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Framework for the Environmental Impact Factor for drilling discharges

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Framework for the Environmental Impact Factor for drilling discharges

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Summary

In order to achieve the oil industries' "zero harmful discharge" goal and selection of cost-benefit based solutions, a methodology was developed in a joint industry project together with external consultants to calculate an Environmental Impact Factor for produced water discharges (EIF_{PW}), based on environmental risk and hazard identification. As a follow up of the EIF for produced water discharges the ERMS project was started in order to develop an EIF for drilling discharges (EIF_{DD}).

The EIF_{PW} addresses environmental risk in the water column, based on toxicity as the only stressor. The EIF for drilling discharges also takes into account risk in the sediment compartment. For the development of a concept for the EIF_{DD}, two approaches for risk calculation in the sediment compartment have been evaluated. The first approach is based on the risk assessment guidelines proposed by the EU-Technical Guidance Document on risk assessment (EU-TGD; EC, 2003). This approach is also applied for the EIF_{PW}. The second approach was focussing on the application of data from sediment monitoring surveys. Following this latter approach, statistical correlations between discharge (volume and characteristics) and observed exposure and effect levels should be developed. After evaluation of the two approaches it was decided that the concept for the EIF_{DD} should be based EU-TGD approach (PEC:PNEC) with some specific extensions. And where possible the available data form the monitoring studies should be used to derive thresholds and for validation purposes.

Hazard identification has indicated several important stressors related to drilling discharges; two stressors in the water column (toxic components and suspended particles) and for four stressors in the sediment (toxic compounds, change in grain size, oxygen depletion and burial). For these stressors estimates of exposures (PECs) and thresholds (PNECs) are made. A comparison of PEC and PNEC results in a risk characterization ratio (RCR). Below a RCR value of one, risk of the specific stressor is generally considered acceptable.

In order to combine and compare the contribution of different stressors to the overall risk, Species Sensitivity Distributions (SSDs) are applied to calculate a risk probability represented by the PAF (Potentially Affected Fraction). PAF values for the separated stressors are combined into a joint risk probability also called the multi stressor PAF (msPAF). The spatial extent (volume or area) where the msPAF exceeds 5% is taken as a basic value for the EIF in the water column as well as in the sediment.

The available data from sediment monitoring studies (MOD database) is used for validation purposes. Alternative thresholds for different stressors in the sediment are derived from this data and compared to the PNEC values based on literature

and laboratory data. Also the EIF for the sediment (area with a risk value beyond the acceptable risk levels) can be compared to biological effects observed during field studies.

The presented concept for the EIF_{DD} presented in this report is elaborated in different studies all carried out under the framework of the ERMS project:

- EIF_{DD} model development (Rye *et al*., 2006a)
- Sediment processes and characteristics (Schaanning & Bakke, 2005; Trannum & Brakstad, 2005 and Rye *et al*., 2006b)
- Toxicity in water and sediment (Frost *et al*., 2006)
- Risk assessment for weighting materials (Smit *et al*., 2006a)
- Non-toxic stressors in the sediment (Smit *et al*., 2006b)
- Validation of risk assessment (Bjørgesaeter, 2005; Brakstad & Trannum 2005 and Grung *et al*., 2005a)
- Management options and the EIF_{DD} (Rye & Ditlevsen, 2005)

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1. Introduction

1.1 Background on the Environmental Impact Factor

In order to achieve the oil industry's "zero harmful discharge" goal and selection of cost-benefit based solutions, a methodology was developed to calculate an Environmental Impact Factor for produced water discharges (EIF_{PW}), as an indicator of environmental risk. The EIF_{PW} is developed as a management tool applied by the oil industry. Its calculation is based on internationally agreed procedures for hazard and risk assessment, as defined by the European Union (EU). The EIF_{PW} helps to identify the potentially most environmental harmful substances in the discharged produced water and to quantify the environmental benefit of different mitigating measures (prognosis).

In this context it is based on the PEC_PNEC ratio approach, also termed risk characterization ratio (RCR). The PEC_PNEC ratio approach compares the Predicted Environmental Concentration (PEC) of a pollutant with the predicted environmental tolerance level or the concentration giving no adverse effect (PNEC).

Different methods exist to define the level of the PNEC. The selection of method to be applied is quite arbitrary but often depends on availability of data. For the EIF_{PW}, PNEC values are defined using assessment factors. These principles are described in the EU Technical Guidance Document (TGD) (EC, 1996 and 2003). In order to calculate the contribution of the different stressors to the overall risk Species sensitivity distributions SSDs are applied. Smit *et al.* (2005a) provides a description of the overall methodology. A description of the development of the EIF for produced water can be found in Smit *et al.*, (2003). Johnsen *et al.*, (2000) describe the use of the EIF for produced water in produced water management.

1.2 Aim of Environmental Impact Factor for Drilling Discharges

The ERMS (Environmental Risk Management System) project has been defined to develop a prognosis tool to estimate ecological risks arising from intended drilling discharges and integrate it with the EIF for produced water. The endpoint of the prognosis tool will be an EIF for drilling discharges (EIF_{DD}) comparable to the EIF_{PW}. The EIF_{DD}, together with the EIF_{PW} should form an integrated model system which enables the oil companies to perform risk calculations for different discharge scenarios during different operations (production and drilling). The development of the EIF_{DD} is built on the same setup as used for the EIF_{PW} (Johnsen *et al.*, 2000). As drilling discharges are likely to pose a risk to biota in both water and sediment, the environmental compartment seabed needed to be included in the model framework of DREAM (Dose-related Risk and Effect Assessment Model) (the EIF_{PW} focuses on the water column only).

A number of specific requirements apply to the development of the EIF_{DD}. These include:

- A variety of stressors, including toxic substances, high particle concentrations and changes in sediment quality and structure must be evaluated;
- Both the water column as the sediment compartment must be taken into account.
- The aspect of time needs to be included; relative short-term stress on the water column vs long-term stress on the sediment.

The main steps taken in the conceptual development of the EIF for drilling discharges were:

- Identification of main stressors related to drilling discharges;
- Development of calculation rules to simulate the exposure of the selected stressors;
- Investigation of effects of the selected stressors and collect the best available data (laboratory and field studies) by literature review: NB. There is a lack of a formal effect evaluation framework for disturbances other than toxicity, since no guidance is available from the TGD or other relevant documents;
- Development risk calculation rules and define the EIF for drilling: NB. there is a lack of a general methodology on how to combine the risks from the different (toxic and non-toxic) stressors into a single risk figure;
- Implementation of data and procedures in a simulation tool.

Substantial data is available on the integrative effects of drilling discharges on the sediment compartment. Many monitoring studies have been carried out in the past which include chemical analysis and biological effect monitoring. This information contains valuable data which could be used in relating stressors to effects. Because effect monitoring ‘measures’ the overall effect of the whole activity this includes effects of other stressors than toxicity as well.

1.3 ERMS concept development process

As a first step in the development of the concept a meeting was organised at the Schiphol A4 Hotel, the Netherlands, on December 2, 2002. This meeting was organized to present and discuss the different approaches for concept development of the EIF for drilling discharges, in order to have a pre-agreed basis for the setup to be discussed at the ERMS workshop on January 2003. From this meeting two possible bases for the EIF for drilling discharges were identified: The PEC_PNEC ratio and the Community Disturbance Index (CDI: Massart, 1997) were selected as potential good indicators to express the EIF_{DD}. At the first ERMS workshop of January 2003 in Trondheim the two different bases were discussed. At this workshop six activities were defined to evaluate the two principles further. As literature data was needed in this evaluation the tasks of concept development and literature research were merged into one development task.

During the 2nd ERMS workshop of June 2003 (Texel) the defined activities were presented and discussed. It was concluded that the EIF should be based on generic risk assessment approaches in line with European guidelines for risk assessment (i.e. PEC_PNEC ratio).

Two main reasons were the basis for this decision: A precondition for the development of the EIF_{DD} was that more or less the same principles should be applied as for the EIF_{PW}. which is completely based on the PEC_PNEC ratio. And secondly the CDI can only be applied to the sediment compartment because the monitoring data is restricted to this compartment while the water compartment is also impacted by drilling discharges.

In order to use the large amount of valuable data from past seabed monitoring studies (on which the CDI approach would be based) it was suggested to define validation activities using this data. The aim of this program was to place the sediment related results from the concept development and literature study dedicated to the PEC/NEC approach in the light of field relevance. Finally, a stepwise risk assessment approach for the EIF_{DD} was defined:

1. The first step was the determination of the EIF_{DD} according to the risk assessment guidelines described in the TGD for risk assessment (PEC_PNEC approach). Threshold values (PNECs) were defined based on literature and, if needed, other data sources, for different stressors both for the water column and the sediment compartment.

2. The second step was a validation step which was only worked out for the sediment related part of the EIF_{DD}. A second set of sediment thresholds was defined based on monitoring data (field derived threshold effect levels: F-TELs) for comparison with PNECs based on toxicity literature data for metals. Calculated risk areas (following the procedures in step 1) will be compared to the extent of observed impacted areas described by indicators representing biological disturbance (e.g. CDI).

This document presents the approach for the development of the EIF for drilling discharges, thereby defining a framework for the evaluation and validation of potential impacts and risks from different disturbances in different compartments (water column and sediment).

2. Risk assessment and the EIF_{DD}

A general framework for the prediction and evaluation of risks from chemical pollution is provided in Figure 1 (Smit *et al.*, 2005b). The triangles in the middle illustrate the stress of toxicants on the biological system. The level of stress can be evaluated by measuring (diagnosis) or by prospective modelling (prognosis). The left branch of the figure represents the diagnosis tools applied in field monitoring. Samples extracted from the field can be analyzed for contamination, tested in bioassays or the health status of the field biota is directly assessed by field monitoring. The right branch of the figure is based on the framework for generic risk assessment. This procedure starts with the identification of main hazards for which exposure and effect levels are assessed. Comparison of both levels results in a risk estimate for the compartment under consideration. In order to assess the impact at the ecosystem level both procedures are frequently applied.

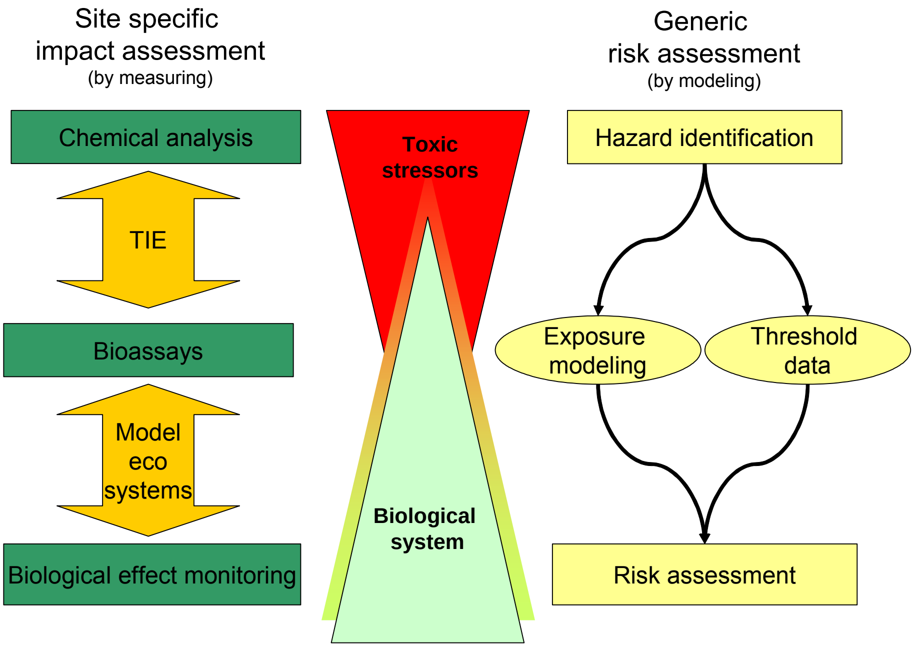

Figure 1 General framework to predict and evaluate the risks of chemical substances at the ecosystem level

The aim of this project is to develop a prognosis tool. Risks on the environment need to be predicted and effects of proposed mitigating measures on the calculated risk need to be evaluated. In the EIF_{PW} the only hazard considered is toxicity. For the evaluation of this stressor the internationally agreed procedures for chemical risk assessment are applied (EC, 2003). As mentioned in the introduction for drilling discharges toxicity is not the only and main hazard of concern. However, for reasons of consistency it is preferred that the same strategy as applied in the EIF_{PW} is used in the development of the EIF_{DD}. Therefore, in this project the general risk assessment rules developed to evaluate toxic stress will be applied on other (non-toxic) stressors.

2.1 Principles of risk assessment

Similar to the international agreed principles for risk assessment (EC, 2003) the following steps are identified for the development of the EIF_{DD}:

1. Hazard identification
2. Exposure assessment
3. Effect assessment
4. Risk assessment
5. Validation

Figure 2 provides an overview of the different activities within the framework of the EIF_{DD} calculation.

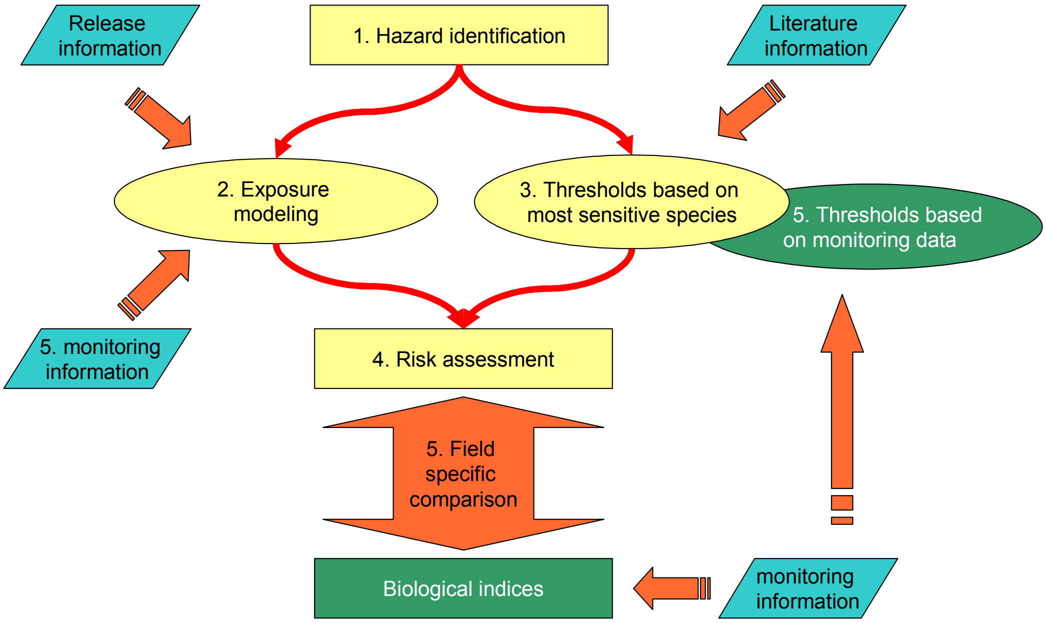

Figure 2 *Framework for the EIF_{DD}. The numbers indicate the different steps in the risk assessment process.*

Hazard identification

To start the risk assessment procedure, a selection of stressors of specific concern must be made. For these stressors the data collection and risk assessment procedures are elaborated. In chapter 3 the main stressors related to drilling discharges are identified.

Exposure assessment

Traditionally, ecological risk assessment is based on the comparison of the exposure of (a part of) the ecosystem to a stressor with the sensitivity of (the same part of) the ecosystem for this stressor (through this specific exposure-route) (Suter, 1993). For the selected hazards it is the challenge to have an adequate estimation of the environmental exposure level. The exposure is represented by the PEC (Predicted Environmental Concentration), and can be obtained by actual field measurements (monitoring data) or by estimations using environmental fate models. Chapter 4 describes how the exposure levels of identified stressors are defined and expressed.

Effect assessment

As the exposure levels of the identified stressors need to be compared with the sensitivity of the ecosystem, estimates of the sensitivity need to be defined for all stressors. The PNEC (Predicted No Effect Concentration), for toxicity usually derived from standardised laboratory tests, represents the sensitivity of the ecosystem. The EU-TGD (EC, 2003) describes two methods to derive a threshold level. Depending on the availability of data either the assessment factor approach or statistical extrapolation methods are applied (Smit *et al.*, 2005a). Chapter 5 describes for each stressor how the threshold level and the variation in species sensitivity are estimated.

Risk assessment

The ratio of exposure and sensitivity (for toxicity the comparison of PEC and PNEC often referred to as the PEC_PNEC ratio or the RCR - Risk Characterisation Ratio) gives an indication of the likelihood of adverse effects to occur as a result of the anticipated exposure level. There has been some criticism of the use of RCRs in risk assessment on the grounds that: (a) they do not directly yield probabilities of adverse effect; (b) they are ecologically naive; (c) they involve so many uncertainties that they cannot be used as a sound basis for risk management. Calow & Forbes (2002) addressed most of these issues with respect to the EIF approach applied to produced water and concluded that RCRs are a good basis for deriving EIFs.

Analogous to the comparison of risks from different toxicants in the EIF_{PW} the risks for the different stressors in the EIF_{DD} will also be aggregated to one integrated risk value. Species Sensitivity Distributions (SSDs) can be applied to calculate risk probabilities, which facilitate the weighting of different risk values (Posthuma *et al.*, 2002; Smit *et al.*, 2005b). The SSD describes the variation in sensitivity of biota to stressors. Potentially Affected Fractions (PAFs) are calculated for the different stressors and the PAF values are combined in a multi stressor PAF value (msPAF). Finally the water volume or the sediment surface where msPAF values exceed the 5% level is taken as a basis for the EIF calculation. This is the same procedure as defined for the calculation of the EIF_{PW}.

There are limitations to the use of the PAF in risk assessment when one considers the lack of ecosystem dynamics, such as food web relationships, incorporated into this concept, with the major focus at the species level of organisation. In other words: "How representative is the PAF for the actual risk in the field?" In this project the PAF represents the ecological risk. However, the challenge that still remains for ecologist and ecotoxicologists is the definition of what effects on the ecosystem are acceptable or unacceptable in relation to the most sensitive endpoints on the species level (Calow & Forbes, 2003). Chapter 6 describes the procedures for risk assessment and EIF calculation is detail.

Validation

The reason for establishing a validation program using field monitoring data is to bridge the gap between risk assessment and monitoring (prognosis vs. diagnosis). The presence of a risk does not necessarily imply occurrence of effects in the environment. However, reducing risks does reduce the probability of the occurrence of effects.

The availability of monitoring data for the sediment compartment facilitates the process of validation at two different levels:

1. Threshold values derived from toxicity literature data can be compared to threshold values derived from this field monitoring data (so called field derived threshold effect levels; F-TEL), and in parallel;

2. Calculated risk areas in the sediment can be compared to the surface area where abundance of biota is significantly altered compared to reference stations.

