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Overall evaluation of offshore drilling fluid technology

Development and application of life-cycle inventory and impact assessment methods

Thesis for the degree of doctor philosophiae

Trondheim, December 2007

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Words of wisdom:

In order for something to become clean, something else must become dirty.
(For some time listed as Imbesi's Law of the Conservation of Filth in Wikipedia)

There ain't no such thing as a free lunch.
(The TANSTAAFL principle; Robert A. Heinlein: The moon is a harsh mistress, 1966)

Abstract

The goal of this thesis is to provide the means for discussion of overall benefits of alternative offshore drilling technologies. Life-cycle assessment is used to assess environmental impact of alternative drilling technologies. Life-cycle assessment is well-suited for relative comparison and it offers the broad perspective necessary to evaluate overall performance.

Several methodological developments are made within the framework of life-cycle assessment to support the evaluation of offshore drilling fluid technology.

Offshore discharges to the marine environment during drilling operations are pulse emissions. The relative marine aquatic ecotoxic impact of pulse emissions compared to continuous emission processes is investigated by transient dispersion modeling.

Occupational health is an important decision objective for offshore operations. Crane-lifts are an important cause of accidents with human health damages on drilling rigs. A characterization factor for offshore crane-lifts is developed to include occupational health in life-cycle assessment.

Long-term release of metals from solid wastes is important for the ecotoxicity of drilling wastes. A review is presented that considers the current and possible solutions to address long-term leaching processes in life-cycle assessment.

An overall evaluation of offshore drilling fluid technology is performed. The study assesses the relative life-cycle performance of alternatives for density control in drilling fluids (ilmenite versus barite), offshore loading systems (crane-lifts versus a hydraulic system), base drilling fluids (water-base versus oil-base), and waste treatment of cuttings drilled with water-based drilling fluid (offshore discharge versus onshore treatment). A well located in the Barents Sea is used as reference.

Results are interpreted using Monte Carlo simulation. Preferred alternatives from an overall evaluation are proposed.

This thesis illustrates the challenges of life-cycle assessment. Most product systems require adaptation and development of methods for proper evaluation of impacts and results that meet requirements for decision objective attributes.

Acknowledgement

The subject of this thesis can be traced back to a student project I did for Statoil ASA in 2002 together with two other students. The task then was to use life-cycle assessment to evaluate various weight agent minerals in drilling fluid. John Eirik Paulsen, in charge of Total Fluid Management in Statoil, was one of the project supervisors. Fortunately for me, and the subject of this thesis, he came from the task with remained interest in the life-cycle approach. I am grateful to John Eirik for his continuous support of my work, although I suspect he often wanted to stress technical and policy issues rather than the assessment methodology.

I am thankful also to Statoil ASA for providing the financial means to work with this thesis.

There are several people that have participated in the development of my work, either by tolerating me as their office roommate, challenging me on the approaches that I have taken, or generally offering a good environment both at work and after. Although the smartest thing to do probably would be to not mention names, I feel I need to list a few: Anders, Stig, Frank, Marte, Håvard, Christian, and Glen, thank you!

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Edgar Hertwich has been my supervisor during the thesis work. He has given me room to follow my own interests, for which I am grateful. Now, when completing the thesis, my feeling of achievement is greater due to this freedom, although it has been a little frustrating during the course of the work.

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Johan Pettersen

Trondhjem, November 2007

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Johan Pettersen, Glen P. Peters and Edgar G. Hertwich (2006): Marine ecotoxic effect of pulse emissions in life cycle assessment. *Environmental Toxicology and Chemistry* **25**(1): 297–303

Appendix C – Paper 2
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List of abbreviations

AP	acidfication potential
BAT	best available technology (according to OSPAR) best available technique (according to the European Council Integrated Pollution Prevention and Control Directive)
BREF	BAT reference, also aquatic ecotoxicity potentials as outlined by the BAT reference document
CML	Institute of Environmental Sciences (CML), at Leiden University (Netherlands)
DALY	disability adjusted life years
DCB	dichlorbenzene, usually used with reference to 1,4-DCB
DTPA	diethylene triamine pentaacetic acid
EP	eutrophication potential
FAETP	freshwater aquatic ecotoxic potential
FU	functional unit
FWT	freshwater aquatic ecotoxic potential
GWP	global warming potential
HC50	concentration hazardous to 50 percent of the ecological community (of species)
HOCNF	harmonized offshore chemical notification format
HT	human toxicity potential
IPCC	United Nations Intergovernmental Panel on Climate Change (www.ipcc.ch)
IPPC	Integrated Pollution Prevention and Control; European Council IPPC Directive
ISO	International Organization for Standardization (www.iso.org)
LC_{50}	concentration lethal to 50 percent of the population
LCA	life-cycle assessment
LCI	life-cycle inventory
LCIA	life-cycle impact assessment
MAT	marine aquatic ecotoxic potential
MAETP	marine aquatic ecotoxic potential
MSR	marine sediment risk
NOEC	no-effect concentration
OB	oil-based
OBM	oil-based mud, oil-based drilling fluid
OLD	ozone-layer depletion potential
OSPAR	Oslo-Paris Convention for the protection of the marine environment of the North-East Atlantic (www.ospar.org)
PAH	polyaromatic hydrocarbons
PLONOR	PLONOR substances are substances considered to Pose Little Or NO Risk
PM nr	ecotoxic potentials for priority metals
ROP	rate-of-penetration
SETAC	Society of Environmental Toxicology and Chemistry (www.setac.org)
SLV	ecotoxicity potentials extracted from soil limit values
SSD	species sensitivity distribution
TET	terrestrial ecotoxic potential
TU	toxic units
WB	water-based
WBM	water-based mud, water-based drilling fluid

1 INTRODUCTION

1.1 Background

The Norwegian economy and the marine environment are strongly dependent of each other. Offshore oil and gas extraction represented about one quarter of the Norwegian gross national product in 2006 (SSB 2007). At a length of 25,000 km, the coastline is a dominant feature of Norway's nature. The fjords are a significant attraction for the tourism industry. The marine fishing sector is cornerstone to Norwegian culture. Fish products stand for about 5% of Norwegian export value and around 0.25 percent of the population list commercial fishing as their main occupation (SSB 2006).

With the remaining discovered oil and gas resources depleting, the petroleum industry is moving to areas previously unavailable due to technical or political reasons. A large part of the global undiscovered oil and gas resources are suspected to be found in the northern areas. These areas are also of high importance to fish stocks and other biological resources (Føyn et al. 2002).

The Arctic ecological system is considered particularly sensitive to environmental changes due to its complex nature and harsh environmental conditions. At the same time, other environmental impacts receive interest on the international and national scene. Former priority issues include ozone layer depletion and acidification. Global warming impacts are receiving increasing interest. While each of these problems deserve attention, it is important to have a systems perspective in mind when addressing them. Solutions should be found that do not solve one problem at the cost of another.

The current evaluation procedure for offshore activities revolves around potential effects on the marine environment from planned and accidental emissions to the ocean. The marine focus is apparent in the scientific literature; see Patin (1999), as well as documents developed for policy support, such as the recent impact assessment that was undertaken by the Norwegian Ministry for Petroleum and Energy prior to reopening the Barents Sea for oil and gas extraction (OED 2002).

The single-issue focus stands in stark contrast to current developments in design of regulatory instruments, which take a broader perspective to environmental decision making. The prescription for best available technique, outlined by the *Integrated pollution prevention and control directive* (European Council 1996) is an example of a governance structure with a systems approach. It asks that solutions be preferred from an overall evaluation of environmental impacts rather than based on evaluation of a single issue.

The environmental policy debate supports the shift to overall evaluations. Norwegian governmental policy documents for the environment cover an extended list of priority issues besides the marine environment (MD 1997; MD 2002b; MD 2005). These issues are addressed by separate policy instruments, but are also covered by systems instruments such as the *Regulation concerning pollution* (Norw.: Forurensningsforskriften). An overall evaluation of drilling fluid technology thereby must address impacts besides those to the marine environment.

The purpose of systems analysis is to evaluate systems level performance on decision objectives. As illustrated by Figure 1, a relative evaluation of the performance of drilling fluid technologies must include complete process alternatives, and needs to consider the overall environmental impacts rather than limit the focus to a subset of impacts. The aim of this thesis is to apply a broad perspective to assess the environmental performance of drilling fluid technologies. A life-cycle perspective is chosen for the reason that application of any process in offshore operations has repercussions related to processes upstream and downstream from the rig. The selection of fluid technology has consequences for the production of chemicals, equipment, and fuels, as well as waste logistics and impacts from waste treatment.

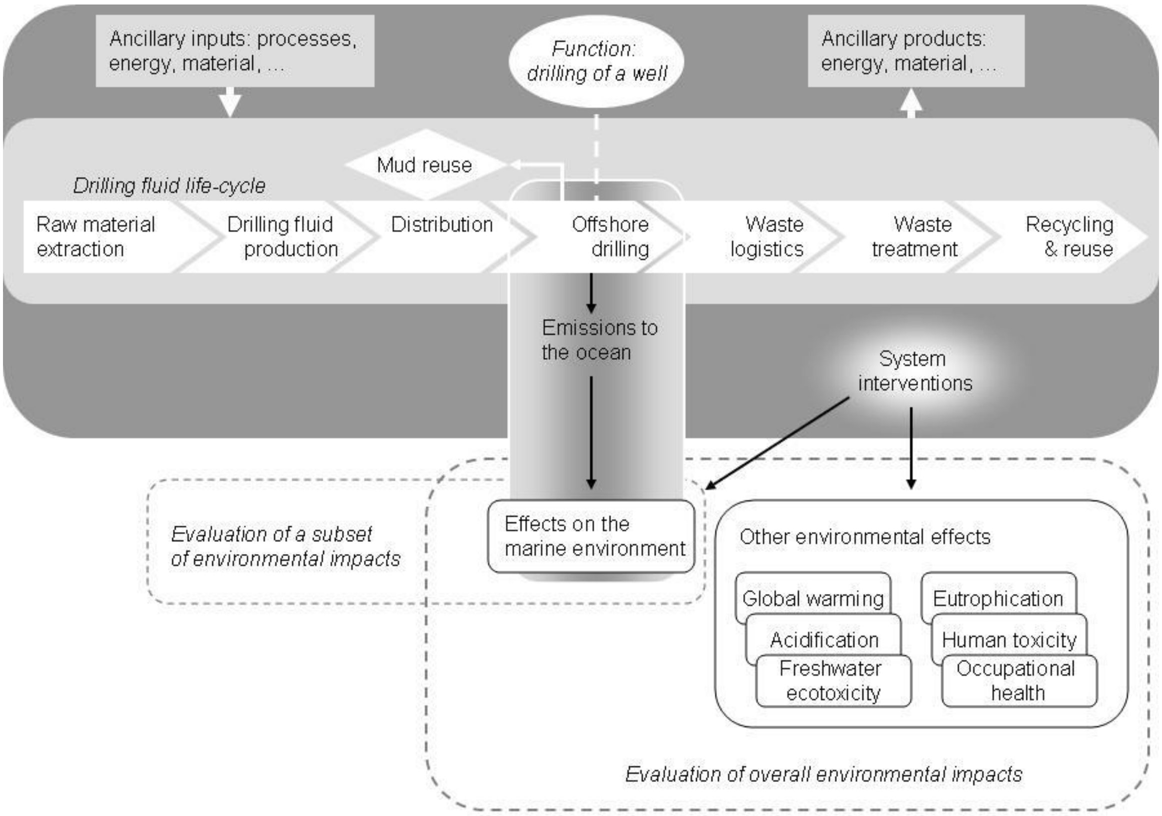

Figure 1: Environmental interventions and impacts in a systems perspective

Some terms are defined briefly for the benefit of readers not familiar with drilling terminology. Drill cuttings, or cuttings, is the solid rock material carved out of the well during drilling. Drilling fluid, often referred to as drilling mud, is a viscous fluid phase that is pumped down-hole through the drill-string and drill-bit. The fluid aids in the transport of cuttings out of the well. It also helps maintain well structure, lubricate the drill-bit and balance pressure down-hole.

1.2 Why life-cycle assessment

Offshore drilling operations are intermittent and complex. Operations may last from weeks to several months, involving a number of various suppliers and stages in the drilling process. Upon completion of the well, the rig is moved to a new location with new technical and environmental challenges. Each well is planned and executed as a separate project, including the choice of drilling chemicals, logistics for chemicals and waste, waste contracts and rig designs.

External parameters that vary from one operation to the next include sub-sea geological properties, weather conditions, ecosystem sensitivity, availability of waste treatment facility options onshore, rig-space limitations, safety considerations, and whether the drilling operation is part of a larger drilling campaign. All aspects influence the design of the drilling fluid technology. Although the crude setup of technology may not change much, fluid systems are continuously upgraded and other technologies constantly evaluated during operations.

About fifty different chemical systems exist for drilling operations, each with specific and different chemical compositions. The systems show different properties with respect to the amount and characteristics of waste they produce, reuse value, recycling ability, and the extent to which they may be separated from the waste that they produce. They also differ greatly in the ecotoxic risk they pose upon offshore discharge or onshore waste treatment.

The alternatives for each operation are practically endless but the function that they provide is the same in every situation: the drilling of a well for exploration or production purposes. The evaluation of a number of alternative methods for serving a function is well suited for life-cycle assessment (LCA).

Vital in application of life-cycle assessment for product system development and comparison is the identification of trade-offs made during design and in selection of alternatives. The assessment must therefore address stakeholder decision objectives.

1.3 Aim of the study

The aim of this study is *to perform comparative life-cycle assessment of offshore drilling fluid technology alternatives*. The term drilling fluid technology encompasses technology applied as part of the drilling fluid itself as well as technology that is used within the life-cycle of the drilling fluid and complementary wastes.

Many attributes relevant for an overall evaluation of drilling fluid technology cannot be assessed within the existing life-cycle assessment framework due to gaps in inventory and impact assessment methods. A prerequisite for achieving the stated goal of this thesis therefore is to bridge the gap between currently available methods to assess environmental and human health impacts with life-cycle assessment and the decision objectives posed by stakeholders to the drilling process. An alternative definition of the aim of this thesis thereby is *to develop and apply inventory and impact assessment methodology for comparative life-cycle assessment of drilling fluid technology alternatives*.

1.4 Structure of the work

Relevant aspects of life-cycle assessment are introduced and discussed in Chapter 2. Chapter 3 provides an overview of oil and gas drilling operations and describes the role of drilling fluids.

The first step towards the aim of the thesis is achieved by identifying the significant decision objectives for evaluation of offshore drilling technology. Gaps must be bridged by methodological development in case life-cycle assessment fails to provide answers on these issues. The identification of gaps in life-cycle assessment methods is the subject of Chapter 4.

Development of inventory and impact assessment methods is discussed in three separate papers. Full papers are attached to the thesis. Chapter 5 provides a synopsis of the main findings, methods used to achieve them, and the respective conclusions related to overall evaluation of offshore drilling fluid technology.

Finally, methods are applied to make overall evaluations. Several technologies are investigated by comparative life-cycle assessment. The study is described in Chapter 6, following the framework of life-cycle assessment. Conclusions from the case study and methodological developments are summarized in Chapter 7.

An overview of the workflow, with respective documents in the thesis and appended papers, is given in Figure 2.

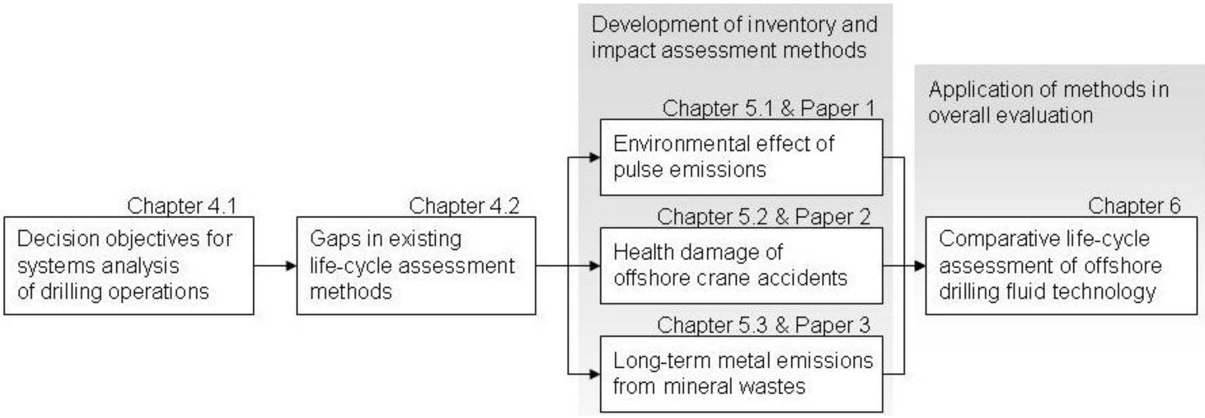

Figure 2: Thesis workflow.

1.5 Scope and limitations

1.5.1 Technology selection

The scope of this thesis is set by the technologies which are considered. One comparison that often emerges in a discussion of drilling fluid technology is that of an oil-based drilling fluid versus a water-based drilling fluid. Oil-based fluids have a continuous phase of mineral or synthetic base oil. Water-based drilling fluids generally present less environmental risk upon discharge, leading to water-based muds being preferred from the principle of substitution towards use of less ecotoxic chemicals. However, water-based drilling fluids generally produce more cuttings waste. (The term *cuttings* refer to the solid rock phase carved out of the well.) Water-based fluids thereby lead to increased emissions in transportation and waste treatment if cuttings are transported to shore. Moreover, transport operations carry occupational health burdens by an increased need for loading operations off rig and at dock.

A new loading system was recently installed on an exploration rig in the Barents Sea. The system is a hydraulic pump unit that replaces the use of containers and crane-lifts to load solid drilling waste onto and off the supply vessel. The hydraulic system removes the need for crane-lifts and thereby reduces risk for crane accidents. It does, however, require more energy than the traditional crane-lifts. A second comparison therefore is the processes required to produce and use the hydraulic system, compared to the savings by the associated reduction in accident risk.

A third comparison is the mineral used to add density to drilling fluids. Here are considered two of the alternatives: barite and ilmenite. Barite ($BaSO_4$) contains heavy metals both as trace metals within the mineral matrix but also as barium part of the matrix itself. The alternative, ilmenite (FeO, MgO)TiO_2), has an ecotoxically benign matrix, but still holds heavy metals within its crystal structure. These two minerals are the main alternatives used to balance the density of drilling fluids. The most important environmental differences are related to differences in production, metal leaching potential and transport needs. Barite is traded globally, while ilmenite used in Norway is mined in Sokndal (Norway).

The reopening of the Barents Sea for oil and gas activities was much debated in Norway. Permits were issued under strict requirements for clean operations. The practice of discharging cuttings drilled with water-based fluids, generally permitted in Norwegian waters, is prohibited within the Barents Sea area. This presented offshore operators with a novel situation in which use of water-based fluids is preferred by regulators from a principle of substitution, while the cuttings drilled with such fluids must be transported to shore for treatment. Onshore treatment represents impacts in terms of occupational accident potential and emissions caused by transportation and treatment. A fourth comparison therefore is the comparison of treating drilling waste from offshore operations in the Barents Sea onshore, or if a better option is to allow them to be discharged offshore at the rig site.

The issues discussed above form the technologies selected for overall evaluation in this thesis. Four comparisons are selected:

- Weight material: ilmenite versus barite
- Loading system: crane-lifts versus the hydraulic system
- Fluid system: water-based versus oil-based drilling fluid for operations in the Barents Sea (where all cuttings must be transported to shore)
- Treatment of wastes from drilling with water-based fluids: onshore treatment versus offshore discharge of cuttings waste

1.5.2 Limitations of work

As has been stated above, the goal is to compare the environmental performance of drilling fluid technology relative to an alternative solution. The perspective of this thesis is thereby comparative rather than absolute. The purpose is limited to overall evaluation from a comparative perspective, with the goal of discerning alternatives. The comparative perspective is, however, on a systems level. The aim is to offer conclusions regarding alternatives being identical in terms of environmental impacts, or recommending one of the alternatives. This has consequences for the system boundaries applied in the evaluation, and drives the development of methods necessary to reach the stated aims.

Technology alternatives are considered in the context of offshore drilling operations. Results thereby are intended applicable to the offshore situation.

There is a growing volume of literature on the marine and onshore ecotoxic risk caused by drilling wastes. The goal of this thesis is to expand the evaluation perspective for offshore activities to include complete life-cycles and environmental impacts besides those caused directly by drilling wastes. Site-specific considerations such as marine areas of particular concern are not considered; e.g., coral reefs and fish spawning sites. Environmental impact potentials are assessed on a systems scale rather than by focusing on the local issues relating to environmental risk and impact. Adjustments are made to accommodate local conditions where possible, but impact assessment by LCA generally does not include spatial considerations. Results presented here must therefore be interpreted accordingly.

Temporal considerations are limited to the current, average situation. Data sources representative of the current situation are therefore preferred.

2 LIFE-CYCLE ASSESSMENT

2.1 Introduction

Life-cycle assessment (LCA) is the assessment of environmental impact through the life-cycle of product systems. Cornerstone to the life-cycle approach is the understanding that environmental impacts are not restricted to localities or single processes, but rather are consequences of the life-cycle design of products and services. The product life-cycle covers all processes from extraction of raw material, via production, use, and final treatment or reuse (Wenzel et al. 1997; Guinée 2001; Baumann and Tillman 2004; ISO 2006). The combination of a quantitative approach and a holistic perspective leads to trade-offs being clearly stated in LCA. It is a systems tool well-suited for environment decision making.

Referred to by many names through its development (Baumann and Tillman 2004), LCA has in the last four decades evolved from the idea of cumulative resource requirements into a scientific field that includes emission inventory methods (Heijungs and Suh 2002) and environmental cause-consequence modeling (Udo de Haes et al. 2002). Many of the first applications, including the first Norwegian use of the life-cycle concept (Nunn 1980), were related to beverage packaging, although early reviews show a large span in the products that were assessed with life-cycle approaches (Nord 1992).

The problem of including all significant processes in life-cycle inventories is a well known in LCA (Norris 2002). Hybrid approaches have been proposed as a method to identify the largest contributing paths and to ensure that all processes are included within the system boundaries (Suh 2004; Suh et al. 2004). Hybrid approaches link process information collected in physical life-cycle inventories with monetary flows in economic models. The combination of LCA and input-output models has shown value as a complementary tool to traditional inventory methods in LCA (Heijungs and Suh 2002; Strømman 2005; Strømman et al. 2006).

Standardization of LCA methodology has been achieved step by step. The SETAC working groups (e.g., Consoli et al. 1993; Barnthouse et al. 1997; Udo de Haes et al. 2002) and other institutions have been vital in this process (e.g., Nord 1992; Nord 1995). The development of international standards has been an important driver for defining the methods of LCA. The first set of standards were published by the International Organization for Standardization in 1997 (ISO 1997), with a revised version complete in 2006 (ISO 2006). For a more thorough description of the historical development of LCA, see Ayres (1995) and Baumann and Tillman (2004).

2.1.1 General framework

The standardized framework for LCA states four consecutive stages, as illustrated in Figure 3 (ISO 2006). The stages are described in some detail here, but the reader is referred to guidelines and textbooks for a thorough introduction (e.g., Wenzel et al. 1997; Hauschild and Wenzel 1998; Guinée 2001; Heijungs and Suh 2002; Baumann and Tillman 2004; ISO 2006).

Goal and scope

The first stage of LCA consists of defining the aim and boundaries for the assessment, and the choice of methods for inventory and impact assessment.

The goal and scope stage includes defining the functional unit (FU). The functional unit is a quantitative measure of the functional requirement(s) that the product or service is designed to fulfill. It is the basis for comparison in LCA, used to evaluate the relative performance of alternative product systems.

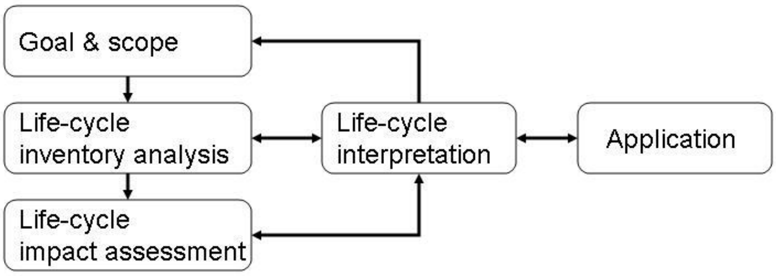

Figure 3: Outline of the stages and iterative approach of life-cycle assessment. (Redrawn from ISO 2006)

Examples of FUs are *15 years of person transport* for transportation systems, *100 m^2·years* for paints and other surface protectors, and *1 GJ at consumer* for energy supply and distribution systems.

Life-cycle assessment may be applied for various purposes, such as product benchmarking, product declaration, process development and policy support. Study designs set important limitations on the applicability of the study to provide answers. An important issue in this respect is the functional unit. Other issues include the level of inventory completeness, temporal and spatial considerations, and impact and inventory assessment approaches.

Limitations in scope may be caused by resource constraints. Spatial and temporal limitations may be applied to suit policy perspectives. Similarly, a study may be undertaken to investigate a few issues of concern, such as energy efficiency rates or CO_2-equivalents, or it may aim at a broad impact assessment. While limitation of the scope is a necessary step towards completing any study, it is vital that the principle of reproducibility is maintained; i.e., that the eventual limitations do not exclude information that may alter the conclusions.

Life-cycle inventory analysis (LCI)

The second stage consists of establishing an inventory that describes the environmental interventions that arise from the product system. Environmental interventions are inputs of resources from the environment to the product system (i.e., energy and material resources), and outputs to the environmental of adverse effect that the product system produces (i.e., emissions). The inventory is balanced to the functional unit.

Life-cycle impact assessment (LCIA)

Once the inventory of environmental interventions is established, the interventions are translated to environmental impact indicators in the third stage of LCA.

The ultimate purpose of LCA is to provide indication of environmental impact potential. Quantitative scores are achieved by application of characterization factors that describe the relative potential of each intervention to adversely affect safeguard objects through defined impact mechanisms. An example is CO_2-equivalents which are used to aggregate the global warming potential of various emissions to air. Each substance is characterized by its potential relative to the global warming potential of CO_2.

The life-cycle impact assessment stage is divided into three consecutive steps. First, environmental interventions are separated according to their cause-and-effect chains, termed impact chains or impact categories in LCA. Interventions may be input-related; i.e., energy and material extracted from the environment, or they may be output-related; i.e., emissions made to the environment. Second, impact scores are aggregated for each impact category by multiplying inventory mass flows with their respective characterization factors and summarizing for each of the impact chains. The last step of life-cycle impact assessment is the weighting of impact scores relative to each other.

Weighting requires relative comparison of different environmental issues; such as comparison of acidifying air-emissions with consumption of material resources. An inherently subjective process, and a voluntary step in life-cycle impact assessment, weighting is not often applied in the scientific literature.

Weighting methods and the selection of impact categories to be considered in an LCA depend on the stakeholders to the study. Identification of stakeholder attributes, and the matching of these with the results produced by the study, is vital to ensure the relevance of any LCA.

Life-cycle interpretation
The final stage of LCA is the interpretation of results. Vital in the interpretation stage is the consideration of uncertainty. Other aspects include the effect and validity of the selected impact assessment methods to fulfill the stated purpose of the study, and the potential bias introduced by inventory sources and approach. The re-visitation of methodological choices validates the outcome of LCA and increases the relevance of LCA for decision support.

Reiteration of goal and scope, inventory and impact assessment stages is an important feature of LCA, as outlined in Figure 3.

2.2 Life-cycle impact assessment

Attributes for decisions analysis by LCA are the environmental impact category indicators used in life-cycle impact assessment (Hertwich and Hammitt 2001b). Category indicators are quantitative scores for the relative potential to cause adverse effect through a predefined impact chain. Indicators are made for each impact chain on the basis of a model that relates stressor (i.e., the intervention) to environmental consequence.

Attributes may be defined at various levels of the cause-consequence chain. If defined at the level of value lost, they generally are referred to as endpoint indicators. Attributes defined at intermediate levels in the cause-consequence chain are midpoint indicators in LCA (Hertwich and Hammitt 2001b; Udo de Haes and Lindeijer 2002).

Several cause-consequence models have been developed within the LCA framework, covering a wide set of impact mechanisms (Guinée 2001; Udo de Haes et al. 2002). Table 1 lists a few impact chains for which characterization factors have been developed, divided by their area of protection (Udo de Haes et al. 1999; Guinée 2001). Impact chains frequently relate to more than one area-of-protection due to the inter-related nature of environmental effects, better described as impact webs (see, e.g., Udo de Haes et al. 1999; Hertwich and Hammitt 2001a).

Models of various resolution and complexity have been used in life-cycle impact assessment. For the example of toxic impacts, impact assessment models may be the application of simplistic assumptions regarding environmental residence times and toxicity thresholds (e.g., Hauschild and Wenzel 1998), or more complex representations of model (like the human toxicity potential, Hertwich et al. 2001). Continuing with the example of toxic impacts, the impact assessment framework characterizes the relative ecotoxicity of a substance as follows

Equation 1: $$S_i^{m,n} = M_i^n F_i^{m,n} E_i^m$$

where S is the impact score for the ecotoxicity of substance i to environmental (recipient) entity m through impact chain n (i.e., exposure pathway or mechanism). Factors to the right side of the equation are M: the amount of intervention (mass loading for ecotoxicity), F: the exposure that results from a unit of intervention (fate factor describing the relative distribution to impact chain n for ecotoxicity), E: dose-response function (ecotoxic effect factor for impact chain n for ecotoxicity). The cause-and-effect chain for each final impact chain m thereby consists of the following steps for a midpoint indicator for ecotoxicity

Equation 2: $$\left\{ intervention_m \underset{Exposure\ model}{\rightarrow} \Delta exposure_{m,n} \underset{Dose-response\ model}{\rightarrow} \Delta stress_{m,n} \right\}_i$$

Table 1: Impact categories in LCA organized by areas-of-protection. The list is not exhaustive.

Area of protection - societal value(s)	Impact categories (chains/pathways/midpoints)
Natural environment - intrinsic value (ecosystems, species) - life support functions	Depletion of biotic resources Impacts of land use Climate change Ecotoxicity Acidification …
Natural resources - economic and intrinsic values - life support functions	Depletion of abiotic resources Depletion of biotic resources
Human health - intrinsic value of human life, economic value	Human toxicity Stratospheric ozone depletion Climate change Noise Accidents …
Man-made environment - cultural, economic and intrinsic values	Loss of materials Loss of catch , crops

Toxicity potentials have been derived for various environmental recipients covering aquatic, sediment and soil compartments and the human population (Hauschild and Pennington 2002; Krewitt et al. 2002). The framework outlined in Equation 1 offers midpoint indicators, indicative of the stress induced upon environmental recipients as a result of an environmental intervention. Stress may be translated to damage by use of damage models, thereby continuing the cause-consequence chain from intervention to final endpoint damage.

A common damage indicator for human health in life-cycle assessment is disability adjusted life years (DALY). Originally developed for health economics (Murray and Lopez 1996), DALY is used as endpoint indicator to make commensurable effects from a diverse set of cause-consequence chains including ionizing radiation (Frischknecht et al. 2000), toxic exposure including effects on the respiratory system and by carcinogenic and noncarcinogenic toxicity (Hofstetter 1998; Pennington et al. 2002; Crettaz et al. 2003; Huijbregts et al. 2005), road noise (Müller-Wenk 2004) and occupational health damage (Hofstetter and Norris 2003).

Endpoint metrics are useful for interpretation of life-cycle inventories as they provide a common scale that encompasses several cause-consequence chains. Reducing the number of categories in impact assessment, endpoint metrics lead to easier identification and comparison of trade-offs. Secondly, endpoint indicators may be better representatives for the decision objectives (Hertwich and Hammitt 2001a). Returning to the example of human toxicity, midpoint indicators for human toxicity are extracted from exposure limit values, derived from laboratory test programs or epidemiological surveys (Hofstetter 1998; Huijbregts et al. 2000; Hertwich et al. 2001). Implemented in LCA they are indicative of the relative potential to cause human toxic effects, but they do not quantify the absolute damage caused by emissions. The DALY framework allows quantification of health burdens in life quality years, a scale to which most people may relate, thereby making results from LCA more understandable (Hertwich and Hammitt 2001a). Such absolute indicators may be important if environmental benefits are compared to other attributes of the system (Hertwich and Hammitt 2001b).

While indicators related to damage may better communicate the scale of impacts, the damage assessment also adds an additional layer to the impact assessment model. The additional modeling of the cause-consequence chain introduces new sources of uncertainty which may blur the comparison of product systems. Product systems that are discernable on midpoint level of impacts may become indiscernible if impacts are quantified in terms of damage (Lenzen 2005).

2.3 Life-cycle assessment as industrial ecology

An often quoted definition of industrial ecology states that it is *"the study of flows of materials and energy in industrial and consumer activities, of the effect of these flows to the environment, and of the influences of economic, political, regulatory, and social factors on the flow, use, and transformation of resources"* (White 1994).

White's definition of industrial ecology carries three aspects: *flows*, the *effect* of the flows to the environment, and societal factors that *affect* such flows. Although not overlapping on all the issues, life-cycle assessment is a tool well defined within the industrial ecology tool box. Life-cycle assessment covers flows between the economical and environmental systems as environmental interventions in the life-cycle inventory (Udo de Haes and Lindeijer 2002), and the life-cycle perspective ensures that inter-industry flows as well as environmental interventions are included within the assessment perspective. Life-cycle assessment therefore produces a comprehensive inventory of the environmental interventions that occur from a product system. Ayres (1995) points out that life-cycle inventories are not comprehensive from a principle of mass conservation and that this practice may lead to results that overlook important impacts. Nonetheless, life-cycle inventories should be comprehensive from the perspective of environmental effects.

Impact assessment is the translation of flows to environmental impact indicators. With some exceptions, notably acidification, eutrophication and certain substances with respiratory effects; see summary in (Potting et al. 2002), life-cycle impact assessment generally does not incorporate the element of thresholds and spatial variation. Impacts are proportional functions of environmental interventions independent of emission pattern and temporal and spatial considerations. The focus lies on the investigation of flows themselves rather than the assessment of effects that flows may cause. The main reason is the wide assessment perspective of LCA, as emissions are aggregated across temporal and spatial scales.

The aspect of change is not strongly emphasized in LCA, although recently several studies have assessed net effects that occur from choices made in system design and development (see, e.g., Jungbluth et al. 2004; Fehrenbach 2005; Ekvall and Andræ 2006; Eriksson et al. 2007; Sandén and Karlström 2007). Such studies are referred to as consequential LCA or change-oriented LCA (Ekvall 2002; Curran et al. 2005; Sandén and Karlström 2007). The traditional, attributional LCA describes the environmental performance of product systems as attributes of the product system design, relying on the use of average data for materials and energy. Marginal data becomes more relevant if change of system designs is assessed with LCA. Marginal situations are functions of the time perspective, market flexibility and trends, and the level of market influence (Ekvall and Weidema 2004). Examples of consequences playing a role in LCA are if former waste fractions become resources, thereby replacing parts of an existing resource system, or changes in energy systems which may have system-wide effects. In the first example, waste oils may be regenerated to replace virgin oils. If the composition of the virgin oil is expected to change over time, assessments should include the effect that such changes have on the performance of the original virgin system that is replaced (see, e.g., Fehrenbach 2005). A second aspect of change-oriented LCA is that boundaries may need to be expanded to include several functions (Ekvall and Weidema 2004). Waste oils may be used as an energy source or it may be regenerated. By selecting one of the life-cycle alternatives, the consequence is that the function not provided by waste oil is replaced by either energy or virgin oil given a system of constant demand.

While factors affecting environmental interventions may be discussed in LCA, the implementation of external factors is not part of the traditional approach. Changes in regulations, trends and policy are generally considered outside of the scope of LCA. Used to support inventory generation, external factor analysis increases the relevance of LCA as policy support, but it is a complementary approach for sensitivity analysis rather than an intrinsic part of LCA.

2.4 Life-cycle assessment as systems analysis

Findeisen and Quade (1985) divide decision making into the following three main elements. First are the alternatives under consideration. In the context of this thesis, alternatives are the options for consideration by comparative LCA. Second are objectives, attributes and criteria, linked together as follows: Objectives are the desires of the decision maker, uttered or implied. The objectives are translated to quantitative measures as either functional requirements (i.e., constraints) which must be met, or attributes on which the performance of alternatives is measured. Criteria are the rules or standards by which the attributes are ranked relative to each other, identical to the framework of characterization factors and weighting schemes in LCA. The third element in decision making is the model that allows us to investigate performance of alternatives on the attributes that are selected. The model that we describe here is the method of life-cycle assessment.

The selection of performance measures constitutes an important part of systems analysis. Performance measure definition should be part of the early stage of projects (ISO-IEC 2002). Various terms have been proposed to separate classes of performance measures in systems engineering (Oliver et al. 1997; Stevens et al. 1998). Keeping with the terminology of Findeisen and Quade (1985), we divide performance measures into *constraints* on the system and *attributes* of the system. Constraints describe the limitations within which solutions must be found, while attributes are the measures used to rank the alternatives.

With reference to LCA, constraints include the functional unit and the industrial and societal environment in which the product system operates. Systems are not brought into being unless in agreement with the boundaries of the constraints (Sproles 2000). Economical constraints, often the constraint deciding the design, may show properties of elasticity. In common systems engineering approaches, economical performance therefore forms part of the attributes of a system. In environmental assessments, however, economical issues are considered constraints on the system design. Optimizing on economy may produce non-dominant solutions for the environmental attributes. Physical and technical constraints affect the viability of system installment. Physical space limitations are very important for rig technology given the limitations in floor area. Technical constraints include system reliability, availability and possible risk aspects. Regulatory constraints include standards, policy and acts of law, all of which must be met for any technology used offshore and elsewhere. Other constraints are, e.g., environmental image and company policy.

Attribute measures are consequences of the physical design of systems. For the case of LCA, attributes constitute a set of environmental impact indicators. In order to be useful for decision support, the results provided by LCA must match with the objectives posed by stakeholders, and be representative of objective performance. They must also carry an aspect of measurability (Keeney 1992; Hertwich and Hammitt 2001a).

It is important to consider problem shifts when implementing environmental policy (Wrisberg et al. 2002). Life-cycle assessment includes processes from cradle-to-grave and covers a potentially large number of environmental impact chains. It is therefore well-suited to identify problem shifts between life-cycle stages, recipients, effects and temporal locations. However, life-cycle assessment is inherently function-oriented, not region-oriented (Olsen et al. 2001; Wrisberg et al. 2002). Shifts due to variation in environmental sensitivity may therefore go undetected because only generic environments are considered.

3 OIL AND GAS DRILLING OPERATIONS

3.1 Rotary drilling and drilling fluids

Oil and gas drilling operations are performed using rotary drilling methods in which a drill string equipped with a rotating drill bit grinds the rock phase while a drilling fluid is injected down the well through the drill string. The drilling fluid returns through the well annulus carrying the rock phase that is drilled out of the well. The rock material carved from the formations is referred to as rock cuttings. A schematic illustration of the rotary drilling system and drilling fluid cycle is given in Figure 4.

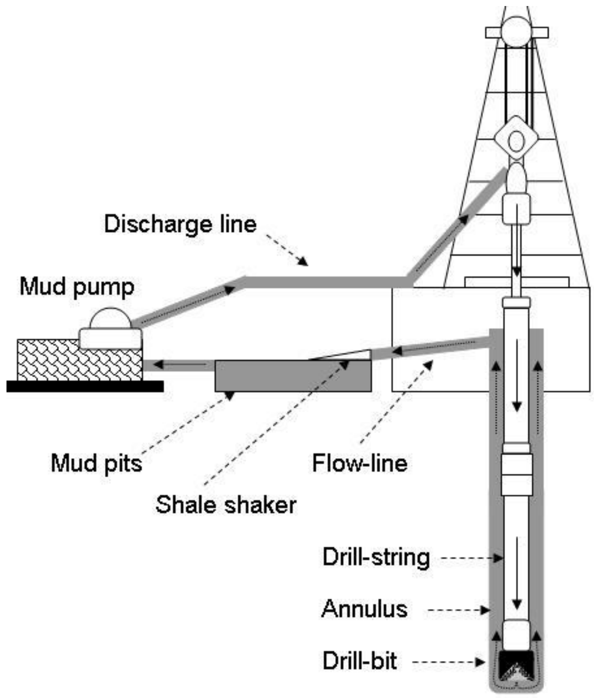

Figure 4: Drilling fluid cycle (Redrawn from Growcock and Harvey 2005).

Drilling fluid is often termed drilling mud for historical reasons. (The first fluids consisted mainly of plain mud.) Drilling operations require the use of many types of fluids for well drilling, completion and cementing. In order to avoid any confusion we specify that in the remains of the text, the term drilling fluid is used to describe chemicals used in the drilling operation to transport cuttings out of the well. The fluid serves several purposes besides supplying a transport phase, most notably it cools and lubricates the drill bit, stabilizes the well walls and maintains down-hole pressure (Bourgoyne Jr et al. 1991; Growcock and Harvey 2005). The latter of these tasks is important in order to avoid blow-out of the well. Pressure is achieved by controlling fluid density, balancing it with the pressure that is experienced down-hole.

Separation techniques are used to remove cuttings from the drilling fluid before the fluid may be re-employed. The fluid is continuously tuned by addition of components according to loss of fluid properties down-hole and changes in ambient well conditions. The down-hole injection, resurfacing of fluid and cuttings, and solids removal processes form the fluid cycle in which fluid is reused in an open loop. Loss to well formations down-hole and as residuals on cuttings is compensated by addition of new fluid. The principal components of the drilling fluid cycle are contaminant-removal equipment, mud pits, mud-mixing equipment (hoppers), and mud pumps (Bourgoyne Jr et al. 1991); see Figure 4. Common equipment used to control fluid contamination includes shale shakers, de-gassers, centrifuges and hydrocyclones (Montgomery 1996; ASME Shale Shaker Committee 2005).

3.2 Stages in the drilling operation

The initiation stage of drilling is termed spudding. Drilling in the spudding stage is usually done using a fluid consisting of sea-water or a seawater-bentonite mud (Montgomery 1996). Spud sections are drilled without return of drilling fluid to the rig deck and cuttings are deposited around the well site. After a drilled depth of typically a few hundred meters, a steel casing is inserted into the well hole and cemented to formations. With the casing locked to the well, a riser system may be installed. The riser is a pipe-connection between the well and rig allowing drilling fluid to be returned onto the rig deck for separation of fluid and solids and reuse of fluid. From the drill-bit down-hole, drilling fluid now travels first to the sea level through the annulus between the drill string and casing, and from sea level to the rig floor through the riser conduct.

The consecutive steps of drilling, inserting casings and cementing are repeated through the depth of the well. Each new casings section is installed by insertion through the previous sections and hinged onto the lowermost casing of the pre-existing section. The well diameter therefore decreases as the sections get deeper.

Changes in formation properties with depth, as well as ambient temperatures and pressure, lead to shifts in the technical specifications for the drilling fluid. The fluid is therefore constantly rebalanced within the drilling of each section. Shift of fluid type may be performed at the completion of sections. Mass balances for drilling fluids therefore are made for each section rather than each well.

3.3 Drilling fluid components

3.3.1 Base fluid systems

Drilling fluids may be separated to classes depending on the continuous phase that is used. Three main classes are identified: gaseous, water-based and oil-based. Gaseous systems are seldom used in offshore operations. Our discussion is therefore limited to water-based (WB) or oil-based (OB) fluids. In general, the fluids consist of a base-fluid phase with clays, minerals and additives in suspension. Water-based fluids have a saline water-solution as base, while oil-based fluids have a hydrocarbon base. The hydrocarbon fluid can be classified according to aromatic content and origin. Early oil-based fluids used diesel or paraffin as base, while most existing drilling fluids have a base of either non-aromatic base of mixed and linear paraffins, or a synthetic hydrocarbon base of ethers, esters or olefins. The latter class of oil-based fluids is often referred to as synthetic fluids. Synthetic drilling fluids were developed to meet stricter requirements regarding occupational exposure limits and environmental persistency. Principal components and characteristics of various generic offshore fluid systems are listed in Table 2.

