# Environmental Risk Assessment of Produced Water Discharges on the Dutch Continental Shelf

P. de Vries & C.C. Karman

Report number C012/09

Institute for Marine Resources and Ecosystem Studies

Wageningen IMARES

Location Den Helder

Client: Rijkswaterstaat Dienst Noordzee
T.a.v. Rik Duijts
Lange Kleiweg 34
2288 GK Rijswijk

Publication date: February 2009

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# Summary

The OSPAR Offshore Industry Committee (OIC) has decided, in its meeting of 2008, to evaluate the possibility of implementing a risk based approach towards produced water management. Currently, Norway has made most progress in this field as it has fully implemented the Environmental Impact Factor as the basis of their biannual reporting obligations. The Netherlands has for as yet mainly followed a source (immission) based approach, and therefore did not adopt a specific risk based approach. In this study an overview is provided of current approaches to assess the ecological risk of produced water discharges and it is investigated how these approaches can be used in the Dutch situation for produced water management as intended by the OIC.

In the risk management cycle, both exposure and effects need to be assessed in order to characterise environmental risk. There are different approaches for the assessment of exposure and effect, but in general a tiered approach is applied. In a tiered approach a first tier screening is performed based on simple, but realistic worst-case assumptions. Most risk assessment tools use Predicted No Effect Concentrations (PNECs) to evaluate effects; where a PNEC is the concentration of a substance below which unacceptable effects on organisms will most likely not occur. Where the Predicted Environmental Concentration (PEC) is evaluated against the PNEC. Higher tier assessment can be performed after first tier screening indicated potential adverse effects. Higher tier studies will be more specific but requires more detail, data and modelling effort. Some applications implement Species Sensitivity Distributions (SSDs) in order to quantify effects as a second tier. SSDs are statistical distributions of No Observable Effect Concentrations (NOECs) and can be used to determine the Potentially Affected Fraction (PAF). Body Burden approaches have been developed as a third tier, where effects on species are based on internal concentrations.

Currently, many tools are available to determine exposure levels resulting from produced water discharges. A number of tools use particle tracking to simulate physical processes involved in the dispersion of the components in produced water. The DREAM / EIF (Norway), PROTEUS (United Kingdom), MIKE (Denmark) and Delft3D all use a similar approach to model the dispersion of discharged components. A mathematically much less demanding exposure assessment is applied in the Dutch Chemical Hazard Unit (CHU). In the CHU, a dilution factor of 1,000 is used to determine the Predicted Environmental Concentration (PEC) at 500 m distance from a reference platform. This dilution factor, originates from the CHARM model, which is the basis of the CHU and is in turn based on realistic worst-case assumptions and has been validated in the field [1]. These models all use PNECs to assess effects, the Norwegian model additionally includes an implementation of SSDs.

For a selection of platforms, 4 to be exact, on the Dutch Continental Shelf (DCS) the risk of the 'natural' components in produced water is characterised, using 2 of the sophisticated dispersion models (DREAM/EIF and Delft3D) and the more simple CHU approach. Both dispersion models result in comparable dispersion patterns, although exact comparison is not possible due to the different processing and presentation of the data. The DREAM model expresses risk as the water volume in which the ms-PAF (multi-substance Potentially Affected Fraction) is equal to or greater then 5%.

It is shown, for production water discharges from Dutch platforms, that simple PEC:PNEC calculations (where the PEC is calculated using the CHARM dilution factor of 1,000) result in similar risk-based ranking when compared to the results of the more complex EIF calculations (which implements the spatial distribution of the produced water components.

Although the intended use of a risk-based approach within the OSPAR framework is not clear, it is most likely to be used to rank chemicals and mitigating measures and potentially assess the extend of the potential effects. For those purposes risk assessment can be used in a tiered approach. Where first tier assessment based on simple PEC:PNEC calculations for screening and ranking purposes. Spatial distribution becomes relevant in higher tier assessment, when the extend of potential effects needs to be evaluated (for instance to determine the effectiveness of mitigating measures). The DREAM / EIF model currently provides a convenient way of determining the extend of effects, as the EIF is defined as the water volume where the ms-PAF is greater than or equal to 5%. The current study also indicates that a simpler meta-model (a simpler model fitted to the results from the complex model) could be developed for this purpose. This, however, requires further research.

# Acronyms and abbreviations

| | |
|---|---|
| BMT | British Maritime Techonology Ltd |
| BTEX | Benzene, Toluene, Ethylbenzene and Xylenes |
| CDV | Critical Dilution Volume, the volume required to dilute a discharged concentration below its PNEC |
| CHARM | Chemical Hazard Assessment and Risk Management |
| CHU | Chemical Hazard Unit, the discharged amount in kg multiplied by HQ |
| CORMIX | CORnell MIXing Zone Model |
| DCS | Dutch Continental Shelf |
| Delft3D | 2D/3D modelling system to investigate hydrodynamics, sediment transport and morphology and water quality for fluvial, estuarine and coastal environments; developed by WLDelft Hydraulics, currently known as Deltares |
| Delft3D-Flow | Delft3D instrument to model hydrodynamics |
| Delft3D-Part | Delft3D instrument to model physical transport and chemical processes |
| DHI | Danish Hydraulic Institute |
| DREAM | Dose-related Risk and Effect Assessment Model |
| EC50 | 50% Effect Concentration |
| ECO lab | Ecological and substance fate model used in combination with MIKE instruments |
| EIF | Environmental Impact Factor, the water volume where the ms-PAF is greater then or equal to 5% |
| EQS | Environmental Quality Standard |
| ERA | Environmental Risk Assessment |
| EU | European Union |
| GIS | Geographical Information System |
| HQ | Hazard Quotient, used in CHARM as the ratio between the PEC at 500 m distance of a reference platform and the PNEC |
| ISO | International Standardisation Organisation |
| MIKE | Unknown acronym for the hydraulic modelling package of DHI |
| MIME | Managing Impacts on the Marine Environment programme |
| ms-PAF | multi-stress or multi-substance PAF |
| NOEC | No Observable Effect Concentration |
| NOGEPA | The Netherlands Oil and Gas Exploration and Production Association |
| OECD | Organisation for Economic Co-operation and Development |
| OIC | Offshore Industry Committee |
| OSIS | Oil Spill Information System, part of the VMIS framework |
| OSPAR | OSlo PARis convention |
| PAF | Potentially Affected Fraction of species, calculated from an SSD |
| PAH | Polycyclic Aromatic Hydrocarbons |
| PEC | Predicted Environmental Concentration |
| PNEC | Predicted No Effect Concentration |
| PROTEUS | Pollution Risk Offshore Technical EvalUation System |
| PROVANN | Unknown acronym for the original dispersion model used as a basis for the DREAM / EIF tool |
| SINTEF | Stiftelsen for Industriell og Teknisk Forskning ved Norges Tekniske Hoegskole / The Foundation for Scientific and Industrial Research at the Norwegian Institute of Technology |
| SSD | Species Sensitivity Distribution, a statistical distribution of toxicity data for a specific compound for different species |

| | |
|---|---|
| SSOR | Sub-Sea Oil Release module of the VMIS framework |
| TGD | Technical Guidance Document |
| TRENDS | The REsource Network facilitating QHSE (Quality, Health, Safety & Environment) Development for a Sustainable Energy Industry |
| US EPA | United States Environmental Protection Agency |
| VMIS | Visual Marine Information System, a modelling framework of BMT |

# Contents

# 1 Introduction

## 1.1 Background and aim

The OSPAR Offshore Industry Committee (OIC) has decided, in its meeting of 2008, to evaluate the possibility of implementing a risk based approach towards produced water management. This decision was made in light of the overall goal of the OSPAR Recommendation 2001/01 for the Management of Produced Water from Offshore Installations, which is to:

- "reduce the input of oil and other substances into the sea resulting from produced water from offshore installations, with the ultimate aim of eliminating pollution from those sources;
- ensure that an integrated approach is adopted, so that reduction in oil discharge is not achieved in a way that causes pollution in other areas and/or other environmental compartments;
- ensure that effort is made to give priority to actions related to the most harmful components of produced water."

Currently, Norway has made most progress in this field as it has fully implemented the Environmental Impact Factor (EIF) as the basis of their biannual reporting obligations and prioritising mitigating measures. The Netherlands has for as yet mainly followed a source (immission) based approach and not adopted a specific risk based approach.

In the present study several risk based approaches for produced water discharges are compared. The level of complexity required for risk based methods for ranking purposes is studied. The contribution of dispersion models, to determine the spatial distribution of produced water components, to ranking is investigated as they are usually a time-consuming step in risk calculations.

## 1.2 Risk assessment: evaluation of the likelihood and severity of effects

This section will go into a bit more detail on Environmental Risk Assessment (ERA) in general, before proceeding to the specific methods for produced water discharges. ERA has become a generally used tool in the evaluation of the potential environmental impact of chemical products or activities. According to the International Standardisation Organisation [2] the term risk can be interpreted as the combination of the probability of an event and the consequences of this event. In the case of environmental risk performed to assess the ecological risks of operational discharges the probability of occurrence of the event is often equal to 1 (discharges are taking place). Therefore, the definition of risk in environmental risk assessment mainly focuses on the consequences.

The definition of environmental risk assessment varies between countries. In 1983 the US National Research Council (USEPA) formulated a widely accepted definition of risk: "the characterisation of the potential adverse health effects of human exposure to environmental hazards". The UK Department of Environment defines risk assessment as "the structured gathering of the information available about risks and the forming of a judgement about them". Risk assessment has since developed into a series of guidelines for conducting ecological risk assessment [3].

Within the EU, risk assessment is defined as: 'A process of evaluation including the identification of the attendant uncertainties, of the likelihood and severity of an adverse effect(s) / event(s) occurring to man or the environment following exposure under defined conditions to a risk source(s)'. A risk assessment comprises hazard identification, hazard characterisation, exposure assessment and risk characterisation, and is an integrated part of the risk management procedure (Figure 1). These four steps of the risk assessment process were first elaborated by the US national research council [4] and are adopted by the EU [5].

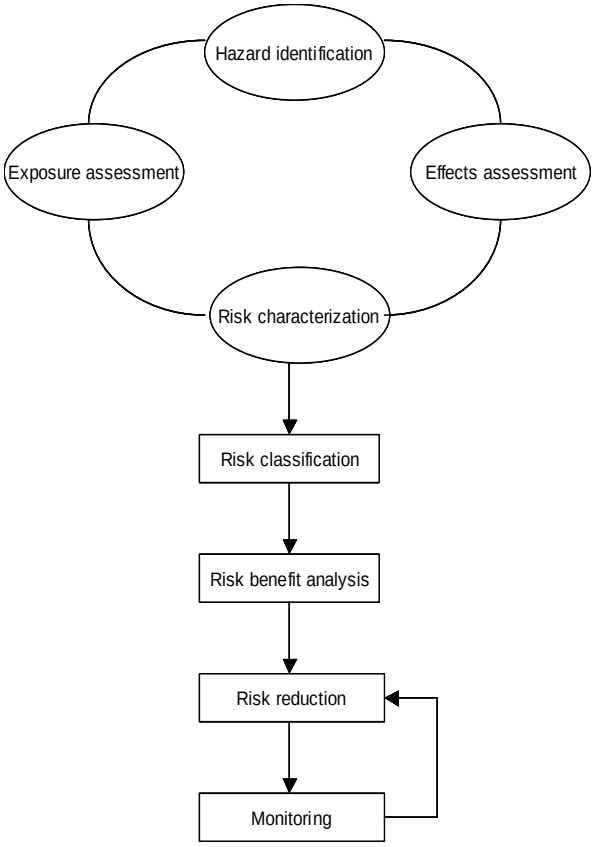


*Figure 1 Steps in the risk management process.*

The above clearly defines risk assessment as a process providing more than just the identification of environmental hazard. It also provides a characterisation of risk by quantifying the likelihood of occurrence, as well as the extent of the consequences.

The guidance on and requirements for environmental risk assessment for US Naval Operations has explicitly included the marine environment. The EU has published the updated Technical Guidance Document (EU-TGD) on risk assessment containing a technical guidance specified to marine risk assessments [5]. OSPAR[1] agreed to adopt the guidelines for risk assessment as described in the EU-TGD as the common EU/OSPAR approach on risk assessment methodology for the marine environment (OSPAR agreement 2003-20). OSPAR currently also adopting REACH (EU directive on Registration, Evaluation and Authorisation of CHemical substances) guidelines. The REACH guidelines on risk assessment [6] are mostly identical to the EU-TGD. There are different approaches

[1] The OSPAR Convention entered into force on 25 March 1998. It replaces the Oslo and Paris Conventions, but Decisions, Recommendations and all other agreements adopted under those Conventions will continue to be applicable, unaltered in their legal nature, unless they are terminated by new measures adopted under the 1992 OSPAR Convention.

for ERA, but in general a tiered approach is applied (Figure 2). A tiered framework for ERA starts out with a screening based on realistic worst-case assumptions. Higher tier assessment is only needed when the initial screening indicates potential effects and/or when a more specific or detailed answer is required. In that case more data and modelling effort is also required. The three generally applied tiers in ERA a described below.

