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Life Cycle Assessment of Desalinated Water for Enhanced Oil Recovery

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MASTER THESIS

for

Stud.techn. Mari Hellvik Kvadsheim

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Life cycle assessment of desalinated water for enhanced oil recovery
Livsløpsvurdering av avsaltet vann for meroljeproduksjon

Background and objective

A number of oil fields in the North Sea have produced as much petroleum as was originally expected. However, there is still a lot of oil in place, it is just more tightly bound to the rock and not under as much pressure as the oil that was produced early. New techniques allow for the production of a higher fraction of this oil by making previously insoluble oil more soluble. Such tail production offers substantial economic benefits. However, the costs are higher because additional effort is required, involving pumping solvents, steam or other material down the pipe to mobilize the oil. This additional effort has environmental costs.

The objective of this thesis is to quantify and assess the energy use and emissions associated with such enhanced oil production, using desalinated water as an example, and to compared it to other fuel production methods, using life cycle assessment as a method.

The following issues are to be considered:

1. What is the present state of knowledge in the LCA of oil production, especially off-shore?
2. What are methods for enhanced oil production?
3. What are the additional steps required for *enhanced* oil production applying the selected method?
4. What are the equipment and operational inputs needed for this production?
5. Conduct an overall assessment.
6. Provide a contribution analysis and assess the relevance of different impacts.
7. Discuss uncertainties, assumptions, and evaluate the quality of your results.
8. Discuss your contribution to the body of LCA knowledge of oil production.

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Within 14 days of receiving the written text on the master thesis, the candidate shall submit a research plan for his project to the department.

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The thesis should be formulated as a research report with summary both in English and Norwegian, conclusion, literature references, table of contents etc. During the preparation of the text, the candidate should make an effort to produce a well-structured and easily readable report. In order to ease the evaluation of the thesis, it is important that the cross-references are correct. In the making of the report, strong emphasis should be placed on both a thorough discussion of the results and an orderly presentation.

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☐ Work to be done in lab (Water power lab, Fluids engineering lab, Thermal engineering lab)
☐ Field work

Department of Energy and Process Engineering, 16. January 2013

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Preface

This thesis was written in the spring of 2013 and finalised my Master of Science degree in Energy and Environmental Engineering at the Department of Energy and Process Engineering at the Norwegian University of Science and Technology (NTNU).

A number of people have supported and encouraged me while writing this thesis. I would like to thank my supervisor Edgar Hertwich for his guidance throughout my work. A special thanks to my co-supervisor Anders Arvesen for always taking the time to answer my questions and for constructive feedback along the way.

I would also like to thank my fellow student Martha Marie Øberg for sharing my joys and frustrations during my work and for always keeping me company at the office. I also want to thank Anne Hellvik Kvadsheim and Bjørn Inge Kvadsheim for giving me motivation and words of encouragement when needed. Last but not least, to Martin Holst; thank you for everything.

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Abstract

Currently, fossil fuels supply 85% of the world's energy demand. Nevertheless, we consume more than we are able to produce from new discoveries of fossil resources. As energy demand is predicted to grow rapidly over the next few decades, the need for new methods to sustain oil production emerges. By using new technology, known as enhanced oil recovery, it is possible to recover oil previously considered too tightly bound to the reservoir rock to be recovered in a profitable way. One such method is low-salinity waterflooding, where desalinated water is injected into the reservoir in order to increase the crude oil recovery. If implemented, this method could result in significant economic benefit, but little is known on the environmental impacts associated with it.

In this thesis, a life cycle assessment of desalinated water for enhanced oil recovery was conducted. Reverse osmosis was chosen as desalination technology and a generic model located in the North Sea was developed based on existing literature. The results show that the operation phase is the largest contributor to environmental impacts due to the generation of power by natural gas-driven turbines on the platform. The chemical treatment process is also a significant contributor to environmental impacts, due to energy inputs and wastes from chemical manufacturing.

The emissions of greenhouse gases from the system were calculated to be 151 kg of CO_2 equivalents for each standard cubic meter of recovered crude oil. This is three times higher than greenhouse gas emissions from oil production without enhanced oil recovery methods, but substantially lower than emissions from oil sands production. It is recommended to implement enhanced oil recovery methods such as low-salinity waterflooding, rather than producing oil from unconventional fossil reserves such as oil sands.

A sensitivity analysis was also conducted, presenting alternative scenarios for power supply, by means of electrification of the platform. The results show that electrification of a platform could offer substantial environmental benefits in terms of reduced emissions of greenhouse gases, depending on the composition of the electricity mix. However, several issues will need to be addressed before this should be implemented on a large scale, in order to ensure that it will indeed reduce global greenhouse gas emissions.