The toxicity range of drilling fluid systems and system components to various species groups varies greatly. Marine species LC_{50} values are reported within the range $10\text{-}10^5$ mg/kg for various fluid systems[1]. Water-based fluids are usually less ecotoxic than oil-based fluid types (Patin 1999).

Oil-based fluids are in many situations preferred for their technical performance. The disadvantages of oil-based fluid systems are higher purchase cost and stricter requirements for treatment of drilling wastes. The main technical advantages of oil-based fluids are (Bourgoyne Jr et al. 1991; Lindland 2006):

- Can be used in water-sensitive formations, such as shale and clay
- Offer better lubrication (which increases the rate-of-penetration)
- Prevent bit balling in clay (i.e., avoid cuttings collecting between the bit and the true hole bottom)

[1] LC_{50} is the concentration lethal to 50 percent of the population of the tested species.

- Maintain test samples in core drilling
- Perform better in high-pressure, high-temperature (HPHT) reservoirs

Table 2: Generic drilling fluid systems (from Growcock and Harvey 2005)

Fluid type	Principal components	Area of use
Water-based fluids		
Simple sea-water	Seawater	Surface hole (spudding)
Spud fluid	Bentonite, water	Surface hole (spudding)
Saltwater	Seawater, brine or saturated saltwater; saltwater clay, starch, cellulosic polymer	Salt formations
Lime or gypsum	Fresh or brackish water; bentonite, lime or gypsum, lignosulfate	Shale drilling, high temperature, salt tolerant
Lignite or lignosulfate	Fresh or brackish water; bentonite, caustic soda, lignite or lignosulfate	Shale drilling, high temperature, salt tolerant
Potassium	Potassium chloride; acrylic, bio or cellulosic polymer, some bentonite	Hole stability, low tolerance to solids, high pH
Low solids	Fresh to high saltwater; polymer, some bentonite	Hole stability, low tolerance to solids and divalent salts
Oil-based fluids		
Oil	Weathered (oxidized) crude oil; asphaltic crude, soap, water 2-5%	Moderate to low pressure wells, strong environmental restrictions
Asphaltic	Diesel oil; asphalt, emulsifiers, water 2-5%	High temperature wells (<315 ºC), strong environmental restrictions
Invert emulsion	Diesel, mineral or low-/nonaromatic mineral oil; emuslifiers, organophilic clay, modified resins, and soaps, 5-40% brine	High temperature wells (<230 ºC), environmental restrictions
Synthetic	Synthetic hydrocarbons or esthers; other products same as invert emulsion	High temperature wells (<230 ºC)

3.3.2 Additives

The most commonly used additives in drilling fluids include (Bourgoyne Jr et al. 1991; Patin 1999; Growcock and Harvey 2005; Ukeles and Grinbaum 2005):

- *Viscosity control:* bentonite, organic polymers (starch, guar and xanthan gum, cellulose, lignosulfate, lignite), phosphates
- *Alkalinity and pH control:* sodium hydroxide (caustic soda), sodium carbonate (soda ash), sodium bicarbonate (baking soda), potassium hydroxide, magnesium oxide, calcium hydroxide (lime), calcium sulfate, acetic acid, citric acid, oxalic acid
- *Contaminant removal:* chrome or ferrochrome lignosulfate (for deflocculation), phosphate (for removal of calcium)
- *Lubrication:* glycols (in WB fluids), glass or polystyrene beads, graphite, oils
- *Shale stabilization (well stability):* various salts, including sodium chloride, calcium chloride, potassium chloride, potassium hydroxide, potassium carbonate
- *Density:* bentonite (in unweighted fluids), barite, ilmenite, hematite, magnetite, siderite, dolomite, calcite (limestone), manganese tetraoxide, salts (sodium chloride, sodium bicarbonate)

3.3.3 *Density agent*

The density of drilling fluids can be controlled by the use of soluble salts, or by adding finely ground mineral phases. Several minerals are used for this purpose, two of which are selected for this particular study: barite and ilmenite. Barite has been the dominant weighting agent in drilling operations world-wide and still is used in most operations. Ilmenite has increasingly replaced barite in Norwegian waters due to its lower trace metal content and the benign nature of its mineral matrix.

Most metal releases to the marine environment from drilling operations originate from the weight agents. Other sources for trace metals in drilling wastes are contaminations in the base fluids and the additive contents, particularly clay minerals and lignosulfates (for chrome).

Barite; (barite), barium sulfate, $BaSO_4$

Barite was introduced as a weight agent in the 1930s. Today, drilling operations are the main applications of barite. Barite is favored for its relative abundance and market availability. About 6 million tonnes is produced and traded globally every year. Almost half of this is sourced from China, 12% from India and 7-8% from USA. Smaller producers include Turkey, Morocco and Iran (Newcaster et al. 2007). Prices have increased markedly in the last years due to lack of barite reserves suitable for drilling applications (Tran 2007).

Main impurities in barite are silica, iron oxide, and carbonates (i.e. limestone and dolomite). Trace metals occur mostly in the form of sulfides (Neff 2005).

Ilmenite; iron titanium/magnesium oxide, (FeO, MgO)TiO_2

Ilmenite is mostly known as a raw material for titanium dioxide (titania, TiO_2). Since 1920, titanium dioxide has been used as a white-color pigment in products including food, make-up, sunscreen and paint (Reck and Richards 1999). Pigment production is by far the main use of ilmenite. Norway holds a large part of the global ilmenite resources, with the largest reserve located in Jøssingfjorden (Titania AS, Sokndal municipality).

Since its introduction in 1979, drilling grade ilmenite used in Norwegian waters is sourced exclusively from the Jøssingfjorden open pit mine (Fjogstad et al. 2002). No processing is required for the production of ilmenite besides crushing and separation. Ilmenite used in drilling fluid is slightly finer ground than the normal ilmenite made by Titania. This is achieved by a simple crushing jet-stream.

Phases of iron and titanium oxides are typical impurities in ilmenite. Trace metals are found both within the main crystal lattice, associated structures and in sulfide phases (Myran 2003). Ilmenite has under certain conditions been restricted from use as magnetic properties of mineral impurities disrupt logging systems down-hole. These problems have been remedied by improved producer practices.

3.4 Regulations

Wide use of water-based fluids was the consequence of a ban on release of cuttings with >1 wt-% of hydrocarbons in Norwegian waters. This effectively was a ban of the discharge of cuttings drilled with oil-based fluids. Most countries, including Norway, allow marine discharges of cuttings containing residues of water-based fluid under the requirement that additives meet limits for toxicity and environmental persistency, although a minimization of marine disposal is requested (OGP. 2003). The most used additive classification system is the PLONOR list published by the OSPAR Convention (OSPAR. 1992). PLONOR substances are substances considered to Pose Little Or NO Risk to the environment based on substance or product characteristics in terms of marine persistency, bioaccumulation potential, acute toxicity and the possibility of endocrine effects.

Greater variation is seen in regional regulations for the discharge of oil-based fluids and cuttings; e.g., OSPAR protocols restrict discharge of oil-based fluids and require a maximum content of oil components of 1 wt-% in cuttings, while discharge of synthetic hydrocarbon drilling fluids is subject to permission in many jurisdictions (OGP. 2003).

The Norwegian sector of the Barents Sea was recently re-opened for exploration and production drilling under the general requirement that cuttings drilled with water-based fluid not be discharged offshore after installation of the riser. Combined with a strong requirement for substitution towards use of environmentally benign drilling fluids in operations in the Barents Sea, this gave a novel situation in which use of water-based fluids is recommended by regulators and at the same time such cuttings must be transported to shore. Injection to sub-sea formations is not an option in this area due to lack of dedicated wells and the risk increase associated with injection to the same well.

3.5 Material flows for drilling fluid

Losses during fluid cycling depend on fluid type, formation properties and solids control performance. In a recent study of loss rates, undertaken by Lindland (2006), average loss per section as residue on cuttings was found to be 30% for water-based and 15% for oil-based fluid systems. The loss to well formations varied greatly between wells, presumably due to formation properties. A general loss rate down-hole of about 5% was indicated for both oil-based and water-based fluids. The fraction of fluid lost as residues on cuttings may be reduced by good solids control performance, while the fraction lost to formations is a direct result of fluid and well characteristics. Given that fluid characteristics are defined by the technical requirements, less can be achieved in terms of the fraction lost to formations down-hole.

Percentage distribution of fluid sources and loss recipients are illustrated in Figure 5. Volumes of fluid and cuttings for a typical well and the entire Norwegian sector are reported later.

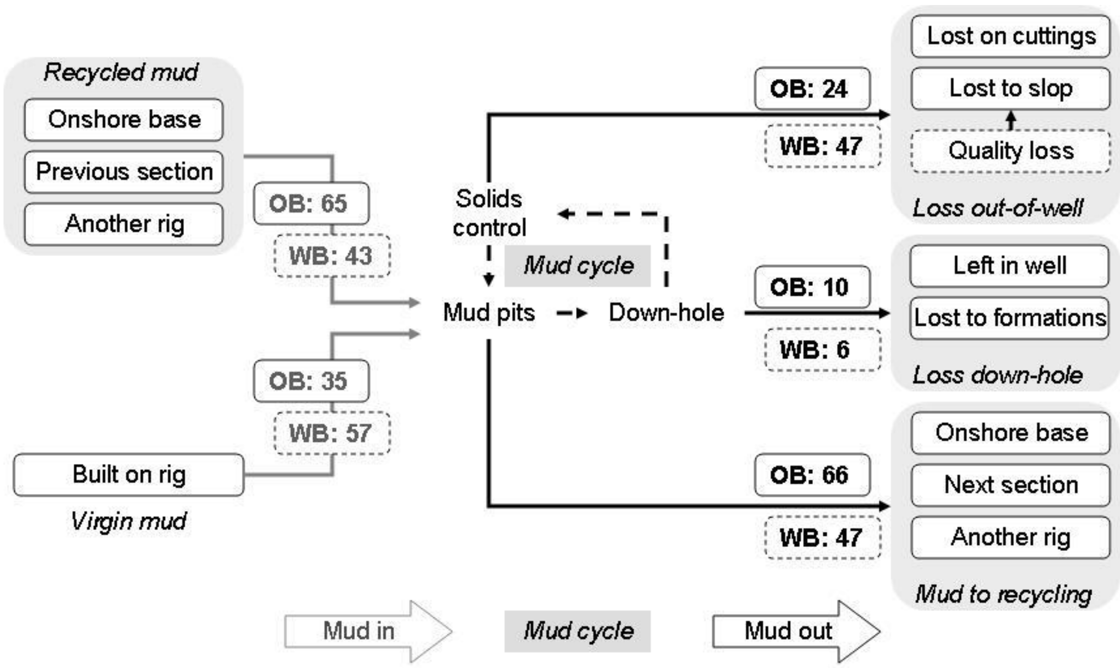

Figure 5: Percentage distributions of source and end-of-life fate for water-based (WB) and oil-based (OB) fluids. Numbers are aggregated on mass for sections drilled in the period 1999-2005. Slop denotes contaminated water, usually including collected rain water, washing water and liquid chemical residues. Source: (Lindland 2006)

According to the percentage distribution in Figure 5 of losses in-well and out-of-well compared to the input of virgin fluid there apparently is an increasing demand for fluid. This result is an artifact of how fluid end-of-life fate is recorded on rig. Remaining fluid in mud pits after completion of each section is generally transferred to the fluid supplier. This is recorded as *to recycling* independent if the supplier later decides that it is unfit for further use.

Drilling fluid remaining after completion of well sections is transferred to a new section in the same or a second well. If loss of fluid quality is substantial, the fluid may be considered unsuited for regeneration by rebalancing of fluid properties. In such cases both remaining fluid and the cuttings produced need to be treated. Due to the commercial value of mineral oil, extraction of oil components from oil-based fluids and cuttings with oil-based fluid residues is an economical option. The oil fraction is separated and used for drilling or other purposes by thermal or thermo-mechanical (i.e., hammer mill) separation. Regeneration technologies for water-based fluids have yet to be commercialized on larger scale. Options for water-based waste include re-injection to sub-sea formations offshore or land-farming or landfill treatment onshore. High salinity in water-based fluids is the main cause of concern in onshore treatment of water-based waste. Re-injection can be performed for cuttings and fluids of both water-based and oil-based origin, in which case the cuttings volume is slurrified and injected into the annulus of a well being drilled or a dedicated injection well.

Including losses in well and in operations out of the well (i.e., in solids control, accidental losses, losses during transportation, losses to waste waters, losses due to quality degradation, etc), average monthly fluid recycling rate per section drilled were about 45 and 68 % for water-based and oil-based fluids respectively in the period 1999-2005 in operations undertaken at Statoil ASA. The variation in monthly rates was significant in this period, particularly for water-based fluids; see Figure 6. Note that some deviation is seen between empirical values for recycling rates based on mass (Figure 5) and frequency (Figure 6).

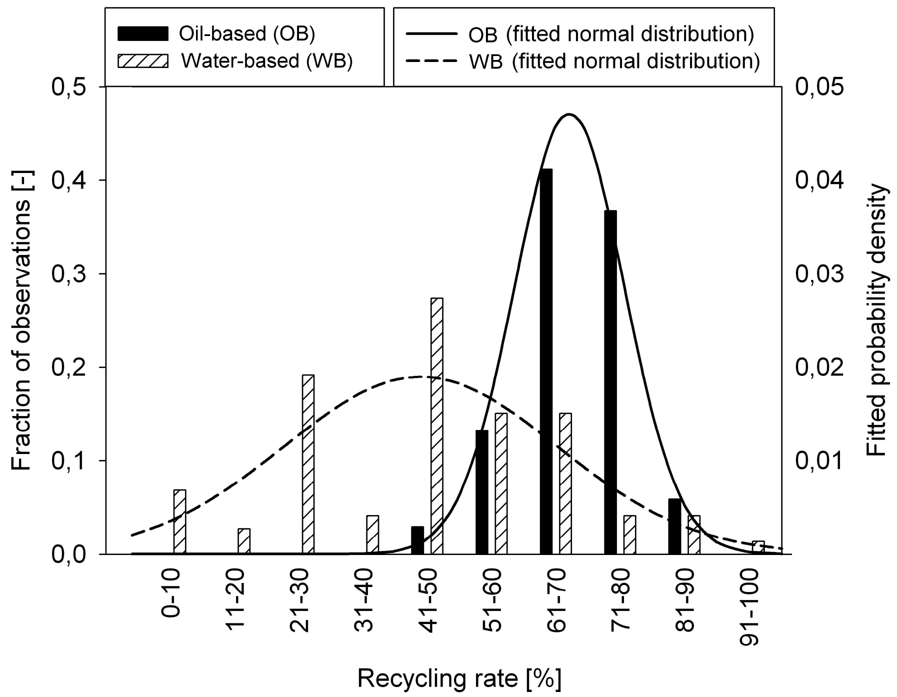

Figure 6: Variation in monthly average recycling of water-based (WB) and oil-based (OB) fluids for well sections in the period 1999-2005. Based on volumes used offshore (Lindland 2006). Parameters for the fitted distributions are OB: mean 68, standard deviation 8.4; WB: mean 45, standard deviation 21.0, all numbers as percentages.

3.6 Wastes and emissions from drilling operations

National numbers for fluid use and end-of-life for fluid and cuttings in Norwegian waters is reported by the Norwegian Oil Industry Association. Numbers for the last three years are summarized in Figure 7. The split of use of water-based and oil-based fluid is about even in all three years. The main end-of-life treatment for water-based fluid is discharge, but about one fifth of the volume is treated by other methods. It is important to note that spud drilling fluid is registered as water-based fluid in the statistics and this contributes a large part of the water-based fluid discharged at site.

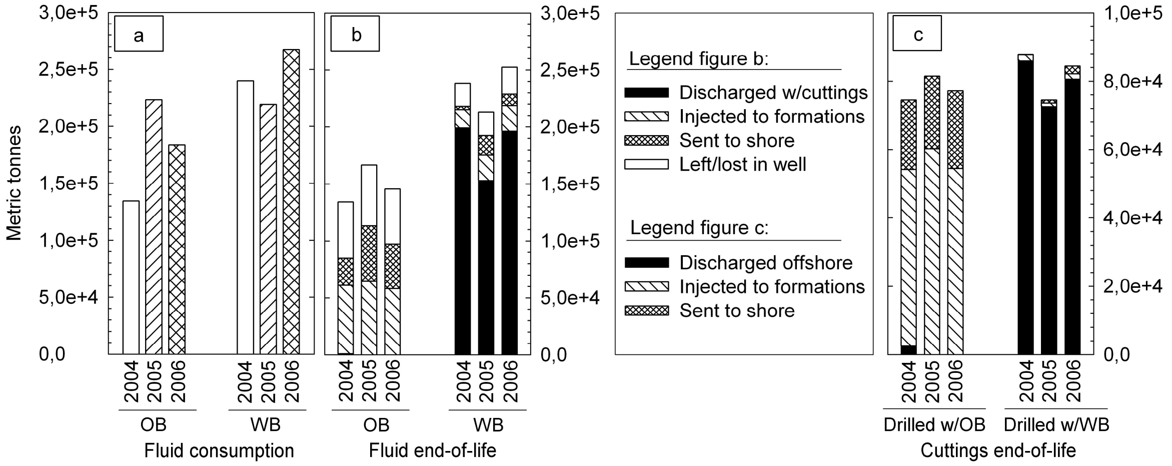

Figure 7: Norwegian situation for a) use of fluid, b) end-of-life for fluid, and c) end-of-life for cuttings (based on OLF 2007). Cuttings transported between installations is included as cuttings re-injection in figure c. OB = oil-based fluids (oil- and synthetic-based), WB = water-based fluids.

The fraction of fluid lost or disposed in well is significant for both fluid systems, but particularly for oil-based fluids. The distribution of end-of-life for oil-based fluids is more or less even between the volumes left/lost in well, treated onshore and injected to offshore sub-sea formations. The pattern of treatment of cuttings waste is largely different for the two fluid systems. Cuttings drilled with water-based fluid are almost exclusively disposed off by offshore discharge. This is not a permitted end-of-life route for cuttings drilled with oil-based fluid. The cuttings have some economic value, as the oily fluid residues may be separated from the solids by thermal or thermo-mechanical treatment. This value is weighted against the costs of transportation. The numbers show that around two thirds of oily cuttings are injected to formations, indicating that reinjection wells generally are available locally, or by little additional logistics.

Based on the industry data presented in Figure 7, there apparently is an accumulation of drilling fluid in the industry. This may be explained by increasing volumes of fluid stored by fluid suppliers as a result of take-back of used fluids for subsequent use. The size of the discrepancy between use and disposal, however, indicates that there may be losses that are not accounted for in the statistics. This discrepancy is particularly pronounced for oil-based fluids.

The available statistics do not separate between the different types of offshore operations. Drilling wastes and emissions are reported for the oil and gas extraction and production sector as a whole. We therefore rely on numbers submitted in drilling permit applications to estimate mass flows for single wells. Material requirements for an example well, based on a well in the Barents Sea, are summarized in Table 3.

It is reported that about 200,000 – 250,000 tonnes of oil-based fluid and cuttings waste is produced annually in Norwegian sectors offshore (OLF 2007). Offshore waste production is not fully covered by national statistics as water-based wastes may be disposed off at site, and the industry data clearly has weaknesses with recording all flows of fluid and cuttings. If waste production with water-based fluids is similar to that of oil-based fluids, this effectively doubles the fluid and cuttings waste production to >500 000 tonnes of waste every year, which is equivalent to about half the mass of construction and demolition (C&D) waste and equivalent of 6 % of all waste production reported for Norway in 2005 (SSB 2006).

Table 3: Estimate of mass flows for an exploratory well in the Barents Sea

Well characteristics:	2 966 m length, with successive section lengths [a, b]
	52 m 36" section (spud section, with 30" casing inserted)
	365 m 26" section (spud section, with 20" casing inserted)
	505 m 17 1/2" section (with 13 3/8" casing inserted)
	992 m 12 1/4" section (with 9 5/8" casing inserted)
	1052 m 8 1/2" section (with 7" casing inserted)
Fuel:	1 920 tonnes diesel[c], resulting in the release of
	6 120.0 tonnes CO_2
	134.4 tonnes NO_x
	9.6 tonnes nmVOC
Steel:	232 tonnes of steel casings cemented to well hole [d]
Chemicals :	2 007 tonnes of various drilling chemicals, including [a]
	1 028 tonnes ilmenite
	322 tonnes cement
	90 tonnes bentonite
	5.45 tonnes washing chemicals
	0.41 tonnes grease (as drill-string dope)
Wastes:	
	530 tonnes cuttings (dry rock) deposited at site [e]
	348 tonnes chemical products deposited at site (as spud mud), including
	• 244.6 tonnes ilmenite
	• 90.9 tonnes bentonite
	• 11.2 tonnes carboxymethyl cellulose
	• 1.7 tonnes soda ash
	1 250 tonnes wet cuttings transported to shore for treatment [f]
	400 - 4 000 m^3 contaminated water (slop) [g]

[a] Source: (Lykling Berge 2004)
[b] Inches refer to well hole diameter ; e.g., the 36" section is drilled with a drill-bit of diameter 36 inches
[c] 32 tonnes diesel per day, 60 operative days
[d] Assuming the following approximate material use in casings: 30" – 150 lbm/ft, 20" – 100 lbm/ft, 13 3/8" – 70 lbm/ft, 9 5/8" – 45 lbm/ft, 7" – 30 lbm/ft (Bourgoyne Jr et al. 1991)
[e] Assuming a hole enlargement factor of 1.3 and cuttings density of 2.6 tonnes per m^3
[f] With the following assumptions: hole enlargement factor: 1.1, cuttings density 2.6 tonnes per m^3, fluid:cuttings volume ratio of 2:1, and fluid density of 1.2 tonnes per m^3
[g] Slop production estimate is 0.3-3 m^3 per tonne drilled cuttings

3.7 Local impacts of offshore drilling

The primary local disturbance from offshore oil and gas drilling operations is the discharge of drill cuttings containing drilling fluid residues. Four environmental parameters have been identified to show strong covariance with distance from platforms (Olsgard and Gray 1995; Peterson et al. 1996):

- Sand and fine-particulate matter: causing burial by sedimentation and effects on suspension feeders
- Hydrocarbons: toxicity by aromatic hydrocarbons and complex mixtures
- Metal concentrations: causing exposures above natural background levels and toxic thresholds at some sites
- Ambient water properties: oxygen depression and enhanced nutrient availability in bottom waters

Reporting on the findings of long-term effects from drilling activity in the Gulf of Mexico, the dominant effects were found to be organic enrichment and metal toxicity (Peterson et al. 1996). Toxic effects from organic constituents seemed less significant; e.g., PAH concentrations remained below threshold values. Sedimentation effects, often localized to the platform site depending on the level of distribution by flow regimes, were generally reported to have short-lasting effects. Macroinfaunal density was increased at platform sites due to increased organic substrate availability for annelid worms. However, amphipod and copepod abundance was reported to decrease, presumably due to sub-lethal toxic responses. These latter effects are typical for crustaceans, which often show responses at modest toxic exposure.

In the North Sea, drill cuttings have shown reduced hydrocarbon and barium (Ba) concentration upon cessation of discharges, indicative of possible resuspension and redistribution of these elements (Olsgard and Gray 1995). Total hydrocarbon and metals associated with barite showed clear correlations with changes in the benthic fauna. However, biological responses were not correlated to the amounts of cuttings discharged. This may be caused by differences in the fluid types used, depths at the site, and flow regimes.

Dissolved contaminants are rapidly dispersed in the marine environment leads. Hence, aquatic exposure is of lower importance. Effects are located to the area surrounding the discharge site, and limited mainly to the benthic community (Peterson et al. 1996; Hurley and Ellis 2004), although indirect effects on the fish population and other mobile species are expected due to loss of food sources. As an example, population density of the brittle star (*Amphiura filiformi*) was significantly reduced close to platforms in the North Sea (Olsgard and Gray 1995).

An important issue for drilling operations is the proximity to sites of particular concern, such as, e.g., coral reefs and fish spawning regions. Reef communities are sensitive to sedimentation and changes in turbidity (Rogers 1990). Several reefs have been identified in Norwegian waters, and many fish species have spawning sites not far from the coast or potential drilling sites in the northern waters (Føyn et al. 2002).

Several studies have investigated the spatial and temporal distribution of the environmental effects of drilling operations. At sites in California, barium concentrations increased above background levels during drilling operations. Concentrations later decreased after completion of drilling operations, although not to concentrations prior to discharges. Suspended sediments regained background levels of Ba within 1 year after cessation. The spatial limit for barium dispersion was within 6 km from the platform (Steinhauer et al. 1994). In the Gulf of Mexico, changes in environmental parameters that affect the benthic community were highest within the nearest 200 m (Peterson et al. 1996). Multivariate analysis applied to North Sea sites, showed a much larger affected area. Discharges and subsequent redistribution gave evidence of contamination 2-6 km off the drill site 6-9 years after completion of drilling operations (Olsgard and Gray

1995). Similarly large areas are reported for hydrocarbon distributions to sediments in the North Sea (Kingston 1992), and for barite in suspended sediments offshore California (Hyland et al. 1994). In a Canadian review, the zone of contaminant detection was generally within 1000 m for single-well sites, and up to 8 km for multiple-well sites, although in some cases larger distributions were reported (Hurley and Ellis 2004). Hurley and Ellis also report that the zone of affected benthic fauna diversity and abundance was considered detectable to 250 m, rarely detectable to 500 m, and seldom observed at 1000 m off the well site for most sources. In terms of temporal distribution, most studies found that baseline conditions were achieved within 12 months after drilling was completed for the area outside the nearest 100 m.

Since the first survey in 1973, the Norwegian Pollution Control Authority has initiated regional monitoring surveys of sediments in Norwegian waters suspected affected by drilling activities. Regular benthic surveys have been conducted since 1982, and water-column measurements since 1999. Findings from the last round of surveys are reported here (based on Mannvik et al. 2005; Nøland et al. 2006; Botnen et al. 2007). Observations indicate that considerable redistribution of cuttings deposits occurs, increasing the area affected by hydrocarbon or metal contamination to 10-100 km^2 in some cases. Measurable improvement has been reported for the recent years in terms of contamination levels and area size, as well as the size of the area with affected benthic communities. This is not unexpected given the reduced drilling activity, substitution to use of less ecotoxic drilling fluids, reductions in cuttings discharges, and the installment of extensive cuttings and water cleaning technology on rigs. Current observations indicate that while contamination may be detectable at large distances from the drill sites, benthic effects are in most cases limited to the closest few hundred meters.

3.8 Prior environmental assessments

Environmental assessments may be prospective, aiming at evaluating alternatives prior to implementation, and retrospective, aiming at assessing the effects that occur as a result of alternatives that have been implemented. Monitoring programs, described in the previous section, are retrospective. The purpose of this thesis is to develop methods that allow prospective assessment of drilling technologies.

Most of the prospective assessments of drilling technology that exist in the literature tackle decision objectives as separate issues. The main focus is the direct impacts occurring at the site (see, e.g., Garland 2005). The existing literature mainly evaluates marine environmental effects and offshore safety. Models have been developed to describe the dispersion of contaminants in the marine environment taking into account substance toxicity and characteristics, emission pattern and local wind and current information (examples are presented by Rye et al. 1998; Rye et al. 2006). Simplistic models have been used for the same purpose (Sadiq et al. 2003a; Thatcher et al. 2005). Probabilistic parameters have also been implemented in marine risk assessment (Sadiq et al. 2003b; Sadiq et al. 2004). Issues beside dispersion include potential ecotoxicity and bioavailability of substances in drilling waste in an offshore or onshore context (e.g., Schaanning et al. 2002; Payne et al. 2006). Results from environmental exposure assessment are interpreted relative to regulatory or environmental thresholds.

Offshore safety aspects have been modeled with great detail in quantitative risk models, including technical and organization aspects (Øien 2001a; Øien 2001b). The most used approach, however, is the application of statistics (e.g., frequency of dropped load, Mazzola 2000). Much of the safety literature for offshore activities exists in the grey literature, or as documents for regulatory support (see, e.g., Vinnem 1999; OD 2003; HSE 2005a). The reports often evaluate whole installations, or the entire industrial sector. Some activities may be investigated separately, such as lifting operations (see, e.g., HSE 2004; Scandpower 2005). Results from safety assessments are in the format of changes in risk levels or accident frequency, related to time or unit operation.

Some attempts have been made at overall evaluation of offshore drilling fluid technologies. These are either qualitative in scope or limited in the assessment of effects (e.g., Meinhold 1999; Paulsen et al. 2005). An example is the environmental assessment of formate brine drilling fluid (METOC 2003). Proposed as an environmentally benign fluid technology, a partial life-cycle assessment was undertaken by the supplier to support the claims for the sodium/potassium formate brine system. The report reports only data related to product flows and potential ecotoxic effects. No evaluation is made regarding other emissions or processes during production and use.

The US EPA has published a study to evaluate end-of-life practices for cuttings drilled with synthetic fluids (EPA 1999). While this study includes environmental effects to other recipients beside the marine environment, it uses an unclear approach in the comparison of alternatives. Environmental attributes included in the evaluation include safety considerations, energy and water consumption, solid waste production, and the aggregated category of *air emissions*. The study would have benefited from using a formalized life-cycle approach, both in the assessment of inventories and in the translation of emissions to decision objectives that are based on environmental impact potentials.

4 DECISION OBJECTIVES AND ATTRIBUTES

4.1 Stakeholder attributes

Participants in the design process for drilling operations include the operator and a number of suppliers and contractors; see Figure 8. The system design is proposed by the operator in cooperation with suppliers. The rig contractor and the fluid supplier generally have the most influence on the final design, although important system components are provided by several other suppliers.

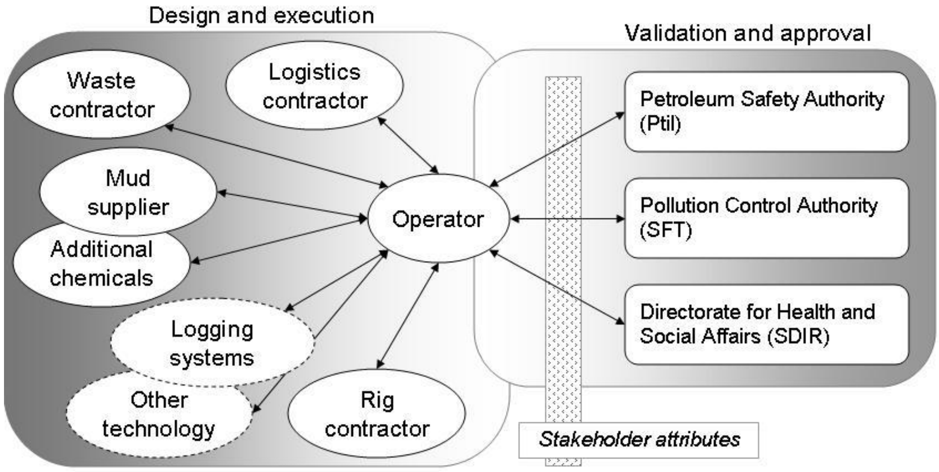

Figure 8: Stakeholders in design, execution and approval of offshore drilling operations

Regulatory authorities validate and approve system designs. Regulators outline the framework that shall be employed when performing drilling operations and in the selection of technology alternatives. The next sections outline the environmental objectives put forward in relevant regulatory documents.

4.1.1 Attributes and criteria in Norwegian law

Three acts of law are identified as important for petroleum activities in Norwegian waters.

The *Act relating to petroleum activities* (i.e., the Petroleum Act, Norw.: Petroleumsloven) demands that petroleum resources be managed for the long-term benefit of the Norwegian society as a whole, and specifies this as taking into consideration welfare, environmental effects and revenues.

A set of regulations have been designed to accompany the Petroleum Act, one of which is the *Regulations relating to health, environment and safety in the petroleum activities*, in short termed the Framework Regulation (Norw.: Rammeforskriften). The purpose of the Framework Regulation is to ensure a high level of protection for health, environment and safety. In effectuating risk reductions, the regulations require that technical, operational or organizational solutions are used that provide the best results according to an individual as well as an overall evaluation. The guideline to the Framework Regulation terms this the principle of best available technology (BAT) and refers to this principle being based in the *Act relating to pollution and waste control*. The overall evaluation is required also by the *Regulations relating to management in the petroleum activities* (the Management Regulation, Norw.: Styringsforskriften).

The *Act relating to pollution prevention and waste control* (Norw.: Forurensningsloven) is designed to reduce existing pollution and quantity of waste so that they do not result in damage to human health or adversely affect welfare or damage the productivity of the natural environment. The act lists emission reduction at source, recycling and use of best overall technology as important principles in achieving these requirements. The complementary regulation, *Regulations relating to pollution control* (Norw.: Forurensningsforskriften), describes best available technique (BAT) as the best option by an *overall evaluation*, with attributes set according to the BAT described by the European Council Integrated Pollution Prevention and Control (IPPC) Directive (European Council 1996). We shall return to this European Directive later, for now we only note that two different BATs have been mentioned: best available *technology* and best available *technique*.

Protection of the health and safety of employees is controlled by the *Act relating to worker protection and working environment etc* (Norw.: Arbeidsmiljøloven), designed with the objective to secure a working environment which affords the employees full safety against harmful physical and mental influences, concurrent with safety, health and welfare standards of society at any time.

4.1.2 Attributes and criteria in Norwegian policy

Norwegian environmental white papers cover an extended list of environmental priority issues including global warming, acidification, release of hazardous substances and waste production (MD 1997; MD 2002b; MD 2005). As stated above, some policy instruments are aimed at reducing overall impacts on such extended lists of issues, although the general approach still is limited to tackling each issue separately. However, the discussion of these attributes on national level is an indication of their priority in Norwegian policy.

The marine environment has become a separately discussed issue in Norwegian policy due to the potential conflict between offshore oil and gas activities and the commercial value of marine biological resources. A specific goal of zero environmentally hazardous discharges to the sea from petroleum activities was therefore introduced to accompany the Petroleum Act. This goal was introduced in 1997 by White Paper 58 (MD 1997) and restated in later White Papers (MD 2002a; MD 2002b). In a joint effort by representatives from regulatory authorities and offshore operators, the goal of zero-discharges was transformed into operative requirements for offshore activities (see SFT 2003). The result was a list of requirements for chemicals used offshore and naturally occurring substances. In general terms the requirements were that no discharges are allowed if the discharge may cause environmental harm. This includes discharges of hazardous substances, or if the discharged material may cause harm by non-toxic mechanisms. An example of the latter is if discharges cover important benthic ecosystems such as coral reefs. In addition to defining the requirements for zero discharge, the final document also lists substitution towards use of less hazardous/harmful activities or substances and use of best available technology (BAT) as important principles. In the definition of BAT, the document refers to Appendix 1 of the OSPAR Convention (OSPAR 1992).

4.1.3 Requirements for an overall evaluation

Several of the acts and regulations described in the previous sections use the term *overall evaluation* and best available technology or technique, both latter terms abbreviated as BAT. The acts concerning waste and pollution point to the European Commissions Integrated Pollution Prevention and Control (IPPC) Directive (European Council 1996) and states BAT as best available *technique*. The zero discharges document points to the OSPAR Commission (OSPAR 1992) and refers to BAT as best available *technology* (SFT 2003). The use of the term technology in the guidelines to the petroleum regulations lead to the conclusion that they, as the zero discharge document does, specify use of the OSPAR BAT. The definition of BAT is important as the two different sources specify different attributes for the selection of best alternative.

The IPPC Directive does not encompass offshore activities. However, it is highly relevant for the offshore industry for two reasons. First, drilling wastes that contain oily residues are classified as hazardous waste according to the European Waste Catalogue (European Council 2002). As such, facilities that are involved in the treatment of oily drilling wastes must use BAT as prescribed by the IPPC Directive. Second, future tightening of regulations concerning offshore activities is expected to be in line with the requirements of the IPPC Directive. Issues regarding resource management; including energy, materials and water, are listed by the IPPC Directive as attributes to the BAT. These are issues also addressed by other European Council Directives, such as the Waste Directive (European Council 2006). In order to achieve improvements in resource management, alternatives for drilling technology must address improvement potentials beyond the treatment of waste. Identification of best overall alternatives must allow flexibility in selection of drilling fluid components and techniques employed offshore, and should not be restricted to down-stream alternatives only. From this reasoning, implementation of a wider evaluation perspective in terms of processes and environmental attributes can be expected in the future, in line with the BAT as described by the IPPC Directive.

The IPPC Directive is the most advanced legal instrument in assisting integrated pollution control. Although it is a source based approach, the directive stresses resource issues, safety and emissions. It integrates polluting emissions to air, water and land within a cradle-to-grave perspective. The guideline for conducting BAT evaluations is published by the European IPPC Bureau. It lists the most important environmental aspects as human and aquatic toxicity, global warming, acidification, eutrophication, stratospheric ozone depletion and photochemical ozone creation (EIPPCB 2005). The list is not exclusive, meaning that other issues may be included in the evaluation if considered significant for the conclusions. An example of such an issue is occupational safety, mentioned specifically in the IPPC Directive. The completed reference documents published by the IPPC Bureau show clearly that LCA meets the structural requirements for a BAT evaluation, and the guidelines for assessment of technologies according to the IPPC Directive draw extensively on the existing LCA literature.

While the IPPC Directive focuses on installations, the OSPAR Convention has a region-based approach to BAT. The aim of the Convention is to ensure a high level of environmental quality in a defined area through control with inputs from all environmental media. It may therefore also be termed an ecosystem-based approach.

Comparing the two sources for BAT attributes, LCA matches the IPPC Directive’s requirements for a holistic perspective and in attributes. Due to the single-attribute focus of the OSPAR Convention, a solution preferred by the OSPAR Convention may not be the best alternative if evaluated according to the attributes specified by the IPPC Directive.

4.1.4 Policy and law with regards to the Barents Sea

The policy of zero discharges with environmental harm applies to the entire Norwegian Continental Shelf, but the interpretation of the zero discharges is particularly strict in the northern areas. A management plan for the northern areas, including the Norwegian Barents Sea and areas offshore Lofoten, was completed in 2006 (MD 2006). The plan states separate criteria for the practices in these areas compared to the Norwegian and North Sea.

Discharge of cuttings is generally permitted in Norwegian waters as long as the oil content is below 1 wt-% and the cuttings contain only substances on the OSPAR PLONOR list. The rule for operations in the Barents Sea area is a physical zero-discharge requirement, based on a precautionary approach. All cuttings, except from the spud sections, must be collected on rig and treated. Dedicated wells are not found in the area for injection of cuttings to formations and cuttings must therefore be transported to shore.

4.2 Necessary impact and inventory method developments

Life-cycle assessment has a strong history of assessing consumer goods. Life-cycle assessment has also proven itself as a useful tool for evaluation of larger product systems, particularly in comparison of energy systems; e.g., the Ecoinvent data-base (Frischknecht et al. 2004) and the European Commission funded Externalities of Energy project (European Commission Undated). Common to all of these applications is that in the degree that they include offshore activities, they consider the offshore activities as extensions of land-based industrial systems, i.e., they assume that such processes may be modeled as continuous processes and assessed analogous to onshore emissions. With some exceptions (e.g., Bergerson and Lave 2005), the aspect of safety is normally not evaluated in LCA. It may however, be included as part of system functionality; e.g., reliability (Winnes and Ulfvarson 2006), or in the degree that regular accidental emissions may be expected; e.g., probability of radioactive releases (European Commission Undated), and loss of water in water supply systems (Landu and Brent 2006).

The next few sections discuss gaps that must be met for LCA to be a useful method for overall evaluation of drilling technology.

4.2.1 The intermittence of offshore operations

Offshore drilling operations are often performed by mobile drilling units. Every single drilling operation is normally completed within two months before the rig is moved to a second site. Marine discharges appear as intermittent flows within the operation itself. While some emission processes may require time to complete, such as leaching from seabed sediment deposits, most emissions disperse immediately to the aquatic phase. The result is short-term plumes with large concentration gradients.

The assessment of ecotoxicity in LCA relies on the use of steady-state multi-compartment distribution models. It has been shown that steady-state models can be used to quantify the toxicity of pulse emissions relative to continuous emissions if effect is proportional to exposure (Heijungs 1995). Recently, ecotoxic effect functions are based on the use of species sensitivity distributions (SSD, see e.g., Huijbregts et al. 2000; Goedkoop and Spriensma 2001; van de Meent and Huijbregts 2005). The underlying assumption for SSDs is that the ecotoxic sensitivity of species follows a statistical distribution, in most cases the log-normal distribution. With ecotoxic effect a function of SSDs, the effect is not proportional to exposure. However, the use of steady-state models still is valid if the exposure is marginal, i.e., a first order Taylor approximation may be used.

While the assumption of marginal exposure may be assumed for largely distributed emissions, it is not valid for emissions from drilling operations. If LCA is to be used to assess drilling fluids, the significance of using steady-state based characterization factors for ecotoxic emissions must be investigated. This issue forms the first part of the methodological development.

4.2.2 Offshore occupational safety

Although accidents and the effect of unsafe working conditions seldom are assessed in LCA, they have a long tradition in the LCA literature. Workplace effects on human health is included early textbooks for life-cycle impact assessment (Nord 1992; Hauschild and Wenzel 1998), and fatal accidents are mentioned as an impact category in the recent guide to ISO LCA (Guinée 2001). Accidents were evaluated in the ExternE project (European Commission Undated). A method to include the working environment in LCA was proposed by Antonsson and Carlsson (Antonsson and Carlsson 1995), relying on statistical records of work injuries and production volumes for a company. A review of approaches to include the working environment in LCA has been presented by a separate SETAC working group (Poulsen and Jensen 2004).

Investigating occupational health impacts for industry sectors in USA, Hofstetter and Norris (2003) conclude that occupational health impacts are significant for the total human health impact of sectors with hazardous working environments. The offshore oil and gas sector belongs to this group of industries, and improving the offshore safety level is an important objective for regulators and operators offshore. For this reason, the inclusion of occupational health impacts in LCA of offshore drilling technologies would add relevance to LCA results and increase the value of LCA as a method for overall evaluation of drilling technology. Furthermore, occupational health is considered an important trade-off caused by the requirement that cuttings from wells in the Barents Sea be transported to shore. Crane-lift accidents are pointed out as the main cause of accidents with health damages from the resulting transport processes (OED 2002).

Crane-lifts are considered an important cause of unsafe working environments offshore. This is the second issue for methodological development.

4.2.3 Long-term emissions from drilling wastes

The models used in life-cycle impact assessment for quantification of the ecotoxic impact of substances have been developed from the perspective of onshore activities and protection of terrestrial and freshwater environments. The models were originally designed for risk assessment purposes and in the degree that they did include the marine environment it was for the near-shore volumes only. In the application of the exposure models from risk assessment to life-cycle impact assessment, the models have been extended and designed to work as closed systems (Huijbregts et al. 2000). This set-up is a direct consequence of a sustainability-based approach in impact assessment, that impacts are included irrespective of temporal and spatial localization.

To ensure proper evaluation of marine aquatic effects, two issues are identified as problematic when LCA is applied to offshore activities. First is the issue of intermittence and the effect that pulse-emissions have upon the ecotoxic effect of marine discharges. Second, drilling waste is a significant source of ecotoxic emissions, particularly of metals (Brügmann 2001). Recent developments indicate weaknesses in current impact assessment models for metal ecotoxicity, related to metal non-degradable nature, speciation, uptake availability, and potential essentiality (Paquin et al. 2003; Heijungs and Koening 2004; Wegener Sleeswijk 2005). In addition, the current practice in LCA renders all metal bound in wastes susceptible to leaching (Finnveden 1999). The reasoning is that the infinite time perspective of LCA gives infinite weathering, in turn giving release bound by total contents only. The assumption of complete release does not follow the principle of LCA of risk objectivity; instead it gives results which are risk conservative for the long-term emission of metals. Since metals are non-destructible and the marine environment is the final sink for waterborne emissions, results give unreasonably high scores for metal marine ecotoxicity.