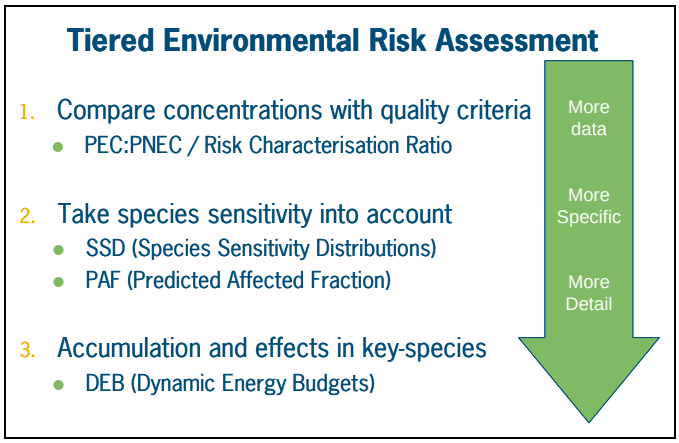


*Figure 2 Three general tiers in a tiered framework for Environmental Risk Assessment*

**Tier 1**:
The first tier screening is performed based on simple, but realistic worst-case assumptions. Most risk assessment tools use Predicted No Effect Concentrations (PNECs) to evaluate effects; where a PNEC is the concentration of a substance below which unacceptable effects on organisms will most likely not occur. Where the Predicted Environmental Concentration (PEC) is evaluated against the PNEC. The PEC:PNEC ratio does not quantify risk in a strict sense but is taken as indicative measure (for further details see Appendix A).

**Tier 2**:
Second tier assessment can be performed if first tier screening indicated potential adverse effects. Higher tier studies are more specific and therefore require more detail, data and modelling effort. Some applications implement Species Sensitivity Distributions (SSDs) in order to quantify effects as a second tier. SSDs are statistical distributions of No Observable Effect Concentrations (NOECs) and can be used to determine the Potentially Affected Fraction (PAF).

**Tier 3**:
Third tier assessment in general focuses on population dynamics of specific key-species. Body Burden approaches have also been developed as a third tier, where effects on species are based on internal concentrations using the DEBtox approach [7].

# 2 Risk assessment tools for produced water discharges

In Europe several methods are used for risk assessment of produced water discharges. In this report, the following methods are discussed:

- DREAM / EIF. Both the Dose-related Risk and Effect Assessment Model (DREAM) and the Environmental Impact Factor (EIF) are implemented in Norway by SINTEF and are available in a single package. Wageningen Imares provided the instruments for risk assessment as applied in those tools, where the first (DREAM) focuses on body burdens while the latter (EIF) works with PEC:PNEC ratios in combination with probabilistic risk assessment.
- PROTEUS is a modelling tool developed in the UK by BMT-Cordah which also expresses risk based on either body burdens or PEC:PNEC ratios.
- MIKE is a Danish product developed by the DHI (Danish Hydraulic Institute ) group, again expressing risk as PEC:PNEC ratios.
- Delft3D, a Dutch product developed by WLDelft Hydraulics (currently known as Deltares) is used for dispersion calculation. By combining it with PNEC values, it can be used as a risk assessment tool.
- CHU (Chemical Hazard Unit) is a methodology based on the CHARM model in which risk is expressed as PEC:PNEC ratios.

This chapter will provide more details on the methods listed above (see for more details TRENDS (The REsource Network facilitating QHSE Development for a Sustainable Energy Industry) study [8]). A description of each model is included, where the most important input and output is listed.

## 2.1 DREAM / EIF (Norway)

### 2.1.1 Introduction

The Norwegian Environmental Risk Assessment (ERA) models, implemented by SINTEF in collaboration with other institutions, are used to assess risks associated with operational and accidental discharges into marine and freshwater environments. In the current study we focus on the operational discharge of produced water. Industry and national environmental authorities have a need for predicting the environmental impacts and risks associated with such discharge scenarios. In particular, the models are used extensively for management decision-making to achieve the goal of “zero (effect) discharges”.

The central concepts within the SINTEF modelling suite are described below:

- Simulation of complex mixtures by inclusion of the individual chemical components in the discharge;
- Realistic representation of governing physical, chemical, and biological processes governing the fates and effects of each chemical component;
- Four-dimensional (space and time) simulations and analysis of physical fates, biological exposure, and risk within a geographical reference frame;
- Inclusion of global databases to facilitate applications world-wide;
- Built-in analysis tools and import-export utilities for ease in both analysis and reporting procedures;
- Unification of model support within a common framework, called the SINTEF Marine Environmental Modelling Workbench (MEMW)

The core models calculate the dynamic 3-dimensional fate of each of the compounds in the discharge, including advection, dissolution, dilution, evaporation, biodegradation, adsorption/desorption, settling, and mixing in sediments. The model includes multiple sources and multiple compounds in the discharge. The ocean currents can either be simulated by separate hydrodynamic models at SINTEF, imported from other hydrodynamic models run elsewhere, or approximated by the user through simple tools in the Graphical User Interface (GUI).

The models are based on Lagrangian particle concepts, operating within a gridded representation of the geographical base map (coastline, bathymetry) selected by the user. The user may develop environmental effects and risks based on:

- The ratio of a Predicted Environmental Concentration (PEC, computed by the model) and a Predicted No Effect Concentration (PNEC) derived from the best available quality controlled toxicological data for each compound in the release in combination with probabilistic risk assessment in which risk is expressed as the Potentially Affected Fraction of species (PAF), or
- Exposure, uptake and depuration calculations for exposed individuals in a population, to allow calculation of body burdens, hazards, and probabilities of lethal and sub-lethal effects based on the DEBTOX approach [9].

The model system also allows for stochastic simulations, in which results from many scenarios, each with a different start point in time, are combined to produce probability distributions for exposure and effects as the basis for risk. This capability is especially useful in design and planning phases. The dispersion model is based on the PROVANN model which has been validated with field data in the past [10].

The tool for chemical releases (DREAM) calculate risks based either on a PEC:PNEC approach consistent with the EU Technical Guidance Document, or based on exposure and body burden calculations for specific biota. Multiple discharges as well as multiple components in the discharges are included.

The model is fully three-dimensional and dynamic in time. The DREAM and EIF models are embedded in a graphical user interface called the MEMW. Model output can be exported to geographical information systems (GISs). A near field module is embedded in the system to simulate underwater blowouts as well as near field plumes of regular discharges.

### 2.1.2 Input

For the ambient: Winds, stratification, currents, shoreline, bottom topography (eventually GIS layers with vulnerable resources present).

For production releases: Location and depth of the discharges, amounts, densities, composition, and durations of the discharges. Properties of each of the compounds to be calculated: PNEC's, biodegradation and evaporation properties, particle sizes and densities. For exposure and body burdens: The distributions and patterns of motions of the biota to be considered.

### 2.1.3 Output

The outputs from the DREAM and EIF models are extensive. The GUI supports a variety of options for presentation of results: Animations of horizontal and vertical cross sections (all selected by the user) for the parameters of interest: Concentrations, currents, risk maps. Diagrams for exposure and/or body burden for the

biota included in the simulations. Diagrams and listing of results can be produced upon user's choice. Tables are produced for presenting the EIF (Environment Impact Factor). Time and space resolution are selected by the user as well.

## 2.2 PROTEUS (United Kingdom)

### 2.2.1 Introduction

PROTEUS specialises in modelling the fates of offshore discharges. The numerical fate modelling involves the computation of the physical dispersion, and biogeochemical transfer of chemical compounds in the marine environment. Turbulent advection and diffusion of chemical plumes both in solid and dissolved phases is computed by PROTEUS. The transfer of contaminant between solid and water phases is also modelled by PROTEUS through adsorption/de-sorption kinetic principles. The predicted exposures of chemicals, at various environmental departments, are dynamically computed in time and space. The environmental risk is also computed as the ratio of the Predicted Environmental Concentrations (PEC) to that of the Predicted No Effect Concentration (PNEC). Highly interactive chemical, toxicity and biological species databases are used by the various modelling components of the PROTEUS system.

PROTEUS is fully three-dimensional and include spatio-temporal variations. It is hosted by the BMT Visual Marine Information System (VMIS) framework and operate within PC Microsoft environments. The PROTEUS system has been validated against laboratory and field data during the three years international research project MIME (1997-2000). Details on the validation study are not available.

### 2.2.2 Input

The VMIS framework automatically provides common databases and servers to support all of BMT ERA tools, including PROTEUS, SSOR and OSIS. These include the following;

- GIS database
- Hydrodynamic server
- Bathymetry server
- Physico-chemical property of oil database
- Physico-chemical property of mud and cuttings
- Chemical database
- Toxicity database
- Biological species database

For the modelling of produced water discharges, the model needs the following input:

- Density of effluent
- Discharge Rate, depth, location, time and duration of discharge, pipe direction and diameter
- Particulate matter Concentration in effluent (if any)
- Chemical Compounds in pipe

### 2.2.3 Output

The output from the model is comprehensive and can be processed by the user form more data analysis using the graph and reporting services within BMT VMIS. The GUI supports a variety of options for presentation of results. These include: Animations of 3D and 2D plumes for any specific discharge scenario. Exposure concentrations, currents, PEC:PNEC, dynamic body burden risk maps. Reports of results can be exported to word document upon user's choice.

## 2.3 MIKE (Denmark)

### 2.3.1 Introduction

The modelling tools at DHI all takes basis in a hydrodynamic and an advection-dispersion model for the area of interest enabling the prediction of the dispersion of an emission in the water body of interest. Thus, the concepts of their modelling tools on ERA is to predict the concentration distribution (both spatial and temporal) in the water body. The risk assessment is carried out by comparing the predicted concentrations with PNEC-values of the substances in question.

The environmental risk assessment in this model is, again, based on the PEC:PNEC concept. The PNECs applied in the model are usually derived on the basis of the principles in the TGD [5]. Evaluation of chronic effects is usually based on a comparison of the time-averaged concentration with the PNEC. Acute effects are usually based on a comparison of the maximum predicted concentration with the PNEC for acute effects, which usually are set to 10-100 times the PNEC for chronic effects. Risk to sea-birds, marine mammals may also be assessed.

The modelling tools at DHI are all based on the DHI hydraulic models, of which MIKE 3 and MIKE SHE are relevant tools for offshore produced water discharges:

- MIKE 3: is a 3-dimensional modelling system for rivers, lakes, estuaries, coast and oceans and other water bodies. MIKE 3 simulates unsteady flow taking into account density variations, bathymetry and external forcing such as meteorology, tidal elevations, currents and other hydrographic conditions;
- MIKE SHE is a 3-dimensional modelling system simulating the entire land phase of the hydrologic cycle.

The environmental modules of the MIKE-models include an advection-dispersion module simulating the spreading of a dissolved or suspended substance in an aquatic environment under the influence of the fluid transport (from the hydrodynamic module) and associated dispersion processes. The substance may be of any kind, conservative or non-conservative, inorganic or organic. The advection-dispersion module solves the advection-dispersion equation for dissolved or suspended substances, which is the mass-conservation equation. The hydrodynamics of the MIKE 3 model has been validated for the Baltic Sea [11].

On the top of the hydrodynamic and advection-dispersion models different approaches on ERA can be used:

- For non-degrading substances and/or substances degrading according to a first order kinetics, a common used method is to simulate the dispersion of one unit source using one ore more hydrodynamic model. One result of the simulations is the predicted concentration distribution in the water body as a function of time. These results can be treated in a post-treatment system enabling many types of

calculations, eg. conversion according to the actual emission rate, calculation of time average concentrations, maximum concentration and risk quotients;

- For other substances with a more complex degradation pathway and/or for a more complex ERA study, ECO lab, a numerical laboratory for ecological and substance fate modelling, is used. A fate model is programmed in a ECO lab template. ECO Lab has an interface to all DHI hydraulic models. Thus, a template can be used for any DHI hydraulic model and any water body of interest. With ECO Lab it is possible for the user to change the equations of the model and to include data generated for instance by other computational modules of the DHI software family. The ECO Lab Templates, are pre-programmed sets of equations developed by DHI comprising Water Quality (BOD-DO relationship, nutrient transport and bacterial fate), eutrophication (nutrient cycling and relationship with primary production) and heavy metal (fate of dissolved and suspended metal in water and sediment).

