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Dynamic modeling of environmental risk associated with drilling discharges to marine sediments

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ABSTRACT

Drilling discharges are complex mixtures of base-fluids, chemicals and particulates, and may, after discharge to the marine environment, result in adverse effects on benthic communities. A numerical model was developed to estimate the fate of drilling discharges in the marine environment, and associated environmental risks. Environmental risk from deposited drilling waste in marine sediments is generally caused by four types of stressors: oxygen depletion, toxicity, burial and change of grain size. In order to properly model these stressors, natural burial, biodegradation and bioturbation processes were also included. Diagenetic equations provide the basis for quantifying environmental risk. These equations are solved numerically by an implicit-central differencing scheme. The sediment model described here is, together with a fate and risk model focusing on the water column, implemented in the DREAM and OSCAR models, both available within the Marine Environmental Modeling Workbench (MEMW) at SINTEF in Trondheim, Norway.

1. Introduction

During offshore drilling operations the vast majority of the waste discharges to the sea are drill cuttings and drilling fluids. Parts of these discharges will remain suspended in the water column while the denser fraction sinks to the seafloor (Neff, 2010). As a result of deposition to the seafloor, adverse environmental effects may be observed in the marine sediments (e.g. Meinhold, 1998). Impacts are primarily due to oxygen depletion (as a result of biodegradation), toxicity of chemicals used in the drilling fluids and physical stress as a result of accumulation of particles on the seabed. Water-based muds (WBM), oil-based muds (OBM), and synthetic-based muds (SBM) are the three main categories of drilling fluids which consist of a base fluid and several chemical components. In addition, the volume of rock cuttings produced during the drilling process may amount to several hundred cubic-meters. Although as the cuttings themselves are non-toxic, there is a potential for burial of organisms during deposition and change of the original sediment characteristics (e.g. grain size distribution) which might cause an alteration of sediment communities. Physical effects of deposition of cuttings on the seabed should therefore be considered in the evaluation of environmental risks in marine sediments (Irvine et al., 2009; Smit et al., 2008a,b; Trannum et al., 2010). Rye et al. (2006, 2008) described the basic equations for calculating the levels of burial of natural sediments, the change of natural grain size, the process of oxygen depletion and toxicity after deposition of drilling discharges. In this paper, we focus on the sediment processes that determine the level of these stressors in the sediment after deposition and how these levels are used for the calculation of environmental risk of drilling discharges.

Early diagenetic processes refer to the physical, chemical, and biological transformations that occur in the surface layer of aquatic sediments following deposition (Berner, 1980; Boudreau, 1997). The theory of early diagenesis is based on the mass conservation of a particular chemical species in the sediment, and is subject to the physical phenomena of burial, bioturbation/dispersion, and degradation. The theory provides a model (i.e. the diagenetic equations) which predicts dynamic concentration profiles of the chemical species in the sediment column.

Diagenetic models have been applied before for environmental modeling and numerous applications and solutions to be found in the literature. Such solutions have been compiled by multiple authors (e.g. van Genuchten and Alves, 1982; Lindstrom and Boersma, 1989; Boudreau, 1997). Moreover one can find further applications of the advection–diffusion–reaction equation for specific conditions. For instance Freijer et al. (1998) have presented

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Nomenclature

B	concentration profile of added chemical in sediment (g/m^3 dry sediment)
C	concentration profile of natural organic matter/carbon in sediment (g/m^3 dry sediment)
D_B	dispersion of organic matter in sediment by bioturbation (cm^2/h)
D_O	diffusion coefficient of oxygen in pore water (cm^2/h)
G	median grain size profile of new mixed sediment (mm)
H	water column depth above sediment (m)
k_B	biodegradation rate of added chemical (1/h)
k_C	biodegradation rate of natural carbon (1/h)
K_O	half-saturation coefficient of oxygen content (mg/L)
L_{tox}	toxicity calculation depth (cm)
L_x	bioturbated mixing depth (cm)
O	concentration profile of dissolved oxygen in pore water (mg/L)
S_m	standard deviation of distribution for risk function (dimensionless)
t	simulation time (h)
w	natural burial rate (cm/h)
X_m	mean of distribution for risk function (dimensionless)
z	vertical position in sediment (cm)
γ_C	ratio of oxygen amount consumed for per amount of organic carbon degraded
γ_B	ratio of oxygen amount consumed for per amount of added chamical degraded
μ	dynamic viscosity of water (cp)
θ	tortuosity factor of sediment pore space (dimensionless)
ϕ	sediment porosity (fraction)
Φ	fractional concentration profile of deposited particulate material in sediment (fraction)

