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Threshold levels and risk functions for
non-toxic sediment stressors: burial, grain size
changes, and hypoxia - summary report -

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Summary

This report is produced within the framework of the ERMS project (Environmental Risk Management System). It provides an overview of the results related to the establishment of threshold values and risk curves for non-toxic sediment stressors to be included in the calculation rules for the Environmental Impact Factor for drilling discharges (EIF_{DD}). As described in Smit *et al.* (2006) the following non-toxic stressors are defined:
– Burial of organisms;
– Change in sediment structure described by the change in grain size;
– Oxygen depletion.

Comparable to the risk assessment for toxic substances, the risk assessment of these disturbances is based on a comparison of the exposure to the selected stressor and a defined threshold for adverse effects derived for this stressor. The Species Sensitivity Distribution (SSD) for the stressor is used to derive an exposure to risk function (See Smit *et al.*, 2005).

In order to derive a threshold and a SSD, literature and monitoring information was collected for marine species. Results of these inquiries were reported by Kjeilen-Eilertsen *et al.* (2004) and Trannum (2004) for burial and grain size changes, respectively. Background information on hypoxia related stress is collected by Beardsley & Neff (2004) (Appendix 1 of this report).

As no formal evaluation procedures exist for non-toxic stressors the risk assessment guidelines for toxicity described in the EU-Technical Guidance Document (EC, 2003) served as a basis. However, the nature of the data and the studied effects made deviation from the guidelines necessary. Based on the collected information the threshold levels for burial, grain size changes and oxygen depletion as presented in Table 1 were defined.

Table 1   Threshold values for non-toxic disturbances in sediment

Stressor	Value
Burial	0.65 cm deposited layer
Grain size change	52.7 μm change in median grain size
Oxygen depletion	20% reduction of integrated oxygen content

Corresponding risk curves were derived which are to be included in the EIF_{DD} calculation module. Evaluation of defined thresholds and risk curves is anticipated after experience is gained with the model by the evaluation of several management options. The relative importance of the different stressors will be evaluated by modeling different scenarios. After that, it can be decided whether further refinement of the results is necessary.

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1.   Introduction

To aid the industry in the development of the "zero harmful discharge" strategy and selection of cost-benefit based solutions, a methodology has been developed to calculate an Environmental Impact Factor (EIF) for produced water discharges, based on environmental risk and hazard assessment (Johnsen *et al*., 2000). The EIF for produced water was developed to <u>identify the most potential environmentally harmful discharges</u> of produced water and to <u>quantify</u> the environmental benefit of different mitigating measures. As a follow up of the EIF for produced water discharges a project has been started in order to develop an EIF for drilling discharges; the ERMS (Environmental Risk Management System) project. The development of the EIF for drilling discharges builds on the present EIF being used for produced water discharges, including both the environmental compartments water and bottom sediment. The concept for the EIF development is described by Smit *et al*. (2006). The EIF for drilling discharges is an integrated measure of the overall probability of damage caused by the different stressors. This implies that different kinds of stress (toxic and physical) are combined. The current report is dedicated to stressors to the sediment other than toxicity caused by drilling discharges.

1.1   Sediment disturbances caused by drilling discharges

In Figure 1 the disturbances to the water column and sediment caused by drilling discharges are shown. The following disturbances to sediment, other than toxicity, are indicated as important to take into account in the EIF for drilling discharges:
-    Burial of sediment biota;
-    Change in sediment structure (represented by a change in grain size);
-    Depletion of the oxygenated layer (hypoxia).

These sediment disturbances are discussed separately in the chapters 2, 3, and 4.
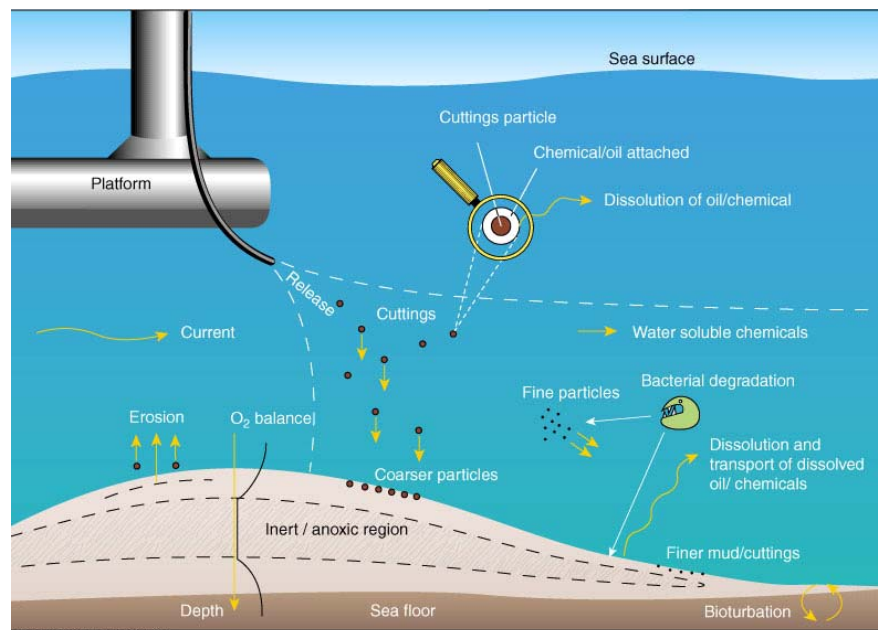
Figure 1   *Overview of short and long term disturbances caused by drilling discharges (figure from Rye et al., 2006).*

1.2   Exposure, thresholds and Species Sensitivity Distributions

The exposure to the non-toxic stressors in the sediment will be modelled as described by Rye *et al.* (2006). In the exposure model the following assumptions are made (See also Smit *et al*., 2006):
–   Habitat change for sediment biota is expressed as the change in median grain size.
–   Burial is expressed as the cumulated thickness of the deposited (mud and cuttings) layer.
–   Oxygen depletion is expressed as the difference in the integrated oxygen concentration in the sediment before and after discharge

Where possible, the framework for risk assessment of toxic substances as set by the EU Technical Guidance Document (EC, 2003) is followed to assess the risk of exposure to non-toxic stressors. This is mainly based on the PEC/PNEC approach (PEC = Predicted Environmental Concentration; PNEC = Predicted No Effect Concentration). Recently, this approach has also been adopted by OSPAR (OSPAR, 2003; ref. nrs. 2002-19 and 2003-20). In order to compare and integrate the different disturbances caused by drilling discharges, preferably, they all should be based on the same principles. It is therefore proposed to apply the PEC/PNEC approach for non-toxic effects as well. The PNEC can be derived from quality assured effect data by applying safety factors or by statistical extrapolation. More detailed information on the application of the PEC/PNEC approach for toxic substances can be found in Neff *et al*. (2006), Smit *et al.* (2005) and Smit *et al.*

(2006).There is, however, no regulatory framework available for other disturbances than toxicity. Therefore, in some cases, it was necessary to deviate from the TGD guidelines. This document describes when and why it was decided to follow an alternative approach than described in the EU-TGD and if relevant provides the details of the alternative approach. Strictly spoken the terms PEC and PNEC should not be used for non-toxic stressors, as these terms refer to a certain concentration. These terms could be replaced by exposure, level or change and threshold. The change is the deviation from the undisturbed situation. However, for reasons of consistency the terms PEC and PNEC are still used in this document. Disturbance – effect relationships and Species Sensitivity Distributions (SSDs) for the stressors are defined. This is based on literature as well as on monitoring information collected in three dedicated studies:

(1) Kjeilen-Eilertsen *et al*. (2004) describe the collected data for burial effects. Based on the conclusions and recommendations described in that report, the final threshold and risk curve for burial as presented here are established.

(2) Trannum (2004) describes the collected data for changes in grain size. Some intermediate threshold values are already described in that study. Chapter 3 of this report presents a final threshold and risk curve for changes in grain size which is to be included in the EIF calculations.

(3) Beardsley & Neff (2004) (Appendix 1) prepared a note on how a PEC/PNEC approach could be applied to oxygen depletion in the sediment. Based on this information, and input from other studies, a threshold value and a risk curve for oxygen depletion are established (chapter 4 of this report).

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 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. Van Straalen & Denneman, 1989, Aldenberg & Slob, 1993; Newman *et al.*, 2000; Van der Hoeven, 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).

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.*, 2005). Here it is assumed that the relation between PAF and actual risk is the same for all stressors under consideration.

Finally, it has to be noted that hardly any standardized effect information for disturbances other than toxicity could be obtained. This is mainly due to the fact that no regulatory framework is available for non-toxic effects. The PEC/PNEC approach for these disturbances is therefore also based on expert opinions by the authors of the different background documents and the other experts present in the different ERMS working groups. Assumptions were made in the derivation of thresholds and sensitivity distributions which, if assessed relevant, should be validated in future by an experimental program. All assumptions are described in the next chapters of this report.

2.   Burial of organisms

2.1   Introduction

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);
–   Burial time;
–   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.

In general, the effect of burial mainly depends on the mobility of organisms in the sediment matrix and on the settling rate of particles. Sedentary organisms, which have no or very limited abilities to move, such as attached barnacles or mussels, are very sensitive. Other species with a low capability to move through the sediment, such as certain bivalve species, may eventually suffer from low oxygen concentrations in the sediment (Essink, 1999).

Most species present in muddy sediments or in high-energy, dynamic sediments are, however, well adapted to changes in their substrate. Especially species with burying behaviour, experience hardly any effect (Bijkerk, 1988).

For most species, the oxygen consumption rate is lower in winter than in summer. This can cause organisms to survive longer in winter after burial. Movement of the organisms, however, is also lower, so it takes longer for the organism to escape

from the layer of burial. The influence of the season on the effect of burial is therefore hard to predict. It depends on the species, location and temperature.

2.2   Effect data

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).