Inventory methods for the long-term release of metals bound in solid wastes should be based on an understanding of the physio-chemical properties of metal deposits. A simple example of the consequence of ignoring the influence of ambient environmental conditions on metal leachability is the potential mobility of lead in cementitious waste, found to be five orders of magnitude higher at leachant pH 5 than at pH 9 (Kosson et al. 2002). Such a variation clearly has the possibility to affect conclusions of a study, and the uncertainty in leaching is significantly larger then what is indicated for immediate emissions in LCA (Lloyd and Ries 2007).

As the drilling industry moves towards use of less hazardous substances, metals are expected to appear as the dominant ecotoxic substances in drilling wastes. The content of hazardous metals in drilling wastes may be large, and emission inventories for drilling operations made by assuming complete release of metals in drilling wastes very possibly overestimate the actual release by several orders of magnitude. The long-term metal emissions from leaching are potential emissions. In other words they may not be observable within the measurement timescale. If potential leaching is set equal to total content, they are potential also in the sense that they are risk conservative. If leaching inventories are compared to immediate emissions one-to-one (which is common practice in LCA) then the total aggregated inventory consists of emissions flows with very different probabilities attached. Assessment of metal ecotoxic contribution from minerals in the drilling fluid compared to that of organic substances relies on the use of a consistent risk perspective for immediate and long-term emission processes.

In order to understand resolve current inventory and impact assessment issues, it is necessary to assess and address the challenges related to the treatment of metal leaching in LCA. The proposed solutions for the treatment of metal leaching in LCA form the third issue for methodological development.

5 METHODOLOGICAL DEVELOPMENTS

5.1 Marine ecotoxic effect of pulse emissions in life cycle impact assessment

5.1.1 Background and aim

Ecotoxicity has a long tradition in life-cycle assessment and several approaches have been proposed to assess the relative ecotoxic potential of substances. The approaches discussed here rely on modeling of ecotoxic impact by two separate stages: exposure and effect assessment. Exposure modeling is performed to estimate the effective dispersion between compartments - aquatic, sediment and terrestrial - and the average residence time in each. It is therefore also referred to as fate assessment as the result is an indication of the environmental fate of emissions. Effect assessment is the translation of exposure to ecotoxic effect potential; i.e., it is a description of the cause-consequence relationship for ecotoxicity.

Linear models are used for fate modeling in ecotoxic impact assessment in LCA (e.g., by Heijungs 1995; Huijbregts et al. 2000; Hertwich et al. 2001), with compartments typically in the scale of regions or continents. Originally intended for steady-state analysis, multi-compartment models are described by a system of first order differential equations. It has been shown that the multi-compartment models accommodate pulse-emissions as long as the ecotoxic effect is proportional to concentration (Heijungs 1995). But, this conclusion does not hold for non-proportional effect functions unless it is assumed that exposure increase is marginal at all points in the environmental compartment that is assessed.

An issue that needs clarification before LCA is applied to offshore drilling operations is if the assumption of marginal exposure is significant for the ecotoxic characterization of pulse emissions relative to continuous emissions.

5.1.2 Ecotoxic effect definition

Several classes of effect models have been used in ecotoxic impact modeling and, although all of them rely on use of the same information, they have important differences (Hauschild and Pennington 2002; Pennington et al. 2004; Pennington et al. 2004). The effect models considered here are based on species sensitivity distributions taking into account the combined effect of multiple substances. They differ in the definition of combined ecotoxic effect.

Various substances, and toxic modes, show different patterns for the distribution of ecotoxic sensitivity in a panel of test species (de Zwart 2002). This distribution may be described by use of probabilistic methods, referred to as species sensitivity distribution (SSD). The approach and benefits of SSD is well documented (Posthuma et al. 2002). Ecotoxic effect is by use of SSD defined as an increase of probability; e.g., in (Goedkoop and Spriensma 2001; Huijbregts et al. 2002; van de Meent and Huijbregts 2005). The cause-consequence relationship for ecotoxicity in LCA has generally been based on chronic ecotoxic test observations, termed no-effect concentrations (NOEC). For SSD functions based on NOEC, the ecotoxic effect of an emission is the increase in probability that a random species is affected by the resulting increased chronic exposure.

Species sensitivity distributions are non-linear, bound within the interval 0-1 as the boundaries for probability. One consequence of using a non-linear cause-consequence relationship is that it requires the exposure prior to the additional emission to be known; i.e., the background exposure. Moreover, if the non-linear SSD is combined with multi-media models it requires use of a first order Taylor approximation.

With the SSD, ecotoxic effect is defined as the change in SSD:

Equation 3: $$Effect_{Function} = \Delta SSD = SSD(C_1) - SSD(C_2)$$

where *SSD* is the species sensitivity distribution and *C* is concentration, changing from an background concentration of C_1 into a concentration C_2 after an emission. Ecotoxic effect as calculated by the SSD effect function is denoted $Effect_{Function}$. The first order derivative is defined as the effect factor, and ecotoxic effect by use of an effect factor ecotoxic effect becomes

Equation 4: $$Effect_{Factor} == \left.\frac{\partial SSD}{\partial C}\right|_{C_1} \cdot \left(C_2 - C_1\right) = E \cdot \Delta C$$

where *E* is the effect factor, equal to the first order derivative of the SSD function, and $Effect_{Factor}$ is the ecotoxic effect as calculated by use of the effect factor approximation.

5.1.3 Effect factor and effect function – different ecotoxic effect

The influence on ecotoxic modeling by use of an effect factor for the SSD is best illustrated graphically. Figure 9 shows the ecotoxic effect as calculated by use of a continuous SSD function (Equation 3), and an effect factor approximation (Equation 4). Since the SSD function yields a probability, effect calculated by use of a continuous effect SSD function converges towards one. No such convergence occurs if the ecotoxic effect is defined proportional to concentration. The difference between these curves is dependent of the size of the concentration increase. While any emission into a multi-compartment regional-sized model per definition is marginal since it is averaged for the entire volume of the compartments, intermittent emissions from an offshore rig induce large concentration gradients in the local environment.

Ecotoxic impact is in life-cycle ecotoxic assessment defined as the time- and volume-integral of ecotoxic effect. The ecotoxic effect is integrated for infinite time over the volume of each compartment in the fate model. The difference between applying a constant effect factor and a continuous SSD effect function in this integral is illustrated in Figure 10. The graph shows volume-integrated effect over time for both approaches, given an initial pulse emission at time zero. In the initial stage of dispersion the conversion of the SSD curve for very high ecotoxic effects results in a lower volume-integrated effect for the ecotoxic effect defined form the SSD function. After some time, dispersion brings the SSD-curve towards the volume-integral of the factor-based effect curve. This is because the dispersion over time converges to an exposure profile resulting from the marginal assumption that is the basis for the effect factor. This may also be observed in Figure 9 for small concentration increases. At some point in time, degradation becomes the limiting factor rather than the dispersion.

The constant decrease of the volume-integral of effect as estimated by the effect factor is due to degradation of the substance that is modeled. The substance modeled in Figure 10 has a relatively short residence time. For substances with longer residence times, the dispersion phase becomes less significant for the final characterization factor, and the degradation phase dominates the time-integrated ecotoxic effect.

Before commencing with the results from the modeling, it is necessary to discuss one more issue with the SSD. With ecotoxic effect a probabilistic function of concentration, two different definitions may be adopted for the combined toxicity of multiple substances. These are referred to as response addition and concentration addition. In the first approach, the ecotoxic effect is a function of the combined probability of independent toxic pathways (responses), each modeled by a separate SSD (Huijbregts et al. 2002; van de Meent and Huijbregts 2005). The latter approach defines a multi-substance SSD for toxic equivalents, found as the toxicity-weighted sum of substance concentrations (Goedkoop and Spriensma 2001). Both methods have been used in LCA and both are considered in this investigation.

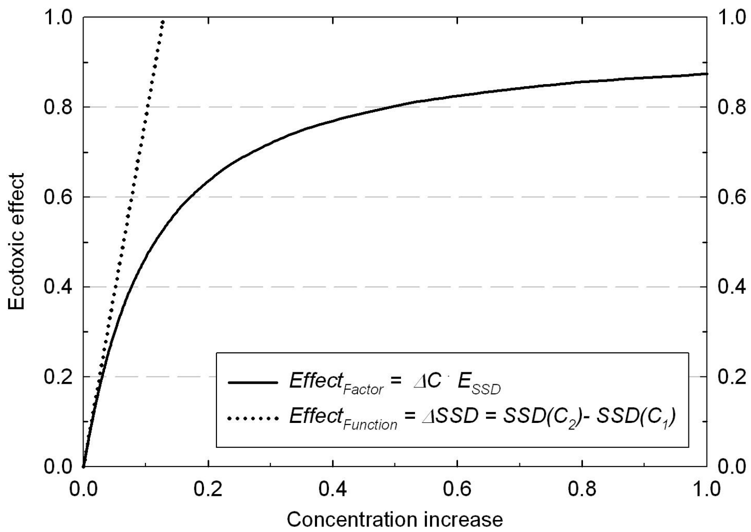

Figure 9: General illustration of the difference between the original effect function (SSD) and a first-order Taylor approximation (effect factor) as a function of concentration increase.

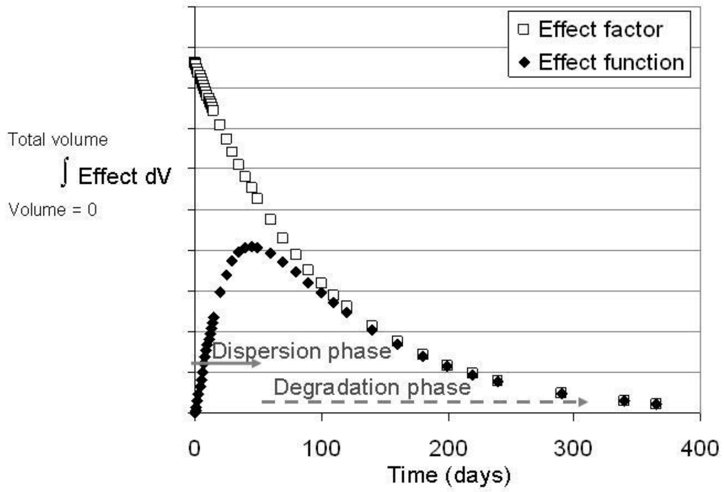

Figure 10: Volume-integrated effect as a function of time in a closed compartment, only including dispersion of the plume and a first order degradation rate.

5.1.4 Model results

The significance of using an effect factor approximation for the SSD effect function on the modeled ecotoxic impact was investigated for several substance properties, emission loads, and background exposure levels.

A transient model was defined, consisting of a closed, finite marine aquatic volume. Dispersion was modeled with commercial software (COMSOL 2004) using dispersion coefficients observed from ocean dye studies. Ecotoxic effect was calculated simultaneous to concentration, allowing time- and volume-integrals to be extracted.

Final results are presented as ratios; *W*, representative of the relation between characterization factors for ecotoxicity (*Q*) calculated by use of the effect factor (Eq. 3) and the effect function (Eq. 2):

Equation 5: $$W = \frac{\int_{Time=0}^{\infty} \int_{Volume} Effect_{Function} dVdt}{\int_{Time=0}^{\infty} \int_{Volume} Effect_{Factor} dVdt} = \frac{Q_{Function}}{Q_{Factor}}$$

Results from the calculations are presented in Figure 11 for both the concentration addition definition of the SSD, and for a combined response and concentration addition definition. The plot shows that if the concentration addition rule is assumed for the ecotoxicity of a mix of substances, the Taylor approximation gives characterization factors in the scale of those calculated by use of the original SSD effect function. However, the opposite conclusion is found if response addition is assumed: characterization factors with the SSD effect function are many orders of magnitude less than the ones calculated by use of the Taylor approximation. The degree of deviance is strongly correlated with the background concentration.

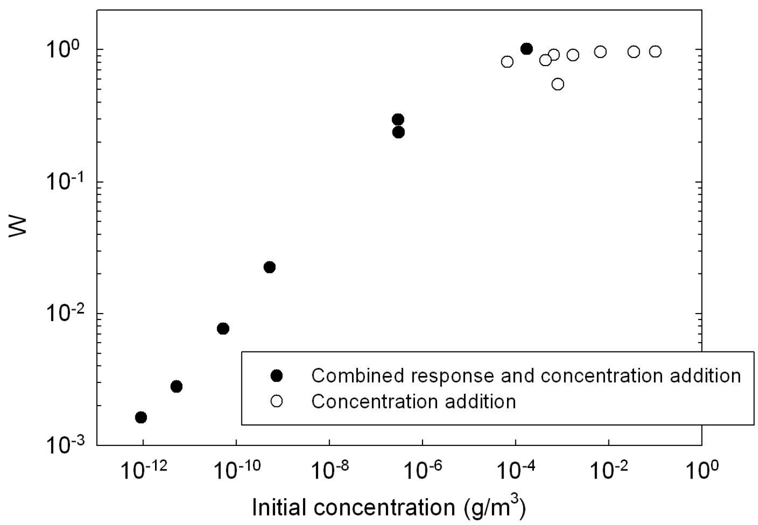

Figure 11: Ratio of characterization factors found by effect function and by effect factor.

The finding is explained by the principle of calculating multi-substance toxicity. The background exposure is relatively large for relevant scenarios with the concentration addition rule. The same may be stated for the multi-substance toxicity by response addition, but if the substance (or toxic mode) that is modeled shows a low background exposure, the first order Taylor approximation is not robust for the pulse that is modeled for the particular substance. This may be explained by the derivative of the two rules for mixture toxicity. Taking the concentration addition, the first order derivative is:

Equation 6: $$\frac{\partial SSD_{CA}}{\partial C_{Subst\,i}} = \frac{\partial SSD_{CA}}{\partial TU} \cdot \frac{\partial TU}{\partial C_{Subst\,i}}$$

where SSD_{CA} is the SSD by concentration addition, *TU* is toxic units, and $C_{subst\ i}$ is the concentration increase for substance *i*. The important parameter seen in this equation is the right-most term, which denotes the change in toxic units as a derivative of concentration of substance *i*. This term is generally constant through the potential changes in concentration for the substance given the relative large exposure present prior to the additional release. But, if the same derivative is expressed for response addition SSD, it should be apparent that it is more sensitive to the changes in concentration for substance i:

Equation 7: $$\frac{\partial SSD_{RA}}{\partial C_{Subst\,i}} = \frac{\partial SSD_{RA}}{\partial SSD_{Subst\,i}} \cdot \frac{\partial SSD_{Subst\,i}}{\partial C_{Subst\,i}}$$

where SSD_{RA} is the SSD by response addition, and $SSD_{Subst\ i}$ is the SSD for the particular substance *i*. The rightmost term in Eq. 7 is the crucial parameter deciding the particular sensitivity of SSD_{RA} towards pulse emissions. With the response addition, background concentrations may be very small for the substance or toxic mode that is investigated. This gives a larger relative span for the potential increase in concentration, and thereby a less robust Taylor approximation.

5.1.5 Conclusions for overall evaluation of drilling fluid technology

The modeling of pulse emissions by transient simulation showed that the existing characterization factors may be used to assess marine intermittent emissions relative to continuous emissions if the concentration addition assumption is used to estimate mixture ecotoxic effect with species sensitivity distributions.

The response addition assumption for the multi-substance species sensitivity distribution is not robust for substances with low background concentration and short residence times in the marine aquatic environment.

Observations of ecotoxic response in single species support use of both concentration addition within toxic modes and independent action for dissimilarly working substances (Altenburger et al. 2000; Backhaus et al. 2000; Faust et al. 2003; Verslycke et al. 2003). Which method is most representative for ecosystem response is uncertain (Backhaus et al. 2003).

If response addition based species sensitivity distributions are to be used to assess the ecotoxicity of offshore pulse emissions in life-cycle assessment, transient simulation must be performed for these substances to ensure that the relative impact of pulse emissions is not overstated relative to continuous emissions.

5.2 Occupational health: offshore crane-lifts in life cycle assessment

5.2.1 Background and aim

Company processes cause impacts to both the external environment and the internal working environment. Traditional life-cycle assessment draws a border between these two, where the workplace is seen a system separate from the entities affected by product life-cycles. Arguments raised for this separation are the assessment of occupational health makes LCA more complex, that the issue is covered by regulatory standards, and that the impacts do not fit within the existing impact assessment categories (Antonsson and Vershoor 2004). However, methodological challenges have been resolved for impact categories currently included in LCA and regulations exist for most impacts assessed with LCA. Furthermore, human health is an area-of-protection shared by cause-consequence chains that work through emissions to the external environment and health impacts caused by occupational hazards. The latter argument has been supported quantitatively for sectors in the US economy (Hofstetter and Norris 2003).

Current methods to assess occupational health in LCA rely on the use of company or sector statistics to relate working environment conditions to products and processes (Poulsen and Jensen 2004). While useful to compare product alternatives, company statistics do not allow process comparison. An important safety aspect for offshore drilling fluid technologies is the use of cranes to move and load cargo and equipment internally on rig and onto or off the supply vessel. Lifting accidents account for approximately one third of all reported incidents and about half of all incidents with health consequences on offshore rigs in UK waters (HSE 2005a; HSE 2005b). Lift operations thereby are a controlling factor for offshore occupational safety, and the means should be developed that allow comparison of offshore technologies with different requirements for the use of crane-lift.

A large number of crane-lifts are required for the transportation of drilling waste from rig to an onshore treatment facility. The aim of this paper was to develop a characterization factor for crane-lifts. The characterization factor should allow comparison of the expected health impacts induced by the accident risk for crane-lifts to health impacts caused by emissions from transportation and treatment processes. The complete modeling approach is described in the appended paper. The description here is limited to an outline of the approach and main findings. Applications of the characterization are given in the case study in Chapter 6.4

5.2.2 Method of approach

Disability adjusted life years (DALY) have been used to assess the relative health damage potential from several cause-consequence mechanisms in life-cycle assessment, including road noise, ionizing radiation, carcinogenic and non-carcinogenic toxicity, as well as respiratory effects (Hofstetter 1998; Frischknecht et al. 2000; Goedkoop and Spriensma 2001; Müller-Wenk 2004). In order to make expected health damage from accidents commensurable to health damages from emission-related impacts, the DALY framework was selected as the category indicator for occupational health.

The DALY framework was originally intended for health economics (Murray and Lopez 1996). Factors are available that describe the relative disability caused by various physical injuries, and these were used to evaluate the damage caused by the injuries observed in statistical records of crane-lift accidents on offshore drilling rigs. Records have been compiled for all incidents on offshore jack-up and semi-submersible drilling rigs in UK waters over an extended time period (HSE 2005a), and forming the primary source of information for the modeling in this paper.

Probabilistic parameters were introduced to relate annual accident frequency to the probability of an accident with human health damage per crane-lift. Health damage is described by:

Equation 8: $$Q = \frac{u}{c}\frac{1}{n_t}\sum_{m}\left(n_i d_i w_i\right)$$

were Q denotes the total expected health damage per crane-lift (DALY per lift), u is the statistically observed frequency of accidents with health damage in the source data (crane-lift accident per time), and c is the intensity of crane-lifts (lifts per time). In total, 177 crane-lift accidents (n_t) with health damages were observed in the source data (in the period 1980-2003). The injury outcome for each of the 177 accidents was classified manually by interpretation of the description of the event given in the source data. Injury outcomes are denoted i in Equation 8. Each injury class is assigned a disability weight w_i and a duration d_i, based on values in the DALY framework (Murray and Lopez 1996). The duration for lifelong injuries was modeled by the reported age at injury for offshore drilling workers (Forbes 1997) and the expected remaining lifetime of males in the UK (GAD 2006). With exception for the duration of recoverable injuries and the disability weights, all factors were treated as probabilistic parameters; i.e., they were assigned probability distributions.

Table 4: Classification of accident records by their health outcome.

Health outcome [i]	Cases[a] [n_i]	Weight[b] [w_i]
Fatalities	2 (+2)	1.000
Amputation – thumb	1 (+1)	0.165
Amputation – finger	4 (+5)	0.102
Amputation – toe	0 (+2)	0.078
Amputation – foot	1 (+0)	0.300
Fracture – face bones	0 (+4)	0.223
Fracture – rib or sternum	0 (+3)	0.199
Fracture – pelvis	1 (+2)	0.247
Fracture – clavicle, scapula, or humerus	1 (+1)	0.153
Fracture – radius or ulna	1 (+2)	0.180
Fracture – hand bones	9 (+16)	0.100
Fracture – patella, tibia, or fibula	3 (+8)	0.271
Fracture – ankle	1 (+4)	0.196
Fracture – foot bones	1 (+14)	0.077
Minor injuries	88 (+64)[c]	0.108[d]

[a] On format: Certain cases (+ Potential cases)

[b] Disability weights from Murray and Lopez (1996, table 4.4)

[c] Modeled so that $\Sigma(n_i)|_i = n_t = 177$

[d] Assumed with equal weight to Open wounds in (Murray and Lopez 1996)

Classification of accidents by injury class is presented in Table 4. Most injuries are interpreted as recoverable injuries with minor damage. Only eight of the 177 cases are fatalities and amputation cases. Uncertainty is assigned to the distribution of outcomes based on the uncertainty in the manual classification of the source data.

5.2.3 Main results

Monte Carlo simulation was used to estimate the uncertainty in the characterization factor for health damage from crane-lifts. Results from simulation of Equation 8 are presented in Figure 12, separated between recoverable injuries (including fractures and minor injuries), amputation cases, and fatalities. The graph shows that minor injuries and fracture cases are not significant for the total health damage of crane-lift accidents. Fatalities dominate the health damage, representative of about two thirds of the total damage.

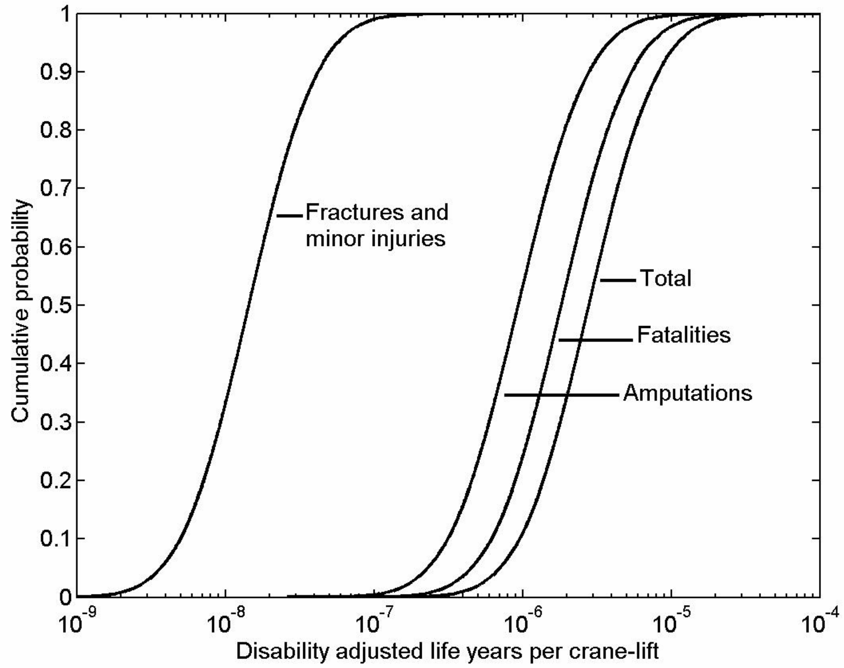

Figure 12: Cumulative probability distribution for the expected health damage per crane-lift.

The main contributor to uncertainty in the characterization factor for offshore crane-lifts is accident frequency. Uncertainty is significant, with a 95% confidence interval described by $\sigma^2 \approx 5$ (geometric standard deviation). This is less than what is indicated for the other impacts chains in LCA; e.g., (Hofstetter 1998; Frischknecht et al. 2000; Hertwich et al. 2000; Huijbregts et al. 2005). The mean characterization factor is $4.0 \cdot 10^{-6}$ DALY per crane-lift, with cumulative percentiles $[P_{2.5}, P_{50}, P_{97.5}] = [5.4 \cdot 10^{-7}, 2.8 \cdot 10^{-6}, 1.5 \cdot 10^{-5}]$..

5.2.4 Conclusions for overall evaluation of drilling fluid technology

The means to include the human health damage from crane-lift accidents in overall evaluation of offshore drilling fluid technology has been developed. Uncertainty in the modeled characterization factor is within the range reported for other impacts chains in LCA. It is therefore operationally applicable for comparative assessment of offshore drilling fluid technology with different crane-lift intensities.

5.3 Metals in life-cycle assessment: current inventory issues and possible solutions

5.3.1 Background and aim

The estimation of long-term metal emissions from wastes is a major source of uncertainty in ecotoxic assessment in life-cycle assessment. There are two main reasons for this. First, current impact assessment methods are not designed for inorganic emissions, and second, long-term leaching processes represent a problem for life-cycle inventory estimation.

Methods are underway in environmental risk assessment and life-cycle impact assessment for improved assessment of metal ecotoxicity. Aquatic phase speciation and ecotoxicity models represent important steps forward in the evaluation of metal ecotoxicity once they are released into the environment (Paquin et al. 2003; Adams and Chapman 2007; Harvey et al. 2007). The field is moving forward, and better characterization factors for the ecotoxic potential of metals relative to each other as well as relative to degradable, organic substances are expected in the near future; e.g., (Wegener Sleeswijk 2005). Issues, remain, however, with the estimation of release from solid wastes.

Given the infinite time-perspective of life-cycle assessment, long-term leaching emissions must be predicted rather than measures when compiling the life-cycle inventory. The long-term perspective of LCA is traditionally interpreted as meaning infinite weathering, resulting in the complete release of metals bound in solid deposits (Finnveden et al. 1995; Hellweg et al. 2005). Leaching estimates for inorganic substances are thereby indicative of potential rather than actual emissions, carrying both a larger uncertainty and a risk conservative bias compared to the evaluation of immediate, measurable emissions and leaching of degradable organic substances. The aim of this paper is to review current approaches in LCA for estimation of long-term metal leaching, and identify possible solutions based on approaches used in related fields.

As the final recipient for waterborne emissions, assessment of marine ecotoxic impact is particularly affected by the methodological gaps discussed above. Moreover, inorganics have very long residence times in the marine environment.

With the intention to forward the main conclusions with respect to an overall evaluation of offshore drilling fluid technology, the following provides a synopsis of the paper. The full paper should be consulted for complete discussion and references.

5.3.2 Current solutions and framework of discussion

As a tool for evaluation of the sustainability of products, the assessment timeframe of LCA is infinite. Impacts should be included in the assessment regardless of temporal and spatial considerations (Udo de Haes et al. 1999). Two options have been proposed to address the issue of long-term inventory estimation within the infinite timeframe; either by introducing uncertainty in inventory modeling or by discounting future emissions. In the first option a separation is made between the inventory that can be stated with some certainty; such as the first 100 years, and the inventory that is linked to higher uncertainties. Long term potentials seen in the literature include zero (Nielsen and Hauschild 1998), the total mass initially placed in the landfill (Moberg et al. 2005), and the fraction expected to be released prior to the next ice age (Doka and Hischier 2005).

Discount rates reduce the significance of emissions that occur in the less certain future compared to emissions that occur today. The introduction of a discount rate is in fact a weighting between future and present impacts and as such it violates the objective perspective asked for in the ISO standards (ISO 2006) and many in the LCA community (Hofstetter 1998; Finnveden 1999; Finnveden and Nielsen 1999). Hellweg and colleagues discuss the consequences of introducing discount rates for leaching processes and argue that positive as well as negative rates may be relevant. They further conclude that the influence of the discount rate should be investigated by scenario analysis (Hellweg et al. 2003).

Any scheme temporal boundary or discount rate is arbitrary. While they may simplify the estimation of long-term emissions, they are inherently subjective and thereby not suited for generic implementation in LCA. Better solutions should be sought, that are based on an understanding of the physical-chemical-geological processes that control metal stability in solid phases and govern metal attenuation and release.

Estimation of leaching from solids is an issue met also in environmental risk assessment and waste management. Drawing on the approaches used in these fields, possible solutions are found using the concepts of geoavailability and mobility. The interpretation of geoavailability and mobility used here is illustrated in Figure 13, as part of the ecotoxic impact chain for metals bound in solid phases. Geoavailable metal is the fraction of total metal that within the boundaries for mobility and dispersivity can be mobilized to bioavailable states. Mobility describes the physio-chemical processes that govern metal fate, while dispersivity refers to the physical processes that drive dispersion (Smith and Huyck 1999)

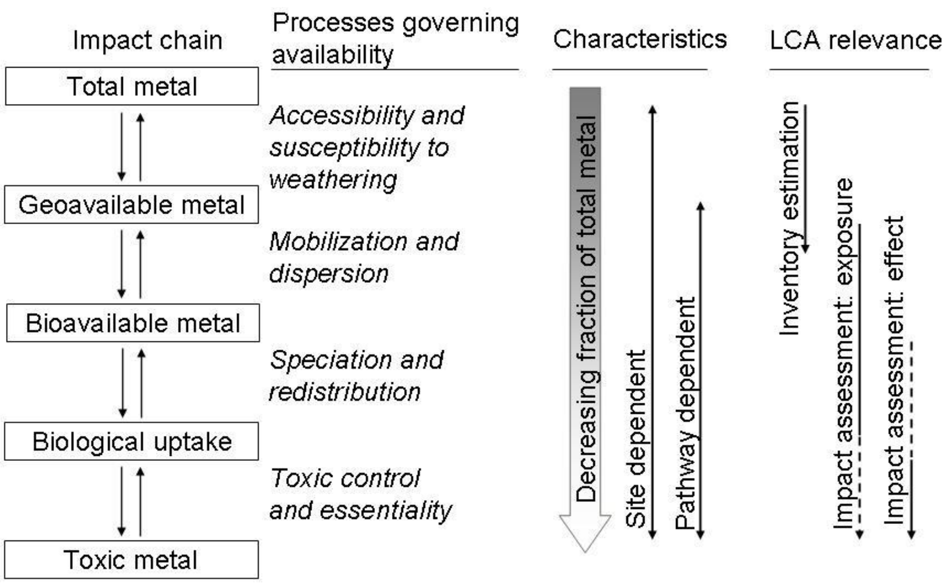

Figure 13: Overview of processes in the metal ecotoxic impact chain, their characteristics, and relation to life-cycle assessment. (Adapted from Smith and Huyck 1999)

5.3.3 Possible approaches

The approaches used in waste characterization and soil and sediment risk assessment are outlined below.

Waste characterization

Waste management relies on the use of characterization tests as decision support for waste and resource management. A large effort has been invested in the harmonization of waste characterization tests (van der Sloot et al. 1997; Grathwohl and Halm 2003; van der Sloot et al. 2004). The base on which the harmonization project rests has an obvious value as a source of end-of-life inventories, but the general conclusions are also relevant for LCA. Contrary to the general assumption of complete release, waste leaching takes as a starting point that three levels of leaching potentials may be identified (van der Sloot et al. 1997):

- The total mass placed in the landfill

- The potentially leachable fraction of metal
- The fraction that is actually released

Standardized tests are used to investigate the different potentials. The focus here is on tests for the potentially leachable fraction as they provide an estimate of the boundary for the long-term leaching potential. Scenario analogy tests are used to assess leachability, often referred to as availability tests (Finnveden 1999; Kosson et al. 2002). Generally used as pass/fail criteria, availability tests are designed to be simplistic in use and to offer risk conservative results.

The main controlling parameters for leaching have been found to be pH and the degree of percolation. The overestimation of availability tests compared to tests for actual leaching may be illustrated by plotting release against these two parameters; see Figure 14. With the exception for metals that form oxyanionic metal species, the release is lowest at neutral or near-neutral pH conditions, and the availability increases with decreasing pH. Given that the percolating volume increases with time, the volume is interpreted as the temporal aspect of leaching. Infinite volume cannot be investigated by experiment, but the release generally converges towards a value less than the total metal. The cumulative release at volume-to-solid ratios of 100-1000 is accepted as the potentially leachable fraction (NEN 1995; Nordtest 1995; OECD 2001; CEN 2004).

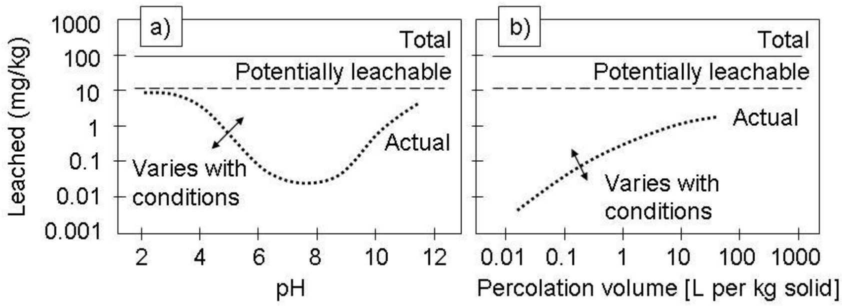

Figure 14: Leaching as a function of a) pH and, b) the percolating volume (redrawn from van der Sloot et al. 1997). Condition variability includes parameters such as organic phases, oxidization-reduction conditions, etc.

The conditions tested with availability tests do not allow large scale changes to the waste material that is leached. Such changes would include variation in oxidization-reduction (red-ox) conditions. A solution to address this issue is to apply sequential extraction.

Sequential extraction

Metals in solids are associated with mineral or organic phases, each mobilizable at certain environmental conditions. Sequential extraction is the selective extraction of metal bound in target geochemical phases (Tessier et al. 1979). It is frequently used in risk characterization of soils and sediments to estimate the mobility of solid-bound metal.

Several schemes have been proposed for sequential extraction, identifying 3-9 different operationally defined phases (Ure et al. 1993; Tack and Verloo 1995; Filgueiras et al. 2002; Sahuquillo et al. 2003). One classification system is given in the leftmost column in Figure 15. The extractants used may be described along a gradient of increased leachant strength. Alternatively, phases may be classified as labile/nonlabile or mobile/nonmobile.

The classification of phases by sequential extraction may be placed within a framework of mobility and geoavailability, as outlined in Figure 13. The least extractable phase by sequential extraction is termed the residual, or refractory phase. For the purpose of leaching, the residual phase is considered inert as it contains the metals bound in stable mineral structures. It is thereby the difference between total metal and geoavailable metal. The geoavailable metal may further be separated into a fraction of high mobility and a fraction of low mobility.

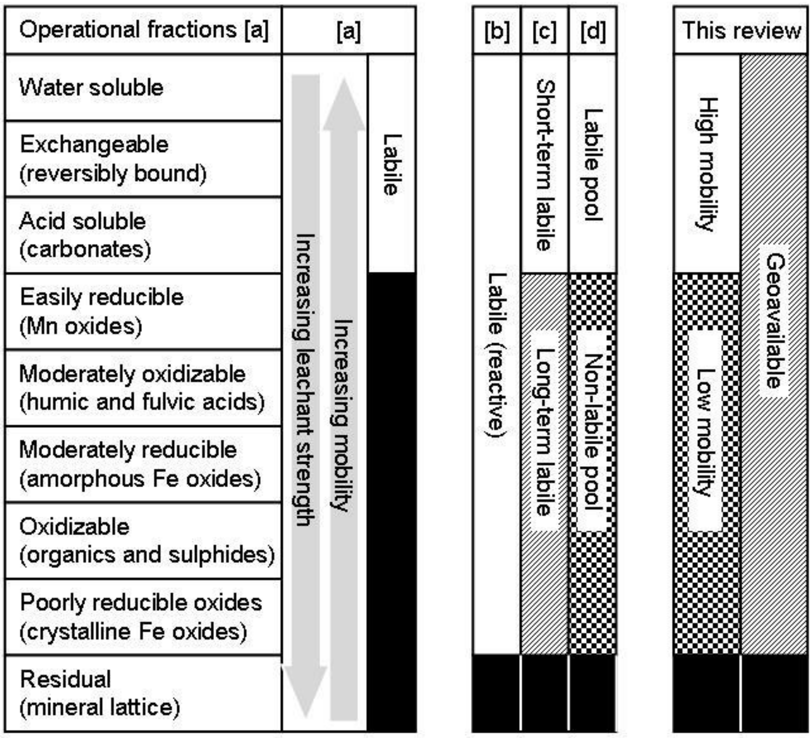

Figure 15: Overview of phases extracted by sequential extraction, and interpretations of metal mobility met in the literature. Notes refer to references; a: (Filgueiras et al. 2002), b: (Manz et al. 1999), c: (Kennedy et al. 1997), d: (Almås et al. 1999).

5.3.4 Geoavailable and mobile metal in barite

Leaching potentials were established for barite, based on a comprehensive review of the available literature (sources include Nelson et al. 1984; Trefry et al. 1986; Deeley 1989; Deuel and Holliday 1998; Fjogstad et al. 2002; Myran 2003; Linjordet et al. 2004; Novatech 2006; Westerlund 2007). In accordance with the classification in Figure 15, content and mobility of ten trace metals was estimated for total contents, geoavailable fraction, and highly mobile fraction. Results are plotted in Figure 16.

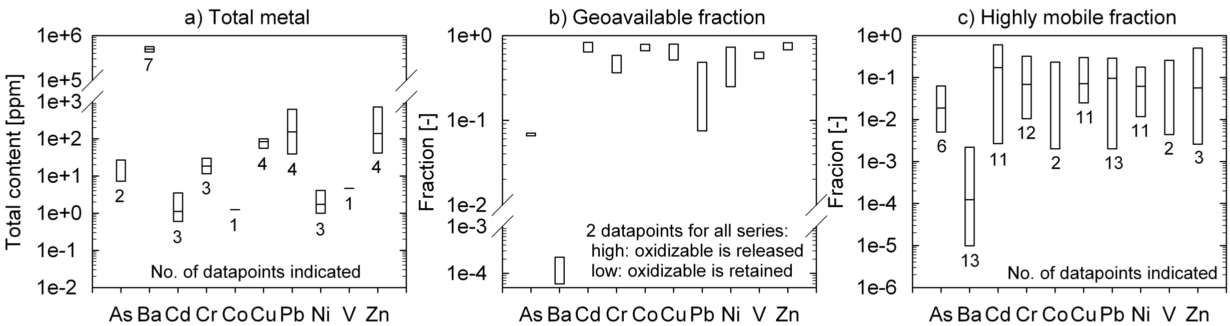

Figure 16: Uncertainty in total, geoavailable and highly mobile metal in barite. Spans cover the high-low interval. The geometric mean is indicated. For the geoavailable metal, high and low estimates are including and excluding the oxidizable fraction respectively.

Highly mobile metal was assumed represented by the leachable metal according to a pH-separated extraction scheme, cation-exchangeable and DTPA extractable, as well as the water-leachable fraction. All these are equal in concept, and assumed equivalent in our calculations, to the sum of exchangeable and carbonate phases by sequential extraction.

As the graph in Figure 16 indicates, various literature sources produce order of magnitude variation in total metal, as well as metal mobility levels. But, it also communicates the importance of considering the state in which metal is bound. If the more certain, highly mobile fraction is used as the leaching potential, large uncertainties still remain for the size of the potentials. The uncertainty in leaching emissions are far higher than those used as generic factors in LCA (see review by Lloyd and Ries 2007). Moreover, if geoavailable metal is used as the definition of long-term leaching potential, it is vital to consider if oxidizable conditions are met, or if only reducing conditions are relevant. And finally, the investigation of geoavailable metal shows the significance of the inert fraction of metals in solid deposits.

The most comprehensive European life-cycle inventory source, ecoinvent (Jungbluth 2004), assumes that the leaching of metals from offshore drilling wastes can be described by the complete dissolution of barite to barium. If this assumption is replaced with release estimated by geoavailable metal (as given in Figure 16), marine aquatic ecotoxic contribution (according to Huijbregts et al. 2000) from offshore drilling operations is reduced by a factor 50. Most life-cycle assessments are dominated by leaching from solid deposits. This simple example shows that the quality of inventories, in the sense of more realistic estimation of emissions, may be greatly increased by better inventory methods. Attenuation and immobilization occurs in solid deposits (Almås et al. 1999; Singh 2007) and should not be disregarded by life-cycle inventories.

5.3.5 Conclusions for overall evaluation of drilling fluid technology

A final solution for estimation of the long-term release of metals from solid deposits has not been found. The geoavailability concept reduces the risk conservative bias for metal leaching from solids, but cannot remove the fact that emissions are predicted rather than measured. Furthermore, local conditions play a vital role for the accuracy and validity of the geoavailable fraction as an indication of the long-term leaching potential.

The large uncertainty and maintaining risk conservative bias for metal emissions from solids underlines the caution that should be used when comparing immediate emissions to those predicted for solid deposits.

6 OVERALL EVALUATION OF OFFSHORE DRILLING FLUID TECHNOLOGY

6.1 Challenges for life-cycle assessment of drilling operations

6.1.1 Variability of drilling operations

That "no man ever steps in the same river twice, for it's not the same river and he's not the same man" (philosopher Heraclitus) is true also for drilling operations: no well may be drilled twice. Hence, all comparative assessments of drilling technologies are comparisons of hypothetical systems. At best, one of the alternatives is the actual outcome of an operation while the others comprise of assumptions based on knowledge of the technical challenges and technology employed in each case.

Issues which may show large and often unpredictable variations with effect on drilling systems include down-hole formations (through its effect on loss rates and reuse potential for fluids, and the generation of cuttings waste), the design and efficiency of the solids control system employed on rig to separate fluids and cuttings, and stops in operations caused by weather conditions. Depending on the drilling rig that is utilized at any location, limitations may exist regarding space and technical boundaries, with effect on the amount of equipment that can be installed on rig for storage, solids control and waste logistics and treatment.

Issues related to geology, weather and rig construct vary from operation to operation. Although greatly affecting the performance of the drilling operation, their influence may not be accurately predicted prior to operations. Comparative assessment of drilling technologies therefore relies on the use of scenarios and average data. We have applied empirical parameters related to composition, use and disposal of wastes, but complete descriptions of the alternatives based on measurements cannot be made. The influence that variability in geology, weather and rig factors has on the life-cycle of drilling fluids must therefore be implemented by use of uncertainty and sensitivity analysis.

6.1.2 The functional unit

What is the function of drilling? There are two specific causes to undertake drilling operations. The first is to investigate if there are resources that justify production; i.e., exploration drilling. The second is to gain physical access to the oil and gas resources; i.e., production drilling. Ultimately, the function of drilling operations is to secure availability of oil and gas resources. The top-level function of drilling may therefore be stated as energy generation or material extraction.

In the case of exploration drilling, better knowledge of resource characteristics may be attained through non-intruding means, such as seismics. Alternatively, solutions may be used that do not require an entire well to be drilled, such as the drilling badger currently under development. The prototype is developed by Badger Explorer ASA (Stavanger, Norway; http://bxpl.com/). The drilling badger is an independent drilling unit which closes the hole after itself as it keeps drilling deeper. Communicating by radio or cord, the badger unit does not produce cuttings waste and drilling fluid is not employed.

For the case of production wells, the physical connection that the well provides cannot be exempted. It can, however, be made with less waste produced. Drilling with smaller well radii, termed slim-hole drilling, will require less cuttings to be carved out of the well, thereby producing less cuttings waste and reducing the use of drilling chemicals.

Both exploration and production wells must be drilled to a predefined depth for the well to serve its purpose. The depth is set by the geology at the site and the location of potential oil and gas resources. With this as a starting point, we define the functional unit for drilling as the service that it provides. A functional unit for offshore drilling operations thereby is:

> *the drilling of a well, given the information available concerning location, depth, formation data, inclination, rig characteristics, waste reuse and logistic, distance to potential waste treatment sites, and other relevant and available information.*

Components of the product system of drilling operations include, although not limited to, the drilling rig, fluid components, logistical systems, waste treatment facility and site, labor force. Each of the components serve as subsystems to the drilling operation and thereby serve separate sub-functions. They may be assigned functional units in LCA. To illustrate this we consider one of the sub-systems: the drilling fluid. The drilling fluid serves to

- maintain pressure down-hole (balanced by the density agent),
- stabilize well walls (by density and surface tension agents),
- carry cuttings to surface (dependent of viscosity, controlled by viscosity agents and the base fluid)
- cool and lubricate the drill-bit (predominantly performed by the base fluid)

The function of drilling fluid is ensured by various subsystems of the fluid, added to the fluid as chemical components. Some components have simple functions, such as pH stabilizers, while others perform several tasks simultaneously, such as the base fluid (see Section 3.1)

For comparative assessment of drilling technology, we have defined the functional units with the functional unit of drilling operations in mind. That is, starting from the single well that is drilled. The functional unit for the base fluid in a drilling fluid thereby is:

> *the function of base fluid for the drilling of a well, given the information available concerning location, depth, formation data, inclination, rig characteristics, waste reuse and logistics, distance to potential waste treatment sites, and other relevant and available information.*

Arguably, our definition of functional units is very similar to reference flows, with the well as the reference flow. The approach that we outline is applicable for comparative LCA of subsystems of the drilling operation as long as the complementary subsystems in the system are kept identical or are replaced by systems which are equal in function. For assessment of large changes in the drilling technology, we would have to separate between exploration and production wells and the product system components used in either of the two types of wells.

