report
ERMS

Environmental Risk Management System

Program participants: - Agip - Conoco Phillips - Exxon Mobil - Hydro - Petrobras - Shell - Statoil - Total

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ERMS Report no. 24

Environmental Risk Management System (ERMS). A summary report

I. Singsaas, M.G.D. Smit, E. Garpestad, I. Skare, K. Bakke, L.Falcao Veiga, M. Buffagni, O.A. Follum, S. Johnsen, U.E. Moltu

SINTEF REPORT

SINTEF Materials and Chemistry

Address: NO-7465 Trondheim, NORWAY
Location: Brattørkaia 17B, 4. etg.
Telephone: +47 4000 3730
Fax: +47 930 70730

Enterprise No.: NO 948 007 029 MVA

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Environmental Risk Management System (ERMS)
A summary report

AUTHOR(S)

Ivar Singsaas, Mathijs G.D. Smit, Eimund Garpestad, Ingvild Skare, Knut Bakke, Leticia Falcao Veiga, Melania Buffagni, Odd-Arne Follum, Ståle Johnsen, Ulf-Einar Moltu.

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ABSTRACT

As a response to the Norwegian authorities' requirements in 1997 of "zero discharges to sea" by the end of 2005, the operating companies on the Norwegian shelf initiated the development of the DREAM (Dose-related Risk and Effect Assessment Model) model, including the framework for calculating the Environmental Impact Factor for produced water (EIF_{PW}). The DREAM model has successfully been used for years as a basis for calculation of EIFs for produced water discharges, assisting the oil companies in choosing among different technological produced water treatment solutions. The success of the EIF_{PW} as an environmental management tool inspired the industry to initiate a Joint Industry Project (JIP), ERMS (Environmental Risk Management System), to develop a similar tool for drilling discharges: the Environmental Impact Factor for drilling discharges (EIF_{DD}). Similar to the EIF_{PW}, the EIF_{DD} was designed to be an environmental risk-based decision support tool, to assist the oil industry in establishing cost-effective mitigation measures for reducing potential harmful discharges to the marine environment.

The concept of the EIF_{DD} is based on the PEC/PNEC ratio approach as described in the European technical Guidance Document (2003) on environmental risk assessment. Its calculation is implemented in the DREAM model. The companies involved in the development are using the EIF_{DD} as a management tool to evaluate the risk connected to discharges and to evaluate the effects of alternate mitigation measures, for instance changing the chemical composition during drilling operations offshore.

KEYWORDS	ENGLISH	NORWEGIAN
GROUP 1	Numerisk analyse, Petroleumsteknikk	Numerical analysis, Petroleum technology
GROUP 2	Miljørisiko, Matematisk modellering	Environmental Risk, Mathematical modelling
SELECTED BY AUTHOR	Environmental Risk Management System	Environmental Risk Management System
	Beslutningstøttesystem	Decision support system

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Summary

As a response to the Norwegian authorities' requirements in 1997 of "zero discharges to sea" by the end of 2005, the operating companies on the Norwegian shelf initiated the development of the DREAM (Dose-related Risk and Effect Assessment Model) model and the Environmental Impact Factor for produced water (EIF_{PW}). The success of the EIF_{PW} as an environmental management tool inspired the industry to initiate the ERMS (Environmental Risk Management System) project to develop a similar EIF for drilling discharges (EIF_{DD}).

The overall objective of the ERMS Joint Industry Program (JIP) was to develop an environmental risk-based decision support tool, to assist the oil industry in establishing cost-effective mitigation measures for reducing potential harmful discharges to the marine environment from drilling discharges. The concept was based on the PEC/PNEC ratio approach as a basis for calculation of Environmental Impact Factors (EIFs), as was successfully developed for produced water discharges through the DREAM program.

For produced water discharges, toxicity in the water column was the main issue. For drilling discharges, non-toxic disturbances in seafloor sediments were evaluated in addition to toxic disturbances. A comprehensive literature study was performed to define PNEC values for toxic stressors and non-toxic disturbances from drilling discharges. Toxicity for the water column and seafloor sediments has been evaluated according to the principles described in the EU Technical Guidance Document (TGD). Threshold values for non-toxic disturbances have been developed from literature and monitoring information collected for marine species. This work is well documented in reports and papers issued as part of the program.

The DREAM model, which was the main deliverable from the DREAM program, has been further developed to include discharges from drilling operations. It has been developed along the same lines as for produced water discharges, defining water volumes and sediment areas with PEC/PNEC ratios higher than unity. Case studies have been performed by the participating oil companies as part of qualification of the model, with good results.

The DREAM model has successfully been used for years as a basis for calculation of EIFs for produced water discharges and to help the oil companies choosing between different technological produced water treatment solutions. The extended model is now being used by the companies to establish a similar basis for calculation of EIFs from drilling operations. The model will be used as a management tool in order to evaluate the environmental risks associated with discharges and to evaluate the effects from mitigation measures, for instance changing the chemical composition during drilling operations offshore.

1 Introduction

1.1 Zero discharge

Following the Norwegian authorities' requirements in 1997 of "zero discharges to sea" within the end of 2005, the operating companies on the Norwegian shelf initiated the development of a modelling tool used for guidance of management decisions for reduction of potential harmful environmental effects associated with produced water discharges. This effort was embodied in the DREAM (Dose-related Risk and Effect Assessment Model) project (Johnsen *et al*, 2000), from which the Environmental Impact Factor for produced water (EIF_{PW}) was developed. The EIF_{PW} is a relative indicator of environmental risk whose purpose is to aid the industry in the development of a "zero harmful discharge" strategy and selection of cost-benefit-based solutions.

1.2 EIF for produced water

The EIF_{PW} was developed as a decision-support tool for environmental management. Its calculation is based on internationally agreed procedures for hazard and risk assessment, as defined by the European Union (EU). 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 in a given volume of water with the predicted environmental tolerance level or the concentration giving no discernible adverse effect (potential no effect concentration, 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 (2003). In order to calculate the contribution of each chemical component to the overall risk, species sensitivity distributions (SSDs) are applied to develop explicit functional relationships to risk for different chemical groups. A description of the development of the EIF_{PW} for produced water can be found in Smit *et al*., (2003). Johnson *et al*., (2000) describe the use of the EIF_{PW} for management of produced water discharges.

1.3 Goal and description of ERMS

The success of the EIF_{PW} as an environmental management tool has inspired the industry to pursue a more holistic environmental risk assessment methodology, able to address a variety of discharges to both atmospheric and marine recipients. As one step in this development process, the ERMS (Environmental Risk Management System) project was established to develop a similar EIF for drilling discharges (EIF_{DD}).

The overall objective of the ERMS Joint Industry Program (JIP) was to develop an environmental risk-based decision support tool, to assist the oil industry in establishing cost-effective mitigation measures for reducing potential harmful discharges to the marine environment from drilling discharges. The concept was based on the PEC/PNEC ratio approach as a basis for calculation of Environmental Impact Factors (EIFs), as was successfully developed for produced water discharges through the DREAM (Dose-related Risk and Effect Assessment Model) program.

The ERMS program was carried out with SINTEF as the contractual partner towards the oil companies. The scientific work has been carried out by the following research organisations: Akvaplan-niva, Battelle, MUST, RF-Akvamiljø, SINTEF, TNO and the University of Oslo. The oil companies have also contributed with significant scientific input to the program. The program was managed by a Steering Committee with professional participation from the oil companies ConocoPhillips, Eni, ExxonMobil, Hydro, Petrobras, Shell, Statoil and Total

The ERMS program was initiated in 2002 and completed in 2007 with a budget frame of more than 28 million NOK. A total of 7 workshops and 15 Steering Committee meetings were held. A User Group consisting of representatives from the oil companies and research organizations has been active during the program and will continues its work with case studies and further qualification of the model developed.

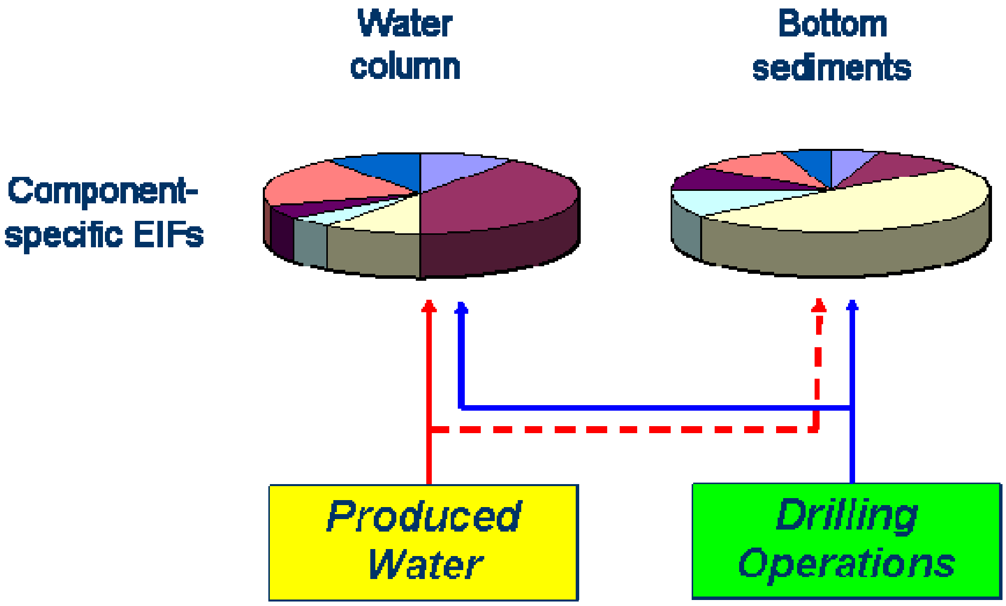

Figure 2.1 The ERMS project has produced a methodology for environmental risk assessment associated with drilling discharges. Subsequent goals include combining these and other EIFs into a holistic risk assessment concept.