The results from this thesis create a basis and a starting point for future research. The environmental impacts associated with *desalination of water* are deemed reliable; however, great uncertainty is linked to the required amount of water per standard cubic meter of recovered crude oil. In order to calculate the environmental impacts from one specific oil field or enhanced oil recovery project, it is necessary to quantify material and energy inputs, emissions and wastes, as well as the exact water-to-oil ratio by mapping and identifying key parameters and properties of the petroleum reservoir in question.

Sammendrag

I dag forsyner fossile brensler 85% av verdens energibehov. Imidlertid bruker vi mer enn vi klarer å produsere med nye funn av fossile ressurser. Ettersom verdens energibehov er forventet å vokse raskt de neste tiårene, vil det oppstå et behov for nye metoder som kan opprettholde oljeproduksjonen. Ved å benytte ny teknologi, kjent som meroljeproduksjon, er det mulig å utvinne råolje som tidligere ble ansett å være for tett bundet til reservoarbergartene til å være økonomisk lønnsom å utvinne. En slik metode er å injisere avsaltet vann ned i et petroleumsreservoar. En implementasjon av denne teknikken vil kunne føre til betydelig økonomisk vinning, men vi vet ennå lite om hvilke miljømessige konsekvenser en slik metode kan ha.

I denne avhandlingen er det gjort en livssyklusanalyse av avsaltet vann for meroljeproduksjon. Omvendt osmose ble valgt som avsaltingsteknologi og en generell modell lokalisert i Nordsjøen ble utviklet basert på eksisterende litteratur. Resultatene viser at driftsfasen er den største bidragsyteren til miljøbelastninger som følge av kraftproduksjon ved naturgass-drevne turbiner på plattformen. Kjemisk rensing er også en stor bidragsyter til miljøbelastninger, grunnet energibruk og avfall fra produksjonen av kjemikalier.

Utslippene av klimagasser ble beregnet til 151 kg CO_2-ekvivalenter per standard kubikkmeter råolje utvunnet med injeksjon av avsaltet vann. Dette er tre ganger høyere enn tilsvarende utslipp fra oljeproduksjon uten meroljeproduksjonsmetoder, men vesentlig lavere enn utslipp fra oljesandproduksjon. Det anbefales å implementere meroljeproduksjonsmetoder, som injeksjon av avsaltet vann, fremfor å produsere olje fra ukonvensjonelle fossile reserver som oljesand.

En sensitivitetsanalyse som presenterer alternative scenarioer for strømforsyning ble også gjennomført. Disse scenarioene involverer at plattformen blir forsynt med strøm produsert på land, såkalt elektrifisering. Resultatene viser at elektrifisering av en plattform kan gi betydelige miljømessige fordeler i form av reduserte utslipp av klimagasser, avengig av sammensetningen av elektrisitetsmiksen. Imidlertid vil flere spørsmål måtte tas opp før dette bør gjennomføres i stor skala for å sikre at dette faktisk vil redusere de totale klimagassutslippene globalt.

Resultatene fra denne avhandlingen skaper et grunnlag og utgangspunkt for fremtidig forskning. Miljøbelastningene knyttet til *avsalting av vann* er ansett som pålitelige, men det er samtidig stor usikkerhet knyttet til den nødvendige mengden vann per standard kubikkmeter utvunnet råolje. For å bergne miljøbelastningene knyttet til et bestemt oljefelt eller meroljeproduksjonsprosjekt er det nødvendig å kvantifisere råmaterialer, energitilførsel, utslipp og avfall, i tillegg til det eksakte vann-til-olje forholdet som trengs ved å kartlegge og identifisere viktige parametre og egenskaper for det aktuelle petroleumsreservoaret.

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Abbreviations

API	American Petroleum Institute
CA	Cellulose-Acetate
CCS	Carbon Capture and Storage
CIP	Clean-In-Place
COFCAW	Combination Of Forward Combustion And Waterflooding
DALY	Disability-Adjusted Life Years
ENTSO-E	European Network of Transmission System Operators for Electricity
EOR	Enhanced Oil Recovery
EROI	Energy Return On Investment
ETS	Emissions Trading Scheme
GWP	Global Warming Potential
HS	High-Salinity
HSW	High-Salinity Waterflooding
HVAC	High Voltage Alternating Current
HVDC	High Voltage Direct Current
ISO	International Organization of Standardization
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LS	Low-Salinity
LSE	Low-Salinity Effect
LSW	Low-Salinity Waterflooding
MED	Multiple Effect Distillation

MPD	Ministry of Petroleum and Energy
MS	Moderate-Salinity
MSF	Multistage Flash Distillation
OG21	Olje og Gass I det 21. århundret
OOIP	Original Oil in Place
PV	Pore Volume
PVC	Polyvinyl Chloride
RO	Reverse Osmosis
SCO	Synthetic Crude Oil
TDS	Total Dissolved Solids
TSO	Transmission System Operator
WTR	Well-to-Refinery Entrance Gate
WTW	Well-to-Wheel

1 Introduction

Today, 85% of the world's energy demand is supplied by fossil fuels such as oil, gas and coal. 32 billion barrels of oil equivalents are produced every year, and still the energy demand is predicted to grow rapidly over the next few decades (Sheng, 2011). We currently consume more fossil fuels than we are able to produce from new fossil resource discoveries. Many scientists believe that we eventually will reach what is known as "peak oil", the point in time where maximum oil production is reached, followed by a steady decline in production. However, there are still on-going debates on if or when this might happen and some even believe that it has happened already (Hall et al., 2008).

Global petroleum resources are vast; they are estimated to 9.6 trillion barrels of conventional crude oil and 3 trillion barrels of unconventional crude oil, but it is technically impossible to recover all of these resources. The recovery factor, or recovery efficiency, of a field is the fraction of crude oil that can be recovered with today's technology. The estimated global average recovery efficiency ranges from 20% to 40%, and there are large regional differences. In the North Sea, the average recovery efficiency is 46%, making it the highest recovery efficiency in the world (Sandrea and Sandrea, 2007).

By using new technology it is possible to recover crude oil previously considered too tightly bound to the reservoir rock to be extracted in a profitable way. Methods which use external materials and/or energy sources to recover crude oil are classified as enhanced oil recovery (EOR) methods. This includes methods such as injection of steam, chemical-infused water or different types of gas (e.g. CO_2, nitrogen gas or flue gas) into the reservoir (Satter et al., 2008).

A promising EOR method is the injection of desalinated water, so-called low-salinity waterflooding (LSW). It is a fairly simple method as nothing is added to the injection water. The method has proven to increase the crude oil recovery efficiency in sandstones. One of the most comprehensive studies reported an average increase in oil recovery by 14% (Lager et al., 2007), but increases up to 25% have also been documented (Morrow and Buckley, 2011). If implemented, this method could result in significant economic benefit. However, little is known on the environmental impacts associated with it.

1.1 Objective and scope

The objective of this thesis is to quantify and assess the life cycle environmental impacts of desalinated water for EOR. This will be done by means of life cycle assessment (LCA) of a generic system for desalination of seawater located on a platform. The geographical scope is limited to the North Sea since many EOR projects are implemented here due to the target set by Olje og gass i det 21. århundret (OG21) and the Norwegian Ministry of Petroleum and

Energy to increase the oil recovery rate on the Norwegian continental shelf in order to sustain societal value creation (OG21, 2010). The focus will be on the required *additional* inputs and steps for EOR, meaning that the existing equipment, inputs and the platform itself will be excluded from the LCA. Furthermore, the oil production which is not the result of EOR on the platform will be excluded as well. The system will include the manufacturing of the different system components, raw materials, energy inputs, transportation, operation and disposal for the LSW technology only. This means that the desalination facility will be seen as isolated from the platform in this analysis.

The generic system for desalination of seawater will be developed based on existing literature. It is chosen to use reverse osmosis (RO) as the desalination technology since it is planned to be used by Statoil on a pilot project on LSW at the Heidrun field in the North Sea (Hegre, 2008). No previous LCAs on LSW by RO were found, and the study will therefore use existing literature in a new context. The energy use and emissions will be quantified and assessed, and compared to the results from other studies and from other oil production methods.

The LCA is conducted by using the software Arda 16.1 which is directly linked to the Ecoinvent database. Arda quantifies the potential environmental impacts over 18 impact categories, based on the ReCiPe hierarchist impact assessment method.

1.2 Structure of thesis

Chapter 1 gives an introduction to the thesis and outlines the objective and scope, while chapter 2 presents the background information and technical aspects regarding oil recovery methods and desalination technologies. This chapter also contains a literature review where previous LCA studies of oil production and of desalination technologies are presented.

The third chapter describes the methodology of LCA and explains its theoretical framework and basic mathematics. Important tools and methods for this thesis are also presented. The life cycle inventory analysis (LCI) is given in chapter 4. The system and its delimitations are presented and all its components are described in detail. The inventory data and the different assumptions are also included here.

Chapter 5 is where the results from the LCA are presented and shown graphically. Chapter 6 presents the sensitivity analysis where the results from the LCA are tested with respect to robustness. It is also chosen to create alternative scenarios for power supply.

Chapter 7 contains the discussion part and this is where the results are interpreted and compared to other studies and oil production methods. This chapter also discusses the data quality and uncertainty for the results. The conclusions of this thesis and recommendation for future work are presented in chapter 8.

2 Background

This chapter presents background information and technical aspects of the different types of oil recovery. Different desalination technologies are also described and a literature review of previous LCA studies of oil production and of RO are given at the end of the chapter.

2.1 Primary oil recovery

Primary oil recovery methods exploit the naturally occurring pressure inside a petroleum reservoir, resulting in a flow of crude oil towards the producing well. This is the result of several physical mechanisms that combined create what is called the *reservoir drive*. One of the mechanisms of the reservoir drive is the natural water drive, which displaces the oil upward in the well. Other mechanisms are the gas-cap drive, which expands the natural gas at the top of the reservoir and displaces the oil downwards towards the producing well, dissolved gas drive, which is the dissolution and expansion of gas in the crude oil, and gravity drainage, which are the movements of oil due to gravitational forces (Tzimas et al., 2005). Figure 1 illustrates how a petroleum reservoir may look like.

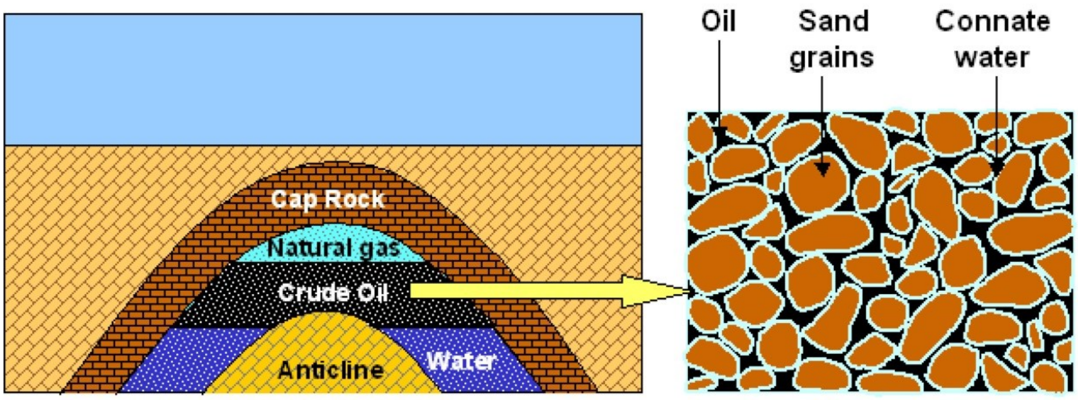

Figure 1: An example of an oil and gas reservoir (Tzimas et al., 2005).

The reservoir drive will eventually diminish as oil and gas are extracted from the reservoir, and pumps will then be used to maintain the production rate. A completion of the primary oil recovery stage is done when it is no longer possible to maintain profitable production rates. The primary oil recovery efficiency typically ranges between 5-15% of original oil in place (OOIP), depending on reservoir characteristics and the properties of the reservoir fluids (Satter et al., 2008).

There are a number of factors that determine how much crude oil is recovered from a reservoir. One of them is the porosity of the reservoir rocks, which is defined by Satter et al. (2008) as a measure of the pore volume compared to the bulk volume of the rock. The rock porosity can be expressed by the following:

$$\phi = \frac{\text{volume of pore space in rock}}{\text{bulk volume of rock}} \tag{2.1}$$

Economically profitable oil reservoirs usually have porosities in the range of 5% to 35% or higher. Gas reservoirs, on the other hand, may have lower porosity values and still be capable of commercial production (Satter et al., 2008).

Permeability is also an important factor in terms of the recovery efficiency. Satter et al. (2008) defines permeability as “a measure of the capability of a porous medium to transmit fluid through a network of microscopic channels under a certain driving force” (Satter et al., 2008, p. 25).

Permeability is usually determined through laboratory tests on core samples by using Darcy’s law. Darcy’s law describes the flow of water through a bed of sand particles and is given by the following equation (Satter et al., 2008):

$$q = KA\frac{h_1 - h_2}{L} \tag{2.2}$$

Where q is the volumetric flow rate, K is the hydraulic conductivity, A is the cross-sectional area of flow, h_1 and h_2 are hydraulic heads at points 1 and 2 and L is the length of porous medium.

One way of increasing the recovered amount of crude oil is by lowering the mobility ratio. This can be done by increasing the viscosity of the displacing fluid or by lowering the viscosity of the oil. The mobility ratio of a fluid is defined as following by (Satter et al., 2008):

$$\mathrm{M} = \frac{\text{mobility of displacing fluid}}{\text{mobility of displaced fluid}} \tag{2.3}$$

The capillary number is also a very important factor for oil recovery efficiencies and it represents the ratio of viscous to interfacial forces (Satter et al., 2008). It is defined as (Schlumberger, 2013a):

$$N_c = Ca = \frac{\mu U}{\sigma} \tag{2.4}$$

Where N_c is the capillary number, μ is the fluid viscosity, U is the fluid velocity and σ is the interfacial or surface tension.