6.1.3 Information availability

An important aspect of performing LCA is the availability of information to describe the product system that is investigated. Data availability is an important challenge for LCA of oil and gas operations. Much of the information exists in the grey literature and a large part of it is withheld from the public due to the value of information relating to resource stocks and technology in this industry.

The information that is disclosed is often reported as aggregates or made anonymous with regards to substances and products. An example is the content and characteristics of the chemicals used in drilling fluids. Material safety data sheets are public information, but chemical components may be stated as a group of substances, e.g., "polyalkylene glycols" in Glydril MC (supplier: M-I Norge AS, Stavanger, Norway). Although chemical component characteristics such as degradability, ecotoxicity and biological accumulation potential must be documented according to the harmonized offshore chemical notification format (HOCNF), this information is communicated to the public only as red/yellow/green indications; see e.g., the applications for drilling permission in the Barents Sea; e.g.

(Lykling Berge 2004; Lykling Berge and Breivik Jakobsen 2005). Furthermore, the flexibility in the HOCNF guidelines allow exclusion of substances not deliberately added (OSPAR. 2003), such as contaminations or hazardous substances within chemical substance groups.

The issue of substance confidentiality and drilling chemical contents affects life-cycle assessment in two ways. First, it makes it difficult to match ecotoxic characterization factors with emission inventories. Characterization factors may be missing for components of the fluid, or the information that is available does not allow substance identification. Second, it makes it difficult to match chemical products with production inventories. Some of the chemicals therefore have been assessed by means of proxy materials in this study.

An important factor for assessing the life-cycle of drilling chemicals is the degree of recycling and disposal for the various fluid systems. The existence and format of recorded data is dependent of the operating company. Some issues are highlighted by Lindland (2006). Few actual measurements are available for the loss of fluids down-hole and as residue on cuttings after separation on shakers and contaminant-removal equipment. Material balances are rather uncertain as these make up the most important loss factors. Moreover, the categorization of material flows has been an issue in earlier and current databases. An example is that fluids may be recorded as recycled if sent to shore for potential reuse, rather than at the rig upon actual reuse. Tracing reuse is a difficult task since the intermediate storage is undertaken by the supplier, and fluids often are upgraded onshore.

Figure 8 (in Section 4) illustrates the concerted effort from several actors that is involved in each drilling operation. The offshore operator, the rig contractor, the fluid supplier and the waste contractor are the most important contributors to the final system design and execution. Each of these have separate and often conflicting commercial interests in the operation, and each employs proprietary technology. The single actor with best knowledge of all phases of the operation is the fluid supplier, who oversees operations on the rig and has information on the exact composition of the fluids. When chemicals are handed to the offshore operator and rig contractor, information is limited to that communicated in HOCNF and material safety data sheets. In the next stage, when drilling waste is transferred to a waste contractor, information is limited to that required for safe transport and treatment of wastes. What begins as a complete list of chemical substances in the hands of the chemical supplier is transformed to a set of risk-phrases (R-phrases) for waste when cuttings waste is handed to the waste contractor. Most of the information regarding composition is lost at the stages in-between.

The large number of actors present at any offshore drilling operation also complicates the compilation of life-cycle inventories. The operator is assigned liability in offshore operations, but the information transfer between contractors and operator is not complete.

6.2 Goal and scope

The purpose of the study is to conduct a comparative assessment of offshore drilling fluid technology. Our scope is not to perform a life-cycle assessment of drilling operations, but to assess the relative impact of offshore technology alternatives on the environment as a whole. Process flow-sheets and results are presented separately for each set of alternatives.

6.2.1 The reference well

The assessment relies on a well definition, with given characteristics that decide fluid use, waste production, and drilling waste end-of-life. The well that forms the basis for comparison is the Uranus exploration well, with the physical characteristics summarized in Table 5.

Table 5: Well characteristics (*the well*) – basis for the functional unit (Lykling Berge 2004; NPD 2007)

Name	Uranus exploration well 7227/11-1S
Location	72º 14' 22.2'' N, 27º 22' 14.9'' E
Area	Barents Sea
Drilling facility	Eirik Raude; semi-submersible drilling rig
Distance to shore	115 km (off Nordkinnhalvøya)
Distance to supply base	~ 250 km (from Polarbase, Rypefjord)
Water depth at site	234 m
Operation period	January 13th – March 24th 2006

Well sections	Section length (m)	Description
36''	52	Spud section
26''	365	Spud section
17 1/2''	505	Drilled with fluid return
12 1/4''	992	Drilled with fluid return
8 1/2''	1752	Drilled with fluid return

The Uranus well is located in the Norwegian Barents Sea. As illustrated in Figure 17, it lies about 115 km north off the coastline. The Polarbase supply base was used to service the rig from shore, located in Rypefjord outside the town of Hammerfest. Drilling waste logistics routes are described in Table 6. Two routes are outlined for the waste logistics to shore. Cuttings waste drilled with water-based fluids is shipped via the supply base to the local Stormoen facility, certified for treatment of industrial waste. Cuttings waste containing residues of oily compounds are classified as hazardous waste according to the European waste list (European Council 2002). Oily cuttings waste must be sent to facilities holding certificates for treatment of hazardous waste. Lack of suited facilities in the northern regions means that such waste currently is shipped to Mongstad, which lies about 50 km from Bergen, 1 500 km south of Hammerfest.

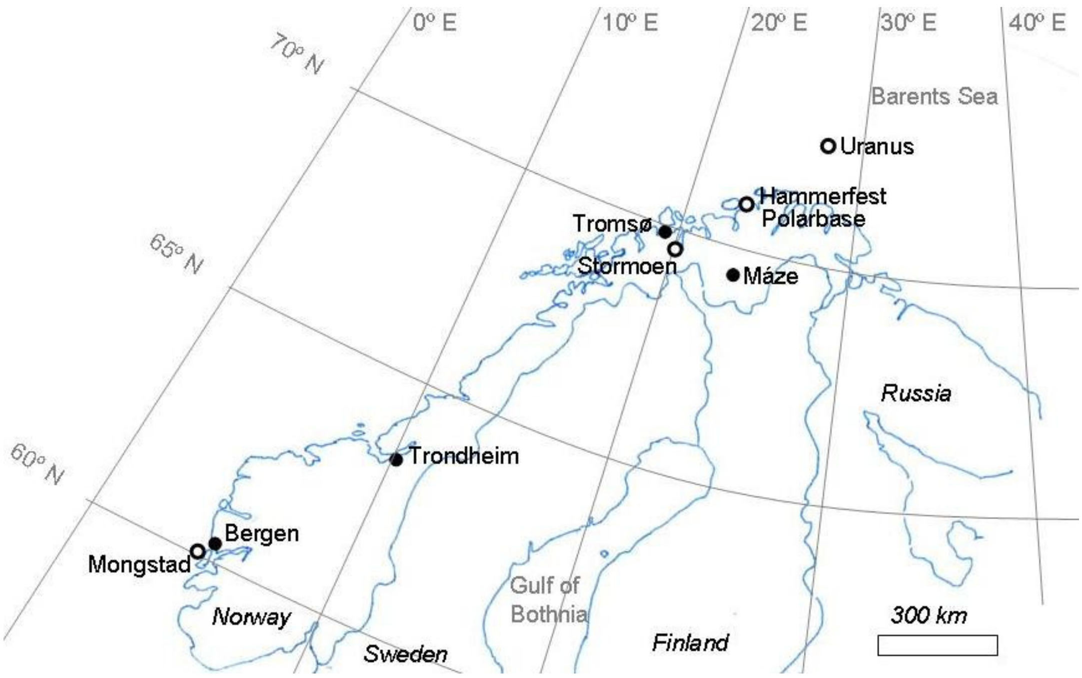

Figure 17: Map of locations. The author is born in Tromsø and raised in the village of Máze.

Table 6: Transport routes for cuttings waste treatment alternatives

No transport – cuttings discharged at site (only applicable to cuttings drilled with WB fluid)	
Transport to local treatment facility – cuttings waste treatment onshore (Stormoen, Balsfjord)	
24h round-trip	Supply-ship to shore (Eirik Raude – Polarbase)
150 nmi	Cargo-ship to facility loading dock (Polarbase – Bergneset)
7 km	Truck transport to facility (Bergneset – Stormoen, Balsfjord)
Transport to non-local treatment facility – cuttings waste treatment onshore (Mongstad)	
24h round-trip	Supply-ship to shore (Eirik Raude – Polarbase)
830 nmi	Cargo-ship to facility loading dock (Polarbase – Mongstad)
1 km	Truck transport to facility (Mongstad dock – Mongstad facility)

Harsh weather conditions are the rule more than the exception in the Barents Sea. Rigs are equipped to handle the local conditions (Paulsen et al. 2005). Still, hard weather may lead to problems with loading drilling waste off rig onto the supply vessel. Given limited storage capacity on rig, the consequence in such cases is that drilling is stopped. The processes induced by a halt in operations are summarized in Table 7.

Table 7: Additional processes induced by a waiting-on-weather incident

Amount	Process	Description
2 days	Additional rig energy	~ 15 tonnes diesel per day
1 trip	Additional supply vessel service	24h sailing, 6h at rig

6.2.2 Functional unit

Characteristics of the reference well are summarized in Tables 5-7. The Uranus reference well is hereafter referred to as *the well*. Functional units are defined from well characteristics, with the following specifications in our comparative assessment:

Weight agent: the function of fluid density control for drilling of the well, given

1. offshore discharge of cuttings drilled with water-based fluid, or
2. onshore treatment of cuttings drilled with water-based fluid

Loading technology: the loading of 1 metric tonne of cuttings waste onto supply ship at rig and off supply ship at supply base

Drilling fluid system: the function of drilling fluid for drilling of the well, given

1. no local treatment site for cuttings drilled with oil-based drilling fluid, or
2. a local treatment site for cuttings drilled with oil-based drilling fluid

Cuttings end-of-life: the end-of-life treatment of cuttings drilled with water-based/ilmenite fluid for the well, given a hypothetical permit to discharge such cuttings and

1. no additional processes induced by waiting on weather (harsh weather), or
2. a 2 day waiting-on-weather incident

6.2.3 Temporal and spatial considerations

Inventories are produced from the desire to describe the current situation in Norwegian operations. Inventory sources are within years 2000-2005. Emissions are modeled within the long-term perspective of LCA. Impact assessment is modeled similarly, with the exception for global warming emissions where a 100 years timeframe is used. Attempts are made at making the impact results more relevant for the temporal location by adjusting characterization factors according to the local situation. See more on impact approach in the next section.

6.2.4 Impact categories and approach

Several impacts are considered in this study, based on the aim of an overall evaluation perspective. As seen from the list below, a mixed midpoint/endpoint set of indicators is used, as a result of the trade-off between indicators being operational (representative of decision objectives and commensurable over several impact chains) and with reasonable uncertainty. The environmental impacts considered in the comparative assessment are:

- The following non-toxic impact categories included in the CML 2 baseline method (Guinée 2001):
    - Global warming potential (GWP), quantified in units of kg CO2-equivalents according to the Intergovernmental Panel on climate Change (IPCC) 100 years perspective (Houghton et al. 2001)
    - Ozone layer depletion potential (OLD)
    - Acidfication potential (AP)
    - Eutrophication potential (EP), according to European average potentials (Huijbregts et al. 2000)
- Ecotoxicity, quantified in units of kg 1,4-dichlorbenzene (DCB) equivalents according to the CML2 method (Huijbregts et al. 2000) and additional methods to assess metal ecotoxicity. See more on this issue below.
- Human health damage, quantified in units of disability adjusted life-years (DALY)

A few adaptations were made to the original methods in order to increase their validity for selecting best alternative.

Health damage

The Ecoindicator 99 hierarchical approach (Goedkoop and Spriensma 2001) was used for emission-related human health impacts, with some adaptations. Health damage from climate change, ozone layer depletion and radiation were removed due to large uncertainty in the connection between impact chain midpoints and endpoint damage.

Characterization factors for the health damage from respiratory effects were adapted according to local population density for all emissions occurring directly within use and end-of-life of drilling fluid. This includes the transport route for cuttings waste, energy generated offshore for loading equipment and other rig applications, as well as energy for onshore treatment of wastes. The original population density is 80 cap per km^2 in the Ecoindicator 99 method, while it is 1.6 cap per km^2 in the three northernmost regions of Norway combined. Further, fate factors for respiratory effects from offshore emissions are reduced by a factor 2 to adjust for the offshore situation. Final characterization factors are presented in Appendix a, Table XIIb-3.

Health damage from crane-lift risk was included, with characterization factors as described in Chapter 5.2.

Human health damages in DALY were complemented with human toxic impacts in the CML2 baseline method (Guinée 2001). The same adjustments were made for respiratory effect as described above for respiratory effects within the Ecoindicator 99 method. Other human toxic impacts were kept identical to the original CML2 method. Adjusted characterization factors are presented in Appendix a, Table XIIb.

Ecotoxic impacts

The ecotoxic assessment framework of the CML2 baseline method; (Huijbregts et al. 2000), was selected as this is the only existing method within LCA that includes a marine environment. Characterization factors were calculated for ecotoxic fluid components in order to include marine aquatic ecotoxicity of drilling fluid discharged to the marine environment. The fluid components modeled by separate characterization factors are listed in Table 8. Components are anonymized for reasons of confidentiality.

Table 8: Marine aquatic ecotoxic potential (MAETP) for components of water-based drilling fluid. Toxicity is assumed log-normally distributed, described here by the geometric standard deviation (σ) and geometric mean.

Fluid component	MAETP	σ^2	Unit
Base fluid	5,58E-03	20.0	kg 1,4-DCB per kg
Drill-string dope	1,05E-04	5.0	kg 1,4-DCB per kg
Casing dope	3,27E-04	5.0	kg 1,4-DCB per kg

The MAETP characterization factors are calculated from acute toxicity test results and degradation rates reported within the harmonized offshore chemical notification scheme; HOCNF (Novatech 2006), following a simplistic approach. Chronic no-effect concentrations (NOEC) were calculated assuming a general acute to chronic ratio (HC50 to NOEC) of 10 (de Zwart 2002). The concentration affecting 5 percent of the aquatic community by chronic exposure was calculated from the short list of toxic test results using statistical coefficients (Aldenberg and Jaworska 2000). The marine aquatic effect factor is the inverse of this concentration. Uncertainty in final characterization factors is assumed equal to the uncertainty in ecotoxic effect. The large uncertainty is due to the low number of species reported in the test data.

Fate factors are found as the inverse of degradation rates. Risk factors for each component is normalized by the risk factor for the reference substance 1,4-dichlorobenzene (1,4-DCB), giving the characterization factor in kg 1,4-DCB per kg component (Huijbregts et al. 2000).

The implementation of uncertainty for the MAETP of fluid components deserves some discussion. There are few species reported in the original data. Moreover, the rather crude approach applied for the acute-to-chronic extrapolation implies that effect factors carry significant uncertainty. Furthermore, substances are assumed fully water-soluble and no fate modeling is performed. Given the importance of the marine environment as a decision objective, the fluid component ecotoxicity is the only impact issue modeled with uncertainty in our assessment.

Life-cycle inventories and impact assessment results are compiled using commercial software (SimaPro, PRé Consultants 2007). The software does not allow modeling of uncertainty in characterization factors. Uncertainty in impact assessment is achieved by setting the uncertainty in emission of these components equal to the estimated uncertainty in characterization factors, thereby offering an indication of the potential influence of the components on the resulting ecotoxic assessment.

Metal ecotoxicity

Alternatives for consideration of metal ecotoxicity were found outside the LCA literature. Norwegian threshold limits have been published for marine sediments (SFT 2007) and soil waste management (MD 2004). These offer an alternative source of relative scores for metal ecotoxicity by marine sediment deposition and onshore terrestrial deposition.

Given the non-degradable nature of metals, effect factors may themselves be used as impact assessment indicators for metals ecotoxicity. Effect factors are supplied by the IPPC BAT reference document (EIPPCB 2005).

A further option is to base the relative score for metal ecotoxicity on political priority substances. As a simple approach we assign relative weights to metals depending on the priority class in Norwegian policy documents (SFT 2004).

Alternative characterization factors for metal relative ecotoxicity are summarized in Appendix a, Table XIIc-5, extracted from the sources listed above.

Non-toxic impacts
Non-toxic impacts were treated analogous to the approach described above for health damages in the Ecoindicator 99 method. As there is no damage modeling for the non-toxic impacts, the changes only relate to the fate factor for air emissions for acidification and eutrophication. Final characterization factors are summarized in Appendix a, section XIIb.

Ecotoxicity of barite
Adjustments must be made for the implementation of barite emissions in the original inventory data for consistent treatment of barite human and ecotoxicity. The original inventory assumes complete release of barium (Ba) in barite ($BaSO_4$) discharged offshore during drilling operations. Since fossil energy systems are important to many product systems, ecotoxicity of drilling operations in the background ecoinvent system appear as important for the entire life-cycle ecotoxicity in the product systems considered here.

The assumption of complete dissolution of barite was replaced with release of geoavailable metal in barite, as described in Section 5.3. This was implemented in the software used for inventory and impact assessment (SimaPro, PRé Consultants 2007) by changing the characterization factor for barite. Factors were changed for marine aquatic ecotoxicity, freshwater ecotoxicity, terrestrial ecotoxicity and human toxicity. Final characterization factors are presented in Appendix a, section XIIa.

6.2.5 System boundaries

A generalized flow-sheet for processes in the life-cycle of offshore drilling fluid technology is outlined in Figure 18. Separate sheets are given for each of the comparative assessments in Chapter 6.4.

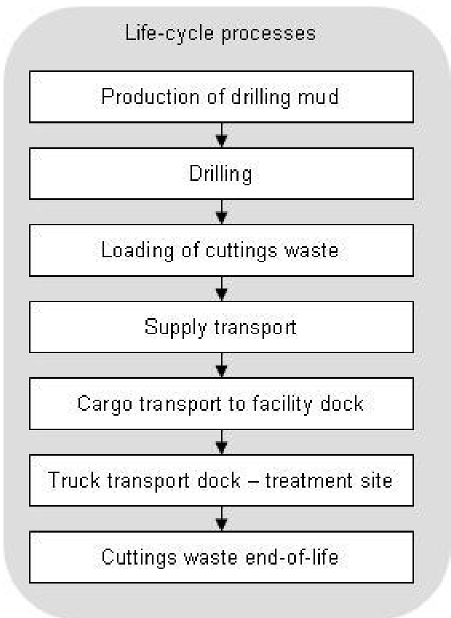

Figure 18: Generalized flow-sheet for the comparative life-cycle assessments.

Foreground focus
The system that is considered includes production, use and end-of-life of technologies from a comparative perspective. Infrastructure is included for all processes. A foreground system focus is maintained. Inventories are therefore not linked to input/output analysis by hybrid analysis. The reasoning behind this approach is twofold. The first reason is that process-LCA is considered to provide the necessary comprehensiveness for relative comparison. The background economic system is similar for all alternatives, and in many of the comparisons the use of the background system is identical. The second reason is related to the eventual users of the results. It is uncertain whether the additional resources required for hybrid analysis provide additional information to decision-makers. Hybrid inventories offer increased comprehensiveness but do not allow localization of impacts. Offshore activities are regulated in a region-oriented perspective. This is apparent, e.g., in the impact assessment undertaken prior to opening the Barents Sea for oil and gas exploration (OED 2002). Inventories are therefore modeled on a process basis only.

System boundary issues
Occupational exposure to hazardous chemicals is not included in this assessment as such information is not available. Nonetheless, exposure to chemicals in the work environment is becoming a hot topic in the policy debate, particularly for offshore installations, and should be an issue for further analysis of drilling technologies.

A second issue not included in the system is the production of waste water, termed slop water. Slop water consists of used tank-wash water, rainwater collected from the rig deck, chemical residues and other liquid wastes. As was outlined in Table 3 (Section 3.5), the slop production may amount to considerable volumes. To some extent, different drilling fluid systems produce different slop volumes. Waste logistics and treatment of slop water is excluded here, reasoning that the difference for the various alternatives is relatively small and information regarding difference in slop water composition due to fluid alternatives is unavailable.

Following the same line of argument as for slop water, rig energy use is excluded except for rig energy related wait-on-weather incidents. Oil-based fluids are generally described as offering faster drilling speeds compared to water-based fluids; i.e., increased rate-of-penetration (ROP), see section 3.3.1. The energy requirements for drilling with oil-based fluids may therefore be lower compared to a water-based fluid. These claims are hard to prove as the fluid alternatives to some extent are used for various applications. External factors such as down-hole formations and solids control efficiency contribute to the effect. Energy use is therefore assumed equal for all fluid systems.

The water-based fluid system assessed here contains chloride salts (Lykling Berge 2004). Chloride salts pose an environmental risk when leaching into a terrestrial environment, but carry no risk in a marine environment. Salt leaching has been identified as an important issue for the onshore treatment of cuttings drilled with water-based fluid (Linjordet et al. 2004). The treatment facility for water-based cuttings is located close to the marine environment, and effects from salt leaching from cuttings are considered to be of small scale for the particular site considered here. While salt toxicity is an important issue in the comparison of end-of-life treatment for cuttings drilled with water-based fluid it is not considered here.

As has been stated earlier, the purpose is to conduct a comparative assessment. As such, the drilling rig and other constant components of the drilling operation can be considered outside the system boundaries.

6.2.6 Allocation rules

Allocation is the partitioning of environmental impacts between products for processes with multiple outputs (ISO 2006). Allocation by weight is applied for all production processes. Benefits are accredited by system expansion for waste treatment processes that generate products by recycling.

Weight allocation implies that in the case of a process providing multiple products, environmental impact is scaled according to the relative mass of product outputs.

Ascribing environmental credit to byproducts is usually referred to as allocation by system expansion (see, e.g., Tillmann 2000) or substitution (Guinée 2001). An alternative interpretation of the practice of system expansion is that it reduces net input when subtracting for materials sent to recycling. This applies when byproducts from the production system may replace one or more of material requirements. An example is illustrated in Figure 19 for the case of treatment of drilling waste containing oil-based drilling fluid. The waste treatment process extracts base oil from wastes, available for use in drilling fluid or other applications of the material. Hence, total requirement for base oil is reduced. Inclusion of this effect requires use of allocation by system expansion.

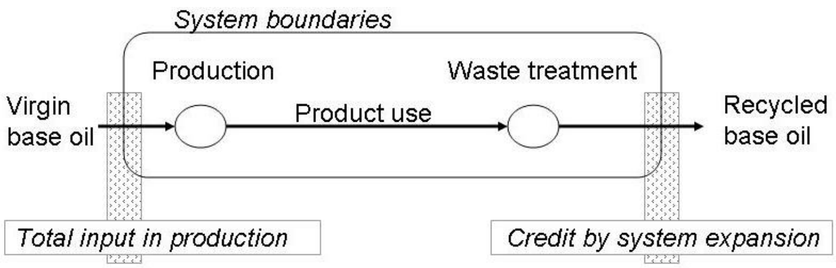

Figure 19: System boundaries and allocation of waste treatment by system expansion

6.3 Inventory assessment

Complete inventory tables are provided in Appendix a, with overview tables in appendix section I. The reader is referred to these for data and references. Main approaches and information sources are summarized in the following sections.

6.3.1 Cuttings and drilling fluid budgets

The production of cuttings waste and drilling fluid consumption was estimated for the reference well separately for water-based and oil-based fluids using common engineering practice (Jensen 2007; Omland 2007). Complete waste and drilling fluid budgets are presented in Appendix a, Tables II-1 through II-4.

6.3.2 Production of drilling fluid

Fluid system specifications were estimated from various sources (Lykling Berge 2004; Paulsen et al. 2005; Omland 2007). Fluid composition was balanced against the density of barite and ilmenite according to common engineering principles (Bourgoyne Jr et al. 1991). Inventories for fluid production are given in Appendix a, Table IV-1 (non-aquous/ilmenite fluid), Table IV-2 (water-based/ilmenite fluid), Table IV-3 (oil-based/barite fluid), and Table IV-4 (water-based/barite fluid).

Substance information was retrieved by combination with material safety data sheets (Fosse 2007). Substance inventories were then mapped against production processes in the ecoinvent database (Frischknecht et al. 2004).

Inventories were compiled for the production of weight agent minerals from reported inventories (Pettersen et al. 2002) and company specific emissions and resource use data (SFT 2007). Inventory results for ilmenite are summarized in Appendix a, Tables Va-1 through Va-4, and for barite in Tables Vb-1 through Vb-5.

Drilling grade ilmenite and barite are commodities traded world-wide. Ilmenite used in Norway is, however, produced in Norway by Titania AS (Sokndal, Norway). Inventories for the production of barite were assembled assuming barite mining in Morocco, with refining of barite at Norbar Minerals AS (Tananger, Norway). These are considered the most probable production routes for barite and ilmenite used on the Norwegian sector.

6.3.3 Transport and energy

Fuel use for the marine transport operations were collected based on a recent drilling operation in the Barents Sea for the supply vessel and cargo transport to local waste treatment site (Folkvord 2006). Fuel use for the cargo transport to Mongstad (the selected non-local treatment site) is estimated from general cargo transport fuel use (Magerholm Fet et al. 2000). Emission inventories for all ship operations were estimated from fuel consumption using emission factors from Cooper and Gustavsson (2004). Infrastructure for ship transport operations were estimated by correlating fuel use with marine transport operation inventories in ecoinvent.

Generic database sources, ecoinvent (Frischknecht et al. 2004), were used for the onshore truck transportation processes.

Fuel use data for offshore and onshore energy generation was extracted from various sources (Lykling Berge 2004; Folkvord 2006). Emission factors and infrastructure were extrapolated from assumed similar ecoinvent processes using fuel consumption data.

Complete life-cycle inventories for the transport operations are presented in the Appendix, Tables VII-1 through VII-6. Unit processes are described in Appendix a, Tables VIII-1 thruogh VIII-6

6.3.4 Treatment of cuttings waste

Leaching data was estimated according to modeled contents of ilmenite and barite in wastes, using geoavailable metal as described in Section 5.3.

The thermo-mechanical cuttings cleaning technology was modeled for treatment of cuttings drilled with oil-based fluid (Soilcare 2007; Thermtech 2007). Life-cycle inventory for treatment of cuttings drilled with oil-based/ilmenite fluid is presented in

Biological degradation was modeled for treatment of cuttings drilled with water-based fluid, according to the approach used at the Stormoen site (Barlindhaug 2006). Inventories for onshore treatment of water-based cuttings waste are presented in Appendix a, sections IXb (water-based/ilmenite fluid) and IXc (water-based/barite fluid).

Inventories for offshore discharge of cuttings drilled with water-based fluid contain only leaching of metals, emission to the ocean of other ecotoxic fluid components, and emissions to air from degradation of organic components. Inventories are presented in Appendix a, section III (content of fluid components in cuttings waste), and section IXa (inventory of emissions from offshore discharge). Ecotoxicity for components is presented in Appendix a, Table XIId-1.

6.3.5 Loading technology

The hydraulic system was used at a recent well in the Barents Sea, and we base our fuel use for this system upon the experience made for this well (Folkvord 2006). Production of the hydraulic system was compiled from information received from the equipment supplier (Samuelsen 2006). Complete inventories for loading systems are given in Appendix a, Tables VI-1 through VI-3.

No production was included for the crane-lift alternative as no additional installation is required. Fuel use was assumed same as what has been reported in US waters (EPA 1999).

6.3.6 Inventory quality

Numerous sources are used to compile the inventories, with differences in the quality of information. Compared to our background database, ecoinvent, some processes are modeled with high precision while others have been compiled from weaker sources. Uncertainty is assigned for each inventory, representative of the uncertainty in:

- inventory uncertainty, i.e., uncertainty in entry value
- inventory completeness, i.e., coverage of all significant interventions
- quality of the link with the background system inventory database, i.e., ecoinvent (Frischknecht et al. 2004)

Inventory quality is summarized in Table 9. The rightmost column indicates an overall judgment of quality of the respective inventories, representative of the all above issues and thereby the value of each inventory for decision support. The following ranking is used: high – good – medium – low.

Table 9: Quality of process life-cycle inventories

Process	Inventory completeness	Inventory uncertainty	Quality of ecoinvent match	Overall LCI quality
Ilmenite production	Good	Medium	Good	Medium
Main issue: direct emissions (medium quality, restricted to reported substances)				
Barite production	Medium	High	Good	Medium
Main issue: direct emissions (low quality, no direct emissions)				
Production of fluid components	High	Medium	Good	Good
Main issue: matching substances to ecoinvent materials (generally good fit)				
Crane-lift system	High	Low	High	High
Main issues: fuel use (high quality)				
Hydraulic pump system	Good	Medium	Medium	Medium
Main issues: fuel use (measured for specific case, high quality), construct (low quality)				
Service vessel	High	Low	Good	High
Main issue: fuel use (measured for specific case, high quality)				
Cargo vessel: Balsfjord	High	Low	Good	High
Main issue: fuel use (measured for specific case, high quality)				
Cargo vessel: Mongstad	High	Medium	Good	Medium
Main issue: fuel use, estimate of general cargo transport (Norway)				
Truck	High	Medium	High	High
Main issue: fuel use, estimate of general truck (Europe)				
Waste treatment, OB cuttings	Medium	High	Medium	Low
Main issues: fuel use (specific to technology, good quality), leaching (modeled from geoavailable metal; high uncertainty), additional inputs and outputs highly uncertain				
Waste treatment, WB cuttings	Medium	High	Medium	Low
Main issues: fuel use (set at zero due to lack of information), leaching (modeled as general process; high uncertainty), additional inputs and outputs highly uncertain				
Offshore discharge, WB cuttings	High	High	Not	Medium
Main issue: leaching (modeled as general process; high uncertainty)				

6.4 Impact assessment

Results from the comparative assessment are presented below. Note that notation in figures may differ from that used in the text for ease of reading the figures. This includes acidification (AP), eutrophication (EP), ozone layer depletion (OLD), freshwater ecotoxic potential (FWT), marine aquatic ecotoxic potential (MAT), and terrestrial ecotoxic potential (TET), and human toxicity according to the CML2 method (HT). Human health impacts assessed wit the adjusted Ecoindicator 99 (hierarchical) method is referred to as DALY in the figures. Simplified notation is also used for the alternative metal ecotoxicity assessment approaches; marine sediment risk (MSR), soil limit values (SLV), effect factors as outlined by the BAT reference document (BREF) and metals listed as priority substances (PM nr).

Results presented here are intended for comparative assessment. Processes identical to both systems under consideration are therefore excluded.

6.4.1 Weight agent mineral

Finely ground minerals are used in drilling fluids to increase fluid density. Two alternatives are considered here: ilmenite and barite. Barite has been the dominant mineral agent for weighted drilling fluids. However, the potentially hazardous metal barium (Ba) constitutes parts of the mineral structure of barite. As an alternative, ilmenite is therefore receiving increasing interest, particularly in operations undertaken in sensitive areas. See more information in section 3.3.3. The process flow-sheet for the comparison of the two alternatives is presented in Figure 20.

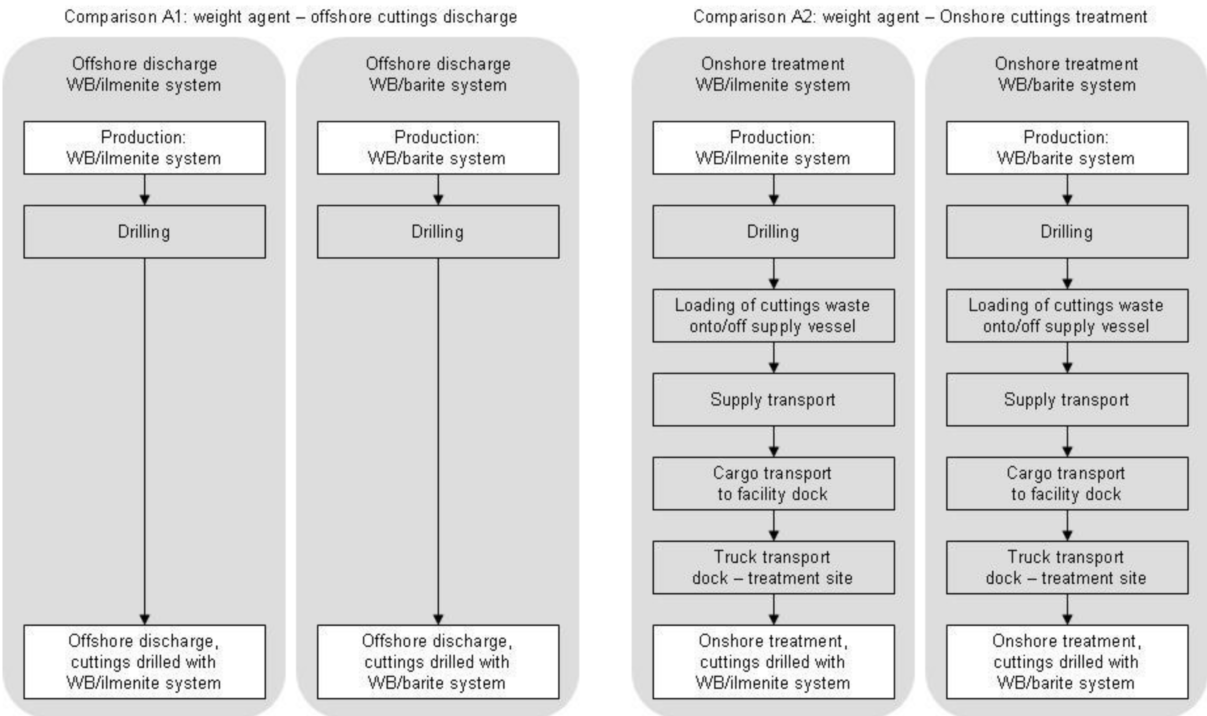

Figure 20: Process flow-sheet for the comparison of ilmenite and barite. Processes with white-fill are modeled in the comparison.

As outlined in the flow-sheet, the comparison of the two alternatives includes two end-of-life options for cuttings waste: onshore treatment or offshore discharge. A just comparison of the two minerals requires that also production is included as the choice of mineral affects the composition of fluid with regards to the other fluid components. Barite has a lower specific gravity. The larger volume relative to ilmenite means that less of the other fluid components is required per volume of drilling fluid.

The toxicity of ilmenite and barite during waste treatment is the major decision objective for this comparative assessment. Metal ecotoxicity is therefore discussed as a separate issue. Challenges and possible solutions to assess the leaching potential of mineral wastes are discussed in section 5.3. The results presented here use the geoavailable metal as basis for leaching potentials.

Uncertainty analysis is only implemented for the leaching stage as the production system is identical for the additional contents in fluid besides minerals. We know that the barite alternative requires less production of unweighted drilling fluid and it is therefore not necessary to model this comparison with Monte Carlo simulations. Furthermore, the motivation for recommending one or the other is ruled mainly by the consideration of ecotoxicity, which is dominated by the leaching of metals from weight agents.

Comparative life-cycle impact assessment results for ilmenite and barite are presented in Figure 21 for the offshore discharge end-of-life and Figure 22 for the onshore treatment end-of-life. Both figures show the effect of the different density of barite and ilmenite, but main differences are related to the production of weight mineral and leaching potentials.

Starting with the offshore discharge scenario; see Figure 21, the advantage for barite in production of other fluid components is offset by the higher energy intensity of barite production. Ilmenite is considered the best option by all impact categories except marine aquatic ecotoxicity (MAT). Leaching inventories assessed by the CML2 ecotoxicity methods favor barite for marine aquatic ecotoxicity. Ilmenite is indicated as the better option for all other toxic impacts, and also for human health damage by the adjusted Ecoindicator 99 method.

For the onshore treatment of cuttings wastes, the results for the toxic categories are changed; see Figure 22. The conclusion for the ecotoxic comparison is dominated by leaching from wastes. The CML2 method considers barite as the preferred alternative for freshwater (FWT) and marine aquatic ecotoxicity (MAT), as well as terrestrial ecotoxicity (TET). Ilmenite maintains the position of best alternative by human toxic assessment according to the CML2 method, and also by human health damage assessment with the adjusted Ecoindicator 99 method.

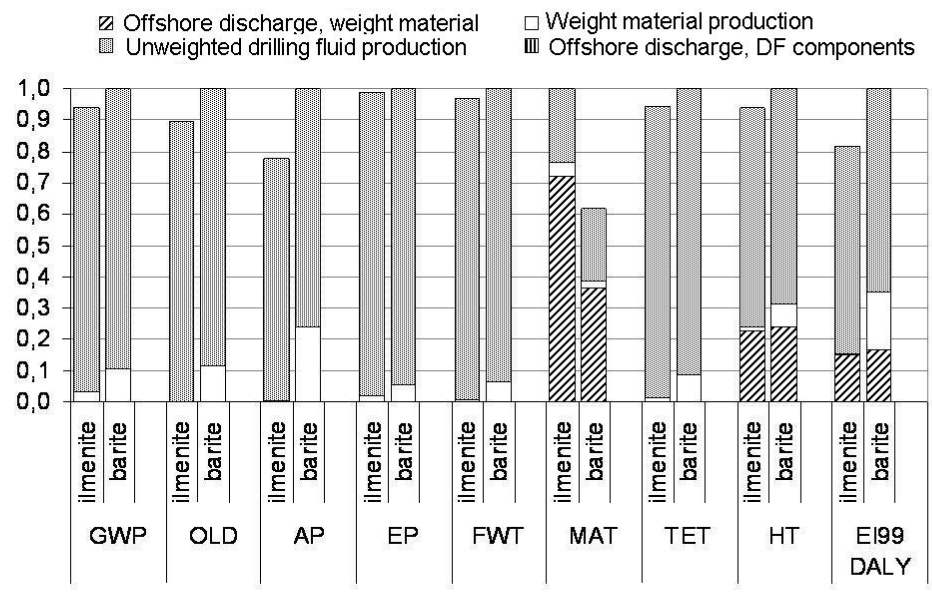

Figure 21: Comparative life-cycle assessment – barite and ilmenite with offshore discharge of cuttings waste. DF = drilling fluid. Results are scaled with the largest impact set equal to 1.

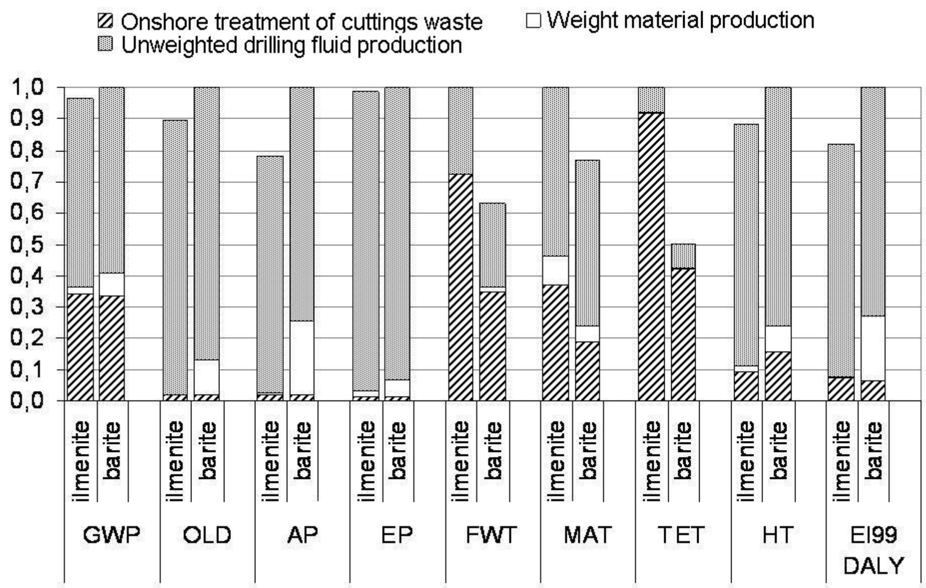

Figure 22: Comparative life-cycle assessment – barite and ilmenite with onshore treatment of cuttings waste. Results are scaled with the largest impact set equal to 1.

Ecotoxicity, and marine aquatic ecotoxicity in particular, is the major decision objective for the selection of weight agent. We shall therefore consider the ecotoxicity of leaching from minerals attached to cuttings waste in more detail. Impact results by application of different metal ecotoxic assessment methods are given in Figure 23. Characterization factors and source references are listed in Appendix a, Table XIIc-5. Looking only at the end-of-life for the mineral alternatives, we see that the different methods emphasize different metals, thereby offering different preferences for the comparison of minerals. For an offshore discharge solution for wastes, the CML2 marine aquatic ecotoxic potentials favor barite, while marine sediment limit values (MSD), effect factors (BREF) and policy measures (PM) recommend ilmenite.

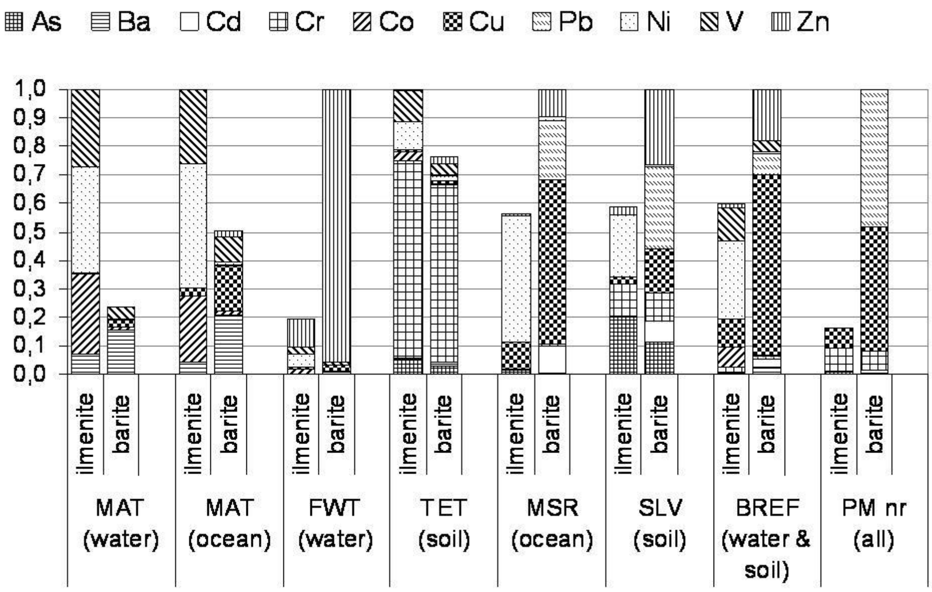

Figure 23: Comparative assessment of metal leaching from barite and ilmenite by use of available impact assessment methods. Recipient is indicated in parenthesis. Results are scaled with the largest impact set equal to 1.

For the onshore treatment scenario, the CML2 method prefers barite for marine aquatic and terrestrial ecotoxicity, but ilmenite is the best solution for freshwater aquatic ecotoxicity. The latter result is dominated by Zn emissions. The alternative metal assessment methods all favor ilmenite; policy measures (PM), soil limit values (SLV) and effect factors (BREF).

The impact assessment does not yield a unison winner. Therefore, leaching inventories are assessed in more detail to see if dominant alternatives may be discerned from metals that the approaches list as most important. Results from this comparison are summarized in Table 10.

Table 10: Ecotoxic assessment of metals leaching from weight agent minerals. Discerning outcomes by inventory Monte Carlo are shaded. Significance is assigned for each method to metals representative of >1 % of total aggregated ecotoxicity for either mineral.

			Significant offshore		Significant Onshore	
Metal	Winner (by % of outcomes)	Priority substance	MSR	MAT	SLV	TET
As	Barite (68%)	Yes	Yes	-	Yes	Yes
Ba	Ilmenite (81%)	-	-	Yes	-	Yes
Cd	Ilmenite (98%)*	Yes	Yes	-	Yes	-
Cr	Barite (58%)	Yes	-	.	Yes	Yes
Co	Barite (91)	-	-	Yes	-	Yes
Cu	Ilmenite (100%)*	Yes	Yes	Yes	Yes	Yes
Pb	Ilmenite (99%)*	Yes	Yes	-	Yes	Yes
Ni	Barite (100%)*	-	Yes	Yes	Yes	Yes
V	Barite (82%)	-	-	Yes	-	Yes
Zn	Ilmenite (90%)	-	Yes	Yes	Yes	Yes
Best option By fraction of discerning metals		Ilmenite (3/3)	Ilmenite (3/4)	Split (1/2)	Ilmenite (3/4)	Ilmenite (2/3)

MAT = marine aquatic ecotoxic potential (CML2); MSR = marine sediment risk values (SFT marine sediment quality guidelines); TET = terrestrial ecotoxic potential (CML2); SLV = soill limit values (SFT soil quality guidelines)
* Discerning comparisons, with 95% confidence (10,000 Monte Carlo runs)

Beginning from the left, the first column in Table 10 is a list of the metals included in the assessment. The next column offers the best mineral alternative for this particular metal, and the percentage of outcomes that favor this winner. Metal-by-metal comparisons that offer conclusive results using a strict 95% confidence selection criterion are indicated by a star. These metal rows are shaded in the table. The first method that is combined with the metal-by-metal comparison is policy measures; column three from the left. Priority substances listed in policy documents are indicated. By application of policy measures to characterize leaching potential from ilmenite and barite, ilmenite is the preferred alternative for three out of four metals (Cd, Cu, Pb). The last metal is not included in the priority list. Policy measures thereby indicate that, for the metals for which we may conclude that the minerals show a difference in leaching potential, ilmenite is the best option.

Using the same approach to the other ecotoxic assessment methods does not offer unison dominance.

6.4.2 Loading system

Crane-lifts are a major driver for accidents on offshore rigs and occupational safety is an issue of priority to operators and regulators. A hydraulic pump system has been developed to replace the use of cranes to load cuttings waste off rig. The system has been used in several operations in the Barents Sea. The main objective for recommending either the hydraulic pump system or the use of crane-lifts is their preference in terms of overall impact to human health. The hydraulic pump system requires production of the unit, and has a higher fuel use per loaded tonne of cuttings compared to use of cranes. The question that must be answered is if the additional energy and infrastructure required for the hydraulic pump is justified by a lower human health impact. A process flow-sheet for the two alternatives is given in Figure 24.

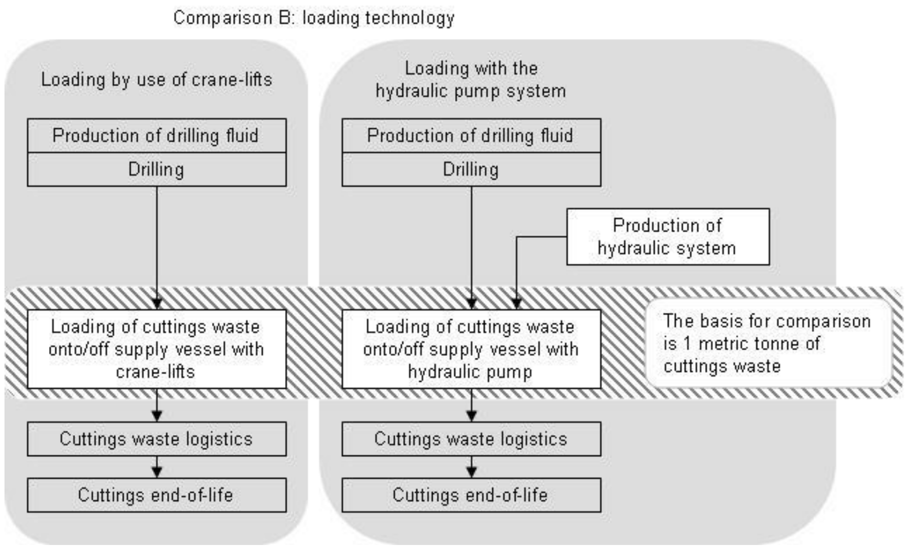

Figure 24: Process flow-sheet for the comparison of loading systems. Processes with white-fill are included in the assessment.

Before we consider the health issue separately, we present results from the comparative assessment of the two options for a greater set of impact categories in Figure 25. As indicated, the hydraulic system requires more energy during operation. Given that the energy source is the same for both systems; i.e., offshore diesel generators, the direct emissions from hydraulic system dominate over those from crane-lifts. This effect applies to energy related impact categories and is increased when production of the hydraulic pump is included. The toxic impacts from the hydraulic system are mainly related to production processes.

The results presented in Figure 25 are in favor of crane-lifts for all impacts besides human health damage. The expected risk of health damage by use of crane-lifts is included in the assessment of impacts to human health; see Figure 25a. From this overall perspective on human health damages, the pump system is the best option due to the health benefits from avoided accidents. A straight forward Monte Carlo simulation on the inventory, while keeping characterization factors restricted to their original value, confirms this conclusion beyond a criterion of 95% confidence.

The uncertainty in the characterization factor for human health damage from crane-lifts has a probability distribution attached. Results above are found by application of the mean characterization factor of $4.0 \cdot 10^{-6}$, with cumulative percentiles $P_{2.5;}$ $P_{50;}$ $P_{97.5;}$ = $5.4 \cdot 10^{-7}$; $2.8 \cdot 10^{-6}$; $1.5 \cdot 10^{-5}$; see Chapter 5.2. Applying the lower bound of the 95% interval of confidence for the expected health damage from crane-lifts, the preference for crane-lifts as the best alternative in terms of damage to human health is maintained, although at a reduced degree of 74% of comparisons.

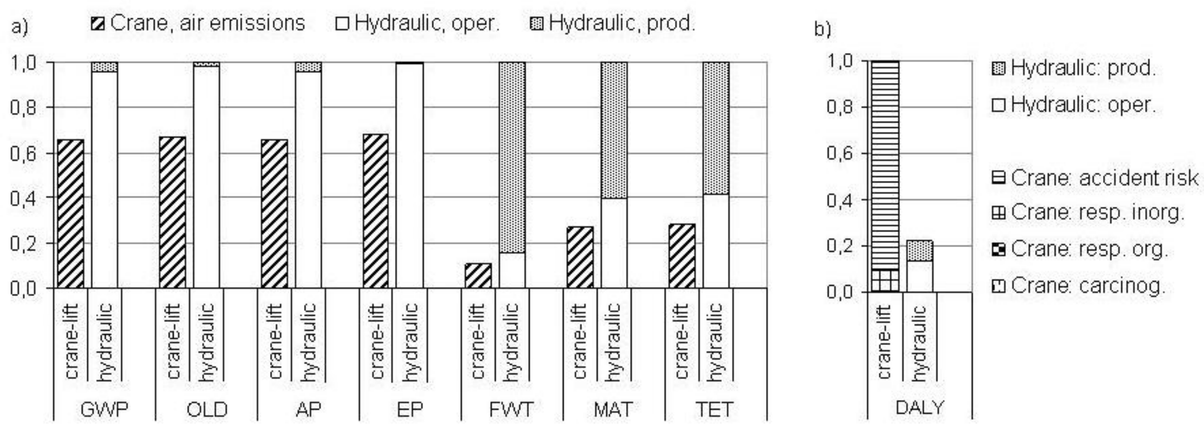

Figure 25: Comparative life-cycle assessment of loading systems for cuttings waste; a) impact assessment split on life-stages, and b) human health damages separated on impact mechanism. Results are scaled with the largest impact set equal to 1. Hydraulic oper. and hydraulic prod. denote operation and production of the hydraulic pump system respectively.

Respiratory inorganics is the major cause of human health damage from loading operations according to both the CML2 method and the adjusted Ecoindicator 99 (hierarchical) method. As previously described, we make adjustments for air emissions occurring offshore and scale population density according to the local situation for all characterization factors related to respiratory effects. However, the characterization factors for respiratory effects are rather uncertain. Source literature lists uncertainty in respiratory inorganic substance effect factors as on the scale of $\sigma^2 \approx 20$ (Hofstetter 1998), as the geometric standard deviation. This corresponds to damages from respiratory inorganics varying within a factor 20 from what is indicated in Figure 25a.

Given that there are two very different impact mechanisms that dominate human health damages from loading operations; crane-lift accidents and respiratory effects from inorganic substances, we investigate their relative sizes in more detail. The median expected health damage per crane-lift is $2.8 \cdot 10^{-6}$ DALY (geometric mean), with log-normal distribution described by $\sigma^2 \approx 5$ (geometric standard deviation). Assigning an analogous distribution to the results for the health damage from respiratory inorganics with $\sigma^2 = 20$, we find that 87% of outcomes support the hydraulic system as the best in terms of DALY. Distributions for DALY from crane-lifts and the hydraulic system are illustrated in Figure 26.

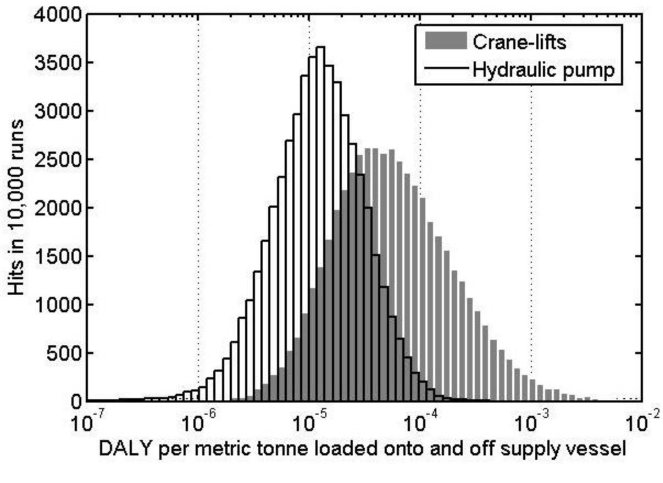

Figure 26: Distribution of disability adjusted life years from using crane-lifts and hydraulic pump to load cutting ont and off supply vessel.

6.4.3 Drilling fluid system

The interpretation of the zero-discharges regime as it is enforced for operations in the Barents Sea has two clear consequences. For the selection of drilling fluid to be used, low-toxic, preferably water-based fluids are to be used from a principle of substitution towards use of less hazardous substances and less risk of harm to the marine environment in case of accidental spills. In addition, the zero-discharges regime requires that cuttings drilled with water-based fluids be transported to shore for treatment. While these two requirements are motivated by a precautionary perspective in relation to marine ecotoxicity, they have consequences which may contradict the policy requirements of applying technology which best protects the environment as a whole. Water-based fluids generally show less capability to maintain well stability, leading to larger volumes of cuttings to be produced compared to oil-based fluids. This in turn requires a larger waste logistic system for bringing cuttings to shore and larger volumes of fluid to be produced per well.

Our third comparison therefore is that of using a water-based (WB) fluid system to drill the well, relative to using an oil-based (OB) fluid system. This is the first comparison that considers true system alternatives, all through from production to end-of-life. The two fluid systems rely on very different production processes. Consistency has been attempted by the use of a single inventory database to cover all production; see Chapter 6.3.1. A simplified process flow-sheet for the two alternatives is given in Figure 27. As seen from the figure, the comparison includes production, waste logistics and onshore treatment. Logistics are included as the two alternatives perform different in terms of waste production. Drilling with water-based fluids generally leads to larger volumes of cuttings being washed out of the well, and the residue of fluid on cuttings is larger for water-based fluids.

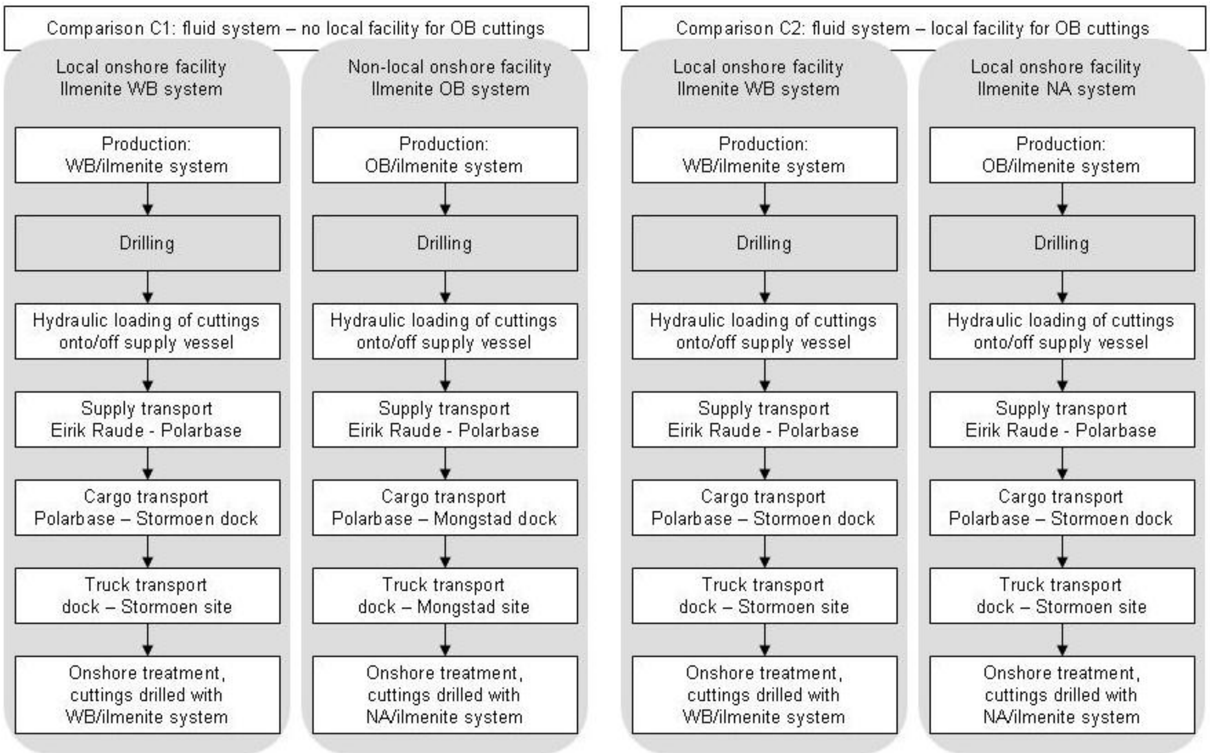

Figure 27: Process flows-sheet for the comparison of water-based (WB) and oil-based drilling (OB) fluid. Processes with white-fill are included in the assessment.

Two end-of-life scenarios are investigated for cuttings drilled with oil-based fluids. The first scenario is the actual situation, where cuttings are transported 1 500 km to

Mongstad for treatment at a certified facility (termed the non-local facility). The second scenario is the hypothetical situation that a facility is located at the same site as the current receiver of cuttings drilled with water-based fluid (the local facility).

Life-cycle impact assessment results for the two alternatives are presented in Figure 28. Results are largely dependent of the extensive transportation requirement for the cuttings drilled with oil-based fluid, termed *Cargo & truck* in the figure. The onshore treatment of cuttings drilled with oil-based fluid is a regeneration process for the oil phase. The negative contributions to ozone depletion and human toxicity are caused by the relatively cleaner process of onshore treatment compared to virgin production of light fuel oil.

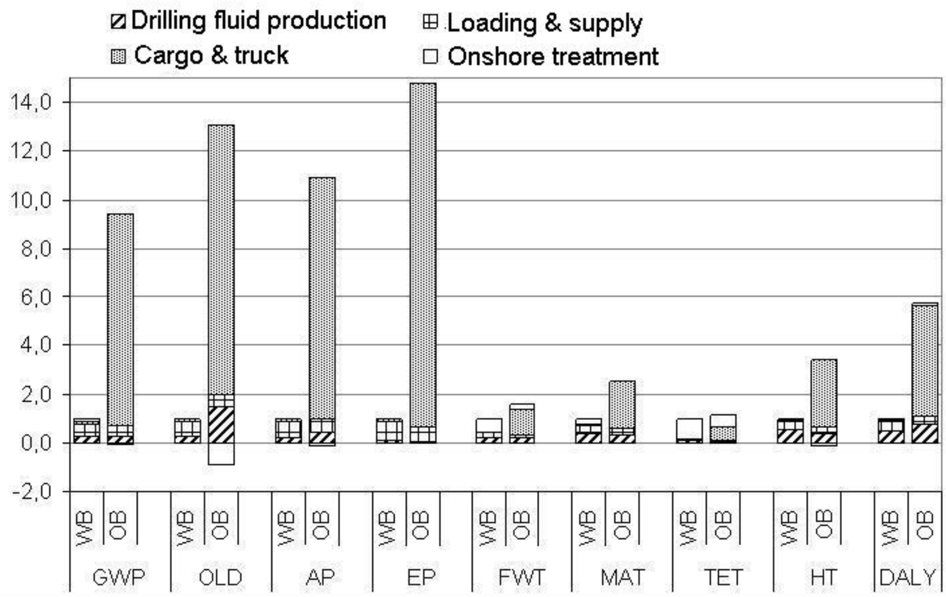

Figure 28: Comparative life-cycle assessment of drilling with an oil-based (OB) and a water-based (WB) fluid system. Impact results are scaled with reference to the performance of the WB alternative, defined as equal to 1. The assessment of disability adjusted life years (DALY) includes risk of health damage from crane-lift accidents, although not particularly significant in this evaluation.

The dominance of impacts from the cargo transport to Mongstad makes uncertainty assessment unnecessary for the comparison of water-based cuttings treated locally and oil-based fluids treated at the Mongstad facility.

Drilling operations currently undertaken in the Barents Sea with oil-based drilling fluid require transportation of wastes to the non-local treatment site for oil-based fluid. We may, however, investigate the sensitivity of our conclusion to the transport distance necessary for oil-based cuttings. Keeping all other parts of the process identical to our previous scenario, we define a hypothetical onshore treatment site for oil-based cuttings at the same location as the current treatment site for water-based cuttings. Monte Carlo simulations are performed for fluid production and the onshore treatment process. Final results are presented in Figure 29.

As presented in Figure 29b and c, the relative performance of water-based fluid for transport operations compared to that of oil-based fluid is a direct consequence of the larger volumes produced by water-based drilling. For the production stage, only a few of the impact categories indicate dominance by either of the alternatives. The exceptions are eutrophication, which is in advantage of the oil-based alternative, and acidification and ozone layer depletion, which discern water-based fluid as the better alternative. The end-of-life stage shows large variations depending on the impact category. Some categories show negative contributions for the oil-based alternative, related to benefits for treatment of oily wastes by including positive credit for regenerated oil by system expansion. The uncertainty in the waste treatment inventory is large compared to the other life-stages, an issue noted also in our inventory uncertainty overview in Table 9.

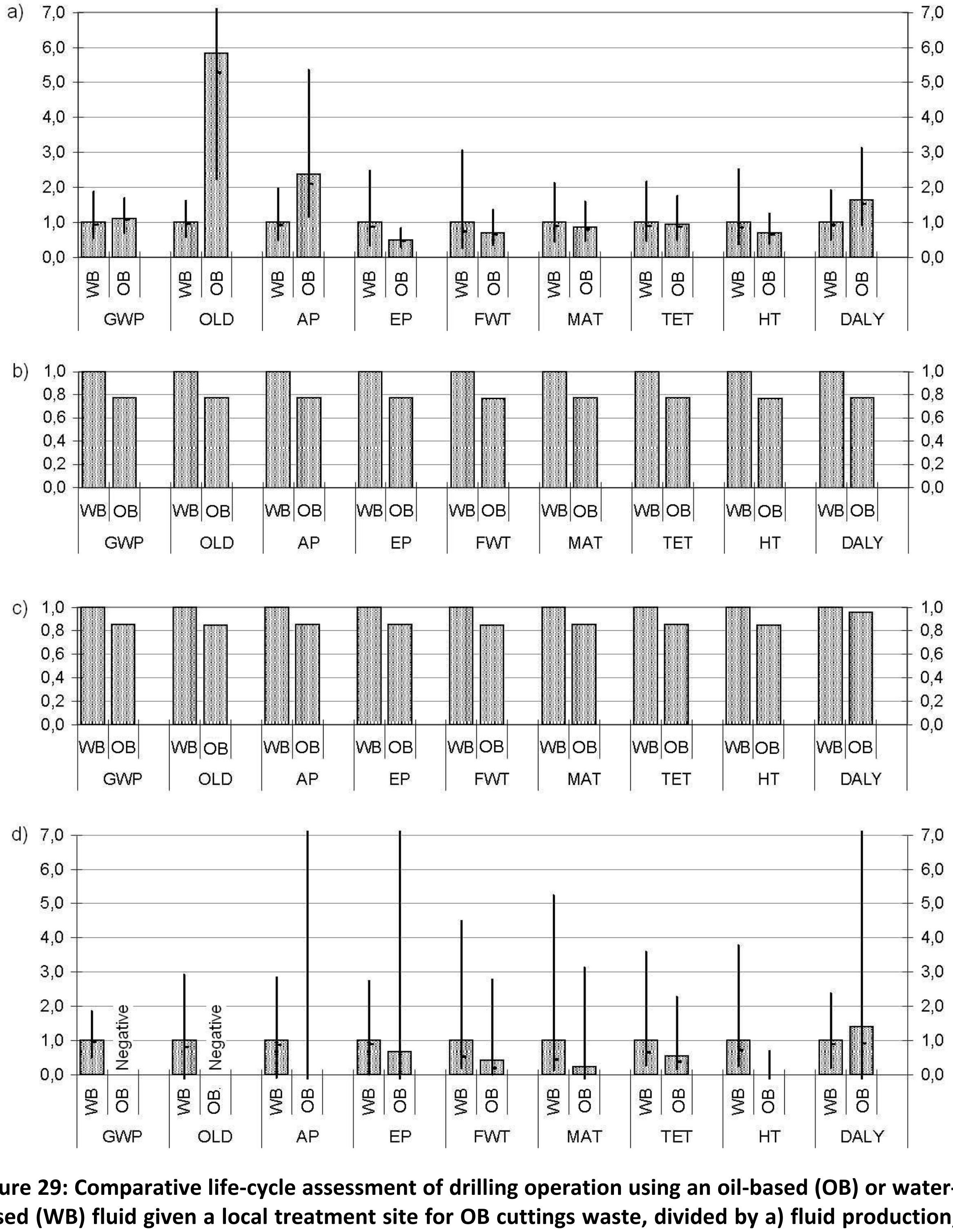

Figure 29: Comparative life-cycle assessment of drilling operation using an oil-based (OB) or water-based (WB) fluid given a local treatment site for OB cuttings waste, divided by a) fluid production, b) loading and supply vessel transport, c) cargo vessel transport and truck onshore, d) onshore treatment of cuttings waste. Results are scaled with the performance of the WB alternative, set equal to 1. Columns in figure a) and d) are mean outcomes by Monte Carlo simulation (1,000 runs), while the outliers represent the 95% confidence interval, with median values indicated by a bar.

The split of life-cycle impacts into system components overlooks the relative size of impacts in each life-cycle stage. Nonetheless, it is an aid in determining best overall alternative as impact categories which do not forward a preferred option by comparison of full life-cycles, may be better interpreted if results for processes close to decision makers are presented separately, such as transport and cuttings waste treatment. Findings and recommendations made based on such a stage-by-stage comparison are summarized in Table 11. By investigating the two alternatives stage-by-stage, the only discernible impact category for production is ozone layer depletion, which indicates water-based as the best alternative. All transportation stages are in favor of oil-based fluid, as this alternative requires a lower volume of wastes to be transported. For the end-of-life stage, the oil-based alternative is discernible as the best alternative for global warming and ozone layer depletion.

Finally, as listed also in Table 11, a Monte Carlo simulation of complete life-cycles for the two alternatives is performed. It is assumed that a local treatment site exists for treatment of oil-based cuttings, as shown in comparison C2 in the flow-sheet in Figure 27. The conclusion from this comparison is that the only discernible impacts, by a criterion of 95% confidence, are global warming, eutrophication and human toxicity. In all three cases the results are in favor of the OB alternative.

Table 11: Comparative assessment of fluid alternatives for oil-based and water-based cuttings treated locally. Preferred alternatives are indicated for each life-cycle stage and over the total life-cycle. Non-decisive outcomes are marked with a hyphen.

	GWP	OLD	AP	EP	FWT	MAT	TET	HT	DALY
Life-cycle stages									
Drilling fluid production	-	WB	-	-	-	-	-	-	-
Loading & supply stage	OB: lower waste production, thereby reduced transportation								
Cargo & truck transport	OB: lower waste production, thereby reduced transportation								
Onshore treatment	OB	OB	-	-	-	-	-	-	-
Total life-cycle	OB	WB	OB	OB	OB	OB	OB	OB	WB
by percent of outcomes (%)	100	65	58	99	89	91	80	96	75
Overall recommendation [a]	**OB**	-	-	**OB**	-	-	-	**OB**	-

[a] Alternatives supported by a 95% confidence criterion over the total life-cycle
WB = water-based; OB = oil-based fluid

6.4.4 Water-based drilling fluid end-of-life

As a degradable, low-toxic fluid, the polyalkylene glycol/potassium chloride drilling fluid system is designed for offshore discharge. Cuttings drilled with water-based fluids can be discharged at site in Norwegian waters outside the Barents Sea area. Operators therefore prefer the use of water-based fluids in many situations due to the less complex end-of-life treatment. Large trade-off impacts are caused by the zero-discharges' requirement for transport to shore of water-based cuttings. A large logistics system must be initiated, and bad weather may amplify the impacts caused by transport operations by the possibility of delays in drilling operations if weather conditions prevent offloading of cuttings stored on rig. And even in the controlled environment of a treatment facility, the onshore treatment of cuttings waste carries environmental impacts.

Our final assessment therefore is the marine aquatic ecotoxic potential caused by offshore discharge of cuttings drilled with the water-based fluid system, compared to impacts that arise from shipping the cuttings waste to shore for treatment at an onshore facility. Our previous assessments were of systems which shared many common processes, while this last assessment involves two systems with largely different impact patterns. An offshore discharge of cuttings waste has no other impact of significance besides those affecting the marine environment, while the alternative system, - bringing the cuttings waste to shore, involves a logistics chain and onshore treatment of cuttings waste. A process flow-sheet for the comparison of end-of-life alternative for water-based (WB) fluid is presented in Figure 30.

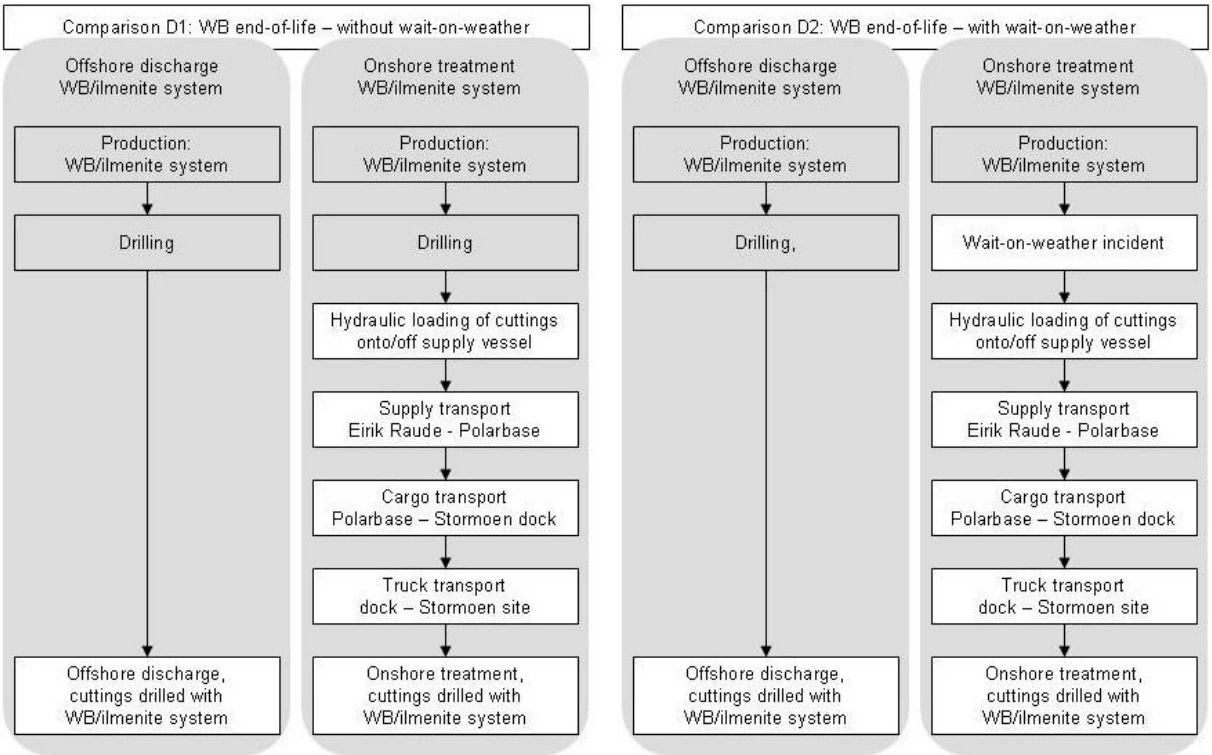

Figure 30: Process flow-sheet for comparison of end-of-life alternatives for cuttings drilled with water-based fluid. Processes with while-fill are included in the assessment.

Because of the skewed impact pattern of the two alternatives it is necessary to establish a common point of reference. Normalization is therefore applied to make the system impacts commensurable; if not in an actual sense then at least in concept. Normalization relates the respective impacts of the two systems to a common scale, here is used the reference of total impacts from emissions in West Europe in year 1995. Impacts induced by treating cuttings drilled with water-based fluid onshore are presented in Figure 31.

Results in Figure 31 are scaled with the impact from the "no wait" scenario set as 1. The normalized results indicate a relatively large significance of global warming emissions, as well as acidification and eutrophication even with the adjusted characterization factors that were applied for emissions from the transport operations. The global warming impacts amount to 603 tonnes of CO_2 equivalents. Compared to the mass flow for the well, this is equivalent to about 10% of the emissions from energy generation on rig (as described in Table 3; Section 3.5).

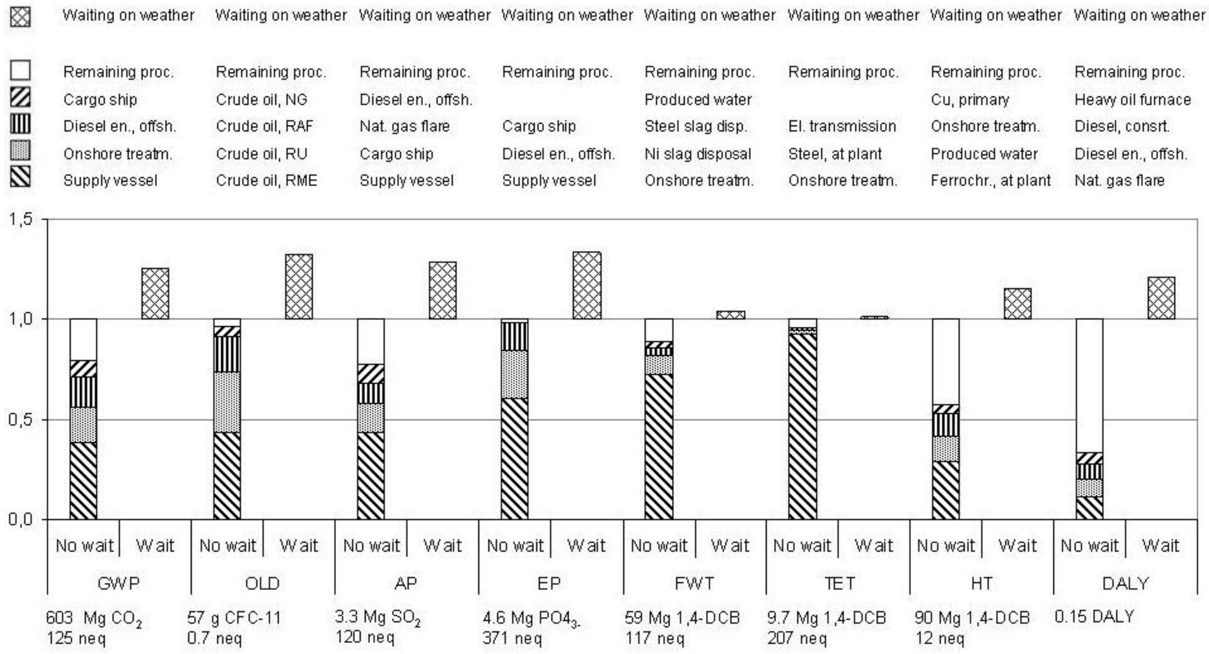

Figure 31: Life-cycle impact assessment of the onshore treatment of cuttings drilled with water-based (WB) fluid. Top contributing processes listed above. Results are scaled to 1 for the "no wait" scenario. Impact scores and normalized results refer to the "no wait" scenario. Normalized impacts are in nano-equivalents (neq; 10^{-9} fraction) of total impacts from West Europe 1995.

The comparison of marine aquatic ecotoxicity (MAT) of the two system alternatives is presented in Figure 32 in terms of best estimate and the contributions made to marine aquatic ecotoxicity from treating the cuttings onshore, with probability distributions presented in Figure 33. Obviously, the leaching from ilmenite offshore has a larger MAT than leaching from ilmenite onshore, and the difference is larger than the potential contributions from the additional processes induced by the onshore treatment. However, there are significant marine ecotoxic effects also from the onshore treatment, and the reductions in MAT from transporting the cuttings to shore are limited to about 40%. This is further reduced to 34% in the case that a wait-on-weather incident arises.

Monte Carlo simulations for marine aquatic ecotoxicity were performed to test the sensitivity to uncertainty in inventories for leaching relative to those of other emissions in the onshore treatment alternative. Cumulative probability distributions are given in Figure 33, separated as marine aquatic ecotoxicity from leaching by offshore discharge and onshore treatment, and for the transport chain to shore. The figure shows clear correlations between the contributions from onshore leaching compared to offshore discharge, which is expected since they are modeled using the same leaching potential. Bringing the cuttings to shore retains some of the metals from reaching the marine environment. A second observation is the much smaller contribution from emissions not occurring from leaching from wastes. This is indicative that the additional transport processes required to bring cuttings waste to shore are less than what is seen as direct emissions from leaching. The purpose of requiring that the cuttings be transported to shore is achieved, i.e., to reduce marine ecotoxic effects. However, it does come at a cost. The trade-off for reducing marine ecotoxic impacts by 35-40% are emissions with relatively large-scale effects, e.g., global warming. Human health impacts from transporting wastes to shore account to 0.15 DALY, equivalent of 0.15 years of life lost. The main part of health damages are related to processes outside the local region, in production of inputs to the transportation and treatment process.

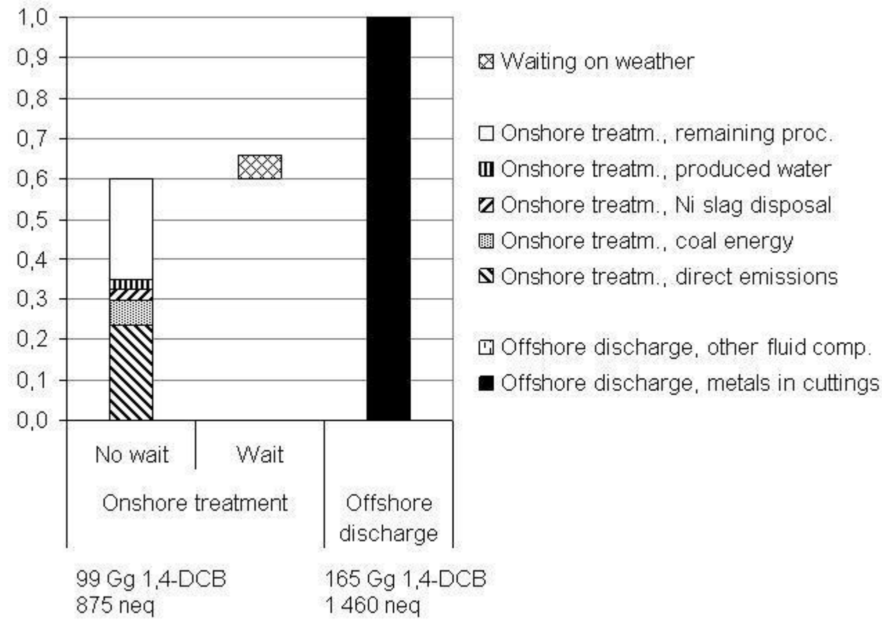

Figure 32: Comparative assessment of marine ecotoxicity of water-based/ilmenite cuttings waste by offshore discharge or onshore treatment. The bars are scaled according to impacts from offshore discharge, set equal to 1. Normalized figures and total 1,4-DCB equivalents for offshore discharge and onshore treatment with no wait on weather only.

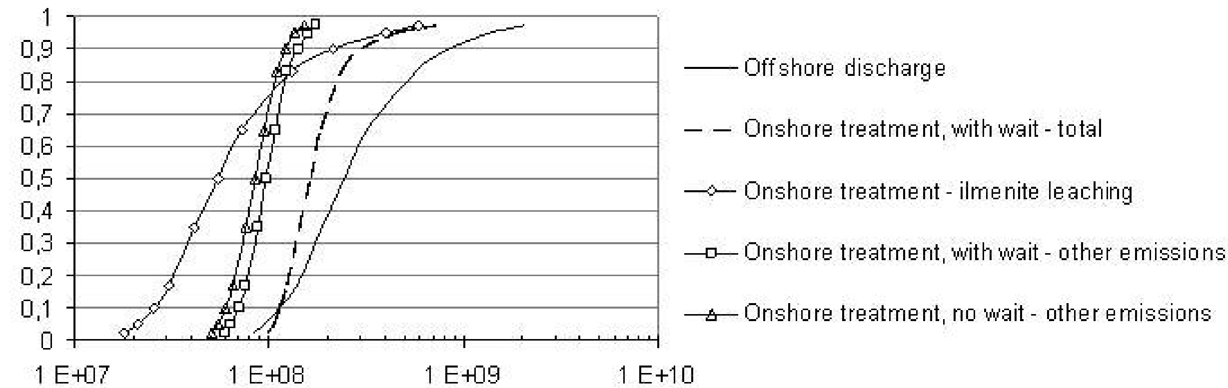

Figure 33: Cumulative probability distribution of marine aquatic ecotoxic potential (kg 1,4-DCB per well) from offshore discharge and onshore treatment of cuttings drilled with water-based fluid.

6.5 Interpretation

6.5.1 Best alternative considering uncertainty in inventories

The above sections discuss results in terms of discernibility and decision objectives. It is also necessary to discuss conclusions related to the quality of the inventories on which they are made, as summarized in Table 9 in Section 6.3.6.

Weight agent mineral

Ilmenite is indicated the best alternative in the production stage, mainly due to the larger transport requirement in the value chain of barite. The production inventories are indicated of medium quality, and do not have the required comprehensiveness for assessment of toxic impacts.

Assessing leaching potential by Monte Carlo simulation identifies only four out of ten metals as having significantly different leaching potentials for barite and ilmenite, applying a 95% confidence criterion. This is an indication that uncertainty in leaching is considerable, but also that leaching potentials may be discerned. The inventory uncertainty table indicates that the uncertainty in the leaching inventories is high, but also that the uncertainty is well documented.

Ecotoxicity caused by onshore treatment and offshore discharge was assessed using several approaches. Neither of the minerals performed as the dominant option by all methods. The ecotoxic impact methods of Huijbregts et al (2000) consider barite the best alternative for marine aquatic ecotoxicity, both by onshore treatment and offshore discharge. All the additional methods used – soil limit values (SLV), marine sediment risk limits (MSR), effect factors (BREF) and priority metal list (PM) – consider ilmenite the best alternative, by offshore discharge as well as onshore treatment. It is difficult to identify an overall best alternative since the methods each indicate different metals as dominant in the relative assessment.

Limiting the leaching inventory to metals for which the leaching potentials are significantly different provides some guidance. From the limited set of metals, ilmenite appears as the best alternative for most metals, although the difference in conclusion for the total aggregated ecotoxicity by the methods remains.

Loading system

The inventories leave no doubt that the continued use of crane-lifts is the best alternative for environmental impacts besides human health. The question is whether the additional processes required for production and use of the hydraulic system outweighs the benefit of a reduced accident risk. The inventory for production of the hydraulic system is of poor quality.

Monte Carlo analysis of combined inventory and impact assessment uncertainty (in Figure 26) supports crane-lifts as the best alternative with a 87% confidence for human health even with the large uncertainty span modeled for respiratory effects.

Reduction of human health damage is the main decision objective for the loading system, for which the hydraulic system is discerned as the best alternative.

Drilling fluid system

Results and uncertainty for the fluid system comparison is discussed for each life stage.

The water-based (WB) fluid offers the only discernible outcome in the production stage and is recommended for ozone depletion potentials in production. This conclusion is sensitive to the quality of the original ecoinvent inventories. Source data for refinery processes are not very recent and this may be the cause of the conclusion towards preference of water-based fluid.

Oil-based fluid produces less drilling waste and is therefore the overall best for all logistic processes independent of inventory quality.

Credits are allotted to the treatment of cuttings with oil-based (OB) fluid for the regeneration of base oil. These shift the conclusion in favor of the oil-based fluid alternative in the end-of-life stage. However, inventories for waste treatment are of overall low quality and may not cover significant impacts, particularly for organic substances with toxic effects. Furthermore, the conclusions based on system expansion by use of the background database (ecoinvent, Frischknecht et al. 2004) are vulnerable to the quality of the database inventories.

The uncertainty indicated for end-of-life processes is high in terms of both calculated impact potentials and inventories (see Table 9 and Figure 29). Still, oil-based fluid is preferred with respect to human toxicity, global warming and eutrophication (with 95% confidence). The results for human health, however, are altered if the adjusted Ecoindicator 99 (hierarchical) method is used (with 75% confidence in water-based fluid as the best alternative). The cause of the different conclusions is the emphasis that the Ecoindicator 99 (hierarchist) method puts on respiratory effects relative to other toxic pathways.

The oil-based fluid is recommended as the best overall alternative due to the better performance in end-of-life stages (transport chain and onshore treatment) and the beneficial products by onshore waste treatment.

Water-based fluid end-of-life
Onshore treatment of drill cuttings retains a fraction of metals in the onshore environment and prevents them from reaching the ocean. The conclusions made here rely on the comparison of the prevented release to the marine environment with emissions related to transporting cuttings to shore and in the onshore treatment process.

Emissions to air induced by transportation and onshore treatment are equivalent to about 10% of the regular emissions from drilling the well (excluding offshore supply; see Table 3). Human health damages are estimated at 0.15 DALY. On the other hand, marine aquatic ecotoxicity is reduced with 34-40% by transporting cuttings to shore rather than discharging cuttings waste at site.

The net benefit in terms of marine aquatic ecotoxic potential by onshore treatment is not insignificant. The associated effects were normalized to total emissions from West Europe, 1995. Emissions to air, leading to global warming, acidification and eutrophication, were found as the most significant effects.

The conclusion regarding the overall evaluation depends on the weighting of marine aquatic ecotoxicity relative to the additional impacts by onshore treatment. Marine aquatic ecotoxic potential is reduced by 34-40% by onshore treatment. Uncertainty in the inventory for onshore treatment is high and completeness in terms of inclusion of all processes is judged as medium (see Table 9). Improvement of the inventory will most likely lead to a further reduction in benefits by onshore treatment due to inclusion of additional inputs to the process.

6.5.2 Boundary issues – inventory comprehensiveness

The conclusions made in the overall comparative evaluation must be interpreted with respect to the issues related to system boundary limits, as discussed in Section 6.2.5. Their potential influence in discussed in the following.