Kranz (1974) studied the effect of burial on bivalve species and showed that the habitats of the species, influence the susceptibility of the individuals to mortality. Species which suffer most from burial with a sediment type different from the native one are the infaunal nonsiphonate suspension feeders, infaunal mucus tube feeders and labial palp deposit feeders. When buried with native sediment, the mucus tube feeders and labial palp deposit feeders seem to be the least affected groups. The group least affected by burial with exotic sediment are infaunal siphon-feeding bivalves. This could be explained by the fact that the members of this group do not demonstrate any significant escape burrowing.

Considering the available data for threshold derivation it must be noted that *depth of burial*, which was measured in the experiments, is not the same as *thickness of the deposited layer*, which is the defined stressor in the exposure model (Smit *et al.*, 2006). The first indicates the thickness of the layer on top of the individual. The second indicates the thickness of the layer covering the sediment. When the individual is living on top of the sediment, it will take some additional layer thickness to bury this individual. In order to bury an adult suspension feeder like a mussel, as tested by Kranz (1974), the deposited layer must at least exceed the dimensions of the exposed shell in order to bury the shell. Using the depth of burial to represent the thickness of the deposited layer is actual an overestimation of the effective level (Figure 2).
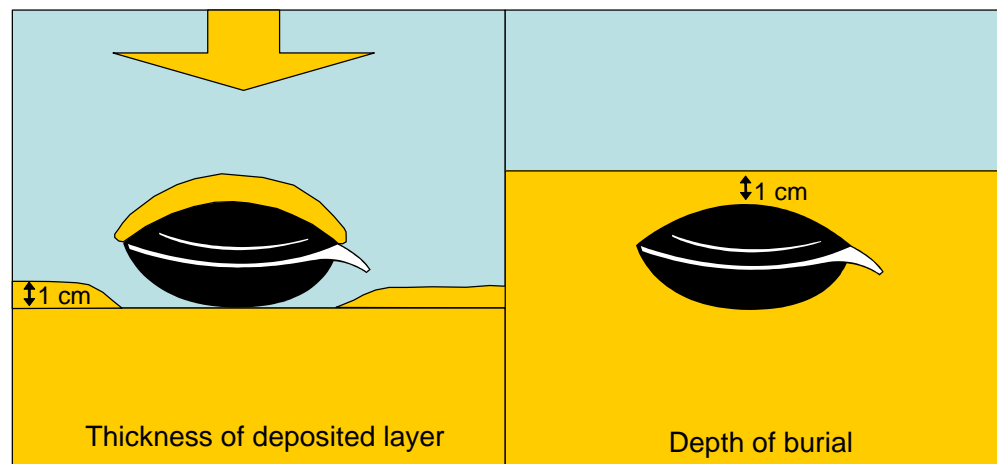
Figure 2   Illustration of the difference between the expression of the PEC (thickness of deposited layer) and the nature of the data used to derive the threshold value (depth of burial)

2.3   Threshold assessment

The PEC represents the exposure, in this case the thickness of the deposited layer. This layer thickness will be modeled by the Marine Environmental Modeling Workbench (MEMW) (Rye *et al*., 2006) on the basis of forecasted release scenarios. The threshold for burial represents the sensitivity of the ecosystem, in this case the threshold value (depth) for adverse effects caused by burial. The threshold for burial is derived from the effect data reported in Kjeilen-Eilertsen *et al*. (2004). 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*., 2005). This methodology, referred to as 'statistical extrapolation methods' facilitates probabilistic risk assessment and is incorporated in the TGD (EC, 2003).

The EU-TGD describes how statistical extrapolation methods can be applied to chronic no-effect data from standardised tests. However, for burial such data is not available. The only data available are effect data (in the range of 0 to a 100% effect) resulting from chronic tests. This implies that it might be necessary to apply additional safety factors to the HC_5. For the selection of the input data, the following has to be noted:
−   In case the effect value is described as 'less than' a certain value, half of that value is taken as input for the SSD, assuming that the value at which no effect is observed (NOEC) is the lowest observed effect value (LOEC) divided by two. This procedure is in line with the Dutch guidelines as described in Traas (2001). For example, For the *Nereis succinea* the threshold value is <30 cm, which means that the threshold value used for the SSD is set at 15 cm.

    – For some of the species (epifaunal suspension feeders, permanently attached to hard substrate) the threshold value was determined at 0 cm, since they could not escape burial of 1 cm depth, which was the lowest exposure level included in the experiments. As it is assumed that effect data are lognormal distributed, a no-effect value of zero cannot be used as an input value in the SSD. As the lowest observed effect level (LOEC) for these species was 1cm, the thresholds are set at 0.5 cm, again assuming that the NOEC is half the no observed effect level (Traas, 2001). Especially in these cases it must be noted that there is a discrepancy in *depth of burial* and *thickness of deposited layer*.

    – In case more than one threshold value is reported for only one species, the average value for that species is taken as the input for the SSD.

Figure 3 shows the cumulative distribution of the species sensitivity (the natural logarithm of the threshold levels reported by Kjeilen-Eilertsen *et al*., 2004). From this sensitivity distribution, the probabilistic value, at which 5% of the species are likely to be affected (HC_5), can be calculated. This value can serve as an intermediate for the PNEC for burial and is determined at a level of 0.96 cm (5 - 95% conf. interval of 0.47 – 1.59 according to the method describe by Aldenberg and Jaworska (2000)). The data points which are used to construct the SSD for burial by native and exotic sediment are listed in Appendix 1.

The reported data indicate that species are more sensitive to burial by exotic sediment than native sediment. A second SSD for burial by exotic sediment only is drawn (see Figure 4). 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 describe by Aldenberg and Jaworska (2000)). This value is expected a better representation of the effects of burial by 'exotic' drilling muds and cuttings. The data points which are used to construct the SSD for burial by exotic sediment are listed in Appendix 1.

For both figures the number of species are equal (n=32). However for some species the data values are different. If more effect values are available for one species, it is common practice to use the geometric mean of the values in the SSD.
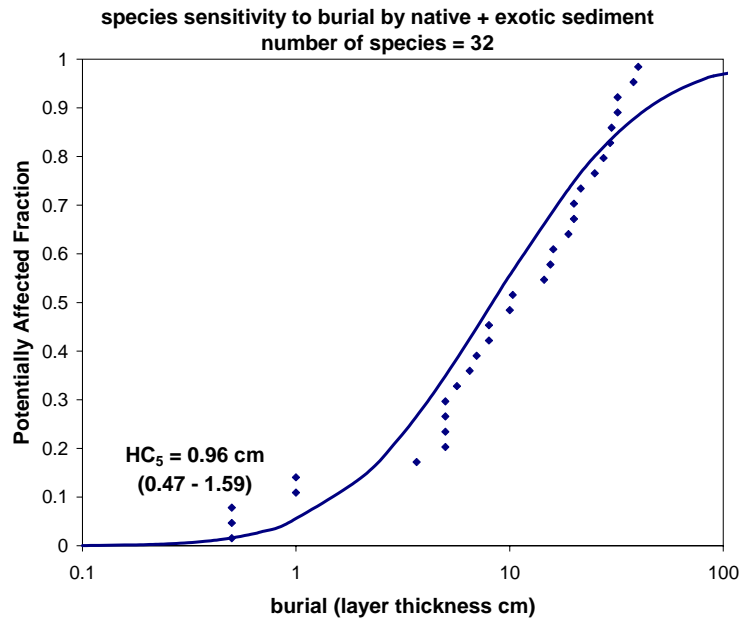
Figure 3   *Species Sensitivity Distribution (SSD) of benthic species based on data on burial by both native and exotic sediments. Data reported by Kjeilen-Eilertsen et al. (2004) (See also Appendix 1).*
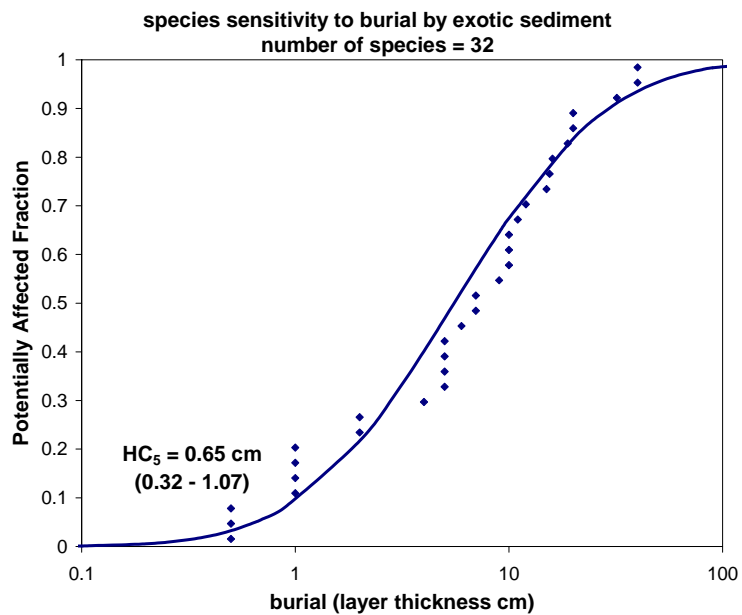
Figure 4   *Species Sensitivity Distribution (SSD) of benthic species for burial by exotic sediment, only. Data reported by Kjeilen-Eilertsen et al. (2004) (See also Appendix 1).*

The level of the PNEC for burial can be derived from the HC_5 by applying safety factors (EU-TGD; EC, 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). The SSD can also be based on other data than NOECs. 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 illustrated by Figure 2).

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).

The values for the Xm (geometric mean of the ln (natural-logarithmic) transformed data) and Sm (standard deviation of the ln (natural-logarithmic) transformed data) describing the SSD for burial by exotic sediments (Figure 4) are respectively 1.71 and 1.30. The values for the Xm and the Sm describing the corresponding PEC:PNEC-to-risk curve (according to Smit *et al.*, 2005) are respectively 2.14 and 1.30.

2.4   Uncertainties in the PNEC

The SSD presented in Figure 4 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.

As there is no effect data available referring to the thickness of the deposited layer, nor on deposition rate, the relevance of the sensitivity data used to derive the PNEC can be discussed. It is reported in a review paper by Berry *et al.* (2003) that overburden or thickness of deposited sediment is an important parameter. However it is also mentioned that the lack of data renders the derivation of threshold values for effect impossible. In addition it is stated that generalisation is impossible as the

sensitivity or dose-response is species specific and substrate dependant. Apart from the arguments mentioned by Berry *et al.* (2003), the procedure described in this report makes use of the best data available and uncertainty is always an integrated part of the risk assessment process. As long as no better data is available the value determined in this report can serve as an estimator for the actual environmental risk.

There is also no quantitative data available on the effects or risks related to size or structure of particles (dose-effect curves). As a consequence there is no possibility to include shape or structure of particles in the exposure (PEC) or the threshold estimates (PNEC) in the risk assessment. In the model, burial is solely described as the thickness of a deposited layer (indifferent of the origin of particles out of which this layer consists). The model is always a simplification of reality. Again, the procedure described in this report makes use of the best data available as no data is available for burial by drilling particles.

For the time being it is suggested to apply the presented threshold of 0.65 cm for modelling the potential impacts for burial and discuss the relevance of risks for burial in relation to other potential impacts (Kjeilen-Eilertsen *et al.*, 2004).

3.   Change in sediment structure (grain size)

3.1   Introduction

Subsequent to the precipitation of mud particles on the seabed, they may be mixed into the top-layer of the sediment as a result of the activity of benthic fauna or physical processes. Following the settlement of drilling particles, benthic communities have been observed to be dominated by opportunistic species (small size, short life spans, high population growth rates), while the infauna was found to be concentrated in the upper portions of the sediment. In contrast, climax communities are composed of more equilibrium-type species (larger size, long life spans, lower population growth), which live deeper in the sediments. Their activities tend to mobilize the sediments. Also benthic communities with intermediate characteristics, concerning their interaction with the sediment, exist (Zajac, 2001). The effects of elevated turbidity and sedimentation on benthic fauna are more significant in environments that have low natural concentrations of fine sediments, particularly in areas dominated by gravel substrata (ICES, 2000).

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. Within the ERMS project a task dedicated to define a threshold for changes in grain size was defined. The results of this task are described by Trannum (2004). This chapter describes the final definition of the threshold and risk function for changes in grain size based on this study by Trannum (2004).

3.2   Effect data

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).

3.3   Threshold assessment

From the analyses reported by Trannum (2004) it was observed that North Sea species are more sensitive to changes in grain size than species from the Norwegian and Barents Sea. The values reported are intermediate values for one final threshold level for grain size changes to be used in the EIF. Transformation of the values reported by Trannum (2004) is needed because:

1. They are defined for three different regions;
    As the EIF-tool will be a generic management tool, no location specific values should be used in the assessments. Clustering the data for different regions reduces the information available to assess the window-of-occurrence for the different species. In order to construct an SSD which represents the variation in the most optimal way, all information of a specific species should be included indifferent of the location where the observations are made. (E.g. If a species A occurs both in the North Sea as the Barents Sea, all observations form both locations should be clustered to assess the grain size preference and range of that species. When the observations from both locations are kept separate, the information on the grain size preference of species A would be reduced.)

2. The window-of-occurrence is expressed as coefficient of variance (COV); The window-of-occurrence indicates the grain size interval which is preferred by a species. The standard deviation represents only 64% of the window-of-occurrence. It was discussed and decided that the 95% interval around the median would be a better estimate of the window-of-occurrence. Besides that, the COV is expressed as a percentage of change. For the derivation of the final PNEC for grain size changes the SSD is based on the absolute width of the

window-of-occurrence (95% interval) in stead of the relative width of the window-of-occurrence as presented in Trannum (2004). The application of a relative window-of-occurrence results in inconsistent risk estimates when increasing or decreasing the median grain size). Expressing the change as an absolute value avoids this problem.

EXAMPLE:
At location A the median grain size is 50 micrometer. There is a specific benthic community adapted to this median grain size (type A). At location B the median grain size is 100 micrometer. Also at this location there is a specific benthic community adapted to this median grain size (type B). If due to a discharge the median grain size at location A changes from 50 to 100 (100% change), the benthic community will change from type A to type B. If due to a discharge at location B the median grain size changes from 100 to 50 micrometer (50% change), the benthic community will change from type B to type A. If the risk is related to the relative change, the risk of changing type A into type B is higher that changing type B into type A. This is very strange because the absolute change in community structure (which is the result of the risk) is equal but opposite. Therefore, the risk cannot be related to the relative change but only to the absolute change (which is equal in both cases (50 micrometer)). Concluding: a change in median grain size from e.g. 50µm to 100µm should result in the same risk as a change from 100µm to 50µm. The absolute change is in both cases 50µm while the relative change in the first case is 100% and in the second case 50%. These relative changes would result in different risk levels, while equal risk levels are expected.

3. Species with limited data points are not excluded;
    If the number of locations where species are observed is limited the window-of-occurrence can be very low. It was decided only to take the species into account which are observed at more than 10 locations. This resulted in 205 different species. A table with the data is included in appendix 2 of this report.

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 (Figure 5).

**Species sensitivity to changes in grainsize
number of species = 300**
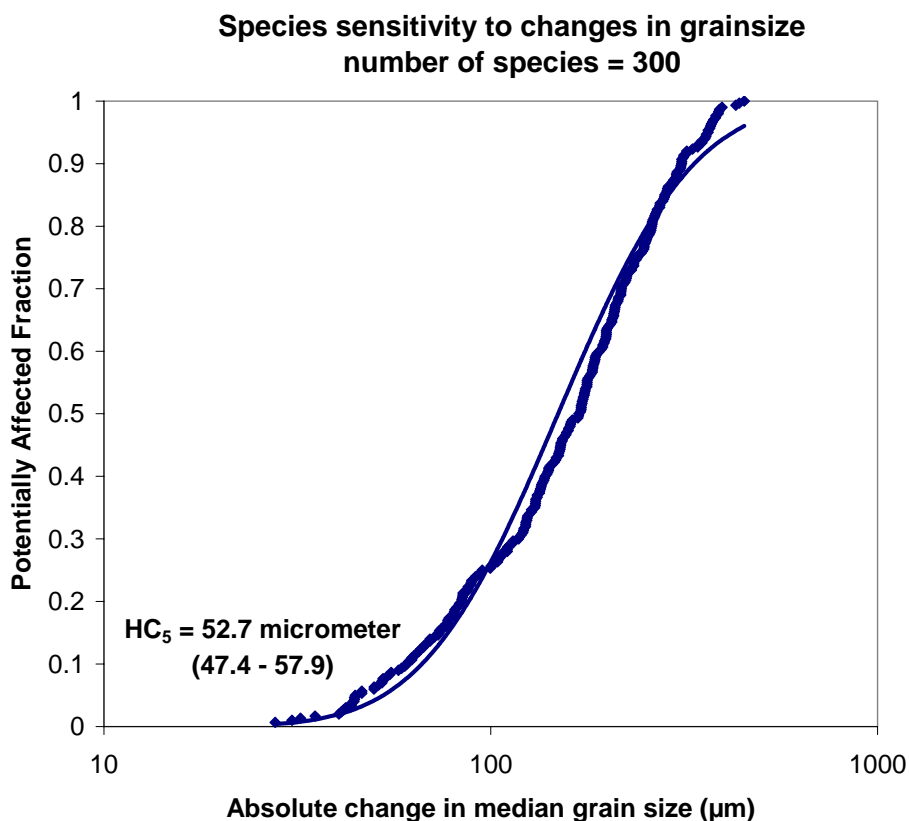
*Figure 5   Species Sensitivity Distribution (SSD) based on the absolute natural grain
size window-of-occurrence (95% interval) of 300 North Sea, Norwegian Sea
and Barents Sea species.*

From the sensitivity distribution presented in Figure 5, 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 describe by Aldenberg & Jaworska (2000).

The SSD presented in Figure 5 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.

The values for the Xm (geometric mean of the ln (natural-logarithmic) transformed
data) and Sm (standard deviation of the ln (natural-logarithmic) transformed data)
describing the SSD for changes in grain size in millimetres (Figure 5) are
respectively -1.90 and 0.63. The values for the Xm and the Sm describing the

corresponding PEC:PNEC-to-risk curve (according to Smit *et al*., 2005) are respectively 1.04 and 0.63.

3.4   Uncertainties in the PNEC

Trannum (2004) recommends to examine whether the most sensitive species (the 5% group of this study where impacts are "accepted"), belong to a particular group or have a particular ecological function. Some species have key functional roles and their removal may therefore have cascading effects in the ecosystem, which means that one ends up with more than 5% risk. Furthermore, if one particular functional group (e.g. species living at the sediment surface) is affected, pelagic species depending on the benthos as a food source may also be affected. The relation between Potentially Affected Fraction (PAF) and actual environmental risk caused by changes in grain size is not known.

The threshold derived for grain size changes could be specified for specific locations or environments with characteristic biota (arctic species, tropical species, deep sea species etc.). However, the data used in this study is not always available for all environments. An evaluation should be performed on the contribution of this stressor after the relative importance of risk from grain size changes compared to potential impacts from other stressors has been indicated.

4.   Hypoxia

4.1   Introduction

Hypoxia is defined as dissolved oxygen in seawater of less than 2.8 mg O_2/l (equivalent to 2 ml O_2/l or 91.4 mM) and below saturation (Wu, 2002). Many ecosystems have reported some type of decline in dissolved oxygen levels through time with a strong correlation with human activities; inputs of nutrients and organic matter (Diaz, 2001). In deep marine areas hypoxia is very dependent on physical conditions. Stratification in combination with hypoxia can occur below the thermocline during summer and during calm weather conditions. The direction of water currents determines the production and supply of organic material. Within the ERMS project a dedicated task was defined to study the possibilities to describe the stress of oxygen depletion in a PEC/PNEC oriented way. Beardsley & Neff (2004) reported their results in a note which is included in Appendix 3 of this report.