an analytical solution to describe leaching and degradation of pesticides in a specific type of column experiment; Kumar et al. (2009) have presented analytical solutions for with variable coefficients. These approaches often include an analytical solution of the diagenetic equation under specific conditions (e.g. steady-state solutions, constant-time invariant model parameters, given initial concentration profiles, specified boundary conditions). However, the diagenetic equation can also be solved numerically, allowing investigators to extend the solutions beyond the limitations of analytical approaches. Boudreau (1997) has reported several different methods and standard computer codes that can be applied. Meysman et al. (2003) investigated complexity and software/code quality of three publicly available models (OMEXDIA, Soetaert et al., 1996; STEADYSED, Wang and Van Cappellen, 1996; CANDI, Boudreau, 1996). These recent generation diagenetic models include all redox zones in the sediment and incorporate extensive species and reactions. Sabeur et al. (2002) attempted to quantify the effect of contaminants on the receiving environment with a so-called long-term model which is an explicit finite difference solution with a forward time/centered space scheme. However, this model ignored any dependency of the governing parameters on environmental impact caused by deposition itself. Rye et al. (2006) developed a sediment model using diagenetic equations to predict the fate of discharges of drill cuttings and mud for the purpose of environmental risk assessment. Following the descriptions from Rye et al. (2006) we developed a methodology to calculate environmental risks in the sediment based on diagenetic equations and incorporated dependency of the controlling parameters.

2. Material and methods

Drilling waste discharges generally sink to the bottom of the sea because of the higher densities of the discharged materials. Even less dense components in the discharge that initially stay in suspension may eventually be carried to the seafloor due to adhesion to or agglomeration with sinking particles. So the marine sediment model should address the environmental risks associated with discharged and deposited materials on the sediment. In other words, the sediment model should compute measures of potential environmental impact (i.e. stressors) and integrate them into the model.

2.1. Environmental stressors in marine sediments

Four environmental stressors have been identified that contribute to the environmental risk of drilling discharges on sediments (Smit et al., 2008a); oxygen depletion, toxicity, burial, and grain size change.

2.1.1. Oxygen depletion in sediment

Biodegradation of both added chemicals and accumulated natural carbon in the sediment results in depletion of dissolved oxygen content in pore water. In order to assess the effect of drilling discharges on the dissolved oxygen profile in pore water, first the undisturbed profile needs to be established. Depletion of the oxygen content after discharge can then be calculated for each time step by estimating the difference of the simulated oxygen profile from the initial one:

$$\text{Oxygen depletion}(t) = \frac{\left[\int_0^L \phi O(z,t)dz\right]_{t>0} - \left[\int_0^L \phi O(z,t)dz\right]_{t=0}}{\left[\int_0^L \phi O(z,t)dz\right]_{t=0}}, \quad (1)$$

where $O(z,t)$ is the dissolved oxygen content of the pore water in mg/L or g/m^3. The porosity, ϕ, is included to account for the fact that only a part of the sediment volume (given by the porosity) is occupied by the pore water. However, by assuming that porosity in the sediment layer is constant porosity is cancelled out. Since the oxygenated layer corresponds to the layer where bioturbation takes place, the oxygen depletion parameter is calculated by integrating the oxygen content over the bioturbation layer. The integrated oxygen content has units of the amount of dissolved oxygen in pore water per the unit area of the sediment surface, i.e. g/m^2. Generally, the oxygen content is close to zero at the sediment depth L. This ensures that approximately all the dissolved oxygen in the sediment is included in the oxygen depletion parameter.

2.1.2. Toxicity in sediment

Chemicals and heavy metals attached to particles from the drilling discharge may have toxic effects on marine organisms. These components may be mixed down by bioturbation into the original sediment layer and their concentrations in the sediment determine the level of toxicity. The concentration in the bioturbation layer is computed by integrating and averaging over this upper layer: