hva spredningen vil være dersom sedimentene tilfredsstiller grenseverdi for trinn 1. Antall skipsanløp (N_{skip}) fra ark 1a er det eneste som ikke er standardverdi i denne utregningen. Resultatet benyttes i resultatark 4 for sammenligning med beregnet spredning basert på sedimentkonsentrasjoner som er lagt inn i ark 1b. <i>Det skal ikke legges inn tall i dette arket.</i></p>
3a. Beregnet spredning	<p>Dette arket viser hva spredningen blir ut i fra de konsentrasjoner og stedsspesifikke data som brukeren har lagt inn. Det er i alt tre spredningsmekanismer som kan være aktuelle: spredning som følge av biodiffusjon (F_{dif}), spredning som følge av oppvirling fra skip (F_{skip}) og spredning som følge av transport via organismer (F_{org}). F_{tot} angir summen av de tre spredningsmekanismene. Regnearket beregner maksverdier og snitt-verdier for spredning, samt tiden det vil ta å tømme det bioaktive sedimentlaget for det gitt stoffet (t_{om}). Resultatene oppsummeres i ark 4. I tillegg viser arket prosentvis fordeling mellom de ulike spredningsmekanismene, slik at man raskt ser hvilken mekanisme som dominerer. <i>Det skal ikke legges inn tall i dette arket</i></p>

Box 4. Information required for implementation of Level 2

Information necessary to implement Level 2 is highlighted in bold (see also spreadsheet part 1a). The remaining points are not used in the calculations but may improve interpretation of the results and understanding of local conditions and will often be included in Level 3.

Physical factors:

- **water depth** (from map, used to assess potential for resuspension, volume calculations)
- **seabed area** (from map, used to calculate total flux, volume calculations)
- **seabed area at a depth of less than 20 m** (calculation of resuspension due to ship propellers)
- **grain size distribution** (measured and used to calculate resuspension)
- **residence time** of water in the basin (calculated, used to calculate concentrations of hazardous substances in the water and risk of ecological impact)
- **ship traffic data** (retrieved, length, depth and position of shipping routes, total sediment area <20 m affected by shipping traffic, docking frequency)
- water content of the sediment (measured, used for porewater volume, resuspension)
- shear strength (measured, interpretation of resuspension)
- current in vicinity of seabed (measured, further spread of hazardous substances from the sediment)
- ongoing construction work (data retrieved)
- area of any hard bottom within the sediment area (surveyed - visually if possible, assessment of actual spread to the water masses)

Chemical factors:

- **hazardous substances in the sediment** (measured in Level 1)
- **organic content of the sediment** (measured, adjustment of partition coefficients and interpretation of results)
- hazardous substances in porewater (calculated, or measured (Level 3), included in flux calculations of biodiffusion and transport via organisms)
- hazardous substances in seawater (calculated, or measured (Level 3)), assessment of whether calculated contribution from the sediments is realistic and of human exposure during bathing)
- oxygen levels in bottom water (measured, interpretation of mobility of metals and of ecological impact),
- sediment redox conditions (measured, interpretation of mobility of metals and of ecological impact)
- sediment respiration (from the literature, data required for calculation of transport in the food chain)
- sedimentation of organic matter (measured, revised calculation of transport in the food chain)

Biological factors:

- **toxicity tests** (measured in Level 1)
- **whole sediment toxicity test** (measured in Level 2)
- benthic infauna composition (measured, interpretation of bioturbation intensity and ecological impact)
- presence of fish and shellfish suitable for human consumption (data retrieved/measured, interpretation of risk to human health)
- hazardous substances in seafood (measured, assessment of the significance of the contribution from sediments)
- survey of particularly valuable or vulnerable stocks (data retrieved, interpretation of the significance of ecological impact)

Socioeconomic factors

- harvesting of fish and shellfish for consumption (data retrieved, risk to human health)
- current and desired usage of area (data retrieved, evaluation of objectives for any remedial actions)

4.2 Assessment of the risk of spreading hazardous substances (Level 2A)

In Level 2, simple calculations are performed of the rate of transport of hazardous substances (flux) from the sediment to other parts of the ecosystem. The transport routes are shown in simplified form in *Figure 5*. The risk assessment uses the calculated transport and the resulting concentrations in different media to assess the extent of the spreading and the consequences for the environment and for human health. The transport of hazardous substances from the sediment is presented both as flux per square metre and as annual transport from the entire sediment area. The processes by which hazardous substances are transported from the sediment to the water, and the methods for calculating this transport, are described in more detail in section 4.2.1 - 4.2.3.

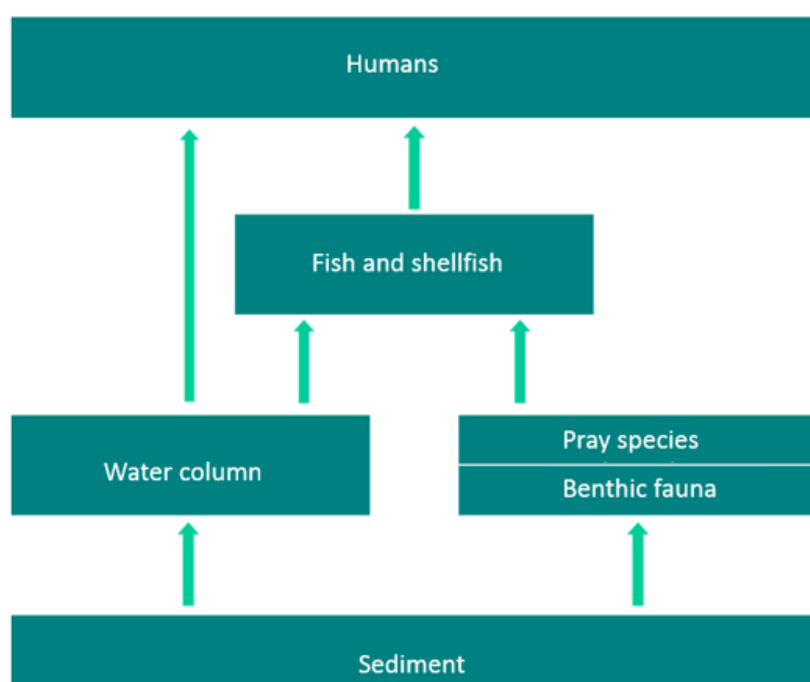


Figure 5. Simplified scheme showing routes for the spreading of hazardous substances from the sediment to other parts of the ecosystem.

4.2.1 Transport of dissolved substances from porewater

Diffusion

This is a physical process that evens out concentration differences without the influence of currents or turbulence. In the context of risk, it is the evening out of concentrations of hazardous substances in the porewater in the sediment and in the water above the seabed that is of importance. Diffusion leads to continuous weak transport of chemical substances from the porewater to the water at the seabed.

Advection

This is the transport of porewater to the overlying water driven by weak currents through the sediment. If the groundwater transport through the sediment layer is sparse, advection will be insignificant.

Biodiffusion

This is enhanced diffusion in the upper part of the sediment because of the benthic infauna either stirring up the sediment and bringing new porewater to the surface or pumping water from the sediment and expelling it as part of respiration and food uptake. This biological activity (bioturbation) leads to a mixture of advection and diffusion known as biodiffusion, which is estimated to be 10 times more intense than pure physical diffusion in natural sediment. Under anoxic conditions in the sediment, there will be little or no bioturbation and, in practice, no difference between physical diffusion and biodiffusion. The calculation is performed as shown in [Box 5](#).

Box 5 Calculation of transport via biodiffusion

Spreading as a result of biodiffusion (F_{diff}) is calculated as shown in the equation below. If no measurement data are available, the default values shown in parentheses should be used.

$$F_{diff} = \frac{n}{\tau} \cdot a \cdot D_s \cdot \frac{C_{pw}}{\Delta x} \cdot 3.15 \cdot 10^8$$

F_{diff} = biodiffusion (mg/m²/year)

n = porosity (0.7)

τ = tortuosity (3)

a = factor by which diffusion rate increases because of bioturbation (10)

D_s = molecular diffusion coefficient (cm²/s, substance-dependent, [Appendix I](#))

C_{pw} = porewater concentration (mg/l, $C_{pw} = C_{sed} [\text{mg/kg}]/K_d$ or measured, see [Box 10](#))

Δx = diffusion length (1 cm)

4.2.2 Transport of substances bound to sediment particles

Resuspension/erosion

This is the transport of hazardous substances bound to sediment particles that become resuspended in the water masses because of agitation of the bottom water. Particles in the clay fraction ($\leq 2 \mu\text{m}$) are considered to contribute most to the transport of particle-bound hazardous substances; the clay fraction is also the fraction of seabed sediment that remains resuspended the longest following agitation. In the context of risk management, resuspension during ship manoeuvring is the most important transport mechanism ([Box 6](#)). In the risk guidelines, propeller-generated erosion is considered to occur only at water depths of less than 20 m. Little is known about the effects of water jets relative to propellers. However, there are indications that during manoeuvring in harbour areas, water jets may cause greater erosion than propellers for a given boat size. Water jets may also cause erosion at greater depths than propellers, but probably over a smaller area. Until more evidence on the effects of water jets is available, it is recommended that in the context

of risk assessment, passenger boats and car ferries with water jets should be classified as large propeller-driven boats in a large harbour ([Box 6](#)), and erosion calculated accordingly.

Box 6 Calculation of sediment transport resulting from propeller-driven resuspension

Sediments at a depth of less than approximately 20 m can be spread as a result of propeller-driven resuspension. Spreading as a result of propeller-driven resuspension by ships (F_{ship}) is calculated based on the number of dockings per year and an estimate of the mass of sediment resuspended. The calculations are performed for each substance.

$$F_{ship} = \frac{2 \cdot N_{ship} \cdot m_{sed} \cdot C_{sed} \cdot (f_{diss} + f_{susp})}{A_{ship}}$$

F_{ship} = spreading as a result of shipping traffic (mg/m²/year)

2 = multiplication factor covering dockings plus departures

N_{ship} = number of dockings per year (port authorities)

C_{sed} = sediment concentration within A_{ship} (mg/kg d.w., measured)

f_{diss} = fraction dissolved, the proportion of the sediment content that can dissolve after resuspension ($10/K_d$, substance-dependent ([Appendix I](#)) or from a leaching test where $L/S=10$, [Appendix VIII](#))

f_{susp} = fraction suspended (sediment fraction < 2 µm, measured)

A_{ship} = total sediment area at a depth < 20 m (< 15 m where traffic consists only of smaller vessels) that is affected by shipping traffic (m², estimated from traffic patterns, independent of number of dockings).

m_{sed} = mass of resuspended fine fraction sediment in dry weight (kg per docking one-way; table below).

Default values for the mass (m_{sed} , kg) of fine fraction sediment that is resuspended per docking are given in the table below. Values are based on a ship covering a default distance of 120 m in water < 20 m deep. Before m_{sed} is inserted into the formula for F_{ship} , the default value must be multiplied by the actual distance travelled by the ship in water < 20 m deep (T in metres) and divided by 120 (performed by the spreadsheet). The sediment type must be selected according to the measured grain size distribution.

$$m_{sed} = \frac{m_{resuspended} \cdot T}{120}$$

T = distance travelled by ship (m, default value is 120 m)

Mass resuspended ($m_{resuspended}$)		Category of harbour	
Sediment type	Large harbour 1)	Industrial harbour 2)	Marina
Silt and clay	2000	1000	150
Sand	200	100	15
Gravel and rock	20	10	1

1) ferries, cruise ships, tugs, etc.

2) freighters, supply boats, etc.

For a more nuanced calculation of resuspension by different types of ships, see [Appendix IX](#).

4.2.3 Transport of hazardous substances through the food web

In addition to direct transport via physical processes, hazardous substances can be transported out of the sediment through uptake by benthic infauna that are then consumed by fish and other animals. This may result in some substances increasing in concentration upon ascent of the food web (biomagnification). The calculation is performed as shown in [Box 7](#). For further information see [Appendix X](#)

Box 7. Calculation of transport via organisms

Spreading because of uptake by organisms and predation (F_{org}) can be calculated from the tissue concentration of hazardous substances in potential prey (C_{bio}) and an estimate of how much of this benthic biomass is consumed by predators. The calculations assume that the benthic biomass remains approximately constant over time. If no measurements are available, the default values shown in parentheses should be used ([Appendix VII](#)).

$$F_{org} = \frac{C_{bio}}{OC_{cbio}} (OC_{sed} \cdot (1 - d) - OC_{resp}) \cdot \frac{1}{1000}$$

- F_{org} = spreading as a result of uptake by organisms (mg/m²/year)
- C_{bio} = tissue concentration in benthic fauna (mg/kg d.w., measured or calculated)
- OC_{cbio} = mass of organic carbon in benthic fauna biomass (0.25 g/g d.w.)
- OC_{sed} = supply of organic carbon to sediment from external sources (200 g/m²/year)
- d = fraction of organic carbon that is not lost due to respiration (0.47 g/g)
- OC_{resp} = organic carbon in the sediment lost due to respiration (31 g/m²/year)

If measurements of tissue concentrations in the benthic fauna are not available, values can be calculated as follows.

$$C_{bio} = \frac{C_{sed} \cdot BCF_{fish} \cdot 5}{K_d}$$

- C_{bio} = tissue concentration in benthic fauna (mg/kg d.w.)
- C_{sed} = concentration in sediment (mg/kg d.w., measured)
- BCF_{fish} = bioconcentration factor water/fish (l/kg w.w., substance-dependent, [Appendix I](#))
- K_d = partition coefficient sediment/water (l/kg, substance-dependent, [Appendix I](#), can be adjusted for organic carbon content, [Box 10](#))
- 5 = factor used to convert BCF_{fish} , which is on a wet-weight basis, to C_{bio} on a dry weight basis. The factor is based on the principle that the dry weight of biological material is typically 1/5 of the wet weight.

4.2.4 Calculation of spreading in Level 2

The equations in Boxes 5-7 provide a basis for calculating the total transport of hazardous substances out of the sediment (Box 8). From this, one can calculate the contribution made by the sediments to the concentrations of hazardous substances in the water masses above ([Box 9](#)) and in the biota/fish. These values in turn make it possible to evaluate the risk to the ecosystem and to human health posed by the sediment. The partition coefficients included in the calculations are listed in [Appendix I](#). It is also possible to use measured coefficients from the relevant sediment, as described in [Box 10](#).

It may be useful to check whether the results of the calculations are plausible. Methods for doing so are outlined in [Box 11](#).

Box 8. Calculation of total flux of hazardous substances and annual transport from sediment

The flux and annual transport of hazardous substances from sediment must be calculated separately for each subarea.

The total flux of a substance from sediment F (expressed as mg/m²/year) is estimated as:

$$F_{tot, ship} = F_{diff} + F_{ship} + F_{org}$$
$$F_{tot, sed-ship} = F_{diff} + F_{org}$$

$F_{tot, ship}$ = total flux of hazardous substances from the sediment area affected by ship activity (mg/m²/year)

$F_{tot, sed-ship}$ = total flux of hazardous substances from the sediment area unaffected by ship activity (mg/m²/year)

F_{diff} = flux due to biodiffusion (mg/m²/year)

F_{ship} = flux due to ship-generated resuspension (mg/m²/year)

F_{org} = flux due to uptake by organisms (mg/m²/year)

To assess the overall flux of hazardous substances, one must distinguish between areas that are affected by ship activity (A_{ship}) and those that are not ($A_{sed}-A_{ship}$, see [section 2.4](#)). The calculation is therefore performed separately for the two sediment areas shown in [Figure 4](#).

Annual transport of hazardous substances

In a remediation assessment, it may be necessary to compare the extent to which hazardous substances originate from the sediments with the contribution from other sources. This is most easily done by determining annual transport. The total transport of a substance out of the sediment U (expressed as mg/year) is estimated as:

$$U_{tot, ship} = F_{tot, ship} \cdot A_{ship}$$
$$U_{tot, sed-ship} = F_{tot, sed-ship} \cdot A_{sed-ship}$$

$U_{tot, ship}$ = total annual transport from the sediment affected by ship activity (mg/year)

$U_{tot, sed-ship}$ = total annual transport from the sediment unaffected by ship activity (mg/year)

A_{ship} = sediment area affected by ship activity (m²)

$A_{sed-ship}$ = sediment area unaffected by ship activity (m²)

U is calculated separately for the same subareas as in the flux calculations above. This provides a basis for determining the annual transport U from areas affected and unaffected by ship activity, as well as the total mass that spreads from the entire area:

$$U_{tot} = U_{tot, ship} + U_{tot, sed-ship}$$

U_{tot} = total annual transport from the entire sediment area (mg/year)

Box 9. Calculation of concentrations in the water

The average concentration of hazardous substances in the water within the sediment area as a consequence of spreading from the sediment can be calculated from the residence time of the water. This spread is equal to the total flux out of the sediment (F_{tot}) minus that taken up by organisms (F_{org}) and is equal to diffusion flux plus spread as a result of ship-generated resuspension ($F_{diff} + F_{ship}$; see [Box 8](#)). If no measurements are available, the default values given in parentheses should be used. The calculation is performed according to the formula:

$$C_{sw} = \frac{(F_{tot} - F_{org}) \cdot A_{sed}}{V_{sea}} \cdot t_r = \frac{F_{tot} - F_{org}}{d_{sea}} \cdot t_r$$

C_{sw}	= concentration in the water (mg/m ³ = µg/l)
A_{sed}	= total sediment area (m ² , calculated from map)
V_{sea}	= water volume above the sediment (m ³ , calculated from area and depth)
d_{sea}	= average water depth in the sediment area (m, measured)
t_r	= residence time of water in the sediment area (0.02 year = approx. 1 week)

The residence time t_r is highly variable and should be calculated based on oceanographic measurements.

The above calculations can then be used to calculate the flux of hazardous substances from the risk area to the water in surrounding areas (this calculation does not include transport through migration of organisms):

$$F_{out} = \frac{C_{sw} \cdot V_{sea}}{t_r}$$

F_{out} = total transport of hazardous substances out of the sediment area (mg/year)

Box 10. Changes to proposed partition coefficients

For organic hazardous substances, K_d values are based on the fraction of organic carbon (f_{oc}) and substance-specific partition coefficients (K_{oc}) normalised to organic carbon ([Appendix I](#)):

$$K_d = f_{oc} \cdot K_{oc}$$

The K_d values specified are based on the sediment having an organic carbon content of 1% ($f_{oc} = 0.01$). K_d should be adjusted if the measured organic carbon content of the sediment deviates substantially from this. In such cases, use of an average f_{oc} value for the samples in a subarea is recommended. The new K_d can then be calculated by multiplying the specified K_d by the measured percentage of organic carbon. For example, the K_d specified for naphthalene is 13 at 1% organic carbon. At 5% organic carbon, the K_d for naphthalene therefore becomes $13 \times 5 = 65$, i.e. the naphthalene is more strongly bound to the sediment.

The specified partition coefficients must cover all conditions and will in many cases overestimate the concentration of hazardous substances in the porewater, and therefore also the risk posed by the sediments. For some areas, it is reasonable to assume that the hazardous substances are legacy contaminants and hence strongly bound to particles, or that the particulate material itself has strong binding capacity (e.g. carbonaceous particulate matter or black carbon). Where this is suspected, the values provided for K_d should be replaced with site-specific measured values. This applies to some PAH-contaminated sediment areas where the contaminants are strongly bound to combustion-related carbon and thus have limited mobilisation potential. Another example is anoxic sediment where metals may be bound so strongly in the form of metal sulphides that they will not be bioavailable in practice, unless the sediments were to become resuspended in oxygen-rich water masses. Determination of site-specific K_d values is described in [Appendix VIII.6](#).

It may also be desirable to replace the values given for the partition coefficients between water and organisms (BCF) and between sediment and organisms (BSAF) with measured coefficients. Direct measurement of BSAF is carried out in Level 3 through the bioaccumulation test described in [Appendix VIII.5](#).

K_d and BCF for heavy metals vary considerably at the local level as a function of the sediment redox conditions. This variation is unpredictable and substituting the default values with measured BSAF values should be considered.

Tips for using the spreadsheet

In the spreadsheet, the K_d values for organic hazardous substances will automatically be adjusted if a value other than the default value for total organic carbon (TOC) (1%) is entered. If measured porewater concentrations are entered, the spreadsheet will calculate site-specific K_d values for the substances measured. These site-specific K_d values will then be used by the spreadsheet in place of K_d values adjusted only for TOC.

NB Remember that porewater concentrations must be entered in the same column in the spreadsheet as the corresponding sediment sample. If not, then K_d will be calculated incorrectly.

Box 11. Verification of the calculations in Level 2

It is recommended that simple checks be performed to verify that the results of the calculations in Level 2 are plausible.

The calculated flux from the sediment can be used to estimate how quickly the ‘store’ of hazardous substances in the sediment will be depleted. In a contaminated area, the annual net loss of hazardous substances from the sediments should be only a small fraction of the total amount present. Where this not the case, the sediments would already have been depleted of hazardous substances. As a default, one can assume that depletion occurs from the store of hazardous substances in the upper 10 cm of the seabed (presumed to be the maximum depth for significant bioturbation). If the fluxes are relatively large such that the store is being ‘rapidly depleted’, the fluxes have either been overestimated, or the sediment is being supplied with considerable quantities of hazardous substances. The estimated time taken to fully deplete the store of a hazardous substance will therefore provide an indication of whether the fluxes have been overestimated. The depletion time is calculated as follows:

$$t_{empty} = \frac{d_{sed} \cdot C_{sed} \cdot \rho_{ww} \cdot f_{dw}}{F_{tot}}$$

- t_{dep} = time taken to deplete the store in the bioactive layer (years)
- d_{sed} = bioturbation intensity (100 mm/m²)
- C_{sed} = concentration in sediment (mg/kg d.w., measured)
- ρ_{ww} = wet sediment density (1.3 kg/litre)
- f_{dw} = dry weight fraction of wet sediment (0.35)
- F_{tot} = total transport of hazardous substances from the sediment (mg/m²/year)

The calculated values for the concentration of hazardous substances in the water masses may also be verified through direct measurements, either of water samples or by using passive samplers. These measurements will show the total concentration of hazardous substances in the water masses. The calculated contribution from the sediments should not exceed this value.