### 2.3.2 Input

For the ambient: winds, bathymetri, bed resistance coefficients, (temperature), hydrographic boundary conditions
For the releases: release rate (both volume rate and linear rate), release duration, position and depth, density
For the pollutants: biodegradation rate, abiotic transformation rate parameters (photolysis: quantum yield, neutral, acidic and alkaline rate constants), $PNEC_{chronic}$, $PNEC_{acute}$ (often set to $10 \cdot PNEC_{chronic}$)

### 2.3.3 Output

The model generates extensive outputs, so it is mostly a matter of defining which parameters being of interest for the user. Thus, a variety of options for the presentation of the results are possible, e.g. currents (velocity and direction), animation of almost any kind, concentrations, maps of concentration, risk quotients etc.

## 2.4 Delft3D (The Netherlands)

### 2.4.1 Introduction

The Deltares (formerly known as WL Delft Hydraulics) chemical fate model can be used as a risk assessment tool, when combined with effect assessment based on PNEC values. The dispersion modelling requires three separate steps. In the first step near field plume behaviour is investigated with CORMIX [12], a model that has been developed by U.S. EPA and is used worldwide. In the second, the Delft3D-Flow is used to model hydrodynamics. It will not be subject to a calibration procedure, although a comparison with the available observations will be carried out. The model was validated during a study in 2001 [13]. Finally, the transport of a tracer is modelled with Delft3D-Part. It simulates transport processes and simple chemical reactions by means of a particle tracking method. The tracks are simulated in three dimensions over time, whereby a dynamic concentration distribution is obtained through averaging of separate particle tracks.

Although the model currently only implements dispersion, it can easily be adapted to evaluate the PECs against the PNECs as has been done in the current study. Although in the current study potential effects are assessed using PEC:PNEC ratios, it can also be assessed with probabilistic risk assessment, using Species Sensitivity Distributions (SSDs). In the future it is intended to develop a module to assess the effect of substances that accumulate along a key food chain.

### 2.4.2 Input

The Flow module of Delft3D is a multi-dimensional (2D or 3D) hydrodynamic simulation program which calculates non-steady flow and transport phenomena resulting from tidal and meteorological forcing on a curvilinear, boundary fitted grid. In 3D simulations, the vertical grid is defined following the so-called sigma coordinate approach. This results in a high computing efficiency because of the constant number of vertical layers over the whole of the computational field. The Delft3D-Part uses the discharge characteristics as input next to the hydrodynamics calculated by the Flow module. For risk calculations, PNECs are required as input, or when probabilistic risk assessment is applied, parameters describing Species Sensitivity Distributions (SSDs) need to be set.

### 2.4.3 Output

The model output is generated in a GIS environment and can be displayed as spatial distribution of concentrations as a function of time. The output data can easily be post-processed in order to present them differently. For instance, the statistical distribution of concentrations or maximum concentrations in time can be determined. Data can also be presented as PEC:PNEC ratios or as multi-substance PAFs (Potentially Affected Fractions) using probabilistic risk assessment.

## 2.5 CHU (NOGEPA, The Netherlands)

### 2.5.1 Introduction

The Chemical Hazard Unit (CHU) is used in the Netherlands for prioritising the replacement of harmful production chemicals. The CHU was introduced as a measure of environmental impact caused by the discharge of chemicals. It allows for an indicative assessment of small amounts of highly hazardous substances or, large amounts of less hazardous substances. Furthermore, the sum of the CHUs (ΣCHU) provides an impression of the total environmental impact. The CHU of offshore chemicals is calculated by multiplying the PEC:PNEC ratio determined with CHARM (Chemical Hazard Assessment and Risk Management) and the discharged load of the chemical (kg).

The CHARM model calculates the Hazard Quotient (HQ) (the PEC:PNEC ratio for a reference platform) at a distance of 500 meters from the platform). For this purpose a dilution of the substance in the discharged water, of a factor 1,000 is assumed to be worst-case and has been validated in the field [1]. In fact, CHARM HQ calculations are based on characteristics of a reference platform and realistic worst case assumptions (Table 1). For production chemicals the concentration in the discharged water is calculated from the administered dose.

*Table 1 Characteristic conditions of the reference platforms (realistic worst case) used in Hazard Assessment, from the CHARM user manual [14].*

| **Parameter** | **Symbol used in CHARM** | **North Sea oil production platform** | **North Sea gas production platform** | **Units** |
|---|---|---|---|---|
| Platform density | | 0.1 | 0.1 | $km^{-2}$ |
| Water depth | | 150 | 40 | m |
| Refreshment rate | R | 0.24 | 0.24 | $d^{-1}$ |
| Corresponding Residual Current Speed | U | 0.01 | 0.01 | $m\ s^{-1}$ |
| Sediment organic carbon content | $f_{oc}$ | 0.04 | 0.04 | - |
| Dilution at 500 m | D | 0.001 | 0.001 | - |

### 2.5.2 Input

In principle the method is developed for production chemicals for which the concentration in the discharged produced water needs to be estimated. The concentration is calculated with CHARM for a reference platform, where only the administered dose is the only input variable. The method can also be applied to other contaminants, as long as the concentration in the discharged produced water is known.

In addition the amount (in kg) of the substance being discharged is input for the calculation. This mass can be determined from the concentration in the discharged water and the volume being discharged.

### 2.5.3 Output

The method provides a CHU for each substance and can be summed for all substances into a single ΣCHU. Where CHU is the PEC:PNEC ratio at a distance of 500 m from the platform multiplied by the amount of substance being discharged in kg.

## 2.6 Summary on available tools

Most ERA models for produced water discharges (DREAM / EIF, PROTEUS, MIKE and Delft3D model) use sophisticated dispersion modelling in order to predict the spatial distribution of concentrations in time. They all express risk based on PEC:PNEC ratios in some form. The EIF also implement a probabilistic risk assessment approach, using SSDs. The CHU differs in that it uses a simple method (a dilution factor based on realistic worst-case assumptions) to predict the environmental concentration. Furthermore, it expresses risk as a PEC:PNEC ratio multiplied by the discharged amount in kg.

# 3 Illustration of the methods

## 3.1 Introduction

Some of the methods described above have been applied to specific platforms on the Dutch Continental Shelf (DCS), in order to illustrate these methods. As indicated, most risk assessment models described above use sophisticated dispersion modelling to describe the produced water discharge. Two of those approaches have been selected to represent this group: the EIF approach and Delft3D.

As mentioned earlier, the CHU expresses risk differently when compared to the other methods. The CHU uses a simple method (dilution factor) to predict the environmental concentration. In the CHU, the PEC:PNEC ratio is multiplied with the discharged amount. As this is a unique way of quantifying risk, it too is calculated for a selection of platforms on the DCS.

Results from the calculations will be compared at the end of this chapter. First, the selection of platforms is presented in section 3.2. As all different models use a PEC:PNEC ratio in some way to express risk, a single set of PNECs is used in the calculations, in order to make the results comparable. The PNECs used in the calculations are listed in section 3.3.

## 3.2 Selection of platforms

To illustrate the risk assessment methods, a number of representative oil and gas platforms on the DCS are selected. The selection is primarily based on the concentrations of 'natural' contaminants in produced water. These concentrations, a result of a separate monitoring study in the years 2005 and 2006, were provided by NOGEPA. Gas platforms were chosen such that a platform with generally relatively high and relatively low concentrations of 'natural' contaminants is represented. In addition a gas platform with relatively high metal concentrations is also included. Furthermore an oil platform with relatively high concentrations of 'natural' contaminants is also added to the selection. As it is not the aim to assess the actual platforms, the platforms have been made anonymous in this study. The selected gas platforms are number one to three, the oil platform is labelled with the number four.

## 3.3 PNEC values

All the ERA methods use PEC:PNEC ratios in some way to express risk. For convenience a single set of PNECs is used in the illustrative calculations with the different methods, as this promotes the comparability of the results. For this purpose, the PNECs that were derived for the Norwegian EIF are used [15-17]. These values have been determined according to the principles of the TGD [5] and are listed in Table 2. The grouping of substances as shown in this table will also be implemented in the other risk methods. It should be noted that most of these PNECs are based on the old EU-TGD guidelines, which means they are not specific for marine waters. When actual risk needs to be assessed (which is not the goal of the present study), the values should be updated with PNECs based on the more recent marine EU-TGD, which uses more strict assessment factors in marine waters.

*Table 2 PNEC values as derived for the Norwegian EIF [15-17] supplemented with Dutch EQS values*

| Substance (group) | PNEC (µg/L) |
|---|---|
| BTEX[2] | 17 |
| Naphthalenes | 2.1 |
| PAH[3] 2-3 ring | 0.15 |
| PAH[3] 4 ring+ | 0.05 |
| Alkylphenol C0-C3 | 2 |
| Alkylphenol C4-C5 | 0.36 |
| Alkylphenol C6+ | 0.04 |
| Arsenic[4] | 32 |
| Cadmium | 0.028 |
| Chrome[4] | 84 |
| Copper | 0.002 |
| Lead | 0.182 |
| Mercury | 0.008 |
| Nickel | 1.22 |
| Zinc | 0.46 |
| Aliphates | 40.4 |

## 3.4 DREAM / EIF model calculations

EIF calculations are performed with the Marine Environmental Modelling Workbench V4.0.1, where discharge characteristics (listed in Appendix C) are used as input for dispersion computation. The model has been developed for the Norwegian situation, where discharge volumes are generally larger than the Dutch situation. In order to model dispersion for the Dutch situation, the resolution in time and space needs to be increased. For platform 2, the discharged volume and concentrations even such that the resolution needed to be set beyond the model specifications. A period of 30 days was simulated.

In order to calculate the Environmental Impact Factor (EIF), the risk for each component is expressed as the Potentially Affected Fraction (PAF) of species (with values between 0 and 1). The PAF is calculated for each chemical component as a function of the concentration, the so-called Species Sensitivity Distribution (SSD). The SSD is a statistical distribution of the sensitivity of species in laboratory tests (see Appendix A for more details). The total risk is calculated by assuming that the chemical components act independently on the organisms and is expressed as the multi-substance PAF (ms-PAF). The model generates maps of this ms-PAF; Figure 3 shows the maximum risk (ms-PAF) for the four selected platforms. The EIF is defined as the number of grid cells (sized 100 x 100 x 10 $m^3$) where the ms-PAF is equal to or greater than 5%. These EIF values are shown in Figure 4.

[2] Benzene, Toluene, Ethylbenzene, and Xylenes
[3] Polycyclic Aromatic Hydrocarbons
[4] This is not a standard EIF chemical and hence there is no EIF PNEC available. Instead, the Dutch Environmental Quality Standard (EQS) (Total concentration in surface water) is used.