2.2 Secondary oil recovery

Secondary oil recovery methods are applied when primary oil recovery methods are no longer profitable. The goal of these types of methods is to increase the natural reservoir drive by injecting fluids into the reservoir. This artificial drive will result in an increase in the reservoir pressure, thus it is possible to recover more crude oil (Tzimas et al., 2005). The principle of secondary oil recovery is given in Figure 2.

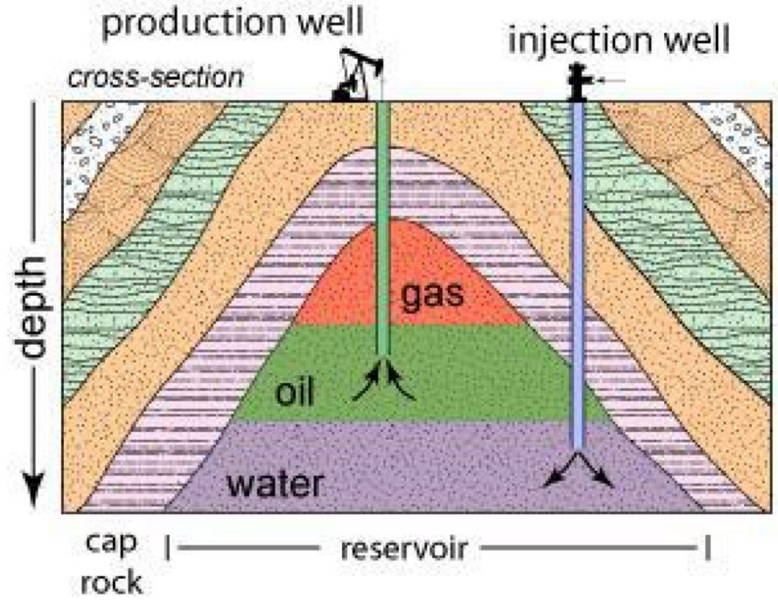

Figure 2: The principle of secondary oil recovery (Myers, 2010).

There are different secondary oil recovery methods, but waterflooding is by far the most common. Waterflooding is a method in which water is injected into a reservoir with the purpose of displacing residual oil. Using water as an injection agent has many advantages; it is easy to inject, it is cheap and available, it is efficient in terms of displacing light to medium gravity oils and it has relatively low investment costs as well as operating costs (Satter et al., 2008).

It is preferable that water injection is initiated early in unsaturated oil reservoirs as this will maintain reservoir pressure above the bubble point so that no gas develops. The reason for this is that the oil recovery rate is higher while the gas is dissolved in the solution. If the reservoir pressure is lower than the bubble point pressure, as it will be in a depleted reservoir, it is likely that there will be a gas cap in the reservoir. When the gas cap is refilled with water the gas will dissolve again. This result in delays in the oil production, thus the net present value of the project will be reduced (Satter et al., 2008).

When performing waterflooding on a reservoir, water will eventually break through at the producing well, resulting in an increase in the water-oil-ratio. This will eventually make the oil production unprofitable. Peak oil production and waterflood response depend on several parameters such as injection rate, fluid properties, well spacing and heterogeneities in the reservoir. Oil recovery by waterflooding will in general yield costs ranging from several cents to several dollars per barrel produced. Compared to other EOR methods this is relatively low (Satter et al., 2008).

In the reservoir, oil and water saturations will change with time and space during the performance of the waterflooding. The change depends on fluid and rock properties, such as fractional flow characteristics, which again depends on the permeability of the oil and water (Satter et al., 2008).

When designing the waterflood operation, there are several elements that need to be included according to Satter et al. (2008), such as a description of the reservoir and rock characterization studies. It is also necessary to map heterogeneities since they control the direction of the fluid flow in the reservoir, as well as zones with permeability that is highly contrasting. In order to achieve optimal oil recovery, the number of wells, their schedule, location and pattern must be appropriately designed. The water injection rate, production rate and reservoir pressure must also be optimized. The key to managing the reservoir in a successful manner is waterflood surveillance, which includes a detailed monitoring and evaluation plan.

Other secondary oil recovery methods include injection of gases, e.g. re-injection of natural gas. One option is to inject natural gas into the gas cap in the reservoir, which will increase the reservoir pressure and displace oil towards the producing wells. The other option is immiscible displacement, where natural gas is injected into the oil bank without being mixed with it. Immiscible gas displacement is less efficient than waterflooding and there are also issues concerning whether or not it is more profitable to sell the gas (Tzimas et al., 2005).

Both waterflooding and the re-injection of natural gas have been successfully demonstrated in reservoirs in the North Sea. Waterflooding typically have a recovery efficiency of about 30%, while the recovery efficiency after primary and secondary oil recovery is between 30% and 50% (Tzimas et al., 2005).

2.3 Enhanced oil recovery

2.3.1 Concept and potential

EOR, also called tertiary oil recovery, includes methods which are applied at the end of life of a petroleum reservoir in order to further augment the recovery of crude oil beyond secondary oil recovery. Satter et al. (2008) presents the following definition; "Enhanced oil recovery processes include all methods that use external sources of energy and/or materials to recover oil that cannot be produced economically by conventional means" (Satter et al., 2008, p. 549).

The additional amount of oil recovered by EOR depends on the characteristics of the reservoir and the properties of the petroleum fluids, but it is suggested that an additional 5-15% of OOIP may be recovered (Tzimas et al., 2005).

2.3.2 Enhanced oil recovery methods

We separate between three different EOR methods (Satter et al., 2008):

- Thermal methods
- Chemical methods
- Miscible methods

Thermal methods

Thermal methods are used mainly for heavy viscous oils and tar sands. Heavy viscous oils are typically in the API (American Petroleum Institute) gravity range of 10°-20° (Satter et al., 2008). The API gravity gives the relative gravity or density of liquid petroleum products in degrees. It can be expressed through the following formula (Schlumberger, 2013b):

$$API = \frac{141.5}{SG} - 131.5 \tag{2.5}$$

In heavy viscous oils, injection of water will have little effect as the water-oil mobility ratio is too high. This leaves injection of heat as a solution in order to recover oil from these types of reservoirs. Thermal EOR production represents about 60% of all EOR production and includes methods such as steam stimulation, steam flooding and in-situ combustion (Satter et al., 2008).

Steam soak

Steam soak is the most successful thermal EOR process, also called cyclic steam injection or "huff-and-puff steam flood". Steam soaking is done by injecting steam with a high rate into a single well for a short period of time, typically for a few weeks. Afterwards, the steam gets to soak in for a few days, before the well is pumped. The cycle is repeated until the oil recovery is unprofitable to carry out (Satter et al., 2008).

This type of process is called a well stimulation process. The steam heats up the oil around the wellbore, causing mobilised oil to flow through the wellbore and allowing it to be recovered. This is most successful when performed on reservoirs with good permeability and viscous oils. However, only a fraction of the reservoir is affected, resulting in a low oil recovery rate (Satter et al., 2008).

Steam flooding

Steam flooding is a process in which steam is continuously injected into a well. This will result in a reduction of the crude oil's viscosity as it is heated, and the oil is then mobilized towards the producing wells. The injected steam will condense as it gives off heat to the crude oil, the fluids in the reservoir and to the base rock. This will result in a steam zone that drives the crude oil towards the well (Satter et al., 2008).

This method is generally used on viscous oils in sandstones with high permeability, or in unconsolidated sands, as lighter crude oils will respond to an ordinary waterflood operation.

There are several requirements for steam flooding; the oil saturations must be high and the steam flooded reservoirs should be shallow due to excessive heat loss in the wellbore. The downside of this method is the high cost per incremental barrel of oil, as about one-third of the oil that is additionally recovered is consumed in order to generate the steam. Steam flooding is usually not performed on carbonate reservoirs as they may contain bottom water and gas caps (Satter et al., 2008).

In-situ combustion

In-situ combustion is a method in which a fire is started in the reservoir. The fire is sustained by injection of air. There are different types of in-situ combustion, the most common being forward combustion. Here, the ignition of the fire occurs at the bottom of the well, while air is injected to lead the fire away from the well. In the fire the heavier parts of the crude oil are burnt at high temperatures around 600°C, while the lighter parts are driven to the wellbore by steam, hot water and combustion gas (Satter et al., 2008).

It is also possible to use a combination of forward combustion and waterflooding (COFCAW) or to use a method called reverse combustion. Reverse combustion involves starting a fire in a well that will later become a producing well, while injecting air into adjacent wells. This has, however never been successfully achieved in any test fields (Satter et al., 2008).

For in-situ combustion the amount of coke deposited is critical; if the amount is too low, the combustion process will not be maintained, but if the amount is too high, the amount of air required in the combustion process will be high. It is also important that the oil saturation and the porosity in the reservoir are high in order to reduce heat loss to the rock (Satter et al., 2008).

There are also some problems related to this method; it is complex, hard to control and expensive. There are also potential environmental problems related to the flues gases which are produced during the fire. Problems related to the operation may also occur such as corrosion or pipe failure. The sweep efficiency will also be poor in thick formations (Satter et al., 2008).