With the defined validation activities the field relevance of the risk assessment is checked. However, the field derived threshold effect levels (F-TELs) cover risk from multiple stressors and cannot be directly compared to the literature based PNEC values (for metals) based on toxicity as the only stressor. It must be kept in mind that the validation activities only relate to the stress to the sediment compartment. Recommendations for further validation and development are taken up in Chapter 7 of this report.

2.2 EIF-drilling discharges: EIF-water and EIF-sediment

As already mentioned, drilling discharges will influence two compartments; water column and sediment. The potential impacts on the two compartments have different time frames. During and shortly after discharge, exposure levels are present in the water column. As long as these exposure levels are present in the water column risks on adverse effects could be present. The duration of water column exposure is in the order of minutes to several days. During and long after the drilling discharges, exposure levels will be significant at the sediment surface. The duration of possible exposure through the sediment is much longer than through the water column effects (order of months and years). The water column risks are generally referred to as acute, while the sediment risks are more chronic. Therefore the risks in both time frames and compartments are expressed in their own EIF for the water column and the sediment. The combination of both EIFs will form the EIF_{DD} (Figure 3).

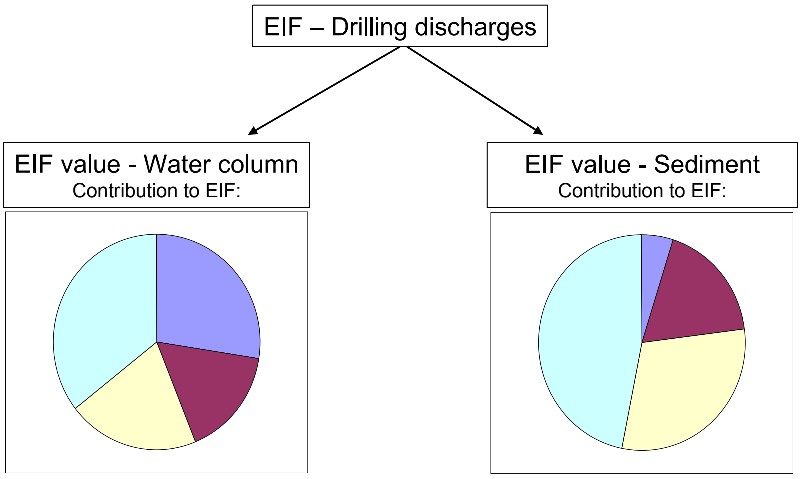

Figure 3 *The EIF_{DD} consists out of two EIFs; One for the water column and one for the sediment. Contribution to both EIFs can be pointed to the different stressors as represented by the pie charts.*

3. Definition of main stressors

3.1 Characterization of drilling discharges

The EIF for drilling discharges should be an integrated measure of the overall probability of damage caused by different stressors. This implies that risks on different kind of effects need to be integrated. But like the EIF for produced water this EIF must also be able to differentiate between the various contributors to environmental risk in terms of percentage contributions from various chemical compounds in the sediment (graphical presentation in a pie chart).

Like the EIF for produced water discharges the EIF for drilling discharges will eventually be used for environmental management. This means that the effect of environmental management actions which can be taken during drilling (mitigating measures) should be displayed by a reduction of the EIF_{DD}. In contradiction to the EIF_{PW} for releases during the exploitation phase, which can be reduced by removing toxic substances from the produced water flow, there are many more options to reduce the EIF_{DD}.

The following management options were identified, but may need to be supplemented:

- Use of Water Based Mud (WBM), Synthetic Based Mud (SBM) or (Oil Based Mud (OBM);
- Composition of the drilling mud;
- Selection of added chemicals;
- Moment of discharge;
- Batchwise or continuous discharge;
- Surface or subsurface discharge;
- Depth and angle of the discharge pipe.

When defining the EIF_{DD} it is important to indicate the main disturbances of the discharges during drilling. For the EIF for produced water the main disturbance is assumed to be the toxicity of produced water components. The effects of temperature, salinity and suspended matter have been neglected in this EIF. But for drilling discharges toxicity cannot be assumed as the main stressor because discharge of OBM is not longer allowed and most chemicals in the drilling mud are from the PLONOR list (added chemicals Posing Low Or No Risk: OSPAR Agreement 2004-10), i.e. they are considered to be non-harmful. Figure 4 gives a mechanistic presentation of the fate of drilling discharges (taken from Rye *et al.*, 2006a).

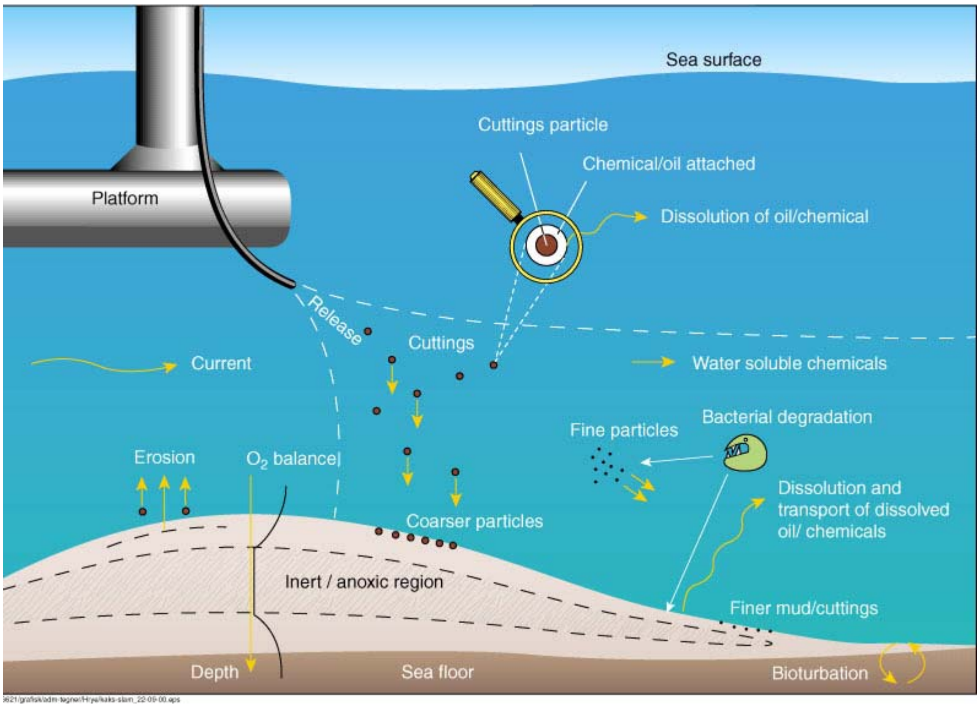

Figure 4 Overview of short and long term disturbances of drilling discharges (taken from Rye et al., 2006a).

3.2 Hazard identification

The hazard identification was the first step in the development of the EIF_{DD}. The main stressors of concern for potential impacts to the environment were identified. Based on literature information, monitoring studies as well as on detailed knowledge of the drilling procedures, the following stressors to the water column and sediments of the marine environment have been indicated. These should be taken into account in the development of the EIF_{DD}:

- Water column;
  - Toxicity of chemical substances;
  - Physical effects of suspended particles;
- Sediment;
  - Toxicity of chemical substances;
  - Burial of organisms and change in sediment structure;
  - Oxygen depletion and consequential increase in sulphide concentration.

To represent these stressors the parameters mentioned in Table 1 have been selected.

Table 1 Defined parameters to represent the identified stressors to be taken into account in the EIF_{DD}

Water column:	**Sediment**
Toxic components concentrations	Toxic components concentrations
Suspended particle concentrations	Oxygen depletion as a result of organic carbon enrichment
	Change in grain size distribution
	Coverage by sedimentation of material

4. Exposure assessment

The exposures of the different stressors will all be assessed by the DREAM model (Rye *et al*., 2006a). As a part of the ERMS project the ParTrack model has been merged into DREAM. Both the water column and the sediment compartment are included in one model environment. Once released from the discharge pipe the mud and cuttings will behave differently. Part of the mud will stay in suspension and will be carried away from the discharge pipe by the currents. The cuttings will sink to the seafloor. Both processes are incorporated in the model (Figure 5). This chapter will describe for the different stressors how the exposure is derived and which level is taken to represent the exposure in a certain grid cell. In order to model the disturbed sediments also processes in the undisturbed sediments was needed to be studied.

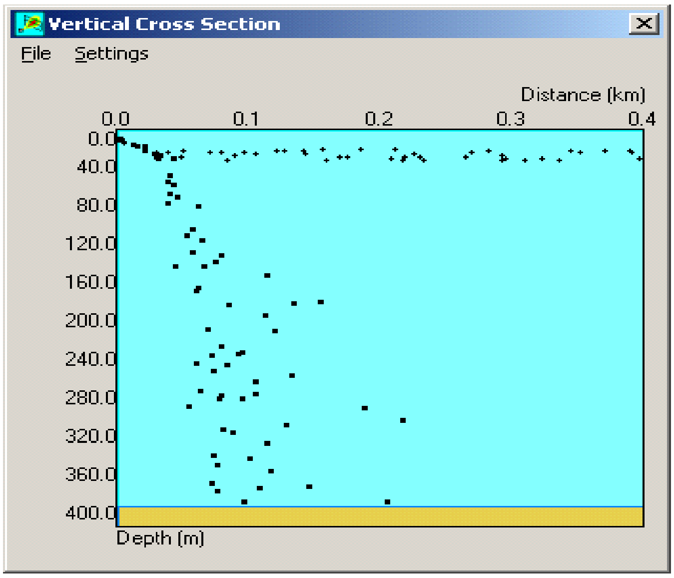

Figure 5 Presentation of modelled particles which partly deposit on the seafloor and partly suspend in the water column (taken from Rye et al, 2006a).

4.1 Exposure in the water column

The concentration in the water column is determined by the physical and chemical properties of the discharged chemicals and under the influence of environmental processes like currents, turbulence, and mixing. In principle, the same approach has been adapted for the drilling discharges that have been used for calculating produced water concentrations. Transport features are built into the simulation model in order to have a proper description of the behavior of the different cuttings and mud compounds/particles in the water column.

The model concept applied is a “particle” (or Lagrangian) approach. The model generates particles at the discharge point, which are then transported with the currents and turbulence in the sea. Different properties of the particles (representing properties like mass of various compounds, densities and sinking velocities) are associated with each particle as shown in Figure 5. Model particles are representing different state variables, such as gas bubbles, droplets, dissolved matter and solid matter. For discharges of drill cuttings and mud, solid particles and dissolved matter will be of particular interest.

4.1.1 Chemicals

Three categories of chemicals associated with drilling waste discharges have been selected for prediction of the possible harm (toxic and non-toxic) to the marine environment of drilling discharges:

- Metals (as ingredients of added chemicals or as part of the barite)
- Natural organic compounds
- Added chemicals (including drilling fluid chemicals e.g. non-PLONOR, PLONOR chemicals etc. (Chemicals Posing Low or No Risk listed by OSPAR, reference nr. 2004-10)

The chemicals included in these three categories are summarized in Table 2. Many chemicals are included in each of the categories. However, most drilling muds and drill cuttings do not contain environmentally significant amounts (sufficient quantities to cause acute or chronic harm to the marine environment) of all these chemicals.

A limited number of chemicals were selected from Table 2 for inclusion in the risk calculation for drilling discharges, based on following criteria:

- The total amount of each chemical used and discharged to the sea from drilling discharges (particularly PLONOR chemicals);
- The potential for the chemical to accumulate in the water column (soluble chemicals) or sediments (low-solubility chemicals) in forms and concentrations that could be toxic (and/or cause other disturbances (burial, oxygen depletion etc.) to marine organisms.

Table 2 Candidate chemicals for use in the risk calculations (EIF) for drilling discharges.

Metals (as ingredients of added chemicals or as part of the barite)	
Arsenic, Barium, Cadmium, Chromium, Copper, Mercury, Nickel, Lead, and Zinc	Both the free-available and attached form of the metal needs to be considered
Natural Organic Compounds	
BTEX	Might be present in the water column following oily cuttings discharge
Naphthalenes	Includes naphthalene and C_1- through C_3-naphthalenes
Other 2-3-Ring PAH	Includes 12 analyte groups, including fluorenes, phenanthrenes, and dibenzothiophenes
≥ 4-Ring PAH	Includes 10 unalkylated PAH, including benzo(a)pyrene
Aliphatic Hydrocarbons	Total petroleum hydrocarbons measured by IR or GC/FID. Sometimes high in cuttings piles
C_0-C_3 Phenols	Might be present in solution in drilling waste plume in water column
C_4-C_5 Phenols	Might be present in solution in drilling waste plume in water column
≥ C_6 Phenols	Might be present if the mud contains alkylphenol polyethoxylate detergents. Focus on C_8- & C_9-phenols
Added chemicals (green chemicals)*	
Barite	Barium sulfate: the most frequently used weighting agent in drilling muds. The most abundant solid ingredient in most muds
Carboxymethyl cellulose	Often used instead of lignosulfonate as a clay deflocculent
Bentonite	Montmorillonite clay. Usually, a major solid ingredient in drilling muds
Portland cement class G	Used in setting casing. Usually not discharged intentionally
Quartz	Silica. Generally considered toxicologically inert
Xanthan gum	Natural plant material for viscosity control
Added Chemicals (other chemicals than the green chemicals)**	
Added chemicals to drilling muds	Yellow substances should be evaluated; black or red substances will be evaluated if use is proved to be necessary from a safety or a technical perspective

* In the OSPAR area is this PLONOR List Chemicals. The six most frequently discharged PLONOR chemicals on the Norwegian Continental Shelf in 2003 are listed.

** Categorisation and colour code for chemicals used and discharged offshore on the Norwegian Continental Shelf.

For WBM in particular, many of the chemicals discharged are water soluble. For these chemicals, the same assumption is used as for the produced water dissolved compounds. They are assumed completely dissolved into the water column. The concentration for each compound can then be calculated from the mass and the number of particles within a unit volume of water for each of the compounds included. The model includes biodegradation and evaporation, adsorption to natural particles in the water and additions due to background concentrations in the water. The model includes these factors for each chemical or component group, dependent on the information available.

For the estimated release of substances, a distinction is usually made between substances that are emitted through point sources at specific locations and substances that enter the environment through diffuse (non-point source) releases (See also the TGD (EU, 2003)). Point source releases have a major impact on the environmental concentration on a local scale and also contribute to the concentration on a larger scale. The concentrations of substances released from point sources are assessed for a generic local environment, so-called “standard environment” (averaged values over 24 hours, or reasonable worst-case values). This means that the local PEC is calculated on the basis of daily release rate, regardless of whether the discharge is intermittent or continuous. It represents the concentration expected at a certain distance from the source on a day when discharge occurs. In principle, degradation and distribution processes are taken into consideration for the local PEC. The regional PEC takes into account the further distribution and fate of the substance over a wider area. It also provides a background concentration to be incorporated in the calculation of the local PEC. This is specifically for natural chemicals such as metals and some organo-metallic compounds. Relevant site-specific information can be used, if it is available, to improve the assessment.

According to the TGD, first priority is given to measured data for determination of PEC, if it is available. These data will be carefully evaluated for their adequacy and reliability according to criteria, such as an evaluation of sampling/analytical methods employed and the geographic and time scale of the measurements. Preferably, the measured data should be compared to the model-estimated PEC and a representative PEC should be based on a measured and a calculated PEC for risk characterisation. For existing substances, the generic “reasonable worst case” exposure concentration should be based on modelling and, measured data from site-specific or monitoring information, can then be used to revise the calculated concentrations.

Dissolved compounds

The EU-TGD (EC, 2003) provides recommendation on how concentrations (the PEC's) are calculated for substances discharged to sea. For dissolved chemicals in the water column, the TGD recommends the following formula:

Equation 1: Calculation of local PEC $_{seawater}$ for the marine environment (TGD approach)

$$PEC_{seawater} = C_{discharge} / DILUTION * (1 + Kp_{susp} * SUSP_{seawater} * 10^{-6}) + PEC_{seawater, regional}$$

in which:

$PEC_{seawater}$ = local concentration in seawater during emission episode $[mg*l^{-1}]$
$PEC_{seawater, regional}$ = regional concentration in seawater $[mg*l^{-1}]$
$C_{discharge}$ = concentration of the substance in the discharge $[mg*l^{-1}]$
$Kp_{susp-water}$ = solid-water partition coefficient for suspended matter $[l*kg^{-1}]$
$SUSP_{seawater}$ = concentration of suspended matter in seawater $[mg*l^{-1}]$
DILUTION = dilution factor [-]

The concentration at the regional scale ($PEC_{seawater,regional}$) is used as background concentration in seawater for the local scale.The DILUTION factor can be estimated analytically for certain cases (say, discharges into rivers). In the ERMS project, the DILUTION is calculated with the DREAM model. For using this formula for PEC calculations, information must be available on the suspended matter in the sea water and on background concentrations (denoted "regional concentrations in sea water" in Equation 1). These contributions can be calculated with the DREAM model if needed. However, experience has shown that it may be difficult to obtain good data on these factors. The content of suspended matter undergoes usually seasonal variations, in particular the organic part of the particle matter. Therefore, in the ERMS approach (for the time being), the concentrations of the discharge compounds (the PEC's) are calculated without the contributions from background concentrations and natural particle matter present in the recipient.

Reduction of concentrations due to biodegradation of the chemicals dissolved is included in the ERMS approach. The formula used in the ERMS approach can thus be written as (Equation 2):

Equation 2: Calculation of local $PEC_{seawater}$ for the marine environment (ERMS approach)

$$PEC_{seawater} = C_{discharge} * \exp(-kt) / DILUTION$$

in which:

$PEC_{seawater}$ = *local concentration in seawater during emission episode* $[mg*l^{-1}]$
$C_{discharge}$ = *concentration of the substance in the discharge* $[mg*l^{-1}]$
k = *biodegradation factor* $[days^{-1}]$
t = *time [days]*
DILUTION = dilution factor [-]

The ERMS approach only includes chemicals (both natural substances and added drilling fluid chemicals) with a partition coefficient $\log P_{ow}$ smaller than 3 as dissolved in the water column. Chemicals with $\log P_{ow}$ coefficient larger than 3 are assumed to be “attached” to particles or to form “agglomerated particles”. These particles will to a large extent descend down on the sea floor and impact on the sediment. The fates of these are treated in section 4.2 of this report.

For the present ERMS approach, the DILUTION factor given in Equation 2 includes the effects from:
- Near field mixing, driven by the momentum and the negative buoyancy of the discharge;
- Far field mixing, driven by the natural turbulence and transport in the recipient;
- Evaporation into the atmosphere (only for compounds where relevant information is available).

Heavy metals in barite

The concentration of dissolved metals originating from barite in the water column is estimated from Equation 3:

Equation 3: Calculation of local PEC $_{seawater}$ for dissolved metals from barite in the marine environment (ERMS approach)

$$PEC_{seawater} = (C_{discharge} / DILUTION) * (FRACTION / Kp_{metal})$$

in which:

$PEC_{seawater}$ = local concentration in seawater of dissolved metal [$mg*l^{-1}$]
$C_{discharge}$ = concentration of barite particles in the discharge [$mg*l^{-1}$](see 4.1.2)
DILUTION = dilution factor for dilution of the discharge in recipient water [-]
FRACTION = fraction of the metal in barite [kg metal.kg^{-1} barite]
Kp_{metal} = partition coefficient between the metal in barite and dissolved metal

The dissolved metal concentrations originating from barite is influenced by two factors:

- The part of the barite that consists of the metal in question (denoted FRACTION)
- The part of the metal in the barite that is bioavailable (that is, dissolved in the ambient water, denoted Kp_{metal}).

These two factors are therefore built into the Equation 3 above (see the last bracket) for the calculated concentration of the dissolved metals originating from barite (ERMS approach). A high Kp_{metal} value indicates low potential of leakage to the water phase and low Kp_{metal} value a higher potential for flux to the water. At present, only metals originating from barite are included in the ERMS approach. However, when other weighing agents are used, the Kp_{metal} values applied to barite may be used. Cuttings and other particle matter may contain heavy metals as well, but the barite is currently assumed to contain the most dominant contributions to the heavy metal risks for discharges of drill cuttings and mud. In Frost *et al.* (2006) guidance on the selection of values for the parameters FRACTION and Kp_{metals} is provided.

4.1.2 Suspended particles

The second stressor in the water column is suspended particles. The weighting agent in the drilling mud is the main source of suspended particles; usually consisting of barite, illminite, bentonite or attapulgite. As a function of discharge volume, composition and some characteristics of the discharge pipe, the concentration of suspended particles in the water column will be calculated by the model. This concentration is a result of different processes determining the horizontal and vertical transport of suspended matter around the discharge location. A part of the suspended matter will remain suspended in the water column and will be carried by the currents over long distances. Another part will settle on the seafloor in the vicinity of the discharge location.

The formula used to calculate the PEC for particle matter in the water column can be written as (Equation 4):

Equation 4: Calculation of local PEC $_{seawater}$ for the particle matter discharged into the marine environment (ERMS approach)

$$PEC_{seawater} = C_{discharge} / DILUTION$$

in which:

$PEC_{seawater}$ = *local concentration of discharged particles in seawater during emission episode* $[mg \cdot l^{-1}]$
$C_{discharge}$ = *concentration of the substance or particles in the discharge* $[mg \cdot l^{-1}]$
DILUTION = dilution factor [-]

The DILUTION factor is the same as in Equation 2, except that the DILUTION factor in Equation 4 also includes fall-out of particles from the plume and down on the sea floor (due to gravity). For particles, the biodegradation factor k in Equation 2 is assumed to be zero (no biodegradation of particles). Equation 4 is applied for all particle contents in the discharge (cuttings, barite, bentonite, etc). The fall-out of particles due to the particle density represent an extra "dilution" of the discharge because the content of the particle matter in the discharge plume is reduced due to the fallout.

Figure 6 is a 2-dimensional snapshot of the dynamic concentrations of suspended matter as calculated by the model. The momentary concentration in each volume grid cell is taken as PECwater for suspended matter.

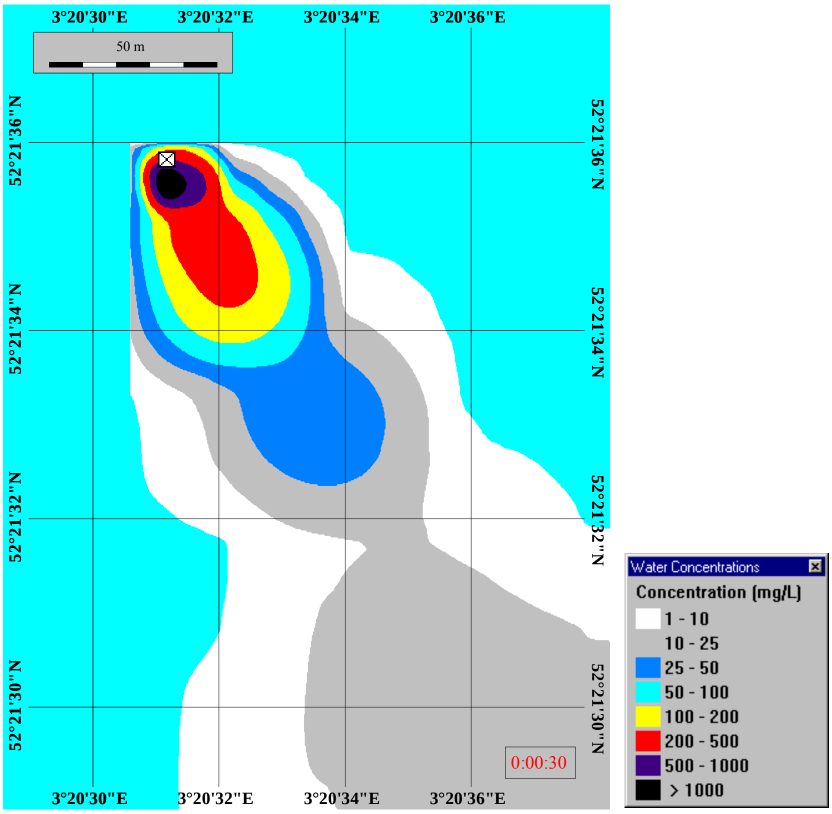

Figure 6 *Two dimensional snap shot of a 3D suspended matter plume (figure from H. Rye at SINTEF)*

4.2 Exposure in the sediment

Due to the particle content of the discharge, parts of the discharge sink down on the sea floor. Once on the seafloor, processes like bioturbation and degradation will change the quality and structure of the sediment. In order to assess the exposure also undisturbed sediment processes have to be incorporated in the model. Chemicals in the discharge may attach to the particles and thus end up on the sea floor as well, even though the density of the chemicals may be smaller than the ambient water. The chemicals may also form larger particle “agglomerates” due to the sticky properties of the chemical (OBM, SBM). The bottom sediment model must therefore be able to calculate the impacts on the original sediment layer caused by the parts of the discharge that deposit on the sea floor.

Particles will sink down on the sea floor with sinking velocities dependent on their size and density. A near field descending plume is included. The inclusion of a three-dimensional (3D) and time variable ocean current field will cause a spread of the deposits on the sea floor, resulting into a deposition with characteristics dependent on the horizontal co-ordinates x and y. Each grid cell on the sea floor then contains the amount of drill cuttings and mud (particles, chemicals) deposited on the sea floor within that cell.

The layer deposited is assumed to be well mixed (homogeneous) and aerated (oxygen content equal to the initial oxygen content of the upper sediment layer). Because one of the stressors is the *change* of oxygen content in the sediment layer due to the discharge, the content of the oxygen in the sediment layer before and after the discharge needs to be calculated.

Because the oxygen content varies with sediment depth, the whole oxygen profile in the vertical needs to be calculated (before discharge) and the time variation of the oxygen content in the sediment as well (during and after discharge). This has to be done for every horizontal grid point at the sea floor due to the fact that the deposition on the sea floor will vary with geographic location (x and y). A full 3D and time variable presentation of the oxygen content in the sediment before, under and after the discharge needs to be established.
The processes that influence on the oxygen profile in the sediment are:

- The amounts of biodegradable matter (mostly biodegradable carbon) in the sediment;
- The diffusion of the free oxygen downwards through the pore water in the sediment; and,
- The consumption of the free oxygen due to the presence of biodegradable organic matter in the sediment.

The biodegradable matter itself (comprising carbon, nitrogen and phosphorus, basically) spreads downwards into the sediment layer due to natural burial and bioturbation (that is, mixing within the sediment layer due to the presence of sediment re-workers).

An approach that is able to describe these processes makes use of the so-called “diagenetic equations”. These are differential equations which have to be solved numerically. One example of such an equation can be written as (simplified version):

$$\frac{\partial C}{\partial t} = \frac{\partial}{\partial z}\left(D \ \frac{\partial C}{\partial z}\right) - K_C \ C + Q_C \qquad \text{Equation 5}$$

where the two last terms on the right hand side are reaction or source terms. The first term on the right hand side is a diffusion term (could represent bioturbation or diffusion D through pore water), while the term on the left hand side expresses the rate of change of the concentration C (which could be a chemical, organic matter in the sediment, oxygen or some content of solid matter). These types of equations are applied to calculate the oxygen balance in the sediment layer before the discharge (Rye *et al*., 2006a).

During and after the discharge, the four stressors defined for the sediment layer have to be calculated (burial, free oxygen depletion, toxicity and change of grain size). These four parameters are calculated for the new sediment layer (including the added deposition on the top) as follows:

- *The toxicity* of the new sediment layer is simply calculated from the content (concentration) of the chemical(s) in the added sediment. These chemicals may however bioturbate into the original sediment layer, causing a mixing between the new and the old sediment.
- *The burial* is represented by the thickness of the new layer added. This parameter is calculated from the depositions of the discharge compounds only.
- *The change of grain size* is a new layer with another median grain size added on the top of the former (natural) sediment layer. These two layers may then start to mix into one another due to bioturbation, causing a distribution of the median grain size in the vertical.
- *The free oxygen depletion* is calculated from re-calculating the new free oxygen profile after discharge. The biodegradation from the added chemicals in the new sediment layer must then be included in addition to the natural biodegradation (present in the sediment layer before discharge). This biodegradation may then cause a reduction of the free oxygen content in the pore water of the sediment layer. The actual reduction of the free oxygen content in the pore water of the sediment layer is calculated by taking the difference between the new oxygen content in the pore water of the sediment after discharge and the oxygen content before discharge.

All these sediment processes are described in the model by means of an implementation of diagenetic equations. In order to describe the processes in sufficient detail, four diagenetic equations are used:

- One for the free oxygen content in the pore water
- One for the natural organic matter in sediment
- One for the added organic matter (chemicals) due to the discharge
- One for the change of the grain size in the sediment.

One of the requirements for the sediment impact model is to calculate the restitution time or the recovery time of the sediment layer after deposition. Examples of processes that will contribute to the recovery of the sediment layer will be:

- Burial by natural deposition after discharge (dilution effect)
- Bioturbation after discharge (dilution effect)
- Biodegradation of the chemicals (removal of chemicals)
- Re-suspension of deposited matter (re-distribution and removal)

The first three of these processes are modeled by means of the diagenetic equations. Both burial, bioturbation and biodegradation effects are included in these equations, including toxicity (which is reduced as a consequence of the biodegradation) and free oxygen depletion (which may occur during the biodegradation of the chemicals). Details on restitution processes are given in Rye *et al*. (2006b).

4.2.1 Chemicals

The concentration in the freshly deposited sediment is taken as the $PEC_{sediment}$; therefore, the properties of the suspended matter are used. The concentration in bulk sediments can be derived from the corresponding concentration in the water column by predictions from Equation 6, assuming thermo-dynamic partitioning equilibrium (Di Toro *et al.,* 1991):

Equation 6: Calculation of local $PEC_{sediment}$ for the marine environment (TGD approach)

$$PEC_{sediment} = [Kp_{susp\text{-}water} / RHO_{susp}] * PEC_{seawater} * 1000$$

in which:

$PEC_{seawater}$ = local concentration in seawater during emission episode [$mg*l^{-1}$]
$Kp_{susp\text{-}water}$ = suspended matter-water partitioning coefficient [m^3*m^{-3}]
RHO_{susp} = bulk density of suspended matter [$kg*m^{-3}$]
$PEC_{sediment}$ = predicted environmental concentration in sediment [$mg*kg^{-1}$]

Highly adsorptive or insoluble, solid substances may not be considered subsequently with the approach described above, as their concentration in water and suspended matter are not in equilibrium because of their strong association with suspended matter; however they may be desorbed or dissolved after ingestion by benthic organisms.

Experience has shown that the sediment may be strongly impacted by discharges of drill cuttings and mud, even though the plume concentrations (the $PEC_{seawater}$) are not in touch with the sea floor at all. This case is not covered by the TGD (2003). The mechanisms that will bring the discharges down on the sea floor are sinking particles and/or chemicals attached to the particles (or chemicals that agglomerate to form particles). These processes are accounted for in the described DREAM model.

To put these processes into a more formal framework, the deposition of the discharges for larger depths can be expressed as (Equation 7):

Equation 7: Calculation of local $PEC_{sediment}$ for the marine environment (ERMS approach)

$$PEC_{sediment} = [1000 / RHO_{susp}] * PEC_{discharge} / DEPOSITION$$

in which:

$PEC_{discharge}$ = concentration in discharge pipe [$mg*l^{-1}$]
DEPOSITION = deposition factor calculated by the model [-]
RHO_{susp} = bulk density of suspended matter in sediment [$kg*m^{-3}$]
$PEC_{sediment}$ = predicted environmental concentration in sediment [$mg*kg^{-1}$]

This formula applies to both chemicals and particles (cuttings, barite, etc.) in the discharge. The chemicals included are only those with a $\log P_{ow} \geq 3$ (chemicals assumed not to be dissolved in the water column).

The “DEPOSITION” factor represents the spreading of the discharge in the ambient, transport by the currents, the sinking down and deposition of the particles (including chemicals) on the sea floor.

No biodegradation or other processes are involved in the deposition calculations, because these processes are assumed to take place over a relatively short period of time, generally. Once the chemicals are deposited on the sea floor, they may start to biodegrade.

The TGD (2003) does not provide any guidance on the re-distribution of chemicals in the sediment. Therefore, a method for calculating time variable PEC in the sediment for toxic stressors has been developed. Once the matter deposited is down on the sea floor, the $PEC_{sediment}$ for chemicals (those with Log $P_{ow} > 3$) is calculated according to the following formula (Equation 8):

Equation 8: Calculation of local $PEC_{sediment}$ for organic chemicals in sediment (ERMS approach)

$$PEC_{sediment} = PEC(t=0)_{sediment} * \exp(-kt) / BIOTURB$$

in which:

$PEC_{sediment}$ = concentration of chemical in the sediment [$mg*kg^{-1}$]
$PEC(t=0)_{sediment}$ = same as above for the initial time step t = 0.
k = biodegradation factor for chemical in the sediment [$days^{-1}$]
t = time [days]
BIOTURB = dilution factor in the sediment due to effects from bioturbation [-]

The PEC $(t = 0)_{sediment}$ for organic matter is calculated based on Equation 7. The k factor in Equation 8 is a biodegradation factor which is not a constant. This factor will vary with the availability of free oxygen in the pore water. If the free oxygen goes to zero, the k factor will go to zero as well.

Equation 8 describes only the time development of the organic chemical concentration in the sediment. However, the risk assessment (comparison of PEC and PNEC) in the sediment will be based on the dissolved chemical in the pore water (and not on the concentration of the chemical in the sediment). The pore water concentration of the organic chemical in the sediment is based on equilibrium partitioning (Equation 9):

Equation 9: Calculation of local pore water concentration $PEC_{porewater}$ for organic chemicals (TGD approach):

$$PEC_{porewater} = [RHO_{susp} /1000] * PEC_{sediment} /Kp_{susp\text{-}water}$$

in which:

$PEC_{porewater}$ = concentration of chemical in the pore water [$mg*l^{-1}$]
$PEC_{sediment}$ = concentration of chemical in the sediment [$mg*kg^{-1}$]
$RHO_{susp\text{-} water}$ = bulk density of suspended matter [$kg.m^{-3}$]
$Kp_{susp\ water}$ = suspended matter-water partitioning coefficient [m^3*m^{-3}]

In Equation 9, the $Kp_{susp\ water}$ is given in non-dimensional form ($m^3.m^{-3}$). Sometimes, the partition coefficient is given in dimensional form, like $L.kg^{-1}$ sediment (Di Toro *et. al*. (1991). The partitioning factor $Kp_{susp\ water}$ should in such a case be written as shown in Equation 10:

Equation 10: Calculation of local pore water concentration $PEC_{porewater}$ for organic chemicals (TGD approach):

$$PEC_{porewater} = PEC_{sediment} /Kp_{susp\text{-}water}$$

in which:

$PEC_{porewater}$ = concentration of chemical in the pore water [$mg*l^{-1}$]
$PEC_{sediment}$ = concentration of chemical in the sediment [$mg*kg^{-1}$]
$Kp_{susp\ water}$ = suspended matter-water partitioning coefficient [$l*kg^{-1}$]

The ionic strength, composition, and pH of seawater, compared with freshwater, have potential effects on the partitioning (Kp) of a chemical among environmental compartments. This is particularly true for ionisable chemicals, such as metals and phenols. To a large extent, these effects are associated with differences in water solubility and/or speciation of the chemical, compared with freshwater.

The Kp for the sediment is given by (Equation 11):

Equation 11: Calculation of Kp $_{sediment}$ of non-ionic organic substances in sediment (TGD approach)

$$Kp_{sediment} = F_{oc} * K_{oc}$$

in which:

K_{oc} = *partition coefficient organic carbon-water [l*kg^{-1}]*
$Kp_{sediment}$ = *partition coefficient for the sediment for non-ionic organic substances [l*kg^{-1}]*
F_{oc} = *weight fraction of organic carbon in compartment [kg*kg^{-1}]*

Equations 7 to 11 describe how the organic chemicals in the discharge deposit on the sea floor and dilute and biodegrade due to processes in the sediment. They also explain how the concentrations of the chemical in the pore water ($PEC_{porewater}$) are calculated. These formulas are implemented in the DREAM model.

Metals

Heavy metals attached to the barite may enter the sediment layer along with the barite particles. These metals may affect the biota in the sediment layer. In the model, the bioavailable part of the metals is determined through equilibrium partitioning, that is, a part of the metals is assumed to be bioavailable through dissolution of the metal into the pore water. In the following, the metal toxicity in the sediment is accounted for, as presently implemented in the model.

The concentration of barite deposited in the sediment initially during the release ($PEC_{sediment}$) is calculated by Equation 7. Once the barite is deposited in the sediment, the PEC for the metals in the sediment (due to the presence of the barite) is calculated according to the following formula (Equation 12):

Equation 12: Calculation of local PEC_{metal} for barite metals in sediment (ERMS approach):

$$PEC_{metal} = PEC(t=0)_{sediment} * FRACTION / BIOTURB$$

in which:

PEC_{metal} = *concentration of barite metal in the sediment* $[mg*kg^{-1}]$
$PEC(t=0)_{sediment}$ = *concentration of deposited barite in the sediment at t = 0* $[mg*kg^{-1}]$
FRACTION = content of the metal in barite $[kg\ metal * kg^{-1}\ barite]$
BIOTURB = dilution factor in the sediment due to effects from bioturbation [-]

The BIOTURB factor is calculated by the model, and represents the downward mixing of the metal in the sediment caused by the moving organisms in the sediment (bioturbation effect). The PEC $(t = 0)_{sediment}$ for deposited barite on the sea floor is calculated by Equation 7.