The processes that determine the oxygen content in bottom waters are:
–   The consumption of oxygen due to degradation of organic materials in the bottom water and sediments. The consumption rate depends on the amount and quality of organic material sedimenting to the bottom and on the temperature;
–   Consumption by infaunal organisms;
–   The supply of oxygen from vertical mixing and horizontal transport processes. The supply rate depends on the hydrographical processes forced by wind, buoyancy and tides (EPA, 2000).

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) (Smit *et al*., 2006; Rye et al., 2006). 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) (Beardsley & Neff, 2004).

4.2   Effect information

Hypoxia degrades bottom habitat through a wide suite of mechanisms. Under conditions of limited oxygen at the bottom, rates of nitrogen (nitrate) and phosphate remineralization, and sulfate reduction increase. The resulting

production of nitrite, ammonia, and sulfide in combination with low oxygen can be lethal to benthic organisms (Buzzelli et al., 2002). Hypoxia may have several sub-lethal effects on organisms and population by impacting: growth, survival, moult, capture success, feeding, development, hatching, motion, respiration and settlement of individual benthic organisms. In general, the critical dissolved oxygen concentration for survival of most benthic organisms is around 2.8 mg O_2/l, while certain crustacean and zooplankton species could tolerate 0.5-1 mg O_2/l for several days to weeks (Wu, 2002).

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 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).

4.3   Threshold assessment

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 Redox Potential Discontinuity (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 6) for modelling the risk of oxygen depletion.

The values for the Xm (geometric mean of the ln (natural-logarithmic) transformed integrated oxygen concentration) and Sm (standard deviation of the ln (natural-logarithmic) transformed integrated oxygen concentration) describing the risk curve for changes in integrated oxygen content (Figure 6) are respectively -0.86 and 0.42. The values for the Xm and the Sm describing the corresponding PEC:PNEC-to-risk curve (according to Smit *et al.*, 2005) are respectively 0.65 and 0.42.

Predicted community changes related to oxygen depletion
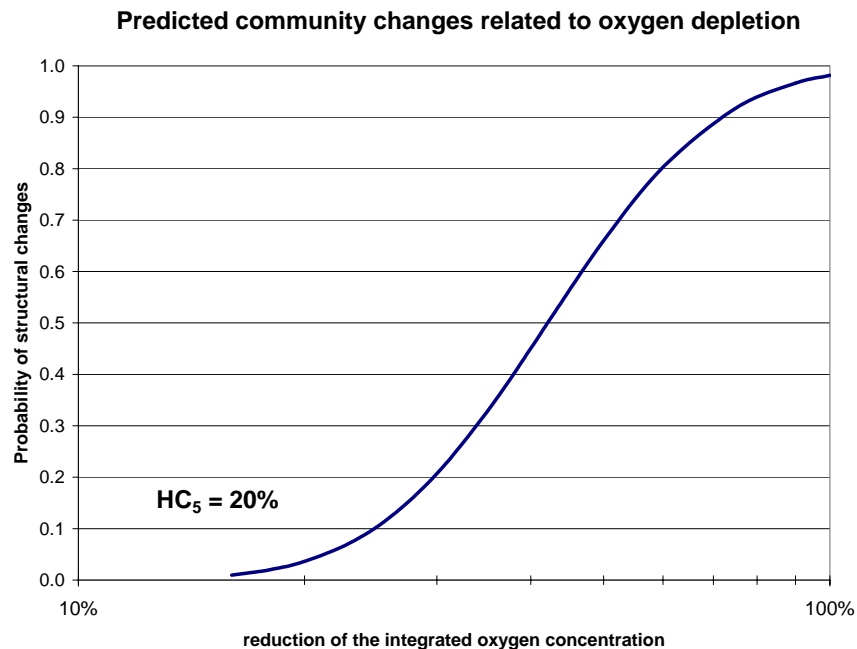
Figure 6   Theoretic risk curve for the reduction of the thickness of the oxygenated layer

4.4   Uncertainties in the PNEC

The SSD presented in Figure 5 together with the threshold level of 20% reduced oxygen were discussed among the experts in the different ERMS-working groups. 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.

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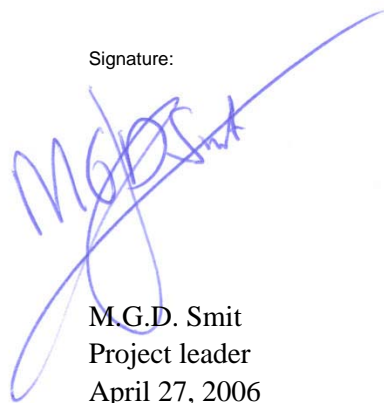
Names and establishments to which part of the research was put out to contract:
-

Date upon which, or period in which, the research took place:
July 2002 – March 2006
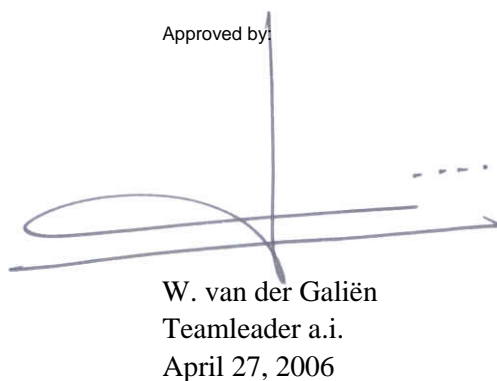
Signature: Approved by:

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Project leader   Teamleader a.i.
April 27, 2006   April 27, 2006

APPENDIX 1:
Data table for burial

Table 2   Data for burial by native and exotic sediments included in Figure 3

Species number	Species name	EP10 (cm)
1	*Crassostrea virginica * *	0.5
2	*Hinnities multirugosus * *	0.5
3	*Modiolus demissus * *	0.5
4	*S. laticauda*	1
5	*Mytilus edulis*	1
6	*Venericardia borealis*	3.7
7	*Acanthocardia echinata*	5
8	*Anadara notabilis*	5
9	*Cerastoderma edule*	5
10	*Laevicardium crasum*	5
11	*Astarte castanea*	5.7
12	*Astarte undata*	6.5
13	*P. longimerus*	7
14	*Cardita floridana*	8
15	*Scoloplos fragilis*	8
16	*Cancer magister*	10
17	*Clinocardium nuttalli*	10.3
18	*Gemma gemma*	14.5
19	*Mercenaria mercenaria*	15.5
20	*Ilyanassa obsoleta*	16
21	*Crangon crangon*	18.8
22	*Heteromastus filiformis*	20
23	*Phaciodes nassula*	20
24	*Nucula proxima*	21.5
25	*Mya arenaria*	25
26	*Yoldia limatula*	27.5
27	*Divaricella quadrisulcata*	29.5
28	*Nereis succinea*	30
29	*Codakia orbicularis*	32
30	*N. sayi*	32
31	*Macoma nasuta*	38
32	*Ensis directus*	40

* for this species an EP10 of 0 was reported. A value of half the LOEC value was used in the construction of the SSD (data is also presented in Kjeilen-Eilertsen, 2004).

Table 3   Data for burial by exotic sediments included in Figure 4

Species number	Species name	EP10 (cm)
1	*Crassostrea virginica* *	0.5
2	*Hinnities multirugosus* *	0.5
3	*Modiolus demissus* *	0.5
4	*Cardita floridana*	1
5	*Mytilus edulis*	1
6	*S. laticauda*	1
7	*Venericardia borealis*	1
8	*Astarte castanea*	2
9	*Phaciodes nassula*	2
10	*Scoloplos fragilis*	4
11	*Acanthocardia echinata*	5
12	*Anadara notabilis*	5
13	*Cerastoderma edule*	5
14	*Laevicardium crasum*	5
15	*Gemma gemma*	6
16	*Astarte undata*	7
17	*P. longimerus*	7
18	*Nucula proxima*	9
19	*Cancer magister*	10
20	*Clinocardium nuttalli*	10
21	*Yoldia limatula*	10
22	*Divaricella quadrisulcata*	11
23	*Codakia orbicularis*	12
24	*Nereis succinea*	15
25	*Mercenaria mercenaria*	15.5
26	*Ilyanassa obsoleta*	16
27	*Mya arenaria*	18.8
28	*Crangon crangon*	20
29	*Heteromastus filiformis*	20
30	*N. sayi*	32
31	*Ensis directus*	40
32	*Macoma nasuta*	40

* for this species an EP10 of 0 was reported. A value of half the LOEC value was used in the construction of the SSD (data is also presented in Kjeilen-Eilertsen, 2004).