Similarly, the estimated concentrations of hazardous substances in organisms can be verified by analysing levels of these substances in fish and shellfish. As in the case of water, the calculated contribution from the sediments should not notably exceed the measured total concentrations.

4.2.5 Assessment of spreading

Threshold values have yet to be established for the acceptable/unacceptable spread of hazardous substances from sediments. This means that it is not possible to assess the risk that spreading will exceed critical values, as can be done for effects on the ecosystem or on human health. What constitutes acceptable/unacceptable spread of hazardous substances from the sediments will depend on the environmental objectives and on any local acceptance criteria that have been set. If acceptance criteria for spreading have not been defined, the consequences in terms of adverse

effects on human health or the ecosystem should be assessed. In such cases, the risk of spreading will be acceptable if the risk of adverse effects on human health and on the ecosystem are both acceptable.

Some examples of how to set acceptance criteria merely for spreading:

- The flux of hazardous substances from the sediment must not exceed a specified value (e.g. 'not more than n kg/year', 'not more than n mg/m²/year').
- Spreading must not exceed that from an agreed reference sediment by more than x per cent; for example, from a sediment that lies just below the threshold values in Level 1. In this case, the threshold value will vary with the fines content of the sediment. This approach is used in the spreadsheet. It is also possible to compare spreading with other threshold values. When this is done in the spreadsheet, a rationale for these threshold values must be provided.
- The flux of hazardous substances from the sediment to adjacent areas must not exceed specified values.
- The spread must not cause the hazardous substances content of the sediments in neighbouring areas to exceed the boundary between NEA environmental quality Classes II and III. Neighbouring areas can be defined based on topography or as a certain number of km², for example.

Several comparisons can be made regarding the spread of hazardous substances because of ship traffic, including:

- The spread of hazardous substances merely due to ship traffic relative to defined acceptance thresholds, or other known sources of pollution.
- The significance of the spread of hazardous substances due to shipping traffic relative to other transport routes (biodiffusion and transport through the food web) within the area affected by shipping (A_{ship}).
- The significance of total spreading from the shipping-affected area relative to total spread from the remainder of the risk area ($A_{\text{ship}}-A_{\text{sed}}$).

These assessments will allow the subareas included in the risk assessment to be ranked according to their importance as sources of the spread of hazardous substances. Nevertheless, it is first and foremost the impact of the spread of hazardous substances that is of concern, not the spread of the substances per se.

4.3 Assessing the risk to human health (Level 2B)

A key factor in assessing the risk to human health is the extent to which hazardous substances in the sediment are bioavailable to benthic fauna, which is the first step in the transport of these substances to humans via the food web. Bioavailability calculations based on measured sediment concentrations and the recommended partition coefficients will yield conservative estimates, i.e. bioavailability values that are probably higher than the real values. Since the actual partition coefficients will vary markedly with sediment conditions, there is also the option of measuring partition coefficients directly (see [Appendix VIII.6](#)). In cases where evaluating the risk to human health of seafood consumption is an important part of the risk assessment, it is recommended that a bioaccumulation test be conducted to directly measure the uptake of hazardous substances by organisms in sustained contact with the relevant sediment. These measurements are described in [Appendix VIII.5](#) and will form part of Level 3. Direct analysis of hazardous substances in local seafood can also be used to validate calculations regarding the risk to human health. These analyses

will also capture exposure to other sources of hazardous substances besides the sediments. Consequently, the analyses will not reveal the contribution to risk posed solely by the sediments, but rather an upper limit for this contribution.

Consumption of fish and shellfish will be the dominant risk factor for human health in most of cases. However, a comprehensive risk assessment must also include other relevant exposure routes linked to current and potential future use of the area. Relevant exposure routes for different types of usage are listed in *Table 4*. If an area is used for bathing or recreation, for example, exposure through oral ingestion and skin contact with contaminated sediment and water poses a potential risk to human health. Evaluating this risk requires knowledge of the concentrations of hazardous substances in the sediment in the recreation area itself.

It should be emphasised once again that the threshold values for risk assessment Level 1 indicate the risk of the substances having an ecological impact, not the risk of an impact on human health. The threshold values in Level 1 may thus be so high for certain substances (e.g. benzo(a)pyrene) that sediments that do not exceed these limits could nevertheless pose an unacceptable risk to human health in Level 2.

Table 4 Exposure routes for assessing the risk to human health posed by different types of marine area usage.

Area usage	Ingestion				Skin contact	
	Sediment	Surface water	Particulate matter	Fish and shellfish	Sediment	Surface water
Conservation area			•	•		•
Bathing	•	•	•	•	•	•
Recreation		•	•	•	•	•
Fish farming		•	•	•		•
Marina		•	•	•	•	•
Port				•		
Industry				•		

Methods for calculating exposure as a result of seafood consumption are presented in [Box 12](#) and through contact with sediment, particles and water in [Appendix IV](#). Calculated life-time exposure is compared to the threshold values specified in the form of MTR/TDI (maximum tolerable risk/lifetime tolerable daily or weekly intake; see [Appendix III](#)). Since humans are exposed to hazardous substances through many routes, it is assumed that not more than 10% of a person's total exposure is due to sediments. Exceptions are TBT and TFT where 100% of exposure is assumed to be sediment-related.

Consumption of fish and shellfish and recreational contact with sediment will vary throughout the country. It may therefore be useful in some cases to include a more detailed estimate of how much of total food consumption consists of local seafood. In the calculations, it is assumed that 50% of the seafood consumed contains the calculated concentrations of hazardous substances (C_{fish}). The information required for these calculations should be available from the local or regional food regulatory authorities ([Box 4](#)).

In Norway, it is the Food Safety Authority that assesses the risk to health based on tolerable daily intake/tolerable weekly intake (TDI/TWI) limits set by international expert groups within WHO/FAO and the EU. For substances for which a threshold value for ingestion cannot be determined

(genotoxic substances; in this context mainly PAH and in particular benzo(a)pyrene), the Norwegian Food Safety Authority conducts its own assessments to decide upon an adequate level of protection. Upon detection of hazardous substances in seafood, a separate expert group, the Norwegian Scientific Committee for Food and Environment, will perform the necessary health-related assessments. Considering this health-related assessment and other relevant information, the Food Safety Authority will consider the need to issue dietary advice for an area.

The calculations in [Boxes 7](#) and [12](#) provide a means of assessing the specific risk of sediments giving rise to unacceptable tissue levels of hazardous substances in fish and shellfish. When planning remedial action, this contribution must be weighed against that from other sources.

Box 12. Human exposure via consumption of fish and shellfish

Indirect human exposure to sediment through the consumption of fish and shellfish can be calculated from anticipated consumption of fish and shellfish and their content of hazardous substances via the formula:

$$IEC_f = \frac{DI_f \cdot CF_f \cdot af \cdot C_{fish}}{BW}$$

- IEC_f = indirect exposure via consumption of fish and shellfish (mg/kg/d)
 DI_f = daily consumption of fish/shellfish (child: 0.028; adult: 0.138 kg w.w./d)
 CF_f = contaminated fraction (0.5)
 af = absorption factor (1)
 C_{fish} = concentration in fish/shellfish (mg/kg w.w., measured or calculated, $C_{bio}/5$ used if appropriate, see [Box 7](#))
 BW = bodyweight (child: 15 kg; adult 70 kg)

By assuming that a human life can be divided into 6 years of childhood and 64 years of adulthood, the total lifetime dose via consumption of fish and shellfish can be calculated as follows:

$$DOSE = \frac{6 \cdot IEC_{fc} + 64 \cdot IEC_{fa}}{70}$$

- IEC_{fc} = total daily indirect exposure of child (mg/kg/d)
 IEC_{fa} = total daily indirect exposure of adult (mg/kg/d)
 $DOSE$ = average lifetime daily exposure (mg/kg/d)

$DOSE$ is compared with 10% of the MTR/TDI values ([Appendix III](#)).

4.4 Ecological risk assessment (Level 2C)

Hazardous substances can affect the ecosystem in multiple ways, but knowledge of these effects is greatly lacking. The primary objective of the threshold values in [Appendix II](#) is to protect at least 95% of the species in an ecosystem even in the event of prolonged exposure. The risk of harm to the ecosystem is considered acceptable if at least 95% of species remain unaffected (M-241/2014). Since the 95% target can only be verified when the effects of a substance on many species are known, it is important to be able to measure directly whether hazardous substances in sediment do have adverse effects. A whole sediment toxicity test is therefore performed in Level 2 (see [Appendix VIII.4.5](#)). This provides a means of directly assessing the risk to organisms that are in prolonged contact with the sediment. The risk of harm to non-sediment dwelling organisms is assessed based on an estimate of the contribution from the sediments to the level of hazardous substances in the water ([Box 9](#)). The risk to the ecosystem posed by a contaminated sediment is therefore based on results from Level 1 and Level 2 combined.

The procedure for evaluating ecological risk is as follows:

- Assess the risk that direct contact with the sediment poses to the biota on the basis of measured sediment concentrations and measured or calculated porewater concentrations, relative to the boundaries between NEA environmental quality Classes II and III for seawater and marine sediments, respectively, in [Appendix II](#), and on the basis of the results of whole sediment toxicity testing as well as the toxicity tests in Level 1.
- Assess the risk posed to organisms in the waters above the sediment on the basis of estimated concentrations of hazardous substances in water originating from the sediments, relative to the thresholds for NEA environmental quality Classes II and III for seawater in [Appendix II](#) and toxicity testing of porewater using *Skeletonema*, *Tisbe* and *Crassostrea* larvae in Level 1.

The threshold values in Level 1 are derived for each substance/substance group separately without assuming any interaction between substances. Toxicity tests, by contrast, reveal the combinational effects on benthic infauna, since the test results reflect the overall effect of all hazardous substances present. It is therefore important to weigh up the risk calculated from concentrations against that calculated from toxicity. If porewater and whole sediment testing reveals toxic effects more than the threshold values, this indicates that there is a risk of harm to the ecosystem even if none of the threshold values for sediment concentration are exceeded. Such effects may be due to the combined impact of the substances analysed plus others that were not. Further studies should be performed in such cases to exclude the possibility that the testing method itself is affecting the results.

If toxicity testing reveals no toxic effects, the real ecological risk is less than that indicated by exceedence of the threshold values for concentrations in Level 1.

5. Risk assessment Level 3

5.1 Level 3 objectives

In some cases, a more comprehensive risk assessment with a stronger local basis than that of Level 2 will be required prior to the planning of remedial action. This is referred to in the guidelines as Level 3 of the risk assessment. The motivation for proceeding to Level 3 may be that there is reason to believe that the Level 2 assessment predicts an unrealistically high risk, or in some other way fails to reflect the actual risk. For example, the hazardous substances in the sediments may be less bioavailable than the proposed partition coefficients suggest, or the flux to other parts of the ecosystem may be lower than that calculated from the default values. Sometimes analyses of hazardous substances in water and organisms will indicate that the contribution from the sediments is much smaller than shown by the Level 2 calculations.

Level 3 could include:

- Selected elements from Level 2, with the aim of verifying and improving the precision of calculations performed in Level 2 with the aid of new local data. An important principle here is that Level 3 calculations should not be more conservative than those in Level 2. Supplementary information to improve the interpretation and evaluation of Level 2 results.

There is great freedom to tailor Level 3 to requirements. In some cases, the purpose of Level 3 will be to verify, and potentially replace, the default values suggested in the guidelines for Level 2 with more reliable, site-specific values generated through new tests. In other cases, one will wish to perform full numerical modelling of the mobilisation, transport and bioaccumulation of hazardous substances to provide the best possible basis for evaluating the impact of the sediments on levels of hazardous substances in the water body, neighbouring areas and local seafood.

There is also the option to proceed directly to Level 3 after Level 1. Although the guidelines do not specify how Level 3 should be performed, there is nevertheless still a requirement for Level 3 to cover the three main assessments of Level 2: risk of the spread of hazardous substances, and the risks posed to human health and to the ecosystem.

The costs of completing Level 3 should be weighed against the potential costs of remediation. In some cases, it may be less expensive to proceed directly to remedial action than to first conduct extensive supplementary tests to improve the risk assessment in Level 2.

The following sections provide advice and guides on how to improve the local foundation for factors for which default values are proposed by the guidelines. There are also suggestions for tests that can be performed to support the interpretation of results related to risk. Requirements and suggestions for testing methodology are presented in [Appendix VIII](#).

5.2 Replacing default Level 2 values with local values

The default values in Level 2 will reflect the local conditions to varying degrees. Some default values are so universal that there is little to be gained from site-specific measurements. Other factors are in practice so dependent on local conditions that the default values may be unrealistically conservative in some situations. [Appendix VII](#) lists the factors for which default values are provided in Level 2, along with each default value, the uncertainty related to its use in a risk area, and suggested approaches for obtaining values that better reflect the local conditions. A simplified qualitative scale is used to describe the degree of uncertainty (high, medium, low).

A numerical analysis examining the sensitivity of the results of the risk assessment to variation in each of the default values has been performed (Saloranta et al. 2011³). This revealed that the partition coefficients K_d and BCF are among the variables that have greatest influence on the results, and thus those for which it is particularly important to obtain realistic values.

5.3 Supplementary tests to aid interpretation

Level 3 may also include tests that are not intended to improve the accuracy of the calculations in Level 2 per se, but rather to aid interpretation of the Level 2 results in order that the risk posed by the sediments may be determined as accurately as possible. Various types of tests shown to be useful for this purpose are described below.

5.3.1 Spreading from the sediments

Key factors to consider when assessing the risk of spreading are local conditions in the seabed, sediment characteristics, ship traffic, and exchange of water and currents.

Surveying seabed conditions

A more in depth characterisation of the seabed in the sediment area may be useful for several reasons. Much of the information that one would like to obtain to be able to interpret risk will also be useful for any future remediation planning. For example, knowledge of the vertical profiles of hazardous substances and other variables will be useful both for interpreting the calculated values for leaching of hazardous substances and for judging the benefits of active remediation versus natural improvement.

The most important components of a physical sediment characterisation are typically an overview of the key sediment types (gravel and rock, sand, or silt and clay) and the stratification down through the sediments of various properties (sediment type, bioactive layer, appearance, colour and odour, water content, redox conditions, content of hazardous and non-hazardous substances, etc). Stratification and profiles are usually characterised through analysis of sectioned core samples, but direct images of the vertical appearance and thickness of different layers can be obtained using Sediment Profile Imaging (SPI). Such imaging can provide the same visual information for approximately the upper 30 cm of the sediments and can generate many vertical images in a short space of time. A large-scale overview of depth, seabed topography, and any subareas with a hard

³ Saloranta TM, Ruus A, Borgå K. 2011. Identification of the most influential factors in the Norwegian guidelines for risk assessment of dispersion of contaminants from sediments. *Integrated Environmental Assessment and Management* 7:657-667.

bed can be obtained by means of echo sounding or other types of acoustic recording (side scan sonar, multibeam sonar) and video mapping. Photographic techniques (stills, video) can also be useful for obtaining an overview of the appearance of the sediment surface and of the larger organisms present there. The latter may for example indicate whether propeller erosion is an important local factor.

Redox conditions

The sediment redox conditions determine the amount of leaching from the sediments, especially of metals. Metals bound as sulphide under anoxic conditions are less soluble than their partition coefficients suggest. In anoxic sediments, the benthic fauna in general are reduced (depending on the thickness of any oxic surface layer) so that bioturbation and hence biodiffusion are lower than the values shown in Level 2. Both factors will lead to the flux of hazardous substances from anoxic sediments being overestimated in Level 2. Knowledge of whether the sediments are anoxic is therefore important for interpreting results regarding spread. A method for direct measurement of redox profiles is described in [Appendix VIII.3.6](#). With experience, a sufficient impression of the redox conditions can in most cases be obtained through observations made during the sampling itself (hydrogen sulphide odour, black layers in the sediment, little fauna).

Measurement of flux of dissolved hazardous substances

Mesocosm experiments (which simulate natural environmental conditions) can be used to obtain a realistic estimate of the natural flux of dissolved hazardous substances into the water masses from the sediments, driven by biodiffusion and advection. In these experiments, undisturbed sediment samples collected using box corers are installed in a laboratory, with a weak flow of water over their surface. The water circulates in a semi-closed system and passes through chambers with passive samplers that accumulate hazardous substances over time. A variety of passive samplers are now available for metals and non-polar/polar organic hazardous substances. Exposure time is in the order of a few weeks. Advection-generated flux can be estimated by manipulating the flow rate or turbulence over the test sediment.

Biodiffusion can also be measured *in situ* using diffusion chambers on the seabed. These are closed units equipped with passive samplers like those in mesocosm experiments. The chambers can operate in either still or turbulent water. There are various types of chambers available, which can be placed in position either by divers or from boats.

Measurement of flux of particle-bound hazardous substances

The transport of hazardous substances on sediment particles is a two-way process: resuspension and subsequent resedimentation. These processes are different for different grain sizes and sediment types. Most important in the context of risk is the fraction that remains in the water masses long enough to have an adverse effect on the site itself or to any neighbouring areas.

Mesocosm experiments can be used to measure particle flux from a natural sediment through erosion as a function of bottom flow and turbulence. Both gross erosion flux (what is resuspended) and net erosion flux (what remains in the water masses after resedimentation) can be simulated. These experiments offer the advantage of allowing flow and turbulence to be manipulated experimentally, but in most cases, they will only reveal small-scale effects.

The significance of ship-generated erosion (from propellers and water jets) is discussed in [Appendix IX](#). To calculate such large-scale erosion, direct measurements in a risk area affected by typical ship traffic are recommended in place of mesocosm experiments. The concentration of suspended fine particulate matter that remains in the water masses in conjunction with a docking and the volume

of water in which the matter becomes dispersed can be measured directly with the aid of horizontal and vertical turbidity profiles and particle analyses in the field during/after dockings. The measurements can be made from a small boat and require access to electronic turbidity logs and water sampling equipment. The schedule for such measurements must be tailored to the specific situation.

To interpret the significance of ship-generated flux, it is important to estimate the actual seabed area affected by ship traffic and the maximum depth at which erosion occurs. Video recording of seabed conditions, possibly supplemented with SPI recording and sampling for grain size analysis, should in many cases indicate how far out from a ship's route the effects of shipping erosion extend, and perhaps also how deep. However, experience in estimating this area of influence is lacking.

The easiest way to measure the resedimentation of resuspended sediment (intensity and location) is to use sediment traps. Sediment traps come in a variety of sizes and designs to ensure that they do not collect too much or too little sediment in relation to what sinks through the water. There are also traps suitable for estimating the horizontal transport of sediment particles along the seabed. The traps remain in position for an extended period, usually 1-2 months, to gather enough material for analysis, and therefore provide a time-integrated picture of the amount and chemical composition of the material that undergoes sedimentation. There are also instruments that can be placed on the seabed to measure the sedimentation of particles optically in approximately real time.

Measurement of flux of hazardous substances out of the sediment area

Level 2 describes how to calculate this flux from the estimated concentrations of hazardous substances in the water masses above the sediment, the total water volume above the sediment, and the exchange rate between this water volume and the surrounding waters. The most challenging measurements are the exchange rate and the residence time, which are two sides of the same coin. A default value is provided for the residence time of overlying water, but since residence time is strongly dependent on local conditions, this value is highly uncertain. It is therefore recommended that local residence time be calculated in Level 2 but, if not, it should at least be calculated in Level 3. Estimates of the real concentrations of hazardous substances can be verified through water analyses or the use of passive samplers on rigs in the water. These will not be able to distinguish between hazardous substances contributed by the sediments and those from other sources, but they will provide a basis for judging whether the estimates seem plausible.

5.3.2 Risk to human health

The Level 2 assessment encompasses calculations of the uptake of hazardous substances by humans via several routes. Many of the default values with the greatest uncertainty concern the usage of an area, as well as diet and recreational patterns that have a direct bearing on the significance of the various exposure routes. Improving the reliability of the risk assessment therefore requires a local survey of these factors. How is the area currently used in terms of fishing and recreation and what is the long-term environmental objective for the area?

Risk from consumption of seafood

Assessment of the risk to health posed by hazardous substances is generally conducted by international bodies such as JECFA (WHO) and EFSA (EU), as well as by the Norwegian Scientific Committee for Food and Environment (VKM). This assessment is based on the content of hazardous substances in relevant fish and shellfish. The Norwegian Food Safety Authority provides dietary advice based on these risk assessments.

Although the overall assessment of an area by the Norwegian Food Safety Authority in principle constitutes a Level 3 assessment of the risk to human health, it does not differentiate between the various sources of hazardous substances in seafood. The challenge for the risk assessment system is to calculate how much of the hazardous substances in seafood may originate from sediments.

The calculations in Level 2 cover transport directly to benthic fauna and then onwards to the first level of predators, as well as transport directly from the water to organisms in the water. At a minimum, Level 3 should involve obtaining local data to improve these calculations. Level 2 presupposes, however, that the tissue concentration in seafood is equal to that in the first level of the food web, i.e. that there is no increase (biomagnification) or reduction in the concentration of hazardous substances upon ascending the foodchain to seafood. Level 3 should therefore include the additional step of estimating the transport of sediment-related hazardous substances from the first predator level to edible species of fish and shellfish. This requires knowledge of which local species are relevant as seafood and how they are connected to sediments via the local ecological food web. Such food web analyses are both uncertain and complicated, in part because we lack sufficient knowledge of the various ecological relationships.

An alternative means of assessing the contribution made by sediments to hazardous substances in seafood is through numerical modelling. Modelling tools, including those tailored to conditions in Norwegian fjords, can be used to estimate the actual flux from the sediments to seafood, and consequently how much the sediments probably contribute to the measured concentrations in seafood. Good modelling approaches can provide a more reliable picture than the Level 2 calculations of the risk that hazardous substances from sediments will lead to exceedance of tolerable intake limits. They can also indicate to what extent the sediments contribute to the need for/continuation of dietary advice. Tailoring such models to a sediment area will usually require a measurement programme to determine key input parameters, possibly also to verify selected results from the model.