Equation 12 describes the time development of the barite metal concentration in the sediment. The pore water concentration of the metal in the sediment is based on equilibrium partitioning (Frost *et. al.*, 2006), as described by Equation 13:

Equation 13: Calculation of local pore water concentration $PEC_{porewater}$ for barite metals (ERMS approach):

$$PEC_{porewater} = [RHO_{susp} / 1000] * PEC_{metal} / Kp_{metal\text{-}porewater}$$

in which:

$PEC_{porewater}$ = *concentration of dissolved metal in the pore water* $[mg*l^{-1}]$
PEC_{metal} = *concentration of metal in the sediment* $[mg*kg^{-1}]$
RHO_{susp} = *bulk density of suspended matter* $[kg.m^{-3}]$
$Kp_{metal\text{-}porewater}$ = *barite metal-pore water partitioning coefficient* $[m^3*m^{-3}]$

The $Kp_{metal\text{-}porewater}$ is given in non-dimensional form ($m^3.m^{-3}$). Sometimes, the partition coefficient is given in dimensional form, like $L.kg^{-1}$ sediment (Di Toro *et. al.*, 1991). The partitioning factor $Kp_{metal\text{-}porewater}$ should in such a case be written as shown in Equation 14:

Equation 14: Calculation of local pore water concentration $PEC_{porewater}$ for barite metals (ERMS approach):

$$PEC_{porewater} = PEC_{metal} / Kp_{metal\text{-}porewater}$$

in which:

$PEC_{porewater}$ = *concentration of dissolved metal in the pore water* $[mg*l^{-1}]$
PEC_{metal} = *concentration of metal in the sediment* $[mg*kg^{-1}]$
$Kp_{metal\text{-}porewater}$ = *barite metal-pore water partitioning coefficient* $[l*kg^{-1}]$

The Equations 12 to 14 describe how the concentrations of the metals in the barite that deposit on the sea floor are calculated. They also describe the calculation of the metal concentrations in the pore water ($PEC_{porewater}$). These formulas are implemented in the DREAM model.

Bioturbation will cause the deposited layer to be mixed into the original sediment. This results in gradients of concentrations of contaminants from the upper layer into the sediment. The model calculates this gradient for each chemical in all surface grid cells. The concentration of a chemical in the sediment ($PEC_{sediment}$) is specified by averaging the content of chemical over the upper three cm of the sediment layer (including the added layer) (Equation 15):

$$\text{Sediment chemical concentration} = \frac{1}{3cm}\int_{0}^{3cm} C(z)\,dz \qquad \text{Equation 15}$$

where C(z) is the concentration of the chemical deposited on the sea floor along with the cuttings (or other type) particles. The concentration of the chemical is allowed to have a sediment depth dependency due to the effects from bioturbation (as for the grain size change). This value will be used in the risk assessment. Figure 7 provides a schematic overview of the concentration of a toxicant over depth in one grid cell.

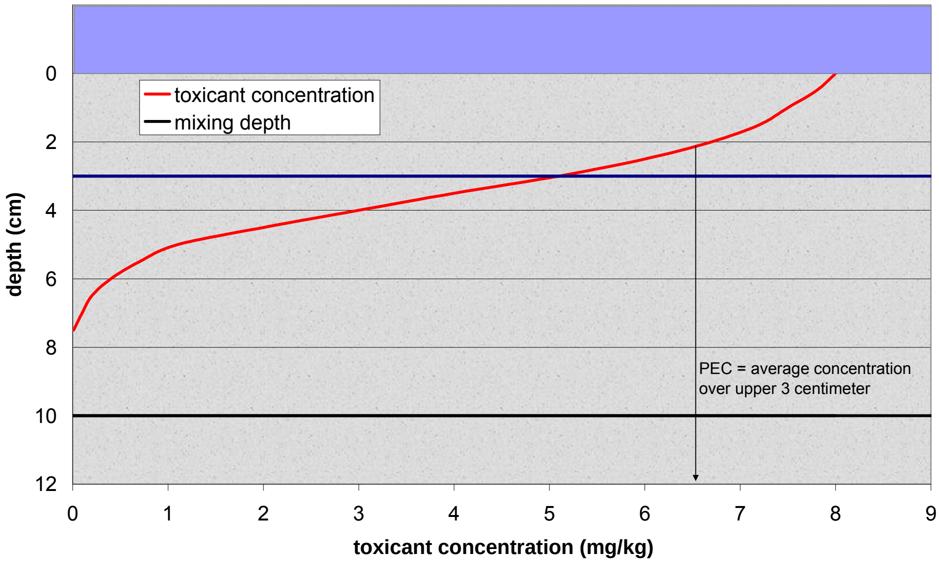

Figure 7 Gradient of sediment concentrations over depth and the selected level of the PECsediment

4.2.2 Burial

The fraction of the suspended matter that settles will, together with the settled cuttings, form a deposited layer on the seafloor. This layer, when formed quickly, could bury sediment biota and will therefore pose a risk. Both the settling velocity and the thickness of the layer determine the final risk on burial. The model will calculate the thickness of the deposited layer over time. During discharge (and shortly after discharge) this layer will be build up. Eventually bioturbation and re-suspension will reduce the layer thickness. The layer thickness in millimetres over time will be used as the "PEC" in the determination of risk on burial.

Burial is defined as the total thickness of the added layer caused by the deposition. This build-up is caused by the particles (grains) in the discharge (cuttings and particles in the weight material). The thickness is calculated from:

$$\text{Burial} = \frac{1}{1-\varphi}\sum_i \frac{M_i}{\rho_i} \qquad \text{Equation 16}$$

where φ is the porosity, M_i is the mass of particle component *i* deposited pr. m^2 of the sediment area, and ρ_i is the density of the particles of class *i*. The summation is carried out over all particle classes *i* deposited on the sea floor. As an example of calculation, assume a porosity of 0.6 and a density of cuttings particles equal to 2500 $kg.m^{-3}$, then the thickness of 1 kg of cuttings deposited over 1 m^2 will result into a thickness (burial) of about 1 mm. The porosity factor is included because the particles are only occupying a part of the total volume of the sediment. Figure 8 presents a schematic overview of the increasing layer thickness in a sediment surface grid cell.

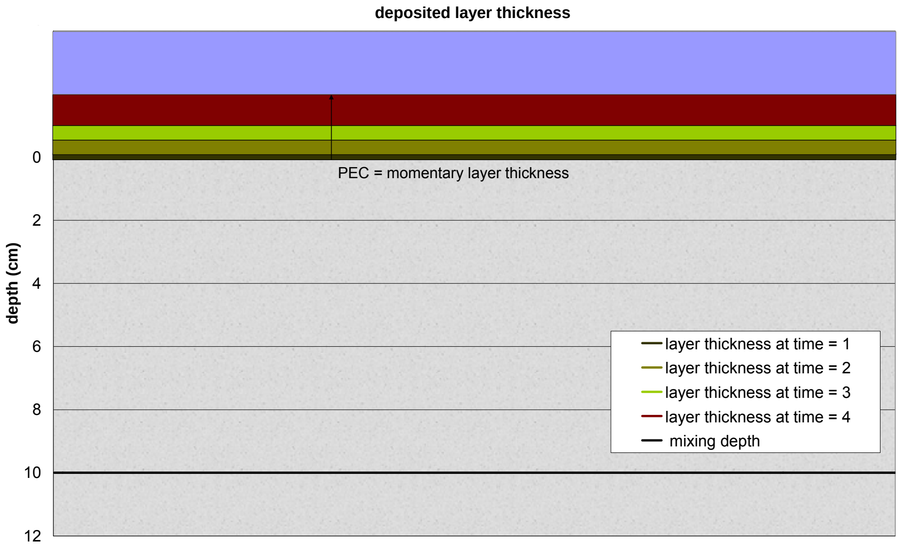

Figure 8 Increased layer thickness over time. The PEC is the momentary layer thickness

4.2.3 Change in grain size

The deposition of particulate matter and cuttings will alter the characteristics of the original sediment. One of the parameters frequently used to describe the sediment structure is the median grain size. After particulate matter and cuttings are deposited on the seafloor, bioturbation will mix the deposited layer with the original sediment. This leads to a gradient of changed median grain size with depth and distance compared to the original median grain size. In field monitoring studies the average median grain size over the upper three centimetres is used to describe the sediment characteristics. The change in the grain size over the upper three centimetres compared to the original grain size was chosen to represent the “PEC” in the risk assessment. *Grain size change is defined as the change of the median grain size* in the sediment, averaged over the upper three cm of the sediment layer (including the added sediment). The parameter is defined as:

$$\text{Grain size change} = \frac{1}{3\,cm}\int_{0}^{3\,cm} D(z)\,dz - D_{original} \qquad \text{Equation 17}$$

where D(z) is the median grain size parameter (diameter) after discharge. $D_{original}$ is the median grain size of the natural sediment on the site before the deposition has taken place (constant value). The new grain size parameter after discharge D(z) may have a sediment depth dependency due to effects from bioturbation. This definition secures that the environmental stress imposed by a finer grain size added on top of coarser sediment, will be similar as for the stress imposed by coarser sediment added on top of finer grain sized sediment. Figure 9 presents the gradient of changed grain size after deposition and bioturbation over depth in one surface grid cell on the seafloor.

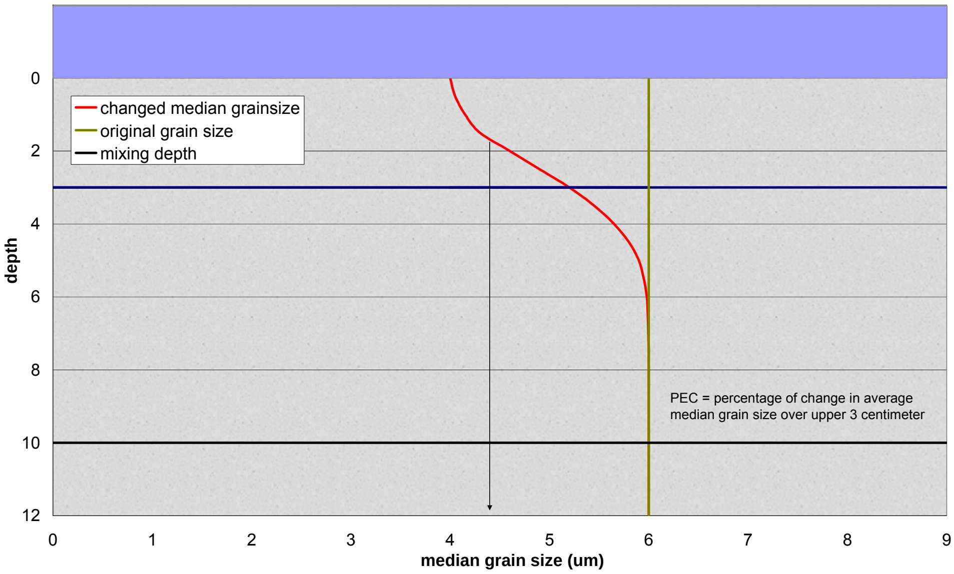

Figure 9 Gradient of altered medium grain size with depth and the PEC level

4.2.4 Reduction of the oxygenated layer

As mentioned earlier the oxygen concentration in the sediment is a result of several complex processes including biodegradation, diffusion, respiration, etc. Schaanning & Bakke (2005) and Rye *et al*. (2006a) describe these processes in order to assess the oxygen concentration of undisturbed sediment. As a result of the discharge and deposition of biodegradable matter from drilling muds, the equilibrium in the original sediment might get disturbed. The reduction in oxygen will pose a risk to sediment biota. The exposure part of the model will describe the change of the oxygen concentration in the sediment. This is represented by a reduction of the oxygenated layer (presented by the redox potential discontinuity (RDP), See Schaaning & Bakke (2005).

It is assumed that the thickness of the oxygenated layer represents the sediment layer were biota is present. A reduction of this layer will pose a risk to sediment biota. The model will describe the thickness of the oxygenated layer expressed as the integrated oxygen concentration over depth (representing the total amount of oxygen in the Redox Potential Discontinuity (RDP-layer). A reduction of the integrated oxygen content will be calculated by the model, and was chosen to represent the “PEC” for oxygen related stress. *The oxygen depletion* parameter is calculated as the reduction of the oxygen content (%) in the sediment layer due to the discharge, integrated over the layer where bioturbation is taking place. The integrated free oxygen content of the layer has unit g O_2 in pore water.m^{-2} sediment surface. Generally, the free oxygen content in the layer is close to zero at the sediment depth L. This secures that approximately all the free oxygen in the sediment is included in the oxygen depletion parameter, defined as

$$O_2 \text{ depletion } (\%) = 100 \frac{\int_0^L \varphi \, O_2(z)\, dz_{\,afterdischarge}}{\int_0^L \varphi \, O_2(z)\, dz_{\,beforedischarge}} \qquad \text{Equation 18}$$

where $O_2(z)$is the oxygen concentration in the pore water (mg.L^{-1} or g.m^{-3}) and φ is the porosity included to account for the fact that only a part of the space (given by the porosity) is occupied by the pore water. Figure 10 presents a schematic overview of the changed oxygen concentration over depth in a sediment surface grid cell. The change in the integrated oxygen content is represented by the shaded part of the picture.

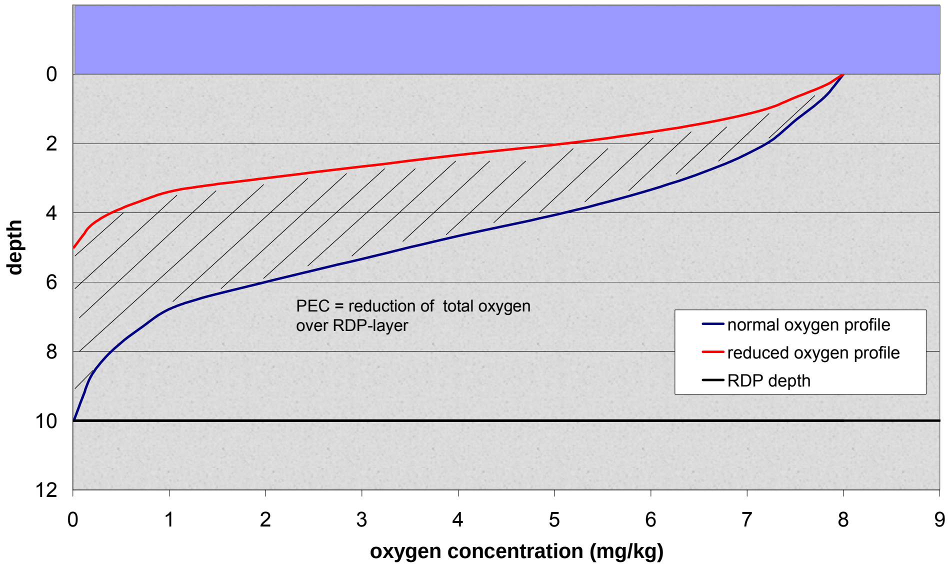

Figure 10 Change in the integrated oxygen concentration over depth. The PEC is the percentage change in the integrated oxygen content over the RDP layer

5. Effect assessment

In order to obtain an indication of the potential effects of drilling discharges, the exposures to the selected stressors will be compared to the levels at which they might cause effects. For toxicity effect levels are mainly obtained from laboratory studies where the sensitivity of biota for a specific toxicant is tested. Many databases containing toxicity data are available. For the disturbances other than toxicity the disturbance-effect relationships and the variation in species sensitivity is not easy to obtain. This is due to the fact that no regulatory framework is available for other disturbances than toxicity requiring structured data collection. Definition of effect levels for the non-toxic stressors will therefore include more inherent uncertainties. Assumptions made at the determination of thresholds and sensitivity distributions for non-toxic stressors are described in Smit *et al*., (2006b). In order to reduce uncertainty, threshold values should be validated experimentally in the future.

In order to estimate the risk level, the exposure level will be compared to a threshold value. The ratio of exposure and sensitivity (PEC_PNEC ratio) is a risk indicator. The risk indicators of the different stressors will be compared by applying the variation in species sensitivity. For each stressor both the PNEC level and a species sensitivity distribution (SSD) must be constructed. In this chapter the generic principles and the information needed on effects is described.

5.1 Effects in the water column

5.1.1 Chemicals

The sensitivity of the environment to toxic stress related to chemical exposure is tested in standardized laboratory tests and mesocosms. The toxic components of drilling discharges in the water column are divided into three classes (Frost *et al*., 2006):

1. Metals in drilling mud (added weighting agents);
2. Added chemicals (additives and base fluids; substances from the PLONOR list – Non-PLONOR chemicals (OSPAR Agreement 2004-10);
3. Other added chemicals if used in high quantities (“green” chemicals or PLONOR chemicals).

Natural organic substances (aromatic and poly aromatic hydrocarbons, etc.) if discharged, are assumed to associate to cuttings and mud particles and sink down to the sea floor and the contributions to the dissolved concentration in the water column is assumed to be limited (log $K_{ow} > 3$). For the components in the three classes the availability of toxicity data differs. For metals many effect concentrations are available while for specific additives only information for a few species is available. Depending on the availability of data, specific rules apply to determine a threshold value (PNEC) for the specific toxicant. The EU-TGD describes how assessment factors and statistical extrapolation can be applied to the data to assess the PNEC. As the drilling discharges affect both the water column and the sediment, PNEC values for both compartments must be defined. In the following section, firstly the approaches recommended for calculation of PNEC for single substances for the water compartment described in the EU Technical Guidance Document (TGD) are presented. Secondly, the approaches for derivation of PNEC values recommended applied for the EIF_{DD} are outlined. In Frost *et al.*, (2006) a detailed description is provided how sediment and water column PNEC values for all toxic stressors in drilling discharges are derived.

Assessment Factors
In principle, the PNEC is determined from the available toxicity data, by applying an assessment factor. PNEC values should be derived from the most sensitive endpoint regardless of the medium. The PNEC is calculated by dividing the lowest LC/EC50 (Lethal Concentration or Effect Concentration causing effect to 50% of the exposed individuals) or NOEC (No Observed Effect Concentration) value by an appropriate assessment factor in accordance with the EU-TGD. The assessment factor is applied to extrapolate from laboratory single-species toxicity test data to multi-species ecosystem effects. The assessment factor addresses a number of uncertainties:

- Interspecies variation (biological variance);
- Short-term to long-term toxicity extrapolation;
- Laboratory data to field impact extrapolation.

Preferably, toxicity data on at least the three selected taxonomic or trophic levels (e.g., algae, crustaceans, and fish) are required to determine a PNEC for a substance. As stated in the TGD, both freshwater and marine toxicity data can be used for calculating a PNEC. Recently, the TGD was extended with a special section for the marine environment. The principle of the PNEC calculation has not been changed, but justification has been made to account for the greater species distribution (higher biodiversity) of the marine environment compared to freshwater. It is assumed that the marine environment has broader species sensitivity than the freshwater environment because of the greater diversity of taxa in the marine environment. Therefore, higher assessment factors are applied for the marine environment, to reflect the greater uncertainty of extrapolation, as described by the EC in the revised TGD (EC, 2003). However, where data are available for additional marine taxonomic groups, for example rotifers, echinoderms, or molluscs, the uncertainties in the extrapolation are reduced and the magnitude of the assessment factor applied to a dataset can be lowered.