APPENDIX 2:
Data table for changes in grain size

Table 4   Data for grain size window-of –occurences included in Figure 5

No.	Species name	Median grain size	95% interval
1	*Abra longicallus*	52.94	138.29
2	*Abra nitida*	69.89	130.28
3	*Abra prismatica*	124.08	177.59
4	*Abyssoninoe hibernica*	70.70	130.61
5	*Acanthicolepis asperrima*	69.35	134.39
6	*Acanthocardia echinata*	105.95	127.33
7	*Acteon tornatilis*	121.95	151.26
8	*Aglaophamus malmgreni*	49.83	69.60
9	*Amaeana trilobata*	68.32	141.26
10	*Amage auricula*	75.17	160.72
11	*Ampelisca brevicornis*	110.41	105.55
12	*Ampelisca gibba*	111.02	153.01
13	*Ampelisca macrocephala*	93.76	76.46
14	*Ampelisca odontoplax*	114.89	172.28
15	*Ampelisca tenuicornis*	114.51	198.89
16	*Ampharete falcata*	97.45	124.08
17	*Ampharete finmarchica*	77.61	149.87
18	*Ampharete lindstroemi*	104.50	214.17
19	*Amphicteis gunneri*	77.96	144.39
20	*Amphilepis norvegica*	33.94	44.45
21	*Amphipholis squamata*	104.46	261.62
22	*Amphiura borealis*	86.76	84.20
23	*Amphiura filiformis*	127.57	216.76
24	*Amythasides macroglossus*	106.21	248.81
25	*Anapagurus laevis*	172.38	285.60
26	*Anobothrus gracilis*	101.04	124.53
27	*Antalis agile*	86.74	167.93
28	*Antalis entale*	136.88	268.63
29	*Antalis occidentalis*	71.85	80.45
30	*Aonides paucibranchiata*	161.25	281.32
31	*Apistobranchus tullbergi*	149.91	351.66
32	*Aponuphis bilineata*	213.03	360.92
33	*Apseudes spinosus*	91.69	269.42
34	*Arctica islandica*	119.46	153.29
35	*Argissa hamatipes*	94.04	58.83
36	*Aricidea catherinae*	108.93	156.34
37	*Aricidea roberti*	87.63	64.30
38	*Aricidea simonae*	99.77	129.89
39	*Aricidea suecica*	127.91	208.58
40	*Aricidea wassi*	97.85	53.94
41	*Aspidosiphon muelleri*	187.80	452.52
42	*Astarte crenata*	89.41	180.09

No.	Species name	Median grain size	95% interval
43	*Astarte sulcata*	150.00	260.48
44	*Astarte sulcata auct.*	139.25	311.47
45	*Astropecten irregularis*	88.25	59.95
46	*Asychis biceps*	54.15	73.06
47	*Atylus nordlandicus*	98.12	194.68
48	*Augeneria tentaculata*	73.82	135.04
49	*Autonoe longipes*	98.46	90.63
50	*Autonoe megacheir*	87.04	147.30
51	*Bathyarca pectunculoides*	86.64	217.23
52	*Branchiomma bombyx*	75.80	80.70
53	*Brissopsis lyrifera*	76.91	27.71
54	*Byblis crassicornis*	81.79	181.52
55	*Byblis gaimardi*	120.59	199.03
56	*Bylgides groenlandica*	42.60	61.18
57	*Calathura brachiata*	63.03	87.38
58	*Campylaspis undata*	97.22	87.40
59	*Capitella capitata*	125.94	234.29
60	*Caulleriella serrata*	83.53	177.27
61	*Cerastoderma minimum*	104.10	194.79
62	*Ceratocephale loveni*	37.15	52.64
63	*Cerianthus lloydii*	146.33	284.29
64	*Chaetoderma nitidulum*	100.70	151.81
65	*Chaetoparia nilssoni*	129.19	203.71
66	*Chaetozone setosa*	99.44	216.77
67	*Chone duneri*	98.96	186.11
68	*Cirratulus caudatus*	82.89	30.64
69	*Cirratulus cirratus*	134.57	240.19
70	*Clymenura borealis*	65.99	110.42
71	*Corophium affine*	91.11	43.58
72	*Cossura longocirrata*	76.90	172.07
73	*Cuspidaria costellata*	111.94	250.87
74	*Cuspidaria cuspidata*	70.15	44.65
75	*Cuspidaria lamellosa*	75.92	151.10
76	*Cuspidaria obesa*	48.68	68.50
77	*Cyclopecten imbrifer*	99.20	125.87
78	*Cylichna alba*	99.45	261.35
79	*Cylichna cylindracea*	101.05	95.02
80	*Dacrydium ockelmanni*	106.44	209.18
81	*Dacrydium vitreum*	60.07	137.10
82	*Dendrotion spinosum*	86.79	123.19
83	*Diastylis cornuta*	113.51	308.24
84	*Diplocirrus glaucus*	103.15	141.18
85	*Ditrupa arietina*	151.89	273.56
86	*Echinocardium cordatum*	96.53	63.34
87	*Echinocardium flavescens*	139.17	266.90
88	*Echinocyamus pusillus*	230.75	364.83
89	*Eclysippe vanelli*	85.96	162.62
90	*Entalina quinquangularis*	34.75	55.29
91	*Eriopisa elongata*	66.99	123.17

No.	Species name	Median grain size	95% interval
92	*Euchone analis*	113.61	209.87
93	*Euchone incolor*	106.16	252.02
94	*Euchone southerni*	140.77	293.77
95	*Euclymene affinis*	84.06	161.47
96	*Euclymene lindrothi*	45.30	76.43
97	*Eudorella truncatula*	80.57	44.31
98	*Eudorellopsis deformis*	98.93	73.71
99	*Eugyra arenosa*	88.73	66.94
100	*Eunice pennata*	88.30	168.66
101	*Eurydice truncata*	93.64	175.33
102	*Exogone hebes*	278.19	390.14
103	*Exogone naidina*	227.53	294.81
104	*Exogone verugera*	171.58	332.23
105	*Gammaropsis cornuta*	297.24	396.12
106	*Glycera alba*	123.50	155.13
107	*Glycera lapidum*	122.91	264.42
108	*Glycinde nordmanni*	174.68	254.37
109	*Gnathia oxyurea*	191.21	381.78
110	*Golfingia margaritacea*	51.47	123.95
111	*Golfingia vulgaris*	80.80	177.52
112	*Goniada maculata*	107.16	174.41
113	*Goniada norvegica*	122.41	196.35
114	*Goniadella bobretzkii*	121.84	175.68
115	*Haliella stenostoma*	72.97	43.56
116	*Haploops setosa*	107.01	205.62
117	*Haploops tubicola*	63.43	80.36
118	*Harmothoe fragilis*	201.99	368.05
119	*Harpinia antennaria*	76.31	101.96
120	*Harpinia crenulata*	63.88	93.45
121	*Harpinia mucronata*	38.59	41.10
122	*Harpinia pectinata*	81.35	142.24
123	*Hemilamprops cristata*	76.30	89.27
124	*Hemilamprops roseus*	92.63	49.88
125	*Hermodice carunculata*	113.39	184.68
126	*Heteranomia squamula*	144.62	371.74
127	*Heteromastus filiformis*	102.78	259.71
128	*Hippomedon denticulatus*	149.54	234.29
129	*Hippomedon propinquus*	89.30	217.87
130	*Hyalinoecia tubicola*	152.22	259.24
131	*Hydroides norvegica*	145.23	239.35
132	*Ilyarachna longicornis*	51.78	83.02
133	*Ischnomesus bispinosus*	40.54	67.44
134	*Janira maculosa*	96.97	156.94
135	*Jasmineira candela*	91.40	149.66
136	*Jasmineira caudata*	114.46	201.69
137	*Jasmineira elegans*	88.67	183.67
138	*Kelliella miliaris*	43.66	60.73
139	*Labidoplax buskii*	97.23	185.36
140	*Laetmatophilus tuberculatus*	59.40	71.51

No.	Species name	Median grain size	95% interval
141	*Lanice conchilega*	127.37	231.13
142	*Laonice sarsi*	105.59	206.78
143	*Leptanthura tenuis*	70.32	84.46
144	*Leptochiton alveolus*	123.33	171.30
145	*Leptophoxus falcatus*	78.36	77.48
146	*Leucothoe lilljeborgi*	180.93	302.10
147	*Levinsenia gracilis*	78.03	146.18
148	*Liljeborgia fissicornis*	93.83	127.38
149	*Liljeborgia pallida*	56.53	86.96
150	*Limatula gwyni*	30.89	32.23
151	*Limatula subauriculata*	352.25	351.47
152	*Limopsis minuta*	79.80	173.58
153	*Lipobranchus jeffreysii*	60.61	84.71
154	*Lucinoma borealis*	107.81	106.69
155	*Lumbriclymene minor*	86.74	178.36
156	*Lumbrineris aniara*	213.61	388.57
157	*Lumbrineris gracilis*	79.47	183.98
158	*Lunatia montagui*	204.32	309.73
159	*Lyonsiella abyssicola*	83.08	122.95
160	*Lysippides fragilis*	89.86	211.50
161	*Macrochaeta polyonyx*	55.03	80.35
162	*Macrostylis longiremis*	62.51	104.54
163	*Maldane sarsi*	54.88	137.66
164	*Melinna cristata*	49.56	69.66
165	*Modiolula phaseolina*	118.55	190.84
166	*Montacuta ferruginosa*	92.71	82.70
167	*Montacuta substriata*	219.18	387.45
168	*Mugga wahrbergi*	113.33	252.95
169	*Myriochele danielsseni*	88.97	151.65
170	*Myriochele fragilis*	81.90	136.27
171	*Myriochele heeri*	104.51	198.23
172	*Myriochele oculata*	123.98	237.61
173	*Mysella bidentata*	122.07	249.29
174	*Mysia undata*	112.54	117.97
175	*Nannoniscus oblongus*	62.69	104.15
176	*Natatolana borealis*	200.08	374.22
177	*Neoleanira tetragona*	56.68	74.57
178	*Nephtys caeca*	112.38	170.54
179	*Nephtys hombergi*	100.37	88.94
180	*Nephtys hystricis*	68.25	158.87
181	*Nephtys longosetosa*	115.37	208.05
182	*Nereimyra punctata*	71.08	91.43
183	*Nothria conchylega*	125.67	281.63
184	*Notolimea sarsii*	119.28	198.45
185	*Notomastus latericeus*	112.38	224.19
186	*Notoproctus minor*	58.58	84.62
187	*Nucula tumidula*	79.05	118.79
188	*Octobranchus floriceps*	109.52	213.23
189	*Onchnesoma squamatum*	96.88	171.49