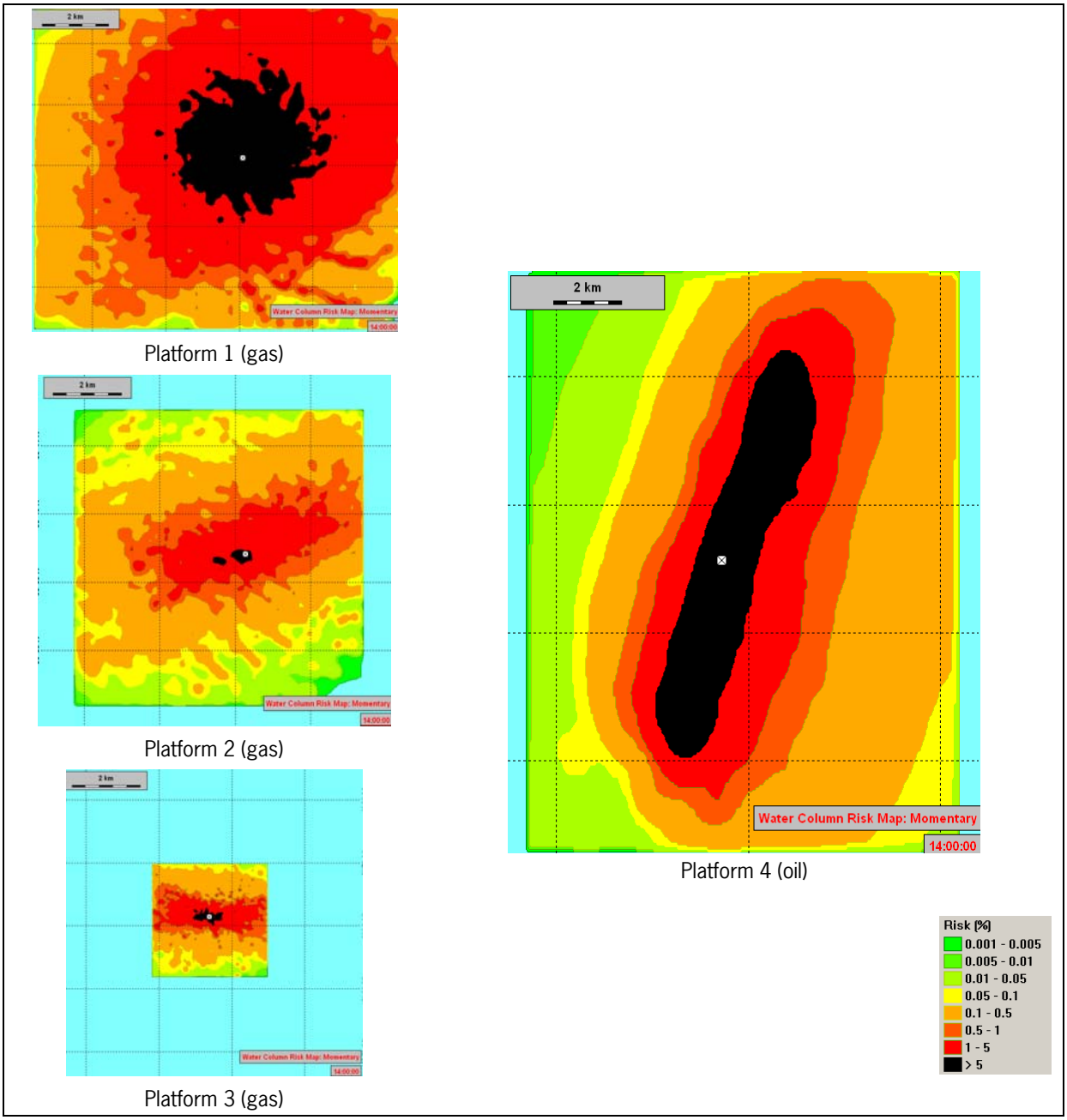


*Figure 3* *EIF risk maps for the selected platforms. Black areas indicate where Multi-substance PAFs (Potentially Affected Fraction of species) are equal to or greater than the generally accepted risk level of 5%. Note that the depicted areas are based on maximum risk values in time, in momentary risk maps the black areas are smaller as discharge plumes rotate around the platform due to tidal currents*

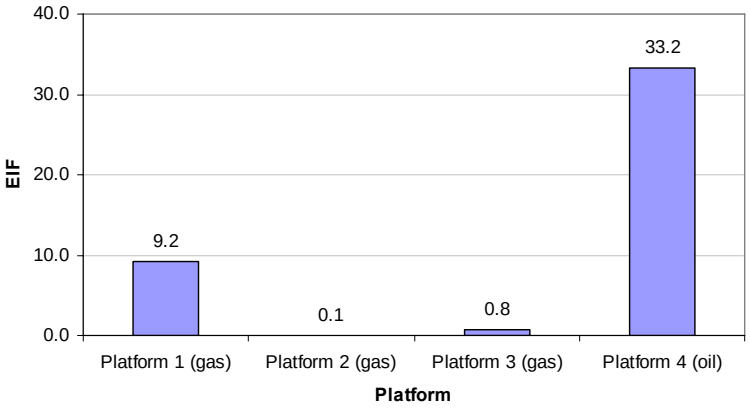


*Figure 4* *EIF (maximum water volume [in 100x100x10 m³ grid cells) where the risk level is equal to or greater than 5%) calculated for the selected platforms, where only 'natural' contaminants are included.*

Based on EIF calculations, the highest risk is expected at platform 4 (Figure 4). Risk is in this case expressed as the EIF (the water volume in which the ms-PAF is equal to or greater then 5%). Platforms 2 and 3 have relatively low EIF values (Figure 4). The model also computes the contribution of the components to the total EIF, the model weighs those contributions with a simple factor to correct for persistent, bioaccumulating components (Figure 5). Based on these calculations, C6+ alkylphenols are ranked relatively high for platform 3 and C0-C3 alkylphenols for platform 2, while for the platforms 1 and 4 respectively lead and copper score relatively high (Figure 5).

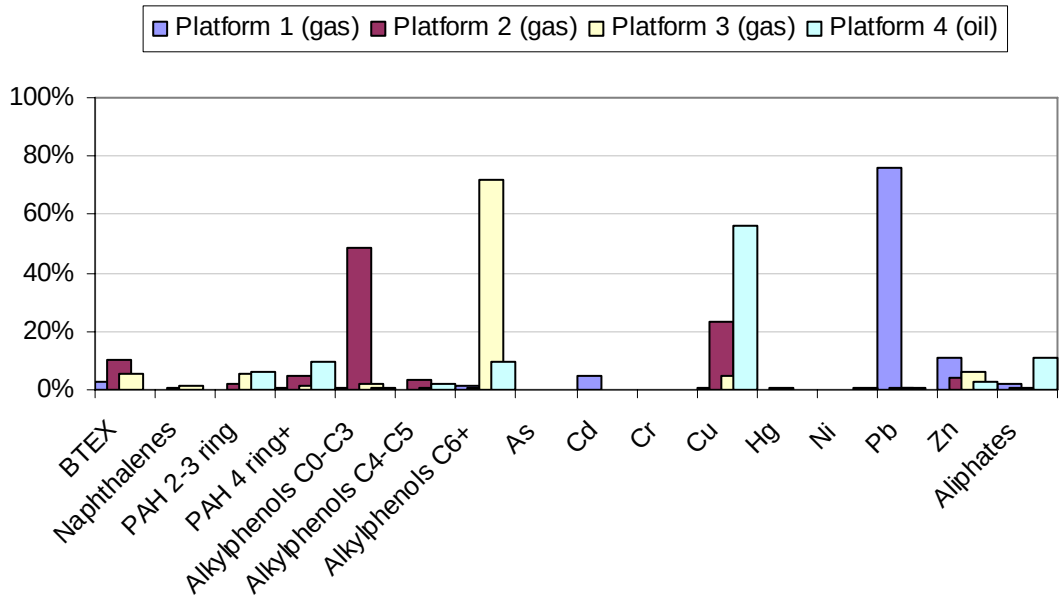


*Figure 5* *Relative contributions of the components to the total EIF; contributions have been weighed with a simple factor to correct for persistent, bioaccumulating components*

## 3.5 Delft3D calculations

The Deft3D model used discharge characteristics (listed in Appendix C) as input. The tool is used to model a single tracer compound, where currently processes such as biodegradation are not considered. The model simulated a period of 14 days which approximates an entire spring neap cycle. The concentration of the tracer in the produced water is set to 1. In order to get actual concentrations from the modelling results the tracer concentration needs to be multiplied by the discharged concentration of the concerning substance. Table 3 shows for each platform the substance with the highest ratio between the discharged concentration and the PNEC. It also shows what dilution factor is required to dilute the substance below its PNEC.

*Table 3 Tracer concentrations equal to the PNEC for substances with the highest PEC:PNEC ratio for each platform*

| **Platform** | **Substance** | **$C_d$ (Discharged concentration) [µg/L]** | **PNEC [µg/L]** | **Tracer concentration equal to PNEC (PNEC/$C_d$)** |
|---|---|---|---|---|
| Platform 1 (gas) | Lead | 13220 | 0.182 | 0.000014 |
| Platform 2 (gas) | C0-C3 alkylphenols | 33142 | 2 | 0.000060 |
| Platform 3 (gas) | C6+ alkylphenols | 659 | 0.04 | 0.000061 |
| Platform 4 (oil) | Copper | 47.7 | 0.02 | 0.00042 |

Calculations were performed with grid cells with a resolution of 500 x 500 $m^2$ (with the exception of platform 4 which was calculated with a resolution of 50x50 $m^2$). Platforms 2 and 3 show that the PNEC is not exceeded for any of the produced water components (for this purpose, compare the tracer concentration equal to the PNEC of the highest ranked substance from Table 3, 0.000060 and 0.000061 for platforms 2 and 3 respectively, with the values depicted in Figure 6). This indicates that no effects are expected to occur at a distance of 500 m and further from the platforms. The EIF calculations show a slightly larger affected area for those two platforms, which can be explained by the cumulative method applied in the EIF: Potentially Affected Fractions (PAFs) of each produced water component is summed into a multi-substance PAF (ms-PAF) in that case. In general the dispersion patterns (shape and size) calculated by both models are comparable (Figure 3 and Figure 6).

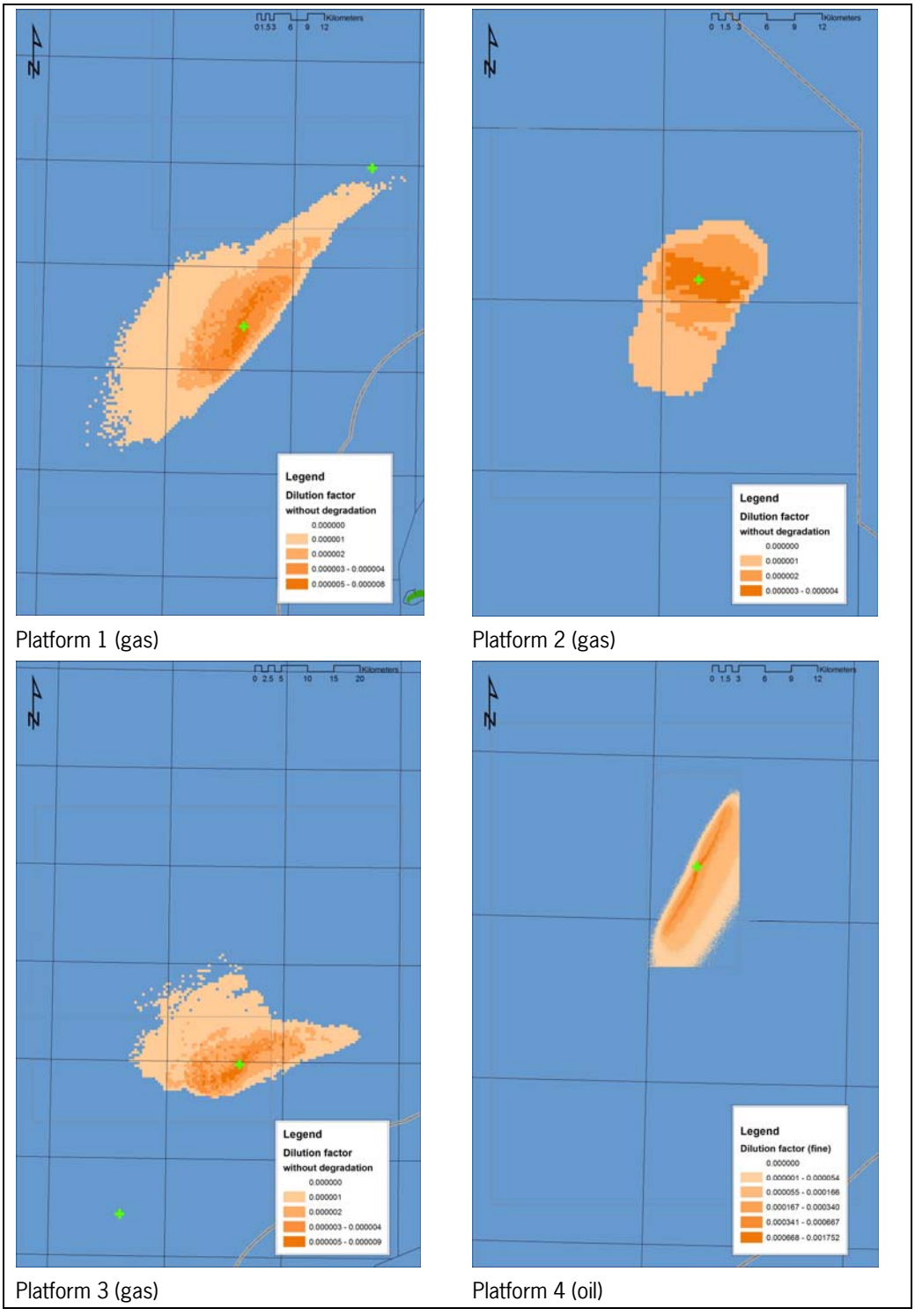


*Figure 6* *Dispersion calculated for a tracer substance with the Delft3D model; where the discharged concentration of the tracer is set to 1, to obtain the concentration of any other substance, multiply dilution factor in picture with the actual discharge concentrations (Appendix B); the extend of potential effects can be determined with the tracer concentrations equal to the worst case PNEC listed in Table 3; note that the legend is different for each platform*

## 3.6 CHU Calculations

Although the CHU is developed for production chemicals, it can also be applied to other produced water components. For this purpose, measured concentrations in produced water (Appendix B) were used as input for the calculations. First, the PEC at 500 m distance is estimated by dividing the measured concentration by the CHARM factor of 1,000. The HQs are now calculated by dividing the PECs by the PNECs as listed in Table 2. Finally the HQs are multiplied by the discharged amount in kg (calculated from the discharged volume and discharged concentration).