Assessment factors decrease in magnitude from higher values for short-term, acute studies from which L(E)C50 values have been derived to lower values for long-term chronic studies from which NOECs have been derived. For long-term studies, the magnitude of the assessment factors also decreases as information on a wider range of species becomes available. The assessment factors described in Table 3 are those that would normally be applied to the datasets available. However, expert judgement may be applied to the interpretation of a dataset, which may result in a lower assessment factor. A full justification must then be provided.

Table 3 Assessment factors proposed for deriving PNEC$_{seawater}$ for the marine water column for different data sets (EC, 2003).

Data set	Assessment factor
Lowest short-term L(E)C50 from freshwater or saltwater representatives of three taxonomic groups (algae, crustaceans and fish) of three trophic levels	10000
Lowest short-term L(E)C50 from freshwater or saltwater representatives of three taxonomic groups (algae, crustaceans and fish) of three trophic levels, + two additional marine taxonomic groups (e.g. echinoderms, molluscs)	1000
One long-term NOEC (from freshwater or saltwater crustacean reproduction or fish growth studies)	1000
Two long-term NOECs from freshwater or saltwater species representing two trophic levels (algae and/or crustaceans and/or fish)	500
Lowest long-term NOECs from three freshwater or saltwater species (normally algae and/or crustaceans and/or fish) representing three trophic levels	100
Two long-term NOECs from freshwater or saltwater species representing two trophic levels (algae and/or crustaceans and/or fish) + one long-term NOEC from an additional marine taxonomic group (e.g. echinoderms, molluscs)	50
Lowest long-term NOECs from three freshwater or saltwater species (normally algae and/or crustaceans and/or fish) representing three trophic levels + two long-term NOECs from additional marine taxonomic groups (e.g. echinoderms, molluscs)	10

Species Sensitivity Distribution (SSD)

Statistical extrapolation can be used to derive a threshold using the variation in species sensitivity (see Aldenberg & Jaworska (2000) for a review). If a large data set with sensitivity values for different taxonomic groups is available, these values can be used to draw a distribution. This distribution that describes the variability of hazard of a stressor to organisms is called a Species Sensitivity Distribution (SSD). This distribution can be presented as a frequency distribution (cumulative normal distribution curves or other similar distribution curves) of NOEC values for species. In general the method works as follows: sensitivity data are log transformed and fitted to a distribution function. It has been shown that the choice of a distribution is quite arbitrary and is mostly done based on best fit results (Kooijman, 1981; Newman *et al*., 2000; Smit *et al*., 2001; Van der Hoeven, 2001 and Wheeler *et al*., 2002). In this report we chose to use the log-normal distribution (natural logarithm). For this cumulative log-normal distribution, sensitivity values for species are fitted to a logarithmic scale. The mean (Xm) of this curve represents the position of the distribution on the x-axis and the standard deviation (Sm) determines the slope of the curve. In terms of the sensitivity of species, the Xm gives an indication of the *mean sensitivity*. The Sm represents the *sensitivity range* or *variation in sensitivity* to a stressor.

The PNEC is defined as a prescribed percentile of that distribution. In general, it is assumed that sufficient test data for use of statistical extrapolation methods will only be available for relatively few substances, primarily in the form of fresh water data. However, a combination of the statistical extrapolation method with the assessment factor approach is also possible (Posthuma *et al.*, 2002).

The method should be applied for all reliable NOECs from chronic/long-term studies, preferably on full life-cycle or multi-generation studies. The minimum species requirements when using the SSD method are at least 10 NOECs (preferably more than 15) for different species covering at least 8 taxonomic groups. Deviations from these recommendations can be made, on a case-by-case basis, through consideration of sensitive endpoints, sensitive species, mode of toxic action, and/or knowledge from structure-activity considerations.

The test data applicable to the most sensitive endpoint should be taken as representative for the species. For equivalent data on the same end-point and species, the geometric mean should be used as the input value for the calculation. If the valid results are considered to be too variable, then grouping into various conditions, should be considered.

The main assumption on the use of SSDs in risk assessment is that the distribution based on a selection of species (for which data is available) is representative for all species (in the field) (Aldenberg & Jaworska, 2000; Posthuma *et al.*, 2002; Forbes & Calow, 2002a and 2002b). Statistical extrapolation methods may be used to derive a PNEC from an SSD by taking a prescribed percentile of this distribution. For pragmatic reasons it has been decided that the concentration corresponding with the point in the SSD profile below which 5% of the species occur, should be derived as an intermediate value in the determination of a PNEC. This 5% point in the SSD is also identified as a hazardous level at which a certain percentage (in this case 5%) of all species is assumed to be affected. The affected fraction of the species is reffered to as the PAF-level (Potentially Affected Fraction), (e.g. Straalen van & Denneman, 1989, Aldenberg & Slob, 1993; Newman *et al.*, 2000; Hoeven van der, 2001; EC, 2003). Attempts to validate this choice of the 5th percentile have been made, however the choice remains quite arbitrary (Okkerman *et al.*, 1993; Versteeg *et al.*, 1999). A 50% confidence interval (c.i.) associated with this concentration should also be derived. The PNEC is then calculated according to Equation 19:

Equation 19: Calculation of PNEC $_{seawater}$ by the SSD method (TGD approach)

$$PNEC_{water} = \frac{5\%\ SSD\ (50\%\ c.i.)}{AF}$$

in which:
c.i. = confidence interval
AF= Assessment factor

AF is an appropriate factor between 5 and 1, reflecting the additional uncertainties identified, such as the overall quality of the database and the endpoints included, knowledge on mode of action of the chemical, etc.

The set of PNEC values which is likely to be applied for the EIF_{DD} is similar to the set in use for the EIF_{PW} (Frost, 2002). This set is based on toxicity data on at least three selected trophic levels (algae, crustaceans and fish). However, the experience from the PNEC values currently being used for metals in the EIF calculation of produced water, is that the PNEC for some metals (e.g., Cu) were lower or in the same range as the natural background concentration in seawater on the Norwegian continental shelf (Frost *et al.*, 2006). Therefore the derivation and use of an alternative set of PNEC values is considered.

The statistical extrapolation approach, Species Sensitivity Distribution (SSD) for determination of PNEC for metals, in accordance to the marine TGD, is being examined in an ongoing OLF project (2006). Provided that sufficient chronic aquatic toxicity data are available for metals, the current extrapolation method is recommended for use in developing PNEC for metals. Therefore, the toxicity threshold values applied for derivation of the, “Integrated Environmental Quality Objectives”, prepared by the Dutch National Institute of Public Health and the Environment (Crommentuijn *et al.*, 1997), were evaluated as “PNEC” values for metals. So-called Maximum Permissible Concentration (MPC_{water}), is the concentration above which the risk for the ecosystem is considered unacceptable (VROM, 1989). MPC_{water}s were derived for eighteen metals taking into account existing national background concentrations following the so-called “added risk approach”. MPCs were derived for water, sediment, soil and air. The MPCs served as a basis for the Dutch government to set generic Environmental Quality Standards (EQS) in The Netherlands. EQS in turn are used by the Dutch government to assess the environmental quality and for other environmental policy purposes (Crommentuijn *et al.,* 1997).

The derivation of PNEC values for added chemicals (non-PLONOR and PLONOR chemicals etc.) is based on a similar approach being applied to EIF_{PW}. However, a few adjustments are introduced to the method. Depending on the number and type of toxicological data available, the PNEC will be derived by using assessment factors following the principles of TGD or the SSD approach. The assessment factor applied can be lowered provided that the substance is released to the environment batch-wise rather than continuously. For intermittent releases, the TGD recommends to lower the assessment factor by a factor of 10, because long-term exposure is not expected. The assessment factor applied needs to be judged for the various discharge scenarios (cementing, BOP testing etc.) and on case-by-case. As the toxicological information is often limited to 3 acute values for added chemicals (non-PLONOR chemicals) the assessment factor approach will be used to derive a PNEC for these products. Frost *et al*. (2006) provides an overview of the different sets of PNEC values which can be used for the EIF_{DD}.

5.1.2 Suspended particles

Discharges of drilling muds and cuttings will result in increased concentrations of suspended particulate matter (SPM) in the water column, of which a substantial part will settle rapidly to the sediment surface. Smaller particles, however, may persist in the water column for longer time and may cause an impact on water column organisms, and organisms inhabiting the sediment but having contact with the overlying water. Particularly in water systems with relatively low suspended sediment concentrations (< 10 mg/l) an increase of the turbidity will lead to an increase of ecological effects (Dalfsen, 1999).

In order to assess the relevance and the potential impact of increased SPM concentrations due to the presence of weighting agents in Water Based Muds (WBM) discharges, a literature review was performed. The aim was to collect information that could contribute to establish a Predicted No Effect Concentration (PNEC) for weighting agents in analogy to the PNECs derived for toxic substances. These PNECs can then be used to derive the “Environmental Impact Factor (EIF) - water column” that will deal with the risk of effects of drilling discharges in the water column (Smit *et al*., 2006a).

The available information covers various taxonomic groups, enabling the use of assessment factors or Species Sensitivity Distributions (SSDs) to derive a PNEC. However, the quality of data is highly variable, because protocolised laboratory tests for suspended matter are lacking. The effect data on weighting agents include different types of particles (e.g. barite, bentonite, attapulgite and WBM) and several types of end-points (lethal and sub-lethal effects). In order to make use of the available acute data, application factors as described in the EU Technical Guidance Document, were applied to derive PNEC values for barite, bentonite, attapulgite and WBM (Note that no regulatory framework is available for the assessment of environmental effects of SPM).

Since the PNEC is supposed to represent the sensitivity of ‘all’ species, and physical effect data are only available for a few species and for limited effect endpoints, assessment factors can be applied as a precautionary measure in accordance with the EU-TGD.

The few NOEC data available were not appropriate for determining a PNEC as the information on the quality of the data was lacking. However, sufficient L(E)C_{50} data were available for at least three taxonomic groups (algae, crustaceans and fish). The PNEC has therefore been based on acute effect data.

When only short-term toxicity data are available, an assessment factor of 10,000 (marine TGD) or 1000 (freshwater TGD) should be applied on the lowest L(E)C_{50} of the relevant available toxicity data, irrespective of whether or not the species tested is a standard test organism. Table 4 provides an overview of the amount of data and the resulting PNEC applying the assessment factor approach.

Table 4 Derivation of PNECs for barite, bentonite, attapulgite and WBMs based on acute toxicity data and by using assessment factors

Type of weighting material	**Barite**	**bentonite**	**attapulgite**	**WBMs**
Number of effect data	30	17	10	82
Number of L(E)C_{50} data	15	12	7	63
Number of tax. groups	5	5	1	4
Lowest effect value (mg/l)	0.5	2.0	25	5
Lowest L(E)C_{50} (mg/l)	32	9.6	2470	2.6
PNEC (mg/l) using assessment factor of 1000	0.032	0.0096	2.5	0.0026

The assessment factor approach results in relatively low PNEC values. For barite the PNEC is a factor of 15 lower than the lowest effect level observed by Cranford *et al*., (1999), which can already be considered as an extreme low value.

As a second approach species sensitivity distributions (SSDs) were based on EC_{50} values for barite, bentonite, attapulgite and WBMs. (Figure 11). The mean (Xm) of the SSD curves represents the position of the distribution on the x-axis and the standard deviation (Sm) determines the slope of the curve. In terms of the sensitivity of species, the Xm gives an indication of the *mean concentration for the physical effects* of suspended mud particles to marine species. The Sm represents the *interspecies variation in sensitivity* of suspended WBM particles for marine species.

Table 5 provides an overview of the data used to construct the SSDs. For attapulgite only fish data was available. Therefore the SSD cannot be considered as representative for general marine biota.

Table 5 *Overview of EC50 data for attapulgite, barite, bentonite and WBMs to construct the Species Sensitivity Distributions (SSDs).Xm and Sm values for the SSD are presented together with the HC5 value*

Type of weighting material	**barite**	**bentonite**	**attapulgite**	**WBMs**
Number of EC_{50} values	20	12	8	63
Number of species with 1 or more EC_{50} values	15	12	7	13
Xm	8.01	7.51	9.22	8.81
Sm	3.05	3.25	2.70	1.05
HC_5	20.0	8.8	1800	79.6
95% confidence interval around HC_5	1.20-100	0.25-58.1	287-3841	5.01-364

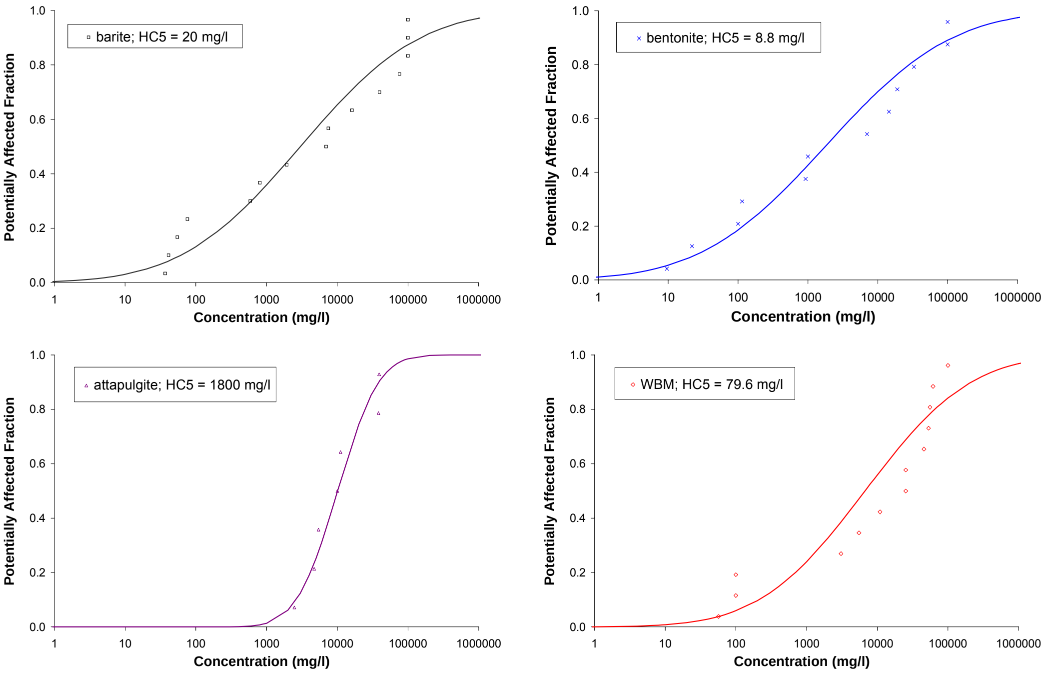

Figure 11 *Effects related to risk distributions for barite, bentonite, attapulgite and WBMs at acute exposure (log-transformed).*

As the SSD for attapulgite is based on fish data only the HC_5 value is not representative for all marine biota. The SSDs of Figure 11 shows that based on the available data species are more sensitive for barite and bentonite than for WBM. However, when statistical analysis (ANOVA, significance level 5%) was performed on the effect concentrations of these WBM particles, no significant differences between effects could be determined (Smit *et al.*, 2006a).

In order to transform the HC_5 value based on acute effect data (EC_{50}) to a PNEC, assessment factors need to be applied (Posthuma *et al.*, 2002; Pennington, 2003). A factor of 10 should be applied to account for the translation from EC_{50} level to no-effect level. A second value of 10 should be applied to extrapolate from acute effects to chronic effects. It can be discussed whether this factor is necessary. It is unclear if chronic physical effects of WBM are likely to occur as the increased turbidity and sedimentation of SPM from discharged WBMs might only have a temporary impact on organisms. Another factor of 10 could be applied to translate laboratory effects to field effects. This factor is also under discussion while most of the data results from non standardised test carried out under semi-field conditions.

To the derived HC_5 levels one assessment factor is applied for the translation from EC_{50} to NOEC level. As mentioned above the relevance of the acute to chronic translation and the lab to field translation can be questioned for this data, only one assessment factor of 10 is applied for these two translation steps. Because of the lack of data on more taxonomic groups an additional assessment factor of 10 is applied to the HC_5 for attapulgite. This results in PNEC levels for barite, bentonite, atapulgite and WBM of 0.2, 0.09, 1.8 and 0.8 respectively (Table 6).

The proposed PNECs for the weighting agents are all lower than the lowest observed effect levels as presented in table 4. For barite the PNEC is a factor of 2.5 lower than the 0.5 mg/l value determined by Cranford *et al*. (1999). The PNEC values for barite, bentonite and WBM derived from the HC_5 are higher than the levels determined with assessment factors. For attapulgite the values are comparable.

Table 6 Overview of assessment factors applied to the HC_5 to derive the PNEC level

Type of weighting material	**barite**	**bentonite**	**attapulgite**	**WBMs**
HC_5 (mg/l)	20.0	8.8	1800	79.6
Proposed assessment factors				
EC_{50} to NOEC level	10	10	10	10
Lab to field & acute to chronic translation	10	10	10	10
Lack of data on different taxa	-	-	10	-
PNEC (mg/l)	***0.20***	***0.088***	***1.8***	***0.8***

When PNECs are derived from SSDs, these values (0.2; 0.09; 1.8 and 0.8 mg/l for barite, bentonite, attapulgite and WBMs respectively) are less conservative and more in line with observed effect levels in the field and in field-relevant exposures. It is suggested to use these values in the calculation of the EIF for drilling discharges. An extended description of the data and PNEC derivation for weighting agents in drilling mud is provided in Smit *et al.* (2006a).

5.2 Effects in the sediment

5.2.1 Chemicals

Substances that are highly hydrophobic or insoluble may be assessed as of low risk for pelagic fauna but can accumulate in sediments to concentrations at which they might exert significant toxic effects. The sediments may act as a permanent sink for highly hydrophobic or insoluble substances that can accumulate in sediments to high concentrations. According to the EU-TGD 'marine paragraph', the general principles as applied to data on aquatic organisms, also apply to sediment data. For sediment risk assessment purposes it is also acceptable to use freshwater effect data alone or in combination with saltwater effect data.

Assessment factors
If results from whole-sediment tests with benthic organisms are available, the PNEC sediment must be derived using assessment factors. Only whole sediment tests with infaunal and epibenthic organisms are considered suitable for use in risk assessment of the marine sediments compartment. No fully internationally accepted, standardised test methods for testing the toxicity of whole sediment are currently available. Due to the generally long-term exposure of benthic organisms to sediment-bound substances, long-term tests with sub-lethal endpoints (like reproduction, growth, emergence, sediment avoidance, and burrowing activity) are regarded as most relevant. The TGD provides an overview of assessment factors applied to acute and chronic whole sediment toxicity tests as presented in Table 7 and Table 8, respectively.

In contrast to the concept applied to the pelagic marine compartment, it is only necessary to have results from one acute sediment test for the assessment factor of 10000 to apply. Furthermore, if only results from short-term tests with freshwater sediment-dwelling organisms are available (at least one), an assessment factor of 10000 also is applied to the lowest value. The $PNEC_{marine\ sediment}$ should also be calculated from the $PNEC_{saltwater}$ using the equilibrium-partitioning method. The lowest PNEC from both approaches is used for further assessment.

Table 7 Assessment factors for derivation of PNEC marine sediment, from short-term sediment toxicity tests (TGD approach: EC, 2003).

Available test results	**Assessment factor**	**$PNEC_{marine\ sediment}$**
One acute freshwater or marine test	10000	Lowest of LC50 /10000 and equilibrium partitioning method
Two acute tests including a minimum of one marine test with an organism of a sensitive taxa	1000	Lowest of LC50 /1000 and equilibrium partitioning method

If, in addition to the results of tests with freshwater benthic organisms, a result from an acute toxicity test with a marine benthic organism (preferably representative of the same taxon that is most sensitive in aquatic freshwater or saltwater tests) is available, then an assessment factor of 1000 is applicable. A PNEC $_{\text{marine sediment}}$ is derived by application of the following assessment factors to the lowest LC_{50} value from acute tests. Once again a PNEC $_{\text{marine sediment}}$ should also be calculated from the $PNEC_{\text{seawater}}$ using the equilibrium partitioning method. A further reduction of the assessment factor is only permitted if results from long-term tests with sediment-dwelling organisms are available. A PNEC $_{\text{marine sediment}}$ is derived by application of the following assessment factors to the lowest NOEC/EC10 value from long-term tests (Table 8):

Table 8 Assessment factors for derivation of PNEC $_{\text{marine sediment}}$ from long-term sediment toxicity tests (TGD approach: EC, 2003).

Available test results	**Assessment factor**
One long term freshwater sediment test	1000
Two long term freshwater sediment tests with species representing different living and feeding conditions	500
One long term freshwater and one saltwater sediment test representing different living and feeding conditions	100
Three long term sediment tests with species representing different living and feeding conditions	50
Three long term tests with species representing different living and feeding conditions including a minimum of two tests with marine species	10

Equilibrium partitioning

In absence of any ecotoxicological data for sediment-dwelling organisms the EU-TGD (EC, 2003) allows a provisional calculation of $PNEC_{\text{sediment}}$ by use of the equilibrium partitioning method. Since the assessment factor approach could not be applied to the ecotoxicological data obtained in the literature review for calculation of $PNEC_{\text{sediment}}$, the equilibrium partitioning approach was evaluated. Additionally, when only acute toxicity data with benthic organisms are available the equilibrium partitioning method also should be applied for determination $PNEC_{\text{sediment}}$ values as a screening approach, according to TGD. The lowest PNEC derived from use of assessment factor or the equilibrium method should be used for the risk calculation. In accordance with the TGD, a PEC_PNEC ratio exceeding 1 indicates that long-term testing with sediment organisms (spiked sediment) should be considered.

The equilibrium partitioning method derives sediment quality criteria (SQC) or PNECs from water quality critera by predicting interstitial water concentrations for the protection of benthic organisms. The assumptions that are made in this method are as follows:

- Sediment-dwelling organisms and water column organisms are equally sensitive to the chemical.
- Concentration of the substance in sediment, interstitial water, and benthic organisms are at thermodynamic equilibrium: the concentration in any of these phases can be predicted using the appropriate partition coefficients.
- Sediment/water partition coefficients can either be measured or derived on the basis of a generic partition method from separately measurable characteristics of the sediment and the properties of the chemical.

Natural organic substances and drilling fluid chemicals
For non-ionic organic substances the sediment quality criteria (SQC) or $PNEC_{sediment}$ is recommended derived by the following formula (US EPA, 1997, US EPA, 2003a and 2003b):

Equation 20: Calculation of SQC ($PNEC_{sediment}$) for organics - EqP approach (US-EPA, 1997)

$$SQC\ (PNEC_{sediment}) = Kp_{sediment} * WQC\ (PNEC_{produced\ water})$$

in which:

SQC = Sediment Quality Criteria ($mg.kg^{-1}$) or $PNEC_{sediment}$
$Kp_{sediment}$ = partition coefficient between sediment and water ($l.kg^{-1}$)
WQC = Water Quality Criteria ($mg.l^{-1}$)

This Equation is recommended applied for derivation of $PNEC_{sediment}$ for added drilling fluid chemicals (e.g. non-PLONOR and some PLONOR chemicals) with log K_{ow} or K_{oc} values > 3 and for naturally occurring substances (PAHs, aliphatic hydrocarbons). The Kp is the partitioning coefficient between sediments ($l.kg^{-1}$) and water, and WQC ($PNEC_{water}$) is the effects-based water quality criteria. Organic carbon appears to be the dominant sorption phase for non-ionic organic substances in naturally occurring sediments and thus controls the bioavailability in sediments (Di Toro *et al.*, 1991). Derivation of Kp values for non-ionic organic substances is suggested expressed as shown in Equation 11.

The $PNEC_{water}$ values for naphthalenes, 2-3 ring PAHs, 4+ ring PAHs and aliphatic hydrocarbons currently applied in the EIF produced water are recommended to be used for aromatic/aliphatic hydrocarbons representing potential oil residues associated with drilling discharges.

The exposure scenario of discharges related to offshore drilling operations is different from the exposure scenario referred to by the TGD. All chemical substances with log K_{ow} or K_{oc} values > 3 are assumed to deposit on the sea bed after release through processes such as agglomeration (forming new and larger particles) and/or attachment to cuttings/mud particles. These processes will cause that the chemicals sink to the sea floor immediately rather than stay suspended in the water column before settling on the seabed (Rye *et al.*, 2006a). This means that K_p related to drilling discharges is expressed as the partitioning between the deposited chemical substance and the porewater in the sediment compartment, and deviates from the TGD where K_p expresses the partitioning of the organic substances between the suspended particles and water (Equation 21). Therefore, the EqP approach applied to non-ionic organics applied by US-EPA (1997) (Equation 20) is recommended to be used for calculation of $PNEC_{sediment}$ for non-ionic organic substances, including both naturally occurring substances such as aromatic and aliphatic hydrocarbons, and added drilling fluid chemicals with log K_{ow} or K_{oc} values > 3. For added chemical substances with a log K_{ow} >5, an additional assessment factor is added to the EqP approach used, to account for ingestion of sediment.

Equation 21: Calculation of $PNEC_{marine\ sediment}$ - EqP approach (EU-TGD)

$$PNEC_{sediment} = [Kp_{susp\text{-}water}/RHO_{susp}] * PNEC_{water} * 1000$$

in which:

$PNEC_{water}$ = Predicted No Effect Concentration in seawater [$mg.l^{-1}$]
$RHO_{susp\ bulk}$ = density of suspended matter [$kg.m^{-3}$]
$Kp_{susp\text{-}water}$ = partition coefficient suspended matter water [$m^3.m^{-3}$]
$PNEC_{sediment}$ = Predicted No Effect Concentration in marine sediment [$mg.kg^{-1}$]

When using the equilibrium method to calculate the $PNEC_{marine\ sediment}$, the $PEC_{marine\ sediment}$ should be determined independent of the value of the K_{oc}. If both the PEC and the PNEC for the sediment compartment can only be assessed using the same K_{oc} or K_{ow} values, no quantitative risk characterization for the sediment should be performed because this would only result in the same risk value as for the water compartment. Under these circumstances the calculated risk (PEC_PNEC ratio) for the aquatic environment can also be used for the sediment. Since in the proposed method the Kp value for a certain substance applied for calculation of PEC and PNEC for the water column and sediment will be different, the PEC_PNEC ratio for these compartments will give different results.

Metals

The EqM-method was originally proposed to develop sediment quality criteria for organic substances, normalising the concentrations in the solid phase to the organic carbon content. The method was further modified for metals and metalloids by use of empirically derived sediment/water partition coefficients. For metals, two approaches utilising partitioning models have been proposed to relate biological effects to interstitial water and sediment concentration (derivation of sediment quality criteria):

- The Dutch approach determining $MPC_{sediment}$ based upon empirically derived $Kp_{sediment}$ values together with water quality criteria (MPA) added to the background concentration in the sediment ($Cb_{sediment}$) (Crommentuijn *et al.*, 1997 and 2000);

- The US SEM/AVS approach, a partitioning model that relates interstitial water concentration and sediment toxicity to the molar ratio of the Simultaneously Extracted Metal (SEM) to acid Volatile Sulphide Concentration (AVS) (US-EPA, 1994a and 1994b).

The similar recommendation as for non-ionic organic chemicals is made regarding the development of $PNEC_{sediment}$ or SQC for metals, except that Kp should be based upon measured (empirical) values (partitioning between sediment -water) and not estimated from f_{oc} and K_{oc}, as for non-ionic organics (US-EPA, 2003). However, it was recommended that background concentrations must be taken into account in SQC for metals which occur naturally in the environment, so the Dutch developed the “added risk approach”, which forms the basis of the recommendations to the EC. The Dutch EqP approach, applied by Crommentuijn *et al.* (1997 and 2000), is recommended for determination of the $MPC_{sediment}$ or $PNEC_{sediment}$ for metals and is outlined in Equation 22 and 23, subsequently:

Equation 22: Calculation $MPC_{sediment}$ for metals - EqP approach (Dutch RIVM - Crommentuijn *et al.*, 1997 and 2000)

$$MPC_{sediment} = Kp_{sediment} * MPA_{water} + Cb_{sediment}$$

In which:

$MPC_{sediment}$ = Maximum Permissible Concentration in marine sediment [$mg.kg^{-1}$]
$Kp_{sediment}$ = partition coefficient sediment and water [$m^3.m^{-3}$]
MPC_{water}= Maximum Permissible Concentration in the surface water [$mg.l^{-1}$]
$Cb_{sediment}$ = background concentration in the sediment [$mg.kg^{-1}$]

where $Kp_{sediment}$ is the partitioning coefficient between sediment particles ($l.kg^{-1}$) and pore water in the sediment compartment and MPA_{water} is the effects-based water quality criteria.

Equation 23: Calculation of $\mathrm{PNEC}_{sediment}$ for metals - EqP approach (Crommentuijn *et al*., 1997 and 2000)

$$\mathrm{PNEC}_{sediment} = \mathrm{Kp}_{sed.\ barite\text{-}seawater} * \mathrm{PNEC}_{water} + \mathrm{Cb}_{sediment}$$

In which:

$PNEC_{sediment}$ = *Predicted No Effect Concentration in marine sediment [mg.kg^{-1}]*
$Kp_{sed.\ barite\text{-}seawater}$ = *partition coefficient barite particles and water [m^3.m^{-3}]*
$PNEC_{water}$ = *Predicted No Effect Concentration in seawater [mg.l^{-1}]*
$Cb_{sediment}$ = *background concentration in the sediment [mg.kg^{-1}]*

The Kp related to drilling discharges is expressed as the partitioning between the deposited particles and the porewater in the sediment compartment and deviates from TGD in that $\mathrm{Kp}_{susp\text{-}water}$ expresses the partitioning between the suspended particles and water (Equation 10), and is also different from the Dutch approach that is based upon the relationship of the substance between the concentration in the solid phase in the sediment (distributions processes in the Dutch environment) and the pore water.

The $\mathrm{PNEC}_{sediment}$ ($\mathrm{MPC}_{sediment}$) is calculated using Equation 23, applying the Dutch approach of EqM-method described by Crommentuijn *et al*. (1997 and 2000). The $\mathrm{PNEC}_{sediment}$ values are based upon measured $\mathrm{Kp}_{barite\text{-}seawater}$ values for the sediment together with water quality criteria (PNEC_{water} or MPA_{water}) added to the background concentration of metals in the sediment.

Field monitoring data

The definition of the threshold values is based on generic guidelines assuming that most sensitive species should be protected. The level of these literature based thresholds are compared to levels in the field where (no)effects of the specific stressors has been observed, so-called field derived threshold effect levels (F-TEL). Monitoring data will be used in order to crosscheck the PNEC values for the sediment derived from literature with alternative PNEC values derived from monitoring data (F-TEL). Two different approaches are followed in this procedure:

1. Identification of locations where the abundance of selected species is reduced by 50% compared to the reference stations. The concentration of several contaminants at these locations is plotted in field-based SSDs. The concentration level where 5% of the selected species are affected is defined as a field based threshold level. Leung *et al*., (2005) and Bjørgesæter (2005) describe the procedure and results in detail.

2. Identification of locations with the highest concentrations of selected contaminants where no significant change in the abundance of selected

species was observed compared to reference stations. Brakstad & Trannum (2005), Grung *et al*., (2005a) and Grung *et al*., (2005b) describe the procedure and results in detail.

Frost *et al*. (2006) provides an overall overview of the different sets of PNECs for different toxic components (groups).

5.2.2 Burial

The potential risk of cuttings contaminated with Water Based Mud (WBM) residues (inert clay, bentonite and barite) settling onto the seabed has been primarily explained by the temporary effects of physical burial of benthic fauna (Daan & Mulder, 1993). A dedicated study to the nature and effects of burial was carried out within the ERMS framework. Results are reported by Kjeilen-Eilertsen *et al*. (2004).

The following factors that determine the effect of burial on species are mentioned (Maurer *et al*.; 1980, Kranz, 1974 and Baan *et al*., 1998):

- Depth of burial;
- Tolerance of species (life habitats, escape potential, degree of mantle fusion and siphon formation, low oxygen tolerance);
- Rate of burial;
- Nature of material (grain size different from native sediment);
- Temperature (mortality rate by burial higher in summer than winter).

Effect data describing the specific impacts related to those factors separately is not available. Only for depth of burial some diffuse data is available for a number of species (Kjeilen-Eilertsen *et al*., 2004). Therefore assumptions have to be made to predict a scientifically sound threshold for burial effects. Besides that, burial can also lead to a chain of other stressors on benthic species communities like oxygen depletion and high sulphide concentrations. These processes are acknowledged (see also Beardsley & Neff, 2004) but not considered in this part, which is to describe the burial-effects only.

Kjeilen-Eilertsen *et al*. (2004) present results of several effect studies related to burial. Results of these studies are mostly expressed as the escape potential (EPn). This potential of a given species can be identified as the probability (n) that the organism will escape a given depth of burial (Kranz, 1974). The threshold values in Kjeilen-Eilertsen *et al*. (2004), with reference to Kranz (1974), are the EP10 values by burial with both exotic and native sediment. Other information is mainly based on studies by Maurer *et al.* (1980; 1981; 1982) and Bijkerk (1988).

A statistical description of the variation in sensitivity (Species Sensitivity Distributions) (SSD) is applied to derive the threshold value. The HC_5 (exposure where 5% of the species are effected) can serve as an intermediate for the threshold level (PNEC) (Smit *et al.*, 2005a). This methodology, referred to as 'statistical extrapolation methods' facilitates probabilistic risk assessment and is incorporated in the TGD (EC, 2003).

The reported data indicate that species are more sensitive to burial by exotic sediment than native sediment. A SSD for burial by exotic sediment is drawn, assuming a log-normal distribution (see Figure 12). The HC_5 for burial by exotic sediment is determined at a level of 0.65 cm (5 - 95% conf. interval of 0.32 – 1.07 according to the method described by Aldenberg & Jaworska (2000)).

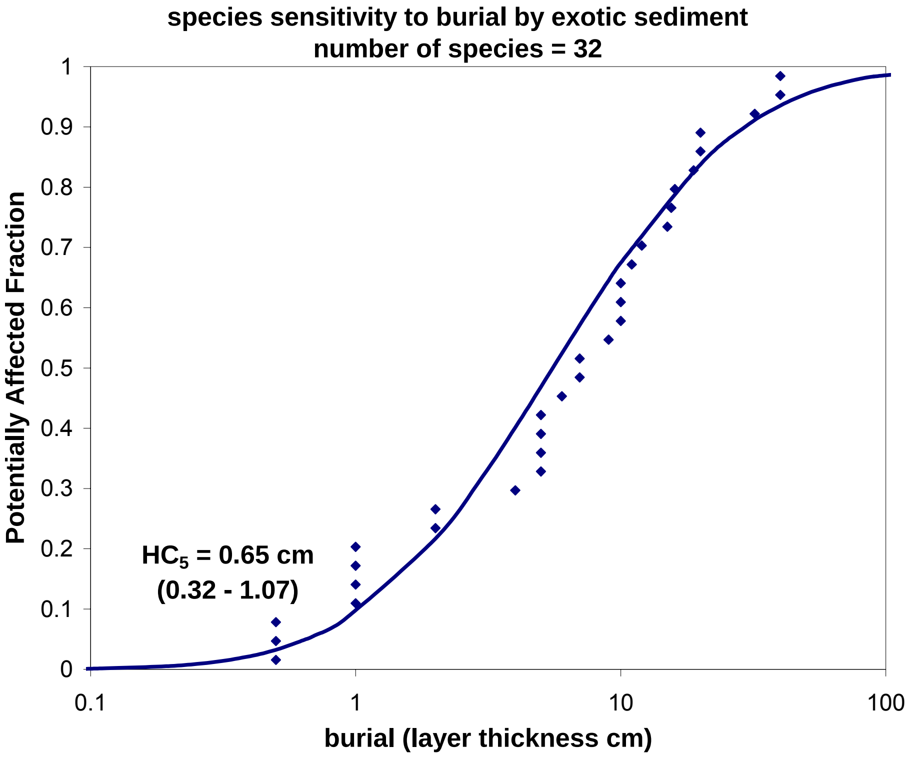

Figure 12 Species Sensitivity Distribution (SSD) of benthic species for burial by exotic sediment, only. Data reported by Kjeilen-Eilertsen et al. (2004).

The level of the PNEC for burial can be derived from the HC_5 by applying safety factors (Posthuma *et al.*, 2002; Pennington, 2003). When the SSD is based on chronic NOEC values no safety factor needs to be applied (except if the number of NOECs is low or the NOECs are not reliable a safety factor between 1 and 5 could be argued for (EU-TGD)). When the SSD is based on other data than NOECs, the same rationale as used for the assessment factor approach (EU-TGD) can be followed; for toxicity a safety factor of 10 is used to go from acute to chronic exposure. A second factor of 10 is applied to extrapolate laboratory data to field data and another factor of 10 to go from effect level to no-effect level. The marine part of the EU-TGD (EC, 2003) even prescribes an additional safety factor of 10 to account for specific sensitive species in the marine environment. The data presented here are chronic *effect* levels. Following the rationale from EU-TGD, this would imply the application of at least two safety factors of 10 to the HC_5 (from effect level to no-effect level and from laboratory to field effects). However, the relevance of this safety factor approach for non-toxic stressors like burial can be questioned. In this case it was decided not to apply any safety factor to the HC_5 to derive the PNEC. There are two reasons for this:

- First, the fact that the data is based on instantaneous burial while in practice the formation of the burying layer is a slow process;
- And second, because of the difference between the exposure in the experiments (thickness of burial) and the defined stressor in the model (thickness of deposited layer) (As described by Smit *et al*., 2006b). The suggested PNEC of 0.65 cm is in the same range as the previous defined threshold level of 1 cm for non-moving sediment species (TNO, 1994).

An overview of the data and procedures applied to derive the PNEC for burial is presented in Smit *et al*. (2006b).

5.2.3 Change in sediment characteristics (grain size)

Although many studies have revealed a relationship between sediment type and infauna community structure, there is considerable variability in species responses to specific sediment characteristics. The studies suggested that the factors ultimately controlling infauna distributions may not be sediment grain size per se or factors correlated to it (such as organic content), but rather interactions between hydrodynamics, sediments and infauna and how these affect sediment distribution, larval supply, particle flux and pore water chemistry (Snelgrove & Butman, 1994). Although the complicity of these processes is acknowledged, in this model the change in median grain size is taken to represent the overall changes in sediment characteristics.

As no (standardized) tests focussing on the impact of altered grain size exist, no experimental data is available to assess a threshold for altered grain size for benthic species (Trannum, 2004). Therefore an alternative data source is used. One parameter used to describe the sediment characteristics at a specific location is median grain size. This parameter is frequently measured in field surveys. As sediment biota has a preference for specific sediments, the presence of specific species can be related to specific ranges of the median grain size. Most species occur at a range of (median) grain sizes. From monitoring data this range of median grain sizes is obtained for species occurring at more than one sample location. These data is used to derive the sensitivity of species to changes in median grain size.

The observed range of median grain sizes per species is defined as the “grain size window-of-occurrence”. This window-of-occurrence is described by an average value and variation (95 percent interval around this average) for the median grain size. This variation is expressed as the range including 95 percent of the observations of this species. Species with a small window-of-occurrence are more sensitive to changes in grain size than species with a wide window-of-occurrence. The window-of-occurrence can serve as a measure of the sensitivity of species to changes in grain size. The information is collected from a review of benthic surveys in the Dutch sector of the North Sea, Norwegian Sea and Barents Sea. The width of the windows-of-occurrence for 246 different North Sea and Norwegian Sea species is determined as well as for 147 Norwegian Sea and 245 Barents Sea species (Trannum, 2004).

The data set containing species occurring at more than 10 locations is used to derive a threshold value for changes in grain size. Based on the *absolute* width of windows-of-occurrence for 300 species a Species Sensitivity Distribution is constructed describing the spread in sensitivity of biota to grain size changes, assuming a log-normal distribution (Figure 13).

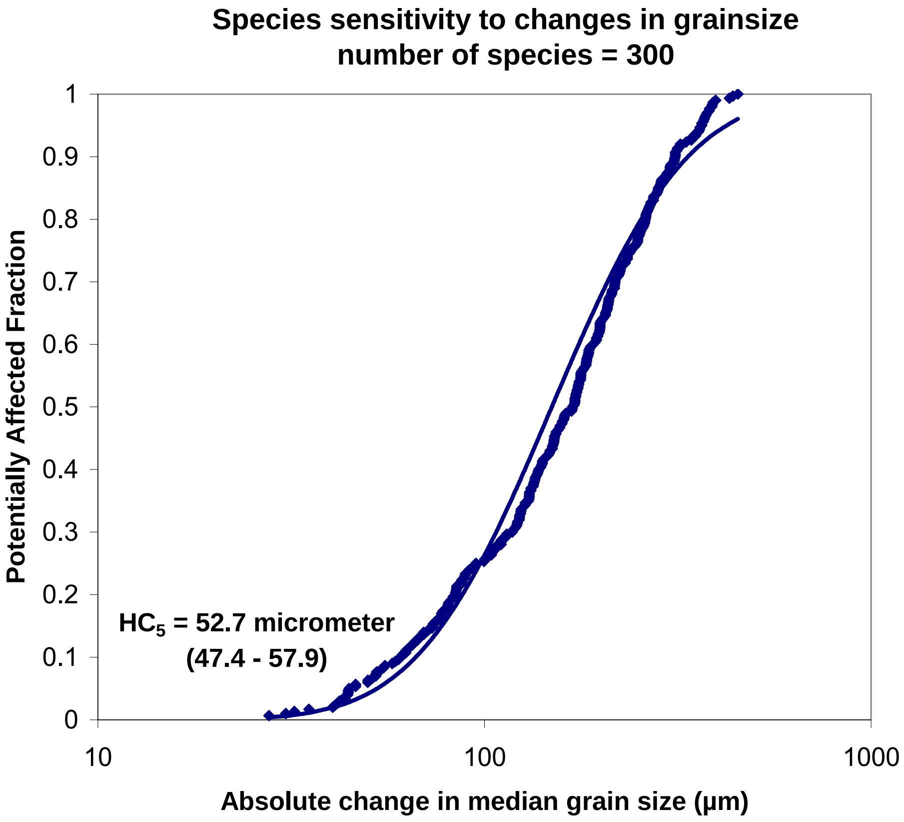

Figure 13 Species Sensitivity Distribution (SSD) based on the absolute natural grain size window-of-occurrence (95% interval) of 300 North Sea, Norwegian Sea and Barents Sea species.

From the sensitivity distribution presented in Figure 13, the probabilistic value at which 5% of the species are likely to be affected (HC_5) can be derived. This value of 52.7 µm can serve as an intermediate value for the PNEC for changes in grain size. The confidence interval around this value (47.4 – 57.9) is calculated according to the method described by Aldenberg & Jaworska (2000).

The SSD presented in Figure 13 together with the level of the HC_5 were discussed among the experts in the different ERMS-working groups before deciding on the use of this SSD as a risk curve to be included in the EIF-drilling model. Taking into account the number of species and the origin of the data (field data from monitoring studies) the application of safety factors on this value of HC_5 was judged as not relevant. Therefore, the HC_5 (52.7 µm) of this SSD is defined as the threshold value for changes in grain size.

An overview of the data and procedures applied to derive the PNEC for changes in grain size is presented in Smit *et al*. (2006b).

5.2.4 Reduction of the oxygenated layer

As described by Beardsley & Neff (2004) the most realistic way to present the stress of reduced oxygen (‘PNEC’) in the sediment would be the reduction of the total oxygen content in the upper sediment layer (RPD- Redox potential Discontinuity). Therefore the ‘PEC’ for oxygen depletion is expressed on the basis of the integrated oxygen content over depth (actually the relative change in the integrated concentration) (See chapter 4 of this report and Rye *et al*., 2006a). In order to relate the change in oxygen content to species sensitivity, a relationship between the oxygenated sediment layer, and species richness needs to be constructed. Most of the effect data cannot be used for that purpose, because it is expressed on the basis of an absolute minimum concentration in pore water (mg O_2 l^{-1}) (Beardsley & Neff, 2004).

Sediments having oxygen-depleted overlying bottom water typically exhibit substantially reduced macrofaunal diversity. Within hypoxic zones the macrofauna exhibit low species richness and very high dominance of view (tolerant) species. Among the macrofauna, many molluscs, crustaceans, echinoderms, and cnidarians appear less tolerant of hypoxia than other taxa, although there are exceptions. No single taxon dominates the macrofauna of low oxygen settings although annelid species are often prevalent. Less information is available concerning the diversity responses to reduced oxygen concentrations of bacteria, small protists (nanofauna), meiofauna, or megafauna. Smaller organisms living entirely within the sediments and with no access to the surface may be confined to hypoxic or even anoxic pore-waters, even when the overlying bottom water is well oxygenated. Yet, foraminifera and a variety of larger metazoans (polychaetes, crustaceans, molluscs, echinoderms) all display abundance peaks close to hypoxic boundaries (Levin *et al.,* 2001). Organisms may have been adapted to lower oxygen in locations with high temperatures and historically reduced oxygen concentrations, or in systems with natural high demands for oxygen (Wu, 2002). An overview of effect data and threshold levels for reduced oxygen is provided by Beardsley & Neff (2004).

If the input of organic matter to the sediment is increased, the fauna will be affected due to reduced availability of O_2 and toxicity of H_2S produced by sulphate reducing bacteria. The actual O_2 concentration in sediments in presumably unaffected control sediment is low and the organisms probably depends on physiological adaptations to periodic residence in low-oxic environments and supply of O_2 from the overlying water via siphons, tubes or irrigated burrows (Schaanning & Bakke, 2005).

In order to represent the “exposure level” of oxygen, the oxygen content of the Redox Potential Discontinuity (RPD) is assumed to decrease from 100 to 0% saturation due to degradation of the organic phase. No effect data is available which directly correlate ecosystem effects with measured concentrations of *integrated* O_2 in sediment layers. However, it can be assumed, that the modelled integrated O_2 profile mimics the redox profile in the sediment (Beardsley & Neff, 2004). Schaanning & Bakke (2005) published data relating a changing redox potential to effects on the macro benthic community. With the relation between integrated oxygen and the redox potential, a bridge can be established between the model and data on effects of various organic phases on redox potentials and the macro benthic community. However, it must be kept in mind that it is only a correlation between the change in community and the change in the redox potential. There might be other factors present that also influence the community structure (e.g. toxicity).

It is not possible to use the threshold data described in Beardsley & Neff (2004) for the derivation of an oxygen threshold for the sediment expressed as integrated oxygen concentration over the RPD. The effect data that is available mainly presents absolute oxygen concentrations related to (pore)water concentrations. This effect data cannot be related to the predicted reduction in the RPD layer. As the study described by Schaaning & Bakke (2005) provide the only (indirect) link between oxygen content, redox potential and species diversity, the results of this study were taken to derive a ‘PNEC’ for oxygen depletion. The observed lowest relative change in the E_h without affecting bentic diversity was 20% (Schaanning & Bakke, 2005). To follow the assumption that the redox potential mimics the oxygen profile, the ‘PNEC’ for oxygen can be set to the same value as the maximum change in E_h where no effects on the benthic community were observed. Therefore a maximum allowable change in the total oxygen content of the RPD is set to 20%. This value is in the line of what is expected by the experts in this field (pers. comm. J. Neff, Battelle)

Also the risk curve for changes in the oxygenated layer cannot be build upon effect data from literature. As long as no data is available a theoretic risk curve is constructed based on the assumptions that the reduction of the oxygenated layer is expected to be more or less linear related to the risk of oxygen depletion. As the 5% risk level corresponds to 20% reduction (threshold level); 50% risk should correspond to a value near the geometric mean of 20% and 100% reduction. At a

100% reduction of the oxygenated layer the risk value will also approach the 100% PAF level (potentially affected fraction). Based on these assumptions a theoretical risk curve is constructed (Figure 14) for modelling the risk of oxygen depletion.

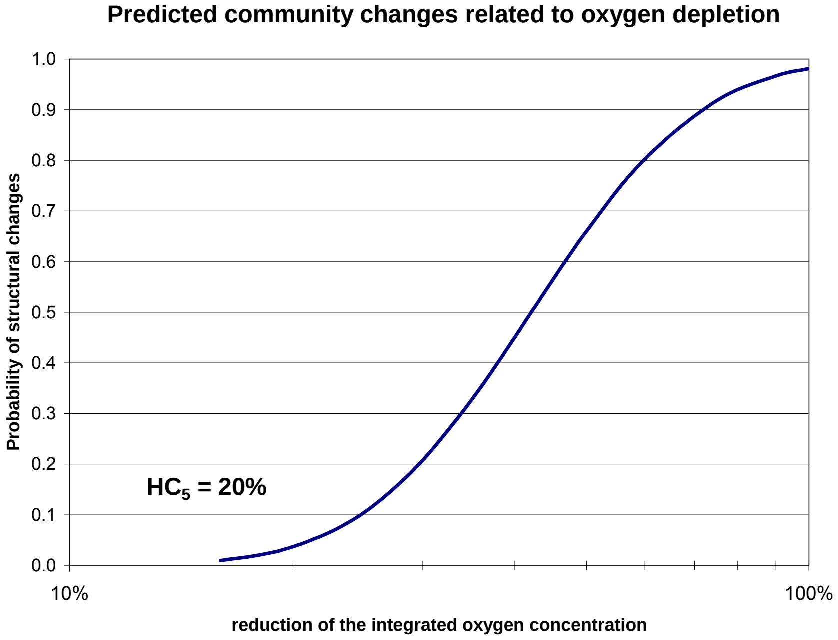

Figure 14 Theoretic risk curve for the reduction of the thickness of the oxygenated layer

The 20% value is a generic level. It is more determined by "expert opinion" (Battelle, TNO and NIVA) than it is related to sound effect data and levels. No quantified significance level for this value can be provided (which sediments, which type of cuttings, which thickness of deposited layer, which temperature, etc.). The fact that normal effect-data could not be applied is a direct result of the chosen way to describe oxygen stress (as described by Beardsley & Neff (2004)). As the 20% value is considered to be a realistic value for a threshold level for hypoxia by several experts, it was decided to apply the theoretic SSD as a risk curve in the EIF-drilling model. It must be clear that the nature of the threshold level for oxygen depletion is different from the threshold levels for other stressors as described in this report. It is recognised that when, as a result of scenario modelling, oxygen depletion is indicated as a main contributor to the overall risk, further research to the relation between integrated oxygen content of the sediment and species richness might be necessary.

An overview of the data and procedures applied to derive the PNEC for changes in grain size is presented in Smit *et al*. (2006b).

6. Risk assessment and integration to EIFs

Chapter 4 and 5 describe for the defined stressors the level of exposure, the threshold values and the SSDs. This information, used in the risk assessment, forms a basis for the definition of the EIF_{DD}. As in practice the EIFs for the water column and the sediment will be calculated separately, this chapter will describe how for both compartments the EIF_{DD} will be calculated and presented.

6.1 PEC_PNEC ratio and PAF

The slope of the SSD, indicating the variation in species sensitivity, and the defined threshold level (PNEC) are both included in the calculation of the EIF_{DD}. In each grid cell and for each stressor, the PEC is compared to the PNEC resulting in a PEC_PNEC ratio. At PEC_PNEC ratios higher than 1, unacceptable effects on organisms are likely to occur. The higher the ratio, the more likely it is, that unacceptable effects may occur.

The PEC_PNEC ratio is only an indicator of risk and for stressors with different modes of action PEC_PNEC ratios cannot directly be compared (Smit *et al.*, 2005a). SSDs provide a mean to calculate a more quantitative and comparable risk indicator: the Potentially Affected Fraction of species (PAF). This PAF value can be explained as the probability that randomly selected species is exposed to a concentration exceeding its chronic NOEC.

In order to express the risk at a given exposure concentration as risk probability (PAF), the slope of the SSD is applied. The relationship between PAF and concentration is calibrated in such a way that the PNEC corresponds to a PAF level of 5%. This 5% is chosen as a cut-off criterion. The exposure of organisms to substances in their aquatic environment is considered acceptable is case less than 5% of the species is at risk (e.g. Straalen van & Denneman, 1989, Aldenberg & Slob, 1993; Newman *et al.*, 2000; Hoeven van der, 2001; EC, 2003). This is inline with the procedure to derive a PNEC from a SSD based on chronic NOECs (with the PNEC being the 5th percentile of a SSD based on chronic NOECs; See Figure 15).

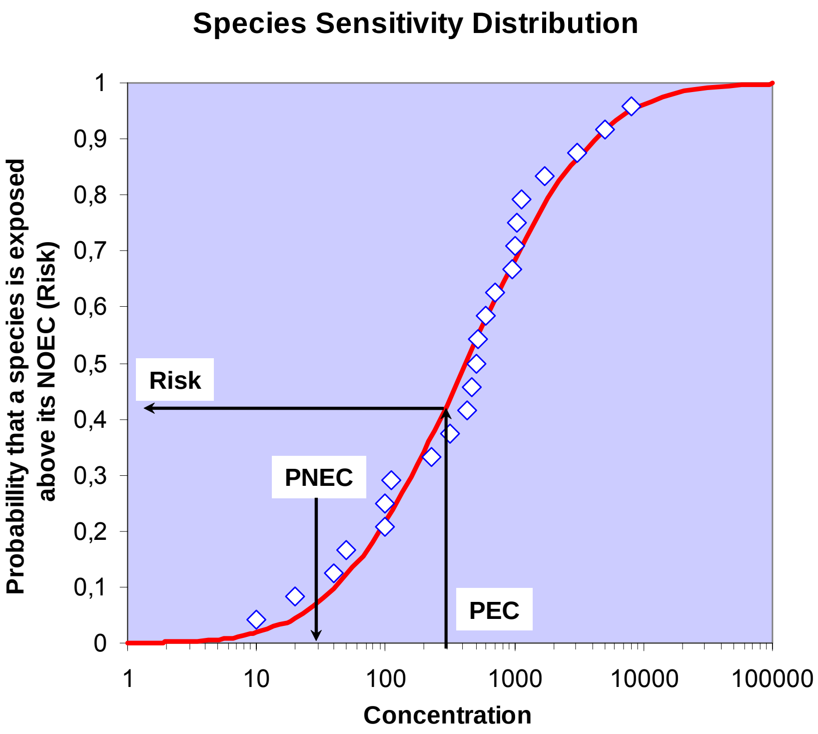

Figure 15 *Use of PNEC and variation of species sensitivity for translating PEC values to risk values. The PNEC level corresponds to a probability value (PAF) of 5%.*

As mentioned above it can be defined that when the PEC is equal to PNEC (PEC_PNEC ratio =1); the probability that a random species is effected by the toxicant is equal to 5% (the risk on adverse effects equals 5%). With the variation of species derived from the SSD, the risk probability can be derived at any level of exposure and, therefore, at any PEC_PNEC ratio. The relationship between the PAF and the PEC_PNEC ratio can be derived from the relation presented in Figure 15. The relation between the PEC_PNEC ratio and the PAF is referred to as a PEC_PNEC-to-risk curve (Smit *et al*., 2005a).

For the determination of the PEC_PNEC-to-risk curve the same characterisation of the variation in species sensitivity is applied (the Sm from the SSD). The Xm of the curve (mean position) is obtained by shifting the distribution with this Sm until an x-value of 1 corresponds to a risk of 5%. Figure 16 presents an example of a PEC_PNEC-to-risk curve.

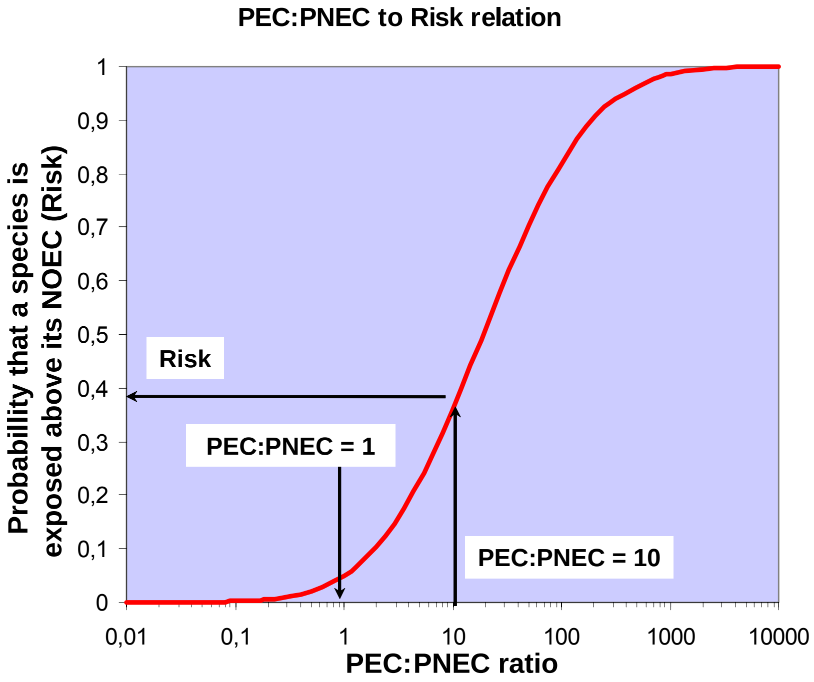

Figure 16 PEC_PNEC-to-risk curve; A PEC:PNEC ratio of 1 corresponds to a probability value (PAF) of 5%.

The relationship between PEC_PNEC ratio and PAF can be calculated according to the following formula (Equation 24):

$$PAF = \int_0^{\ln PEC:PNEC} \left\{ \frac{1}{S_m * \sqrt{2 * \pi}} * e^{\frac{-(\ln PEC:PNEC - X_m)^2}{2 * S_m^{\ 2}}} \right\} \text{..........................Equation 24}$$

in which:
PAF = risk (probability that a species will be affected)
Xm = mean of the distribution for which PEC_PNEC ratio =1; risk = 5%
Sm = standard deviation of the logarithmically transformed data
PEC = exposure concentration

For all stressors PAF levels will be calculated corresponding to the predicted levels of exposure per grid cell. An important challenge in the development of the EIF_{DD} is in deriving a scientifically robust approach for combining risks for, not only different chemicals, but for the combination of toxic and non-toxic stressors associated with drilling impacts. For combining risks for different toxic chemicals, it may be argued that additivity of effects is a pragmatic working assumption. There is some evidence that, as the number of chemicals in a mixture increases; the more likely the mixture is to approximate additivity (Warne & Hawker, 1995).

The total sum of the risks (joint risk probability or multi stressor PAF (msPAF) for all stressors per grid cell is calculated assuming independent action. For the total risk for two stressors, the sum of risks can be calculated from the formula:

$$msPAF(A+B) = PAF(A) + PAF(B) - PAF(A) * PAF(B) \qquad \text{Equation 25}$$

where PAF(A) is the risk probability or environmental risk value for stressor A and PAF(B) is the risk probability or risk value for stressor B. For a larger number of stressors, the sum of risks can be calculated from the generalized formula for the sum of probabilities PAF(A), PAF(B), ….. PAF(i) (Equation 26):

$$\text{msPAF(sum i)} = 1 - \prod_i \{1 - PAF(i)\} \qquad \text{Equation 26}$$

If the value of the msPAF in a grid cell exceeds a value of 5%, the volume or area of the grid cell is included in the calculation of the EIF.

There are limitations to the use of SSDs in risk assessment when one considers the lack of ecosystem dynamics, such as food web relationships, incorporated into the assessment model, with the major focus at the species level of organisation. In other words: "How representative is the PAF for the actual risk in the field?" Besides that the question was raised how representative the selected species are, on which the SSD is based, for specific environments (Forbes & Calow 2002a and 2002b). The challenge that still remains for ecologist and ecotoxicologists is the definition of what effects on the ecosystem are acceptable or unacceptable in relation to the most sensitive endpoints on the species level. Therefore, developments in risk assessment models should focus on the translation from laboratory species to field communities. In addition, these uncertainties in the risk assessment procedure should always be stated clearly (Calow & Forbes, 2003).

In this project the PAF represents the ecological risk. An important advantage of the use of the PAF-levels is the fact that it facilitates the combination of resulting risks from the different stressors into one probabilistic risk value (msPAF; multi stressor PAF) (Smit *et al.*, 2005a). However, it should be acknowledged that the approach of combining PAFs for different chemicals not only assumes that effects are additive, but that the relationship between PAF and actual risk is similar for different stressors. Here it is assumed that the relation between PAF and actual risk is the same for all stressors under consideration.

6.2 EIF$_{DD}$-water

During and shortly after discharge, chemicals and particles from the drilling mud will be present in the water column. These exposures may lead to effects in the water column as a result of toxicity of the chemicals and/or as a result of an increase in suspended matter concentrations. For the calculation of the EIF$_{DD}$-water both risks will be calculated and combined. The procedure, as described in section

6.1, is applied. This is the same procedure as applied for the calculation of the EIF_{PW} (Johnsen *et al.*, 2000). The only add on to this procedure is that suspended matter is added to the stressors in the same way as an additional toxicant group would be added. For each time and for each grid cell, the concentration of the selected toxicants and the suspended matter is calculated. With the application of threshold levels and SSDs the exposure values are translated into risk values expressed as potentially affected fractions (PAFs). These values indicate the probability that a randomly selected species is exposed above its threshold level. The sum of the risks (joint risk probability or multi stressor PAF (msPAF) for all compounds in the discharge is calculated assuming independent action.

If the value of the msPAF in a grid cell exceeds a value of 5%, the volume of the grid cell is included in the calculation of the EIF_{DD}-water. For the water soluble drilling chemicals in the water column, the EIF_{DD}-water is defined in the same way as for produced water: The water volume where the msPAF > 5%, divided by a unit volume equal to (100m x 100m x 10m =) 10^5 m^3 of recipient water. In addition, the EIF water volume is adjusted upwards by a factor of two for those compounds that have a low biodegradation factor in combination with a high bioaccumulation potential. Details are given in Johnsen *et al.* (2000).

Although the same calculation rules for the risks and the EIF_{DD}-water are adapted compared to the EIF_{PW}, some aspects are different (E.g. the time variability of the EIF_{DD}-water). In reality, the drilling is carried out in time intervals with no drilling in between. These time intervals between the actual drilling periods will be neglected in the calculations. In this way, the time duration aspect is taken care of. Therefore, not only the actual (variation of) EIF_{DD}-water should be reported, but also the *duration* of the period where the EIF_{DD}-water is larger than zero (See also Rye *et al.*, 2006a). The EIF_{DD}-water will be presented as a time series of EIF values for the duration of the discharge and the time that risks are present in the water column. The highest EIF_{DD}-water during the simulation period indicates the worst case situation. For this highest EIF_{DD}-water a pie chart can be presented as shown in Figure 17.

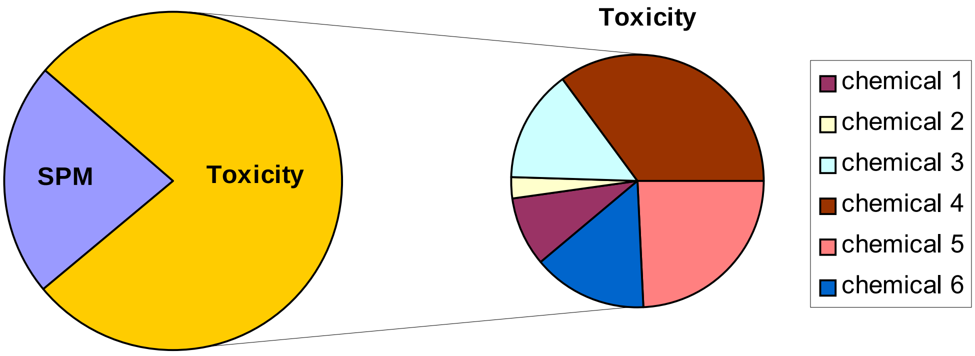

Figure 17 Graphical presentation of the EIF_{DD}-water.

6.3 EIF$_{DD}$-sediment

After drilling activities and related discharges, drilling mud and cuttings have mainly settled on the sea floor. It will remain there for a relatively long period posing a potential risk to the original fauna in the sediment. Risks of muds and cuttings may occur as a result of the toxicity of chemicals (from settled particles), by a change in the sediment structure and quality or by depletion of oxygen in the sediment. The sum of all risks should be reflected in an integrated EIF$_{DD}$-sediment.

For each time step and for each sediment grid cell, the concentration of the selected toxicants, the oxygen profile, the median grain size and the deposited layer thickness is calculated. With the application of the dedicated threshold and variation in species sensitivity, the exposure values are translated into risk values (PAFs). These values indicate the probability that a randomly selected species is exposed above its threshold level. For each stressor in the sediment this will lead to a risk area. The area where the PAF exceeds the 5% can be graphically presented (Figure 18).

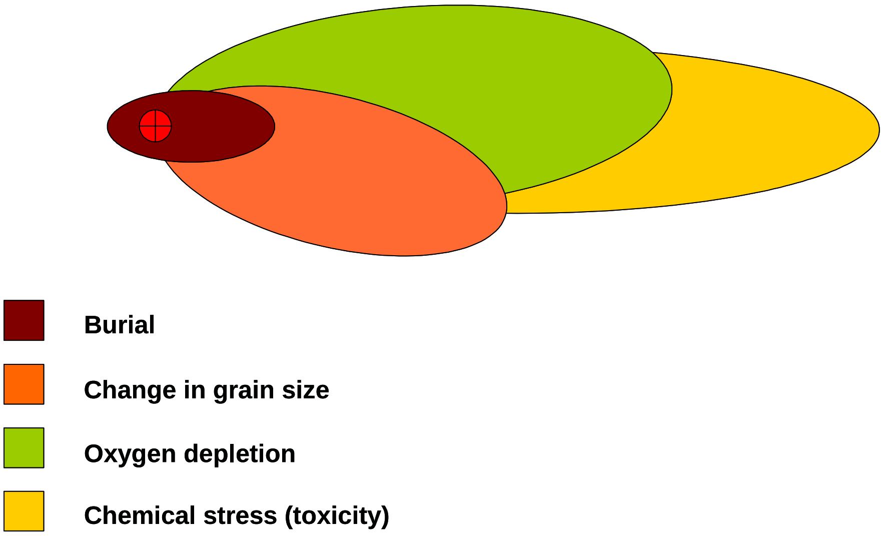

Figure 18 Graphical presentation of the areas where the PAF value for the different selected stressors exceeds 5%.

The final EIF$_{DD}$-sediment will be determined by the overall area were the combination of the risk probabilities (msPAF) exceeds 5%. It is recognized that the defined stressors in the sediment are not independent from each other. For example a change in oxygen concentrations could influence the availability of toxicants in the sediment. For the time being these interactions and synergistic or antagonistic stress mechanisms are not taken into account (Smit *et al.*, 2005a). The sum risk (joint risk probability) for all stressors (toxic and non-toxic) is calculated assuming independent action.

EIF$_{DD}$-sediment is defined as the sediment surface where the msPAF exceeds 5%, divided by a unit area equal to (100m x 100m) 10^4 m^2 of recipient sediment. In the future decisions can be made on the use of weight factors in order to discriminate between the severities of impacts from the different stressors.

The time of exposure in the sediment compartment is much longer compared to the exposures in the water column. This however also depends on the nature of the stressor. Due to biodegradation concentrations of toxic components might deplete while the mixing of deposited particles with the original sediment can cause a permanent change in the sediment structure at the discharge location. This implies that not only the value of the EIF$_{DD}$-sediment will vary (reduce) over time but also the contribution of the different stressors to the overall risk will vary. Therefore, not only the actual (variation of) EIF$_{DD}$-sediment should be reported, but also the *duration* of the period where the EIF$_{DD}$-sediment is larger than zero and the variation in the contribution to the EIF$_{DD}$-sediment (Figure 19) (See also Rye *et al.*, 2006a).

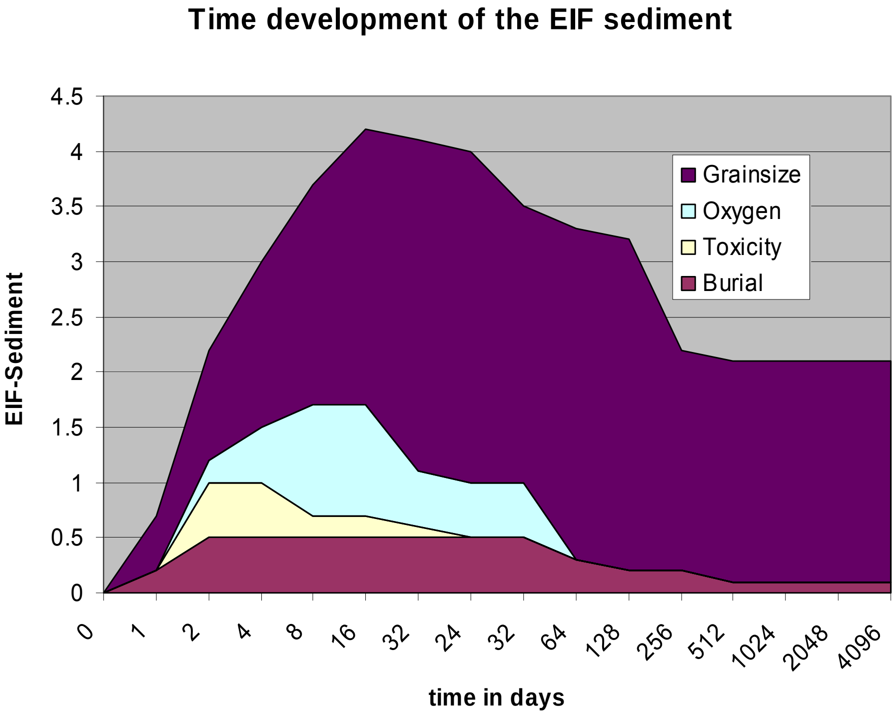

Figure 19 Time development of the EIF$_{DD}$-sediment together with the contribution of the four defined stressors to the EIF.

The highest EIF$_{DD}$-sediment during the simulation period indicates the worst case situation. For this highest EIF value a pie chart can be presented as shown in Figure 20.

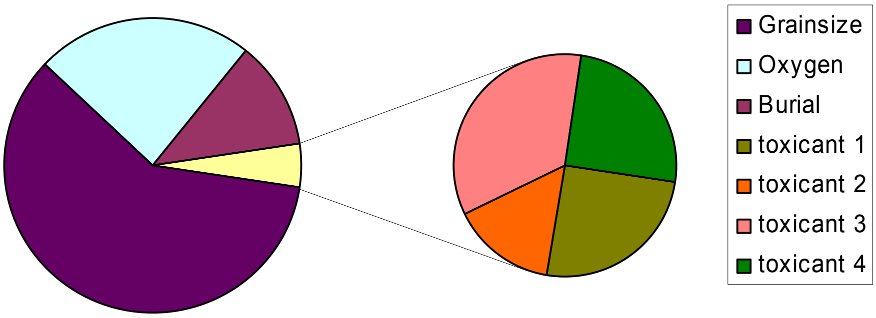

Figure 20 Contribution to the EIF_{DD}-sediment of the different stressors at the maximum EIF level. Note that the toxic stress can be assigned to different toxicants.

6.4 From EIF_{DD}-water and EIF_{DD}-sediment to EIF_{DD}

As described earlier, the EIF_{DD} consists of two parts; a water volume in which the joint risk probability for exposure to toxicants and suspended matter exceeds the 5% level. And, the sediment surface area where the joint risk probability for exposure to toxicants, changes in grain size, oxygen depletion and burial exceeds the 5% level.

The overall objective of environmental management for offshore practices is to reduce the EIF_{DD}. As long as we have two values for the EIF (one for water which is related to acute effects and one to the sediment which is related to chronic effects) a way of weighing the two values should be defined. Focus should be on a reduction of both the time scale as well as value of the EIF.

Discharging close to the sediment floor could reduce the acute EIF for the water column effects. This may, however, result in a high chronic EIF for the sediment compartment. In contrast, discharging close to the water surface would reduce the EIF for the sediment, but increase the EIF for the water column. This dilemma indicates that both EIFs should be compared in a quantitative way.

It is yet unclear how the EIFs should be compared. A complication is that the EIFs differ in their expression (volume vs. area), and in time-scale (acute vs. chronic). It is suggested first to gain experience with the behaviour of both EIF values that form the EIF_{DD} (Rye & Ditlevsen, 2005). The need of a comparison frame work is decided upon later.

7. Recommendations

Several steps can be taken in order to improve and validate the model developed for the calculation of the EIF_{DD} as described in this report. Improvement of the model should focus on a (1) further improvement of calculation rules, (2) setting up experimental programs for collecting data for a better threshold estimation and (3) validation of the risk assessment results.

7.1 Improving the EIF calculation rules

The EIF_{DD} (and also the EIF_{PW}) is defined as the volume or area over which the msPAF exceeds 5%. It neglects differences in the relative magnitudes of impacts. An improvement would be to map the actual msPAF for each stressor in space, indicating the severity of possible effects in the sediment and/or water column.

The EIF_{DD} comprises an EIF for the water column and an EIF for the sediment. In some cases comparison of weighting the EIFs for the two compartments will be necessary. This should be done on a case by case basis taking into account the severity of possible effects and the extent (area and/or volume). Procedures for a sound comparison of EIFs need to be developed.

Within the ERMS project several sets of PNECs for toxicity are defined. Each set has its own characteristics (e.g. uncertainty and validity). A selection of the PNECs to be applied is more a political than a scientific decision. However, it must be kept in mind that, because all stressors are compared, the procedures to derive the PNECs for the different stressors should be more or less comparable. A final selection of the PNECs for toxicity to be used for the EIF_{DD} needs to be made. Different sets of PNECs could be applied for different purposes.

The EIF_{DD} assumes that the effects of the different stressors are additive (e.g., that the amount of suspended solids in the water column does not influence the toxicity or bioavailability of chemicals in the water; that low levels of oxygen in the sediment do not influence toxicity or bioavailability of sediment toxicants). It should be considered the extent to which the additivity assumption is likely to be ‘worst-case’, whether there are situations for which it may lead to underestimates of risk, and whether there could be a way to refine the assumption. In order to do so, the assumed relationship between PAF and risk needs to be validated for all stressors. This can be done in an experimental program focussing on multi-stress situations in combination with additional literature studies.

7.2 Experimental program

Given that the number of stressors to be considered in the EIF_{DD} is relatively small, it is recommended that effort be devoted to collecting further test data on relevant marine species so that uncertainties associated with the SSDs for the different stressors can be reduced. However, for most stressors the best available data is already applied. This implies that if additional data is required an experimental program should be setup. This experimental program could focus on the collection of effect data for burial (thickness of the deposited layer including deposition rate in stead of depth of burial), effect data for oxygen stress and combination of stressors.

Especially the risk function for oxygen needs attention. At the moment only a few effect data on the integrated oxygen content in the sediment is available. Additional data is required to reduce the uncertainty in risk function for oxygen stress, or the way to express the exposure should be reconsidered.

7.3 Validation of risk estimates

The EIF sediment is expressed as an area where the risk exceeds an acceptable level. This area can be compared to the level of impact on benthic life being expressed by the value of biological indices, derived from monitoring information (MOD). This would result in generic and/or field specific relationships between the predicted risk and the observed field effects.

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