Risk to human health from contact with sediment and water

The risk of adverse effects on health from contact with sediments, particles and water is closely related to the usage pattern of the sediment area. If the area is used for bathing, information on local bathing habits will be important in assessing the risk of contact with sediment and particles. Useful local information will include the location of bathing sites (proximity to sediment areas), whether the area is suitable for all age groups (sandy beaches or rock formations), as well as statistics on the frequency and duration of bathing (total exposure time). Furthermore, direct analyses of the hazardous substances content of the sediments on bathing beaches and in suspended material in the upper layers of water during the bathing season may be useful. All of this provides a basis for replacing the default values in [Appendix IV](#) with site-specific values.

5.3.3 Risk of ecological effects

Assessment of ecological risk in Level 2 does not involve any specific factors with default values, beyond the K_d values used in deriving the sediment threshold values. Making improvements to these values in Level 3 has thus been covered previously in these guidelines. The risk of adverse ecological effects in Level 2 is linked to exceedance of the threshold values for effects in the water (boundary between NEA environmental quality Classes II and III for seawater) or in sediments (boundary between NEA environmental quality Classes II and III for marine sediments, as well as the threshold values for toxicity) shown in [Appendix II](#). An aim of the Level 3 risk assessment should therefore be to clarify whether the observed exceedance does in fact harm the ecosystem in a given situation. In

other words, are there signs of adverse effects on the ecosystem or environmental stress on organisms or populations?

For a Level 3 assessment, there are various testing methods available that have proved in practice to be sensitive enough to detect at least significant adverse ecological impact. Some of these effect parameters are natural compensatory responses to environmental stress and will not necessarily lead to an impact on populations or ecosystems. However, they signal that a stressor is present. The tests may include analysis of benthic and pelagic ecology in both the sediment area and neighbouring areas. Measurements of the health status of individual organisms through so-called biomarker analyses may be appropriate for detecting sublethal environmental stress. Biomarker analyses can reveal abnormal anatomical, genetic, biochemical and physiological traits that can often be linked to exposure to substances or groups of substances. Similar methods at the biotope or ecosystem level are the analysis of biodiversity, the prevalence of specific indicator species and changes in community structure, the latter with the aid of multivariate data analysis. Through comparisons with locally known or expected natural ecological conditions, one can judge whether ecological conditions or the health status of organisms have already been affected.

However, determining the risk that this is due to sediments is very difficult, and requires considerable discretionary judgement. Since our understanding of the relationships between environmental stressors and biological effects in natural complex ecosystems is generally limited, there is little possibility of establishing direct links between hazardous substances in the sediment and local ecological status in a reliable manner, and thus being able to verify that the ecological risk estimated in Level 2 or Level 3 is real. This problem arises first and foremost in cases where the sediments are found to pose an unacceptable risk and ecological effects are detected, but where other sources of stress could also play a part. A reliable assessment of the impact of the sediments almost requires the sediments to be the only notable source of stress detected.

If the ecological risk from the sediments is found to be unacceptable, but there is no evidence that local communities or populations are affected, it should be assumed that the risk from the sediments has been overestimated (especially since there will often be other stressors present besides hazardous substances from the sediments).

If the risk from the sediments is found to be acceptable and yet adverse effects on the local ecosystem are detected nonetheless, it should be assumed (given that the risk assessment is intended to be conservative) that other sources of contaminants besides the sediments are responsible for the ecological effects.

6. Relationship between Levels 2 and 3

Level 3 is less conservative, more strongly based on local conditions and more reliable than Level 2. Unacceptable risk in Level 2 may therefore turn to acceptable risk in Level 3. The objective of both assessment levels is the same: they must provide grounds for deciding whether remediation of sediments in an area is necessary. They may also help to prioritize measures across sediment sites. The results from the two levels will thus be used in the same way, but Level 3 will provide more reliable grounds for decision-making. Where conclusions in Level 3 deviate from those in Level 2, Level 3 should therefore be decisive. If the risk calculated in Level 3 is still unacceptable, suitable

remedial action must be planned and if necessary implemented to reduce the risk to an acceptable level.

7. Reporting the risk assessment

The results of the risk assessment must be documented in writing. The structure of a main technical report is outlined in [Appendix VI](#). The report must contain complete documentation of the risk assessment carried out in Levels 1 and 2. The target group for the main report will be problem owners/clients, environmental authorities, and possibly the research and consultancy communities.

Although the main technical report will include a summary, it may also be desirable to produce a separate summary report. This should preferably be approximately 10 pages and should focus on:

- Objectives and assumptions
- Implementation
- Key results, preferably in the form of figures and tables
- Conclusions and recommendations.

The target group for the summary report will be administrative personnel and the public.

Appendix VI does not include reporting from Level 3, as this will depend on the content and scope of the risk assessment. The main report for Level 3 should contain at least as much detail as the reports for Levels 1 and 2.

Appendices

Appendix I – Index of hazardous substances physical/chemical data

A risk assessment system based on the equilibrium distribution between sediment and water is directly dependent on the partition coefficients used. Table 1 shows the partition coefficients for sediment/water K_d and water/fish BCF that are used in these guidelines. The diffusion coefficient ($D_{\text{molecular}}$), which is used to calculate biodiffusion, is also shown.

Table 1 Physical/chemical data on selected hazardous substances

Name	Molar wt g/mol	$D_{\text{molecular}}$ cm ² /s	log K_{ow} L/L	log K_{oc} L/kg _{oc}	K_d sed at TOC 1% L/kg d.w.	BCF _{biota} L/kg w.w.
Metals						
Arsenic	74.9	9.1E-06		-	6607	4
Lead	207.2	9.5E-06		-	154882	424
Cadmium	112.4	7.2E-06		-	130000	623
Copper	63.5	7.1E-06		-	24409	100
Chrome total (III + VI)	52.0	6.0E-06		-	120000	20
Mercury	200.6	8.8E-06		-	100000	100
Nickel	58.7	6.6E-06		-	7079	270
Zinc	65.4	7.0E-06		-	110000	1000
PAH						
Naphthalene	128.2	8.6E-06	3.3	3.1	13	515
Acenaphthalene	152.2	7.6E-06	4.0	3.4	26	509
Acenaphthene	154.2	7.6E-06	3.9	3.7	51	1000
Fluorene	166.2	7.2E-06	4.2	4.0	102	1658
Phenanthrene	178.2	6.8E-06	4.5	4.6	372	14893
Anthracene	178.2	6.8E-06	4.7	4.5	295	1900
Fluoranthene	202.3	6.2E-06	5.2	5.0	977	4800
Pyrene	202.3	6.2E-06	5.0	4.8	589	88157
Benzo(a)anthracene	228.3	5.7E-06	5.9	5.7	5012	33457
Chrysene	228.3	5.7E-06	5.8	5.6	3981	6088
Benzo(b)fluoranthene	252.3	5.3E-06	5.8	5.9	8319	11138
Benzo(k)fluoranthene	252.3	5.3E-06	6.1	5.9	7943	11138
Benzo(a)pyrene	252.3	5.3E-06	6.1	5.9	8318	11138
Indeno(1,2,3-cd) pyrene	276.3	5.0E-06	6.7	6.4	23442	11138
Dibenzo(a, h)anthracene	278.4	5.0E-06	6.6	6.3	19498	50119
Benzo(ghi)perylene	276.3	5.0E-06	6.6	6.0	10233	11138

Table I - Continued from previous page.

Name	Molar weight g/mol	D _{molecular} cm ² /s	log K _{ow} L/L	log K _{oc} L/kg _{oc}	K _d sed at TOC 1% L/kg d.w.	BCF _{biota} L/kg w.w.
Other organic substances						
DDT	354.5	4.2E-06	6.9	6.8	62159	50000
Tributyltin (TBT ion)	290.1	4.8E-06	3.8	3.0	11	6000
Lindane	290.8	4.8E-06	3.5	3.6	37	1300
Hexachlorobenzene	284.8	4.9E-06	5.7	5.1	1300	42000
Pentachlorobenzene	250.3	5.4E-06	5.2	4.6	400	5300
Trichlorobenzene	181.4	6.7E-06	4.1	3.1	14	1140
Hexachlorobutadiene	260.8	5.2E-06	4.8	4.0	112	17000
Pentachlorophenol	266.3	5.1E-06	3.0	3.5	34	770
Octylphenol	206.3	6.1E-06	4.1	3.4	27	634
Nonylphenol	220.4	5.9E-06	4.5	3.7	54	1280
Bisphenol A	228.0	5.7E-06	3.4	2.9	7.2	67
Tetrabromobisphenol A	543.9	3.1E-06	5.9	4.7	497	1234
Pentabromodiphenyl ether	564.7	3.0E-06	6.5	5.8	5659	35000
Hexabromocyclododecane	641.7	2.7E-06	5.6	4.7	457	18100
Perfluorinated octyl sulphonate (PFOS)	500.1	3.3E-06	3.4	3.0	10	2790
Diuron	233.1	5.6E-06	2.8	2.6	3.55	2
Irgarol	253.1	5.3E-06	4.0	3.1	14	250
PCB7	375.7	4.0E-06	5.7	5.5	3211	24950
Triphenyltin	350.0	4.2E-06	3.4	3.3	19	1100
Dodecylphenol with isomers	262.4	5.2E-06	7.1	5.0	1100	823
Di(2-ethylhexyl) phthalate (DEHP)	390.6	3.9E-06	7.5	5.2	1650	840
Perfluorooctanoic acid (PFOA)	413.1	3.7E-06	4.3	2.1	1.3	4
C10-13 chloroalkanes	337.0	4.3E-06	6.0	5.3	1995	1600
Chloroparaffins (medium chain)	529.5	3.1E-06	7.0	6.9	76168	1087
Dioxins and dioxin-like compounds*	322.0	4.5E-06	6.8	6.7	48457	41540
Decamethylcyclopentasiloxane (D5)	370.8	4.0E-06	8.0	5.2	1500	7060
Tris(2-chloroethyl) phosphate (TCEP, organophosphorus flame retardant)	285.5	4.9E-06	1.8	2.0	1.1	5.1
Diflubenzuron	310.7	4.6E-06	3.9	3.7	46	320
Teflubenzuron	381.0	4.0E-06	5.4	4.4	261	640
Triclosan	289.5	4.8E-06	4.8	4.0	93	8700
Alachlor	269.8	5.1E-06	3.0	2.0	1.1	50
Chlorfenvinphos	359.6	4.1E-06	4.0	2.7	4.8	170
Chlorpyrifos	350.6	4.2E-06	5.0	3.6	44.4	1374
Endosulfan	406.9	3.8E-06	4.7	4.2	145	5000
Trifluralin	335.3	4.3E-06	5.3	3.9	86	5674

* comprises dibenzodioxins, dibenzofurans and dioxin-like PCB compounds (DL-PCBs); see footnote 9 in Annex I to Directive 2013/39/EU priority substances in the field of water policy.

Appendix II – Threshold values for ecological risk

Threshold values in water and sediment corresponding to the threshold between NEA's environmental quality Classes II and III are documented in NEA's report M-241/2014. The table covers threshold values for concentration only, not for toxicity (see [Box 3](#)).

Table II Threshold values for ecological risk in water and sediment

Name	Class II/III water µg/L	K _{oc} L/kg _{oc}	K _d sed at TOC 1% L/kg d.w.	Background sediment mg/kg	Class II/III sediment mg/kg
Metals					
Arsenic	0.6	-	6607	15	18
Lead	1.3	-	154882	25	150
Cadmium	0.2	-	130000	0.2	2.5
Copper	2.6	-	24409	20	84
Chrome total (III + VI)	3.4	-	120000	60	660
Mercury	0.047	-	100000	0.05	0.52
Nickel	8.6	-	7079	30	42
Zinc	3.4	-	110000	90	139
PAH					
	µg/L			µg/kg	µg/kg
Naphthalene	2.0	1300	13	2	27
Acenaphthalene	1.3	2600	26	1.6	33
Acenaphthene	3.8	5100	51	2.4	96
Fluorene	1.5	10200	102	6.8	150
Phenanthrene	0.51	37200	372	6.8	780
Anthracene	0.10	29500	295	1.2	4.6
Fluoranthene	0.006	97700	977	8	400
Pyrene	0.023	58900	589	5.2	84
Benzo(a)anthracene	0.012	501200	5012	3.6	60
Chrysene	0.070	398100	3981	4.4	280
Benzo(b)fluoranthene	0.017	831900	8319	90	140
Benzo(k)fluoranthene	0.017	794300	7943	90	135
Benzo(a)pyrene	0.0002	831800	8318	6	183
Indeno(1,2,3-cd) pyrene	0.0027	2344200	23442	20	63
Dibenzo(a, h)anthracene	0.00060	1949800	19498	12	27
Benzo(ghi)perylene	0.00082	1023300	10233	18	84
Total PAH 16				< 300	2000

Table II - Continued from previous page.

Name	Class II/III µg/L	K _{oc} L/kg _{oc}	K _d sed at TOC 1% L/kg d.w.	Background µg/kg	Class II/III µg/kg
Other organic substances					
DDT	0.025	6215900	62159	-	15
Tributyltin (TBT ion)	0.00020	1100	11	-	35
Lindane	0.0020	3700	37	-	0.074
Hexachlorobenzene	0.013	130000	1300	-	17
Pentachlorobenzene	0.001	40000	400	-	400
Trichlorobenzene	0.4	1400	14	-	5.6
Hexachlorobutadiene	0.0030	11200	112	-	49
Pentachlorophenol	0.40	3400	34	-	14
Octylphenol	0.010	2700	27	-	0.27
Nonylphenol	0.30	5400	54	-	16
Bisphenol A	0.15	715	7.2	-	1.1
Tetrabromobisphenol A	0.25	49700	497	-	108
Pentabromodiphenyl ether	0.0000000024	565900	5659	-	62
Hexabromocyclododecane	0.00080	45709	457.09	-	34
Perfluorinated octyl sulphonate (PFOS)	0.00013	1000	10	-	0.23
Diuron	0.20	355	3.55	-	0.71
Irgarol	0.0025	1400	14	-	0.036
PCB7	-	321119	3211	-	4.1
Triphenyltin	0.0019	1900	19	-	35
Dodecylphenol with isomers	0.0040	110000	1100	-	4.4
Di(2-ethylhexyl) phthalate (DEHP)	1.3	165000	1650	-	10000
Perfluorooctanoic acid (PFOA)	9.1	125	1.3	-	71
C10-13 chloroalkanes	0.40	199526	1995.26	-	800
Chloroparaffins (medium chain)	0.050	7616755	76168	-	4600
Dioxins and dioxin-like compounds*	0.0000000019	4845739	48457	-	0.00086
Decamethylcyclopentasiloxane (D5)	0.17	150000	1500	-	44
Tris(2-chloroethyl) phosphate (TCEP, organophosphorus flame retardant)	6.5	110	1.102	-	72
Diflubenzuron	0.0040	4600	46	-	0.20
Teflubenzuron	0.0025	26062	260.62	-	0.0004
Triclosan	0.10	9300	93	-	9.3
Alachlor	0.30	112	1.12	-	0.30
Chlorfenvinphos	0.10	480	4.8	-	0.50
Chlorpyrifos	0.030	4440	44.4	-	1.3
Endosulfan	0.00050	14500	145	-	0.073
Trifluralin	0.030	8551	85.51	-	1600

* comprises dibenzodioxins, dibenzofurans and dioxin-like PCB compounds (DL-PCBs); see footnote 9 in Annex I to Directive 2013/39/EU priority substances in the field of water policy. TEQ: toxic equivalents according to the World Health Organisation 2005 "Toxic Equivalence Factors".

Appendix III – Human exposure threshold values

Human exposure threshold values based on exposure through consumption of seafood and direct ingestion of or skin contact with sediment, water and suspended matter. MTR/TDI are the threshold values when sediment-related exposure is the only source of hazardous substances. 10% MTR/TDI corresponds to the threshold values when only 10% of the exposure is sediment-related. TDI is the Norwegian Food Safety Authority's lifetime tolerable daily intake of hazardous substances (only available for a selection of substances).

Table III Human exposure threshold values based on lifetime dose, direct exposure (recreation) and consumption of fish. The lower of MTR value or the Norwegian Food Safety Authority's TDI is chosen. Threshold values in sediments apply to exposure to 10% MTR/TDI.

Name	MTR/TDI [µg/kg/d]	MTR/TDI 10% [mg/kg/d]
Metals		
Arsenic	1	1.0E-04
Lead	3.6	3.6E-04
Cadmium	0.5	5.0E-05
Copper	163	1.6E-02
Chrome total (III + VI)	5	5.0E-04
Mercury	0.71	7.1E-05
Nickel	50	5.0E-03
Zinc	500	5.0E-02
PAH		
Naphthalene	40	4.0E-03
Acenaphthalene	50	5.0E-03
Acenaphthene	500	5.0E-02
Fluorene	40	4.0E-03
Phenanthrene	40	4.0E-03
Anthracene	40	4.0E-03
Fluoranthene	50	5.0E-03
Pyrene	500	5.0E-02
Benzo(a)anthracene	5	5.0E-04
Chrysene	50	5.0E-03
Benzo(b)fluoranthene	5	5.0E-04
Benzo(k)fluoranthene	5	5.0E-04
Benzo(a)pyrene	0.5	5.0E-05
Indeno(1,2,3-cd) pyrene	5	5.0E-04
Dibenzo(a, h)anthracene	0.5	5.0E-05
Benzo(ghi)perylene	30	3.0E-03
Total PAH 16	-	-

Table III - Continued from previous page.

Name	MTR/TDI [µg/kg/d]	MTR/TDI 10% [mg/kg/d]
Other organic substances		
DDT	10	1.0E-03
Tributyltin (TBT ion)	2.5	2.5E-04
Lindane	1	1.0E-04
Hexachlorobenzene	0.16	1.6E-05
Pentachlorobenzene	0.65	6.5E-05
Trichlorobenzene	8	8.0E-04
Hexachlorobutadiene	0.2	2.0E-05
Pentachlorophenol	3	3.0E-04
Octylphenol	0.000067	6.7E-09
Nonylphenol	50	5.0E-03
Bisphenol A	1000	1.0E-01
Tetrabromobisphenol A	1000	1.0E-01
Pentabromodiphenyl ether	1000	1.0E-01
Hexabromocyclododecane	100	1.0E-02
Perfluorinated octyl sulphonate (PFOS)	0.15	1.5E-05
Diuron	7	7.0E-04
Irgarol	23	2.3E-03
PCB7	0.02	2.0E-6
Triphenyltin	0.25	2.5E-05
Dodecylphenol with isomers	50	5.0E-03
Di(2-ethylhexyl) phthalate (DEHP)	48	4.8E-03
Perfluorooctanoic acid (PFOA)	1.5	1.5E-04
C10-13 chloroalkanes	100	1.0E-02
Chloroparaffins (medium chain)	4	4.0E-04
Dioxins and dioxin-like compounds*	0.00001	1.0E-09
Decamethylcyclopentasiloxane (D5)	250	2.5E-02
Tris(2-chloroethyl) phosphate (TCEP, organophosphorus flame retardant)	120	1.2E-02
Diflubenzuron	12	1.2E-03
Teflubenzuron	10	1.0E-03
Triclosan	250	2.5E-02
Alachlor	5	5.0E-04
Chlorfenvinphos	0.5	5.0E-05
Chlorpyrifos	10	1.0E-03
Endosulfan	6	6.0E-04
Trifluralin	24	2.4E-03

* comprises dibenzodioxins, dibenzofurans and dioxin-like PCB compounds (DL-PCBs); see footnote 9 in Annex I to Directive 2013/39/EU priority substances in the field of water policy. TEQ: toxic equivalents according to the World Health Organisation 2005 "Toxic Equivalence Factors".

References:

Baars, A.J., R.M.C. Theelen, P.J.C.M. Janssen, J.M. Hesse, M.E. van Apeldoorn, M.C.M. Meijerink, L. Verdam and M.J. Zeilmaier, 2001, Re-evaluation of human-toxicological maximum permissible risk levels. RIVM report 711701025. National Institute of Public Health and the Environment, Bilthoven.

TDI/TWI values for metals and DDT were determined by the Joint FAO/WHO Expert Committee on Food Additives (JECFA), the value for 2,3,7,9-TCDD by the EU Scientific Committee on Food (SCF) and the value for TBT by WHO.

Appendix IV – Methods for calculating human exposure to contaminated sediment

Box IV.1 Human exposure via oral ingestion of sediment

Direct exposure via oral ingestion of sediment occurs through hand or mouth contact in shallow water. Thirty incidences of bathing per year are assumed. Exposure can be calculated using standard parameters where measured values are not available.

$$DEI_{sed} = \frac{f_{exp} \cdot DI_{sed} \cdot af \cdot C_{sed}}{BW}$$

DEI_{sed} = direct exposure via oral ingestion of sediment (mg/kg bw/d)

f_{exp} = fraction of time exposed (30 d/365 d)

DI_{sed} = ingestion of sediment (child: $1 \cdot 10^{-3}$ kg d.w./d, adult: $3.5 \cdot 10^{-4}$ kg d.w./d)

af = absorption factor (1)

C_{sed} = concentration in sediment (mg/kg d.w., measured)

BW = bodyweight (child: 15; adult: 70 kg)

Box IV.2 Human exposure via ingestion of surface water

Direct exposure via ingestion of seawater while swimming can be calculated using standard parameters where measured values are not available. Thirty incidences of bathing per year are assumed.

$$DEI_{sw} = \frac{f_{exp} \cdot DI_{sw} \cdot af \cdot C_{sw}}{BW}$$

DEI_{sw} = direct exposure via ingestion of seawater (mg/kg bw/d)

f_{exp} = fraction of time exposed (30 d/365 d)

DI_{sw} = ingestion of seawater (child and adult: $5 \cdot 10^{-2}$ l/d)

af = absorption factor (1)

C_{sw} = concentration in seawater (mg/l, measured or calculated, [Box 9](#))

BW = bodyweight (child: 15; adult 70 kg)

If no data from seawater analyses are available, these values can be estimated in accordance with [Box 9](#), or the concentration in the sea can be set equal to porewater concentration (C_{pw}) as a 'worst case' scenario:

$$C_{pw} = \frac{C_{sed}}{K_d}$$

C_{sed} = concentration in sediment (mg/kg d.w., measured)

K_d = partition coefficient sediment/water (substance-dependent, [Appendix I](#))

Box IV.3 Human exposure via ingestion of particulate matter

Direct exposure via ingestion of particulate matter occurs at the same time as ingestion of water while swimming. This can be calculated using standard parameters where measured values are not available. Thirty incidences of bathing per year are assumed.

$$DEI_{pm} = \frac{f_{exp} \cdot DI_{sw} \cdot I_{pm} \cdot af \cdot C_{pm}}{BW}$$

DEI_{pm} = direct exposure via ingestion of particulate matter (mg/kg bw/d)

f_{exp} = fraction of time exposed (30 d/365 d)

DI_{sw} = ingestion of seawater (child and adult: $5 \cdot 10^{-2}$ l/d)

I_{pm} = particulate matter content of water ($3 \cdot 10^{-5}$ kg/l)

af = absorption factor (1)

C_{pm} = concentration in particulate matter (mg/kg d.w., measured or calculated;
metals: $C_{pm} = 1.5 \cdot C_{sed}$; organic: $C_{pm} = 2 \cdot C_{sed}$)

BW = bodyweight (child: 15; adult 70 kg)

Box IV.4 Human exposure via skin contact with sediment

Absorption through the skin is thought to be insignificant for metals. For organic compounds, exposure is calculated based on 30 incidences of bathing per year. The calculation considers differing surface areas of skin in contact with the sediment for children and adults, as well as different amounts of sediment per unit area. Contact time is set to 8 hours before the sediment is washed off, for example by showering.

$$DES_{sed} = \frac{f_{exp} \cdot SA_{sed} \cdot mf \cdot SAD_{sed} \cdot SAB_{sed} \cdot ET_{sed} \cdot af \cdot C_{sed}}{BW}$$

DES_{sed} = exposure via skin contact with sediment (mg/kg/d)

f_{exp} = fraction of time exposed (30 d/365 d)

SA_{sed} = skin surface area for exposure to sediment (child: 0.17; adult: 0.28 m²)

mf = matrix factor (0.15)

SAD_{sed} = skin adherence factor for sediment (child: $5.1 \cdot 10^{-3}$; adult: $37.5 \cdot 10^{-3}$ kg/m²)

SAB_{sed} = skin absorption rate for sediment (child: 0.01; adult: 0.005 hours⁻¹)

ET_{sed} = exposure time, skin to sediment (8 hours/d)

af = absorption factor (1)

C_{sed} = concentration in sediment (mg/kg d.w.)

BW = body weight (child: 15; adult 70 kg)

Box IV.5 Human exposure via skin contact with water

Uptake via the skin is thought to be insignificant for metals. Exposure calculations for adults are based on 30 incidences of bathing per year. The calculation considers differing surface areas of skin in contact with the sediment for children and adults. Contact time is set at 2 hours of swimming for children and 1 hour for adults.

$$DES_{SW} = \frac{f_{exp} \cdot SA_{SW} \cdot SAB_{SW} \cdot ET_{SW} \cdot af \cdot C_{SW}}{BW}$$

- DES_{SW} = exposure via skin contact with water (mg/kg/d)
 f_{exp} = fraction of time exposed (30 d/365 d)
 SA_{SW} = skin surface area for exposure to sediment (child: 0.95; adult: 1.8 m²)
 SAB_{SW} = skin absorption rate for seawater (l/m²/hour, calculated)
 ET_{SW} = exposure time, skin to seawater (child: 2; adult: 1 hours/d)
 af = absorption factor (1)
 C_{SW} = concentration in seawater (mg/l, measured or calculated)
 BW = bodyweight (child: 15; adult 70 kg)

The rate of absorption through the skin from seawater will be substance-dependent and can be estimated as follows:

$$SAB_{SW} = \frac{5000 \cdot (0.038 + 0.153 \cdot \log K_{ow}) \cdot e^{(-0.016M)}}{(5000 + (0.038 + 0.153 \cdot \log K_{ow})) \cdot 1.5}$$

K_{ow} = octanol/water partition coefficient ([Appendix I](#))

If no data from seawater analyses are available, these values can be estimated in accordance with [Box 9](#), or the concentration in the sea can be assumed to be equal to the porewater concentration (C_{pw}) as a 'worst case' scenario:

$$C_{pw} = \frac{C_{sed}}{K_d}$$

Box IV.6. Total human exposure

Total exposures for children (TCH_{sed}) and adults (TAD_{sed}) are calculated by summing the different exposure routes as follows:

$$TCH_{sed} = DEI_{sed} + DEI_{sw} + DEI_{pm} + DEH_{sed} + DEH_{sw} + IEI_f$$

$$TAD_{sed} = DEI_{sed} + DEI_{sw} + DEI_{pm} + DEH_{sed} + DEH_{sw} + IEI_f$$

For IEI_f see [Box 10](#). By assuming that one is a child for 6 years and an adult for 64 years, a lifetime dose can be calculated as follows:

$$DOSE = \frac{6 \cdot TCH_{sed} + 64 \cdot TAD_{sed}}{70}$$

TCH_{sed} = total daily exposure of child to sediment (mg/kg/d)

TAD_{sed} = total daily exposure of adult to sediment (mg/kg/d)

DOSE = average lifetime daily exposure (mg/kg/d)

DOSE is compared with 10% MTR human values ([Appendix III](#)).

Appendix V – Checklist for performance of risk assessments at Level 1 and Level 2

Simplified checklist for performance of risk assessments

A. Retrieval of existing information

→. Describe the sediment area in question

- Geographical location
- Definition of total sediment area for risk assessment
- Definition of sub-areas (if relevant)
- Delimitation of areas affected by propeller erosion
- Aggregate seabed area, area with sediment
- Depth, topography
- Current use of area (recreation, fishing, transport, etc.)
- Ship sailing route (harbour category (see [Box 6](#)) traffic density, fairways, length and depth profile along fairways)
- Relevant neighbouring marine areas
- Overview of existing sediment data

→. Environmental objectives/remedial action objectives

- Describe environmental objectives that have been set for the area
- Are there other environmental considerations and objectives with a bearing on the risk assessment?
- Future planned/desired use of area

B. Acquiring new data

→. Field sampling programme

- Determine the number of stations
- Decide on and plot in station locations
- Determine the number of samples necessary
- Determine the field equipment needed (tools and instruments)
- Make a fieldwork plan (field protocol)

→. Physical, chemical, toxicological analysis programme

- Decide on the parameter list
- Decide on analytical methods
- Choose a laboratory (accredited)
- Establish procedures for data processing and storage and position plotting (GIS)

→. To implement a fieldwork plan, chemical analyses and toxicity tests

C. Execution of Level 1

→. Data analysis

- Calculate the average content of hazardous substances for the area
- Compare the results with the threshold values in the guidelines
- Identify any exceedances
- Determine whether an exceedance of a threshold value applies to only one station, and whether there are grounds for delimiting a sub-area for further risk assessment.
- Decision that the risk is acceptable, or move on to Level 2

D. Execution of Level 2

→. Decide whether the emphasis should be on spreading, human health or adverse ecological effects

→. If desired: establish individual acceptance criteria for spreading

→. Generate supplementary data for Level 2

- Acquire the local data necessary for performing the calculations (see [Box 4](#))
- Carry out the required whole sediment toxicity testing

→. Level 2 calculations

- Make data files for use in the guideline spreadsheets
- If relevant, enter local coefficients and constants into the spreadsheet
- Perform calculations of spreading (diffusion, resuspension, biological transport)
- If relevant, check the probability of the calculation results ([Box 12](#))

→. Evaluate the risk of spreading (Level 2A)

- If specific criteria have been set: identify any exceedance of these

→. Evaluate the risk to human health (Level 2B)

- Decide which exposure routes are to be covered by the risk assessment
- Summarise the flux results that cover these routes
- Determine whether the calculated level of hazardous substances in organisms is consistent with measured values (where measurements exist)
- Determine whether the calculated level in organism tissue exceeds the consumption threshold values

→. Evaluate the risk of adverse ecological effects (Level 2C)

- Determine whether the results of Level 1 exceed the PNEC values for adverse ecological effects
- Determine whether the results of the whole sediment testing indicate that the sediments cause adverse ecological effects
- Calculate the probable average level of hazardous substances in the water above the sediment
- Determine whether these levels exceed the threshold between NEA's Classes II and III for seawater values for adverse ecological effects

→. **Evaluate overall risk from the sediments based on Level 2**

- Identify exceedances of any spreading acceptance criteria
- Identify exceedances of threshold values for adverse effects on human health
- Identify exceedances of threshold values for adverse effects on benthic infauna
- Identify exceedances of threshold values for adverse effects on aquatic organisms
- If relevant: identify the risk of harm to neighbouring areas
- Describe total risk attributable to sediments in relation to environmental objectives
- Decide whether the sediment area has acceptable risk
- If not - choose whether to perform remediation assessment or to carry out Level 3

E. Execution of Level 3

- Evaluate the usefulness of Level 3 surveys in light of the uncertainty in Level 2
- Establish the level of ambition for Level 3: replacement of default values or full numerical modelling

A general checklist for continued execution of Level 3 is inappropriate, because activities have to be adapted to the individual situation.

Appendix VI – Structure of a risk assessment report

Title: Risk assessment of contaminated sediments in

Summary

Introduction

Description of the area being assessed

The following topics must be covered if possible:

- Geographical location (plotted on an appropriate scale)
- General description of the area (topography, depth, current conditions, type of seabed, ecological significance, protection status, current use of area (recreation, fishing, transport etc.), known discharges and pollution sources, environmental quality.
- Ship sailing route
- Definition of total sediment area and if relevant sub-areas with an estimate of total sediment area (A_{sed})
- Map the main area affected by ship traffic
- Estimated seabed area affected by main ship route (A_{ship})

Desired environmental quality

The following topics must be covered:

- Any established environmental and remedial action objectives
- Any relevant environmental considerations; see Guidelines for management of contaminated sediments

Risk assessment Level 1

The following topics must be covered:

- Plotting in of the sediment stations used in the assessment (maps)
- Description of method used for sediment studies (field procedures, field measurements, chemical and toxicological analytical programme).
- Quality control procedures.
- Tables of results for chemical characterisation and toxicity with specified exceedance of threshold values (Spreadsheet 4, Table 1).
- Other results used in interpreting Level 1
- Level 1 conclusion (acceptable risk, or on to Level 2)
 - Compliance with/exceedance of threshold values
 - Conclusions concerning risk due to sediments
 - Delimitation of sub-areas for further risk assessment, where applicable

Risk assessment Level 2

The following topics must be covered:

- Arguments for prioritising of risk elements, if applicable (spreading, human health and ecological effects).
- Description of methods, and results of supplementary data acquired
- Local parameters used and arguments for these.
- The result of the whole sediment testing (details must be shown in an appendix) with a conclusion regarding exceedance of threshold values.
- Results relating to risk of spreading
 - Table of estimated spreading of hazardous substances, in all and via the three transport routes (diffusion, suspension and organisms). Both fluxes (F) and annual

- transport (U) are specified for each substance for the area affected by ships (A_{ship}) and the area not affected by ships ($A_{\text{sed}} - A_{\text{ship}}$), (Spreadsheet 4 Tables 2a and 2b).
 - Evaluation of the spreading estimates (Is the depletion time probable? Spreadsheet 3a)
 - Comparison of the spreading with threshold values (Spreadsheet 4, Table 2a) and if relevant acceptance criteria for spreading.
 - Figure showing relative importance of spreading routes (Spreadsheet 4 figure “Ex Dia max/middle spreading”).
 - The significance of the risk of propeller-induced resuspension.
- Results relating to risk to human health
 - What sort of human exposure is relevant in the risk area (seafood, bathing etc.).
 - Table of calculated total lifetime exposure with exceedance of threshold values (Spreadsheet 4, Table 3).
 - Figure showing relative importance of exposure routes (Spreadsheet 4 figure “Ex Dia max/middle human”).
- Results relating to risk of ecological effects
 - Describe the risk of effects on benthic fauna based on
 - exceedances in Level 1
 - table of measured/calculated porewater concentrations with exceedances of PNEC_w (Spreadsheet 4, Table 4)
 - results of the Level 1 toxicity tests and the whole sediment testing (Spreadsheet 4, Table 5).
 - Describe the risk of effects on water bodies based on
 - table of measured/calculated seawater concentrations with exceedances of PNEC_w (Spreadsheet 4, Table 6)
- Overall risk assessment Level 2
 - Risk to prioritised ecological/human resources
 - Risk in relation to environmental objectives
 - Uncertainty in conclusions and hence consequences

Conclusions and recommendations

The following topics must be covered in this chapter:

- Overall conclusion of the risk assessment
- Ranking of sub-areas for remediation assessment
- Recommendations for next steps in process (Level 3, including remediation planning)

Appendix VII – Default values used in Level 2 and adaptation to local conditions

Default values used in Level 2 and adaptation to local conditions

An overview is provided below of the default values used in Level 2 and proposals for adapting them to local conditions within a realistic range (given in brackets). A simplified classification is also given of variability (Limited, Medium, Wide) of the factors considered. The overview must be regarded as a guide and is intended to indicate the uncertainty associated with using default values in a particular situation.

Before the start of a study to improve the certainty of the factors, a sensitivity analysis should be conducted to determine the importance a change in a factor (within the probable range) will have for the outcome of the calculations. A large change in the numerical value of a factor will not necessarily have an equally large effect on the final result of the calculations. This can be simply illustrated, for example by doubling or halving a parameter in the spreadsheet.

For more detailed information about sensitivity analyses, reference is made to the literature (for example, Saloranta et al., 2011⁴).

Factor 1	Default value Level 2	Variability
n = porosity	0.7 (0.4-0.7)	Medium
Calculated by determining bulk density (ρ_b) and grain density (ρ_s)		

Factor 2	Default value Level 2	Variability
τ = tortuosity	3 (2-5)	Medium
Can be verified by direct determination of <i>in situ</i> diffusion rate		

Factor 3	Default value Level 2	Variability
a = the factor by which the diffusion rate increases because of bioturbation	10 (1-500)	Wide
It should be possible to use discretion to change the factor on the basis of observation/measurement of redox conditions and the presence of macrofauna (animals > 1 mm). The factor can be set at 0 for completely anoxic sediments. It can also be verified by direct determination of the <i>in situ</i> diffusion rate		

Factor 4	Default value Level 2	Variability
D_s = molecular diffusion coefficient (cm ² /s)	Appendix I Substance-dependent	Limited
Can be obtained from the literature or estimated from the substance's molecular weight and chemical structure		

Factor 5	Default value Level 2	Variability
C_{pw} = porewater concentration (mg/l)	$C_{pw} = C_{sed} [mg/kg]/K_d$ Substance-dependent	Wide

⁴ Saloranta TM, Ruus A, Borgå K. 2011. Identification of the most influential factors in the Norwegian guidelines for risk assessment of dispersion of contaminants from sediments. *Integrated Environmental Assessment and Management* 7:657-667.

Draft procedure; see appendices VIII.2.5 and VIII.6		
Factor 6	Default value Level 2	Variability
Δx = diffusion length	1 cm (0.05-2)	Medium
The diffusion length is determined by the thickness of the benthic boundary layer between sediment and water, which depends in its turn on grain size, seabed geometry (roughness) and current velocity. Can be verified by direct determination of <i>in situ</i> diffusion rate. For a more detailed explanation, see: Boudreau and Jørgensen (2001).		

Factor 7	Default value Level 2	Variability
f_{diss} = dissolved fraction, the fraction of the hazardous substance content that can dissolve in water after resuspension	10/ K_d	Wide
Leaching test at L/S=10; see Appendix VIII.3.1		

Factor 8	Default value Level 2				Variability
m_{sed} = quantity of resuspended fine fraction sediment in dry weight, kg per port call one way (assumed distance 120 m)	Sediment type/harbour category	Large harbour	Industrial harbour	Small boat harbour	Wide
	Silt and clay	2000	1000	150	
	Sand	200	100	15	
	Gravel and stones	20	10	1	
This must be based on direct measurements of turbidity and the quantity of suspended material during manoeuvring of ships in the area in question, if relevant supplemented by sedimentation tests in the laboratory. See Appendix IX .					

Factor 9	Default value Level 2	Variability
C_{bio} = tissue concentration in benthic fauna (mg/kg d.w.) (measured or calculated)	Calculation using K_d and BCF	Wide
A bioaccumulation test should be sufficient for most purposes. The alternative is chemical analysis of the biomass of local benthic fauna. Collection by means of a grab usually provides too little sample material. Epibenthic sleds are recommended.		

Factor 10	Default value Level 2	Variability
OC_{cbio} = quantity of organic carbon in the benthic fauna biomass	0.25 g/g d.w. (0.2-0.3)	Limited
The variation among different fauna is small, and there is little need for local measurements. Can be analysed directly on samples of local sediment fauna.		

Factor 11	Default value Level 2	Variability
OC_{sed} = addition of organic carbon to the sediment from external sources	200 g/m ² and year (10-200)	Medium
This is most easily measured using sediment traps near the seabed. The measurements should cover different production seasons. Analysis of TOC in trap material; see Appendix VIII.3.5		

Factor 12	Default value Level 2	Variability
d = fraction of added organic carbon that is not lost through respiration in sediment	0.47 (0.3-0.6)	Medium
Can be calculated from the difference between measured TOC in sediment traps and measured TOC in the sediment just below the bioactive layer. Can also be measured directly through degradation tests.		

Factor 13	Default value Level 2	Variability
OC_{resp} = organic carbon lost through respiration in the sediment	31 g/m ² /year (10-70)	Medium
Done by measuring total oxygen consumption or CO ₂ release over time in degradation tests with local sediments, either in mesocosm tests or by using a respiration chamber <i>in situ</i> .		

Factor 14	Default value Level 2	Variability
BCF_{fish} = bioconcentration factor water/fish (l/kg w.w.)	Appendix I Substance-dependent	Wide
Most reliably calculated by combining the results of the bioaccumulation test (yields BSAF) with analysis of porewater concentrations (yields K_d). Both must be done on the <i>in situ</i> sediments. Can also be calculated from analyses of fish/shellfish from the area combined with water analyses.		

Factor 15	Default value Level 2	Variability
K_d = partition coefficient sediment/water (l/kg)	Appendix I	Wide
1. Adjusted for measured organic carbon content (Box 10) 2. Direct measurement through porewater analysis (see Appendices VIII.0 and VIII.0). 3. Indirect measurement by means of passive samplers (see for example Appendix VIII.0)		

Factor 16	Default value Level 2	Variability
V_{sea} = water volume over the sediment (m ³)	None	Wide
Most easily calculated from data on seabed area, topography and depth		

Factor 17	Default value Level 2	Variability
t_r = residence time of water in the sediment area	0.02 year (0.003-0.1)	Wide
Calculated based on vertical density profile and flow measurements, preferably with a profiling flow meter. In technical terms, time spent is total water volume in the area divided by the rate of water transport over the interface with the area outside. The result of the calculations must be interpreted considering topography, density profiles and oceanographic experience.		

Factor 18	Default value Level 2	Variability
Maximum bioturbation depth (bioactive layer)	upper 10 cm (0-20)	Medium
The bioactive layer will normally vary from 0 to about 20 cm, depending on the type of fauna present. Alternatives: - Calculated on the basis of chemical vertical profiles (sectioned core samples, standard ISO TC 147/SC6 2003) - Read visually from sediment profile images taken with an SPI camera - Measured by testing trace elements in mesocosms		

Factor 19	Default value Level 2	Variability
f_{dw} = fraction dry matter from wet sediment	0.35 (= 35%) (0.3-0.6)	Limited
Calculated as 1 - measured fraction water		

Factor 20	Default value Level 2	Variability
Distribution of hazardous substance with sediment depth	Concentration in the whole bioactive layer the same as measured in Level 1	Wide
Sectioned core samples (standard ISO TC 147/SC6 2003), analyses as for surface samples in Level 1		

Factor 21	Default value Level 2	Variability
r_{sed} = sedimentation rate new sediment	3 mm/m ² /year	Medium
This is most easily measured using sediment traps near the seabed. The measurements should cover different production seasons. Analysis of total dry weight of precipitate. Can also be estimated by isotope dating of sediment cores. This shows the average sedimentation rate over a number of years going back in time, but not the current state.		

Factor 22	Default value Level 2	Variability
Concentration of hazardous substances in sedimenting material	Same as measured in the sediments in Level 1	Wide
This is most easily measured by analysing material in sediment traps near the seabed. The measurements should cover different production seasons.		

Factor 23	Default value Level 2	Variability
Proportion of total human exposure to hazardous substances attributable to sediments.	10 %, for TBT 100% from seafood	Wide
In principle, this is a factor that will vary for each compound as a function of content in the different types of nutrients. The nearest source of information is the Norwegian Food Safety Authority.		

Factor 24	Default value Level 2	Variability
DI_f = daily ingestion of fish / shellfish	Child: 0.028 kg ww/d Adult: 0.138 kg ww/d	Medium
Information from the Norwegian Food Safety Authority, or local or regional food safety authorities, or separate survey of local eating habits		

Factor 25	Default value Level 2	Variability
KF_f = contaminated fraction of consumed seafood	0.5	Medium
The factor indicates that 50% of seafood comes from the contaminated area, the rest comes from outside and is not contaminated. Local or regional food safety authorities should be able to judge the importance of locally caught seafood, and hence the durability of the default value.		

Factor 26	Default value Level 2	Variability
af = absorption factor	1	Limited
The factor indicates that all exposure leads to absorption. Local conditions have little effect, and supplementary investigations are hardly necessary.		

Factor 27	Default value Level 2	Variability
BW = body weight	Child: 15 kg; Adult 70 kg	Limited
The default value is used in risk assessments of chemicals, and does not vary locally		

Factor 28	Default value Level 2	Variability
C_{fish} = concentration of hazardous substance in fish (mg/kg w.w.)	None	Wide
Analysis of tissue level, for example, according to guidelines issued by OSPAR		

Factor 29	Default value Level 2	Variability
f_{exp} = fraction exposure time for recreation	30 d/yr	Medium
Local values must be based on a population survey (living conditions, recreation pattern)		

Factor 30	Default value Level 2	Variability
DI_{sed} = intake of sediment	Child: $1 \cdot 10^{-3}$ kg d.w./d, Adult: $3.5 \cdot 10^{-4}$ kg d.w./d	Limited
Local variation is hardly of significance, and supplementary surveys of little benefit		

Factor 31	Default value Level 2	Variability
DI_{sw} = intake of seawater	Child and adult: $5 \cdot 10^{-2}$ l/d	Limited
Local variation is hardly of significance, and supplementary surveys of little benefit		

Appendix VIII - Sampling and analytical methods for contaminated marine sediments

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1. Background and aim

The aim of the appendix is to describe the sampling and analytical methods recommended for use in procuring the data necessary for conducting a risk assessment according to the Norwegian Environment Agency's Guidelines. The choice of method has as far as possible been based on Norwegian Standards. In the absence of a Norwegian Standard, emphasis is placed on using national and international standards that are either approved or being drawn up, or methods that are incorporated in major monitoring programmes or that are well documented in international journals. Should there be a wish to use methods other than those proposed (see below), these should also adhere to Norwegian or international standards and meet the same precision and sensitivity requirements. The recommended methods are largely already in use in Norwegian environmental monitoring.

The document covers analyses of the parameters used to carry out Levels 1 and 2 of the risk assessment. Methods that are relevant for a Level 3 are not described in detail, only procedures for testing bioaccumulation potential and porewater extraction, and proposals for an approach to improving the precision of the calculation tool compared with Level 2. This is because Level 3 will have to be adapted to the individual sediment conditions, and the choice of method is therefore free in principle. However, if Level 3 consists only of an extension of the tests performed in Levels 1 and 2 (more analyses etc.), the same sampling and analysis methods are required.

There are often several methods for both sampling and analysis that are used for the same purpose. Alternative sampling methods and collection equipment may yield samples that differ in both content and quality, and alternative analytical methods may vary in precision and accuracy. If different sediment areas are to be compared with respect to both classification of environmental quality and environmental risk, the underlying data should be generated using the same methods and with the same quality requirements.

In many cases, there are several sampling and analytical methods that could yield virtually the same results, and both sampling and analytical methods for hazardous substances are also undergoing continuous development. It is therefore not a requirement that the methods given in this appendix be used, if the laboratory can document that alternative methods yield sufficiently accurate results with satisfactory sensitivity and precision. Intercalibration, the use of certified reference material and other quality control procedures are employed for this purpose. Emphasis is placed on the use of well documented analytical methods and laboratories that perform well in intercalibration exercises (ring tests) and can document their results satisfactorily. Laboratories that perform analyses should be accredited for the methods used. In these cases, the quality requirements are incorporated in the accreditation. There may be a limited number of laboratories with accredited methods for some parameters. In such cases, it is important that the analyses of these parameters are performed according to the same QA/QS principles the laboratory has for its accredited analyses.

The document does not set specific requirements for quantification thresholds for the chemical analyses; these are implicit, as the results must be used for comparison with the threshold values laid down in the classification and risk system, where it should be possible to quantify exceedance of environmental quality Class II.

2. Field and sampling methods

2.1 Fieldwork

General guidelines for carrying out fieldwork in connection with environmental mapping are given in Norwegian Standard NS 9420 (see overview of applicable standards in chapter 0).

2.2 Location and number of stations

The selection of stations for sampling sediments for risk assessment must take place in accordance with NS-EN (Norwegian Standard-European Norm) ISO 5667-19:2004. Sampling points (stations) must be located to be as representative as possible of the area to be assessed. This is carried out on a discretionary basis, drawing on knowledge of the area; first and foremost, bottom type (hard bottom/sediments), topography and depth conditions (from maritime charts). A preliminary survey using echo-sounding, sonar, video camera or other sediment profile camera may be useful for deciding whether the area should be divided up into sub-areas to be sampled individually, and for determining how deep the biologically active layer (bioactive layer) extends in the sediment.

The number and location of stations will always be based to some extent on discretion. The risk guidelines require a minimum of five stations from each area, with each station representing a maximum area of 10 000 m² (for areas deeper than 20 m: 40 000 m²). In the case of complex areas that must be divided up into sub-areas it may be necessary to increase the number of stations. The number and location of stations also depends on whether there are existing sediment analyses from the area that may be included in the risk assessment. If no major changes have taken place in an area (seabed activities, changes in discharges etc.), and if previous analyses are of satisfactory quality, sediment data from the last 10 years can be used.

NS-EN ISO 5667-19 also provides guidelines about the need for parallel samples. The risk guidelines recommend analysing a composite sample of 4 parallel individual samples for each station (see [section 3.2.1](#)). Keeping a sub-sample of each of the parallel samples for possible later individual analyses should also be considered, for example if information is wanted about irregular distribution of contamination within the station. However, it is the average conditions at each station that are of interest for the risk assessment. Historical data will not often have been collected in the same way, i.e. as a mix of 4 parallel samples. It is therefore important to check the underlying data before mixing data sets. If these data meet the requirements in the paragraph above, it should be possible to use them along with new data. However, it should be borne in mind that the spreadsheet attaches the same weight to all sediment data entered, and it is self-evident that a composite sample provides a more representative picture of a station average than a single sample. One should be able to assume that this difference is of little significance in a risk assessment.

2.3 Sampling equipment and handling of samples

General requirements regarding choice of sampling equipment and handling of samples are given in NS-EN ISO 5667. Also given are the types of samplers that are suitable for different types of tests and sediment conditions.

The risk guidelines recommend that sampling cover the uppermost 0-10 cm of the sediment. In most cases, this will comprise the bioactive layer, and it will often also encompass some of the sediment below it. A standard 0.1 m² van Veen grab will collect down to a sediment depth of about 10 cm in a muddy sediment but only to about 5 cm in a sandy sediment. Core or box samplers therefore have to be used in sandy sediment to secure samples 10 cm deep.

For sediments that are to be used for whole sediment testing, it is important to avoid layers that contain H₂S. These anoxic sediments will not contain fauna anyway, and any hazardous substances in them are therefore not expected to be dispersed further through the food web.

One important requirement is that all sediment samples be taken with the least possible disturbance of the surface layer. This means that the sampler must be completely open (core sampler) or be lowered slowly (grab sampler) until it reaches the sediment, so that it does not generate a “front wave” that blows away the surface layer. Sample quality can be checked visually when the sample comes up. A good sample should have an undisturbed sediment surface and clear water above the sample.

Sampling from boats is difficult in seabed areas with a lot of stones in addition to finer sediments. In such cases, diving for samples may be advisable, because a diver can see and select areas with sediment. This sampling is normally limited to areas less than 30 metres deep. In practice, open cylinders 30-50 cm in length and 5-10 cm in diameter are used for sampling. The cylinders should be of clear plastic to enable ready inspection of sample quality. They are pressed carefully down into the seabed and closed at the top with a rubber cork before being drawn up. As soon as the cylinder is free of the seabed, a similar cork is inserted at the bottom and the sample brought to the surface without being tipped over.

A composite sample from a station is made by homogenising each of 4 individual samples and then extracting the same quantity of sediment from each for the composite sample. Since the amount extracted may be only a few grams in some cases, thorough homogenisation is important.

A composite sample of 0.3 litre wet sediment is sufficient for the minimum of physical and chemical analyses required for a Level 1 risk assessment. A total of about 10 litres of wet sediment is needed to carry out the Levels 1 and 2 toxicity tests.

The samples for chemical analysis are transferred to sterilised jars with (sterilised) aluminium foil lining the lid. Alternatively, they can be packed directly into sterilised aluminium foil and stored individually in sealed plastic bags. Samples for toxicity tests are stored in clean plastic buckets with lids. The samples are stored frozen.

Requirements regarding observations and registrations during field collection are set out in NS-EN ISO 5667-19.

More stringent requirements must be applied for some chemical substances, such as siloxanes, with respect to personal care products and other sources of contamination (see for example the procedures for NEA's environmental sample bank). It is also important to bear in mind the risk of contamination when preparing samples, prior to analysis. Personnel who handle samples (samplers and laboratory analysts) must not use personal care products that may contain the chemicals in question.

2.4 Sampling for analysis of hazardous substances in biological material.

Extraction from fish tissue samples must take place in accordance with OSPAR (1999).

2.5 Extraction of porewater for toxicity testing

Porewater should be extracted within a maximum of 1-2 weeks of collection. A porewater volume sufficient to cover all the Level 1 tests should be extracted. Porewater is extracted from sediments by centrifuging a homogenised sediment sample at about 1000 g for 45 minutes. The supernatant is decanted off. After resting for about 30 minutes to allow deposition of any suspended particles, the porewater is suctioned off with a pipette or similar and transferred to sample flasks for further use. It will normally be necessary to filter the porewater through 0.2 µm prior to testing to remove microorganisms that may affect the tests, particularly the *Skeletonema* test. If the tests cannot be conducted immediately, the porewater sample can be stored frozen at about -20°C.

Characterisation of the porewater must be performed to provide a basis for interpreting the results. The following parameters are recommended: pH, salinity, TOC, DOC, ammonia and H₂S, and if appropriate also sulphide and nutrients. Any H₂S in the porewater sample should be removed by bubble aeration before the tests.

2.6 Organic extraction for the DR CALUX test

Non-polar hazardous substances (that dissolve in organic solvents) are extracted from the sediments by means of accelerated solvent extraction (ASE). The following procedure is recommended: Two 20 g sediment samples are extracted, and distributed among 3-4 cells, at a temperature of 100°C and pressure of 2000 psi. Extraction takes place with the aid of cyclohexane and dichloromethane (1:1 w/w ratio) in three static cycles, and acetone and dichloromethane (1:1 w/w ratio) in one static cycle. The extraction time per cycle is 5 minutes. Three ml of the extract is evaporated to dryness with the aid of N₂ gas and the contents dissolved in 250 µl ultra-pure dimethyl sulphoxide (DMSO) with the aid of ultrasound. If the contents do not dissolve, a further 250 µl of DMSO is added and the process is repeated. The extracts in DMSO solution are used in the test.

3. Physico-chemical analytical methods

Examples of analytical methods/principles for relevant parameters are given below. As mentioned, it is not a requirement that the methods given in this appendix be used, if the laboratory can document that alternative methods yield sufficiently accurate results with satisfactory sensitivity and precision, through intercalibration, the use of certified reference material and other quality control procedures. Emphasis is placed on the use of well documented analytical methods and laboratories that perform well in intercalibration exercises (ring tests) and can document their results satisfactorily. Laboratories that perform analyses should be accredited for the methods used. In these cases, the quality requirements are incorporated in the accreditation.

There may be a limited number of laboratories with accredited methods for some parameters. In such cases, it is important that the analyses of these parameters are performed according to the same QA/QS principles the laboratory has for its accredited analyses.

3.1 Sampling and performance of leaching tests

The test is recommended as a measure of the mobilisation potential of dissolved hazardous substances in connection with resuspension of sediments. The procedures are described in NS-EN 12457 “Characterisation of waste. Leaching. Compliance test for leaching of granular waste materials and sludges”.

3.2 Measuring shear strength

This measuring is described in NS 8015.

3.3 Grain-size distribution

The method for determining grain-size distribution is described in Buchanan (1984). First the sand fraction is separated from the clay and silt by wet-sieving the untreated sample through a 63 μm sieve. The sand fraction ($> 63 \mu\text{m}$) is sieved again in the dry state using a series of Wentworth sieves with mesh sizes from 2000 μm to 63 μm (for example, sizes 2000, 1000, 500, 250, 125 and 63 μm) and each fraction is weighed. The weight of the total $< 63 \mu\text{m}$ fraction is determined from the dry weight after freeze-drying. This provides a basis for calculating the percentage $< 63 \mu\text{m}$ (silt and clay). Grain distribution within this fines fraction (2-63 μm) can be analysed in various ways. The use of electronic particle counters is the most common today.

The weights of all the fractions are determined to the nearest 0.01 g, and a cumulative weight percentage distribution over the whole size range is calculated for each station. The calculations are then used to determine the median particle diameter and standard deviation, the skewness of the distribution and the kurtosis (how narrow or wide the distribution is).

3.4 Water content

A weighed wet sediment sample is desiccated to complete dryness at 105°C (about 24 hours), after which the sample is weighed again. Water content can also be determined by freeze drying, which in many cases is a step in the preliminary treatment prior to other analyses.

3.5 Organic carbon

Samples of 0.5 to 10 mg are extracted, and the carbonate content is removed by adding 1 M HCl dropwise until no further effervescence occurs. It must be possible to homogenise dried samples to powder form. Dried samples are weighed in tin capsules which undergo complete combustion in oxygen-saturated helium gas at about 1800°C in a CHN analyser. Excess oxygen is removed by means of copper at about 650°C. The combustion gases then pass through a chromatography column, and the CO₂ gas is detected by a hot wire detector. The carbon content is calculated based on the integrated area under the peaks. The results are normally calculated as percentage of dry weight sediment.

In many contexts, gravimetric analysis of total organic content (loss on ignition) is used as an approximate measure of the organic carbon content. This is not a satisfactory basis for the risk analysis, as it must be possible to assess the validity of the partition coefficient K_d with respect to the content of organic carbon, not organic matter.

3.6 Redox

The redox ratio of the sediment (Eh) can be measured with electrodes that are inserted directly into an undisturbed sediment sample (preferably core sample) without the addition of chemicals. An appropriate procedure is described in NS 9401:2007.

3.7 Metals

3.7.1 General

The samples are analysed after digestion with nitric acid in accordance with NS 4770. The sediment is freeze-dried or dried to constant weight at 105°C, or at 40°C if Hg is also to be determined. A weighed quantity of dried material is sieved through a 0.5 mm sieve. The < 0.5 mm fraction is digested with nitric acid in an autoclave in accordance with the standard. Alternatively, 0.5-1.0 g of dried, homogenised sample or about 2 g of wet sample is weighed in a digestion flask and 20 ml 7 mol/l nitric acid added. The flask is closed tightly, and digestion takes place in an autoclave as described above. After cooling, the sample is diluted directly in the flasks by adding 80 ml deionised water, mixed and left to stand until undissolved material is deposited. The analysis is then performed on the clear liquid phase. The actual determination is carried out with ICP-AES, ICP-MS, FIMS (Hg) or with an atomic absorption graphite furnace.

3.7.2 Mercury (Hg)

This method, the cold vapour method, is based on NS 4768, which is comparable with NS 1483. Samples that cannot be analysed immediately must be frozen or freeze-dried immediately after arrival.

Mercury must be in ionised form in the sample solution for the cold vapour technique to be used. A reducing agent (stannous chloride - SnCl_2 or sodium borohydride - NaBH_4) is mixed with the sample and converts the ionic mercury into metallic mercury (Hg). The mercury vapour is conducted into a spectrophotometer where mercury is quantified based on absorbance at 253.7 nm. One advantage of this technique is that non-specific background absorption and matrix interferences are minimal.

An alternative method is to conduct the mercury vapour into an atomic fluorescence spectrometer after the reduction step with stannous chloride or sodium borohydride. The mercury atoms become excited by radiation with a wavelength of about 254 nm and are quantified based on the intensity of the fluorescence radiation they emit.

3.7.3 Cadmium (Cd), lead (Pb), nickel (Ni), copper (Cu), zinc (Zn), arsenic (As) and chrome (Cr)

Super-pure nitric acid is added to samples and they are digested at high pressure and temperature in a microwave-based digestion unit (UltraClave). An UltraClave has one reaction chamber, which ensures that all samples are digested at the same temperature. There is also limited sample handling, which minimises the risk of contamination.

Metal concentrations can be determined by means of an inductively coupled plasma mass spectrometer (ICP-MS).

There is also a metal analysis method based on Norwegian Standard NS-EN ISO 11885 that employs ICP-AES.

3.7.4 Organotin compounds (tributyltin; TBT and triphenyltin; TFT)

The analysis is performed according to Norwegian Standard NS-EN ISO 17353. This method is based on alkaline digestion of aqueous samples of organotin compounds with sodium tetraethylborate and extraction with hexane. The extract is purified with silica. After concentration, tetra-substituted organotin compounds are separated by means of capillary GC-MS, flame photometric detection (FPD) or atomic emission detection (AED). The concentration of butyl- and phenyltin compounds is determined by calibrating the whole procedure with an internal standard mixture. In order to be used in connection with risk and classification, the concentrations must be given as $\mu\text{g/kg}$ dry weight, not as $\mu\text{g Sn/kg}$ dry weight (i.e. concentration of compound, not concentration of tin).

3.8. Organic hazardous substances

3.8.1 PAH (polycyclic aromatic hydrocarbons)

Risk assessment and environmental quality classification are based on analysis of the 16 compounds on the US EPA's list of priority PAHs (PAH_{16}). The concentration must be given for each compound (naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benzo(a)anthracene, chrysene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene, indeno(1,2,3-cd) pyrene, dibenzo(ah)anthracene, benzo(ghi)perylene and the sum of the individual compounds. When summing for classification purposes, concentrations below the

quantification threshold are counted as 0. The method is covered by Norwegian Standard NS-EN 9815. An internal standard is added to the samples, preferably in a quantity equivalent to what is expected in the sample, and extraction takes place with an organic solvent. Soxhlet extraction is performed only on freeze-dried material and with dichloromethane. Alternatively, ASE 200 is used for extraction, and can be used on both wet and dry material. When extraction takes place on wet samples, total residue is determined on a subsample. GPC is used to eliminate interfering substances from all extracts. After purification, extracts are analysed with GC/MSD. Quantification is carried out with the aid of an external and an internal standard. The method is based on GC-FID and is less suitable if the sample contains large quantities of mineral oils.

3.8.2 PCBs (polychlorinated biphenyls)

Risk assessment and environmental quality classification are based as a minimum on analysis of the seven congeners (PCBs 28, 52, 101, 118, 138, 153 and 180) that are normally regarded as most environmentally questionable (PCB₇). Both the concentration of the individual congeners and the sum of these concentrations must be reported. When summing for classification purposes, concentrations below the quantification threshold are counted as 0.

Internal standards are added to the samples for PCB analysis and extraction takes place with organic solvents. Preparation of the samples, extraction and purification of extracts can be done in the same way for PAHs and PCBs. GC/ECD is used to analyse the extracts. Quantification is carried out with the aid of an external and an internal standard.

3.8.3 DDT

This and similar compounds are extracted and determined along with PCBs.

3.8.4 Bromo-organic compounds (PBDEs; HBCDD and TBBPA)

Analysis is carried out according to international standard ISO 22032. The standard covers the determination of selected polybrominated diphenyl ethers (PBDEs) in sediment and sludge with the aid of GC-MS in electron impact (EI) or negative ion chemical ionisation (NCI) mode. If GC-EI-MS is used, the method can be used to determine concentrations in the range 0.05-25 µg/kg of tetra- and octabromo-congeners, and 0.3-100 µg/kg of decabromodiphenyl ether. Concentrations about ten times lower can be quantified with GC-NCI-MS.

Brominated diphenyl ethers are extracted from the dried sample with an organic solvent such as toluene or a mixture of acetone and hexane. The extract is cleaned, for example by means of preparative multilayer silica gel chromatography. The extract is concentrated prior to analysis with GC-EI-MS or GC-NCI-MS. The concentrations are determined with the aid of an internal standard.

HBCDD can be analysed by adding isotope-labelled HBCDD standard to the samples for quantification. The samples are then extracted with organic solvents under a stream of nitrogen, followed by cleaning procedures employing concentrated sulphuric acid and a silica column to remove fat and other interferences. LC-ToF or LC-QToF can be used for quantification. The HBCDD concentration that is used for risk assessment is the sum of all isomers (α-HBCD, β-HBCD, γ-HBCD and others).

If desired, TBBPA can be analysed together with bisphenol A. A mixture of isotope-labelled phenols used in quantification is added prior to extraction. Organic solvents are used for extraction from the

samples, and the extracts are concentrated under nitrogen followed by a clean-up procedure in an SPE column to eliminate interferences prior to analysis. After elution from the SPE column, the extracts are further concentrated under nitrogen before they are analysed. LC-QToF or LC-ToF can be used to analyse samples. High-resolution mass spectrometry offers the highest sensitivity and very little risk of incorrect identification due to interfering compounds.

3.8.5 Dioxins and dioxin-like PCBs

PCDF/PCDD and non-*ortho* PCBs are analysed using the methods described in Schlabach et al. (1993), Oehme et al. (1994) and Schlabach et al. (1995). In brief, the method involves homogenising the samples in Na₂SO₄ before extraction by direct elution with cyclohexane and dichloromethane. ¹³C-labelled 2,3,7,8-substituted PCDD/PCDF is added as an internal standard and the samples are cleaned up in a multi-column system with various types of silica, aluminium oxide and active carbon. GC/MS is then used to determine the compounds. The results are reported as toxic equivalents in accordance with WHO 2005 (Van den Berg et al. 2005).

3.8.6 Hexachlorobenzene (HCB) and pentachlorobenzene

These compounds are extracted along with the PCBs and determined together with them.

3.8.7 Chlorinated paraffins (C10-C13 chloroalkanes and medium-chain chlorinated paraffins)

As a rule, the sediment sample should be dried before extraction. If not, a homogenous aliquot must be taken for separate dry weight determination. In the absence of ¹³C-labelled compounds of chlorinated paraffins, a ¹³C-labelled PCB mixture is added as an internal standard in connection with quantification. Soxhlet extraction, first with acetone and then with cyclohexane, is performed on the sample. After extraction the samples are concentrated, treated with concentrated sulphuric acid and cleaned up in a silica column. A recovery standard is added prior to GC/MS analysis. The determination is performed with the aid of gas chromatography combined with high resolution mass spectrometry operating in negative chemical ionisation mode (GC/HRMS-NCI).

3.8.8. Lindane (hexachlorocyclohexane)

This compound can be extracted and determined together with PCBs and reported as ΣHCH (Σα-, β-, δ-, ε-, γ -HCH).

3.8.9 Octylphenol, nonylphenol and dodecylphenol

The method is described by Meier et al. (2005). An internal standard (deuterated alkyl phenol) is added to the sample, which is homogenised, extracted with dichloromethane and cleaned up with GPC. Phenols are converted to pentafluorobenzyl derivatives and extracted with GC/MS.

Alternatively, alkyl phenols can be analysed with other phenol compounds (bisphenol A and/or tetrabromo-bisphenol A). A mixture of isotope-labelled phenol standards is initially added to the samples for quantification. Organic solvents are then used to extract the samples and a nitrogen stream is used to concentrate them. After cleaning up in an SPE column, the samples are analysed with LC-QTOF.

3.8.10 Dodecylphenol (with isomers)

Solvent extraction is used on the samples. The extracts then undergo cleaning to remove sulphur and other potential disturbances, for example by means of a silica or GPC column. The extracts are concentrated under a nitrogen stream prior to analysis. Analysis is performed by means of GC/MS, for example.

3.8.11 Chlorophenols (pentachlorophenol)

Chlorophenols are extracted from sediments by means of liquid-liquid extraction of the protonated form after acidification, or as the corresponding phenolate ion in an alkaline buffer solution. Methanolic alkali is widely used for extraction, although other solvents can equally well be used. When gas chromatography is used, the phenol compounds must first be derivatised in an alkaline aqueous solution. When HPLC is used for quantifying, derivatisation is not necessary.

3.8.12 Hexachlorobutadiene

Solvent extraction is used on the samples. The extracts then undergo clean-up to remove sulphur and other potential disturbances, for example by means of a silica or GPC column. The extracts are concentrated under a nitrogen stream prior to analysis. Analysis is performed by means of GC/MS, for example.

3.8.13 Trichlorobenzene

Solvent extraction is used on the samples. The extracts then undergo clean-up to remove sulphur and other potential disturbances, for example by means of a silica or GPC column. The extracts are concentrated under a nitrogen stream prior to analysis. Analysis is performed by means of GC/MS, for example.

3.8.14 Alachlor

Solvent extraction is used on the samples. The extracts then undergo clean-up to remove sulphur and other potential disturbances, for example by means of a silica or GPC column. The extracts are concentrated under a nitrogen stream prior to analysis. Analysis is performed by means of GC/MS, for example.

3.8.15 Chlorfenvinphos

Solvent extraction is used on the samples. The extracts are then cleaned up to remove sulphur and other potential disturbances, for example by means of a silica or GPC column. The extracts are concentrated under a nitrogen stream prior to analysis. Analysis performed by means of LC/MS, for example.

3.8.16 Chlorpyrifos

Solvent extraction is used on the samples. The extracts are then cleaned up to remove sulphur and other potential disturbances, for example by means of a silica or GPC column. The extracts are concentrated under a nitrogen stream prior to analysis. Analysis is performed by means of LC/MS, for example.

3.8.17 Endosulfan

Solvent extraction is used on the samples. The extracts are then cleaned up to remove sulphur and other potential disturbances, for example by means of a silica or GPC column. The extracts are concentrated under a nitrogen stream prior to analysis. Analysis is performed by means of GC/MS, for example.

3.8.18 Triflualin

Solvent extraction is used on the samples. The extracts are then cleaned up to remove sulphur and other potential disturbances, for example by means of a silica or GPC column. The extracts are concentrated under a nitrogen stream prior to analysis. Analysis is performed by means of GC/MS, for example.

3.8.19 DEHP

Solvent extraction is used on the samples. The extracts are then cleaned up to remove sulphur and other potential disturbances, for example by means of a silica or GPC column. The extracts are concentrated under a nitrogen stream prior to analysis. Analysis is performed by means of GC/MS, for example.

3.8.20 Perfluoroalkyl substances (PFOS and PFOA)

A mixture of isotope-labelled PFAS is added prior to extraction. The samples are extracted with organic solvents. Buffers are used for pH control. The extracts are cleaned up by means of solid phase extraction (SPE) and concentrated under nitrogen prior to analysis. PFOS are analysed by means of LC/MS/MS (ESI negative mode), or by LCqTOF-MS.

3.8.21 Bisphenol A

A mixture of isotope-labelled phenols used in quantification is added prior to extraction. Organic solvents are used for extraction from the samples, and the extracts are concentrated under nitrogen followed by a clean-up procedure in an SPE column to eliminate interferences prior to analysis. After elution from the SPE column, the extracts are further concentrated under nitrogen before they are analysed. GC-MS, LC-QToF or LC-ToF can be used to analyse samples. High-resolution mass spectrometry offers the highest sensitivity and very little risk of misidentification due to interfering compounds.

3.8.22 D5 (decamethylcyclopentasiloxane)

Liquid-liquid extraction can be used to extract and quantify siloxanes, as well as headspace extraction techniques. Gas chromatography with mass spectrometry detection (GC-MS) can be used to analyse siloxanes. Attention is again drawn to the strong risk of contamination associated with handling samples for siloxane analysis.

3.8.23 Triclosan

Organic solvents are used for extraction from samples. Extracts are cleaned up to remove interferences before undergoing repeated concentration. A switch is then made to methanol/water. Extracts are analysed with LC/MS/MS.

3.8.24 TCEP

Solvent extraction is used on the samples. The extracts are then cleaned up to remove sulphur and other potential disturbances, for example by means of a silica or GPC column. They are concentrated under a nitrogen stream prior to analysis. Analysis is performed by means of GC/MS, for example.

3.8.25 Diflubenzuron

Solvent extraction is used on the samples. The extracts are then cleaned up to remove sulphur and other potential disturbances, for example by means of a silica or GPC column. They are concentrated under a nitrogen stream prior to analysis. Analysis is performed by means of LC/MS, for example.

3.8.26 Teflubenzuron

Solvent extraction is used on the samples. The extracts are then cleaned up to remove sulphur and other potential disturbances, for example by means of a silica or GPC column. They are concentrated under a nitrogen stream prior to analysis. Analysis is performed by means of LC/MS, for example.

3.8.27 Irgarol (cybutryne)

An internal standard for use in quantification is added prior to extraction. Organic solvents are used for extraction from the samples, and the extracts are concentrated under nitrogen followed by a clean-up procedure in a solid phase extraction column to remove interferences before analysis. LC/MS/MS can be used to quantify Irgarol.

4. Toxicity tests

Most of the tests use pure seawater in some way. This document does not include specific requirements regarding seawater quality. The laboratory must be able to provide evidence that seawater quality is under control and fulfils the necessary requirements for ensuring that there is no effect on the results.

4.1 Toxicity to *Skeletonema costatum*

This toxicity test is performed in accordance with International Standard ISO 10253. Concentrated stock solutions of nutrient salts are added to porewater (see section 0 on extraction) and diluted with pure seawater to which the same nutrient salts have been added to various concentrations between 10-100% or lower if necessary. Controls in pure seawater medium are also included in the experimental setup. The solutions are seeded with algae from an exponentially growing *Skeletonema costatum* culture and incubated under constant light at approximately 20°C. Algal growth is recorded through counting or other indirect methods for 3 days and the growth rate calculated. Growth inhibition in the different cultures is calculated as the reduction in growth rate relative to that of the control cultures. Growth inhibition is plotted against the concentration of porewater, and the concentration that gives 50% growth inhibition (EC₅₀) is determined. If growth inhibition from undiluted porewater is <50%, the risk is considered insignificant. From EC₅₀, TU =

100/EC₅₀ is calculated to obtain a unit proportional to the toxicity. The test must be conducted on at least three replicates for each porewater concentration.

4.2 Toxicity to *Tisbe battagliai*

This toxicity test is performed in accordance with ISO (1999). Water Quality - Determination of acute lethal toxicity to marine copepods (Copepoda, Crustacea) International Standard, ISO 14669. The standard describes determination of acute toxicity to one of three specified species of marine copepod. For the porewater test, the copepodite stage of the species *Tisbe battagliai* is recommended. The experimental animals are exposed to a dilution series of porewater with filtered seawater at 20°C and in alternating light/darkness. The percentage mortality is recorded after 24 and 48 hours to calculate 24 hr LC₅₀ and 48 hr LC₅₀. TU is calculated as 100/LC₅₀. For the test to be valid, mortality under control conditions (pure seawater) must not exceed 10%.

4.3 Toxicity to *Crassostrea gigas* larvae

The toxicity test is based on the following standards:

- Ecological Effects Test Guidelines OPPTS 850.1055 Bivalve Acute Toxicity Test (Embryo-Larval). EPA712-C-96-160 April 1996
http://www.epa.gov/ocspp/pubs/frs/publications/OPPTS_Harmonized/850_Ecological_Effects_Test_Guidelines/Drafts/850-1055.pdf
- ASTM (2004) [ASTM E724 - 98\(2004\) Standard Guide for Conducting Static Acute Toxicity Tests Starting with Embryos of Four Species of Saltwater Bivalve Molluscs](http://www.astm.org/Standards/E724.htm)
<http://www.astm.org/Standards/E724.htm>

The test measures the acute toxicity of porewater (48 hr EC₅₀) to the development of larvae of the Pacific oyster *Crassostrea gigas*. This species is also found in some places along the coast of Norway. Larvae are produced directly for the test from adult males and females by raising the temperature to stimulate spawning. Suspensions of eggs and sperm are mixed and the resulting suspension of fertilised eggs is kept at 20°C until use. The suspension is added to a dilution series of porewater with filtered seawater and, after 48 hours, samples are taken and fixed. Larvae with normal and abnormal development are identified and counted under a microscope. The percentage of abnormally developed larvae is used to calculate EC₅₀. TU is calculated as 100/LC₅₀. For the test to be valid, mortality under control conditions (pure seawater) must not exceed 30%.

4.4 Dioxin Receptor CALUX assay

The DR CALUX assay is a reporter gene test that reveals activation of AhR (the aryl hydrocarbon receptor) following the binding of dioxin-like substances (flat PCBs and dioxins). The resulting complex interacts with DNA in the cell nucleus. The analysis converts this effect to a proportional production of the enzyme luciferase, which is measured in a luminometer through light emission following addition of the substrate luciferin. The assay uses a special cell line (DR CALUX cells) that is commercially available. The results of the test are given as TCDD equivalents. For further information on the DR CALUX test, see <http://www.biodetectionsystems.com>

4.5 Whole sediment tests

These tests have not been described in Norwegian or international standards. Both tests measure the behaviour and survival of a test organism in direct contact with the sediment in question. A mortality rate of over 20% is considered significant and is used in Level 2 as the threshold value for unacceptable ecological risk.

4.5.1 Test with *Arenicola marina*

Arenicola marina (lugworm) is a brushworm that lives buried in the sediment and is found in the intertidal zone and to a depth of 20 metres. The worms live in U-shaped mucous-lined burrows and eat free organic matter, large amounts of which pass through their intestines daily. The undigested material can be seen on top of the sediment in small 'spagetti-like' mounds. The species is widespread along the entire Norwegian coast and is found from the Mediterranean Sea to the Arctic.

The sediments to be tested are homogenised, for example, with the aid of an electric drill onto which a paint mixer has been mounted (for sediment use only). Samples of whole sediment are divided between 3 replicate vessels (small cubes, plastic) to a depth of at least 7 cm in each. A pure 'reference sediment' is used as a control. Pure seawater is added to a depth of at least 12 cm from the bottom of the vessels. Oxygen is provided by bubbling air through the samples (aquarium pumps with a weak air flow). The lugworms are added to the vessels the following day, five individuals to each aquarium. Over the course of the experiment, the behaviour and activity of the experimental animals is observed by recording the number of animals that do not burrow down into the sediment and the volume of 'excrement piles' on the sediment surface. The mass of excrement in the test aquaria is classified relative to that in the control sediment using a simple semi-quantitative scale: 2: equal to the control, 1: clearly less than the control and 3: no excrement. Exposure is discontinued after 10 days. Individual worms are carefully sifted from the sediment and the number of living and dead individuals is recorded.

4.5.2 Test with *Corophium volutator*

Corophium sp. is a small crustacean (amphipod) that lives in tubes in sediment, often in dense colonies. It forms a U-tube in the sand with the aid of its own excrement. At low tide it withdraws into the tube, and the opening can often be seen at the surface of the sediment. *Corophium* sp. is found from the Mediterranean to the coast of Norway.

The sediments in question are homogenised as in the *Arenicola* test. Then 250-300 ml of the samples is transferred to 3 replicate beakers. Pure seawater is added to make the volume up to 800 ml. Oxygen is provided by bubbling air through the samples (aquarium pumps with a weak air flow). Twenty individual amphipods are placed in each beaker the following day. The time at which *Corophium* are added is noted for each beaker. The ability of the amphipods to bury themselves is recorded by observing the number of individuals on the surface of the sediment and in the water column after 1 day. The exposure is terminated after 10 days and the number of individuals on the surface of the sediment and in the watercolumn is recorded. Individual *Corophium* are sieved carefully from the sediment/water and the numbers of living and dead individuals is recorded.

5. Bioaccumulation test

It is most advisable to estimate bioavailability by measuring uptake or accumulation of specific hazardous substances in benthic fauna.

The measurements are made in a standardised system for testing the bioavailability of hazardous substances in marine sediments. The test system was developed by the US EPA (Lee et al. 1991) and adapted to Norwegian conditions (Hylland, 1996). Examples of use are given by Knutzen et al. (1995), Skei and Andersen, (1996), Johnsen et al. (1996), Skei et al. (2002) and Schaanning et al. (2002). Two benthic fauna species, the ragworm, *Hediste diversicolor*, and the netted dog whelk, *Hinia reticulata*, are used. Both are very common in shallow waters along the Norwegian coast and tolerate low salinities. Neither species lives directly off sediment. *H. diversicolor* probably lives mainly off smaller organisms. *H. reticulata* is a scavenger and predator but may also use organic material from sediments. The reason two organisms are used is that there may be major differences across species with respect to accumulation of hazardous substances. Ragworms and molluscs represent two important groups in marine sediment systems.

The test is carried out in glass aquariums with a base of about 300 cm². The sediments in question are homogenised with the aid of an electric drill mounted on a paint mixer (used only for sediments). About 1.2 litres of sediment are added to each aquarium (3 aquariums per sediment/station). At the same time, samples of the sediments are taken out for chemical analysis. The aquariums are placed in a temperature-regulated water bath and clean seawater is added to the sediment. The water phase is constantly replaced during the test. Twenty ragworms and ten snails are placed in each aquarium. After 30 days of exposure, the animals are removed and the ragworms are kept in a beaker of clean seawater for 6-12 hours to expel any residual sediment from their guts. The snails are removed from their shells with the aid of a nutcracker. Experience indicates that there is no need for the snails to expel their gut contents (they do not eat sediment particles). All the animals are then distributed into test jars and kept frozen prior to analysis. Standard procedures for analysis of biological material are adhered to when performing chemical analyses of the organisms.

6. Porewater measurement

Experience indicates that K_d values may vary substantially in sediment with legacy contamination or where the contamination is strongly bound up with carbon-containing particulate matter or soot carbon (up to a factor of 100 or more). Instead of using standard K_d values, it may therefore be advisable to determine partition coefficients by measuring the porewater concentration and calculating the K_d value based on the sediment/water ratio of contaminant concentration:

$$K_d = \frac{C_{sed}}{C_{pw}}$$

6.1 Organic hazardous substances

The concentration of hazardous organic substances in porewater is usually measured by means of a long-term shaking test, in which sediment and water are mixed well and shaken until equilibrium is reached between sediment and water phases. The concentration in the water phase is then measured. One drawback of this method is that relatively large quantities of sediment are needed to achieve a porewater concentration above the detection threshold. It may also be difficult to remove colloidal particles, giving rise to an apparently higher porewater concentration. One option is to mix a third solid, which also achieves equilibrium in the sediment and water phases, into the sediment/water mixture. Substances that lend themselves to solid phase extraction of PAH, PCBs, dioxins and PBDEs, and which are well documented in the literature, are LPDE (low density polyethylene), POM (polyoxymethylene) and SPMD (semipermeable membrane device) using PDMS (polydimethylsiloxane) as a passive sampler. Concentrations of hazardous organic substances in solution are calculated from the quantity of substance absorbed by the solid phase polymer and established partition coefficients between the polymer and water:

$$C_{pw} = \frac{C_{polymer}}{K_{polymer}}$$

The advantages of these polymer-based extraction methods are that there is no sorption onto dissolved organic carbon or soot carbon, and that it is possible to measure low porewater concentrations. The partition coefficient $k_{polymer}$ depends on both the type of material to be measured and the type of polymer used.

6.2 Metals

The variation in K_d values in hazardous inorganic substances is due to the sediment properties, pH, redox conditions, and concentrations of ligands such as SO_4^{2-} , Cl^- , S^{2-} or of dissolved organic matter (DOM). Other cations such as Ca^{2+} and Mg^{2+} will also affect K_d . It is therefore very difficult to measure the porewater concentration of metals in a shaking test with a mixture of water and sediment without changing the natural sediment conditions. One advantage of analysing heavy metals is that relatively little water is required, so that it is possible to obtain a sufficiently large water sample by centrifuging the sediment sample. Clean plastic flasks are filled with sediment immediately after sampling and centrifuged. The water phase is taken off, acid added, and it is then ready for analysis.

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8 Relevant standards

NS-EN 1483. Water quality. Determination of mercury. Norwegian Standards Association.

NS 4764:1980. Water analysis - total residue and total fixed residue in water, sludge and sediments. Norwegian Standards Association. [Norwegian text]

NS 4770:1994. Water analysis - Determination of metals by atomic absorption spectrophotometry, atomization in flame - General principles and guidelines. Norwegian Standards Association. [Norwegian text]

NS-EN ISO 5667-19:2004. Water quality - sampling - Part 19: Guidance on sampling in marine sediments (replaces NS 9422). Standard Norge

NS 8015:1988. Geotechnical testing - Laboratory methods - Determination of undrained shear strength by fall-cone testing (withdrawn and incorporated in NS ICS 93.020). Norwegian Standards Association. [Norwegian text]

NS 9410:2016. Environmental monitoring of benthic impact from marine fish farms. Standard Norway. [Norwegian text]

NS 9420:1988. Rules for fieldwork in connection with environmental monitoring and mapping Norwegian Standards Association. [Norwegian text]

NS-EN 16665:2014. Water quality - Guidelines for quantitative sampling and sample processing of marine soft-bottom macrofauna. Standard Norge. [Norwegian text]

NS-ISO 11338-2:2003 (replaces NS 9815:1995). Stationary source emissions - [...] Part 2: Sample preparation, clean-up and determination. Standard Norge. [Norwegian text]

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NS-EN ISO 17353:2005. Water quality - Determination of selected organotin compounds - Gas chromatographic method. Standard Norge. [Norwegian text]

OSPAR 1999. JAMP Guidelines for monitoring contaminants in biota. OSPAR Ref 1999/2.

ISO 10253:2016. Water Quality - Marine algal growth inhibition test with *Skeletonema costatum* and *Phaeodactylum tricornutum*. International Organization for Standardization.

ISO 22032:2006. Water quality - Determination of selected polybrominated diphenyl ethers in sediment and sewage sludge - Method using extraction and gas chromatography/mass spectrometry. International Organization for Standardization.

Appendix IX - Transport because of ship-induced resuspension

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1. Background

One of the routes by which hazardous substances from sediments are dispersed is ship-induced resuspension. Field observations have shown that the quantity of sediment that becomes resuspended varies considerably and is unpredictable. This study provides an estimate of resuspension by drawing on experience from field surveys and literature studies.

Many of the literature studies are collated in Bjerknes (2002). There is a large quantity of literature from some decades ago about erosion of sandy, non-cohesive sediments, but considerably less material on cohesive clay sediments.

The main questions that this account aims to address are the flow velocities generated by ships docking and leaving docks, and the extent to which these flow velocities cause resuspension of bottom sediments. The significance of water depth, sediment state and ship size for resuspension are also of interest.

2 Sediment erosion - general

Erosion of bottom sediments occurs when the force (shear tension) in overlying water (lifting and drag) is greater than the resistance of the sediments due to the weight of the sediment grains and friction (van Rijn, 1993).

When particles begin to be swirled up from the sediment surface, the shear stress has reached a threshold value known as “critical shear stress”. Shear stress τ (force/area) is often expressed in terms of frictional velocity U_* (length/time unit). The relationship between shear stress and frictional velocity can be expressed as:

$$U_*^2 = \frac{\tau}{\rho},$$

where ρ is water density.

Waves and turbulent flow (often generated by ships) cause shear stresses at the sediment surface and are the dominant erosional forces. Site-specific sediment characteristics such as particle size distribution, particle density, cohesivity, water content and biological activity control resistance to erosion. These properties have a bearing on friction against the seabed, which in turn creates turbulence. The relationship between shear stress and flow velocity U in turbulent flow can be expressed:

$$\tau_0 = \rho \cdot c \cdot U^2$$

The coefficient c is a function of distance z to the sediment surface and the irregularities or roughness of the sediment surface.

$$c = \left(\frac{\kappa}{\ln(z/z_0)} \right)^2$$

κ is von Karman's constant (0.4) and z_0 is a measure of roughness, empirically determined as being $z_0 = D/30$ where D is a typical size of irregularities such as surface formations or particle diameter (Bjerkeng 2002).

Most erosion studies consider non-cohesive (sandy) sediments, where the sediments behave like individual grains. Critical flow velocities for these sediments are given in Table IX.1 after Bjerkeng (2002).

Table IX.1. Relationship between increasing particle size (D) and critical velocity (U_c) and shear stress (τ_c) for particles with a density 2.6 times the density of water (after Bjerkeng 2002).

D μm	U_c m/s	τ_c Pa
0.5	0.004	0.01
2	0.005	0.02
15	0.007	0.05
62.5	0.010	0.09
300	0.013	0.17
<300	0.014	0.19
750	0.019	0.37
3 000	0.048	2.3
7 000	0.083	6.8
7 000	0.082	6.8
30 000	0.171	29.1
100 000	0.312	97

However, the sediments in Norwegian fjords and harbours are cohesive in most cases, i.e. the sediment particles do not behave like individual particles but stick together and form aggregates or flakes because of the electrochemical repulsive and attractive forces acting between the particles. Natural sediments in Norwegian fjords and harbours usually consist of a mixture of clay, fine silt and sand. Cohesion begins to play a part when a sediment contains more than 5-10% clay (% by weight) (Dyer 1986).