Figure 7 shows the resulting CHU values per component for the four selected platforms. To get an indication of the total risk, the CHU of the components can be summed into the ΣCHU as presented in Figure 8.

The ΣCHU needs to be interpreted with caution as the calculations are based on a reference platform, with realistic worst-case bathymetric conditions. The bathymetric conditions of the actual platforms were not used in these calculations.

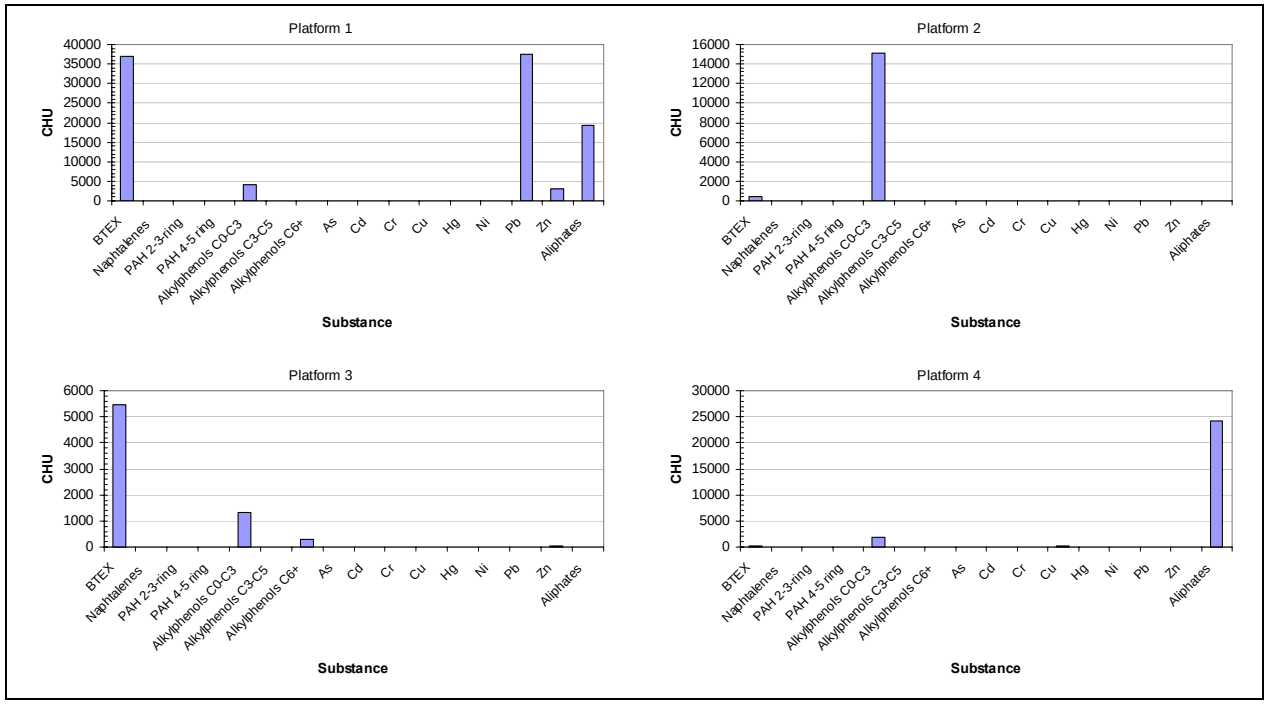


*Figure 7 CHU values for each platform per produced water component*

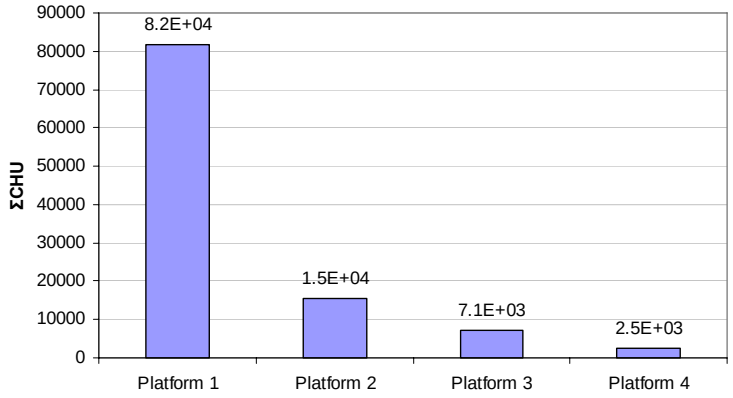


*Figure 8 Sum of Chemical Hazard Units of all 'natural' produced water components per platform*

## 3.7 Comparing the different methods

### 3.7.1 Discharged amount versus environmental risk

Evidently there is a large difference between the EIF and the CHU (see Figure 4 and Figure 8). Based on the EIF calculation the highest risk is expected for platform 4, while the CHU expects it to be lowest for that platform. This large difference can be explained by the way these variables express risk. Although both methods (EIF and CHU) use PEC:PNEC ratios to quantify risk, the CHU multiplies the ratio with the discharged amount. Resulting rankings will therefore be focused on the load, which can be suitable for the Dutch immision based policy.

The focus of the CHU on discharged amounts is also illustrated by the formulas in Table 4: the load is squared while the other terms are not. When the CHU is compared with the discharged amount (left part of Figure 9), there appears to be a strong correlation between the two as can be expected from the formulas listed in Table 4. Therefore, prioritising substances based on either CHU values or discharged amounts is expected to result in very similar rankings (see for example Figure 9). The HQ and the Critical Dilution Volume (CDV) (the volume of water required to dilute the discharged water to a concentration below the PNEC) both incorporate the discharged amount (see Table 4) and provide a more intuitive alternative to characterise risk. For each platform, both the HQ and CDV will lead to the same ranking, as the difference between the two is the discharged volume, which is identical for all components on the same platform (see Table 4 and Figure 9).

*Table 4 Formulas for different risk assessment approaches*

| Formula | Remarks |
|---|---|
| Ld = load (kg/year) | Discharged amount |
| $V_d$ = Volume discharged produced water ($10^9$ L/year) | |
| HQ = $PEC_{500m}$/PNEC<br>= [Ld / (1,000 x $V_d$)] / PNEC | Hazard Quotient |
| CHU = Ld x HQ<br>= $Ld^2$ / (1,000 x $V_d$ x PNEC) | Chemical Hazard Unit |
| CDV = Ld / PNEC | Critical Dilution Volume |

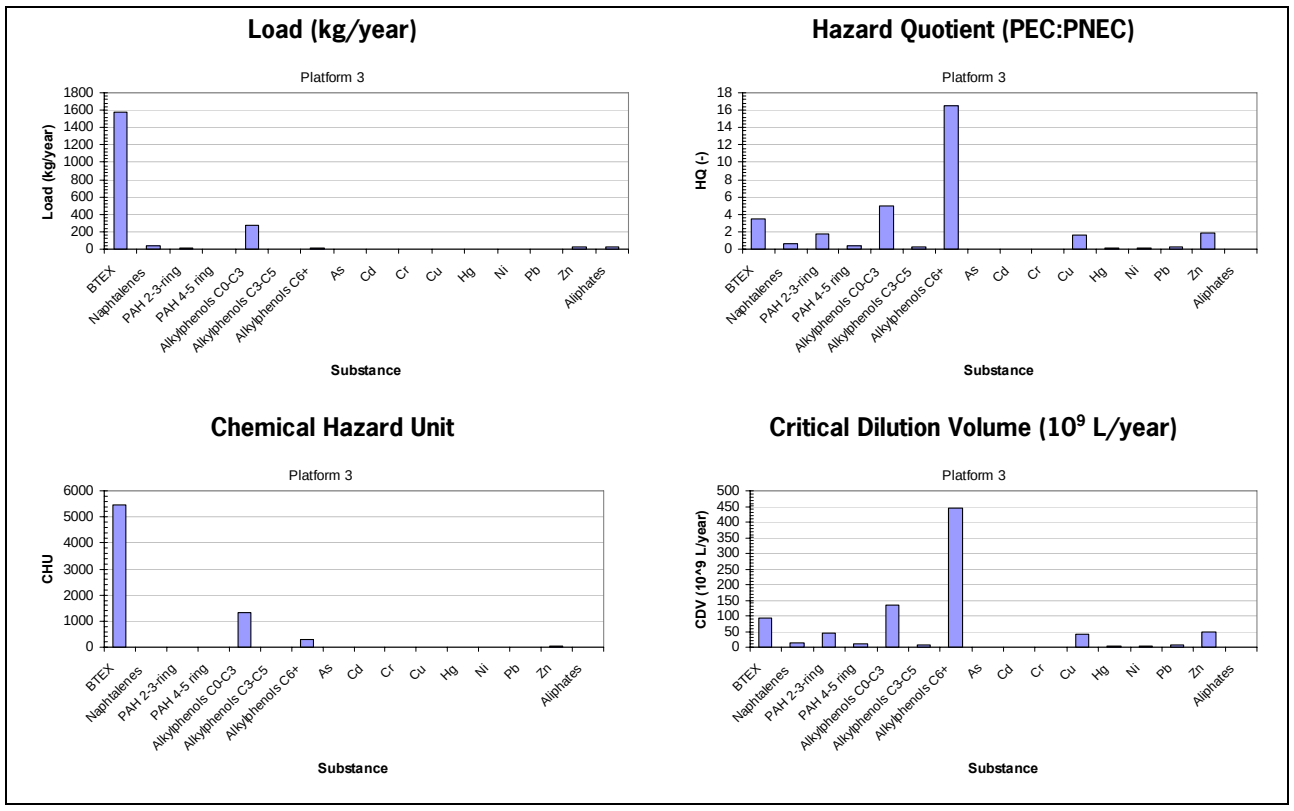


*Figure 9 Relations between different approaches in risk assessment, illustrated for the situation at platform 3 as an example*

### 3.7.2 Ranking and spatial distribution

The HQ provides a simple way of expressing risk, whereas the Environmental Impact Factor (EIF) also includes the spatial distribution. As indicated in the previous section, the CHU generates fundamentally different results when compared with the HQ and is therefore not included in this section. Substances are ranked both using the HQ and the EIF. Substances with the highest contribution are ranked 1, the substance with the lowest contribution is ranked 16 (as there are 16 produced water components considered in the current study). Results of this ranking

exercise are presented in Figure 10.