1.4 Reading guide

The objective of this summary report is to give an overview or the ERMS program and not to go into detail about the different subjects. More detailed information can be found in the other 27 reports prepared as part of the program and the papers submitted for publication (as listed in Section 9).

Chapter 2 providse information about the drilling process and the discharges taken into consideration in the EIF_{DD}. Chapter 3 describes the concept of the model including the main risk assessment principles. A brief overview will be presented of the model framework and how fates, thresholds and the final EIF_{DD} figure are calculated. In Chapter 4 some results of scenario calculations with the EIF_{DD} are presented. Validation of several parts of the EIF_{DD} framework was also a part of the ERMS project. These activities are described in Chapter 5. Chapter 6 will provide an overview of future developments focussing on the improvement and validation of the EIF model. Chapter 7 presents the acknowledgements and Chapters 8 and 9 provide references and a full list of the reports resulting from the ERMS project.

2 Drilling process and discharges

2.1 Drilling operations

Before the drilling operations are initiated, a heavy wall metal pipe with a diameter of 30” (the conductor pipe) will be driven into the seabed to a depth of approximately 50 meters. This pipe should maintain the stability of the shallow drilling hole and protect against eventual pollution of groundwater and seawater. The actual drilling will be carried out inside of the conductor pipe.

Normal drilling operations consist of:

- drilling as efficiently as possible, adding drilling pipes as the well deepens;
- pulling out the drill string from the well to install a new drilling bit and running it back to bottom (tripping);
- running and cementing casing after each drilled section;
- evaluating the formations drilled by cores, logs;
- completing the well for production.

During drilling, a mixture of water, clay, weighing material and chemicals (called a drilling fluid or drilling mud) is used. The drilling mud is pumped down into the drill pipes and lifts the cuttings made by the bit to the surface. Furthermore, the drilling mud provides cooling and lubricating of the bit, it keeps the cuttings in suspension when drilling is interrupted, it provides stability of the well wall and it prevents natural gas or fluids from entering the well from the penetrated layers.

2.2 Drilling discharges

The largest-volume solid wastes generated during drilling of oil and gas wells offshore are the drilling muds. The cuttings and mud discharges originate from the drilling process, including cementing, maintenance and testing operations. Different types of mud are used, mainly:

- Oil-based mud (OBM)
- Synthetic base mud (SBM)
- Water-based mud (WBM)

Other types of discharges or ingredients of interest are:

- Dope (for lubricating the drill string)
- Contingency chemicals
- Cementing chemicals
- Chemicals for testing the Blowout preventer (BOP)
- Completion chemicals
- Weight materials like barite

Bulk discharges of WBM are permitted in some areas. Drilling muds are usually reprocessed and recycled as much as possible during drilling. Eventually, they are altered by exposure to high temperatures and pressures in the well or by dilution with water and clay-sized cuttings particles to the point where they can not be recycled. Then, they may be discharged to the environment (if permitted by local regulations), re-injected into a well, sent to shore for reprocessing or disposal in a land fill, incinerated (OBM), or applied to agricultural land as a soil amendment (WBM only).

Most drilling of offshore oil and gas wells in the North Sea (including the Norwegian Sector), the Gulf of Mexico, and other offshore production areas is achieved with WBM (Frost *et al*., 2006) This is due to strict regulations on discharges of OBM and SBM. Discharge of diesel based drilling fluids was prohibited within the OSPAR region[1] in 1984, while discharges of OBM as contamination on cuttings have been prohibited in Norway since 1993 (and 1996 within the OSPAR area). The use of SBM in the North Sea has been minor after 2001.
Figure 2.2 gives a mechanistic overview of the fate of drilling discharges (Rye *et al*., 2006; ERMS Report no. 18).

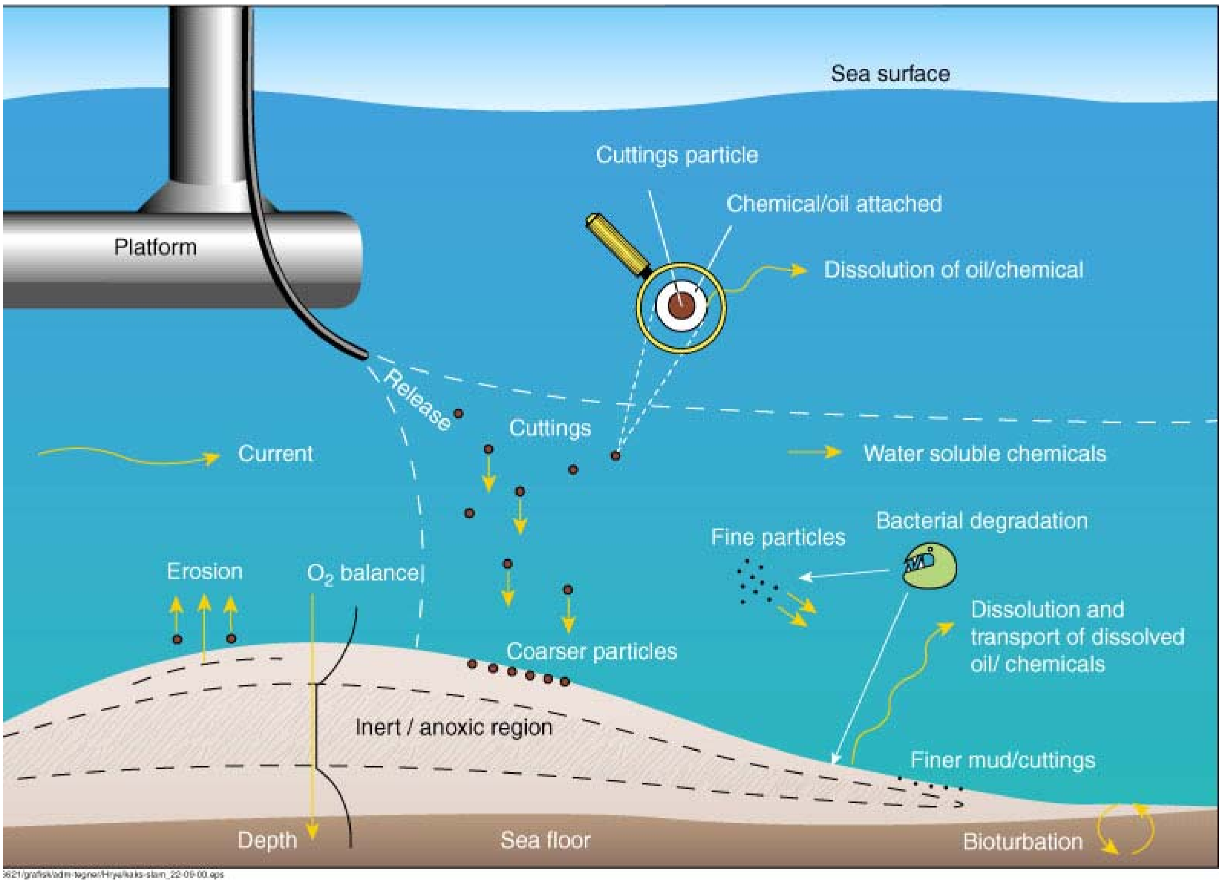

Figure 2.2 Overview of short- and long-term disturbances of drilling discharges

Effects from drilling discharges have the potential to influence upon two compartments: the water column and sea-floor sediments. The potential effects can be divided into toxic and non-toxic disturbances. Based on literature information, monitoring studies as well as detailed knowledge of drilling procedures the following stressors have been identified for drilling discharges:

- Water column:
  - Toxicity of chemicals
  - Physical effects of suspended matter
- Sediment
  - Toxicity of chemicals
  - Burial of organisms and change in sediment structure
  - Oxygen depletion and consequential increase in sulphide concentration

[1] The 1992 OSPAR Convention is the current instrument guiding international cooperation on the protection of the marine environment of the North-East Atlantic. It combined and up-dated the 1972 Oslo Convention on dumping waste at sea and the 1974 Paris Convention on land-based sources of marine pollution.

3 Structure and design of the EIF_{DD}.

3.1 Concept

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

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

Figure 3.1 provides an overview of the main activities within the framework of the EIF_{DD} calculation.

The toxicity component of EIF for drilling discharges has been developed in accordance with the principles of risk and hazard assessment described by the European Union (EU) in the Technical Guidance Document (TGD) (EC, 1996, 2003). Environmental risks for chemicals in different marine environmental compartments are estimated by calculation of PEC/PNEC ratios. The PEC (Predicted Environmental Concentration) is an estimate of the concentration of a chemical to which the biota will be exposed during and after the discharge of the chemical. The PNEC (Predicted No Effect Concentration) is the concentration of the chemical in the environment below which it is unlikely that adverse effects on the biota inhabiting a particular environmental compartment will occur. The ratio of the PEC to the PNEC indicates the likelihood of the occurrence of adverse effects from drilling discharge chemicals in the water column and sediments.

Two approaches have been evaluated for the development of a concept for the EIF_{DD}. The first approach is similar to the approach used in the EIF_{PW}, relying on extensive review and analysis of published toxicological data to determine PNEC values. The second approach takes advantage of the extensive benthic survey data available through monitoring studies. The convergence of these approaches in establishing toxicological parameters for the present model is discussed in Section 5.