No.	Species name	Median grain size	95% interval
190	*Onchnesoma steenstrupi*	89.88	182.92
191	*Ophelina abranchiata*	59.18	100.08
192	*Ophelina acuminata*	98.47	135.27
193	*Ophelina cylindricaudata*	88.88	213.50
194	*Ophelina modesta*	74.08	106.90
195	*Ophelina norvegica*	20.83	7.81
196	*Ophiacantha abyssicola*	79.99	130.75
197	*Ophiodromus flexuosus*	109.75	121.85
198	*Ophiopholis aculeata*	102.12	186.94
199	*Ophiura affinis*	146.93	303.07
200	*Ophiura sarsii*	70.20	84.26
201	*Orbinia sertulata*	82.14	49.93
202	*Owenia fusiformis*	140.00	258.28
203	*Paradiopatra fjordica*	113.56	208.00
204	*Paradiopatra quadricuspis*	77.48	223.39
205	*Paradoneis lyra*	90.46	221.99
206	*Paramphinome jeffreysii*	89.73	200.03
207	*Paramphitrite tetrabranchia*	117.56	219.83
208	*Pareurythoe borealis*	92.15	188.43
209	*Pectinaria auricoma*	123.54	231.37
210	*Pectinaria koreni*	105.15	110.27
211	*Peresiella clymenoides*	166.68	358.04
212	*Perioculodes longimanus*	90.55	65.79
213	*Phascolion strombi*	152.26	321.28
214	*Phaxas pellucidus*	100.85	88.77
215	*Phisidia aurea*	132.79	273.27
216	*Pholoe inornata*	150.56	286.31
217	*Pholoe pallida*	103.89	114.06
218	*Phoronida spp.*	94.06	62.54
219	*Phoronis muelleri*	240.56	369.92
220	*Phyllodoce groenlandica*	129.71	265.63
221	*Phyllodoce longipes*	117.15	227.20
222	*Phyllodoce rosea*	115.43	139.73
223	*Phylo norvegica*	57.89	133.57
224	*Pista lornensis*	129.54	274.17
225	*Poecilochaetus serpens*	225.64	302.04
226	*Pogonophora spp.*	71.18	52.32
227	*Polinices pulchellus*	120.47	121.41
228	*Polycarpa fibrosa*	86.05	99.71
229	*Polycirrus arcticus*	93.45	160.28
230	*Polycirrus medusa*	86.21	168.98
231	*Polycirrus norvegicus*	122.58	246.69
232	*Polydora coeca*	99.76	173.24
233	*Polydora socialis*	101.72	147.70
234	*Poromya granulata*	120.56	196.99
235	*Praxillella praetermissa*	130.51	287.75
236	*Prionospio cirrifera*	97.28	217.74
237	*Prionospio dubia*	55.17	62.46
238	*Prionospio fallax*	104.82	121.05

No.	Species name	Median grain size	95% interval
239	*Proclea graffi*	42.29	52.42
240	*Protodorvillea kefersteini*	140.43	311.54
241	*Protomystides exigua*	67.05	131.79
242	*Pseudomystides limbata*	279.01	438.73
243	*Pseudopolydora paucibranchiata*	130.74	279.89
244	*Rhodine loveni*	48.20	57.76
245	*Sabellides octocirrata*	73.84	150.52
246	*Samytha sexcirrata*	116.22	262.37
247	*Samythella neglecta*	89.35	171.34
248	*Scalibregma inflatum*	87.73	174.64
249	*Scolelepis tridentata*	201.62	321.29
250	*Scoletoma magnidentata*	131.94	315.49
251	*Scoloplos armiger*	90.63	130.46
252	*Sige fusigera*	88.78	35.14
253	*Similipecten similis*	200.61	430.17
254	*Siphonodentalium lobatum*	37.85	42.20
255	*Solariella amabilis*	121.29	225.34
256	*Sosanopsis wireni*	204.55	376.77
257	*Sphaerodorum gracilis*	120.96	177.38
258	*Sphaerosyllis hystrix*	396.34	343.66
259	*Sphaerosyllis tetralix*	73.56	81.59
260	*Spio decorata*	97.43	44.44
261	*Spio filicornis*	299.12	309.97
262	*Spiochaetopterus typicus*	74.05	94.18
263	*Spiophanes bombyx*	140.70	244.59
264	*Spiophanes kroyeri*	113.99	224.55
265	*Spiophanes urceolata*	80.22	73.63
266	*Spiophanes wigleyi*	156.62	314.45
267	*Spisula elliptica*	215.98	296.61
268	*Sthenelais limicola*	118.86	140.40
269	*Streblosoma intestinale*	168.00	359.79
270	*Syllidia armata*	100.76	186.07
271	*Synchelidium sp.*	96.54	77.99
272	*Terebellides stroemi*	86.21	198.91
273	*Tharyx killariensis*	90.55	133.90
274	*Therochaeta flabellata*	100.96	205.55
275	*Thyasira croulinensis*	103.61	256.63
276	*Thyasira equalis*	35.86	40.50
277	*Thyasira eumyaria*	33.36	46.44
278	*Thyasira ferruginea*	57.48	54.81
279	*Thyasira flexuosa*	92.65	46.43
280	*Thyasira granulosa*	46.88	77.45
281	*Thyasira obsoleta*	87.96	183.20
282	*Thyasira pygmaea*	76.83	158.54
283	*Thyasira succisa*	164.84	342.90
284	*Timoclea ovata*	153.76	249.25
285	*Tmetonyx cicada*	134.69	278.08
286	*Tmetonyx similis*	76.83	110.36
287	*Trichobranchus roseus*	125.64	311.78

No.	Species name	Median grain size	95% interval
288	*Trypanosyllis coeliaca*	115.84	209.58
289	*Tryphosites longipes*	205.16	364.81
290	*Unciola planipes*	185.62	299.95
291	*Urothoe elegans*	102.93	131.85
292	*Vargula norvegica*	75.30	111.87
293	*Westwoodilla caecula*	121.44	234.22
294	*Yoldiella acuminata*	46.23	65.14
295	*Yoldiella fraterna*	54.14	114.21
296	*Yoldiella intermedia*	36.77	41.68
297	*Yoldiella lenticula*	44.93	51.34
298	*Yoldiella lucida*	58.12	120.02
299	*Yoldiella propinqua*	100.36	193.05
300	*Yoldiella tomlini*	208.21	380.82

APPENDIX 3:

Battelle's background document for the establishment of Non-Toxic (Sediment Disturbance) Thresholds for Drilling Muds and Cuttings - Oxygen Depletion

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1.0 Background

Biodegradable organic matter often has a greater effect than sediment texture and deposition rate or, in some cases, toxic chemicals on the structure and function of benthic biological communities in accumulations of drilling muds and cuttings (cuttings piles) on the sea floor. Bacteria and fungi indigenous to marine sediments degrade the organic matter and, in the process, may deplete the oxygen in the pore water of near-surface layers of sediment and generate potentially toxic concentrations of sulfide and ammonia (Wang and Chapman, 1999; Wu, 2002). Sediments containing high concentrations of biodegradable organic matter, particularly if they are fine-grained, often are hypoxic or anoxic and have markedly altered benthic communities compared to nearby oxygenated sediments. As the organic matter decreases as a result of microbial degradation, the oxygen concentration in the surface layers of sediments increases, resulting in a succession of the benthic community toward a more stable state. This process is called organic enrichment and results in substantial changes in sediment physical/chemical properties and in benthic community structure (Pearson and Rosenberg, 1978). Along a gradient of decreasing sediment organic matter concentration with distance from a source of sediment organic enrichment, such as a sewage outfall or a platform discharging drilling mud/cuttings, the depth of oxygen penetration and the redox potential increase in sediments and the number of species, faunal abundance, and biomass (the SAB ratio) of the benthic community change in characteristic ways.

A typical oxygen penetration depth (the depth at which O_2 concentration drops below about 0.1 mg/l) in uncontaminated fine-grained continental shelf sediments is 1 to 5 cm The concentration of biodegradable organic matter usually is lower in deep-sea sediments, which often have a high porosity, and O_2 penetration depth may increase to more than 10 cm.

Oil-based and synthetic-based drilling muds (OBM and SBM) contain a large amount of biodegradable organic matter. When OBM or SBM cuttings accumulate in a pile on the sea floor, oxygen in the pore water is consumed rapidly, resulting in a steep gradient of decreasing oxygen concentration with depth in the pile. The slope of the vertical dissolved oxygen and redox potential (Eh) gradient in sediment and, therefore, the depth at which Eh reaches 0 mV (the redox potential discontinuity: RPD) and oxygen concentration drops below about 0.1 mg/l (where the sediment becomes reducing and anoxic) depends on the oxygen concentration

in the overlying water, the permeability of the sediment, and the availability of biodegradable organic matter and reducible inorganic substrates in the sediment. Cuttings piles containing high concentrations of organic matter from OBM and SBM often become anoxic within about 1-2 mm of the surface (Figure 1). This steep gradient of decreasing oxygen concentration with depth in the cuttings pile is caused by a combination of rapid microbial degradation of labile organic matter and the low permeability of the cuttings material, which slows diffusion of oxygen into the pile from the overlying water (Schimmield *et al*., 2000; Schimmield and Breuer, 2000). Because the permeability of the cuttings pile sediments usually is low, the mass or the cuttings pile (the height of the cuttings pile) and the rate of cuttings deposition have little effect on the RPD depth in the pile.

Water based mud (WBM) cuttings do not support large populations of bacteria (Dow *et al*., 1990). Dow *et al*. (1990) layered WBM or OBM cuttings on natural sediments in mesocosms. There was only a slight and short-term reduction in the RPD depth in sediments containing WBM cuttings, whereas in sediments containing OBM cuttings, RPD depth decreased to near the sediment surface and did not return to normal in one year.
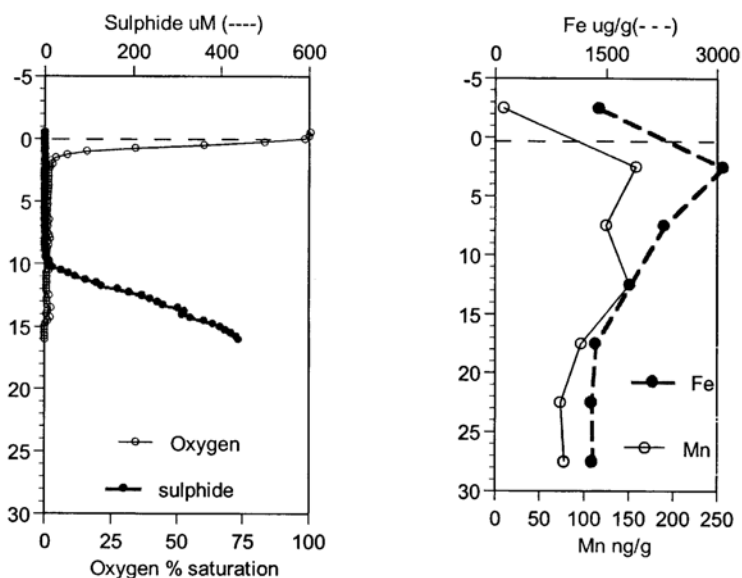

Figure 1. Vertical profiles of oxygen, sulphide, iron, and manganese concentrations in core from the apex of the Beryl A cuttings pile, about 65m from the platform. From Schimmield and Breuer (2000).

In a WBM cuttings accumulation, sulphate reducing bacteria can use carbohydrates and acetate (from WBM solids) as an organic carbon source. Most of the organic chemicals in WBMs are biodegradable under aerobic conditions (Table 1). Organic chemicals in settled solids from mineral oil- and diesel fuel-contaminated WBM solids containing 2,000 to 7,000 mg/kg total organic carbon have a high COD and BOD (Breteler *et al*. 1988). Vrdolkak (1993) reported that exposure to the

suspended particulate phase of WBMs caused stimulation of growth of bacteria and yeasts. Microbial production initially was depressed by some WBMs, but then was stimulated upon longer exposure. This could have been caused by the toxicity to microbes of some mud ingredients. Chrome lignosulphonate was toxic to the bacteria at concentrations of 15,000 to 30,000 mg/L and Thin X (a potassium chloride/polymer mixture) was toxic at a concentration of 100,000 to 250,000), probably due to ionic effects. When these slightly toxic ingredients were diluted, bacterial growth increased under aerobic conditions. Anaerobic biodegradation is slower and less energetically efficient than aerobic degradation.

Table 1. Rates of microbial biodegradation of several organic components of WBMs used in the Mediterranean Sea. Degradation rates were measured as chemical oxygen demand (COD) and 28-day biological oxygen demand (BOD). From Terzaghi et al. (1998).

Chemical Product	COD (mg O_2/mg OC)	BOD (mg O_2/mg OC)	% Degraded in 28 Days
Lignosulphonate	0.923	0.249	27
Modified Starch	0.732	0.531	73
Soltex	0.667	0.094	14
XC-Polymer	0.832	0.355	43
Mor-Rex	0.990	1.134	>100
Carboxymethylcellulose	0.750	0.496	50
PAC	0.750	0.212	21
Detergent	0.238	0.197	83
Defoamer	3.389	1.078	32
Sodium Acetate	0.572	0.716	92

Most of the benthic fauna of continental shelf and slope sediments live in the upper few cm of the sediment column, particularly if the sediments are fine-grained. The organisms that live in the near-surface benthos depend on dissolved oxygen for normal respiratory function. If the oxygen concentration in surface layers of sediment falls below a critical threshold level, the more sensitive members of the benthic community will be lost and possibly replaced by hypoxia-tolerant opportunistic species. Low oxygen concentrations in sediments may have a variety of sublethal effects on benthic fauna, including reduced oxygen consumption, growth rate, and reproductive success. Since many benthic fauna utilize the oxygen in the thin oxidized layer of sediment, the best way to express the critical oxygen concentration for benthic fauna is as the depth-integrated amount of oxygen per unit area of sediment or as the RPD depth (assuming normoxic bottom water).

Organic enrichment of sediments often produces a reduction in the number of species and an increase in the abundance of a few stress-tolerant species (Pearson and Rosenberg, 1978). Schaanning *et al.* (1996) observed this in the NIVA simulated seabed chambers dosed with some types of SBM and OBM cuttings. Effects of thin layers of SBF cuttings on the benthic fauna in the test chambers were examined in some of the simulated seabed studies. Fauna were enumerated at

the beginning and end of the tests. Two ester SBMs had the greatest impacts on the benthic fauna in the chambers (Table 2). Exposure to a thin layer of ester II (Petrofree) resulted in a substantial decrease in the number of species and individuals in the chambers and low diversity indices. In the Petrofree chambers, most species of benthic fauna were eliminated or their numbers were greatly reduced compared to controls. The abundance of a pollution-tolerant polychaete increased. It represented more than half the individuals in the Petrofree chambers. Several of the species that disappeared from the sediments are known to be intolerant of low oxygen concentrations in sediment pore water. Mineral oil was less harmful than ester II to the benthic fauna; Ester I (Anco Green) and IO were essentially without effects on the benthic fauna in the exposure chambers and effects of LAO were minor.

Table 2. Effects of SBM cuttings layered (1.4 to 1.8 mm) on natural sediments in NIVA simulated seabed chambers on characteristics of benthic communities after 187 days. From Schaanning et al. (1996).

SBF Cuttings	No. Species	No. Individuals	Diversity (H)	Diversity (ES_{100})
Control	36 – 39	281 – 856	2.97 – 3.65	17.74 – 22.94
Ester I	14 – 35	283 – 809	2.20 – 2.31	11.12 – 13.66
Ester II	4 – 6	32 – 83	0.87 – 1.90	---
IO	30 – 36	588 – 647	2.70 – 3.33	13.89 – 19.71
LAO	22 – 26	308 – 338	2.88 – 3.16	14.33 – 16.44
Mineral Oil	18 – 20	226 – 309	2.51	13.18 – 13.65

The harmful effects of the SBM cuttings on benthic fauna, as measured by diversity indices, were correlated to sediment redox potential (Eh), an indication of oxygen availability in sediments (Figure 2). Sediments with an Eh approaching 0 mV are hypoxic. The lowest Eh values were produced by the esters that are highly biodegradable. The other SBM base chemicals and mineral oil biodegrade slowly and did not markedly decrease sediment Eh.
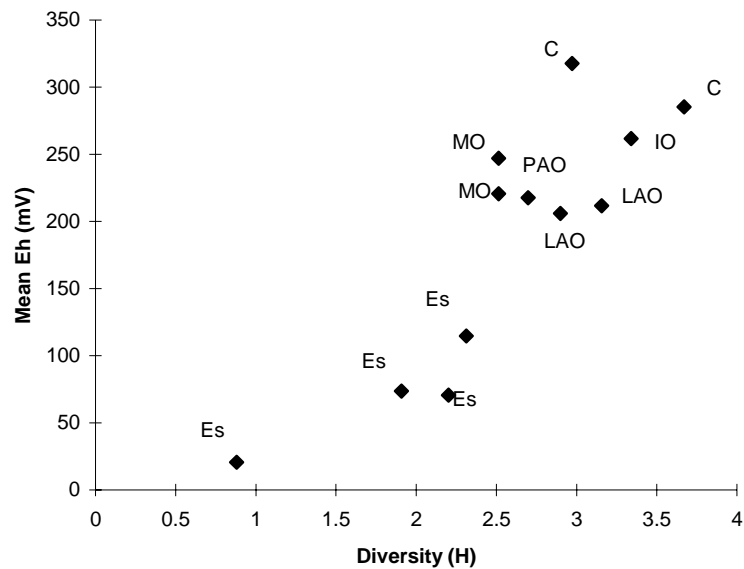
*Figure 2.   Relationship between redox potential and diversity for NIVA seabed
             simulation studies (C=control, MO=mineral oil, Es=ester, PAO, IO,
             LAO = olefins). From Schaanning et al. (1996)*

There were changes in the activity of oxidative enzymes (glutathione reductase and catalase) in the tissues of polychaete worms *Hediste (Nereis) diversicolor* following exposure to SBM cuttings (Schaanning *et al.*, 1996). The enzyme activity responses indicated that the worms exposed to Anco Green (ester), LAO, and IO cuttings were experiencing oxidative stress, probably resulting from a decrease in oxygen concentration in the sediments. The worms did not survive exposure to Petrofree (ester). Benthic invertebrates in the NIVA chambers were adversely affected by a reduction in sediment oxygen concentration caused by organic loading of the sediments with SBF base chemicals, not by the toxicity of the chemicals.

However, it should be kept in mind that many benthic invertebrates, particularly those that construct and irrigate burrows (esp. crustaceans and polychaetes) or have respiratory siphons (bivalve mollusks), extract oxygen from the overlying water column. High-resolution depth profiles of oxygen concentration versus depth in sediment (Figure 1) rarely are measured in monitoring studies of impact of drilling waste discharges on the sea floor environment. Therefore, it will be difficult to model or monitor effects of oxygen depletion on benthic fauna in this way. There are some data available on the effects of oxygen concentrations on survival of benthic organisms and on benthic community structure and function. In these studies, average oxygen concentration often is measured or estimated for pore water in the surface layer (0 to about 5 cm) of sediment. Thus, critical oxygen

concentration will be measured as the concentration of oxygen in sediment pore water below which benthic fauna suffer significant mortality or sublethal effects.

To better understand the ecological impacts associated with drilling muds, the ERMS team is attempting to identify the critical oxygen concentration at which the survival or structure of the endemic benthic community will be compromised. Historical studies suggest that the critical oxygen concentration is dependent on a site's physical characteristics (temperature, salinity) and frequency and longevity of low oxygen events. These variables should be taken into account while trying to refine the nontoxic oxygen concentration benchmark in regions impacted by drilling muds.