Chemical methods

Chemical methods are really just modifications of waterflooding and will therefore require the same conditions as ordinary water injection. Processes that are characterized as chemical EOR methods include polymer flooding, polymer/surfactant flooding and caustic flooding (Satter et al., 2008).

Polymer flooding

Polymer flooding involves injecting water, to which water-soluble polymers have been added, into the reservoir. Both synthetic polymers and biopolymers can be used and the concentration is generally in the range of 250-2000 mg/L. This process enhances volumetric sweep efficiency by increasing the viscosity and decreasing the mobility of water. This allows for a much larger volume of the reservoir to get in contact with the polymer-augmented water.

The residual oil saturation will however not be reduced when adding a polymer to the water. The flooding should be started before the water-oil-ratio is too high. This method has an economic advantage over conventional waterflooding, even if the total recovery is the same for the two methods; the oil displacement is more efficient in the early phases, resulting in higher oil production early on (Satter et al., 2008).

For oils with high viscosity, a higher concentration of polymers in the water is required in order to achieve control over the mobility. Polymer flooding should be avoided when dealing with reservoirs with extensive fractures (Satter et al., 2008).

Polymer/surfactant flooding

Polymer/surfactant flooding, or so-called micro-emulsion flooding or micellar/polymer flooding, is described by Satter et al. (2008) as a process in which slug is injected into the reservoir. The slug is made up of water, a surfactant, an electrolyte and a co-solvent. It may also contain a hydrocarbon. After the slug, polymer-augmented water is injected. This method recovers oil by lowering the interfacial tension between the water and the oil, resulting in enhanced mobility, emulsification and solubilisation of the oil.

Polymer/surfactant flooding is preferably used on light oils, where a relatively homogenous formation is desired, as well as an area sweep of more than 50%. The reservoir should not contain a high amount of clays, anhydride or gypsum and formation water chlorides should be less than 20 000 ppm, while the concentration of divalent ions should be lower than 500 ppm (Satter et al., 2008).

One drawback of this method is that systems that are available only have optimal performance for a very restricted set of conditions. Also, the system is complex and expensive , with a risk of degradation of chemicals at high temperatures, chromatographic separation of chemicals and/or interactions between surfactant and polymer (Satter et al., 2008).

Caustic flooding

Caustic flooding is also known as alkaline flooding. Chemicals, such as sodium hydroxide, sodium silicate or sodium carbonate are injected into the reservoir, leading to a reaction between the chemicals and the organic petroleum acids forming surfactants in situ. For this process it is desirable with sandstone reservoirs and oils in the API gravity range 13°-35°. Another requirement is that the oils have moderate oil gravity and that the content of organic acids is high enough so that the reaction between the chemicals and acids can occur. This method should not be performed on carbonate formations as they may potentially consume a large amount of the chemicals. This is also true for clays, minerals and silica. A known problem with caustic flooding is scale formation in the producing wells (Satter et al., 2008).

Miscible methods

Miscible methods are described by Satter et al. (2008) as methods that involve injecting gas or a solvent that is miscible with oil into the reservoir. This includes injection of hydrocarbon gas, carbon dioxide, nitrogen, flue gas and partial miscible/immiscible gas floods. The result is a reduction of the interfacial tension between the oil and the solvent and of the efficient microscopic displacement efficiency.

Hydrocarbon miscible flooding

Hydrocarbon miscible flooding involves creating a miscible flood in the reservoir by injecting light hydrocarbons. There are three different methods in which this is done, one of them being a method where slug, consisting of liquefied petroleum gas, is injected into the reservoir. This is followed by the injection of lean gas and sometimes water in a water-gas-alternating mode (Satter et al., 2008).

Enriched (condensing) gas drive is a second method, where natural gas enriched with ethane through hexane is injected, followed by the injection of lean gas and sometimes water. This creates a miscible zone that moves the oil towards the well (Satter et al., 2008).

High pressure (vaporizing) gas drive is a method in which lean gas is injected into the reservoir at high pressures. When the crude oil is displaced, the ethane through hexane components vaporize from the crude oil. This means that we get multiple contact miscibility (Satter et al., 2008).

For hydrocarbon miscible flooding the requirements for the minimum depth is determined by the pressure necessary to maintain the miscibility. This depends on the oil composition, and ranges from about 1200 psi for the liquid petroleum gas process to 3000-5000 psi for the high pressure gas drive. It is also desirable with a formation that has a steep dipping. The drawbacks for this method is that it requires a great amount of expensive products and that viscous fingering may occur, resulting in poor horizontal and vertical sweep efficiency. Also, one risks that the solvents are trapped in the reservoir and thus not possible to recover (Satter et al., 2008).