Slop water treatment is a relevant system boundary issue for the comparison of water-based and oil-based fluid. It is uncertain how including slop water would affect the conclusions made in this comparison.

Occupational toxic exposure is expected to be most important for the oil-based fluid alternative. Inclusion of workplace exposure may provide additional input to the evaluation of human health impacts, possibly by altering conclusions in favor of water-based fluid as the preferred alternative compared to drilling with oil-based fluid.

Rig operations were considered equal for all alternatives, also in the comparison of water-based and oil-based fluid. If included, the aspect of drilling speed is expected to shift the impact pattern in favor of the oil-based fluid system for impacts related to fossil energy use (that is global warming, acidification, eutrophication, and respiratory effects).

A foreground focus was maintained in this evaluation, excluding any combination with economic modeling; i.e., hybrid-LCA. The influence of this boundary limit is not expected significant in most of the comparisons, except for the evaluation of end-of-life treatment of cuttings drilled with water-based fluid. A more comprehensive system description, as achieved by hybrid approaches, will further decrease the marine ecotoxic benefit of onshore treatment of cuttings relative to the offshore discharge alternative.

Ecotoxicity of fluid components besides metals in weight agents were not included in the leaching inventories for cuttings waste. The omission of the organics was investigated by sensitivity analysis. The analysis concluded that organics may be a significant contribution to oil-based fluid freshwater ecotoxicity, although not dominant. All other ecotoxic impacts remained unaffected by organic leaching. Details are given in Appendix a, Tables XI-1 to XI-4.

6.5.3 Impact assessment validity

Global warming potential is perhaps the only impact category in life-cycle assessment for which consensus exists. The 100 year time-horizon CO_2-equivalents (as outlined by Houghton et al. 2001) is generally recommended as the midpoint indicator for global warming (Potting et al. 2002).

Eutrophication and acidification in background processes were assessed by application of European average characterization factors, as described by (Huijbregts et al. 2000). The adjustments made for emissions during transportation and combustion offshore and in onshore treatment processes have the effect that processes in the foreground system are reduced in significance compared to those occurring in production processes; the latter containing inventory data from ecoinvent. Differences in impact patterns due to differences in the requirement for transport operations are thereby reduced. These adjustments favor water-based fluid in the comparison with oil-based fluid, and onshore treatment in the comparison with offshore discharge. Acidification and eutrophication are not main decision objectives in this evaluation, although eutrophication is identified as an argument for the use of oil-based over water-based fluid. This argument would be strengthened had Norwegian characterization factors been used since Norway is more sensitive to eutrophication than the European average scenario (Huijbregts et al. 2000).

The validity and significance of the methods applied to assess ecotoxicity – and particularly metal ecotoxicity – have been covered earlier; see the impact assessment section (Section 6.4.1) and paper 3 (Section 5.3).

The disability adjusted life years (DALY) concept provides a framework that allows direct comparison of occupational health damages and damages due to emissions to the external environment. The uncertainty in health damage modeled in Section 6.4.2 shows that it is operationally applicable for this comparison.

Specific local conditions such as areas or ecosystems of particular concern are not considered in this assessment. Land use is also excluded. Local effects such as deposition and coverage of benthic ecosystems are thereby not considered.

6.5.4 The implication of allocation rules

Allocation has been used in several processes related to production, transport and end-of-life. The most important allocation cases in the foreground system are listed in Table 12. Allocation practices have some consequence for the conclusions, particularly for the comparison of fluid systems.

Onshore treatment of cuttings drilled with oil-based fluid is a regeneration processes for the base oil. Regeneration products are assumed to replace light fuel oil. Given the value of the synthetic oil used in drilling fluids, it may be more realistic to assume that a product with higher product requirements is replaced, in which case the credit allocated to the oil-based alternative should be higher than what is estimated here.

Allocation practices for production processes do not play a significant role in the final conclusions. Transport operations are modeled with allocation by load, either assuming a dedicated vessel for the cuttings waste, or by application of generic load factors. This is consistent with the approach in the ecoinvent database (Frischknecht et al. 2004).

Table 12: Allocation practices used in the evaluation

Process	Allocation rule	Judgment of practice
Ilmenite production	Weight	Little consequence, weight mineral is the dominant output by value and weight
Barite production	Weight	Little consequence, weight mineral is the dominant output by value and weight
Supply vessel	100%	Significant consequence. Supply vessel transports all necessary commodities for crew, operations and waste. Drilling waste represents the largest portion by weight
Cargo vessel Balsfjord	Dedicated vessel	Significant consequence. The cargo vessel operates as a dedicated vessel for drilling waste in one direction (100% allocated to the transport of cuttings waste), and returns carrying other commercial goods (100% allocated to the fish feed cargo)
Cargo vessel Mongstad	Generic load factor	Significant consequence. The process is modeled as generic cargo transport, with load factor 60%
End-of-life	System expansion	Significant consequence as credit is allotted for beneficial products from waste treatment. The practice is selected in order to favor recycling of resources

7 CONCLUSIONS

The aim of this thesis was stated in the introduction as *to perform comparative life-cycle assessment of offshore drilling fluid technology alternatives*. To be able to achieve this it was necessary to develop methods for inventory and impacts assessment that allow assessment of attributes corresponding to decision objectives posed by stakeholders to offshore drilling activities. The developments have been applied in overall evaluation of a short list of offshore drilling fluid technology alternatives, based on a reference well located in the Norwegian Barents Sea. Conclusions from the case study and methodological developments are summarized below.

7.1 Best alternatives by overall evaluation of offshore drilling fluid technology

Ilmenite appears as a better alternative than barite for impacts not related to toxicity. Conclusions regarding human toxicity and ecotoxicity are largely dependent of the method used to assess toxic impacts. Uncertainty in leaching potentials is high, although four out of the ten metals modeled show significantly different leaching potentials – three of which are in favor of ilmenite. Dominance of either of the alternatives can not be identified for the toxic impacts due to the different relative ecotoxic potential assigned by the assessment methods used for metals.

Human health is the main decision objective for evaluation of loading technologies, for which the hydraulic system is discerned as the best alternative. Other impacts are all in favor of the crane-lift alternative.

The oil-based fluid system is considered the best alternative by an overall evaluation if cuttings drilled with water-based fluid are treated within same transport distance as the treatment for cuttings drilled with water-based fluid. Main reasons for this are the reduced need for transportation to shore compared to the water-based fluid system and benefits associated with regeneration of oil in cuttings waste. The conclusion is reversed strongly in favor of water-based fluid if oil-based cuttings are shipped to Mongstad, which is the current solution for oily cuttings waste.

The overall benefits to the marine environment by requiring that cuttings drilled with water-based fluid are transported to shore are estimated within 34-40% by process LCA, in terms of marine aquatic ecotoxicity. The additional inputs required for transportation of cuttings to shore and onshore end-of-life treatment are significant. Global warming emissions from these processes amount to about 10% of the total emissions from the rig.

7.2 Scientific contributions

The significance of assuming that pulse emissions may be assessed by steady-state multi-compartments models has been investigated. It was found that the concentration addition approach for species sensitivity distribution (used by Goedkoop and Spriensma 2001) is robust for pulse emissions. The conclusion for the response addition (outlined by Huijbregts et al. 2002; van de Meent and Huijbregts 2005) is sensitive to the background concentration for the substance, or toxic mode of action, that is modeled.

Occupational health damages were estimated for the work-situation considered most affected by changes in drilling technology and most often reported in the accident statistics for offshore rigs, namely crane-lifts. Crane-lifts were identified as the main cause of overall health damages from loading technologies, but not dominant in the life-cycle of drilling fluid technologies.

Current and possible solutions to estimate life-cycle inventories for long-term metal release processes have been reviewed. The study shows that significant improvements can be achieved for leaching inventories for inorganic substances in solid deposits.

Geoavailability seems a promising approach, although large uncertainties remain in the interpretation of metal mobility and geoavailability.

An attempt has been made to implement and interpret uncertainty in a consistent manner in the case study. Uncertainty has been shown to provide valuable input to the interpretation of results, either as indication of alternatives performing equally well, or by enabling results that discern dominant alternatives.

The subject of this thesis, offshore drilling operations, has provided several challenges that could not be met by existing life-cycle assessment methods. The developments have been made within the framework of life-cycle assessment, proving that LCA provides is a versatile backbone. However, the need for methodological development is also proof that few systems can be fully assessed by existing methods alone, and that LCA needs to accommodate adaptations to increase its applicability as a tool for decision support. Life-cycle assessment methods must be applied based on case specific challenges. This relates to inventory estimation as well as impact assessment.

7.3 Other lessons

The concept that LCA shall consist of a standardized approach for inventory and impact assessment and maintained a non-complex method is flawed. Adaptation of methods is a natural way forward for LCA. Life-cycle assessment is moving in this direction, as shown by the updated ISO standard (ISO 2006). Supporting procedural documents are developed for specific LCA applications, such as environmental product declarations.

If LCA is used for decision support, results need to be communicated integrated with the associated confidence in conclusions. Life-cycle assessment is an inherently relative method, either comparing life-cycles or product systems. Results are on the level of impact potential rather than actual damage, mainly due to issues with spatial localization of emission points. Nonetheless, LCA does offer the necessary resolution for identification and discussion of trade-offs between alternative product systems.

The waste hierarchy is often referred to as a guiding principle in waste management. In brief, the waste hierarchy prescribes first removal of waste production, then reduction in waste volume, and finally recycling of wastes before disposal (European Council 2006). As described in Section 6.1.2, alternatives exist that do not produce cuttings (the drilling badger), or require a smaller cuttings volume to be removed from formations (slim-hole drilling). Waste production is also reduced by drilling with oil-based fluid. As shown in Section 6.4.3, the difference in volume of cuttings by drilling with water-based fluid or oil-based fluid is significant for the overall performance of these alternatives. The benefit of oil-based fluid is reduced, and the overall performance shifted strongly in favor of water-based fluid, if transport distance for oil-based fluid is maintained at the current route (i.e., oily cuttings from the Barents Sea shipped to treatment at Mongstad). In other words, waste logistics matter for the validity of the waste hierarchy as a guiding principle.

The waste hierarchy prescribes recycling over disposal of waste resources. Recycling requires that the waste material contains reuse value. Oily cuttings have clear value as the hydrocarbon residue may be regenerated. The commercial value of regenerated oil has lead to several commercial initiatives towards reuse of the oil phase in cuttings drilled with oil-based fluid. Similar value is not seen in cuttings drilled with water-based fluid. The organic phase in water-based fluid consists of compounds selected for their degradability, such as glycols. Separation is not commercially interesting, and the simplest and most efficient treatment is biological degradation of organics. The treatment of water-based cuttings thereby carries no recycling value besides the use of the solid material as landfill cover or filler material or other filler purposes.

The water-based fluid can be characterized as a *designed-for-discharge* solution, i.e., it is designed to be compatible with the marine environment by offshore discharge of cuttings waste. Environmental compatibility is dependent of the end-of-life fate. Salt in the water-based drilling fluid poses no risk to the marine environment, but is an issue for onshore treatment of cuttings waste (Linjordet et al. 2004). The oil-based fluid carries end-of-life value, and as such can be termed a *designed-for-recycling* solution. This is a situation where the requirement of substitution, i.e., that fluids be composed of substances with little environmental hazard, contradicts the principles of the waste hierarchy. Net environmental impact of selecting either base fluid is assessed with life-cycle assessment, and conclusions fall in favor of oil-based fluid if cuttings are treated onshore, i.e., that recycling options should be sought. Waste logistics matter in this comparison. The minimal transport is achieved by offshore discharge. The best overall solution therefore is water-based fluid and its intended end-of-life option, which is offshore discharge.

Technology has been proposed for collection of the cuttings produced prior to installing the riser (SFT 2006), i.e., cuttings in spud sections (see Section 3.2). Not all proposed technologies can be used at all locations. Collection of cuttings from spud sections is energy intensive, and volumes are very large given the large radii of spud sections and the content of sea-water. The results presented in this thesis show the large trade-offs caused by transport to shore. Unless areas of particular interest are expected affected (such as corals), sediment deposit of spud cuttings is the best solution by overall evaluation.

The case of cuttings from spud sections is typical for the issues that have been discussed in this thesis. It is a good illustration of the benefit of a systems perspective. Technology exists that may improve a particular risk aspect, but the improvement comes at a cost to other environmental issues. This is the situation for the loading system that is assessed in Section 6.4.2, where health burden improvements involve increased impact to all other issues, including global warming impacts, toxicity, and acidification. Analogously, onshore treatment of cuttings drilled with water-based fluid reduces impacts to the marine environment but involves transport and treatment processes that carry resource requirements and emissions to air and the terrestrial environment (see Section 6.4.4). It may seem that if wastes are transported to shore, impacts to the marine environment are removed entirely, but life-cycle assessment shows that this is not true. The net reduction is less than half of the marine ecotoxic potential by offshore discharge, at the cost of incurred global warming impacts, acidification and terrestrial ecotoxicity.

7.4 Further research

Several issues have not been considered in this thesis but would be natural subjects in further study of offshore drilling technology or to other applications of an overall evaluation perspective. Such issues include:

- The influence of incentives on increased recycling and reuse. Business incentives are used in offshore contracts today, both for chemicals (Lindland 2006; Paulsen et al. 2006) and safety aspects (Osmundsen et al. 2006). The overall effect of such incentives to increase performance has not been evaluated here, but such mechanisms form a natural part of the industrial ecology field (see e.g., Røine 2005)

- The melding of environmental and technical modeling of offshore solids control technology. Good solids control operation is vital for efficient recycling of fluids and fluid waste characteristics that may be influenced by management decisions. Currently, solid control units are operated with the objective of optimizing fluid properties, possibly at a needless level of fluid loss with cuttings.

- Other offshore drilling technology. Some alternative technologies are listed in Section 6.1.2. A natural extension is the evaluation of slim-hole drilling, i.e., drilling with slimmer well radii. Such an assessment would require the system to be expanded to include steel casings in the assessment.

- Occupational health impacts besides crane-lifts, such as occupational toxic exposure. Commensurable damages to the existing health damage approach may be achieved by implementing separate compartments for the working environment, along the approach taken for dwellings by Meijer et al (2005)

- Additional marine environmental impacts. The evaluation made here of impacts to the marine environment relies on the use of marine aquatic ecotoxicity. Other impacts, such as deposition leading to benthic land transformation and occupation, have not been assessed. Current marine risk assessment models incorporate effects of particulate exposure in the water column (Rye et al. 2006). These mechanisms can be included in life-cycle impact assessment models to extend the understanding of impacts to the marine environment in LCA.

- The consistent use of uncertainty to select best option. Uncertainty in impact assessment is not systematically implemented here due to resource constraints. A framework should be developed to use uncertainty to support decisions based on LCA results.

- Metal ecotoxic effect assessment. Challenges remain with the assessment of metal ecotoxicity, particularly for the marine environment. The current methods assume toxic effect of total dissolved metal, which is flawed (Adams and Chapman 2007). Free-ion based mechanistic models have been proposed, but overlook potentially significant pathways for ecotoxicity. Furthermore, the ocean compartments should be developed as integrated part of exposure models and not just represent the final recipient. There is a clear lack of marine focus in multi-compartment models, see e.g., the unit world model (Harvey et al. 2007)

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Appendix A – Case study tables

Appendix A – Case study tables

No.	Section description
I	Comparative assessments in this study
II	Drilling fluid and cuttings waste budgets
III	Ecotoxic components in cuttings waste
IV	LCI – drilling fluid production
V	LCI – weight material production a) ilmenite production b) barite production
VI	LCI – loading systems
VII	LCI – transport operations for cuttings waste
VIII	LCI – transport operations, unit processes
IX	LCI – onshore treatment of cuttings waste a) onshore treatment of cuttings drilled with oil-based/ilmenite fluid b) onshore treatment of cuttings drilled with water-based/ilmenite fluid c) onshore treatment of cuttings drilled with water-based/barite fluid
X	LCI – offshore discharge of cuttings waste a) discharge of cuttings drilled with water-based/ilmenite fluid b) discharge of cuttings drilled with water-based/barite fluid
XI	Sensitivity analysis – ecotoxicity of organic substances in cuttings waste
XII	LCIA – adjustments made for characterization factors a) barite toxicity b) other adjusted characterization factors c) approaches to assess metal marine ecotoxicity d) marine aquatic ecotoxic potential of drilling fluid components

Functional unit: the function of weight agent for the drilling of the well, onshore treatment of WB cuttings waste

		Mineral	Base fluid	Loading	Waste	Location	Waiting
A1	Ilmenite - onshore	ilmenite	WB	-	onshore	-	-
	Barite - onshore	barite	WB	-	onshore	-	-

Table I-1: Ilmenite onshore inventory

Amount	Unit	Distribution	Process
632	m3	log-norm: σ^2 = 1.4	Water-based (WB) fluid, with ilmenite
1 462	metric t	log-norm: σ^2 = 1.2	Onshore treatment of WB/ilmenite cuttings

Table I-2: Barite onshore inventory

Amount	Unit	Distribution	Process
632	m3	log-norm: σ^2 = 1.4	Water-based (WB) fluid, with barite
1 462	metric t	log-norm: σ^2 = 1.2	Onshore treatment of WB/barite cuttings

Functional unit: the function of weight agent for the drilling of the well, offshore discharge of WB cuttings waste

		Mineral	Base fluid	Loading	Waste	Location	Waiting
A2	Ilmenite - offshore	ilmenite	WB	-	offshore	-	-
	Barite - offshore	barite	WB	-	offshore	-	-

Table I-3: Ilmenite offshore inventory

Amount	Unit	Distribution	Process
632	m3	log-norm: σ^2 = 1.4	Water-based (WB) fluid, with ilmenite
1 462	metric t	log-norm: σ^2 = 1.2	Offshore discharge of WB/ilmenite cuttings

Table I-4: Barite offshore inventory

Amount	Unit	Distribution	Process
632	m3	log-norm: σ^2 = 1.4	Water-based (WB) fluid, with barite
1 462	metric t	log-norm: σ^2 = 1.2	Offshore discharge of WB/barite cuttings

Functional unit: the function of loading cuttings waste for the well onto and off supply vessel

		Mineral	Base fluid	Loading	Waste	Location	Waiting
B	Crane-lifts	-	-	cranes	-	-	-
	Hydraulic pump	-	-	hydraulic	-	-	-

Table I-5: Crane-lift inventory

Amount	Unit	Distribution	Process
1	metric t	none, by definition	Loading of cuttings onto and off supply vessel, crane-lifts

Table I-6: Hydraulic system inventory

Amount	Unit	Distribution	Process
1	metric t	none, by definition	Loading of cuttings onto and off supply vessel, hydraulic pump

Functional unit: the function of base fluid for the drilling of the well, given non-local OB treatment

		Mineral	Base fluid	Loading	Waste	Location	Waiting
C1	WB onshore - near	ilmenite	WB	hydraulic	onshore	Balsfjord	-
	OB onshore - far	ilmenite	OB	hydraulic	onshore	Mongstad	-

Table I-7: WB onshore near inventory

Amount	Unit	Distribution	Process
632	m3	log-norm: σ^2 = 1,4	Water-based (WB) fluid, with ilmenite
1 462	metric t	log-norm: σ^2 = 1,2	Loading of cuttings onto and off supply vessel, hydraulic pump
9	p	log-norm: σ^2 = 1,2	Offshore supply, round-trip to Barents Sea well
7	p	log-norm: σ^2 = 1,2	Cargo vessel transport, Polarbase-Bergneset
10 234	tkm	Triangle: 8 187 - 12 281	Transport, truck
1 462	metric t	log-norm: σ^2 = 1,2	Onshore treatment of WB/ilmenite cuttings

Table I-8: OB onshore far inventory

Amount	Unit	Distribution	Process
441	m3	log-norm: σ^2 = 1,4	oil-based (OB) fluid, with ilmenite
1 076	metric t	log-norm: σ^2 = 1,2	Loading of cuttings onto and off supply vessel, hydraulic pump
7	p	log-norm: σ^2 = 1,2	Offshore supply, round-trip to Barents Sea well
1,65E+06	tkm	log-norm: σ^2 = 1,2	Cargo vessel transport, Polarbase-Mongstad
1 076	tkm	Triangle: 861 - 1 291	Transport, truck
1 076	metric t	log-norm: σ^2 = 1,2	Onshore treatment of OB/ilmenite cuttings

Functional unit: the function of base fluid for the drilling of the well, given local OB treatment

		Mineral	Base fluid	Loading	Waste	Location	Waiting
C2	WB onshore - near	ilmenite	WB	hydraulic	onshore	Balsfjord	-
	OB onshore - near	ilmenite	OB	hydraulic	onshore	Balsfjord	-

Table I-9: WB onshore near inventory

Amount	Unit	Distribution	Process
632	metric t	log-norm: σ^2 = 1,4	Water-based (WB) fluid, with ilmenite
1 462	metric t	log-norm: σ^2 = 1,2	Loading of cuttings onto and off supply vessel, hydraulic pump
9	p	log-norm: σ^2 = 1,2	Offshore supply, round-trip to Barents Sea well
7	p	log-norm: σ^2 = 1,2	Cargo vessel transport, Polarbase-Bergneset
10 234	tkm	Triangle: 8 187 - 12 281	Transport, truck
1 462	metric t	log-norm: σ^2 = 1,2	Onshore treatment of WB/ilmenite cuttings

Table I-10: OB onshore near inventory

Amount	Unit	Distribution	Process
441	m3	log-norm: σ^2 = 1,4	oil-based (OB) fluid, with ilmenite
1 076	metric t	log-norm: σ^2 = 1,2	Loading of cuttings onto and off supply vessel, hydraulic pump
7	p	log-norm: σ^2 = 1,2	Offshore supply, round-trip to Barents Sea well
6	p	log-norm: σ^2 = 1,2	Cargo vessel transport, Polarbase-Bergneset
7 529	tkm	Triangle: 6 023 - 9 034	Transport, truck
1 076	metric t	log-norm: σ^2 = 1,2	Onshore treatment of OB/ilmenite cuttings

Functional unit: the treatment of cuttings waste drilled with WB/ilmenite fluid, given no wait incident

		Mineral	Base fluid	Loading	Waste	Location	Waiting
D1	WB offshore	-	-	none	offshore	at site	-
	WB onshore	-	-	hydraulic	onshore	Balsfjord	-

Table I-11: WB offshore inventory

Amount	Unit	Distribution	Process
1 462	metric t	log-norm: σ^2 = 1,2	Offshore discharge of WB/ilmenite cuttings

Table I-12: WB onshore near inventory

Amount	Unit	Distribution	Process
1 462	metric t	log-norm: σ^2 = 1,2	Loading of cuttings onto and off supply vessel, hydraulic pump
9	p	log-norm: σ^2 = 1,2	Offshore supply, round-trip to Barents Sea well
7	p	log-norm: σ^2 = 1,2	Cargo vessel transport, Polarbase-Bergneset
10 234	tkm	Triangle: 8 187 - 12 281	Transport, truck
1 462	metric t	log-norm: σ^2 = 1,2	Onshore treatment of WB/ilmenite cuttings

Functional unit: the treatment of cuttings waste drilled with WB/ilmenite fluid, given one wait incident

		Mineral	Base fluid	Loading	Waste	Location	Waiting
D2	WB offshore	-	-	none	offshore	at site	none
	WB onshore	-	-	hydraulic	onshore	Balsfjord	2 days

Table I-13: WB offshore inventory

Amount	Unit	Distribution	Process
1 462	metric t	log-norm: σ^2 = 1,2	Offshore discharge of WB/ilmenite cuttings

Table I-14: WB onshore near inventory, with waiting on weather (2 days incident)

Amount	Unit	Distribution	Process
1 462	metric t	log-norm: σ^2 = 1,2	Loading of cuttings onto and off supply vessel, hydraulic pump
10	p	log-norm: σ^2 = 1,2	Offshore supply, round-trip to Barents Sea well
7	p	log-norm: σ^2 = 1,2	Cargo vessel transport, Polarbase-Bergneset
10 234	tkm	Triangle: 8 187 - 12 281	Transport, truck
1 462	metric t	log-norm: σ^2 = 1,2	Onshore treatment of WB/ilmenite cuttings
1,28,E+06	MJ	Triangle: 1,02E+6 - 1,54E+6	Diesel, burned in diesel-electric generating set offshore

Calculated fluid use and volume of cuttings produced

Reference: fluid and cuttings in sections with fluid return (and wet cuttings transport to shore)

Representive of a hypothetical well, described by section lengths and fluid system

Sources
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Tor Henry Omland. 2007. Personal communication. Statoil ASA, Stavanger, Norway.
Aud Lykling Berge. 2004. Søknad om utslippstillatelse for planlagte utslipp ved boring av letebrønn 7227/11-1S Uranus (PL202). Stavanger, Norway, Statoil ASA.

Table II-1: Cuttings waste for the oil-based fluid system

OB	σ^2	unit *	Description
1 076	1,2	metric t	Cuttings with fluid residue, transported to shore
686	1,2	m3	Equivalent volume, including bulk expansion

Table II-2: fluid loss for the oil-based (OB) fluid system

OB	σ^2	unit	Description
343	-	m3	fluid lost as residue on cuttings
97	-	m3	fluid lost down-hole (i.e., lost to formations)
441	1,4	m3	Total fluid loss

Table II-3: Cuttings waste for the water-based fluid system

WB	σ^2	unit *	Description
1 462	1,2	metric t	Cuttings with fluid residue, transported to shore
923	1,2	m3	Equivalent volume, including bulk expansion

Table II-4: fluid loss for the water-based (WB) fluid system

WB	σ^2	unit	Description
599	-	m3	fluid lost as residue on cuttings
32	-	m3	fluid lost down-hole (i.e., lost to formations)
632	1,4	m3	Total fluid loss

Uncertainty approach

All uncertainty indications are based on rough estimation. The potential variation of fluid volume due to fluid on cuttings (m3/m3) is within 0.5-2 for NA and 2-3.5 for WB fluid systems, A distribution with $\sigma^2 = 1.2$ covers this variation . A distribution described by $\sigma^2 = 1.4$ is found to represent the variation in fluid losses. Both fluid loss and cuttings mass is in our model a direct result of the ratio of fluid on cuttings (m3 per m3 dry cuttings). This variation is controlled by case-specific parameters such as formation properties, solids control efficiency, technical problems during operations, etc.

Note on the amount of fluid lost to formations

We apply average factors for the loss of fluid to formations, with relatively small variations for this fraction. While this is a valid approach for our comparative assessment, it is a less realistic situation for the actual outcome of operations. The loss of fluid to formations is highly variable from operation to operation, from being of little significance in most operations, to very large volumes if problems arise during drilling; an example is if the well caves in, thereby cutting of fluid circulation.

Underlying assumptions

Table II-5: Well description (Lykling Berge 2004)

Section	Length m	Diameter inches	Theor. vol. m3	Description
36"	52	36	34	Spud-section, discharged at site
26"	365	26	125	Spud-section, discharged at site
17 1/2"	505	17,5	78	Drilled with riser, cuttings sent ashore
12 1/4"	992	12,25	75	Drilled with riser, cuttings sent ashore
8 1/2"	1752	8,5	64	Drilled with riser, cuttings sent ashore
Sum	3666	-	377	All sections
Sum	3249	-	218	Sections with cuttings sent ashore

Table II-6: Factors affecting cuttings waste production; generic factors for WB and OB fluid systems (Omland 2007)

NA	WB	unit	Description
1,05	1,1	-	Wash-out factor (hole enlargement)
1,5	2,5	m3/m3	fluid-on-cuttings volume ratio fluid:cuttings
1,2	1,1	-	Bulk expansion factor; volume increase
2,6	2,6	sg	Specific gravity of formations; metric t per m3
1,4	1,4	sg	Specific gravity of fluid; metric t per m3

Intemediate calculalations

Table II-7: Intermediate calculations for volume and mass

OB	WB	unit	Description
218	218	m3	Cuttings volume (dry cuttings); theoretical hole
229	240	m3	Cuttings volume (dry cuttings); with wash-out
595	623	metric t	Cuttings mass (dry cuttings); given formation density
343	599	m3	fluid residue volume
481	839	metric t	fluid residue mass; assuming given fluid density
572	839	m3	Cuttings volume (with fluid residue, wet)
686	923	m3	Cuttings volume (with fluid residue, wet); incl. bulk exp.
1 076	1 462	metric t	Cuttings mass (with fluid residue, wet)

Table II-8: fluid lost on cuttings for oil-based (NA) and water-based (WB) fluid systems

OB	WB	unit	Description
481	839	metric t	Mass of fluid lost as residue on cuttings
343	599	m3	Volume of fluid lost as residue on cuttings

Table II-9: fluid lost down-hole for oil-based (OB) and water-based (WB) fluid systems *

OB	WB	unit	Description
0,03	0,01	m3 per m	Lost to formations, m3 per m drilled
97	32	m3	Total lost to formations; indicative values

* Numbers are for wells in Norwegian waters, Statoil ASA, sept 2006 - sept 2007 (Jensen 2007), indicative of volume of fluid lost to formations per meter drilled. Volumes intentioOBlly left in well are not included. The numbers support a difference between OB and WB fluid systems although the apparent disrepancy may be caused by the two systems being used for different purposes, i.e., with different probabilities for the occurrence of problems leading to loss. We do, however, not have information to investigate this further but include them in our fluid inventory as an indication of the portion of fluid loss, and that must be replaced by virging

Ecotoxic components in drilling fluid

Table III-1: Dope use for the well, drilled with water-based fluid (Lykling Berge 2004)

Type	kg use	kg in fluid	kg per metric t fluid*	σ^2
Drill-string dope	400	40,0	2,7E-02	5,0
Casing dope	13	1,3	8,9E-04	5,0

* Assuming 1 462 metric t of cuttings produced for the well

Table III-1: Contents of base fluid in cuttings drilled with WB fluid

	amount	σ^2	
fluid residue on cuttings	599	-	m3 in section with return to well (not spud)
Base fluid in barite fluid	50,6	-	kg per m3 WB/barite fluid
Base fluid lost as residue on barite cuttings	30 317	20 *	kg per well for WB/barite fluid system
Base fluid lost as residue on barite cuttings	20,7	20 *	kg per metric t of cuttings **
Base fluid in ilmenite fluid	51,6	-	kg per m3 WB/ilmenite fluid
Base fluid lost as residue on ilmenite cuttings	30 898	20 *	kg per well for WB/ilmenite fluid system
Base fluid lost as residue on ilmenite cuttings	21,1		kg per metric t of cuttings**

* We model the uncertainty in MAETP for the base fluid through an assigned uncertainty in base fluid loss with cuttings. See also comment on uncertainty in Appendix XIId.
** Assuming 1 462 metric t of cuttings produced for the well

All other components in the drillling fluid are classified as OSPAR PLONOR, i.e., green according to the SFT colour scheme (Lykling Berge 2004). "Green" substances are considered benign and not reported with toxicity test results in the HOCNF reports.

Oil-based drilling fluid, with ilmenite

Reference: 1 m3 of oil-based (OB) drilling fluid, with ilmenite

Representive of a specific oil-based fluid system built on rig for a select drilling situation in Norway, year 2006

Sources: fluid inventories on product level and Material Safety Data Sheets

Table IV-1: Inputs per m3 oil-based/ilmenite fluid

Substance in MSDS	CAS no	kg	σ^2	Matching ecoinvent process
Ilmenite	-	370	1,5	None, modelled by own inventory
Distillates (petroleum hydrotreated middle)	64742-46-7	540	1,5	Diesel, low-sulphur, at regional storage/RER
Polyamide	-	16,4	1,5	Nylon 6, at plant/RER
Mineral oil	-	5,46	1,5	Light fuel oil, at regional storage/RER
2-butoxy ethanol	111-76-2	2,05	2,0	Ethylene glycol, at plant/RER
2-(2-botoxy ethoxy)ethanol	112-34-5	2,05	2,0	Diethylene glycol, at plant/RER
Fatty acid blend	-	11,7	1,5	Fatty acids, from vegetarian oil, at plant/RER
Amido-amine lignite complex	-	2,34	2,0	Ammonia, liquid, at regional storehouse/RER
		21,1 *	2,0	Pulverized lignite, at plant/DE
Calcium hydroxide	1305-62-0	29,4	1,5	Lime, hydrated, packed, at plant/CH
Calcium chloride	10043-52-4	20,9	1,5	Calcium chloride, CACl2, at regional storage/CH
Quarternary alkyl ammonium bentonite	-	10,7	1,5	Bentonite, at processing/DE
Calcium carbonate	-	11,4	1,5	Limestone, milled, loose, at plant/CH
Graphite	7782-45-5	8,55	1,5	Graphite, at plant/RER

* Lignite powder in units of MJ. We assume that the amine-lignite complex is 50/50 wt-% ammonia/lignite.

Water-based drilling fluid, with ilmenite

Reference: 1 m3 of water-based (WB) drilling fluid, with ilmenite
Representive of a specific water-based fluid system built on rig for a select drilling situation in Norway, year 2006
Sources: fluid inventories on product level and Material Safety Data Sheets

Table IV-2: Inputs per m3 water-based/ilmenite fluid

Substance in MSDS	CAS no	kg	σ^2 *	Matching ecoinvent process
Citric acid	77-92-9	3,74	2,0	Acetic acid, 98% in H20, at plant/RER
Xanthan gum	11138-66-2	5,75	2,0	Chemicals organic, at plant/GLO
Polyalkylene glycol (Mw = 240)	-	51,6	3,0	Triethylene glycol, at plant/RER
Ilmenite	-	429	1,5	None, modelled by own inventory
Potassium chloride	7447-40-7	201	1,5	Potassium chloride, as K2O, at regional storehouse/RER
Polyanionic cellulose polymer	9004-32-4	14,6	1,5	Carboxymethyl cellulose, powder, at plant/RER
Sodium carbonate	497-19-8	1,19	1,5	Soda, powder, at plant/RER
Sodium hydrogen carbonate	144-55-8	2,99	2,0	Soda, powder, at plant/RER

* the larger uncertainty for polyalkylene glycol is due to poor specificity

Oil-based drilling fluid, with barite

Reference: 1 m3 of oil-based (OB) drilling fluid, with barite

Representive of a specific oil-based fluid system built on rig for a select drilling situation in Norway, year 2006

Sources: fluid inventories on product level and Material Safety Data Sheets

Table IV-3: Inputs per m3 oil-based/barite fluid

Substance in MSDS	CAS no	kg	σ^2	Matching ecoinvent process
Barite	-	387	1,5	None, modelled by own inventory
Distillates (petroleum hydrotreated middle)	64742-46-7	532	1,5	Diesel, low-sulphur, at regional storage/RER
Polyamide	-	16,1	1,5	Nylon 6, at plant/RER
Mineral oil	-	5,37	1,5	Light fuel oil, at regional storage/RER
2-butoxy ethanol	111-76-2	2,02	2,0	Ethylene glycol, at plant/RER
2-(2-botoxy ethoxy)ethanol	112-34-5	2,02	2,0	Diethylene glycol, at plant/RER
Fatty acid blend	-	11,5	1,5	Fatty acids, from vegetarian oil, at plant/RER
Amido-amine lignite complex	-	2,30	2,0	Ammonia, liquid, at regional storehouse/RER
		20.8 *	2,0	Pulverized lignite, at plant/DE
Calcium hydroxide	1305-62-0	28,9	1,5	Lime, hydrated, packed, at plant/CH
Calcium chloride	10043-52-4	20,6	1,5	Calcium chloride, CACl2, at regional storage/CH
Quarternary alkyl ammonium bentonite	-	10,5	1,5	Bentonite, at processing/DE
Calcium carbonate	-	11,2	1,5	Limestone, milled, loose, at plant/CH
Graphite	7782-45-5	8,41	1,5	Graphite, at plant/RER

* Lignite powder in units of MJ. We assume that the amine-lignite complex is 50/50 wt-% ammonia/lignite.

Water-based drilling fluid, with barite

Reference: 1 m3 of water-based (WB) drilling fluid, with barite

Representive of a specific water-based fluid system built on rig for a select drilling situation in Norway, year 2006

Sources: fluid inventories on product level and Material Safety Data Sheets

Table IV-4: Inputs per m3 water-based/barite fluid

Substance in MSDS	CAS no	kg	σ^2 *	Matching ecoinvent process
Citric acid	77-92-9	3,67	2,0	Acetic acid, 98% in H20, at plant/RER
Xanthan gum	11138-66-2	5,64	2,0	Chemicals organic, at plant/GLO
Polyalkylene glycol (Mw = 240)	-	50,6	3,0	Triethylene glycol, at plant/RER
Barite	-	447	1,5	None, modelled by own inventory
Potassium chloride	7447-40-7	197	1,5	Potassium chloride, as K2O, at regional storehouse/RER
Polyanionic cellulose polymer	9004-32-4	14,4	1,5	Carboxymethyl cellulose, powder, at plant/RER
Sodium carbonate	497-19-8	1,17	1,5	Soda, powder, at plant/RER
Sodium hydrogen carbonate	144-55-8	2,93	2,0	Soda, powder, at plant/RER

* the larger uncertainty for polyalkylene glycol is due to poor specificity

Calculation of ilmenite and barite use in WB and OB fluids

Ilmenite and barite use, and the resulting fluid composition

Table IV-5: Density of weight minerals (Omland 2007)

ilmenite	4,8	sg (metric t per m3)
barite	4,2	sg (metric t per m3)

Table IV-6: Further assumptions for calculation of barite and ilmenite use

OB and WB fluid density	1,4	sg (metric t per m3)

Table IV-7: Barite use for the oil-based (OB) fluid system

Content of ilmenite in original WB fluid	0,370	metric t per m3
Corresponding content of barite	0,387	metric t per m3
Ratio for other components barite:ilmenite	0,984	input per m3; relative to ilmenite fluid

Table IV-8: Barite use for the water-based (WB) fluid system

Content of ilmenite in original OB fluid	0,429	metric t per m3
Corresponding content of barite	0,447	metric t per m3
Ratio for other components barite:ilmenite	0,981	input per m3; relative to ilmenite fluid

Comment

The original inventories are made with ilmenite as the weight agent. fluid engineering principles were used to estimate the corresponding inventories for barite as the weight agent (Bourgoyne et al. 1991). Barite use and content of fluid additives and base fluid was estimated from an assumed fluid denisty (sg. 1.4) and barite and ilmenite densities (sg. = specific gravity).

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Uncertainty approach:

We assume log-normal distributions with a general distribution described by $\sigma^2 = 1.5$. This is indicative of the variation in fluid composition, as seen in the variation between WB fluid systems specified by Lykling Berge (2004) and Paulsen et al. (2005) . The factor is increased to 2.0 for substances if there is a poor match with ecoinvent materials. A factor 3.0 is used if the MSDS does not specify the exact substance that is used, i.e., if there is poor availability of information.

Comments:

It may seem from the inventories above that the oil-based fluid has a higher density than the water-based. This is because water is added to the water-based fluid, thereby increasing the density of the fluid system as most of the components listed are minerals or brine components.

Some variation may be caused by our inventories being based on built volumes, thereby not taking into account that the inventory shows amounts of chemicals adedd to an existing fluid base. Furthermore, we have not considered that the fluid properties are balanced to accomodate changes in technical specifications for the fluid as the drilling commences .

The barite fluid systems are estimated from the original ilmenite fluid inventories by use of specific gravity for ilmenite (4.8) and barite (4.2), and the assumption that both fluid systems; WB and OB, have a specific gravity of 1.4.

Life-cycle inventory for the production of Ilmenite
Reference: 1 metric tonne of Ilmenite at producer
Company Titania AS, Sokndal (Norway)
Source Titania AS emissions report published at www.sft.no/bmi
Period 2003-2005

The ilmenite used in offshore drilling appliactions undergo little additional treatment compared to Ilmenite for titanium dioxide production. The only difference is a simple jet-stream set up to reduce particle size. This process is not included.

Table Va-1: Allocation key, production volumes (metric tonnes). Allocation by weight.

Year	Ilmenite	Magnetite	Sulphur ore	Key
2003	859 016	10 775	7 659	0,98
2004	866 601	14 474	8 184	0,97
2005	811 125	12 662	7 590	0,98

Table Va-2: Energy sources (MJ per metric tonne Ilmenite, allocated)

Year	Natural gas	Fuel oil
2005	225,5	13,6
Average	225,5	13,6
Distribution	Triangle	Triangle
Variation	200-250	12-15.6

Assumptions: 46.894 MJ per kg natural gas, 40 MJ per kg oil. New energy systems were installed 2003-2004, therefore we use 2005 data only.

Table Va-3: Emission to water per tonne ilmenite produced (allocated); average 2003-2005

Substance	Mean	Distribution	St. dev.
Cd	2,41E-07	Normal	8,38E-08
Cu	5,23E-05	Normal	3,25E-06
Co	2,16E-04	Normal	1,07E-04
Ni	5,51E-03	Normal	1,56E-03
N-tot	3,61E-02	Normal	1,45E-02
Olje	7,00E-03	Normal	9,49E-03
Zn	1,20E-04	Normal	1,02E-04
Susp part	1,12E+00	Normal	2,41E-01
PO3	2,32E-02	Normal	6,58E-03

Table Va-4: Waste produced, per tonne ilmenite (allocated); average 2003-2005

Fraction	Mean	Distrib.	St. dev.	Treatment
Oil waste	4,59E+00	Normal	7,83E+00	Incinerat.
Hazardous	3,72E-02	Normal	4,30E-02	Incinerat.
EE Waste	1,09E-02	Normal	2,65E-03	Recycling
Cardboard	7,81E-03	Normal	3,24E-04	Recycling
Paper	3,65E-02	Normal	1,76E-02	Recycling
Steel	9,84E-02	Normal	3,81E-02	Recycling
Wood	1,36E-01	Normal	3,16E-02	Incinerat.
Tailings	2,51E+03	Normal	8,94E+01	Overburden

Life-cycle inventory for the production of Barite

Reference: 1 metric tonne of Barite at producer

Company	Norbar Minerals AS, Tananger (Norway)
Sources	Norbar Minerals AS emissions report published at www.sft.no/bmi Johan Pettersen, Victor Okezie, Sven Otto Kråkvik. 2002. Life cycle assessment of weight materials in drilling fluid. Statoil, Stavanger,
Period	2000-2004

The barite data does not include any emissions to water. Such data is not available for neither mining nor refining processes. Mining operations are located in the Sahara desert, and waterborn emissions are not considered important. Waterborn emissions from refining are unfortunately not included in the SFT reports, but may in any occasion be considered of small scale due to the process at Tananger mainly being a refining process with little or no tailings produced.

Barite production is divided into extraction and refining. Extraction of barite is done in Morocco, where raw mineral is mined and transported to the coast. The raw barite is then shipped to Norway for refining at the Norbar Minerals facility (Tananger).

Operation of mining plant

Location	Selmou, Morocco
Company	Norbar Minerals AS
Source	Pettersen et al 2002

Table Vb-1: Allocation key, production volumes (metric tonnes). Allocation by weight.

Year	Barite	Bentonite	Key
2001	140 890	9 818	0,93

Table Vb-2: Inputs per metric tonne raw barite mineral extracted, allocated

Process	Fuel	Amount	Distribution	Units
Trucks	Diesel	69,6	Triangle: 60-80	tkm
Generators	Fuel oil	29,5	Triangle: 25-35	MJ
Compressors	Fuel oil	9,9	Triangle: 5-15	MJ
Rail	-	400	Triangle: 350-450	tkm
Ship	-	3600	Triangle: 3000-4200	tkm

Operation of refining plant

Location	Tananger, Norway
Company	Norbar Minerals AS
Source	Norbar Minerals AS emissions report published at www.sft.no/bmi

Table Vb-3: Allocation key, production volumes (metric tonnes). Allocation by weight.