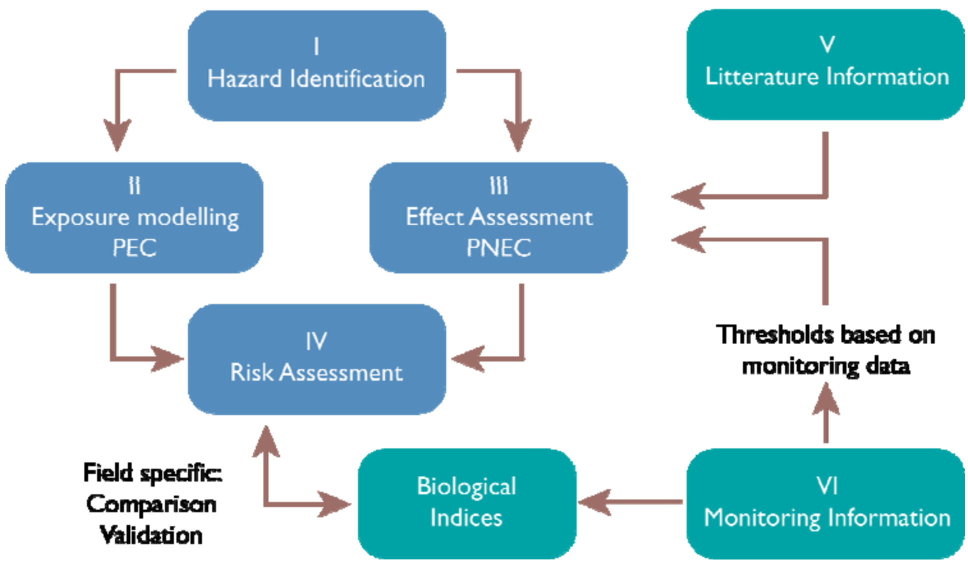

Figure 3.1 Framework for the EIF_{DD} indicating the different steps in the risk assessment process.

3.2 Fates

An important step in the risk assessment process is to quantify the exposure for all stressors. The exposure is represented by the PEC (Predicted Environmental Concentration) and can be obtained by actual field measurements (monitoring data) or by estimation using environmental fate models. To represent the exposure of the defined stressors the following parameters have been selected:

Water column:
- Toxic components concentrations
- Suspended matter concentrations

Sediment:
- Toxic components concentrations
- Oxygen depletion as a result of organic carbon enrichment
- Change in grain size distribution
- Coverage by sedimentation of material - burial

The exposures of the different stressors will be assessed by the DREAM model and are described in detail in ERMS Report no. 18 (Rye *et al*., 2006).

The DREAM model has been extended to include the impacts on the sediment in addition to the impacts in the water column. The near-field module has been extended to include:

- Blowouts or pipeline leaks in deep or shallow waters,
- Produced water releases,
- Releases from drilling operations,
- Combinations of these.

Figure 3.2 shows a general layout of the model structure for the impact calculations.

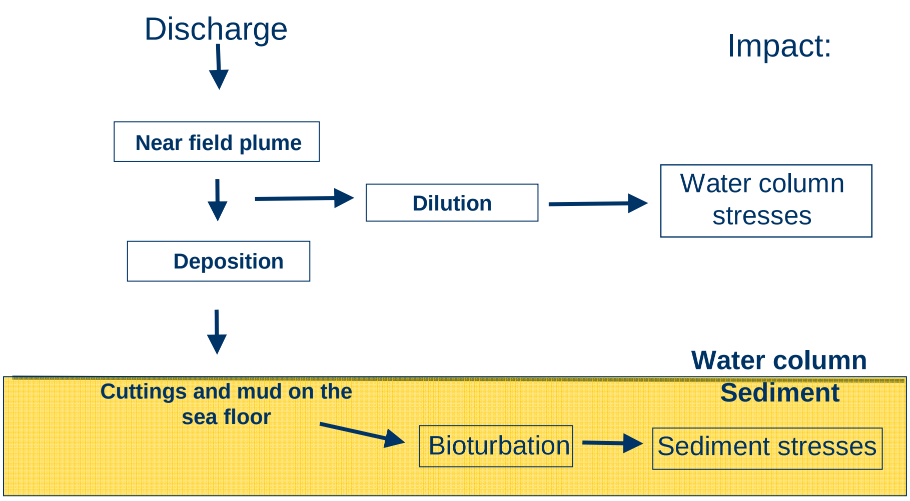

Figure 3.2 Layout of the model structure for calculations of potential impact.

3.2.1 Water column

The concentrations in the *water column* are determined by the physical and chemical properties of the discharged chemicals under the influence of environmental processes like currents, turbulence and mixing. In principle, the same approach has been adapted for drilling discharges that has been used for produced water.

Most drilling muds and cuttings do not contain sufficient quantities of the chemicals to cause acute or chronic harm to the marine environment. A limited number of chemicals was therefore selected for inclusion in the risk calculations for drilling discharges, based on the following criteria (Frost et al., 2006):

- 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.

Components that can be found in drilling muds and cuttings:

1. Metals (from added chemicals or from barite)
2. Natural organic compounds
3. Added chemicals
4. Clay particles from weighting agents. The weighting agent in the drilling mud is the main source of suspended particles; usually consisting of barite, illminite, bentonite or attapulgite.

The EU Technical Guidance Document (EU-TGD, EC 2003) provides recommendations on how the concentrations (PECs) are to be calculated for discharges to sea. The equations included in the calculation of the EIF_{DD} are mainly based on these recommendations. Exceptions are described by Rye *et al.* (2006).

3.2.2 Sediment

Due to the particle content parts of the discharge sink on the sea floor. Once on the sea floor, processes like bioturbation and degradation will change the quality and structure of the sediment. In order to assess the exposure also undistributed sediment processes have to be incorporated in the model.

The depositions to the sea floor are caused by different contributions:

- *The cuttings particles* sink to the sea floor in accordance with their sinking velocity (given by their sizes and densities).
- *The particles in the weighting material* (an example is barite) are also assumed to sink to the sea floor in accordance with the sinking velocity of the particles (given by their sizes and densities).
- *The chemicals* in the discharge with log P_{ow} (or log K_{ow} or log K_{oc}) > 3. These are assumed primarily to deposit on the sea floor, either as “attached” to cuttings particles (or to other particle groups) or as “agglomerates” (that is, particles formed as clusters consisting of the chemicals, the cuttings particles and the particles in the weighting material).

In addition, *the heavy metals in the barite* are assumed to be “attached” to the barite particles and will thus move along with the barite. For the part of the barite that deposits within the model domain, the toxicity of the metals (in the barite) is calculated for the sediment. The toxicity of any metals in the cuttings particles is neglected.

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

- *The burial* is represented by the thickness of the new layer added. This parameter is calculated from the deposition of the discharged compounds only (not including natural deposition). The fraction of the suspended matter that settles will, together with the settled cuttings, form a deposition 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. 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 toxicity* of the new sediment layer is simply calculated from the content (concentration) of the chemical(s) in the added sediment. Bioturbation will cause mixing between the new and the old sediment, such that chemicals in the new layer will also be mixed to greater depth in the sediment..
- *The free oxygen depletion* is calculated by comparing the new free oxygen profile after discharge with that in the undisturbed sediment. 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 change of grain size* (introduction of “exotic” sediment). A new layer with another median grain size is added on the top of the former (natural) sediment layerdue to deposition from the drilling release. 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 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 and described by Rye *et al.* (2006).

Heavy metals attached to the barite may enter the sediment layer along with the barite particles. These metals may impact the biota in the sediment layer. In the model, the bioavailable 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.

The processes taking place in the sediment are calculated using the conservation equations for mass and momentum, known as the “diagenetic” equations when applied to the sediments. These are differential equations which have to be solved numerically.

3.3 Effect levels and thresholds

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. 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*., (2006) (ERMS Report no. 9). 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.

PNEC (Predicted No Effect Concentration) is the concentration of a chemical in the environment below which it is unlikely that adverse effects on the biota inhabiting a given environmental compartment will occur.

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_{50} (Lethal Concentration), EC_{50} (Effect Concentration) or NOEC (No Observed Effect Concentration) value by an appropriate assessment factor in accordance with the EU-TGD.

The distribution that describes the variability of hazard of a stressor to organisms is called a Species Sensitivity Distribution (SSD).

3.3.1 Effects in the water column

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).

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.

Discharges of drilling muds and cuttings will result in increased concentrations of suspended particulate matter (SPM) in the water column and may cause an impact on water column organisms, and organisms inhabiting the sediment but having contact with the overlying water. 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 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 proposed PNECs for the weighting agents are all lower than the lowest observed effect levels (0.2; 0.09; 1.8 and 0.8 mg/l for barite, bentonite, attapulgite and WBMs respectively). An extended description of the data and PNEC derivation for weighting agents in drilling mud is provided in Smit *et al.* (2006) (ERMS Report no. 6).

3.3.2 Effects in the sediment

<u>Chemicals</u>

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

In absence of any ecotoxicological data for sediment-dwelling organisms the EU-TGD (EC, 2003) allows a provisional calculation of $PNEC_{sediment}$ 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 $PNEC_{sediment}$, the equilibrium partitioning approach was evaluated.

The equilibrium partitioning method derives sediment quality criteria (SQC) or PNECs from water quality criteria 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.

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

<u>Burial</u>

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. 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. A statistical description of the variation in sensitivity (Species Sensitivity Distributions) (SSD) is applied to derive the threshold value at a level of 0.65 cm (5 - 95% conf. interval of 0.32 – 1.07).

An overview of the data and procedures applied to derive the PNEC for burial is presented in Smit *et al*. (2006) (ERMS Report no. 9).

<u>Grain size change</u>

Many studies have revealed a relationship between sediment type and infauna community structure, there is considerable variability in species responses to specific sediment characteristics. 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). 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 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).

Based on this information a species sensitivity distribution is derived for changes in grain size leading to a PNEC of 52.7 μm (47.4 – 57.9).

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

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 Rye *et al*., 2006) A value of 20% is considered to be a realistic value for a threshold level for hypoxia by several experts.