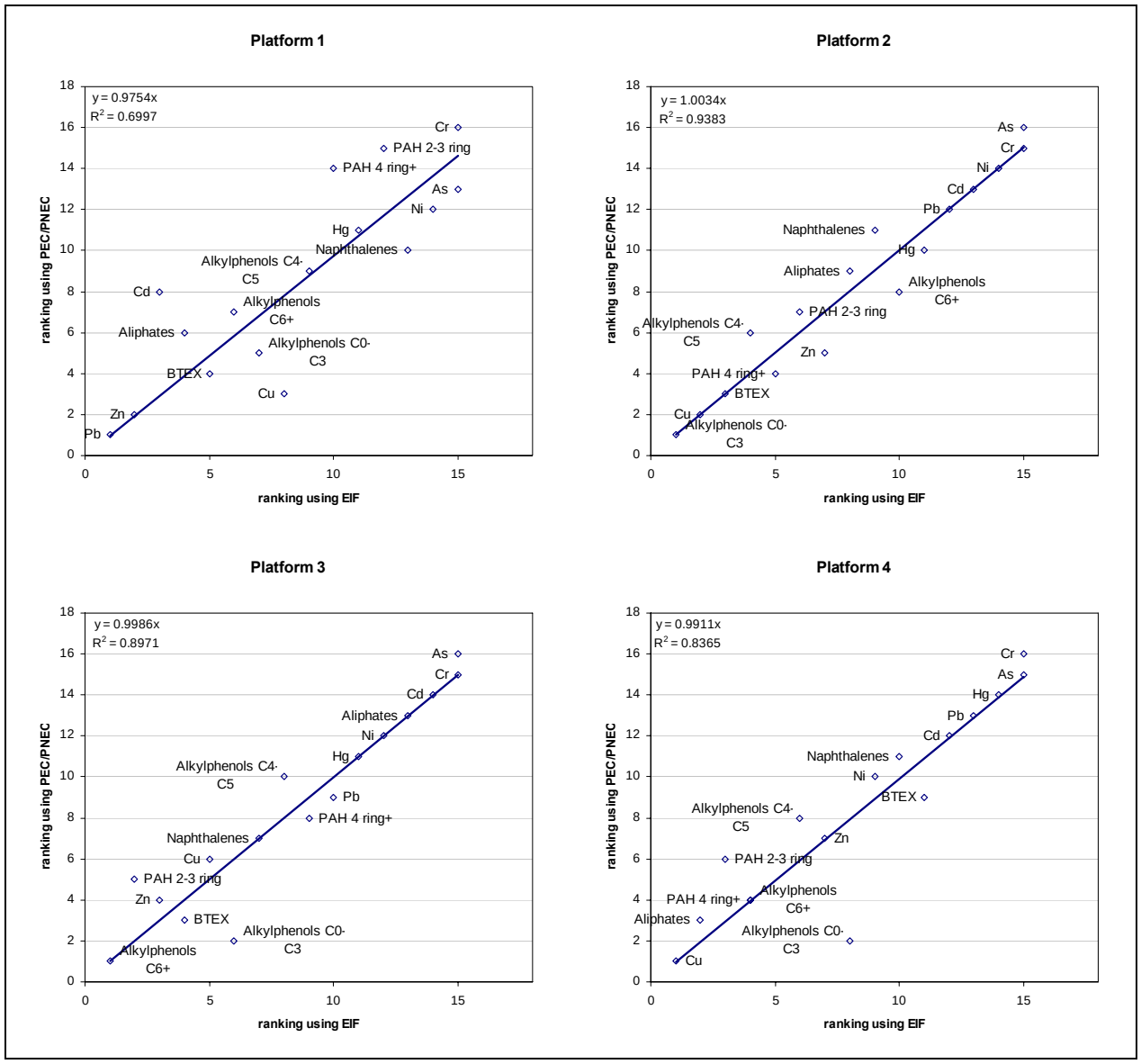


*Figure 10 Ranking using different risk-based approaches. The 16 included produced water components are ranked with the numbers 1 up to 16, where a substance ranked 1 has the highest contribution to the overall risk. The horizontal axis shows the ranking of the substances. Straight lines are fitted with linear regression with the intercept fixed to 0.*

Apparently, as indicated by Figure 10, both rankings with and without a spatial component (EIF and HQ respectively) are highly correlated. For each platform, the substance with the highest HQ is also the substance with the highest contribution to the EIF. Mostly, organic substances are ranked differently by the two methods, which is not surprising as biodegradation (metals don't biodegrade) is implemented in the EIF but nut the current HQ calculations. Other aspects such as bioavailability (the fraction of a substance available to an organism to exert an effect) are expected to be at least equally as important as spatial distribution. For copper for instance, Stauber et al. [18] estimated the bioavailability of copper between 50% and 80% in coastal marine waters. Furthermore, the uncertainty in PNECs, expressed as high assessment factors (ranging from 10 up to 10,000 to

translate effects in laboratory test to effects in the marine field), strongly affects the risk-based ranking of substances. Therefore, in a tiered approach risk assessment can be refined by reducing assessment factors (by specifically expanding the ecotoxicity dataset), or determining the bioavailability of the substances. Spatial distribution of the substances can also used to refine risk assessment, however, this will be most useful to assess the extend of effects and not so much for ranking purposes.

### 3.7.3 Impact assessment

As indicated in the previous section, spatial distribution is relevant for impact (extend of effects) assessment. The EIF defines the impact as the water volume where the ms-PAF equals or exceeds 5%. Such an indication can be used to determine the effectiveness of taken mitigating measures: i.e., 'is the affected area with ms-PAF > 5% reduced?' Figure 11 shows that there is a correlation between the sum of Critical Dilution Volumes (CDVs) and the total EIF. This correlation is only based on a small dataset ($R^2$ = 0.97, n = 4 platforms). An additional study on more platforms and scenarios would be interesting, to see whether this correlation would still hold. In that case a simpler meta-model could be developed to calculate the extend of effects.

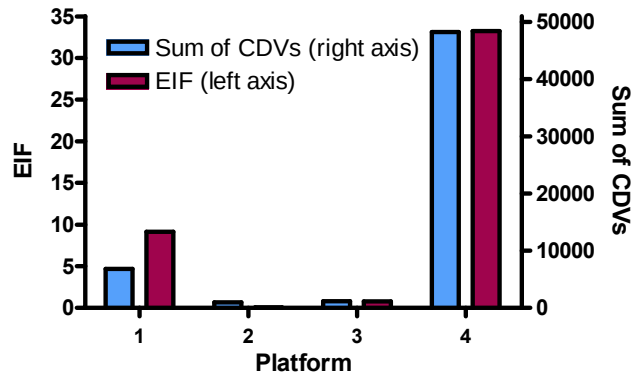


*Figure 11 Environmental risk expressed as the sum of Critical Dilution Volumes (calculated without dispersion modelling) and risk expressed as the Environmental Impact Factor (calculated with dispersion modelling)*

Obviously, when the specific location of potential effects is relevant (for instance in or near Marine Protected Areas), the spatial distribution of produced water components is too. In that case, a simple meta-model is not sufficient, as not only the extend of potential effects (size of the affected area) needs to be known, but also the location where those effects are anticipated. Also, when discharge plumes from different platforms overlap, spatial effects become relevant. However, in the Dutch situation with relatively small discharges, it is expected not to be the case.

# 4 Conclusions and Recommendations

From this study, based on 4 platforms on the DCS, it can be concluded that:

- A first tier assessment (based on relatively simple PEC:PNEC calculations) is initially sufficient for ranking substances (and mitigating measures consequently).
- To assess impact, the extend of potential effects, spatial distribution of produced water components is required, for which a dispersion model can be used.
    - Most dispersion models share a similar mathematical basis, and are expected to generate comparable results (although this has never been studied). The DREAM / EIF model currently provides a convenient way of determining the extend of potential effects, as the EIF is defined as the water volume in which the ms-PAF is greater than or equal to 5%.
- A tiered approach is a risk-based approach that is flexible in the level of detail that is required. It is suitable for both screening / ranking substances (first tier) and assessing the extend of potential effects (second tier).

Based on this study and the conclusions above, the following is recommended that:

- OSPAR specifies the intended use for the risk-based approach: is the approach needed for screening of potential effects, ranking substances / mitigating measures and / or assessing impacts (extend of potential effects)?
- Dispersion calculations are only performed when the risk-based approach requires this: determining the extend of potential effects requires time consuming distribution calculations with models that are not publically available. The current study indicates that a simpler meta-model (a simpler model fitted to the results from the complex model) could be developed. This, however, requires further study.
- With the development of a risk-based approach, sufficient attention should go to the assessment of persistent and bioaccumulating substances. In the currently applied first tier assessment (HQ calculations) these aspects are not explicitly accounted for. In the DREAM / EIF model a simple weighing factor is applied as a pragmatic solution, which needs further scientific validation.

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# 6 Justification

Rapport C012/09
Project Number: 439.51022.01

The scientific quality of this report has been peer reviewed by the a colleague scientist and the head of the department of Wageningen IMARES.

Approved: Dr. T.C. Klok
Senior scientist

Signature: 

Date: February 2009

Approved: Drs. J.H.M. Schobben
Head of department Environment

Signature: 

Date: February 2009

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| Number of copies: | 12 |
| Number of pages | 47 |
| Number of tables: | 4 |
| Number of graphs: | 9 |
| Number of appendix attachments: | 4 |

# Appendix A: PNEC and the limitations of the PEC:PNEC ratio

**Definition and estimation of the PNEC**

Although the detailed definitions of risk differ some among the users of risk assessment methodologies, the basics of environmental risk assessment are universal. It is based on the comparison of the exposure of (a part of) the ecosystem to a chemical with the sensitivity of (the same part of) the ecosystem for this chemical (through this specific exposure-route) (Suter, 1993) (see figure A1 for a schematic overview). The exposure is represented by the predicted environmental concentration (PEC), and can be obtained by actual field measurements (monitoring data) or by estimations using environmental fate models. The toxicity threshold represents the sensitivity of the ecosystem, and is usually derived from results of standardised toxicity tests.

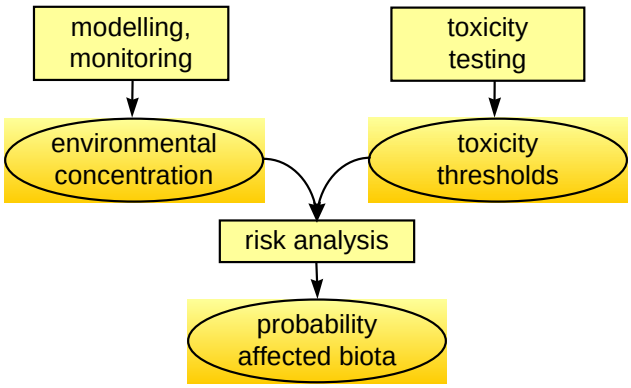


*Figure A1* *General framework for environmental risk assessment, based on the comparison of an environmental concentration with the sensitivity of the environment.*

One of the main challenges in environmental risk assessment is to have an adequate estimation of the sensitivity of the environment towards the toxicant. Usually one fixed value is derived to represent the sensitivity of the environment to a specific toxicant. This threshold is often referred to as the PNEC (Predicted No Effect Concentration). The definition of the PNEC according to the EU Technical Guidance Document (EC, 2003) is:

*The concentration below which unacceptable effects on organisms will most likely not occur.*

To perform an environmental risk assessment, choices have to be made about how information on the sensitivity of biota is used to estimate the PEC. The EU-TGD describes two ways to do this. The first is making use of assessment factors the second uses statistical extrapolation methods defining a Species Sensitivity Distribution (SSD). To illustrate the two methods figure A2 presents a frequency diagram with toxicity data for different species (NOECs; No observed effect concentrations, or EC50s; concentrations at which 50 % of the exposed individuals shows an effect). Figure A2-A shows how the lowest observed effect concentration is divided by an assessment factor (AF) to define the PNEC. Figure A2-B shows how a PNEC is derived from an SSD based on all available toxicity data. These two approaches are described further in the following.

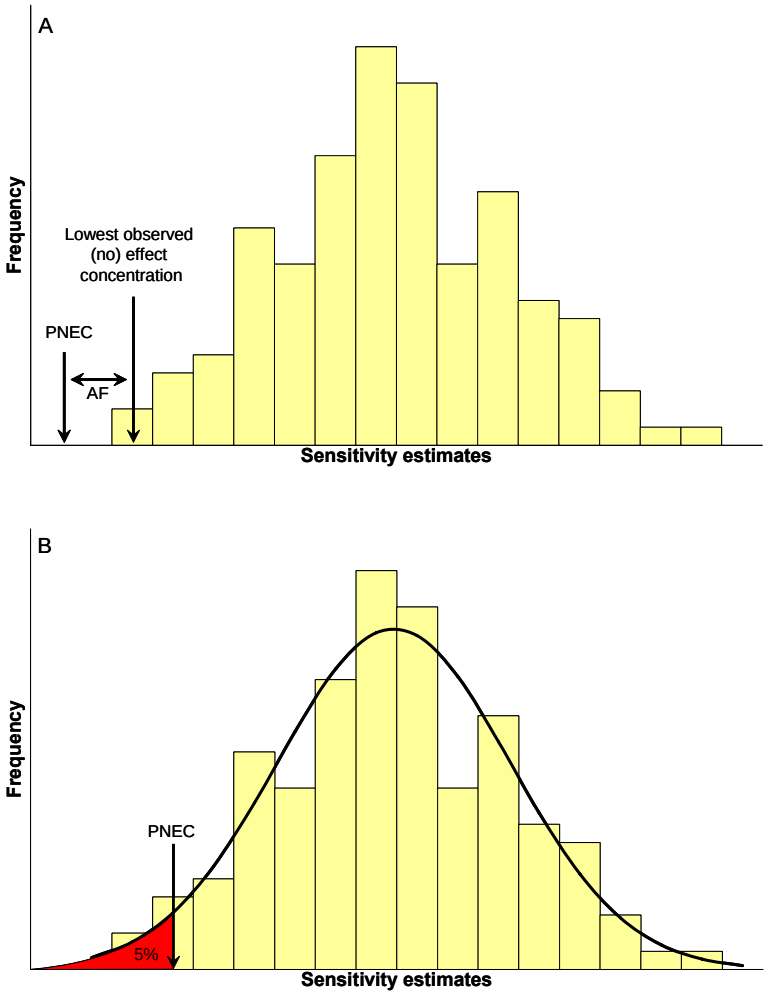


*Figure A2 Two methods to derive a PNEC value from available toxicity data. Figure A: PNEC derivation using an assessment factor. Figure B: PNEC derivation using Species Sensitivity Distributions*

**The use of an assessment factor**

When the lowest observed, toxicity value from the set of available data together with an assessment factor is used to derive the PNEC, its value is largely determined by the value of one data point for one single species. Assumptions are made concerning the aquatic environment which allow, however uncertain, an extrapolation to be made from single-species (short-term) toxicity data to ecosystem effects. It is assumed that:

- Ecosystem sensitivity depends on the most sensitive species, and;
- Protecting ecosystem structure protects community function.