Because of the strong bonds in cohesive sediments, the sediment grains are not eroded in the same way as in non-cohesive (sandy) sediments. Erosion of cohesive sediments is usually described as consisting of two main types: Type I surface erosion and Type II bulk erosion (Mehta et al. 1982). Type I erosion occurs when critical shear stress increases down through the sediment because of consolidation. In Type I erosion, the erosion rate is initially high but then gradually slows. Type II erosion may also be high initially but does not slow gradually; the sediments continue to erode. This happens when the shear stress substantially exceeds a critical value (Paterson & Black, 1999).

In experimental studies of erosion of cohesive sediments, aggregates have been observed to be gently lifted up from the seabed at very slow current velocities. They form thin, almost misty layers (the fluff layer), but they are not transported away (Lintern et al., 2002). As the current velocity increases, the size of the aggregates that are lifted increases. At a critical velocity, layers of sediment loosen. If the velocity increases further, the surface becomes fluid and layers of sediment continue to erode. At very high velocities, the surface collapses and the erosion penetrates several centimetres deep. This course of events has also been observed during experiments with sediments from Oslo Harbour (unpublished data and observations from erosion experiments at NIVA's marine research station Solbergstrand in 2002).

In the literature, the critical flow velocity for the fluff layer in surface sediments is estimated to be 0.5 cm/s (Ziervogel and Bohling 2003) and < 0.3 m/s (Amos et al., 1997). Repeated frequent disturbances of the sediments prevent consolidation and hence increase erosivity. Erosion experiments on cohesive sediments *in situ* indicate a critical flow velocity for erosion of 30 cm/s for initial resuspension and a halving to 15.7 cm/s for second-time resuspension (Widdows et al., 1998a). With repeated resuspension tests at weekly intervals, the critical velocity fell further to 14 and 12 cm/s.

A literature review by Bjerkeng (2002) and references in the same review show that the critical flow velocity for erosion of cohesive sediments varies from 0.1-1 m/s. The variations in erosivity depend on various sediment properties:

- Some studies show that a certain amount of sand in clay reduces erosivity. Indications are that the critical shear stress may be 30-40 times greater for clay containing some sand than for pure clay or pure sand (Mitchener and Torfs 1996).
- Given that clay minerals flocculate when salt is introduced, it may be expected that the salinity of the porewater has a bearing on the erosion of cohesive sediments. Experiments by Gularte et al. (1980) have shown that the critical shear stress for erosion of cohesive sediments increases up to a salinity of 10. This may be of significance for harbour sediments at river mouths.
- Biological activity in sediments or on the surface of sediments may have both a stabilising and a destabilising effect. Benthic diatoms are regarded as being one of the most important organisms for stabilising sediments, because they excrete extracellular polymeric substances (Holland et al., 1974, and Grant et al., 1986). Benthic macrofauna may also have a stabilising effect because sediment particles are cemented together by the secretion used to make tubes on and in the sediment (Yingst and Rhoads, 1978). In most cases, biological activity leads to destabilisation of the sediments through pellet production, grazing and bioturbation (Gerdol and Huges, 1994; Widdows et al., 1998b; Andersen, 2001).

The critical shear stress or threshold value for erosion is a measure of the erosivity of a sediment surface and says something about the stability of the whole upper sediment layer. Another measure is the erosion rate, defined as the quantity of material eroded per unit time and area for a given shear stress (Amos et al., 1997). The latter describes the stability of the surface sediments under the uppermost layer. Studies by Amos (1997) show that the average erosion rate varies between $1.4 - 7.4 \times 10^{-4} \text{ kg/m}^2/\text{s}$, irrespective of flow strength and the shear strength of the sediment.

3. Ship-induced resuspension

In harbour areas, currents and waves generated by ships are one of the causes of resuspension of bottom sediments. The question then is what flow velocities arise along the seabed when ships dock and leave the dock. This will depend on ship size and draught, mode of propulsion (propeller or water jet), engine power and propeller size and design. Ship movements is often causing sediment erosion and resuspension in three different ways (Bjerkeng 2002):

- Outflow from the propulsion unit (propeller or water jet)
- Suction in to propeller or water jet
- Flow around moving ship to compensate for the displacement of the sunken volume.

3.1 Modelling of water flow ensuing from propeller propulsion

Using available models (Liou and Herbich 1976, Blaauw and van de Kaa 1978) adapted and revised in Bjerkeng (2002) and in Klingenberg Holme (2002), it is possible to calculate the

increase in flow velocity ΔV_{prop} directly behind a propeller as a function of propeller diameter (D) and power applied (W). The calculations are because there is overpressure in front of an operating propeller and underpressure behind it. The pressure difference drives the boat forwards. Behind the propeller, where the water pressure has equalised, the velocity of the water flow (ΔV_{back}) will be twice as high as the increase in water velocity through the propeller (ΔV_{prop}):

$$\Delta V_{back} = 2 \cdot \Delta V_{prop}$$

The diameter of the jet (D_0) behind the propeller, where the pressure is equalised, will also be reduced:

$$D_0 = \frac{D}{\sqrt{2}}$$

The water from a propeller exits as a rotating, turbulent jet. The jet entrains surrounding water because of the viscosity of the water and expands in circumference with increasing distance from the propeller, while the kinetic energy gradually dissipates with the transition to swirling and turbulence (Bjerkeng 2002). The velocity distribution in the jet then changes gradually to a Gaussian or normal distribution (Klingenberg Holme 2002). Within a distance x of the starting point, the core of the jet will have the original velocity, but this core has a linearly decreasing radius:

$$r_0(x) = \frac{D_0}{2 - cx}$$

According to Blaauw and van de Kaa (1978), the establishment zone for a propeller jet is about 2.8 times the diameter D_0 of the jet. This yields the coefficient $c = 0.18$.

Around the core, where the velocity is normally distributed, there will be a velocity for $x < x_0$ given by:

$$V(x, r) = \Delta V_{back} \exp \left(- \frac{1}{2} \left(r - \frac{D_0}{2} + cx \right)^2 / (cx)^2 \right)$$

At distance $x_0 = \frac{D_0}{2c}$, the whole jet will be normally distributed, and the central velocity will decrease with increasing jet radius. For $x > x_0$, the velocity is given by:

$$V(x, r) = \Delta V_{back} \cdot \frac{D_0}{2cx} \exp \left(- \frac{r^2}{2(cx)^2} \right)$$

3.2 Water jets

Today's high-speed boats, both passenger ships and some car ferries, are largely driven by water jets. Knowledge of the effect of erosion due to ships comes largely from propeller propulsion, and the model above is based on this propulsion.

There are various types of water jet, and the design may vary, depending mainly on how the reversal shield is designed and functions. In ships with water jets, water is drawn in through a hole in the bottom of the hull, after which it passes a "blade" that ejects water at high speed in the form of a jet. A water jet with a diameter of 1 metre may have an engine power behind it of for example 20 000 HP. A propeller-driven ship with an engine power of this size could have a propeller diameter of 4-8 metres. The velocity in a water jet is also much higher than that produced by propellers. The usual water velocity from a propeller is 8-10 m/s, whereas a water jet may easily reach 25 m/s.

The water jet outlet is mounted at the surface of the water, and therefore lies high above the ship bottom, unlike a propeller. Consequently, erosion from a water jet is not a problem during forward motion. Only when the ship brakes and manoeuvres does the erosion power become significant. When the engine is running, and the boat is lying still, the water jet is normally active. In order for the boat not to move, the blade is in a position that directs the water jet straight downwards. This may cause a powerful erosion effect. The same happens when the ship brakes or backs. The jet is then directed diagonally downwards, often at around 45 degrees, and the engine power is increased. Figure IX.1 shows flow velocities at various distances from a large water jet engine with a jet exit velocity of 25 m/sec. The figure indicates that such a water jet could even erode sediments at depths greater than 20 m, but that the area eroded in an incident may be relatively limited. The erosion problem is known in expert circles, and the design of the reversing blade is therefore being studied, with a view to limiting the vertical velocity without overly impeding reversibility.

Until a better basis has been obtained for estimating the effects of water jets, it is recommended that in the context of risk, passenger boats and car ferries with jets be classified as large, propeller-driven boats and erosion calculated accordingly.

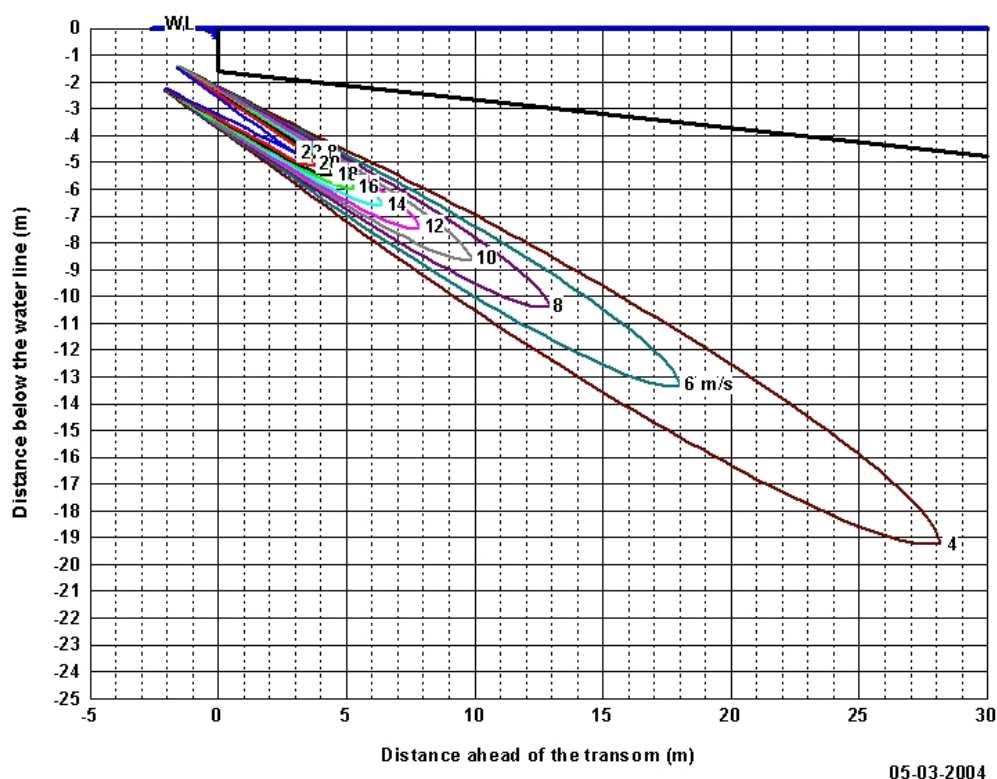


Figure IX.1 Vertical and horizontal extension of flow velocities under a Kamewa water jet with a jet exit velocity of 25 m/sec. (T. Lundestad, personal communication)

3.3 Flow velocities generated by propeller-driven ships

When ships are in motion, the water flow velocity generated by the propeller will be less than if the ship was lying still or manoeuvring at the dock. When the propeller is used to accelerate, brake or turn a ship that is lying still or moving slowly, the drawing of water into the propeller may also cause a “tornado” effect (Bjerkeng, 2002). It is difficult to calculate the velocities that may arise at the seabed in cases like this, but we assume that they are within the range in [Figure IX.2](#).

In the following, the velocities that may develop at the seabed have been calculated from specific data for two types of ship: M/S Color Line Christian IV (in the following called Color Line) and F/F Trygve Braarud (University of Oslo) (Table IX.2). The ships are of different sizes, and perhaps represent the extremes of propeller-driven ships that normally dock in Norwegian harbours. The results are shown in [Figure IX.2](#).

Table IX.2. Ship-specific data for M/S Color Line and F/F Trygve Braarud

Type of ship	Gross tonnage	Breadth / Length (m)	Draught (m)	Engine power (kW)	Propeller diameter (m)	Propeller depth (external edge blade) (m)
M/S Color Line	22000	24.7 / 153.4	5.8	7750	3.6	5.3
F/F Trygve Braarud	106	7.2 / 22	3	610	1.52	2.7

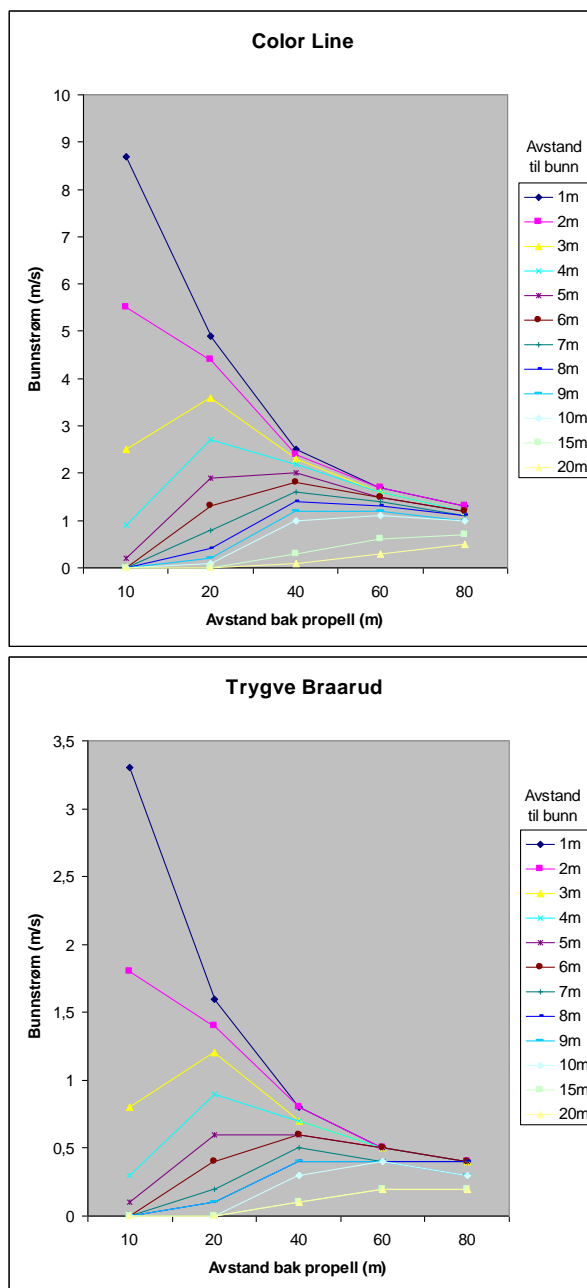


Figure IX.2. Flow at seabed as a function of distance behind propeller and increasing depth below the propeller, for Color Line and Trygve Braarud. "Bunnstrøm (m/s)" = Flow along seabed (m/s); "Avstand bak propell (m)" = Distance behind propeller (m) and "Avstand til bunn" = Distance from seabed.

The calculations are based on the equations in the introduction, and it is assumed that the ships effectively have no velocity when docking. This is a conservative choice designed to yield the highest velocities for the waters. The calculations are presented in Figure IX.2 and show that "Color Line" can generate flow velocities at the seabed of up to 8.7 m/s when there is only 1 m clearance between propeller and underlying seabed. The corresponding velocity for "Trygve Braarud" is 3.3 m/s. The figure shows that with a clearance of 5 m between propeller and seabed, the flow velocity is considerably reduced, to 2 m/s and 0.6 m/s, respectively, for the two ship types.

With a clearance of 5 m between propeller and seabed, the greatest flow velocity arises 20-40 m behind the propeller. When the clearance is narrower, the highest flow velocities occur 10-20 m behind the propeller. The difference in flow velocity between the different clearances diminishes with increasing distance behind the propeller. For "Color Line", the flow velocity at the seabed will be reduced to 1.3-0.5 m/s 80 m behind the propeller with clearance between the propeller and the seabed of from 1 to 20 m. Similarly, the flow velocity in the case of "Trygve Braarud" will be reduced to 0.2-0.4 m/s.

The water depth at docks in most harbours is 10-15 m. This means that in most cases there will be clearance of <1-5 m between propeller and seabed at the dock. Even smaller boats, such as "Trygve Braarud", could thus generate flow velocities at the seabed that exceed the critical velocity for resuspension and erosion of cohesive sediments. The critical velocity can be assumed to be 0.1-1 m/s. Thus, the water depth below the propeller is the deciding factor for how high the flow velocity will be at the seabed.

These calculations estimate the velocity in the core of the jet behind the ship. As mentioned, the radius of the core jet decreases with growing distance from the ship. There is also a slowing of water velocity out to the sides, and the velocity is normally distributed. The steepness or shallowness of this curve determines the area of influence of the propeller. In order to be able to calculate the total amount of sediment that is resuspended per ship docking, it is also necessary to know the area of influence. In the following, we assume an area of influence approximately equivalent to the ship's breadth.

In order to be able to calculate how much sediment is resuspended per docking, it is also necessary to know the erosion rate or how deep down in the sediment the erosion extends. According to Amos (1997) an erosion rate of between $1.4 - 7.4 \times 10^{-4} \text{ kg/m}^2/\text{s}$ can be expected, depending on flow strength and the shear strength of the sediment. Visual observations by Lintern et al. (2002) showed that a high flow velocity can erode several centimetres down into the sediments. On the basis of these observations and the authors' own observations from experimental erosion studies at NIVA's marine research station Solbergstrand (unpublished data), we assume the following erosion depth, depending on the clearance between propeller and seabed (Table IX.3). When erosion is strong, the sediments are torn off in flakes, and erosion extends several cm down into the sediments, an estimated maximum of 5 cm. At lower velocities, only the surface fluff layer will be eroded (0.3 cm). This is assumed to take place when there is approximately 10 m clearance between propeller and seabed. The fluff layer is normally a few mm.

Table IX.3. Proposed erosion depth as a function of clearance between propeller and seabed, and associated quantity eroded.

Clearance under propeller (m)	Erosion depth (cm)	Mass eroded (kg w.w./m ²)	Mass eroded (kg d.w./m ²)
1	5	65	22.8
2	2	26	9.1
5	1	13	4.6
8	0.5	6.5	2.3
10	0.3	3.9	1.4
15	0.2	2.6	0.9
20	0.1	1.3	0.5

3.4 Experience from field studies

Field studies in Sandefjord (DNV 2005), Bispevika (Magnusson 1995), the mouth of the Skien River (Molvær 2002) and in Kristiansand outside the docks of Glencore Nikkelverk AS, the former Falconbridge Nikkelverk and Elkem Carbon Fiskaa (Bjerkeng 2002) were based on flow and turbidity measurements. The total quantity of resuspended particles per ship docking is estimated based on the area of influence of particle clouds and the quantity of particles in the clouds generated by a single docking. These studies show that the quantity of resuspended sediments varies substantially, from 2800 kg in Sandefjord, between 40 and 165 kg in Bispevika, Oslo, 400 kg at the mouth of the Skien River, 60-100 kg at Elkem Carbon and 100-200 kg at Falconbridge Nikkelverk.

The reasons for the variation are complex, but location (openness), water depth and sediment type are of great importance. The approach to Sandefjord harbour is long and shallow, and there is sediment all the way, although here, too, the seabed is reported to be eroded / swept away outside the docks (DNV, 2005). By way of contrast, the area outside Elkem Carbon Fiskaa consists partly of hard bottom in the shallowest areas, and sediments, mainly sand, in deeper waters. As mentioned, repeated erosion increases the erosivity of the sediments. Depending on the residence time of the water in the area in question, the area may also be drained of erodible sediments, or only the coarse fraction or solid clay may be left.

3.5 Modelling versus field studies in Sandefjord

In the following, we want to check whether there is agreement between calculations based on field measurements and calculations based on the risk tools described in previous chapters. The field studies used for the comparison are from DNV's (2005) measurements of turbidity and calculations of the sediment quantity that is resuspended in connection with ferry dockings in Sandefjord.

3.5.1 Calculations based on field studies in Sandefjord

The Color Line shipping company has the ships Color Viking and Bohus in scheduled traffic from Sandefjord. Color Viking is the larger of the two, with a length of 137 m and breadth of 24 m. DNV (2005) estimated the area of influence of the propeller water as being 25.5 m wide at full engine power (17200 hp = 12642 kW). Color Viking's departure was estimated to cause resuspension of about 2800 kg of sediment. The calculations are based on a measured elevated particle concentration in the water over an area with a diameter of 150 m, which corresponds to an area of 17663 m² (DNV 2005).

If we assume that 2800 kg is resuspended from 17663 m², this corresponds to mean resuspension of the upper 0.3 mm of the seabed. This is based on 1 mm/m² corresponding to 0.57 kg/m²⁵

Resuspension of 0.3 mm or 0.17 kg/m² within the area of influence results in a particle concentration in the whole water mass of about 20 mg/l, which is consistent with the measurements of DNV (2005).

Sandefjord harbour is not more than 8 m deep; i.e., the clearance between propeller and seabed is about 2 m. According to the calculation tool ([section 3.3](#)), the flow velocities in the shallow areas outside the docks will be well over the critical velocity for resuspension of cohesive sediments. According to [Table IX.3](#), erosion will resuspend about the upper 2 cm of sediments of this kind, not merely the upper 0.3 mm. This means either that the area of influence is strongly overestimated (cloud 150 m in diameter does not represent the area that was resuspended) or that the amount of resuspended material has been underestimated (more was resuspended than was measured in the water masses) or a combination of the two.

3.5.2 Calculations based on model observations

We attempt to estimate the area of influence of Color Viking's propeller. DNV (2005) estimates the breadth of the centre jet to be 25.5 m 100 m behind Color Viking. The ship has two propellers aft, so the two jets will overlap. We assume that the area of influence of the propeller jet will correspond to the breadth of the ship. If we also assume that the ship affects the seabed from 500 m away from the dock, we get an area of influence of 12500 m² (*Figure IX.3*)

⁵ These figures are taken from a ¹³⁷Cs-dated core from the docks of Jotun Fabrikker further out in the fjord. This was a good dating, where the maximum concentration of ¹³⁷Cs, corresponding to the emissions from Chernobyl in 1987, was found 8.5 cm down in the sediment, which means an addition of 4 mm/year. It was furthermore calculated that annual sediment growth was 2.3 ± 1 kg/m²/year (Bakke and Helland, under preparation).

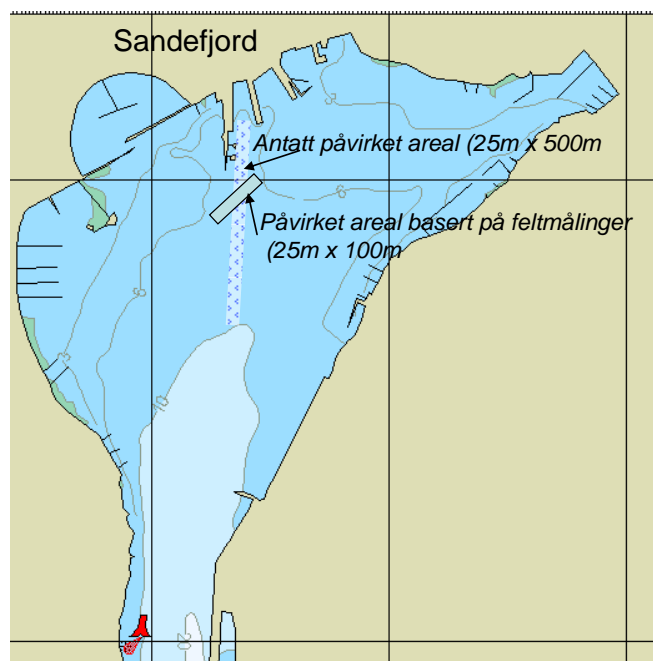


Figure IX.3. Sandefjord dock with shaded area for assumed ship-induced resuspension; see text for further information. Assumed affected area (25 m x 500 m) and affected area according to field measurements (25 m x 100 m)

We also know that the velocity of the central jet has a normal distribution, and that there is about 2 m of clearance between propeller and seabed. From [Table IX.3](#) we assume that the centre of the jet erodes down to a sediment depth of 2 cm, and that the external edge of the jet erodes down to 0.3 cm ([Figure IX.4](#)). This gives an average erosion depth of 0.85 cm in the area of influence. For each metre the ship moves, about 0.2 m³ of sediment is resuspended (25 m² * 0.0085 m). If we assume wet sediment has a self-weight of about 1.3 g/cm³ when the water content is about 65%, this gives 260 kg wet and 90 kg dry sediment for each metre the ship moves. We assumed the area of influence to be 500 m long. Along this stretch, about 130 tonnes of wet and 45 tonnes of dry sediment are resuspended. This is far more than the 2800 kg estimated by DNV (2005).

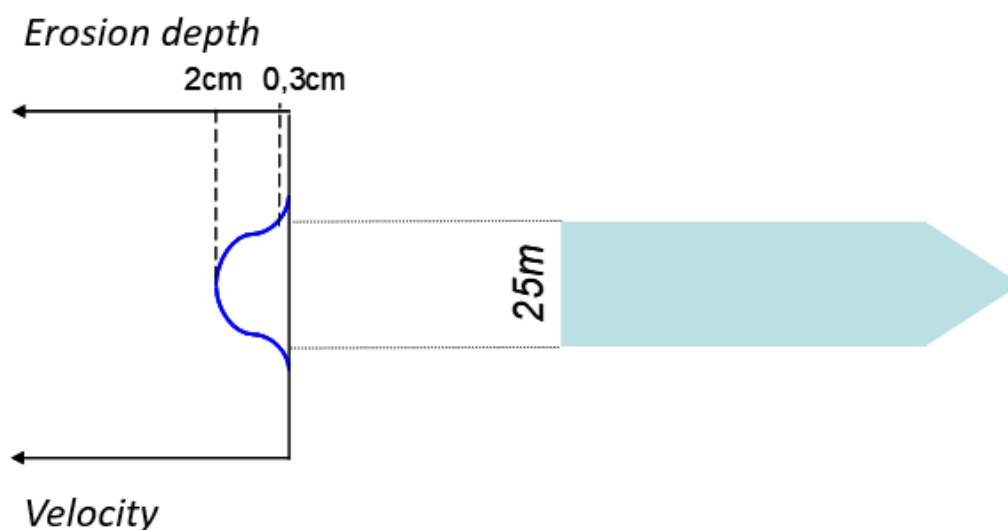


Figure IX.4. Schematic diagram of the velocity and erosion curve in the propeller jet behind a ship

3.5.3 Comparison of calculation methods

Depending on how coarse-grained the bottom sediments are, much of the coarsest fraction, sand and gravel will deposit rapidly after resuspension. The measurements carried out in the various harbours have been taken using both fixed installations and hand-held equipment. In both cases, the measurements were taken a distance away from the resuspension area and a little after resuspension had taken place. We can therefore assume that the measurements reflect the silt and clay fractions of the sediments, which remain suspended in the water. We do not have concrete information about the grain distribution in the sediments outside the Sandefjord docks. If we assume the same distribution as outside the Jotun Fabrikker docks, the sediments consist of about 10% clay and 10% silt. If DNV picked up both the silt and the clay fractions in 2005, a total of close to 14 tonnes of particles were resuspended, and if only the clay fraction remained suspended in the water, close to 28 tonnes were resuspended in all. The quantity of suspended particles based on field measurements is thus of the same order of magnitude as in the calculations based on the risk tool.

The comparison indicates that it is important to take account of the grain size distribution of the sediments when calculating resuspension. In the context of spreading, it is the hazardous substances associated with the silt and clay fraction that are of importance. This fraction is recorded in field measurements as the most important, since the coarser fractions deposit more rapidly on the seabed.

4 Calculation tools and default values

Based on [Table IX.3](#), the relationship between clearance and total resuspended sediment quantity can be described by means of the equation

$$F_{\text{ero}} = 24.78 \cdot D_{\text{sea}}^{-1.24}$$

Where F_{ero} is resuspended sediment (kg dry weight/m²) and D_{sea} is clearance depth (m), i.e. the distance between propeller and seabed. If the clearance is known and it is assumed that erosion acts laterally to an extent corresponding to the width breadth of the vessel, the total weight of the material that becomes suspended for each metre of the boat's movement can be calculated. If the distance moved by the vessel and fraction of fine sediment (silt and clay) along this route are known, it is possible to calculate the total quantity of fine material that is resuspended in a docking process. It may be assumed that this material will remain in the water for a while before redepositing, and that it can disperse to other areas. This is of the greatest importance in the context of risk.

Since there is a wide range of vessel sizes, clearances and fine material fractions, we propose a simple classification for use in a risk assessment.

The fairway area from 20 m depth and in to a harbour is classified into three depth ranges: 5 to 10 m (average 7.5 m), >10 to 15 m (average 12.5 m) and >15 to 20 m (average 17.5 m). The distance moved by the ship and typical fines fraction (< 63 µm) in the sediments is estimated for each category.

We further propose that vessel size should be divided into three categories (Table IX.4)

Table IX.4 Vessel size classification

Vessel category	Hull length (m)	Hull breadth (m)	Propeller depth (m)
Large	>150	> 20	6
Medium	50-150	10	4.5
Small	< 50	< 5	3

The quantity of resuspended fine material per docking or departure can be calculated for each of these vessel categories, combined or separately, for the three depth categories.

$$m_{sed} = \sum_{i=1}^3 (24.78 \cdot (D_i - Pd)^{-1.24}) \cdot Br \cdot f_{si} \cdot T_{ri}$$

m_{sed} : kg fine material resuspended per docking for each vessel category (kg dry weight)

D_i : average depth in category i (m)

Pd : propeller depth (m)

Br : hull breadth (m)

f_{si} : fraction < 63 µm in depth category i

T_{ri} : distance moved by ship in depth category i (m)

Total resuspension per day or year can then be calculated from the traffic pattern. This will be a relevant calculation tool for use in Level 3 of the risk assessment.

We have used the calculation tool to arrive at default values for some typical shipping traffic situations that are included in Risk Guidelines Level 2 [Box 6](#)). Given an assumed fairway stretch of 40 m for each of the depth ranges above, and a sediment fines (silt/clay) content of 60%, 40% and 20% from the deepest to the shallowest range, the total calculated quantity of resuspended fines per docking will be 3000 kg, 900 kg and 350 kg for large, medium and small vessels.

We have also set up 3 categories of harbours: large harbours (ferries, cruise ships, tugboats etc.), medium-sized harbours (cargo boats, supply boats etc.) and small harbours (small boat harbours). Assuming that these harbours will be visited by boats of various sizes, we have rounded the total quantity of resuspended fine fraction per docking as described above to 2000 kg, 1000 kg and 150 kg for the three types of harbour ([Table IX.5](#)). This is for harbours with mainly silt and clay sediments. For harbours with mainly sandy sediments we have set the resuspended fine fraction at 10% of these figures, and for harbours with gravel and stones at 1% of them. The default values in the risk guidelines will then be (kg dry weight resuspended fine fraction per docking):

Table IX.5 Default values for resuspended fine fraction per docking in three types of harbours

Sediment type	Harbour category		
	Large harbour ¹⁾	Industrial harbour ²⁾	Small boat harbour
Silt and clay	2000	1000	150
Sand	200	100	15
Gravel and stones	20	10	1

1) Ferries, cruise ships, tugboats, etc.

2) Cargo boats, supply boats etc.

5. Conclusions and recommendations

A review of available literature and data shows that ship movements can generate flow velocities at the seabed which far exceed the erosion rate of cohesive sediments. The magnitude of the flow velocity along the seabed depends strongly on the clearance between propeller and seabed. Even boats regarded as small in a shipping context may generate a high flow velocity along the seabed if the clearance is limited.

There is a clear need to procure information about the erosion pattern of water jet motors. Little is known about this at present. The water jet effect should be studied before the next revision of the risk guidelines. For the present, it is recommended that in the context of risk, passenger boats and car ferries with water jets be classified as large, propeller-driven boats and erosion calculated accordingly.

The literature and the authors' own observations establish as probable that erosion may take place several centimetres down into the sediments at high flow velocities. However, there is a need for *in situ* empirical data to enable more accurate estimation of the erosion depth.

Erosion as a consequence of shipping has been best examined in Sandefjord harbour (DNV 2005). The calculations show that it is probable that the area of influence of actual propeller erosion is far less extensive than the area of influence of the particle cloud that is observed in the water. Estimates indicate that the area of influence is about twice the length of the ship (L) x the breadth (B): (2LxB).

The calculations from Sandefjord also show that turbidity measurements that up to now have been conducted in various harbours in connection with shipping probably represent the fine fraction that remains suspended in the water a while after resuspension. The silt/clay fraction may typically constitute about 20% in harbour areas. We can moreover assume a sedimentation rate of 0.5 cm/s for the medium-sized coarse fraction (coarse silt / fine sand - 75 µm). Assuming a depth of 8 m and limited water movement, particles of this size will deposit during approximately 30 minutes and hence locally.

The comparison of the model observations with the calculations from Sandefjord harbour show reasonable consistency with respect to estimating the quantity of fine fraction sediment that is resuspended in connection with a docking, and the model observations are used to

derive formulae for simplified calculation of resuspension because of local conditions and type of traffic. This will be a relevant Level 3 assessment. The formulae are then used again to derive recommended default values for three types of harbour: (large, medium, small boat) and three sediment types in these harbours (silt and clay, sand, gravel and stones) for use in Level 2 ([Box 6](#)).

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Appendix X – Theoretical basis for risk assessment methodology

The risk assessment system is based on an equilibrium distribution between sediment, water and organism (Fig. X.1). This distribution is directly dependent on the chemical properties of the substances, including the partition coefficients used.

The guidelines include the following data on the substances in question:

- Molar weight
- Solubility
- Octanol/water partition coefficient, K_{ow}
- Partition coefficient corrected for organic content of the sediment, K_{oc}
- Water/sediment partition coefficient, $K_{d, sed}$, based on 1% TOC in the sediment
- Molecular diffusion rate, $D_{molecular}$
- Bioconcentration factor, BCF_{fish}
- Biota sediment accumulation factors, BSAF

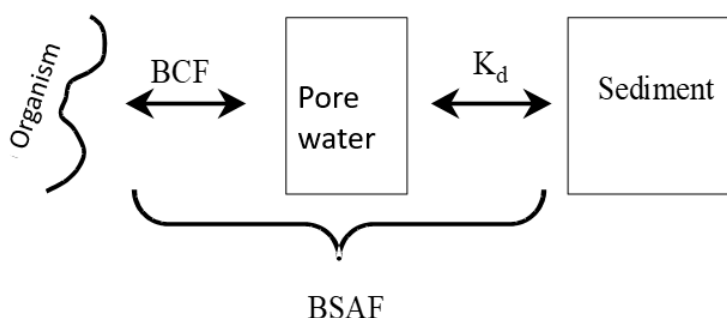


Figure X.1. Schematic model of the distribution of hazardous substances between sediment, porewater and organisms.

In the following sections, a more detailed description is given of the most important parameters.

Partition coefficients, K_d

Partition coefficients, K_d , describe the distribution between the concentration of a substance in sediment (C_{sed}) and porewater (C_{pw}) at equilibrium;

$$K_d = C_{sed} / C_{pw} \quad (X.1)$$

It has been shown that for organic compounds, K_d is proportional to the content of organic matter in the sediment:

$$K_d = K_{oc} \cdot f_{oc} \quad (X.2)$$

where K_{oc} is the partition coefficient corrected for the organic matter of the sediment and f_{oc} is the fraction of organic carbon in the sediment (1% TOC gives $f_{oc} = 0.01$). An empirical relationship is derived between K_{oc} and K_{ow} (DiToro, 1985) expressed by:

$$\text{Log}_{10}K_{oc} = 0.00028 + 0.983\text{log}_{10} K_{ow} \quad (\text{X.3})$$

This equation can be used if there are no direct measurements of K_{oc} .

Bioconcentration factor, BCF_{fish}

Both BCF and K_d are functions of the solubility of the substance in water. This solubility can be expressed as the partition coefficient between water and octanol, K_{ow} , which has been found for most hazardous substances. A high K_{ow} means that the substance is hydrophobic, with limited solubility in water and tends to accumulate in the fatty tissue of organisms. BCF is related to the water/octanol partition coefficient K_{ow} as follows (EU TGD, 2003):

$$\text{log BCF} = 0.85 \text{ log } K_{ow} - 0.70 \quad (\text{X.4})$$

However, this relationship only applies up to a certain value of $\text{log } K_{ow}$ ($= 6$) where BCF reaches an upper limit and then falls again, X.2.

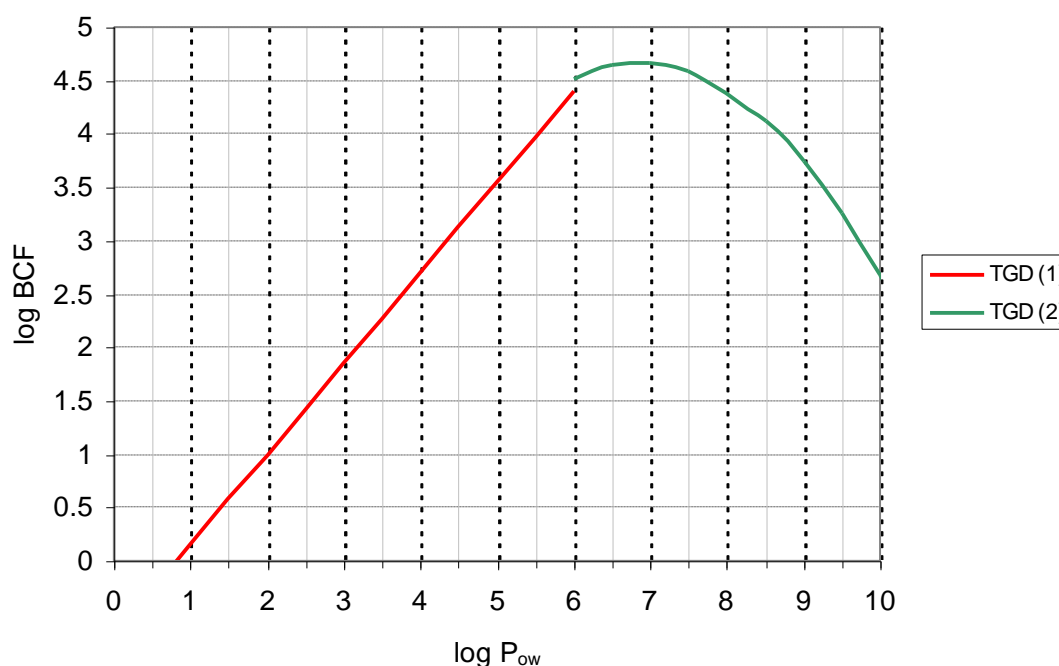


Figure X.2. The relationship between hydrophobicity ($\text{log } K_{ow}$ shown here as $\text{log } P_{ow}$) and the bioconcentration factor BCF (EU TGD, 2003).

Diffusion

The following mechanisms are important for transport of substances from sediment without erosion or sedimentation:

- Molecular diffusion (pure physical diffusion)
- Biodiffusion (enhanced diffusion due to the activity of benthic fauna)

According to Næs et al. (2001), diffusion rates with different mechanisms can be summed up by the figures in Table X.1. The most important factor is that bioturbation may increase the diffusion rate by several orders of magnitude compared with pure molecular diffusion. Diffusion calculations based on leaching tests and measurements of porewater concentration using different transport models show that molecular diffusion may be of great significance for the diffusion flux. Diffusion is entered as one of the three mechanisms for spreading of hazardous substances from sediments. See [section 4.2.1](#) for calculation of this transport.

Table X.1. Diffusion rates for different mechanisms (Næs et al., 2001).

Mechanism	Diffusivity (cm ² /s)
Molecular diffusion without retardation (purely hypothetical)	10 ⁻⁶ - 10 ⁻⁵
Without retardation and corrected for tortuosity	10 ⁻⁶
Molecular diffusion with retardation (realistic)	10 ⁻¹³ - 10 ⁻⁸
Biodiffusion	10 ⁻⁸ - 10 ⁻⁶

Bioavailability

There are several empirical values for partition coefficients between water, sediment and biota in the literature. Common to empirically derived bioconcentration factors (BCF) and biota sediment accumulation factors (BSAF), is that they are largely dependent on the conditions under which the experiments were conducted. Calculation of environmental flux from a sediment to organisms based on partition coefficients from the literature must therefore be conservative.

Using observations of equilibrium between sediment and organisms is only recommended for estimating the tissue level in organisms that live in sediment (infauna). The estimates for other organisms should be based on:

- Estimate of flux of hazardous substance between sediment and overlying water
- Observations of dilution of hazardous substances in the water body
- BCF values (for the transition between estimated water concentration and tissue concentration).

BSAF values in the literature were collated by the Norwegian Defence Research Establishment (FFI 2001), but the values vary rather unsystematically depending on organism, type and structure of hazardous substance and concentration, and across different experiments with the same species. This is because factors that have not been described influence the BSAF.

For organic and other non-polar hazardous substances, BSAF is inversely proportional to the organic content of the sediment (expressed as a fraction or percentage by weight). Several examples show that normalisation of the concentration of hazardous substance in the sediment against TOC and normalisation of the concentration in the organisms in relation to fat content reduce the variability of BSAF. This is considered in the guidelines.

The above variability means that when the BSAF values in the literature are used, BSAF options with a conservative inclination should always be chosen in the context of risk.

More detailed assessments with respect to BSAF are given below for organic hazardous substances and metals.

Organic hazardous substances

The accumulation factor from sediment to organisms, BSAF, is described by the partition coefficient, K_d , from sediment to water, and by the bioconcentration factor BCF, from water to organisms, as follows.

$$BSAF = BCF/K_d \quad (X.5)$$

We then get:

$$\log_{10}BSAF = \log_{10}BCF - \log_{10}K_d = (a-0.983)\log_{10}K_{ow} - \log_{10}f_{oc} + (b-0.00028) \quad (X.6)$$

In practice coefficient a is -1 , and thus:

$$BSAF \sim 10^b/f_{oc} \quad (X.7)$$

i.e. in principle, BSAF for an organic compound is independent of the substance's hydrophobicity (expressed as K_{ow}) and inversely proportional to the sediment's organic content. This is a major simplification of reality nonetheless, because the partition coefficient between sediment and water is often much larger in contaminated sediments than is indicated by theoretical calculations based on K_{ow} .

Metals

The form of metals is of fundamental importance to biological uptake and toxicity. Metals occurring as pure metal, in deposits, in the crystalline structure of minerals and in clay particles or minerals are not regarded as being bioavailable (Waldichuk, 1985). The most bioavailable forms are metals in ionic form and bound to carbonate. Metals bound to sulphides and adsorbed onto organic matter may be weakly bioavailable. It is also likely that much of the metals in porewater occur as complexes with organic material. Thus, they have reduced bioavailability compared with free metal ions. This means that BSAF factors from the literature are only valid if the metals are in the same form as in the experiments upon which the BSAF values are based. It is very difficult to determine the form of metals in a sediment sufficiently reliably to be able to calculate its bioavailability. Routine analysis of sediments yields only concentration, not form. For K_d , too, it is recommended that measured values be used rather than values based on the literature in site-specific risk analysis of metals in sediment (Lijzen et al., 2001).

There is much to indicate that metals in sediment generally have low bioavailability. The conclusion of an extensive review of the risk of biological uptake of hazardous substances from disposal sites for drilling waste on the bottom of the North Sea was that calculated and probable leaching of metals was too small to have any effects (Hartley et al., 2003). There are examples of 70-99% of metals present in clean sediments not being bioavailable. With few exceptions, the current advice on eating fish and shellfish from Norwegian fjords is also based on organic hazardous substances, not metals. The most direct method of obtaining a measure

of the bioavailability of metals in a sediment is therefore to conduct bioaccumulation tests on the sediments in question. An impression of whether the metals are bound as sulphides can also be obtained by judging/measuring whether the sediments are anoxic.

Direct analysis of the concentrations of hazardous substances in porewater probably provides the best measurement of the bioavailable fraction.

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