3.4 Risk assessment and EIF_{DD} calculation

In the risk assessment step the PEC is compared to the PNEC resulting in a PEC over PNEC ratio in each grid cell and for each stressor. At a PEC over PNEC ratio higher than 1, unacceptable effects on organisms are likely to occur (EU, 2003). The higher the ratio, the more likely it is, that unacceptable effects may occur. However, the PEC over PNEC ratio is only an indicator of risk and for stressors with different modes of action PEC over PNEC ratios cannot directly be compared (Smit *et al*., 2005). The SSDs provide a mean to calculate a more quantitative and comparable risk indicator: the Potentially Affected Fraction of species (PAF). The PAF value can be explained as the probability that randomly selected species is exposed to a concentration exceeding its chronic no effect level at a certain level of exposure (Figure 3.3). The exposure of organisms to substances is considered acceptable is case less than 5% of the species is at risk (corresponding to a PEC/PNEC ratio of 1). For all stressors PAF levels will be calculated corresponding to the predicted levels of exposure per grid cell.

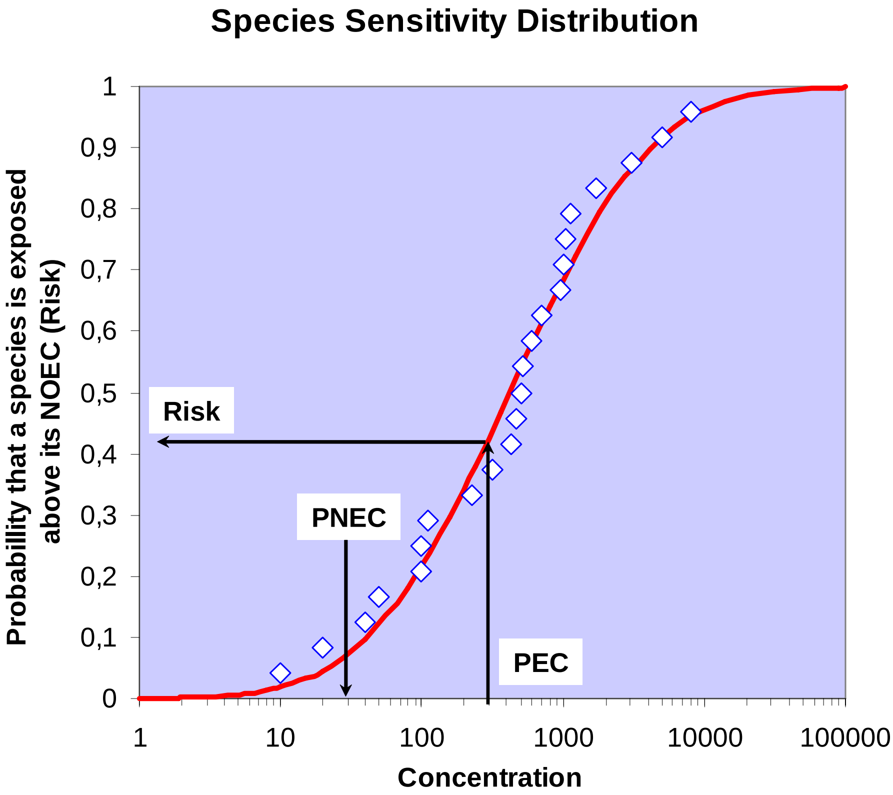

Figure 3.3 An example SSD; The PNEC corresponds to a probability value (PAF) of 5%. At any PEC a Potentially Affected fraction of Species (PAF) can be calculated.

In model grid cells in the water column PAFs for exposure to chemicals and suspended clay particles will be calculated. In model grid cells for the sediment PAFs will be calculated for exposure to chemicals, burial, grain size change and oxygen depletion. For the calculation of the combined risk related to the exposure from toxic and non-toxic stressors associated with drilling impacts additivity is a pragmatic working assumption. Potentially Affected Fractions (PAFs) calculated for the different stressors are combined in a multi stressor PAF value (msPAF) or joint risk probability. The msPAF per grid cell is calculated assuming independent action using the following equation:

$$msPAF(A+B) = PAF(A) + PAF(B) - PAF(A) * PAF(B)$$

where PAF(A) is the risk probability for stressor A and PAF(B) is the risk probability for stressor B.

For a larger number of stressors, the msPAF is calculated from the generalized formula for the sum of probabilities PAF(A), PAF(B), ….. PAF(i) according to the equation:

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

If the value of the msPAF in a water column grid cell exceeds 5%, the volume of the grid cell is included in the calculation of the water column part of the EIF_{DD} (EIF_{DD}-water). The EIF_{DD}-water is defined in the same way as the EIF_{PW}: 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 described in Johnsen *et al*. (2000). In contrast to produced water discharges, drilling discharges are not continuous. Therefore, not only the value of the EIF_{DD}-water should be reported, also the duration the EIF_{DD}-water is larger than zero (See also Rye *et al*., 2006). 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 3.4.

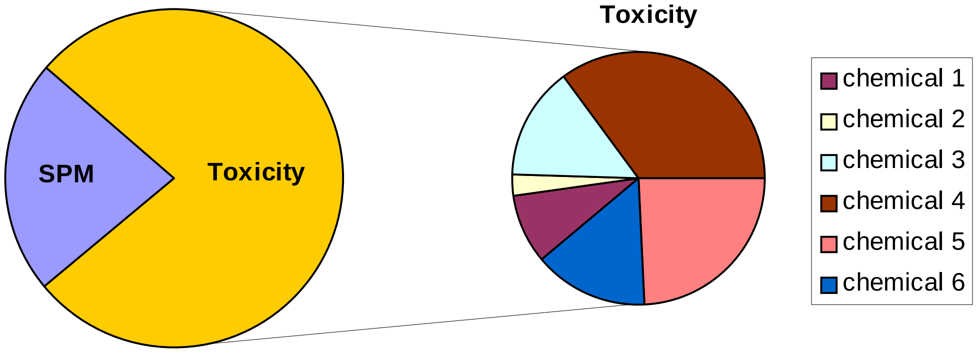

Figure 3.4 Graphical presentation of the EIF_{DD}-water.

The PAF value of the different stressors to the sediment as well as the msPAF exceeding 5% can be mapped resulting in a risk area.

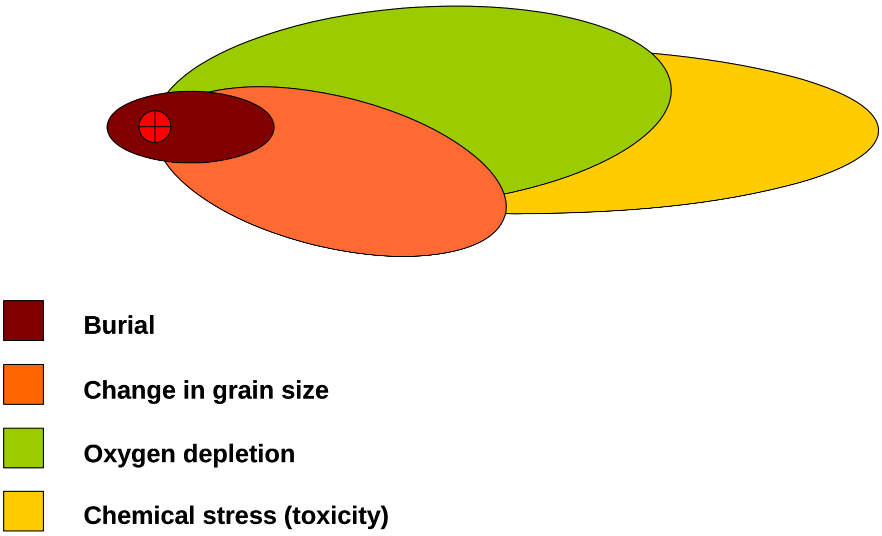

Figure 3.5 Graphical presentations of the areas where the PAF value for the different selected stressors exceeds 5%.

The sediment area where the combination of the risk probabilities (msPAF) exceeds 5% will contribute to the sediment related part of the EIF_{DD} (EIF_{DD}-sediment). 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.

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 one single value of EIF_{DD}-sediment should be reported, but also the variation of the EIF_{DD}-sediment value over time and the variation in the contribution to the EIF_{DD}-sediment (Figure 3.6).

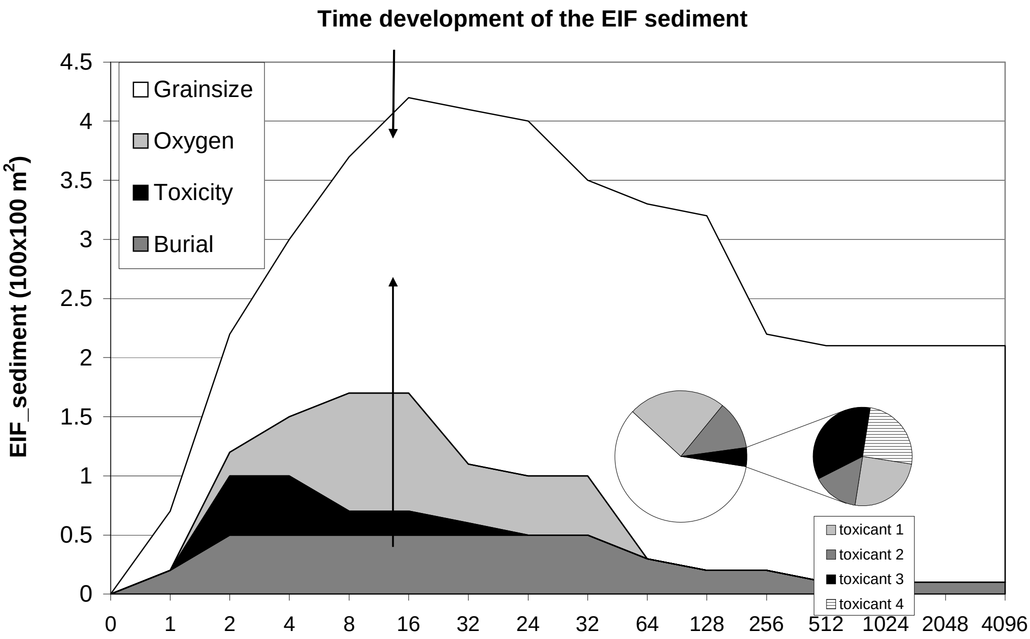

Figure 3.6 Graphical presentation of an example of the influenced sediment area (represented by msPAF > 5%) and the contribution of each stressor to the overall risk related to this theoretical drilling discharge.