These two assumptions have important consequences. By establishing which species is the most sensitive to the toxic effects of a chemical in the laboratory, extrapolation can subsequently be based on the data from that species. Furthermore, the functioning of any ecosystem in which that species exists is protected provided the structure is not sufficiently distorted as to cause an imbalance. It is generally accepted that protection of the most sensitive species should protect structure, and hence function. Schindler (1987) and others concluded that functional measures were relatively insensitive and that changes in the composition of the community were better indicators of ecosystem stress. In stressed ecosystems, shifts in community composition often preceded

changes in ecosystem function. Risk assessment using PEC:PNEC ratio completely focuses on the preservation of ecological structure. Important ecosystem dynamics like food web relations are not included.

For most substances, the pool of data from which to predict ecosystem effects is very limited as, in general, only short-term toxicity data are available. This pool of data is even reduced when strict quality criteria are applied to assess the intrinsic value of the data (Klimisch et al., 1997). In the light of the precautionary principle the lowest available toxicity threshold concentration is often not regarded as being protective for the complete ecosystem. On the basis of uncertainties, e.g., variation among species or between laboratory and field, one could therefore argue to apply an assessment factor to extrapolate the lowest available toxicity value to a PNEC. This value is considered being protective for sensitive species not present in the available dataset. In these circumstances, it is recognized that, while not having a strong scientific validity, empirically derived assessment factors must be used. Assessment factors have also been proposed by the US-EPA and OECD. In applying such factors, the intention is to predict a concentration below which any unacceptable effect will most likely not occur. It is not intended to be a level below which the chemical is considered to be safe. However, again, it is likely that an unacceptable effect will not occur (EC, 2003).

In principle, the PNEC is calculated by dividing the lowest LC/EC50 or NOEC value for three trophic groups of marine organisms by an appropriate assessment factor in accordance with the EU-TGD. The assessment factors are applied to extrapolate from laboratory single-species toxicity data to multi-species ecosystem effects. The assessment factors therefore address a number of uncertainties:

- Intra species variation (biological variance);
- short-term to long-term toxicity extrapolation;
- laboratory data to field impact extrapolation.

When only short-term toxicity data are available, an assessment factor of 1000 will be applied on the lowest L(E)C50 of the relevant available toxicity data, irrespective of whether or not the species tested is a standard test organism. A lower assessment factor will be applied on the lowest NOEC derived in long-term tests with a relevant test organism (EC, 2003).

*Table A1 The assessment factor scheme as used for calculating PNEC values (EC, 2003)*

| **Available toxicity data** | **Assessment factors** |
|---|---|
| At least one short-term L(E)C50 from each of three trophic levels (algae, crustaceans and fish) | 1000 |
| Long-term NOEC from one trophic level (either fish or crustaceans) | 100 |
| Long-term NOEC from species representing two trophic levels (fish and/or crustaceans and/or algae) | 50 |
| Long-term NOEC from at least three trophic levels (fish, crustaceans and algae) | 10 |
| Field data or model ecosystem | Reviewed on a case to case basis |

**Species Sensitivity Distributions**

The second method uses a statistical extrapolation method taking into account the variation in species sensitivity to define a PNEC (see Aldenberg & Jaworska (2000) for a review). If a large data set with NOECs from long-term experiments for different taxonomic groups is available, these values can be used to draw a distribution (as

presented in figure A2). This distribution that describes the variability of hazard of a substance to organisms is called a Species Sensitivity Distribution (SSD). In general the method works as follows: toxicity data are log transformed and fitted to a distribution function. For the description of dose-effect curves, several distribution functions have been proposed for this; Weibull distribution (Kodell and Felton, 1991), log-logistic (Kooijman, 1987), lognormal (Wagner and Løkke, 1991) etc. It has however 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). Figure A2-B presents a Species Sensitivity Distribution described by a log-normal distribution. For this cumulative log-normal distribution, NOEC 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 width of the curve. In terms of the sensitivity of species, the Xm gives an indication of the mean toxicity expressed as the mean NOEC value of a substance. The Sm represents the toxicity range or variation in sensitivity of a substance among species. The proposition that the distribution of sensitivity based on data from a selection of species (tested in laboratory experiments) is representative for the distribution of sensitivity for all species (in the field) is the main assumption on the use of SSDs in risk assessment (Aldenberg & Jaworska, 2000; Posthuma et al., 2002; Forbes & Calow, 2002a and 2002b).

Statistical extrapolation methods may be used to derive a PNEC from a 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 concentration (HC) at which a certain percentage (in this case 5%) of all species is assumed to be affected (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).

Statistical extrapolation can only be applied in cases where sufficient NOECs of good quality for sufficient species are available. Confidence can be associated with a PNEC derived by statistical extrapolation if the database contains at least 10 NOECs (preferably more than 15) for different species covering at least 8 taxonomic groups. (EC, 2003; Posthuma et al, 2002). If this is not the case the estimation of the PNEC and the variation between species becomes unreliable and assessment factors or other approaches should be used.

**PEC:PNEC ratio: Only an indicator of risk**

In the process of ecological risk assessment the (fixed) value of the PNEC is compared to an estimate of the exposure (PEC) e.g. the average or minimum exposure concentration.

With the ratio of PEC and PNEC (often referred to as the PEC:PNEC ratio or the RCR - Risk Characterisation Ratio) as endpoint for risk assessment it will be obvious that the import of this assessment endpoint is related to the definition of the PNEC. A PEC:PNEC ratio higher than 1 indicates that unacceptable effects on organisms are likely to occur; the higher the ratio, it is more likely that unacceptable effects may occur. When we relate this endpoint to the definition of risk as stated in the introduction of this report ("quantification of likelihood and severity of effects"), it becomes clear that the PEC:PNEC ratio does not comply with this definition. The severity of 'unacceptable effects' is not defined nor is the probability level of 'most likely'. Consequently the ratio itself does not provide a quantification of the environmental risk. (See also Scholten et al., 2000). The PEC:PNEC ratio is just an indication of the likelihood. This is OK for identification of possible impacts and for prioritisation. However, it does not provide any characterisation of the expected impact.

Typical risk assessment models, based on the PEC:PNEC approach, which are applied for the use of chemicals by the oil and gas industry, are EUSES (European Union System for the Evaluation of Substances) and CHARM (Chemical Hazard Assessment and Risk Management). Both models are specially developed for use in protective hazard assessments and prioritizing of chemicals and are adopted by regulators in the EU. EUSES is developed in the EC for risk assessment of new and existing substances and pesticides (Vermeire et al., 1997) and based on the Dutch USES model (Jager & Visser, 1994). Within OSPAR the hazard assessment module of CHARM is the mandatory tool for ranking offshore chemicals on the basis of their relative calculated risk with the aim of selecting the environmentally most friendly production, drilling, or work-over chemical (Thatcher et al, 1999).

For both models the endpoint of the risk assessment is a PEC:PNEC ratio. The severity of effects and likeliness of the occurrence of these effects are therefore undefined. This makes these models suitable for ranking but unsuitable for impact assessment. In order to quantify the likelihood and characterise the impacts the PEC:PNEC ratio can be a base, but is definitely not the endpoint. Existing risk assessment models with an assessment endpoint based on a PEC:PNEC ratio can be applied for prioritising. When it is the objective to assess the actual impacts the PEC:PNEC ratio is not sufficient.

Also for contaminated area's the PEC:PNEC ratio is frequently applied to indicate the likelihood of occurrence of environmental impacts. In light of the above, it can be questioned whether the use of the PEC:PNEC ratio for this purpose is relevant. As matter of screening, the analysis using the PEC:PNEC ratio will be sufficient, however to indicate actual impacts, other methods (bioassays) should be applied.

Developments in risk assessment endpoints are needed to make the risk assessment more quantitative leading to a tiered approach The following steps can be identified in this tiered approach:

PEC:PNEC ratio; quick screening of a worst case situation

Probabilistic risk assessment; this approach uses the same toxicity data as used in the PEC:PNEC approach but results in a more quantified risk estimate (probability that a species is exposed above its chronic NOEC).

Mechanistic risk assessment; this approach uses new information from the same toxicological experiments as used in the PEC:PNEC approach, but actual biological mechanisms are implemented.

Ecological modelling in risk assessment; this approach uses the state of the art knowledge on biology, ecology and toxicology and integrates principles to estimate effects on populations, food chains, ecosystems.

**Probabilistic risk assessment**

When statistical extrapolation methods are used to derive the PNEC from an SSD curve, its value will correspond to a probability of 5% of a random species being exposed above its chronic no effect concentration (which can be statistically tested and be regarded as a confidence interval) (e.g. Van Straalen & Denneman, 1989, Aldenberg & Slob, 1993; Newman et al., 2000; Van der Hoeven, 2001; Aldenberg & Jaworska, 2001)). In stead of using just the PNEC value from the SSD its also possible to use the whole SSD to represent the sensitivity of the ecosystem (Schobben and Scholten, 1993, ). When the SSD is based on chronic NOECs for the most sensitive endpoints for different species, a comparison of the SSD with a PEC does give a quantification of the likelihood effects (see Verdonck 2003, for an overview) (e.g 10% probability that a random species is negatively effected).

The use of probabilistic methodologies still leaves subjects open for interpretation. The challenge that remains for ecologist and ecotoxicologists is the definition of (un)acceptable effects in relation to the most sensitive

endpoints on the species level related to adverse effects to communities. ERA using SSDs completely focuses on the preservation of ecological structure. Important ecosystem dynamics like food web relations are not included. However, it is assumed that ecological functioning is also protected when the ecological structure is protected. Schindler (1987) and others concluded that functional measures were relatively insensitive and that changes in the composition of the community were better indicators of ecosystem stress. In stressed ecosystems, shifts in community composition often preceded changes in ecosystem function. However, predicting ecosystem function has a higher value because it integrates responses of component populations (Baird et al., 2001). Developments in risk assessment models should therefore focus on the translation from single species level to ecosystem level.

However, even in this revised state there are limitations 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. Besides that the question was raised how representative the 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 unacceptable effects on the ecosystem in relation to the most sensitive endpoints on the species level. Thus developments in risk assessment models should focus on the translation from laboratory species to field communities. And besides this uncertainties in the risk assessment procedure should always be stated clearly. (Calow & Forbes, 2002).

**Mechanistic risk assessment**

This type of risk assessment is founded on theoretical biological and toxicological principles. Distributions are no longer used to describe the results of toxicity experiments. The mechanistic equations in this model together with experimental data make it possible to relate exposure time and concentration to effects for a selection of representative species. With the right selection of representative species a prognosis of effects on the ecosystem level can be obtained from the estimated effects on the (representative) species level. Finally a risk estimate for the ecosystem can be derived based on variable exposure over (a certain period of) time (Jager and Kooijman (in press), Kooijman and Bedeax, 1996).

**Ecological modelling in Risk assessment**

Mechanistic risk assessment quantifies the levels of effect for a selection of species. This information can be incorporated in ecological models in which food web relations and ecosystem structures and functioning is incorporated. With these models realistic perditions on the severity of effects on the ecosystem level can be made.

It will be clear that the higher the tier of the risk assessment the more specific information will be needed on species, systems and toxic properties. Hence, there must be a justification for such a tiered approach which can be found in the area or toxicant of special concern.

Taking into account the proposed endpoints for monitoring of biological effects, the developments is risk assessment can be related to these endpoints. For impacts on the seabed this will mean an endpoint in risk assessment on the ecosystem level to be related to community indices as measured in monitoring. For impacts on the water column this will mean an endpoint on the species level, which can be probabilistic or mechanistic with relevant toxicity endpoints, to be related to biomarker responses.

**Conclusions**

The PEC:PNEC ratio is very well suited for screening purposes and prioritising hazards as it is applied in CHARM and EUSES. However, it does not comply with the definition of risk in various guidelines for risk assessment. It hardly provides a quantification of the likelihood of the occurrence of effects and does certainly not provide a characterisation of the extend of effects. The PEC:PNEC is only an indicator of risk.