Year	Barite	Bentonite	Key
2002	105 997	7 546	0,93
2003	121 130	9 195	0,93
2004	127 637	10 615	0,92

Table Vb-4: Barite refining, energy sources (MJ per metric tonne Barite); average 2002-2004

Year	Natural gas	Light fuel oil
Mean	65,5	59,6
Distribution	Normal	Normal
St.dev.	8,6	7,8

Table Vb-5: Waste produced in refining, per tonne barite (allocated); average 2002-2004

Fraction	Mean	Distrib.	St. dev.	Treatment
Oil waste	1,12E-02	Normal	7,31E-03	Incinerat.
Hazardous	1,80E-03	Normal	1,13E-03	Incinerat.
Textiles	8,67E-04	Normal	1,38E-04	Insinerat.
Cardboard	3,88E-03	Normal	2,72E-03	Recycl.
Paper	6,77E-04	Normal	3,52E-04	Recycl.
Steel	1,23E-01	Normal	1,48E-02	Recycl.
Wood	1,50E-02	Normal	3,33E-03	Incinerat.
Organic	2,97E-04	Normal	1,48E-04	Incinerat.
Rubber	6,06E-03	Normal	3,98E-03	Incinerat.

Loading of cuttings onto and off supply vessel, hydraulic pump

Reference: 1 metric tonne cuttings loaded onto and off supply vessel; replaces 11 crane-lift tonnes
Representive of Norway, Europe; year 2000-2006
Case-specific fuel use data

Table VI-1: Inputs per metric tonne cuttings loaded with hydraulic pump *

Amount	Unit	Distribution	σ^2	Process
4,04E+02	MJ	Log-normal	2,0	Diesel, burned in diesel-electric generating set offshore
8,05E-07	p	Log-normal	2,0	Diesel-electric generating set production 10MW/RER/I
1,67E-03	kg	Log-normal	2,0	Electronics for control units/RER
5,78E-04	metric t	Log-normal	1,5	Reinforcing steel, at plant/RER
1,00E-04	metric t	Log-normal	1,5	Chromium steel 18/8, at plant/RER

* Fuel use: Folkvord (2006) and estimate, infrastructure: Samuelsen (2006) and estimate. Offshore energy assumed supplied by system equivalent to "Diesel, burned in diesel-electric generating set/GLO U" (ecoinvent). The system is assumed to have a lifetime capacity of 60,000 metric tonnes of cuttings, and has a total weight of

Loading of cuttings onto and off supply vessel, crane-lift

Reference: 1 metric tonne cuttings loaded onto and off supply vessel; 11 crane-lift tonnes in total
Representive of World; year 2000-2006
Average fuel use data

Table VI-2: Inputs and outputs per metric tonne cuttings loaded with crane-lift

Amount	Unit	Distribution	Parameter	Process
2,77E+02	MJ	Log-normal	1,5 (σ^2)	Diesel, burned in diesel-electric generating set offshore
11	p	Trangle	9-13	Crane-lift accident risk units

* Fuel use: EPA (1999) and estimate (11 container lifts, 4.5 tonne cuttings in container); no infrastructure as cranes are existing structures on all rigs. Offshore energy assumed supplied by system equivalent to "Diesel, burned in diesel-electric generating set/GLO U" (ecoinvent).

Crane-lift

Reference: 1 crane-lift
Representive of World; year 2000-2006
Average fuel use data

Table VI-3: Inputs and outputs per crane-lift *

Amount	Unit	Distribution	Parameter	Process
1,13E+02	MJ	Log-normal	1,5 (σ^2)	Diesel, burned in diesel-electric generating set offshore
1	p	defined as reference		Crane-lift accident risk units

* Fuel use: EPA (1999); no infrastructure as cranes are existing structures on all rigs. Offshore energy assumed supplied by system equivalent to "Diesel, burned in diesel-electric generating set/GLO U" (ecoinvent).

Transport of cuttings for the well, drilled with oil-based fluid, to onshore treatment site situated locally
Reference: 1 076 metric tonnes of cuttings with OB fluid residue, transported to local treatment site
Representative of specific well; year 2000-2006. Route: Eirik Raude (rig) - Stormoen deposit (Balsfjord)
Case-specific transport route

Table VII-1: Cuttings to be transported to treatment; OB fluid system

OB	σ^2	min	max	
1 076	1,2	896	1 291	metric tonnes

Table VII-2: Transport operations per reference unit, OB fluid system to local facility

Amount	Unit	Min	Max	Distribution	
7	p	6	8	Triangle	Offshore supply, round-trip to Barents Sea well
5	p	5	6	Triangle	Cargo vessel transport, Polarbase-Bergneset
7 529	tkm	6 023	9 034	Triangle	Transport, truck

Transport of cuttings for the well, drileld with oil-based fluid, to onshore treatment site situated not-locally
Reference: 1 076 metric tonnes of cuttings with OB fluid residue, transported to local treatment site
Representative of specific well; year 2000-2006. Route: Eirik Raude (rig) - Mongstad
Case-specific transport route

Table VII-3: Transport operations per reference unit, OB fluid system to non-local facility

Amount	Unit	Min	Max	Distribution	
7	p	6	8	Triangle	Offshore supply, round-trip to Barents Sea well
1,65E+06	tkm	1,32E+06	1,98E+06	Triangle	Cargo vessel transport, Polarbase-Mongstad
1 076	tkm	861	1 291	Triangle	Transport, truck

Transport of cuttings for the well, drilled with water-based fluid, to onshore treatment site situated locally
Reference: 1 462 metric tonnes of cuttings with WB fluid residue, transported to local treatment site
Representative of specific well; year 2000-2006. Route: Eirik Raude (rig) - Stormoen deposit (Balsfjord)
Case-specific transport route

Table VII-4: Cuttings to be transported to treatment, WB fluid system

WB	σ^2	min	max	
1 462	1,2	1 218	1 754	metric tonnes

Table VII-5: Transport operations per reference unit, WB fluid system without wait incident

Amount	Unit	Min	Max	Distribution	
9	p	8	11	Triangle	Offshore supply, round-trip to Barents Sea well
7	p	6	9	Triangle	Cargo vessel transport, Polarbase-Bergneset
10 234	tkm	8 187	12 281	Triangle	Transport, truck

Transport of cuttings for the well, drilled with water-based fluid, to onshore treatment site situated locally; including a 2 day waiting on weather incident
Reference: 1 462 metric tonnes of cuttings with WB fluid residue, transported to local treatment site; including 2 days additional rig energy production (15 metric t per day) and 1 additional supply service round-trip.
Representative of specific well; year 2000-2006. Route: Eirik Raude (rig) - Stormoen deposit (Balsfjord)
Case-specific transport route

Table VII-6: Transport operations per reference unit, WB fluid system with wait incident

Amount	Unit	Min	Max	Distribution	
10	p	9	12	Triangle	Offshore supply, round-trip to Barents Sea well
7	p	6	9	Triangle	Cargo vessel transport, Polarbase-Bergneset
1,02E+04	tkm	8,19E+03	1,23E+04	Triangle	Transport, truck
1,28E+06	MJ	1,03E+06	1,54E+06	Triangle	Diesel, burned in diesel-electric generating set offshore

Table VII-7: Transport distances for the cuttings transport chain (Barlindhaug 2006)

nmi *	km	Mode	Capacity	Route
270	500	Supply vessel	160 metric t	Polarbase - Eirik Raude - Polarbase
150	278	Cargo vessel	200 metric t	Polarbase-Bergneset
-	7	Lift truck	Not relevant	Bergneset-Stormoen
830	1 537	Cargo vessel	Not relevant	Polarbase-Mongstad

* nmi = nautical mile (= 1.852 km)
Polarbase is the local supply for the drilling operation
Eirik Raude is the drilling rig
Bergneset is the dock for the local treatment site (Stormoen deposit, Balsfjord)
Mongstad is the non-local treatment site (Mongstad base, 50km off Bergen)

The lift truck and long-distance cargo vessel are modelled as pure transport operations; i.e., by tkm.

Offshore supply, round-trip to Barents Sea well

Reference: 160 tonnes cuttings carried from rig to Polarbase

Representive of Norway, Europe; year 2000-2006. Case-specific fuel use data.

Table VIII-1: Inputs per offshore service round-trip *

Amount	Unit	Min	Max	Distribution	Input
7,90	metric t	6,32	9,48	Triangle	Marine diesel, low-S
160,00	metric t	defined as reference			Cuttings loaded onto and off supply vessel
4,87E-05	p	distribution as original data			Maintenance, transoceanic freight ship/RER/I **
4,01E-06	p	distribution as original data			Operation, maintenance, port/RER/I **
4,01E-08	p	distribution as original data			Port facilities/RER/I **
4,87E-05	p	distribution as original data			Transoceanic freight ship/OCE/I **

* Folkvord (2006): fuel use per trip is 6.65 metric tonne while sailing (24 h round-trip), and 1.25 metric tonne during dynamic positioning at rig (6 h service time). ** Scaled from the ecoinvent process "Transport, transoceanic freight ship/OCE" according to fuel use.

Table VIII-2: Direct emissions to air per offshore service round-trip

Substance	Emission	Uncertainty	Triangle boundaries *		Substance	Emission	Uncertainty	Triangle boundaries *	
	g/trip	[%]	Min	Max		g/trip	[%]	Min	Max
Nox	477 951	5-10	430 156	525 746	Se	3,95E-04	>50	1,30E-04	1,19E-03
CO	47 628	10-20	38 102	57 153	Zn	7,90	20-50	3,95	11,85
nmVOC	8 654	10-20	6 923	10 384	PCB	3,81E-03	>50	1,26E-03	1,14E-02
Sox	63 202	20-50	31 601	94 802	Diox/Fur	3,81E-05	>50	1,26E-05	1,14E-04
NH3	171	20-50	86	257	Benz(a)pyr	0,0381	20-50	0,0190	0,0571
PM2.5	8 706	>50	2 873	26 118	Benz(b)flu	0,0763	20-50	0,0381	0,1144
Pb	1,185	>50	0,391	3,555	Benz(k)flu	0,0381	20-50	0,0190	0,0571
Cd	0,0395	>50	0,0130	0,1185	Indenopyr	0,0763	20-50	0,0381	0,1144
Hg	3,95E-04	>50	1,30E-04	1,19E-03	PAH-4	0,2266	20-50	0,1133	0,3399
As	0,237	>50	0,078	0,711	HCB	3,03E-04	>50	1,00E-04	9,09E-04
Cr	0,395	>50	0,130	1,185	CO2	2,51E+07	5-10	2,26E+07	2,76E+07
Cu	13,4	>50	4,4	40,3	CH4	174	20-50	87	261
Ni	7,90	20-50	3,95	11,85	N2O	1 175	20-50	588	1 763

Sailing and dynamic positioning are set to "at sea" and "manoeuvring" respectively in Cooper and Gustavsson (2004) .

Cargo vessel transport, Polarbase-Bergneset

Reference: 200 tonnes cuttings carried from Polarbase to Bergneset dock

Representive of Norway, Europe; year 2000-2006. Case-specific fuel use data.

Table VIII-3: Inputs per cargo trip Polarbase - Bergneset *

Amount	Unit	Min	Max	Distribution	Process
2,14	metric t	1,71	2,57	Triangle	Marine diesel, low-S
85	lifts	68	102	Triangle	Crane-lifts
1,32E-05	p	distribution as original data			Maintenance, transoceanic freight ship/RER/I **
1,09E-06	p	distribution as original data			Operation, maintenance, port/RER/I **
1,09E-08	p	distribution as original data			Port facilities/RERI **
1,32E-05	p	distribution as original data			Transoceanic freight ship/OCE/I **

* Fuel use (Barlindhaug 2006), crane-lifts (4 lifts per container, Scandpower 2003; 6 m3 cuttings per container, Folkvord 2006; with density 1.57 metric t per m3). ** Scaled from the ecoinvent process "Transport, transoceanic freight ship/OCE" according to fuel use.

Table VIII-4: Direct emissions to air per cargo trip Polarbase-Bergneset

Substance	Emission	Uncertainty	Triangle boundaries *		Substance	Emission	Uncertainty	Triangle boundaries *	
	g/trip	[%]	Min	Max		g/trip	[%]	Min	Max
Nox	135 419	5-10	121 877	148 961	Se	1,07E-04	>50	3,53E-05	3,21E-04
CO	11 436	10-20	9 149	13 723	Zn	2,14	20-50	1,07	3,21
nmVOC	2 078	10-20	1 662	2 493	PCB	1,05E-03	>50	3,46E-04	3,15E-03
Sox	17 136	20-50	8 568	25 704	Diox/Fur	1,05E-05	>50	3,46E-06	3,15E-05
NH3	47	20-50	24	71	Benz(a)pyr	0,0105	20-50	0,0052	0,0157
PM2.5	2 091	>50	690	6 272	Benz(b)flu	0,0210	20-50	0,0105	0,0315
Pb	0,321	>50	0,106	0,964	Benz(k)flu	0,0105	20-50	0,0052	0,0157
Cd	0,0107	>50	0,0035	0,0321	Indenopyr	0,0210	20-50	0,0105	0,0315
Hg	1,07E-04	>50	3,53E-05	3,21E-04	PAH-4	0,0621	20-50	0,0311	0,0932
As	0,064	>50	0,021	0,193	HCB	8,35E-05	>50	2,76E-05	2,51E-04
Cr	0,107	>50	0,035	0,321	CO2	6,81E+06	5-10	6,13E+06	7,49E+06
Cu	3,6	>50	1,2	10,9	CH4	42	20-50	21	63
Ni	2,14	20-50	1,07	3,21	N2O	323	20-50	162	485

Fuel use is 2 142 kg per trip Polarbase - Bergneset (Barlindhaug 2006). Emission factors for "at sea" in Cooper and Gustavsson (2004) are used.

Cargo vessel transport, Polarbase-Mongstad

Reference: 1 tkm cargo transport of cuttings, average data

Representive of Norway, Europe; year 2000-2006. Average situation fuel use.

Table VIII-5: Inputs per tkm cargo transport Polarbase - Mongstad*

Amount	Unit	Min	Max	Distribution	Process
1,11E-05	metric t	8,88E-06	1,33E-05	Triangle	Marine diesel low-S
2,76E-04	lifts	2,21E-04	3,32E-04	Triangle	Crane-lifts
6,84E-11	p	distribution as original data			Maintenance, transoceanic freight ship/RER/I **
5,64E-12	p	distribution as original data			Operation, maintenance, port/RER/I **
5,64E-14	p	distribution as original data			Port facilities/RERI **
6,84E-11	p	distribution as original data			Transoceanic freight ship/OCE/I **

* Fuel use (Fet et al. 2000), crane-lifts (4 lifts per container, Scandpower 2003; 6 m3 cuttings per container, Folkvord 2006; with density 1.57 metric t per m3). ** Scaled from the ecoinvent process "Transport, transoceanic freight ship/OCE" according to fuel use.

Table VIII-6: Direct emissions to air per tkm cargo transport Polarbase-Mongstad

Substance	Emission	Uncertainty	Triangle boundaries *		Substance	Emission	Uncertainty	Triangle boundaries *	
	g/tkm	[%]	Min	Max		g/tkm	[%]	Min	Max
Nox	0,7021	5-10	0,6319	0,7723	Se	5,55E-10	>50	1,83E-10	1,67E-09
CO	0,0593	10-20	0,0474	0,0712	Zn	1,11E-05	20-50	5,55E-06	1,67E-05
nmVOC	0,0108	10-20	0,0086	0,0129	PCB	5,44E-09	>50	1,80E-09	1,63E-08
Sox	0,08884	20-50	0,04442	0,13327	Diox/Fur	5,44E-11	>50	1,80E-11	1,63E-10
NH3	2,44E-04	20-50	1,22E-04	3,66E-04	Benz(a)pyr	5,44E-08	20-50	2,72E-08	8,16E-08
PM2.5	0,01084	>50	0,00358	0,03252	Benz(b)flu	1,09E-07	20-50	5,44E-08	1,63E-07
Pb	1,67E-06	>50	5,50E-07	5,00E-06	Benz(k)flu	5,44E-08	20-50	2,72E-08	8,16E-08
Cd	5,55E-08	>50	1,83E-08	1,67E-07	Indenopyr	1,09E-07	20-50	5,44E-08	1,63E-07
Hg	5,55E-10	>50	1,83E-10	1,67E-09	PAH-4	3,22E-07	20-50	1,61E-07	4,83E-07
As	3,33E-07	>50	1,10E-07	1,00E-06	HCB	4,33E-10	>50	1,43E-10	1,30E-09
Cr	5,55E-07	>50	1,83E-07	1,67E-06	CO2	35,3	5-10	31,8	38,8
Cu	1,89E-05	>50	6,23E-06	5,66E-05	CH4	2,17E-04	20-50	1,08E-04	3,25E-04
Ni	1,11E-05	20-50	5,55E-06	1,67E-05	N2O	0,00168	20-50	0,00084	0,00252

We assume transport by an average cargo vessel, as decsribed by Fet et al. (2000); capacity of 5 175 tonnes, 29 km per hour speed, fuel use of 1 tonne per hour. Load factor is set at 0.6. Emissions "at sea" in Cooper and Gustavsson (2004).

Sources:

Cooper D, Gustavsson T. 2004. Methodology for calculating emissions from ships: 1. Update of emission factors. Swedish Meterological and Hydrological Institute (SMHI), Norrköping, Sweden

Fet AM, Michelsen O, Johnsen T. 2000. Environmental performance of transportation - a comparative study. IØT-report nr. 3/2000 Norwegian University of Science and Technology (NTNU), Department of Industrial Economics and Technology Management, Trondheim, Norway

Barlindhaug, J. 2006. Personal communication. Perpetuum Waste Management

Folkvord, TS. 2006. Personal communication. Statoil AS, Stavanger, Norway

Scandpower. 2003. Risk evaluation of handling of cuttings on Snøhvit. Report no. 27.207.306/R1. Scandpower Risk Management AS, Kjeller, Norway

Fuel use is estimated from various sources, indicated for each process. Emission factors from Cooper and Gustanvsson (2004) are used for all ship transport operations. The original inventories indicate uncertainty for all substance entries, with values as given in Table VIII-7 below. We use these uncertainty indications directly in our inventories, as boundaries to a triangle distribution.

Table VIII-7: Distribution of uncertainty in ship emission factors (Cooper and Gustavsson, 2004)

	Triangle distribution	
	Min	Max
5-10%	Value*0.9	Value*1.1
10-20%	Value*0.8	Value*1,2
20-50%	Value*0.5	Value*1.5
>50%	Value*0,33	Value*3

Onshore treatment of cuttings waste, drilled with oil-based/ilmenite fluid
Reference: 1 metric tonne of cuttings treated, drilled with the oil-based (OB) fluid system
Representative of Soilcare facility; Mongstad, Norway; 2007
Energy use specific for this technology; thermomechanical cuttings cleaner (TCC)

Table IXa-1: Oil-based/ilmenite cuttings waste characteristics; all wt-%

65,0 %	Solids
17,5 %	Oil-phase
17,5 %	Aqueous phase
90 %	Base oil in oil-based phase
11,8 %	Ilmenite in cuttings to treatment

Table IXa-2: Calculation for ilmenite content in oil-based/ilmenite cuttings

0,370	metric t per m3 OB fluid
0,319	m3 fluid per tonne cuttings to treatment ; fluid on cuttings ratio is 1.5 (m3/m3)
0,118	metric t ilmenite per tonne cuttings to treatment

Table IXa-3: Inputs for treatment, per metric tonne oil-based/imenite fluid *

Amount	Unit	σ^2	Distrib.	Process
117	kWh	1,5	log-norm.	Electricity, low voltage, production NO, at grid/NO
420	MJ	1,5	log-norm.	Diesel, burned in diesel-electric generating set/GLO
0,18	m3	1,5	log-norm.	Treatment, rainwater mineral oil storage, to wastewater treatment class 2/CH

* Energy use: 700kWh per metric tonne of cuttings with 50/50 diesel/electric energy supply

Table IXa-4: Direct emissions to surface water, per tonne oil-based/ilmenite cuttings *

Substance	kg	σ^2	Distrib.
As	2,14E-04	4,0	log-norm.
Ba	2,93E-03	39,6	log-norm.
Cd	6,62E-06	5,3	log-norm.
Cr	1,48E-03	1,8	log-norm.
Co	2,19E-03	55,0	log-norm.
Cu	1,33E-03	2,3	log-norm.
Pb	5,89E-05	25,4	log-norm.
Ni	5,74E-03	3,0	log-norm.
V	1,08E-03	5,3	log-norm.
Zn	1,42E-03	13,5	log-norm.

* Calculated from an assumed fluid content. Solids are reused for various applications (road filler, construction material, etc). Inorganics are assumed to be released to surface water over time, according to the assumptions for geoavailable metal (Chapter 5.3).

Table IXa-5: Biproducts from treatment, per metric tonne oil-based/ilmenite cuttings*

Amount	Unit	σ^2	Distrib.	Material
150	kg	1,20	log-norm.	Light fuel oil, at regional storage/RER
650	kg	1,20	log-norm.	Gravel, crushed at mine/CH

* Use as gravel material is assumed as the baseline scenario for solids; we assume a regeneration efficiency of 95% for the base oil.

Onshore treatment of cuttings waste, drilled with water-based/ilmenite fluid
Reference: 1 metric tonne of cuttings treated, drilled with the water-based (WB) fluid system
Representative of general facility; Norway; 2007
Treatment consists of degradation of organics, solids reused as material

Table IXb-1: Water-based/ilmenite cuttings waste characteristics; all wt-%

60 %	Solids
40 %	Aqueous phase
17,6 %	Ilmenite in cuttings to treatment

Table IXb-2: Calculation for ilmenite content in water-based/ilmenite cuttings

0,429	metric t per m3 WB fluid
0,410	m3 fluid per tonne cuttings to treatment ; fluid on cuttings ratio is 1.5 (m3/m3)
0,176	metric t ilmenite per tonne cuttings to treatment

Table IXb-3: Inputs for treatment, per metric tonne water-based/ilmenite cuttings*

Amount	Unit	σ^2	Distrib.	Process
1 000	kg	2,0	log-norm.	Process-specific burdens, sanitary landfill

* The original ecoinvent process for the sanitary landfill has been altered with respect to electricity. We use NO grid supply for low and medium voltage.

Table IXb-4: Direct emissions to air, per metric tonne water-based/ilmenite cuttings*

Substance	kg	σ^2	Distribution
CO2	73,3	2,00	log-normal

* Calculated from an assumed fluid content. The treatment consists of anaerob digestion of organics, with capture and combustion of methane. We assume ideal capture, with complete combustion of TOC to CO2. Total organic carbon (TOC) is reported by Amundsen and Sørheim (2006).

Table IXb-5: Direct emissions to surface water, per tonne water-based/ilmenite cuttings *

Substance	kg	σ^2	Distribution
As	3,18E-04	4,0	log-norm.
Ba	4,36E-03	39,6	log-norm.
Cd	9,86E-06	5,3	log-norm.
Cr	2,20E-03	1,8	log-norm.
Co	3,26E-03	55,0	log-norm.
Cu	1,98E-03	2,3	log-norm.
Pb	8,78E-05	25,4	log-norm.
Ni	8,55E-03	3,0	log-norm.
V	1,61E-03	5,3	log-norm.
Zn	2,11E-03	13,5	log-norm.

* Calculated from an assumed fluid content. Solids are reused for various applications (road filler, construction material, etc). Inorganics are assumed to be released to surface water over time, according to the assumptions for geoavailable metal (Chapter 5.3).

Table IXb-6: Biproducts from treatment, per metric tonne water-based/ilmenite cuttings*

Amount	Unit	σ^2	Distrib.	Material
600	kg	1,20	log-norm.	Gravel, crushed at mine/CH

* Use as gravel material is assumed as the baseline scenario for solids.

Onshore treatment of cuttings waste, drilled with water-based/barite fluid
Reference: 1 metric tonne of cuttings treated, drilled with the water-based (WB) fluid system
Representative of general facility; Norway; 2007
Treatment consists of degradation of organics, solids reused as material

Table IXc-1: Water-based/barite cuttings waste characteristics; all wt-%

60 %	Solids
40 %	Aqueous phase
18,3 %	Barite

Table IXc-2: Calculation for barite content in water-based/barite cuttings

0,447	metric t per m3 WB fluid
0,410	m3 fluid per tonne cuttings to treatment ; fluid on cuttings ratio is 1.5 (m3/m3)
0,183	metric t barite per tonne cuttings to treatment

Table IXc-3: Inputs for treatment, per metric tonne water-based/barite cuttings *

Amount	Unit	σ^2	Distrib.	Process
1 000	kg	2,0	log-norm.	Process-specific burdens, sanitary landfill

* The original ecoinvent process for the sanitary landfill has been altered with respect to electricity. We use NO grid supply for low and medium voltage.

Table IXc-4: Direct emissions to air, per metric tonne water-based/barite cuttings *

Substance	kg	σ^2	Distribution
CO2	71,9	2,00	log-normal

* Calculated for an assumed fluid content. The treatment consists of anaerob digestion of organics, with capture and combustion of methane. We assume ideal capture, with complete combustion of TOC to CO2. Total organic carbon (TOC) is reported by Amundsen and Sørheim (2006) for fluid with ilmenite. Organic content of fluid with barite is estimated from fluid density (1.4) and the density of barite and ilmenite as weight agents in WB fluid.

Table IXc-5: Direct emissions to surface water, per tonne water-based/barite cuttings*

Substance	kg	σ^2	Distribution
As	1,80E-04	6,4	log-norm.
Ba	2,13E-02	1,2	log-norm.
Cd	1,69E-04	7,3	log-norm.
Cr	1,98E-03	2,6	log-norm.
Co	1,80E-04	5,2	log-norm.
Cu	1,22E-02	1,7	log-norm.
Pb	1,37E-02	11,5	log-norm.
Ni	2,33E-04	4,5	log-norm.
V	5,42E-04	5,2	log-norm.
Zn	2,10E-02	11,3	log-norm.

* Calculated from an assumed fluid content. Solids are reused for various applications (road filler, construction material, etc). Inorganics are assumed to be released to surface water over time, according to the assumptions for geoavailable metal (Chapter 5.3).

Table IXc-6: Biproducts from treatment, per metric tonne water-based/barite cuttings *

Amount	Unit	σ^2	Distrib.	Material
600	kg	1,20	log-norm.	Gravel, crushed at mine/CH

* Use as gravel material is assumed as the baseline scenario for solids.

Offshore discharge of cuttings waste, drilled with water-based/ilmenite fluid
Reference: 1 metric tonne of cuttings treated, drilled with the water-based (WB) fluid system
Representative of general facility; Norway; 2007
Cuttings discharged to the marine environment at site

Table Xa-1: Direct emissions to ocean, per tonne water-based ilmenite cuttings*

Substance	kg	σ^2	Distribution
As	3,18E-04	4,0	log-norm.
Ba	4,36E-03	39,6	log-norm.
Cd	9,86E-06	5,3	log-norm.
Cr	2,20E-03	1,8	log-norm.
Co	3,26E-03	55,0	log-norm.
Cu	1,98E-03	2,3	log-norm.
Pb	8,78E-05	25,4	log-norm.
Ni	8,55E-03	3,0	log-norm.
V	1,61E-03	5,3	log-norm.
Zn	2,11E-03	13,5	log-norm.
Drill-string dope	2,74E-02	5	log-norm.
Casing dope	8,89E-04	5	log-norm.
Base-fluid	21,1	20	log-norm.

* Calculated from an assumed fluid content. Inorganics are assumed to be released to the water column over time, according to the assumptions for geoavailable metal (Chapter 5.3). Marine aquatic ecotoxic potential (MAETP) of fluid components has been calculated and is reported in Appendix XIId. The uncertainty for the MAETP of these components is modelled through the uncertainty in release and this is the reason for the relatively large variation for fluid components.

Offshore discharge of cuttings waste, drilled with water-based/barite fluid
Reference: 1 metric tonne of cuttings treated, drilled with the water-based (WB) fluid system
Representative of general facility; Norway; 2007
Cuttings discharged to the marine environment at site

Table Xb-1: Direct emissions to ocean, per tonne water-based/barite cuttings *

Substance	kg	σ^2	Distribution
As	1,80E-04	6,4	log-norm.
Ba	2,13E-02	1,2	log-norm.
Cd	1,69E-04	7,3	log-norm.
Cr	1,98E-03	2,6	log-norm.
Co	1,80E-04	5,2	log-norm.
Cu	1,22E-02	1,7	log-norm.
Pb	1,37E-02	11,5	log-norm.
Ni	2,33E-04	4,5	log-norm.
V	5,42E-04	5,2	log-norm.
Zn	2,10E-02	11,3	log-norm.
Drill-string dope	2,74E-02	5	log-norm.
Casing dope	8,89E-04	5	log-norm.
Base-fluid	20,7	20	log-normal

* Calculated from an assumed fluid content. Inorganics are assumed to be released to the water column over time, according to the assumptions for geoavailable metal (Chapter 5.3). Marine aquatic ecotoxic potential (MAETP) of fluid components has been calculated and is reported in Appendix XIId. The uncertainty for the MAETP of these components is modelled through the uncertainty in release and this is the reason for the relatively large variation for fluid components.

Sensitivity analysis: the significance of organics in cuttings waste

The following information is available for the organic contents of cuttings waste

Table XI-1: Organics in oil-based cuttings, after treatment

Total hydrocarbons	500-2000	ppm in cuttings after TCC
Fraction PAH	2%	estimate
PAH in cuttings	10-40	calculated ppm PAH in cuttings
PAH in cuttings	0.01-0.04	kg per tonne cuttings

Table XI-2: Organics in water-based cuttings, prior to treatment

	mg per kg TS in	kg per tonne cuttings	σ^2
Acenaphtylene	0,14	3,68E-04	5
Acenaphtene	0,22	5,79E-04	5
Fluoren	0,29	7,63E-04	5
Phenanthrene	0,28	7,37E-04	5
Antracene	0,01	2,63E-05	5
Total solids (TS)	38 %	-	5

* Uncertainty is pure estimate, only used to test our sensitivity towards organic remains.

Our original inventories assume that these organics are degraded during the treatment
We investigate if this assumption has any significnce on the outcome.

Results from the sensitivity analysis - organics in water-based fluid

Table X1-3: Ratio for (Organics in WB fluid) / (WB/ilmenite fluid end-of-life)

FWAETP	3,23E-04	Organics / Onshore treatment; organics released to soil
MAETP	2,61E-08	Organics / Onshore treatment; organics released to soil
TETP	4,22E-05	Organics / Onshore treatment; organics released to soil
MAETP	2,04E-04	Organics / Offshore discharge; organics discharged to ocean

Conclusion: metals in cuttings waste outweight the eventual significance of organic substances.

Results from the sensitivity analysis - organics in oil-based fluid

Table XI-4: Ratio for (Organics in OB fluid) / (Metals in OB fluid)

FWAETP	35	Organics / Leaching of metals from OB/ilmenite fluid
MAETP	7,70E-03	Organics / Leaching of metals from OB/ilmenite fluid
TETP	1,28E-05	Organics / Leaching of metals from OB/ilmenite fluid

Conclusion: organic components in OB fluid may be significant for freshwater ecotoxicity according to our crude attempt at assessing the uncertaty in organic content. But, the oil that is used in the fluid is hydrotreated, meaning that the polyaromatic content is reduced compared to crude oil. We have not been able to identify figures for the actual content of PAH in the oil-based base fluid or oil-based drilling fluid.

Adjusted characterization factors for barite

The original characterization factors for barite in SimaPro are estimated from barium contents assuming complete release of Ba in Barite (BaSO4), as shown in Table XIIa-1 and XIIa-2 below.

Table XIIa-1: Marine aquatic ecotoxic potential (MAETP), fresh water aquatic ecotox (FWATP) and human toxicity potential (HTP) for barium (Ba) and barite as implemented in SimaPro for the CML2 method

category	MAETP	MAETP	FWETP	HTP	HTP
recipient	water	ocean	water	water	ocean
ratio *	1,700	1,706	1,701	1,698	1,702

Table XIIa-2 cont.: Content of barium (Ba) in barite

wt-% ratio	1,700

* Ratio of barium:barite

In order to have consistency for inventories that link to ecoinevnt processes, we use our inventory of long-term mobilizable metal in barite to calculate a corrected CF for barite that is equivalent to what is used for the leaching potential from barite in drilling operations modelled in this study, set equal to geoavilable metal (Chapter 5.3).

TableXIIa-3: Geoavailable metal in barite

Metal in barite					
	g per kg	kg per kg		g per kg	kg per kg
As	9,84E-04	9,84E-07	Cu	6,65E-02	6,65E-05
Ba	1,16E-01	1,16E-04	Pb	7,46E-02	7,46E-05
Cd	9,21E-04	9,21E-07	Ni	1,27E-03	1,27E-06
Cr	1,08E-02	1,08E-05	V	2,96E-03	2,96E-06
Co	9,84E-04	9,84E-07	Zn	1,14E-01	1,14E-04

TableXIIa-4: Adjusted characterization factors for barite, calculated from geoavailable metal contents; kg 1,4-DCB per kg barite.

recipient	MAETP *	FWATP *	HTP *
water	1,47E+02	3,29E+00	8,51E-02
ocean	3,10E+02	-	1,22E-01

Adaptation of characterization factors to offshore and local situation

Source: SSB (2006): Statistisk Årbok 2005. Statistics Norway, Oslo, Norway. Table 48: http://www.ssb.no/aarbok/tab/tab-048.html. Accessed 2. sept. 2007.

Health effects in Ecoindicator 99

The following adjustments are applied for human health effects for local air emissons
- Climate change, Radiation & Ozone layer are excluded
- Fate factor *(1/2) for respiratory emissions offshore
- Population numbers corrected for respiratory effects acording to local situation

Table XIIb-1: Adjustment factors for offshore emissions

	EI99	West	North
Population *	80	36	1,6
Fate factor **	0,5	0,5	0,5
Total factor	1	0,225	0,010

* The original population density is 80 cap per km2 in Ecoindicator 99. Norwegian densities are from SSB (2006); Norway west includes Rogaland and Hordaland, Norway north includes Nordland, Troms and Finnmark. Norway North is used in this
** Estimate, given that ship emissions occur off shore.

Table XIIb-2: Adjustment factors for onshore emissions

	EI99	West	North
Population *	80	36	1,6
Fate factor **	1	1	1
Total factor	1	0,450	0,020

* The original population density is 80 cap per km2 in Ecoindicator 99. Norwegian densities are from SSB (2006); Norway west includes Rogaland and Hordaland, Norway north includes Nordland, Troms and Finnmark. Norway North is used in this
** No adjustment made for the fate of emisisons occutting onshore

We have three sources of air emissions offshore or regionally in Norway:
- Offshore ship emissions: emissions inventory of Cooper and Gustavsson (2004)
- Onshore truck emissions: transport, lorry 16t/RER (ecoinvent process)
- Offshore diesel energy: diesel, burned in diesel-electric generating set (ecoinvent process)

A substance list is compiled for these three sources. Adjusted characterization factors implemented in in SimaPro for the Ecoindicator H method are listed in Table XIIb-3.

Table XIIb-3: Adjusted characterization factors for respiratory health effects in Ecoindicator 99, Hierarchical perspective (H)

	CAS	EI-99 DALY / kg	N-offshore* DALY / kg	N-onshore* DALY / kg
Respiratory organics				
Benzene	000071-43-2	4,68E-07	4,68E-09	9,36E-09
Benzo(a)pyrene	000050-32-8	2,10E-06	2,10E-08	4,20E-08
Dioxines		2,10E-06	2,10E-08	4,20E-08
Methane	000074-82-8	1,28E-08	1,28E-10	2,56E-10
Methane, fossil	000074-82-8	1,28E-08	1,28E-10	2,56E-10
nmVOC		1,28E-06	1,28E-08	2,56E-08
PAH	130498-29-2	2,10E-06	2,10E-08	4,20E-08
Polychlorinated biphenyls	001336-36-3	2,10E-06	2,10E-08	4,20E-08
Toluene	000108-88-3	1,36E-06	1,36E-08	2,72E-08
Xylene	001330-20-7	2,21E-06	2,21E-08	4,42E-08
Respiratory inorganics				
Ammonia	007664-41-7	8,50E-05	8,50E-07	1,70E-06
Nitrogen dioxide	010102-44-0	8,87E-05	8,87E-07	1,77E-06
Nitrogen oxides	011104-93-1	8,87E-05	8,87E-07	1,77E-06
Particulates, < 10 um		3,75E-04	3,75E-06	7,50E-06
Particulates, < 2.5 um		7,00E-04	7,00E-06	1,40E-05
Particulates, > 2.5 um, and < 10um		3,75E-04	3,75E-06	7,50E-06
Sulfur dioxide	007446-09-5	5,46E-05	5,46E-07	1,09E-06
Sulfur oxides		5,46E-05	5,46E-07	1,09E-06

* N-offshore refers to offshore emissions in th north of Norway , N-onshore refers to onshore emissions in the north of Norway

Impact categories in CML2

As an estimate of the impact of offshore emissions relative to onshore emissions, a general factor of 0.5 is used to down-scale the effect of offshore emissions for eutrophication and acidification in the CML2 baseline method. For human toxicity, same approach is applied as previously for Ecoindicator 99 for substances with respiratory

TableXIIb-4: Adjusted characterization factors for eutrophication; emissions to air according to the CML2 baseline method

Substance	CML2	N-offshore	N-onshore
Ammonia	3,50E-01	1,75E-01	3,50E-01
Nitrogen dioxide	1,30E-01	6,50E-02	1,30E-01
Nithogen oxides	1,30E-01	6,50E-02	1,30E-01

TableXIIb-5: Adjusted characterization factors for acidification; emissions to air according to the CML2 baseline method

Substance	CML2	N-offshore	N-onshore
Ammonia	1,60	0,80	1,60
Nitrogen dioxide	0,50	0,25	0,50
Nithogen oxides	0,50	0,25	0,50
Sulfur dioxide	1,20	0,60	1,20
Sulfur oxides	1,20	0,60	1,20

TableXIIb-6: Adjusted characterization factors for human toxicity; emissions to air according to the CML2 baseline method

Substance	CML2	N-offshore	N-onshore
PAH	5,72E+05	5,72E+03	1,14E+04
Ammonia	1,00E-01	1,00E-03	2,00E-03
Nitrogen dioxide	1,20E+00	1,20E-02	2,40E-02
Nitrogen oxides	1,20E+00	1,20E-02	2,40E-02
Particulates, < 10 um	8,20E-01	8,20E-03	1,64E-02
Particulates, < 2.5 um	8,20E-01	8,20E-03	1,64E-02
Particulates, > 2.5 um, and < 10	8,20E-01	8,20E-03	1,64E-02
Sulfur dioxide	9,60E-02	9,60E-04	1,92E-03
Sulfur oxides	9,60E-02	9,60E-04	1,92E-03

Alternative methods to assess metal ecotoxicity in LCIA

Sediment limit values: used in risk assessment of marine sediments
Source: SFT. 2007. Veileder for klassifisering av miljøkvalitet i fjorder og kystfarvann - utkast 15.02.07. SFT: Oslo, Norway.

Table XIIc-1: Marine sediment risk limits. Limit values for sediments (Class III: susceptible to chronic effects at long-term exposure); mg per kg sediments (dry wt)

	low	high	geomean	1/geomean *
As	52	190	99	1,01E-02
Ba	-	-	-	-
Cd	2,6	17	7	1,50E-01
Cr	560	20 000	3 347	2,99E-04
Co	-	-	-	-
Cu	51	120	78	1,28E-02
Pb	83	700	241	4,15E-03
Ni	43	120	72	1,39E-02
V	-	-	-	-
Zn	360	1 800	805	1,24E-03

* geomean = geometric mean. The inverse of geomean is used as the effect factor for metal ecotoxicity by sediment presence.

Soil limit values: used in soil waste management
Source: Ministry of the Environment. 2004. Regulation concerning the limitation of pollution (Forurensningsforskriften). MD: Oslo, Norway.

Table XIIc-2: Soil limit values; soil for most sensitive uses. Regulation Section 2, App. 1; mg/kg

	Norm	1/Norm
As	2	5,00E-01
Ba	-	-
Cd	3,0	3,33E-01
Cr	25	4,00E-02
Co	-	-
Cu	100	1,00E-02
Pb	60	1,67E-02
Ni	50	2,00E-02
V	-	-
Zn	100	1,00E-02

* The inverse of the norm value is used as the effect factor for metal ecotoxicity by soil presence.

Industry limit values: the approach for best available technique (BAT)
Source: EIPPCB (2005). Integrated pollution prevention and control. Reference document on economic and cross-media effects. European IPPC Bureau: Sevilla, Spain.

Table XIIc-3: Factors to assess aquatic ecotoxicity within the cross-media assessment framework*

	PNEC	EF (LCA)
	mg/ltr	ltr/mg
As	2,40E-02	4,17E+01
Ba	5,80E-02	1,72E+01
Cd	3,40E-04	2,94E+03
Cr	8,50E-03	1,18E+02
Co	2,60E-03	3,85E+02
Cu	1,10E-03	9,09E+02
Pb	1,10E-02	9,09E+01
Ni	1,80E-03	5,56E+02
V	8,20E-04	1,22E+03
Zn	6,60E-03	1,52E+02

* PNEC = predicted no-effect concentration. EF (LCA) is the effect factor in LCA; equal to the inverse of PNEC.

Policy measures: priority substances in policy
Source: SFT. 2004. Prioriterte miljøgifter. Status i 2001 og utslippssprognoser. SFT: Oslo, Norway.

Table XIIc-4: Priority metals in Norwegian policy *

	Priority class			
	Class A	Class B	New subst.	All classes
As			x	1,00
Ba				0,00
Cd		x		1,00
Cr		x		1,00
Co				0,00
Cu		x		1,00
Pb		x		1,00
Ni				0,00
V				0,00
Zn				0,00

* Class A: emissions to be significantly reduced and best avoided by 2005; Class B: emissions to be reduced by 50-90% by 2010; New subst.: emissions to be significantly reduced by 2010

Summary table: methods to assess metal ecotoxicity

TableXIIc-5: Alternative characterization factors for metal ecotoxicity *

	Mar. sed. risk	Soil limit	BAT-REF	Priority metals
Abbrev.	MSR	SLV	BREF	PM
Recipient	ocean	soil	water	all
As	1,01E-02	5,00E-01	4,17E+01	1,00
Ba	-	-	1,72E+01	0,00
Cd	1,50E-01	3,33E-01	2,94E+03	1,00
Cr	2,99E-04	4,00E-02	1,18E+02	1,00
Co	-	-	3,85E+02	0,00
Cu	1,28E-02	1,00E-02	9,09E+02	1,00
Pb	4,15E-03	1,67E-02	9,09E+01	1,00
Ni	1,39E-02	2,00E-02	5,56E+02	0,00
V	-	-	1,22E+03	0,00
Zn	1,24E-03	1,00E-02	1,52E+02	0,00

* MSR applies to the marine environment only; SLV applies to soil toxicity only; BREF applies to generic aquatic ecotoxicty; PM applies on generic level

Marine aquatic ecotoxicity of water-based fluid components

References:
Huijbregts MAJ (1999): Priority assessment of toxic substances in the frame of LCA. Development and application of the multi-media fate, exposure and effect model USES-LCA. Interfaculty department of environmental science, University of Amsterdam, Amsterdam, The Netherlands.
Aldenberg T, Jaworska JS (2000): Uncertainty of the Hazardous Concentration and Fraction Affected for Normal Species Sensitivity Distributions. Ecotoxicology and Environmental Safety 46:1-18.
Novatech (2006): CHEMS - Harmonized offshore chemical notification format. Novatech AS (www.novatech.no), Stavanger, Norway

Effect factors are calculated according to Aldenberg and Jaworska (2000), with characterization factors calculated by a simplistic approach using the estimated marine residence time as a function of degradation, normalized to the MAETP of 1,4-DCB according to the framework of Huijbregts (1999). Substance information is extracted from the CHEMS database (Novatech 2006). MAETP = marine aquatic ecotoxic potential, in units of kg 1,4-DCB equivalents per kg component

Table XIId-1: Marine aquatic toxicity of base fluid	wt-%	MAETP
Base fluid	1,00	5,58E-03

Table XIId-2: Marine aquatic toxicity of drill-string dope	wt-%	MAETP
Component A	0,20	1,42E-04
Component B	0,20	3,33E-04
Component C	0,03	1,95E-04
Component D	0,01	7,27E-04
Other components	0,57	0,00E+00
Drilling-string dope in total	1,00	1,05E-04

Table XIId-3: Marine aquatic toxicity of casing dope	wt-%	MAETP
Component A	0,45	7,27E-04
Other components	0,55	0,00E+00
Casing dope in total	1,00	3,27E-04

Appendix B – Paper 1

Johan Pettersen, Glen P. Peters and Edgar G. Hertwich (2006): Marine ecotoxic effect of pulse emissions in life cycle assessment. *Environmental Toxicology and Chemistry* **25**(1): 297–303

Paper 1 is not included due to copyright.