4 Application of the EIF_{DD}

4.1 Input data for the simulations.

The updated DREAM model is used to calculate the environmental risks of drilling an exploration well using WBM. Five sections are involved, including two pilot holes and one “plug and abandonment” operation. The upper two drilling sections (36” and 26”) are discharged directly on the sea floor, while the lower three drilling sections (17 ½”, 12 ¾” and 8 ½”) are discharged from the drilling rig.

The example case given here is extracted from Chapter 7 in the ERMS report No. 18, “*Documentation report for the revised DREAM model*”.

Grain size distributions for the cuttings and barite particles used in the calculations are shown in Tables 4.1 and 4.2. To the extent that these particles reach the sea floor, they form the basis for calculating the stress caused by the change of grain size and burial. The median diameter of the natural sediment on site is assumed to be 0.03 mm.

Eight different discharges are defined for the exploration well. Each of these has its own composition of release and duration. The discharges comprise cuttings, barite and bentonite for the particle groups and lubrication for the drill string and a drilling chemical. In addition, PLONOR (= *Pose Little Or NO Risk* to the environment) chemicals and water were present in the discharges. The lubrication chemicals were assumed to be “attached” to the cuttings particles.

Therefore, they followed the cuttings particles down to the sea floor in the simulations. The discharge of the lubrication chemicals was estimated based on consumption of the chemicals per meter well drilled, both for drill string lubrication and for casing lubrication.

Table 4.1 Grain size distributions of cuttings particles and their sinking velocities. The sinking velocity is determined from particle diameter and density.

DRILL CUTTINGS				
Diameter	Weight	Density	Velocity	Velocity
Mm	%	SG	m/s	m/day
0.007	10	2.4	1.9E-05	1.7
0.015	10	2.4	8.8E-05	7.6
0.025	10	2.4	2.5E-04	21.2
0.035	10	2.4	4.8E-04	41.6
0.05	10	2.4	9.8E-04	84.9
0.075	10	2.4	2.2E-03	191.0
0.2	10	2.4	1.6E-02	1356.5
0.6	10	2.4	5.7E-02	4898.9
3	10	2.4	2.1E-01	17988.5
7	10	2.4	3.2E-01	27483.8

Table 4.2 Grain size distributions of barite particles and their sinking velocities. The sinking velocity is determined from particle diameter and density.

DRILLING MUD				
Diameter	Weight,	Density,	Velocity,	Velocity,
mm	%	tonnes/m3	m/s	m/day
0.0007	10	4.2	4.4E-07	0.04
0.001	10	4.2	9.1E-07	0.08
0.002	10	4.2	3.6E-06	0.31
0.003	10	4.2	8.2E-06	0.71
0.005	10	4.2	2.3E-05	1.96
0.009	10	4.2	7.4E-05	6.35
0.014	10	4.2	1.8E-04	15.37
0.018	10	4.2	2.9E-04	25.41
0.028	10	4.2	7.1E-04	61.49
0.05	10	4.2	2.3E-03	196.08

The discharge period is about 10 days in this example simulation. This time span is generally shorter than the time used when drilling an exploration well. In the simulations, only “effective” drilling time is included, that is, when drilling is actually taking place. This drilling time is calculated from a drilling penetration rate downwards of order 10 – 25 m/hour for typical well sections. The penetration rate is somewhat dependent on the diameter of the well section drilled.

The sediment model is run for 10 years, because sediment processes are generally much slower than the time scale of the actual deposition on the sea floor.

4.2 Stresses calculated for the sediment.

The four stressors defined for the sediment layer were calculated: burial, toxicity, oxygen depletion and change of grain size. The results from the calculation of these are considered below.

Stress caused by burial. Figure 4.1 shows the deposit (in mm thickness) on the sea floor after the completion of all the 8 releases from the various drilling sections.

Figure 4.2 shows the grids that exceeded the PNEC limit of 0.65 cm added thickness. Grid cells where impacts from burial are likely to occur are those colored red (closest to the discharge point).

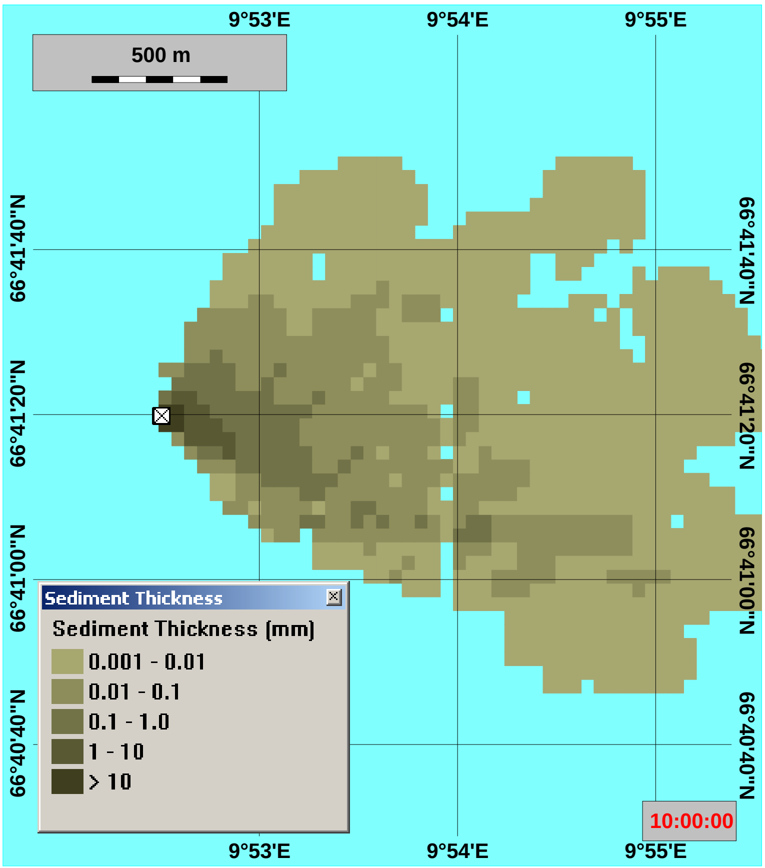

Figure 4.1 Deposition of discharge (layer thickness) at the end of the discharge period (after 10 days).

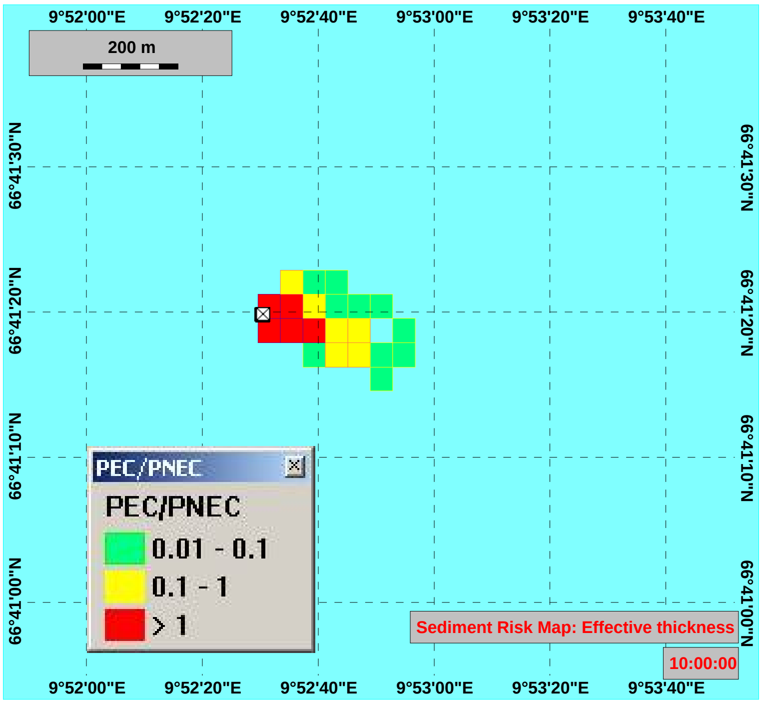

Figure 4.2 Grids that exceed the PNEC level for burial at the end of the discharge period (red color). Grid points are approx. 50 x 50 m.

Stress caused by toxicity in the sediment. The chemicals that are brought down into the sediment layer are two "dope" chemicals used for lubrication. However, the concentration of these chemicals in the sediment is rather low (averaged over the upper 3 cm of the sediment layer). Concentrations of both are well under the toxicity limits (PNEC > 300 ppm) for these chemicals in the sediment layer. Figure 4.3 shows the concentration of the sum of these two chemicals at the end of the discharge period.

Both lubrication chemicals that are used are however biodegradable, so the concentration of these chemicals reduces rapidly with time. Figure 4.4 shows the concentration of the "dope" chemicals as a function of time for the grid point with the largest concentration. The chemicals are both biodegraded within one year simulation time.