For more sophisticated and realistic assessment of ecological risks the PEC:PNEC ratio should be developed further towards a more quantified way of risk assessment. A tiered approach could be developed for areas and toxicants of special concern. When this tiered approach is connected to the endpoints of biological effect monitoring, this monitoring could be used as a validation of the risk assessment process. This will make it possible to optimize risk assessment methodologies and defend minimization of expensive monitoring efforts in the future. In short, a joint development of both risk assessment and monitoring is necessary in order to promote integrative science (Chapman, 1995).

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# Appendix B: Produced water components

*Table B1* *Concentration of BTEXs in discharged produced water of the 4 selected platforms (mg/L) as measured in the years 2005/2006; values provided by NOGEPA.*

| Substance | Platform 1 (gas) | Platform 2 (gas) | Platform 3 (gas) | Platform 4 (oil) |
|---|---|---|---|---|
| Benzene | 111.8 | 9.9 | 53.9 | 1.0 |
| Ethyl benzene | 0.4 | 0.2 | 0.1 | <0.1 |
| m-Xylene and p-Xylene | 1.5 | 0.8 | 0.6 | <0.1 |
| o-Xylene | 2.0 | 0.4 | 0.6 | <0.1 |
| Toluene | 11.1 | 4.3 | 3.3 | 0.3 |
| **sum** | **126.8** | **15.6** | **58.5** | **<1.6** |

*Table B2* *Concentration of Naphthalenes in discharged produced water of the 4 selected platforms (µg/L) as measured in the years 2005/2006; values provided by NOGEPA.*

| Substance | Platform 1 (gas) | Platform 2 (gas) | Platform 3 (gas) | Platform 4 (oil) |
|---|---|---|---|---|
| 1-Methylnaphthalene | 22 | 9 | 94 | 6 |
| 2,6-Dimethylnaphthalene | <2 | <2 | 14 | 3 |
| 2-Isopropylnaphthalene | <2 | <2 | <2 | 2 |
| 2-Methylnaphthalene | 7 | 13 | 82 | 7 |
| Acenaphthene | <2 | <2 | <2 | <2 |
| Acenaphthylene | <2 | <2 | <2 | <2 |
| Naphthalene | 350 | 83 | 829 | 15 |
| Other $C_2$ Naphthalenes | 7 | 4 | 72 | 11 |
| Other $C_3$ Naphthalenes | 14 | 7 | 100 | 58 |
| **sum** | **<408** | **<124** | **<1197** | **<106** |

*Table B3 Concentration of PAH 2-3 ring in discharged produced water of the 4 selected platforms (µg/L) as measured in the years 2005/2006; values provided by NOGEPA.*

| Substance | Platform 1 (gas) | Platform 2 (gas) | Platform 3 (gas) | Platform 4 (oil) |
|---|---|---|---|---|
| 1,2,6-Trimethylphenanthrene | <2 | <2 | <2 | <2 |
| 4-Ethyldibenzothiophene | <2 | <2 | <2 | <2 |
| 4-Methyldibenzothiophene | <2 | <2 | 6 | <2 |
| 4-Propyldibenzothiophene | <2 | <2 | <2 | <2 |
| 9-Ethylphenanthrene | <2 | <2 | <2 | <2 |
| 9-Methylphenanthrene | <2 | <2 | 8 | 2 |
| Anthracene | <2 | <2 | 2 | <2 |
| Dibenzothiophene | 3 | <2 | 27 | <2 |
| Fluorene | <2 | <2 | 54 | <2 |
| Other $C_2$ Dibenzothiophenes | <2 | <2 | 4 | <2 |
| Other $C_2$ Phenanthrenes | <2 | <2 | 20 | 12 |
| Other $C_3$ dibenzothiophenes | <2 | <2 | 4 | 5 |
| Other $C_3$ Phenanthrenes | <2 | <2 | 5 | 9 |
| Other Methyldibenzothiophenes | <2 | <2 | 4 | <2 |
| Other Methylphenanthrenes | <2 | <2 | 26 | 4 |
| Phenanthrene | 7 | <2 | 86 | 3 |
| **sum** | **<38** | **<32** | **<254** | **<55** |

*Table B4 Concentration of PAH 4+ ring in discharged produced water of the 4 selected platforms (µg/L) as measured in the years 2005/2006; values provided by NOGEPA.*

| Substance | Platform 1 (gas) | Platform 2 (gas) | Platform 3 (gas) | Platform 4 (oil) |
|---|---|---|---|---|
| Benz(a)anthracene | <2 | <2 | <2 | <2 |
| Benzo(a)pyrene | <2 | <2 | <2 | <2 |
| Benzo(b)fluoranthene | <2 | <2 | <2 | <2 |
| Benzo(g,h,I)perylene | 2 | <2 | <2 | 3 |
| Benzo(k)fluoranthene | <2 | <2 | <2 | 2 |
| Chrysene | <2 | <2 | <2 | <2 |
| Dibenz(a,h)anthracene | <2 | <2 | <2 | <2 |
| Fluoranthene | <2 | <2 | <2 | <2 |
| Indeno(1,2,3,cd)pyrene | 4 | <2 | <2 | 6 |
| Pyrene | <2 | <2 | <2 | <2 |
| **sum** | **<22** | **<20** | **<20** | **<25** |

*Table B5 Concentration of alkylphenol in discharged produced water of the 4 selected platforms (µg/L) as measured in the years 2005/2006; values provided by NOGEPA.*

| Substance | Platform 1 (gas) | Platform 2 (gas) | Platform 3 (gas) | Platform 4 (oil) |
|---|---|---|---|---|
| C1 - C3 Alkylphenol | 13032 | 29788 | 8953 | 1057 |
| Other C1 - C3 Alkylphenol | 1403 | 3354 | 929 | 367 |
| **sum C1-C3** | **14435** | **33142** | **9882** | **1424** |
| C4 - C5 Alkylphenol | 94 | 8 | 34 | 17 |
| Other C4 - C5 Alkylphenol | 80 | 106 | 57 | 33 |
| **sum C4-C5** | **174** | **114** | **91** | **50** |
| C6 - C9 Alkylphenol | 14 | <2 | 44 | <2 |
| Other C6 - C9 Alkylphenol | 68 | <2 | 615 | 18 |
| **sum C6-C9** | **82** | **<4** | **659** | **<20** |

*Table B6 Metal concentrations in discharged produced water of the 4 selected platforms (µg/L) as measured in the years 2005/2006; values provided by NOGEPA.*

| Substance | Platform 1 (gas) | Platform 2 (gas) | Platform 3 (gas) | Platform 4 (oil) |
|---|---|---|---|---|
| Arsenic (As) | 28.6 | 0.52 | <0.5 | 87.1 |
| Cadmium (Cd) | 167 | <0.5 | <0.5 | <0.5 |
| Chromium (Cr) | 10.6 | 14.6 | 10.6 | 45.9 |
| Copper (Cu) | 18.8 | 32.1 | 31.7 | 47.7 |
| Lead (Pb) | 13220 | 8.46 | 51 | 1.83 |
| Mercury (Hg) | 4.18 | 0.69 | 1.06 | <0.06 |
| Nickel (Ni) | 60.1 | 17.2 | 78.6 | 62.7 |
| Zinc (Zn) | 5859 | 155 | 836 | 73.5 |

*Table B7 Concentration of oil in water components in discharged produced water of the 4 selected platforms (mg/kg) as measured in the years 2005/2006; values provided by NOGEPA.*

| Substance | Platform 1 (gas) | Platform 2 (gas) | Platform 3 (gas) | Platform 4 (oil) |
|---|---|---|---|---|
| Oil | 143 | 26 | 84 | 27 |
| Aromatic Portion | 2 | 22 | 83 | 4 |
| Aliphatic Portion | 141 | 4 | 1 | 23 |

# Appendix C: Discharge characteristics

This appendix contains some discharge characteristics of the selected platforms, which was used as input for the calculations in this report.

*Tabel C1 Discharge characteristics of the selected platforms*

| **Platform** | **Discharged volume [$m^3$/year][5]** | **Discharge depth [m][6]** | **T[7] [°C]** | **Salinity[8] (g/L)** | **Diameter discharge pipe [m]** |
|---|---|---|---|---|---|
| Platform 1 (gas) | 39181 | 1[9] | 32 | 36 | 0.1016 |
| Platform 2 (gas) | 27444 | 21 | 56 | 67.9 | 0.1016 |
| Platform 3 (gas) | 27042 | 1[9] | 22 | -[10] | 0.1016 |
| Platform 4 (oil) | 1849411 | 12 | 70 | 83.7 | 0.254 |

The orientation of the discharge pipes was not provided. The pipes were assumed to be directed downward in a vertical position.

[5] Average of the reported values for the years 2005 and 2006
[6] Depth relative to the water surface
[7] Temperature of the discharged produced water
[8] Salinity of the discharged produced water
[9] The produced water is discharged above sealevel, which cannot be handled by the models. Therefore, a discharge depth of 1 m below the water surface is assumed
[10] Salinity is unknown, the weight density of the water is known in stead: 1.000 kg/L

# Appendix D: OSPAR requirements

This Appendix lists OSPAR requirements, regarding risk assessment and provides a brief overview whether the models comply to these requirements and how (Table D1). Keeping in mind the TGD[11], OSPAR intends to use risk assessment, when setting priorities and in assessing the nature and extent of the programmes and measures and their time scales, using the criteria given below[12, 13]:

a) persistency;
b) toxicity or other noxious properties;
c) tendency to bioaccumulation;
d) radioactivity;
e) the ratio between observed or (where the results of observations are not yet available) predicted concentrations and no observed effect concentrations;
f) anthropogenically caused risk of eutrophication;
g) transboundary significance;
h) risk of undesirable changes in the marine ecosystem and irreversibility or durability of effects;
i) interference with harvesting of sea-foods or with other legitimate uses of the sea;
j) effects on the taste and/or smell of products for human consumption from the sea, or effects on smell, colour, transparency or other characteristics of the water in the marine environment;
k) distribution pattern (i.e., quantities involved, use pattern and liability to reach the marine environment);
l) non-fulfilment of environmental quality objectives.

11 EC. 2003. Technical Guidance Document on Risk Assessment in support of Commission Directive 93/67/EEC on Risk Assessment for new notified substances Commission Regulation (EC) No 1488/94 on Risk Assessment of existing substance Directive 98/8/EC of the European Parliament and Council concerning the placing of biocidal products on the market. *Rep. EUR 20418 EN/2*, European Commission Joint Research Centre.

12 OSPAR. 2007. Annex III - On the Prevention and Elimination of Pollution from Offshore Sources. *Convention for the Protection of the Marine Environment of the North-East Atlantic (1992 OSPAR Convention)*.

13 OSPAR. 2007. Appendix 2 - Criteria Mentioned in Paragraph 2 of Article 1 of Annex I and in Paragraph 2 of Article 2 of Annex III. *Convention for the Protection of the Marine Environment of the North-East Atlantic (1992 OSPAR Convention)*.

*Table D1 An overview of how the different methods account for OSPAR requirements as listed in paragraph in the text above; dashes indicate the method doesn't account fort hat specific aspect*

| **OSPAR criteria** | | **CHU** | **DREAM** | **EIF** | **PROTEUS** | **MIKE** | **Delft3D in combination with effect assessment** |
|---|---|---|---|---|---|---|---|
| a | Persistency | - | First order degradation constants | First order degradation constants | Not indicated by TRENDS report | First order degradation constants, or more complex in ECOLAB | First order degradation constants (optional) |
| b | Toxicity | PNEC | Added mortality rate, based on internal concentrations (DEBTOX) | PNEC | PNEC | PNEC | PNEC |
| c | Bioa-ccumulation | - | Bioconcentration as first order uptake kinetics; Biomagnification not accounted for | Simple weighing factor | Bioconcentration included in body burden maps; Biomagnification not accounted for | - | - |
| d | Radioactivity | - | - | - | - | - | - |
| e | PEC:PNEC ratio | At 500 m distance from platform | - | In 3D | In 3D | In 3D | In 3D |
| f | Eutrophication | - | - | - | - | Is only used as input for complex fate studies with ECOLAB | - |
| g | Changes, durability and irreversibility | - | - | - | - | - | - |
| h | Interference harvesting | - | - | - | - | - | - |
| i | Taste and smell | - | - | - | - | - | - |
| j | Distribution pattern | Only PEC:PNEC at 500 m | Spatial distribution (3D) of affected species | Spatial distribution (3D) of PEC:PNEC | Spatial distribution (3D) of PEC:PNEC and body burdens | Spatial distribution (3D) of PEC:PNEC | Spatial distribution (3D) of PEC:PNEC |
| k | Non-fulfilment of EQSs | When PNEC is set to EQS | - | When PNEC is set to EQS | When PNEC is set to EQS | When PNEC is set to EQS | When PNEC is set to EQS |