Appendix C – Paper 2

Johan Pettersen and Edgar G. Hertwich (In review): Occupational health impacts – offshore crane-lifts in life-cycle assessment. Submitted to *International Journal of LCA*

Paper 2

Occupational Health Impacts: Offshore Crane-Lifts in Life Cycle Assessment

In review at *The International Journal of Life Cycle Assessment*

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ABSTRACT

Background, Aim and Scope. Crane-lifts are a major cause of accidents on offshore oil and gas (O&G) rigs. Health impacts from crane-lift accidents should be included in comparative life-cycle assessment (LCA) of O&G technologies if the alternatives differ in the use of crane-lifts. Recently, several indicator sets have been published for occupational health impacts on industry sector level. Although easily attainable, sector level indicators in many cases do not allow product system comparisons as they lack the required foreground system resolution.

Materials and methods. Accident records for mobile offshore petroleum installations were used to develop an empirical occupational health indicator for crane-lifts in LCA. Probabilistic parameters were introduced in the procedure and results were calculated by Monte Carlo simulations. Health impacts were quantified in disability adjusted life years (DALY) by classification of health outcomes based on the description of events offered by the source data. The characterization factor for offshore crane-lifts is applied in three comparisons to evaluate the significance of crane-lifts to human health impacts from drilling technology.

Results. The mean occupational health impact per crane-lift is $4.0 \cdot 10^{-6}$ DALY, with cumulative percentiles $\{P_{2.5}, P_{50}, P_{97.5}\} = \{5.4 \cdot 10^{-7}, 2.8 \cdot 10^{-6}, 1.5 \cdot 10^{-5}\}$. Analogously, the fatal accident frequency is described by $\{P_{2.5}, P_{50}, P_{97.5}\} = \{7.7 \cdot 10^{-9}, 3.9 \cdot 10^{-8}, 2.0 \cdot 10^{-7}\}$, with mean $5.5 \cdot 10^{-8}$ lives lost per crane-lift.

Discussion. The uncertainty in the results is caused mainly by the random nature of accidents; i..e, variability in accident frequency. The influence of external factors such as weather conditions and rig space limitations on accident probability was not investigated. Applications of the characterization factor indicate that although crane-lifts may not be significant to the overall health impact of the life-cycle of drilling fluids, they are important to the occupational safety for employees on offshore drilling rigs. A comparative LCA of technologies for loading and off-loading drilling wastes to/from drilling rigs shows that a recently developed hydraulic system performs better than the traditional crane-lift alternative in terms of human health impacts.

Conclusions. Although relatively large, the uncertainty found for health impacts from crane-lifts is less than what is indicated for other human health impact chains. The health burden from recoverable injuries was insignificant for the total burden from crane-accidents.

Recommendations and Perspectives. In further work of quantifying occupational health impacts in DALY using accident statistics it is advised to see if records of non-recoverable injuries (fatalities and amputation cases) can be used to simplify the damage assessment procedure.

Keywords: Crane-lifts; disability adjusted life years; fatality; injury; life-cycle impact assessment; risk; working environment; fatal accident rate (FAR), Monte Carlo; health

BACKGROUND, AIM AND SCOPE

This paper is part of an effort to use life-cycle assessment (LCA) in the evaluation and selection of drilling fluid chemicals and drilling waste technology. Drilling fluids are used in oil and gas (O&G) drilling operations to move rock carvings out of the well, stabilize the well walls, and to cool and lubricate the drill bit. The fluid composition at any drill site varies according to geology and local regulations concerning use of chemicals and treatment of drilling wastes. The present focus of regulations is directed towards impacts at the drill site or in relation to the treatment of waste. There is a growing understanding that environmental interventions occur throughout the life-cycle of drilling fluids, and that an overall evaluation of drilling technology must include chemical production, use and reuse value of fluids, as well as waste treatment technologies. The large variation in possible fluid compositions and waste treatment options, combined with differences in infrastructure for treatment of drilling waste present at potential drill sites, calls for the use of a holistic tool for the environmental assessment of drilling technologies.

Two aspects have emerged as especially important in the regulation of offshore O&G activities in the North Sea: ecotoxic impacts from planned and accidental spills, and the safety of the offshore workforce. Both aspects have to be treated within the LCA framework if LCA is to be used by offshore operators in communication with external stakeholders. Discharges during drilling are intermittent, and the issue of marine pulse emissions in LCA is discussed by Pettersen et al. (2006) who investigate the significance of assuming marginal effect using potentially affected fraction of species (PAF) for multiple substances in life-cycle impact assessment of marine discharges. They conclude that the concentration-additive approach used in Eco-indicator 99 (Goedkoop et al. 1998) is robust for pulse emissions, while the response-additive approaches of Huijbregts et al. (2002) and van de Meent and Huijbregts (2005) potentially overstates the ecotoxic impact by several orders of magnitude.

Mechanical lift operations cause a large fraction of accidents on offshore O&G units. In the North Sea, they constitute 25% and 40% of all reported incidents, and 50% and 68% of incidents with person injuries on fixed and mobile units respectively (DNV 2005a, 2005b). Crane-lifts are hence one of the main drivers for accidents in the offshore O&G industry. This paper is an effort to include health impacts from crane-lifts in LCA by development and application of a characterization factor for the health impacts caused by crane-lift accidents.

Life-cycle assessment is conventionally concerned with impacts caused by product systems upon the outside world. The consistent exclusion of internal impacts in LCA is artificial when environmental mechanisms in principal are the same (e.g. occupational toxic exposure) and in any case clearly opens up possibilities for system sub-optimization. An example of the value of complementing LCA with an assessment of occupational health aspects was recently presented in this journal by Schmidt et al. (2004a) for house insulation alternatives.

Poulsen and Jensen (2005) summarize recent efforts to include occupational health in LCA. They recommend that the practitioner select the method depending on the goal and scope of the assessment; either by incorporating working environment into the conventional life-cycle assessment framework, or discussing it separately within the life-cycle approach. If the purpose of the assessment is to quantify trade-offs introduced by changes in technology, it is our view that internal and external impacts should be in compatible metrics throughout the life-cycle. For instance, reduction in crane-accidents can be realized through better safety management. Still, principally it is achieved by replacing cranes with other means of loading of cargo. In order to assess the performance of alternative technologies, human health impacts (occupational and external) from all options should be in the same metric. This is offered by the disability adjusted life years (DALY). Developed for the World Bank and the World Health

Organization (Murray and Lopez 1996) and originally designed for health economics, the DALY concept has been used for various impact chains in LCIA. Applications include human-toxicity (Crettaz et al. 2002, Hofstetter 1998, Huijbregts et al. 2004, Goedkoop et al. 1998, Meijer et al. 2005, Pennington et al. 2002), ionizing radiation (Frischknecht et al. 2000) and road noise (Müller-Wenk 2004); as well as occupational health impacts in the US input-output (I/O) table (Hofstetter and Norris (2003).

Occupational health impacts may be included in LCIA by relating records of fatalities, injuries and illnesses to product outputs from sectors or single companies (Poulsen and Jensen 2005, Hauschild and Wenzel 1998). Occupational health impacts may be quantified as direct impacts occurring within the sector or company (e.g. Schmidt et al. 2004b, Hauschild and Wenzel 1998, Antonsson and Carlsson 1995), or including repercussions in the whole economy (Hofstetter and Norris 2003). The latter approach requires the use of an I/O model. Hybrid-LCA, as described by Heijungs and Suh (2002), accommodates combination of process and sector data in LCA. While I/O indicators are easily available, comparative LCAs call for quantification of health impacts on unit process level for the foreground system. The detail with which the offshore O&G industry reports accidents allows establishment of a quantitative relationship between unit processes and injury characteristics such as frequency and health consequence. In this work we use data reported by the O&G industry to develop an empirical characterization factor for the human health impacts from crane-lifts. Damage to human health is quantified in DALY.

The source data originates from the North Sea area and results are principally to be used in the context of offshore O&G activities. The factor is fit for use in risk assessment of offshore processes since the methodology that we apply draws on the methods of this field.

The significance of crane-lifts to occupational health is illustrated through application of the characterization factor in three comparisons. First, health impacts occurring from crane-lifts compared to the total occupational health burden offshore; second, crane-lifts compared with human health impacts from other unit processes in the drilling fluid life-cycle; and third, a comparative assessment of technologies for loading of drill cuttings aboard a service vessel, the options being crane-lifts or a recently developed hydraulic system.

1 MATERIALS AND METHODS

The UK Health & Safety Executive has compiled incident records for floating and fixed offshore petroleum units on the UK continental shelf for the period 1980-2003 (DNV 2005a, 2005b). Data for floating (i.e. mobile) units were selected in this work as mobile units normally are employed when drilling in new areas. Every incident is recorded with year of incident, rig type, mode of operation, number of people injured, and a brief description of the event. The data-base of 3,105 incidents, of which 817 resulted in person injury, is the most complete compilation of offshore accident records for the period. Unfortunately, the classification of accidents in the data-base groups all incidents from lifting operations into the class of crane-lift incidents. Separation between accidents related to lifts performed with cranes and lifts performed with other equipment, such as the drilling derrick or draw-works, must therefore be done before the data can be used to quantify the impacts caused by crane-lifts.

According to the database compiled by DNV (2005a), there were 588 incidents which resulted in injury to personnel on floating (i.e. mobile) units in the period 1980-2003. Of these, 399 are classified as caused by or involving lifting equipment. In this work we are only interested in injuries caused by crane-lifts, so the cases involving derrick operations and draw-works were excluded based on the description of events given in the accident records. This resulted in a set of 165 cases of crane-lifts causing injury to personnel. Text searches in the cases classified as having zero personnel injuries identified 12 additional cases. The end set consisted therefore of 177 cases from the period 1980-2003. These are hereafter referred to as crane-lift injury-events (CIE) and form the basis for both exposure and effect assessment.

Impact assessment in LCA makes a separation between exposure and effect. In our case, exposure represents the frequency with which crane-lift accidents occur. The source data reports accident frequency per rig year. In order to establish the frequency of accidents with personnel injuries per crane-lift, we have to establish a connection between accident frequency and stressor activity. The stressor that we consider in this case is one single crane-lift. The relationship was achieved by estimation of the average annual number of lifts made on a subset of offshore rigs. Homogeneity in the source data was ensured by restricting it to a single type of mode of action within a select group of rig types and to the period 1990-2003. Exposure assessment therefore was based on semi-submersible (SS) and jack-up (JU) rigs in drilling mode. The drilling operation is a fairly generic mode of operation, and semi-submersible (SS) and jack-up (JU) rigs are similar in that they both are mobile and are predominantly used in drilling operations. Together, SS and JU rigs represent the bulk of rigs used in exploration drilling.

While source data variability is a problem in exposure assessment, it is a source of validity in accident outcome compilation. Accounts of typical health outcomes represent the effect assessment in our framework. To achieve data that is representative of the outcome of crane-lift accidents, all crane-lift accidents recorded in the period 1980-2003 were used in the effect assessment. Although limited to crane-lifts, the data encompasses all rig types and modes of operation. Disability adjusted life years (DALY), following the framework of Murray and Lopez (1996), was used in the damage assessment. Age weighting and discounting of life years was not performed.

Monte Carlo analysis has become the norm when accounting for uncertainty in LCA (e.g. Geisler et al. 2005, Ciroth et al. 2004) and LCIA (e.g. Huijbregts et al. 2004, Hertwich et al. 2000), and was also used here. With the exception of the duration for recoverable injuries and the disability weight of accident outcomes, all parameters were treated as independent distributions.

The next sections describe the impact assessment procedure in detail. Section 1.2 outlines the exposure assessment procedure while the method for effect and damage assessment is described in Section 1.3. Results from the Monte Carlo analysis and applications of the characterization factor are given in Section 2.

1.1 Exposure assessment

A homogenous dataset helps reduce uncertainty in the exposure assessment. We selected semi-submersible (SS) and jack-up (JU) rigs for the exposure assessment as they have similar activity profiles and represent the main share of mobile drilling rigs used in UK waters. They also represent the main share of crane-lift accidents. Both are employed in drilling operations and perform a large number of crane-lifts per hour. Crane-lift injury-events on SS and JU units were extracted from the dataset and combined with years of active drilling on SS and JU rigs. Drilling years were calculated for UK waters using data from RigPoint (ODS-Petrodata 2005). A log-normal distribution was fitted to the frequency of injury-events per drilling year in the period 1990-2003, as illustrated in Fig. 1. The result is a distribution for the number of CIE per year of active drilling.

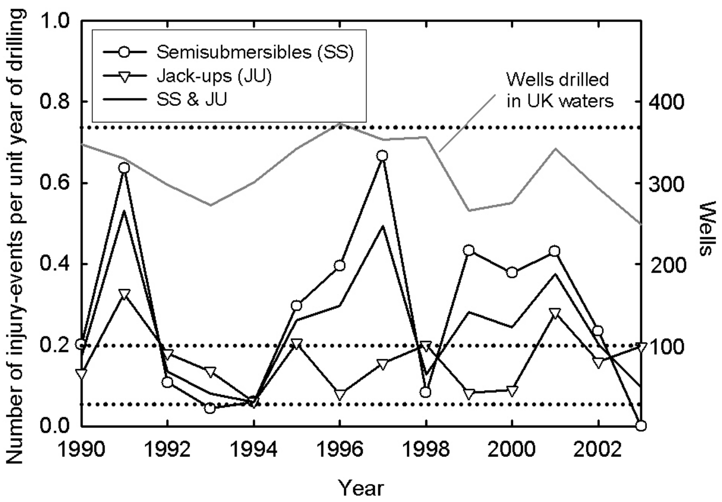

Fig. 1: Injury-events per unit year for jack-ups (JU) and semi-submersible (SS) drilling rigs. Dotted lines indicate the geometric mean and boundaries of the interval of 95% confidence for the distribution fitted to the aggregated scores for JU and SS rigs. The large increase of reported injury-events in 1997 is though to be an artifact of the introduction of the RIDDOR95 reporting scheme in April 1996. Drilling activity figures are from the UK Department of Trade and Industry (http://www.dti.gov.uk/) and include all well and unit types

Normalization to CIE per crane-lifts was achieved by estimating the number of lifts performed per hour. Crane-lift intensity on the rig varies greatly, from zero lifts per hour in quiet periods to peaks of up to fifty during loading of supplies. On average, approximately 8-10 crane-lifts are performed per hour (Eikill GO, Statoil ASA, personal communication). This is within the interval reported for crane-lift intensity on fixed installations by Safetec (2005). Taking into considerations that the intensities average out over one rig year, a log-normal distribution with a mean of 9 and a 99th percentile of 25 was assumed. Values >30 crane-lifts per hour were removed from the set. This gives a quite wide distribution, equivalent to our uncertainty in the average crane-lift intensity.

Equations used in the assessment procedure are listed in Table 1. Note that equation 1 is balanced for 8760 hours per year. Parameters used in Monte Carlo simulations are listed in Table 2.

1.1 Effect and damage assessment

Health outcomes found in the 177 CIEs were classified based on the description of events. The result is presented in Table 3. Equations used in the effect and damage assessment are listed in table 1; i.e., equations 2 and 3. Remaining parameters used in the Monte Carlo simulations are listed in Table 2. Disability weights and the durations for the recoverable injuries were modeled as defined parameters; i.e., set constant.

Given its expected influence on the end result, we find it necessary to discuss the remaining lifetime separately. A program initiated by the Norwegian Oil Industry Association (OLF; `http://www.olf.no/arbeidsliv/aldringoghelse/`) investigated the age distribution of the Norwegian offshore workforce. Average age was between 45-50 years for the various operators, and one of the operators (Norsk Hydro) reported a female representation of about 20%. Several studies have reported higher accident rates among young employees offshore compared to more experiences employees (e.g. Forbes 1997, Mueller et al. 1987). In order to include this aspect, the age distribution reported by Forbes (1997) for the age at injury was preferred over an age distribution of the entire workforce reported by, e.g., OLF. The age distribution was combined with life tables for males in the UK reported by GAD (2006) for 2002. The result is an average remaining lifetime at the time of the accident of 47 years, with standard deviation 6.1

years. Male life expectancy was used in the simulations from the observation that all cases found in the production of Table 3 that specified gender, indicated male victims.

Table 1: Equations used to assess exposure, effect and health damage

Equation		Metric
1	$F = \frac{u}{8760c}$	CIE per crane-lift
2	$E_i = \frac{n_i}{n_t} = \frac{n_i}{177}$	Outcome i per CIE
3	$D = \sum_i \left(E_i d_i w_i\right)$	DALY per CIE
4	$Q = F \cdot D = \frac{u}{8760c} \frac{1}{n_t} \sum_i \left(n_i d_i w_i\right)$	DALY per crane-lift

where
i = indicates health outcome of type i; see table 3 for list
F = number of cases of injury to human health per crane-lift
u = CIE per year of drilling
c = crane-lifts per hour (note: 8760 hours per year)
E_i = number of health outcomes of type i per CIE; i.e., effect factor for health outcome i
n_t = total number of CIE = 177
n_i = number of health outcomes of type i in the total set of CIE
D = DALY per CIE; i.e., damage factor for health outcome i
d_i = duration of health outcome i
w_i = disability weight for health outcome i
Q = damage to human health per crane-lift
CIE = crane-lift injury-event; DALY = disability adjusted life years

Table 2: Parameters in the Monte Carlo simulations

Parameter	Distribution[a]
u	L[ξ, φ] = [-1.62, 0.67][b]
c	L[ξ, φ] = [2.1, 0.50][c]
n_t	177
n_i	U[Certain cases, Certain cases + Potential cases]
$d_{lifelong}$	N[μ, σ] = [47.0, 6.1]
$d_{fracture}$	As listed by Murray and Lopez (1996, Annex table 3)
d_{minor}	0.024[e]

[a] L = log-normal distribution, U = uniform distribution, N = normal distribution

[b] Fitted to CIE frequencies for semi-submersible and jack-up rigs in the period 1990-2003. Values are in natural log-scale

[c] Values are in natural log-scale (lifts per hour)

[d] The recovery period specified by Murray and Lopez (1996, Annex table 3) for Open wounds

Table 3: Injuries found in 177 crane-lift injury-events

Health outcome [i]	Cases[a] [n_i]	Weight[b] [w_i]
Fatalities	2 (+2)	1.000
Amputation – thumb	1 (+1)	0.165
Amputation – finger	4 (+5)	0.102
Amputation – toe	0 (+2)	0.078
Amputation – foot	1 (+0)	0.300
Fracture – face bones	0 (+4)	0.223
Fracture – rib or sternum	0 (+3)	0.199
Fracture – pelvis	1 (+2)	0.247
Fracture – clavicle, scapula, or humerus	1 (+1)	0.153
Fracture – radius or ulna	1 (+2)	0.180
Fracture – hand bones	9 (+16)	0.100
Fracture – patella, tibia, or fibula	3 (+8)	0.271
Fracture – ankle	1 (+4)	0.196
Fracture – foot bones	1 (+14)	0.077
Minor injuries	88 (+64)[c]	0.108[d]

[a] On format: Certain cases (+ Potential cases)

[b] Disability weights from Murray and Lopez (1996, table 4.4)

[c] Modeled so that $\Sigma(n_i)|_i = n_t = 177$

[d] Assumed with equal weight to Open wounds

2 RESULTS AND DISCUSSIONS

2.1 Injury-events per crane-lift

In order to calculate the number of injury-events per crane-lift, 200,000 Monte Carlo simulations of equation 1 in Table 1 were performed according to the distributions listed in table 2. Mean value of the resulting distribution is $3.9\cdot10^{-6}$ injury-events per crane-lift with cumulative percentiles $\{P_{2.5}, P_{50}, P_{97.5}\} = \{5.5\cdot10^{-7}, 2.8\cdot10^{-6}, 1.4\cdot10^{-5}\}$.

Knowing that approximately 90% of the CIEs indicate falling objects as the secondary cause to the accident, we conclude that the injury-rate from falling objects per lift (i.e. dropped load) estimated in this work fits well with the frequency of dropped load used in risk assessment in offshore engineering; e.g. $2\cdot10^{-5}$ dropped objects per lift indicated by Mazzola (2000). Some discrepancy is expected between these two results as the rate found here includes crane-lifts only while previous estimates have been based on accidents caused by all types of lifting equipment. In addition, the factor quantified here only includes incidents with injuries to personnel.

Sample correlation coefficients were calculated according to Morgan and Henrion (1990, p 208) for the contribution to injury-event frequency from CIEs per unit per year (u, 0.74) and crane-lifts per year (8760·c, -0.42). The results show that the uncertainty in the results is dominated by the random nature of accidents (variability in u) and not uncertainty in the estimation parameter c.

2.2 Health impact per injury-event

Disability adjusted life years per injury-event was calculated by 200,000 Monte Carlo simulations of equation 3 in Table 1. Years lost due to premature death and disability adjusted life years from amputations are illustrated in Fig. 2. The average contribution from mortality is 65.5% of the total burden on average, while amputation cases on average account for 34.0% of the total burden from crane-accidents. This leaves 0.5% for the fracture cases and minor injuries together. The result corresponds with the findings of Hofstetter and Norris (2003, Appendix pp. 10) who concluded that about two thirds of the burden of disease from occupational injuries in the economy is due to fatalities. It can be added to this that in this particular case only lifelong disabilities were found significant to the total health burden. This is not unexpected given the duration of the lifelong injuries compared to the recoverable injuries.

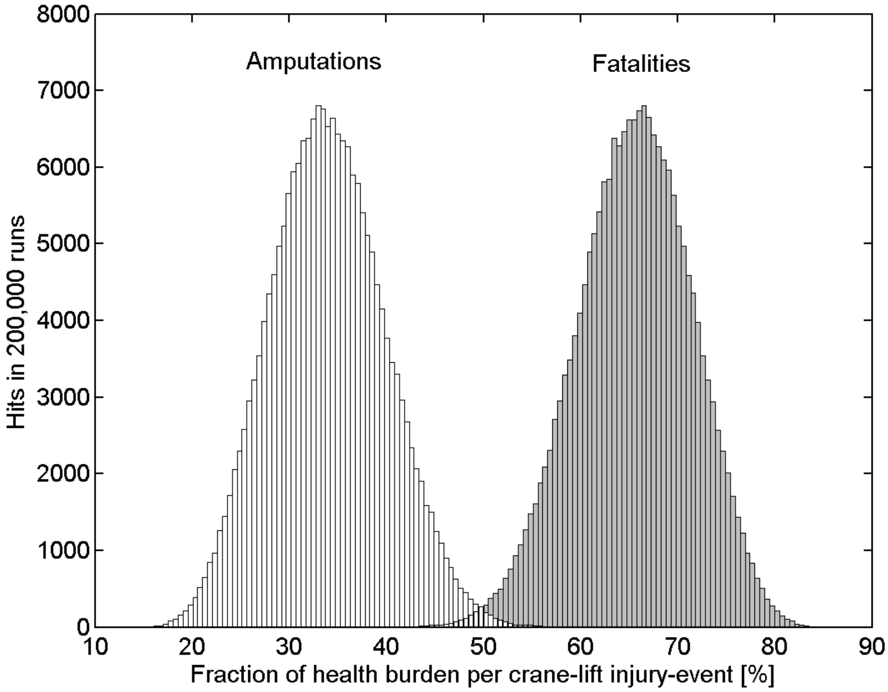

Fig. 2: Contribution from mortality and amputation cases

Discounting would reduce the importance of lifelong injuries for the burden per crane-lift. In further work of quantifying occupational health impacts in LCIA with DALY using statistical records it is advisd to see if records of lifelong injuries could be used to simplify the effect assessment procedure.

2.3 Health impact per crane-lift

A set of 200,000 Monte Carlo runs of equation 4 in Table 1 gave an average health damage per crane-lift of $4.0\cdot10^{-6}$ DALY, with cumulative percentiles $\{P_{2.5}, P_{50}, P_{97.5}\} = \{5.4\cdot10^{-7}, 2.8\cdot10^{-6}, 1.5\cdot10^{-5}\}$. The final distribution is illustrated in Fig. 3.There is significant uncertainty in the result. The 95% confidence interval spans a factor of plus/minus 5 from the median, corresponding approximately to the variation in the number of injury-events per crane-lift

The probability of fatal accidents is often used in risk assessment of technical systems. The mean value for crane-lifts is $5.5\cdot10^{-8}$ fatal accidents per lift, with cumulative percentiles $\{P_{2.5}, P_{50}, P_{97.5}\} = \{7.7\cdot10^{-9}, 3.9\cdot10^{-8}, 2.0\cdot10^{-7}\}$.

The uncertainty in the results compares well to what is found in other methods for quantification of human health impacts in LCIA. For instance, Hertwich et al. (2000) show that parameter uncertainty alone produces a ratio of 10 to 10^3 between the 90th and 10th percentiles in potential doses in human exposure models. Most impact assessment methods show uncertainty in their results by use of σ^2, indicating that a log-normal distribution is assumed. The factor σ^2 in such cases is the factor which, if multiplied or divided by the expected geometric mean, gives the boundaries for the interval of 95% confidence. Huijbregts et al. (2004) indicate a σ^2 of 5 (carcinogenic) and 11 (non-carcinogenic) for human health combined damage and effect factors for toxic substances. Frischknecht et al. (2000) estimate a σ^2 from 15^2 to 65^2 for the human health damage from ionizing emissions depending on substance and emission scenario. Hofstetter (1998) reports σ^2 to be from 15^2 to 50^2 for health damages from the respiratory effect of various inorganic substances. Although these values are reported for different parameters in health damage models in LCIA, and examples of less uncertainty exist; e.g. Müller-Wenk (2004) who indicated an uncertainty in scores for DALY from road noise of plus/minus a factor 2, the 95% confidence intervals in the final characterization factors typically span 1 to 3 units of magnitude.

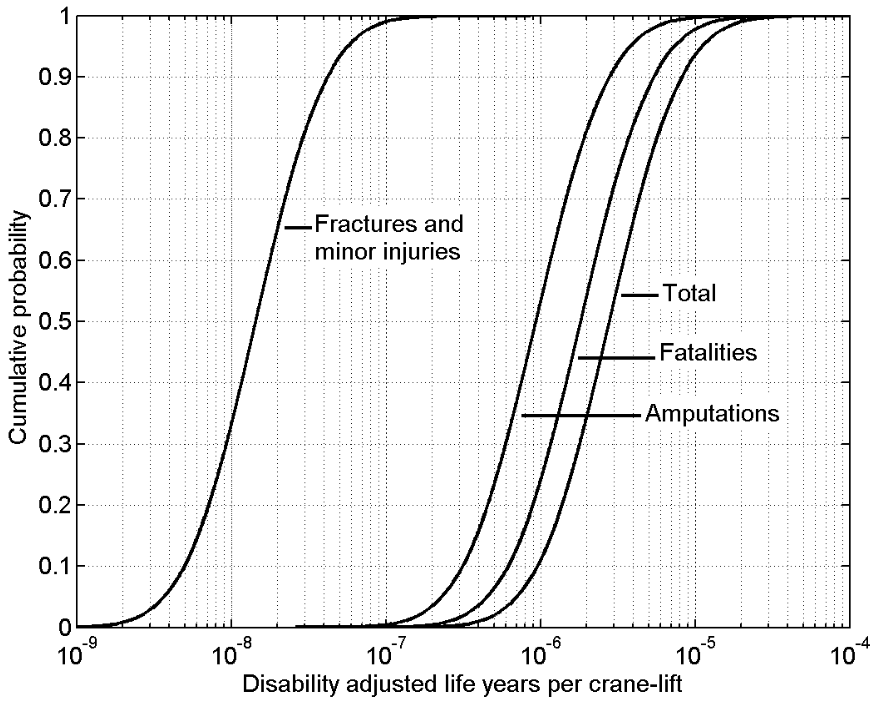

Fig. 3: Health burden per crane-lift from recoverable injuries, amputation cases, fatalities and in total

Sample correlation of the final damage factor and input variables was calculated according to Morgan and Henrion (1990, pp 208). Results showed a linear contribution of 0.13 from health outcomes in total (DALY per injury-event) and -0.98 from the distribution of injury-events per unit per year, pointing to the conclusion that the distribution of health burden per accident contributes less to the uncertainty in the result than the distribution of accident frequency.

2.4 Applicability of the factor

Injury-event frequency depends on the context in which lift operations are performed. Placing of equipment during operations inherently is a more complex operation than simple loading of containers. Other influencing factors include weather conditions, space limitations on rig and obstructions in the lift zone (which may differ from rig to rig), stress level depending on drilling speed and technical challenges, etc. This must be kept in mind when using the factor. From the assumptions in the characterization procedure, application of the indicator should be restricted to crane-lifts on offshore drilling units, possibly also to mobile rigs.

2.5 Crane-lift significance

With the indicator developed for health impacts from crane-lifts it is possible to compare crane-lifts with other health impacts from offshore operations. Three comparisons are investigated in the following sections, illustrating the significance of crane-lifts to human health impacts on the rig (i.e., employee safety levels), as a process in the end-of-life of drilling fluids, and in the selection of loading technology alternative.

2.5.1 Employee safety

In order to compare the significance of the estimated characterization factor with the industry data of Hofstetter and Norrris (2003), a simplistic LCA reference stream of 1 day of drilling is defined. A mean number of approximately 10 lifts per hour in active operations was assumed earlier in this paper, accounting to 240 lifts per 24-hour period. With the confidence interval estimated for DALY per lift, the daily burdens from crane-lifts account to between 0.00013 and 0.036 DALY.

Average day-rate for JU and SS rigs in UK waters in 1997; the source year for the I/O transactions of Hofstetter and Norris, was US$ 82,200 (ODS-Petrodata, 2005). Unfortunately, direct (0^{th} tier) burdens from drilling (BEA sector 110601) are not part of the dataset of Hofstetter and Norris. Petroleum and mineral extraction services (BEA 110602) and engineering, architectural, and surveying services (BEA 730302) are deemed the closest proxy sectors, resulting in daily burdens of 0.074 (mineral extraction) and 0.008 DALY (engineering services) per day. This indicates that crane-lifts may constitute a significant part of the total occupational health burden for offshore workers.

2.5.2 End-of-life contribution

The Barents Sea was recently opened for petroleum drilling under requirements that drilling chemicals are not discharged to the sea. Re-injecting to sub-sea formations the rock phase carved from the well (i.e., cuttings) and chemical residues on cuttings, a common solution in the North Sea, is not an option in the Barents area due to lack of a dedicated well and suitable formations for injection to same well. The drilling waste must therefore be brought to shore for treatment. Approximately 1,000 metric tonnes of cuttings with residues is produced per well.

Intermediate storage and transportation to treatment facility of the rock carvings from a well in Norwegian seas typically requires about 6 lifts each of 220 containers. The cuttings transport chain consists of:
1) Ship transport in two stages: i) rig – onshore supply base (by supply vessel); ii) supply base – treatment facility port (by container vessel)
2) Truck transport: port – treatment facility (10 km road transport)

Total fuel use in the ship transport operations was 100 liter diesel per tonne cuttings for a recently drilled well in the Barents Sea. Ship emission factors are assumed equal to the marine diesel vessel at sea described by Cooper and Gustafsson (2004). Truck transport emissions are found in Ecoinvent v1.01 (16 tonne lorry; Frischknecht et al. 2003[1]). Low-sulphur diesel is assumed for all operations.

Characterization factors from the Eco-indicator 99 method (Hierarchist, Goedkoop and Spriensma 2002) are used as they offer results in units of DALY. The following adjustments are made to the characterization factors for human health impacts: i) impacts from radiation, ozone layer depletion and climate change were excluded, ii) fate factors for respiratory effects from direct ship emissions are reduced by a factor of 2 to adjust for the offshore situation, and iii) damage factors for direct emissions are adjusted according to regional population densities. Human health impacts from the transport chain are illustrated in Fig. 4.

Crane-lift health impacts are quantified in the figure using the factor for 97.5% cumulative probability for two reasons; it is an indication of the priority set on offshore safety compared to the other impact chains under consideration, and it illustrates an upper boundary to the significance of crane-lifts to the overall health impacts. The 95%

[1] The lorry fuel use is 0.23 kg/km fully loaded and 0.19 kg/km empty. With a 10 km distance loaded and on return, the fuel use is 0.7 kg diesel per ton of cuttings when carrying 6 tons on each trip.

confidence interval for the respiratory health effect of inorganics spans a factor of 15^2 to 36^2 (Hofstetter 1998). Depending on location, the conclusion is that health burdens from crane-lifts range from significant to not significant in the end-of-life transport chain.

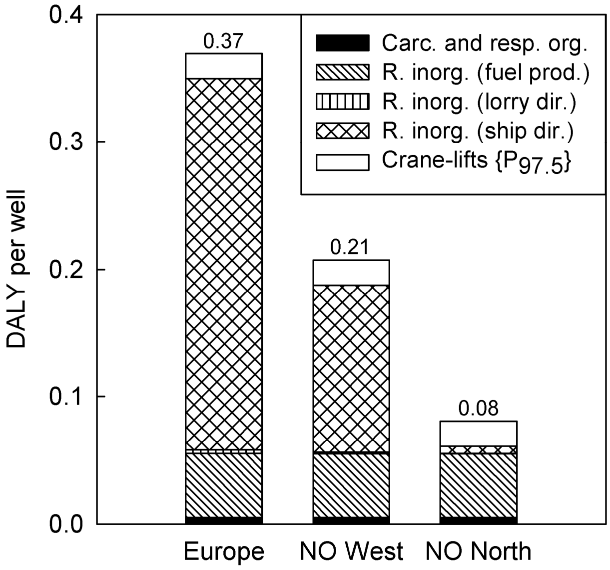

Fig. 4: Health impacts from the end-of-life transport chain. The fate factor is adjusted for offshore respiratory emissions and local population densitities; Europe (European average, 80 cap. per km^2), West Norway (NO West, 36 cap. per km^2), North Norway (NO North, 1.6 cap. per km^2). Carc. = carcinogenics, R: = respiratory, inorg. = inorganics, org. = organics, dir. = direct emissions

2.5.3 Loading technology comparison

Reduction in the number of lifts performed can be achieved by using other means of loading cargo off and aboard ship. A hydraulic system (i.e., pump system) was recently installed on a drilling rig for loading cuttings off rig onto supply vessels, and off vessel at port. Although not included in this evaluation, a second benefit of the hydraulic system is that it is a closed system. It thereby reduces occupational exposure to particulates and chemicals. Table 4 summarizes the production inventory for the pump technology. Fuel use is approximately 31 kg diesel per tonne of cuttings.

The functional unit in this case is the loading off ship of 1metric tonne of cuttings. System boundaries were kept simple and database sources were used (Ecoinvent v1.01, Frischknecht et al. 2003). The Eco-indicator 99 method (Goedkoop and Spriensma 2002) was used to characterize health impacts from emissions, applying the same modifications as described in the previous section.

Rigs are pre-equipped with cranes and no additional system therefore is needed for the crane-lift alternative. The crane consumes 8.33 gallons diesel per hour and operates at an activity rate of 10 lifts per hour (EPA 1999). Each lift loads a container bearing 4.5 tonne of cuttings. This translates to 0.22 lifts per tonne and a fuel consumption of 0.59 kg diesel per tonne of cuttings.

For the crane-lift alternative we find that health impacts are dominated by crane accidents; between $1.2 \cdot 10^{-7}$ and $3.3 \cdot 10^{-6}$ DALY per tonne cuttings, compared to emissions; within $8 \cdot 10^{-10}$ to $5 \cdot 10^{-9}$ DALY per tonne cuttings depending on rig location. In contrast, the hydraulic system represents between $2.6 \cdot 10^{-8}$ and $2.5 \cdot 10^{-7}$ DALY per tonne. Although these results do not include the uncertainty in the emission-related health impacts, they indicate that the hydraulic system does offer a better solution in terms of human health burdens for the onsite personnel and possibly also over the product system life-cycle.

Table 4: Design inventory for the hydraulic system (as communicated by the supplier; personal com., Ørjan Samuelsen, KMC Oiltools)

Component[a]	Weight (tonnes)	Ecoinvent v1.01 process
Engine (diesel generator)	1.0	*Diesel-electric generating set production 10MW/RER/I*, Equivalent to 20.7t. The unit is scaled to 1 tonne
Instruments	0.4	One unit of *Electronics for control units/RER*, equivalent to approx. 2 kg. The remaining weigh is assumed as low alloy steel: *Reinforcing steel, at plant/RER*
Two compressor units	6.0	*Chromium steel 18/8, at plant/RER*
Mechanical components (tanks, screws, skids, container)	34.4	*Reinforcing steel, at plant/RER*

[a] The system has an expected lifetime performance of 6 wells per year over a period of 20 years, giving 120 wells in total. Each well is assumed to require the loading of about 1000 tonnes of cuttings (factor = $8.3 \cdot 10^{-6}$ product systems per tonne)

3 CONCLUSIONS AND OUTLOOK

Accident records were used to develop an empirical characterization factor for offshore crane-lifts. The mean health damage is $4.0 \cdot 10^{-6}$ DALY per crane-lift, with cumulative percentiles $\{P_{2.5}; P_{50}; P_{97.5}\} = \{5.4 \cdot 10^{-7}, 2.8 \cdot 10^{-6}, 1.5 \cdot 10^{-5}\}$. Although uncertainty related to the characterization factor is significant, it is less than what is indicated for other human health impact chains currently included in LCA. The spread in the result is mainly caused by the random nature of accidents (variability), but is also attributed to the estimation procedure (parameter uncertainty).

The contribution to disability adjusted life years (DALY) from recoverable injuries was found insignificant in the case of crane-lifts. In further work of quantifying occupational health impacts in DALY from accident statistics it is advised to see if records of lifelong injuries can be used to simplify the damage assessment process.

Results indicate that crane-lifts are important to the occupational health impacts for employees on offshore petroleum units, and that they are significant to the life-cycle performance of offshore drilling technologies with respect to human health impacts.

The characterization factor for crane-lifts will be used in future case-studies of offshore drilling technologies.

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Appendix D – Paper 3

Johan Pettersen and Edgar G. Hertwich (In review): Metals in life cycle-assessment – current inventory issues and possible solutions. Submitted to *Environmental Science and Technology*

Paper 3 is not included due to copyright.

Electronic supplement

Critical Review: Metals in Life-Cycle Assessment – Current Inventory Issues and Possible Solutions

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BRIEF BACKGROUND

The issue of metal mobility in drilling waste deposits is a returning subject for the marine risk assessment community [1]. Although metal contents in sediments surrounding oil installations are increased compared to natural background levels [2], studies conclude that metal mobility generally is low in marine deposits [2-4].

CALCULATION PARAMETERS

Reported total trace metal content in Barite and Ilmenite vary with a factor 5 to 30 between literature sources for various metals [5-9], largely due to variations in mineral quality [1, 8, 10]; e.g., total trace metal contents in Ilmenite formations [11].

Total metal content is assumed to follow a log-normal distribution with parameters: Total = $f(\mu, \sigma)$. The geometric mean (mode; μ) and standard deviation on log scale (σ), are reported using several sources. Distribution parameters and references for total metal are listed in Table S1.

Two scenarios are investigated for the geoavailable metal: 'high' and 'low'. The high value assumes release of all sequentially extracted fractions but for the residual. The low value assumes that sulfides (in the oxidizable fraction) are retained solids. Source for sequential extraction of Barite and Ilmenite is [6]. Geoavailable metal is calculated as the product of geoavailable fraction and total metal, where total metal is distributed as described in Table S1.

In our assessment, the short-term mobile fraction is assumed represented by the leachable metal according to a pH-separated extraction scheme [7], the cation exchangeable and DTPA extractable fractions [10], and water leachable fraction [12, 13]. All these are equal in concept (and assumed equivalent in our calculations) to the sum of exchangeable and carbonate fractions in the sequential extraction scheme [6, 14]. Highly mobile metal is assumed to follow a log-normal distribution with parameters: Highly mobile fraction = $f(\mu, \sigma)$. Based on the source values, the geometric mean (mode; μ) and standard deviation on log scale (σ) are calculated, as reported in Table S2. Values occurring that exceed total content (highly mobile fraction >1) are set as complete release of total metal (highly mobile fraction =1). Highly mobile metal is calculated as the product of the highly mobile fraction and the total metal.

Most sources report results for pure weight agent mineral; notably [5-9], while the other sources investigate metals in drilling fluids or in drilling waste and soil aggregates. By considering the sources interchangeable we assume that the majority of metal in drilling wastes are attributable to the weight agent. This is not completely true, but figures reported by Nelson et al. indicate that a large portion of the metals do originate from the weight agent [10].

Table S1: Distribution parameters for total metal. Values in natural log scale (ln) of parts per million (ppm; g per kg)

	Ilmenite				Barite			
	μ_T ln(ppm)	σ_T ln(ppm)	No. of datapoints	References	μ_T ln(ppm)	σ_T ln(ppm)	No. of datapoints	References
As	0,607	0,695	1	[6]	2,638	0,930	2	[6, 7]
Ba	4,228	1,840	2	[6, 7]	13,17	0,0979	7	[8, 10]
Cd	-2,796	0,832 [a]	1	[6]	0,106	0,994	3	[6, 7, 9]
Cr	3,743	0,287	5	[5, 6]	2,919	0,477	3	[5, 6, 9]
Co	3,284	2,004	3	[6, 7]	0,222	0,821 [b]	1	[6]
Cu	2,652	0,411	6	[5-7, 9]	4,430	0,273	4	[5-7, 9]
Pb	0,387	1,617	3	[5-7]	5,042	1,223	4	[5-7, 9]
Ni	4,069	0,544	6	[5-7, 9]	0,556	0,747	3	[5, 6, 9]
V	3,238	0,832 [a]	1	[6]	1,538	0,821 [b]	1	[6]
Zn	2,954	1,300	6	[5-7, 9]	4,937	1,211	4	[5-7, 9]

[a] Dispersion set equal to the average (arithmetic) dispersion of metal in Ilmenite for which 3 or more datapoints were found (i.e., based on Cr, Co, Cu, Pb, Ni, Zn)

[b] Dispersion set equal to the average (arithmetic) dispersion of metal in Barite for which 3 or more datapoints were found, excluding Ba (i.e., based on Cd, Cr, Cu, Pb, Ni, Zn)

Table S2: Distribution parameters for the highly mobile fraction of metal

	Ilmenite				Barite			
	μ_M ln(ppm)	σ_M ln(ppm)	No. of datapoints	References	μ_M ln(ppm)	σ_M ln(ppm)	No. of datapoints	References
As	-2,087	1,518	3	[6, 7, 12]	-3,980	0,965	6	[6, 7, 10, 13, 14]
Ba	-3,501	2,338	3	[6, 7, 12]	-9,008	1,787	14	[6, 7, 10, 13, 14] [a]
Cd	-2,561	0,268	2	[6, 7]	-1,757	1,459	12	[6, 7, 10, 14]
Cr	-3,646	0,376	2	[6, 12]	-2,684	1,122	13	[6, 10, 13, 14]
Co	-3,859	1,659	2	[6, 7]	-3,833	3,369	2	[6, 13]
Cu	-2,891	0,344	3	[6, 7, 12]	-2,638	0,774	12	[6, 7, 10, 13]
Pb	-3,231	2,861	3	[6, 7, 12]	-2,350	1,399	14	[6, 7, 10, 13, 14]
Ni	-3,158	0,458	3	[6, 7, 12]	-2,786	0,908	12	[6, 10, 13]
V	-2,782	1,025	2	[6, 12]	-3,390	2,873	2	[6, 13]
Zn	-3,432	2,113	2	[6, 7]	-2,872	2,748	3	[6, 7, 14]

[a] Value for Ref.[6] calculated using total metal as reported by [8]; Ref. [14] based on 'true total'

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