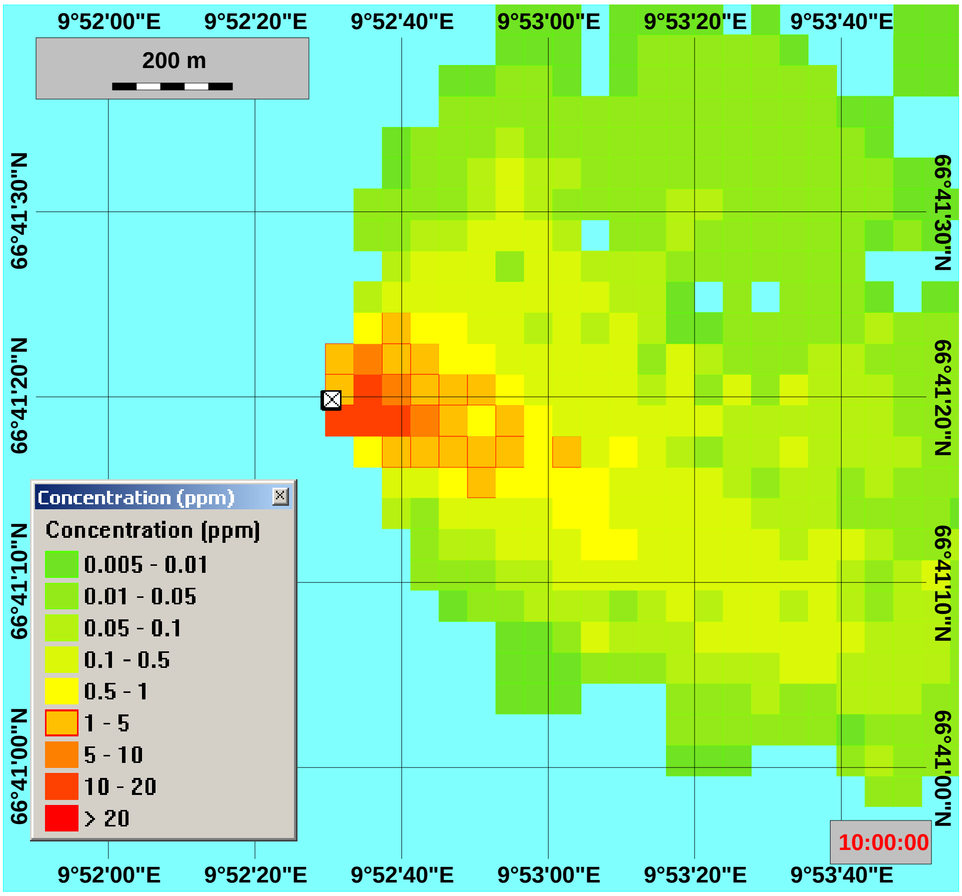

Figure 4.3 Concentration of the "dope" chemicals in the sediment layer at the end of the discharge period (10 days). Max concentration of the "dope" is about 16 ppm (mg/kg sediment)

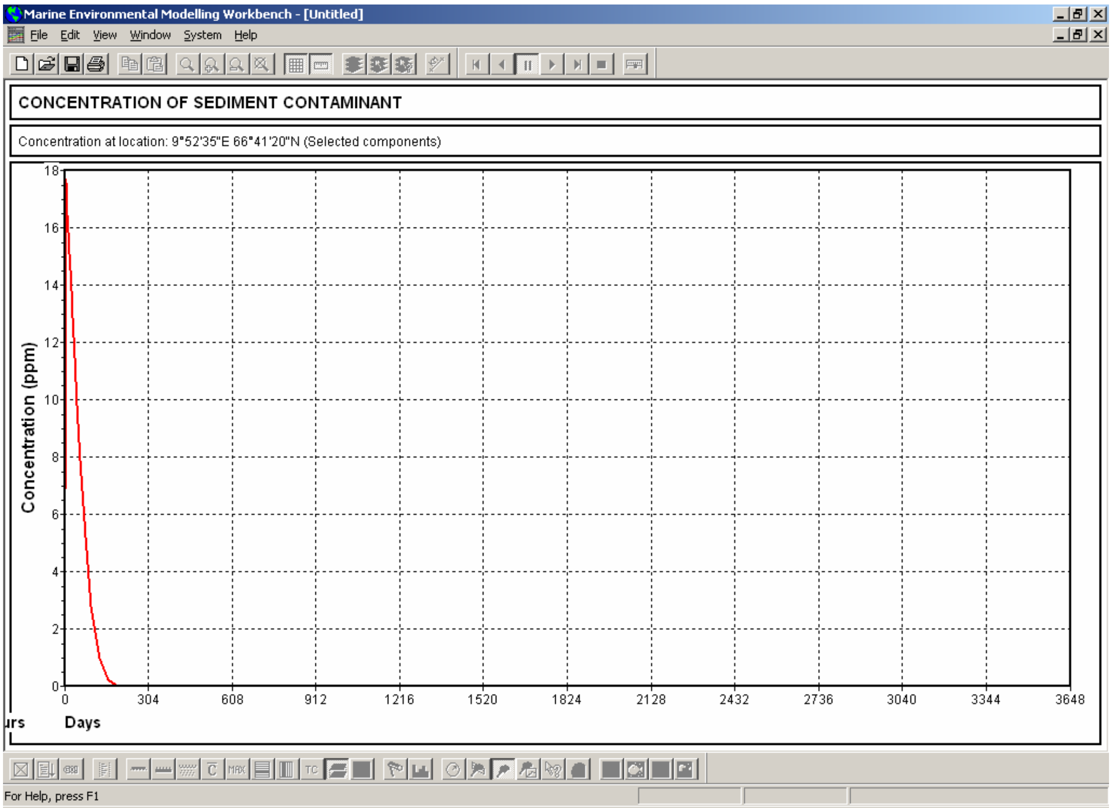

Figure 4.4 The time development of the "dope" concentration in the sediment layer for the grid with maximum concentration. Horizontal time scale in days. The concentration is biodegraded down to zero within about 6 - 8 months.

Stress caused by oxygen depletion in the sediment. Although the lubrication chemicals did not contribute to risk in terms of toxicity, the biodegradation consumes oxygen in the sediment layer. This consumption may cause a reduction of the oxygen content in the sediment.

The PNEC level for oxygen reduction in the sediment layer is set to 20 % reduction of the pore water oxygen content in the layer (in terms of mg O_2/m^2 sediment surface). This level is surpassed in some of the grid points temporarily. Figure 4.5 shows the reduction of the oxygen content in the sediment layer after about 20 days.

Figure 4.6 shows the time development of the oxygen content in the grid point with the maximum concentrations of "dope" chemicals. The 20 % level of reduction of oxygen content (compared to the oxygen content before discharge) is surpassed in a relatively short time interval (some months) just after the discharge period has ended. After the chemicals have biodegraded, the oxygen level returns to more normal levels.

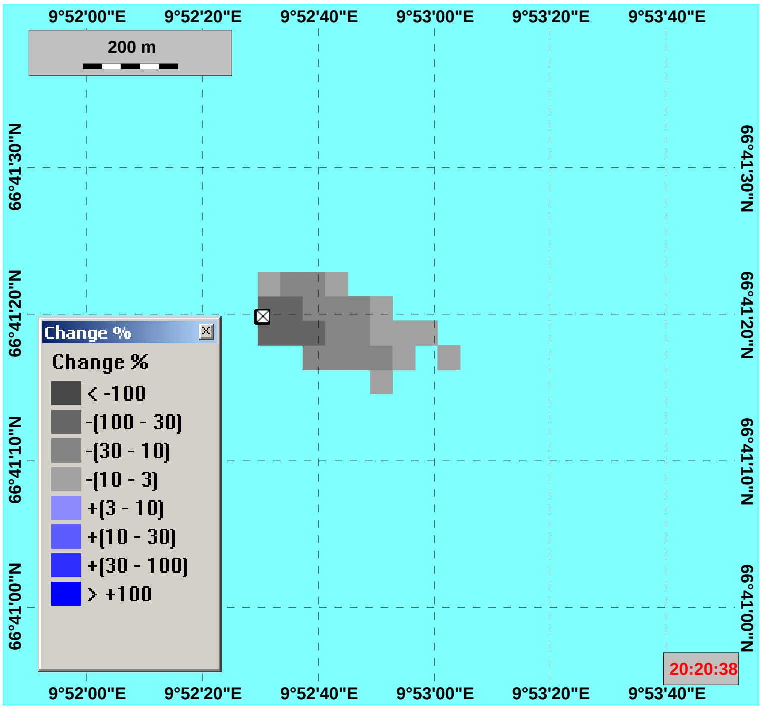

Figure 4.5 Oxygen depletion in the sediment layer after about 20 days of simulation time, caused by the degradation of the lubrication chemicals.

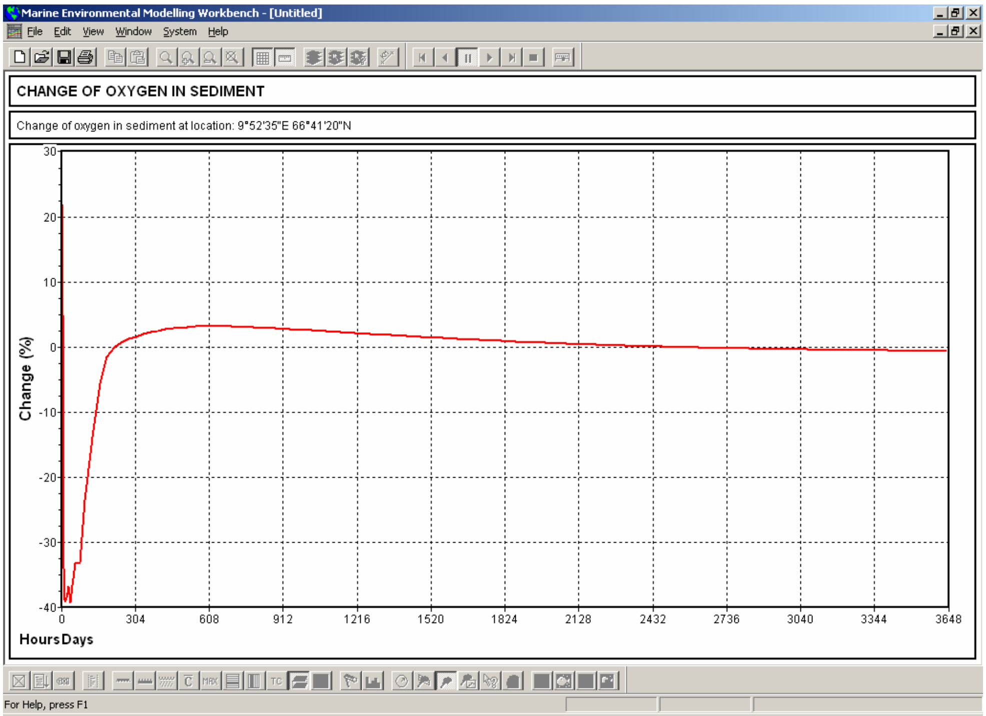

Figure 4.6 Oxygen depletion calculated as a function of time for the grid point with the largest concentration of the lubrication chemicals. Horizontal scale in days. The PNEC level of 20 % reduction of the oxygen content in the sediment layer is surpassed for a short time period (some months) in the beginning of the sediment impact simulation period.

Sediment stress caused by changes in median grain size. Natural sediment in the actual location has been specified to be about 0.03 mm diameter median grain size. The cuttings in particular have larger grain sizes than the natural sediment on this site. Therefore, the median grain size for some grids will change due to the deposition of the cuttings particles. Figure 4.7 shows the change of grain size in the sediment at the end of the discharge period (after 10 days).

Figure 4.8 shows one example (one grid point) of the time development of the grain size change for the upper 3 cm of the sediment layer. The risks due to the change of the median grain size for the upper 3 cm of the sediment layer are reduced somewhat over time due to the return of the original sediment present below 3 cm sediment depth. The reason for this is the effects of the bioturbation, bringing the original sediment back to the sea floor.

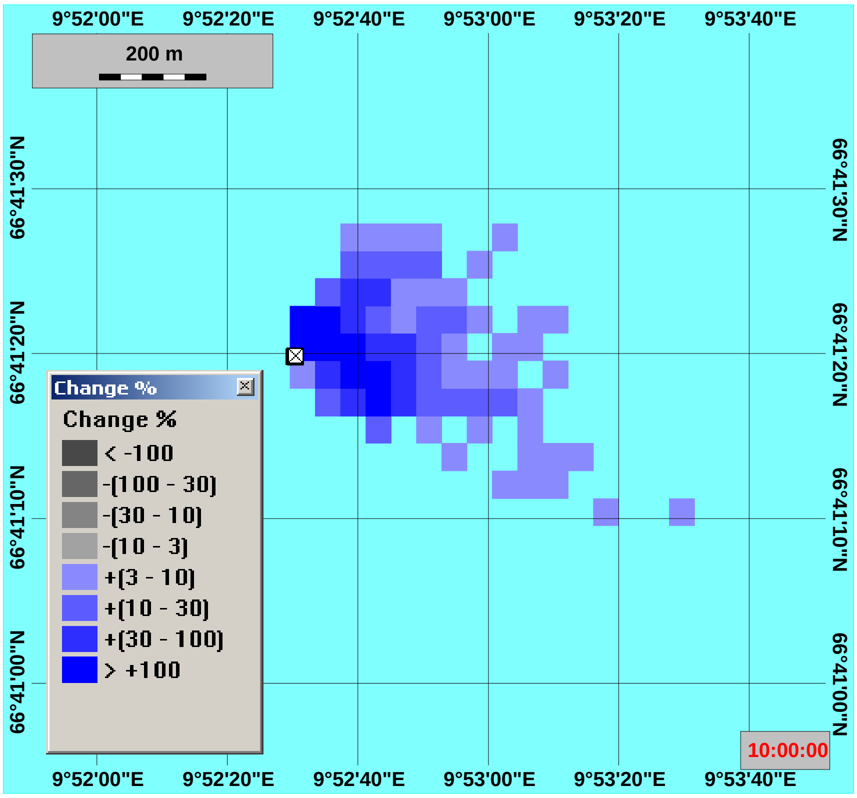

Figure 4.7 Change of grain size in the sediment at the end of the discharge period. The median grain size is averaged over the upper 3 cm of the sediment layer.

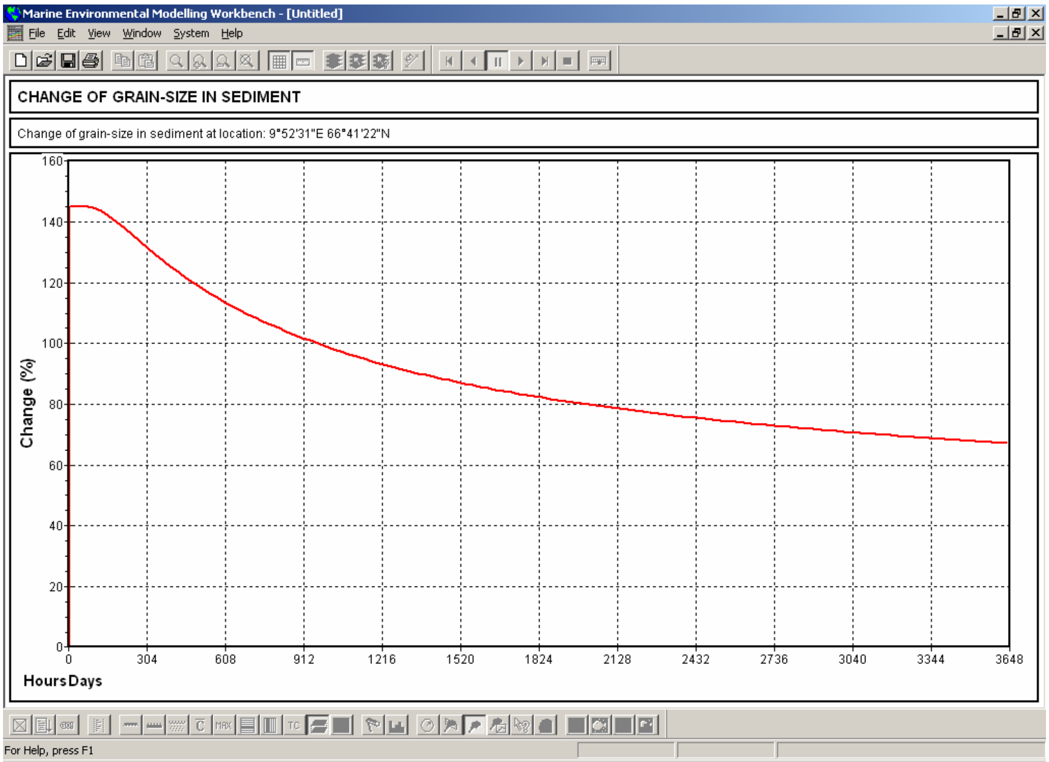

Figure 4.8 Time series for change of grain size for one selected grid point. The reduction of grain size change with time is due to effects from bioturbation, mixing the added particles downwards while the original particles mix towards the sediment surface.

5 Validation and evaluation

5.1 Validation of toxicity threshold values

Task 5 in the ERMS project was initiated for the purpose of validation of the toxicity threshold values derived from the literature review. Two different approaches, the species sensitivity distribution (SSD) approach and the moving window modeling (MWM) approach, have been carried out to establish field-derived threshold effect levels (F-TEL) based on existing sediment data from the NCS (Brakstad *et al*., 2005 (ERMS report no. 13), 2006; Smith *et al*., 2005 (ERMS report no. 3); Grung *et al*., 2005 (ERMS report no. 14); Bjørgesæter, 2006 (ERMS report no. 15)).

The data used is collected from the Norwegian Oil Association's (OLFs) Miljøovervåkings-database (MOD) containing complete datasets from environmental monitoring in the vicinity of petroleum installations on the NCS since 1990. The database covers selected heavy metals and hydrocarbons, grain size and more than 2000 benthic species from depths ranging from 63 to 1500 meters.

Additionally, the field-derived threshold effect levels (F-TEL) based upon the species sensitivity distribution and the moving window modelling approach, have been compared with TELs derived from the US and Canadian Sediment Quality Guidelines (SQG).

Generally, there was good correlation between the PNEC values derived from the equilibrium partitioning (EqP) method and the F-TELs derived from field data on the NCS. On the other hand TELs from the US and Canadian Sediment Quality Guidelines (SQG) is generally higher for all the selected components. This might be due to the fact that the US and Canadian SQGs were derived from coastal and laboratory species, so both the fauna and the environmental conditions are different. Interactions between the investigated component, adsorption (and less bioavailability) and other components present in the field can also lead to lower TELs compared to laboratory data were only one component is present.

Even though there was a general good correlation between the values calculated from the EqP approach and the field derived data, this was not the case for Cr and Hg. For these two components the EqP approach was less conservative than the field-derived data.

5.2 Validation by field experiment

As a part of the ERMS project, a field experiment was conducted. The purpose of the field trial was to collect field data for comparison with model results. The Sleipner field in the North Sea was selected because a drilling program was planned at the Sleipner Vest Alfa Nord (SVAN) location (Jensen *et al*., 2004; Trannum *et al*., 2004 and 2006). This location is about 18 km northwest of the existing Sleipner A and T platforms.

At the same time, IRIS-Akvamiljø has been granted by the Norwegian Research Council (NFR) to carry out a project termed "Validation" over the NFR PROOF Program. This project is aimed at validation of methods for carrying risk analysis offshore. Because the ERMS project is aimed at developing numerical models for carrying out risk analysis for discharges to sea offshore, it was decided that IRIS-Akvamiljø should join the ERMS project by deploying cages with sea scallops and blue mussels close to the discharge site. Then the methods validated by IRIS-Akvamiljø could be tested on the real field case, by comparing risks deduced from the responses on the biota with the risks calculated by the numerical models developed as a part of the ERMS project (Berland *et al*., 2006).

One of the purposes of the field trial was to compare numerical model results with the measurements carried out in the field. Feeding the model with the amounts of discharge from the drilling rig (mud and cuttings) as well as the winds, currents and the stratification as observed during the field trial, the expected concentrations of cuttings, barite and a non-PLONOR chemical were calculated. Due to presence of the stratification in the water masses and the presence of the cuttings and the mud in the discharge, the discharge has a much larger density than the ambient water. The discharge plume will therefore sink down until the density of the plume equals the density of the ambient water. This happens in the depth interval 30 – 40m below the sea surface. Figure 5.1 shows a vertical cross section of the plume calculated for the discharge at the SVAN field.

The concentrations for the discharge (the sum of cuttings and barite particle concentrations are shown) tend to have a maximum at some distance below the sea surface. It was therefore expected that it will be cages at 40 m depth that will experience the largest particle stresses.

Evidence for the downwards sinking of the plume can be seen from Figure 5.2, which shows the vertical profiles of temperature, salinity and turbidity collected near the shell cages. A marked increase of turbidity is observed at 45 – 50 m depth (blue line), which corresponds reasonably well with the entrapment of the underwater plume close to 40 m depth.

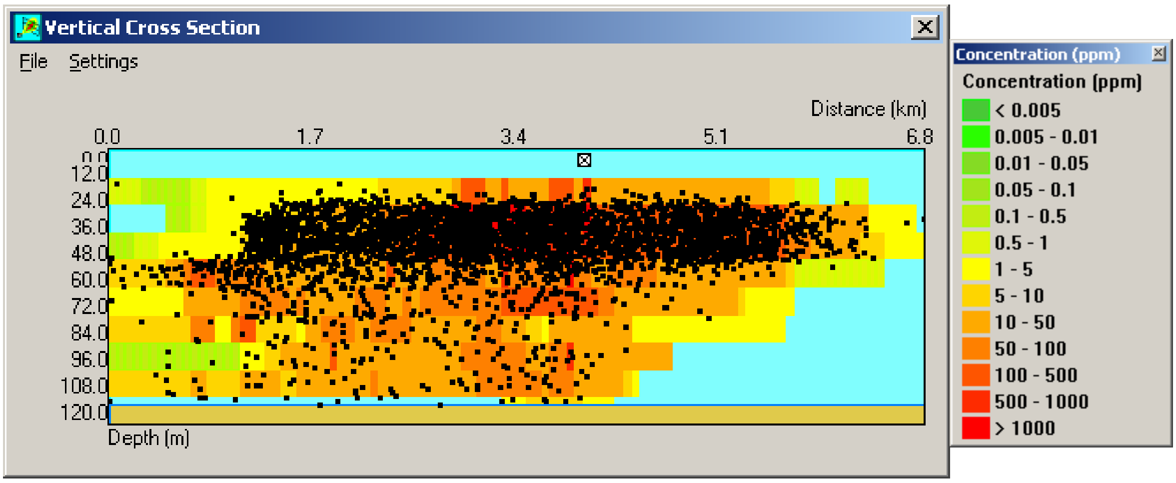

Figure 5.1 *Numerical modeling with the revised DREAM model. A vertical cross section of the plume calculated for the SVAN field release. The discharge point is shown with a cross inside a square (with a cross inside) at 5 m depth at the figure. The near field plume from the discharge will sink down to about 35 – 40 m depth. At this level, the discharge spreads out and is transported away with the currents.*

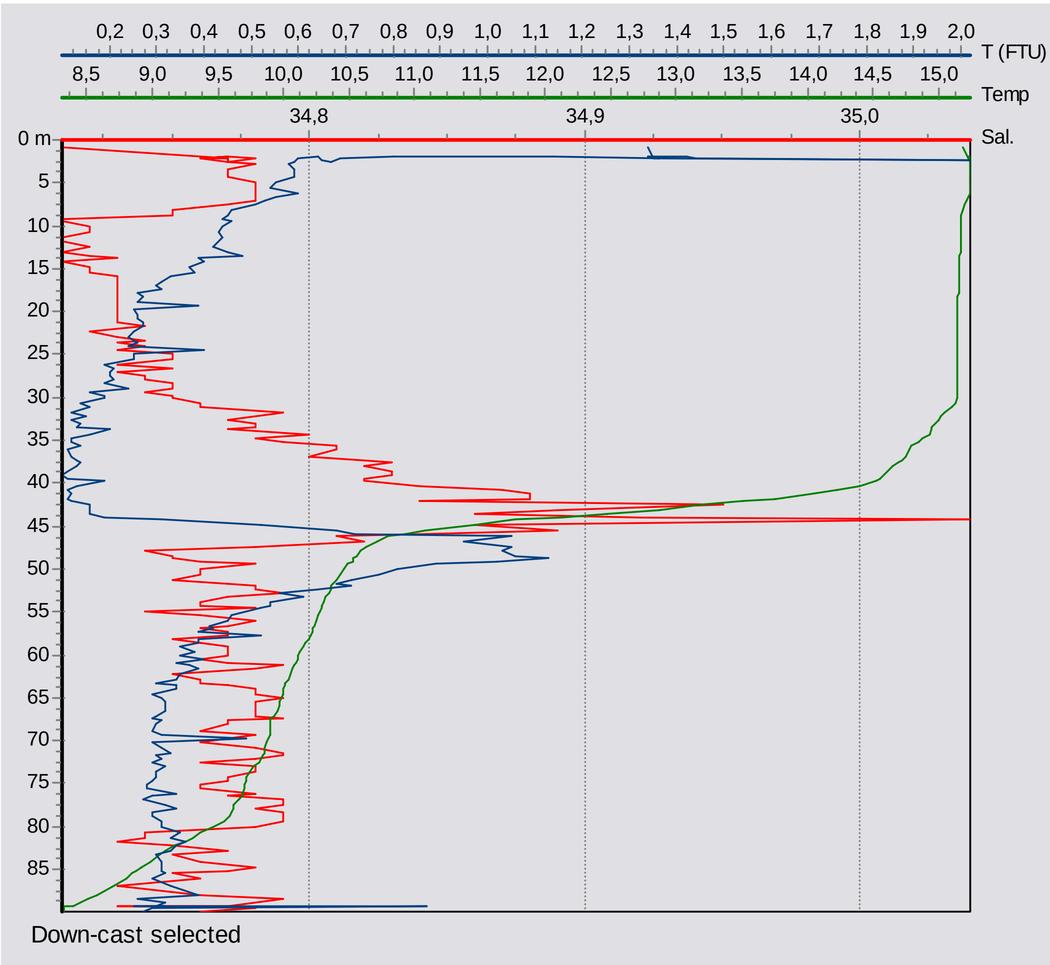

Figure 5.2 *CTD profile close to the discharge site 10 September 2003, at time 1200. The vertical profiles collected at shell cage location B are assumed to be representative for shell cage location A as well. Turbidity (blue), temperature (green) and salinity (red).*

In addition, as a supplement to the measurements and simulations that were carried out, an ROV (= *Remote Operating Vehicle*) was operated from the drilling rig to observe the discharge of drill cuttings and mud. The observations were taped to be examined after the field trial. One of the purposes of these recordings was to examine the sizes and behavior of the particles (and also the possible presence of flocculated material within the plume area). A ruler was also mounted in front of one of the cameras in order to observe particle sizes in the underwater plume.

An observation made by the ROV is also worth mentioning. From the position of the ROV just below the underwater plume at 40 m depth, it was evident that particle matter was leaving the plume area and moved downwards as individual particles. By closing in on these particles with the close-up camera (including the ruler mounted in front), the sizes were observed to be of order some mm, and the particles had a flake-like structure. These particles were most probably cuttings particles, originating from drilling in shalestone or clay layers. It was not observed any indications of (flocculated) barite particles from the ROV recordings.

The results from the biomarker responses showed that drill cuttings and mud cause biological impact as demonstrated with the applied methods. The practical approach of using organisms deployed in cages seems useful for screening of drill cutting and mud discharges. Statistical significant increase of DNA in the comet tail were found for exposed 40m mussels, but not for the 20m exposed group, compare to reference group at 20m. The conditions in the exposed 40m zone during the one month stay, involves mussels in exposure conditions that produced genetically damages in the DNA

The results from the field program were reported jointly for both the ERMS part and the NFR/PROOF part in the ERMS Report no. 20, "*Experimental validation of drilling effects in the field*".

6 Future developments

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.

6.1 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.

6.2 Increased consistency of computational methods between EIF_{PW} and EIF_{DD}

For historical reasons, there are a number of inconsistencies between the two methodologies, such that realistic parameters are not entered into the chemical database for all components in a release. Examples are the solubility for natural and anthropogenic substances in produced water releases (set to a high value for the EIF_{PW}), and solubility for substances in drilling discharges with partition coefficient $K_{ow} > 1000$ (set to zero for EIF_{DD}). Such arbitrary inputs contribute to confusion among users, as reflected in the results from a recent inter-user test (Laurence et al, 2007; in preparation). These inconsistencies also reduce the realism of the outcome, and make it more difficult to compare results for different releases. Increased consistency is therefore a desirable goal.

6.3 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.

6.4 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.

7 Acknowledgement

This report is a summary of the work performed in the Environmental Risk Management System (ERMS) program. This program has been financed by the oil companies: ConocoPhillips, Eni, ExxonMobil, Hydro, Petrobras, Shell, Statoil and Total. The companies are acknowledged for financial support as well as scientific input during the program. Contractors in the program have been: Akvaplan-niva, Battelle, MUST, RF-Akvamiljø, TNO and University of Oslo, with SINTEF as the co-ordinator of the program.

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