2.0 Objective
The objective of this task is to identify the critical oxygen concentrations at which benthic species may be adversely impacted in order to better assess ecological impacts of drilling muds to benthic communities.

3.0 Strategy
A review of published literature was conducted to identify impacts of oxygen depletion on benthic macrofauna and to discern specific benchmark oxygen concentrations at which the macrofauna seem to consistently be impacted. Oxygen availability is significantly affected by an environment's temperature and salinity, so in cases where oxygen concentration data was gathered in contrast to oxygen tension, temperature and salinity are noted where possible. The articles reviewed for this exercise are listed in Section 5.0.

4.0 Results
Oxygen depletion and its impacts on aquatic organisms has been studied for a number of decades. A number of researchers have specifically focused on the "critical" concentration of oxygen, or the level at which organisms are adversely impacted or flee a particular habit. They have found that it is difficult to develop a comprehensive critical oxygen concentration, since oxygen consumption and tolerance can vary with water temperature, salinity and other general environmental characteristics (Staaland and Mostad, 1972), as well as life history characteristics of the fauna, such as developmental stage, reproductive status, nutrition, and previous history of exposure to low dissolved oxygen concentrations (Morrison, 1971). Furthermore, fewer studies have been performed on the impacts of oxygen depletion on benthic communities. A few groups have reviewed the research done in this area and attempted to identify critical oxygen concentration trends. These findings, along with the issues that should be considered while evaluating the results, are discussed below and illustrated in the table following the text.

In the late 1970's and early 1980s, researchers identified several widespread areas of oxygen depletion and its impacts on biological resources (i.e. New York Bight,

Gulf of Mexico). The Ocean Assessments Division of the National Oceanic and Atmospheric Administration (NOAA) reacted by initiating a review in 1984 of the existing knowledge of the distribution of serious oxygen depletion in U.S. coastal waters (LUMC, 1985). For the Nationwide Review Project, NOAA established that all coastal waters with oxygen values less than 4 mg/l were noted in their study as oxygen depleted. The authors of the final report note that oxygen concentrations of 2-3 mg/l may be more realistic benchmarks of oxygen depletion in the warmer waters of the Gulf of Mexico. Furthermore, they report that field studies have shown that oxygen concentrations affecting populations of demersal fishes and macroinvertebrates are closer to 2-2.5 mg/l. The areal extent of bottom water hypoxia (dissolved oxygen concentration ≤ 2 mg/l) on the continental shelf of the US Gulf of Mexico in some summers exceeds 9,500 km^2 (Rabalais *et al* ., 1004).

Another major review of the effects of hypoxia on benthic fauna was published in 1995 (Diaz and Rosenberg, 1995). Based on references to this review in more recent publications, this seems to be the current "state of the science" on the ecological effects and behavioral responses of benthic macrofauna to marine benthic hypoxia.

Diaz and Rosenberg (1995) define hypoxia as dissolved oxygen concentrations ranging between 0-2 mg/l, suggesting that benthic infauna usually can tolerate oxygen concentrations down to at least 2mg/l without major effects. They do note however that many benthic environments are subject to a strong vertical oxygen gradient, meaning that the oxygen levels in the sediments decline quickly as a result of oxygen demands. They also note that this oxygen gradient is subject to temporal and spatial variations at the particular site. They state: "There is a complicated interplay between oxygen concentration and sediment geochemistry that regulates the response of organisms to declining dissolved oxygen concentrations. For example, prolonged hypoxia or anoxia allow sulphate-reducing bacteria, typically found deeper in sediments during normoxia, to survive in surface sediments where they can produce potentially lethal concentrations of hydrogen sulfide."(Diaz and Rosenberg, 1995)

They also note that dissolved oxygen concentrations critical to most benthic organisms vary depending on the environment. For example, in stagnant or semi-stagnant areas such as fjords and protected embayments, the dissolved oxygen concentration critical to most benthic organisms appears to be about 1.96 mg/l (Diaz and Rosenberg, 1995). In areas with seasonally varying hypoxia, such as estuaries and open coasts, the critical oxygen concentration is close to 0.98 mg/l with the reduction from 0.98 to 0.56 mg/l being critically important. Furthermore, offshore benthos from environments that are exposed to extended periods of hypoxia have been noted to tolerate even lower oxygen levels. Critical oxygen levels in offshore areas have been reported as low as 0.14 mg/l (Volcano 7 site) and 0.28 mg/l(offshore of the Peruvian Coast). The benthic fauna in these areas may have evolved to adapt to the extended periods of low oxygen.

Diaz and Rosenberg (1995) do provide a range of critical oxygen concentrations cited in the literature which serve to connect acute effects (mass mortality of benthic species) to a specific oxygen concentration. The values range from 0.28 mg/l to 1.96 mg/l. Study specifics are provided in the table below (in bold), but they continue to stress that the duration of the hypoxic state, sediment quality (i.e. sulfide content), temperature and other physical conditions will all impact the critical oxygen concentration on a site specific basis.

In a more recent review, Wu (2002) reviews the chronic responses (biochemical, physiological and behavioral) to different levels of hypoxia Wu (2002) retains the definition established in the 1995 review that "hypoxia is defined as dissolved oxygen less than 2.8 mg/l". He further examines how organisms respond in biochemically, physiologically or behaviorally to cope with low oxygen and at what concentrations these responses are initiated. The results of this synthesis are summarized in Table 3.

Table 3. Critical oxygen concentrations for benthic species established in historical literature.

Species	Region	Critical Oxygen Concentration (mg/l)	Study Citation
Benthic invertebrates (up to 37 different species)	Black Sea, Gulf of California, continental borderland basins of California	<0.14 is azoic; 0.42-1.4 supports low diversity of small soft bodied infauna; >1.4 species diversity increases significantly.	Nichols (1976)
Benthic mortality	Gullmarsfjord, Swedish West Coast	0.28-1.12	Josefson and Widborn, 1988 in Diaz and Rosenberg, 1995
Benthic mortality	North Sea, Danish and German Bight	0.98	Rachor and Albrecht, 1983, Niermann *et al*, 1990, Arntz 1981 in Diaz and Rosenberg, 1995
Benthic mortality	Kiel Bay, Germany	1.4	Arntz 1981 in Diaz and Rosenberg, 1995
Benthic mortality	Kattegat, Sweden-Denmark	1.26-1.4	Baden *et al*, 1990, Joesefson and Jenson, 1992, Rosenberg *et al* 1992 in Diaz and Rosenberg, 1995
Benthic mortality	Louisiana-Texas Continental Shelf	1.96	Harper *et al*, 1981, 1991, Boesch and Rabalais, 1991.
Acartia tonsa, Zooplankton	NA	.18	Wu, 2002; EPA, 2002
Americamysis bahia, Crustacea	NA	1.25	Wu, 2002; EPA, 2002
Ampelisca abdita, Crustacea	NA	0.9	Wu, 2002; EPA, 2002

Callinectes sapidus, Crustacea	NA	0.76	Wu, 2002; EPA, 2002
Cancer irroratus, Crustacea	NA	2.04	Wu, 2002; EPA, 2002
Carcinus maenus, Crustacea	NA	0.38	Wu, 2002; EPA, 2002
Centropages hamatus, Zooplankton	NA	0.14	Wu, 2002; EPA, 2002
Crangon septemspinosa, Crustacea	NA	0.91	Wu, 2002; EPA, 2002
Crassostrea virginica, Bivalve	NA	1.19	Wu, 2002; EPA, 2002
Dyspanopeus sayi, Crustacea	NA	2.04	Wu, 2002; EPA, 2002
Eurypanopeus depressus, Crustacea	NA	1.91	Wu, 2002; EPA, 2002
Eurytemora affinis, Zooplankton	NA	0.91	Wu, 2002; EPA, 2002
Homarus americanus, Crustacea	NA	2.54	Wu, 2002; EPA, 2002
Labidocera aestiva, Zooplankton	NA	0.38	Wu, 2002; EPA, 2002
Labinia dubia, Crustacea	NA	2.15	Wu, 2002; EPA, 2002
Loligo pealiii, Octopus	NA	1.53	Wu, 2002; EPA, 2002
Mercenaria mercenaria, Bivalve	NA	0.75	Wu, 2002; EPA, 2002
Mulinia lateralis, Bivalve	NA	0.9	Wu, 2002; EPA, 2002
Mytilus edulis, Bivalve	NA	1.4	Wu, 2002; EPA, 2002
Octopus burryi, Octopus	NA	2.99	Wu, 2002; EPA, 2002
Palaemonetes pugio, Crustacea	NA	1.07	Wu, 2002; EPA, 2002
Palaemontes vulgaris, Crustacea	NA	1.7	Wu, 2002; EPA, 2002
Rithropanopeus harrisii, Crustacea	NA	0.51	Wu, 2002; EPA, 2002
Spisula solidissima, Bivalve	NA	0.44	Wu, 2002; EPA, 2002
Tortanus discaudatus, zooplankton	NA	0.28	Wu, 2002; EPA, 2002

Given the variability in responses of benthic invertebrates to dissolved oxygen concentration and the resulting critical oxygen concentrations, it is doubtful that a technically sound critical concentration can be established for all North Sea cuttings accumulations. Although sediments in much of the central and northern North Sea are well-oxygenated sands, there are depositional areas where fine-

grained sediments, often containing elevated concentrations of organic matter, accumulate. It is probable that the benthic fauna of the different sediment types have different low-oxygen tolerances and will respond differently to organic enrichment and the resulting decrease in RPD depth. The best approach for developing a critical oxygen concentration for dissolved oxygen in surface sediments may be to establish a critical difference in concentration or RPD depth between the cuttings accumulation area (usually within about 200-500 m of the discharge) and a nearby reference area, 1-3 km distant, where the absence of cuttings is indicated by background concentrations of barium.

5.0 References

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