Carbon dioxide flooding

Carbon dioxide flooding involves injecting carbon dioxide into the reservoir, typically in large quantities; 15 vol% or more. CO_2 generates miscibility by extracting the light to intermediate components of the crude oil under high pressure, similar to the mechanisms of high pressure gas drive in hydrocarbon miscible flooding. However, CO_2 flooding can be performed on a larger amount of reservoirs at lower miscibility pressure as CO_2 flooding extract a wider range of crude oil components compared to hydrocarbon miscible flooding (Satter et al., 2008).

The generation of miscibility between the oil and CO_2 is dependent on the reservoir pressure. Normally, CO_2 is soluble in crude oils at reservoir pressures and it reduces the viscosity of the oil, resulting in multiple contacts between the crude oil and CO_2. This allows them to flow together due to low interfacial tension. To improve the mobility ratio between the displacing phase and the oil, water is injected in a water-alternating-gas mode (Satter et al., 2008).

Drawbacks related to the CO_2 flooding process is poor mobility control as a result of the low viscosity of CO_2, corrosion in the producing wells and the high amount of CO_2 required per incremental barrel produced. CO_2 also need to be re-pressurised before in can be recycled and it is necessary to separate it from the oil before it is sold (Satter et al., 2008).

Nitrogen and flue gas flooding

Nitrogen and flue gas flooding is described by Satter et al. (2008) as a method in which nitrogen or flue gas are injected into the reservoir. The lighter components of the crude oil are then evaporated resulting in oil displacement. Depending on the composition of the oil and the reservoir pressure, the displacement may be miscible or immiscible. Large volumes of nitrogen and flue gas may be injected due to their low costs, but nitrogen has low viscosity and poor solubility in oil and CO_2 injection is therefore preferred as an EOR method.

This method requires steeply dipping, deep reservoirs with light oils and high pressure. Disadvantages are corrosion for the injection of flue gas and viscous fingering. It is also necessary to separate the non-hydrocarbons from the produced gas (Satter et al., 2008).

2.4 Low-salinity waterflooding

LSW is an EOR method where desalinated seawater is injected into a petroleum reservoir. The interest in this method has its origin in 1997, when Tang and Morrow reported that laboratory tests showed that LSW increased the crude oil recovery rate in sandstones. This is called the low-salinity effect (LSE). Tang and Morrow (1999) did repeated waterfloods on a Berea sandstone core and were able to identify necessary conditions for the LSE to occur. They found that the reservoir needs to contain a significant clay fraction, connate water needs to be present and that exposure to crude oil is necessary in order to create mixed-wet conditions.

Figure 3 shows the recovery curves for LSW at initial water saturation. This is called secondary-mode LSW. The amount of oil recovered is given as a percentage of the OOIP. The results are from laboratory tests on Berea-sandstone cores where HS is high-salinity synthetic reservoir brine, MS is moderate salinity dilution with one tenth of the strength of HS and LS is low-salinity dilution with one hundreth of the strength of HS. The ionic composition is the same for the connate and the injected water (Morrow and Buckley, 2011).

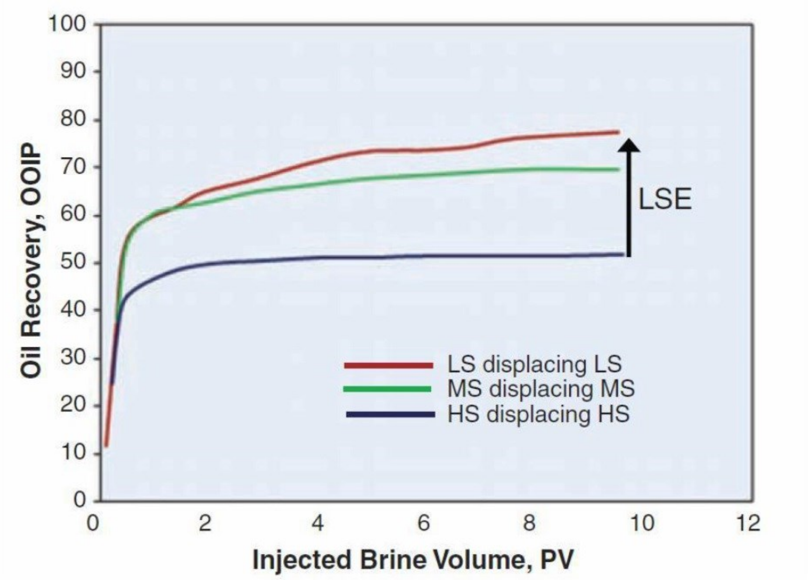

Figure 3: Waterflood recovery efficiency versus pore volume (PV), showing LSE for LSW at initial water saturation (Morrow and Buckley, 2011).

There is a particular interest in applying LSW after HS waterflooding (HSW), so-called tertiary-mode LSW, which is LSW at residual oil saturation. Figure 4 shows one of the more promising results for reservoir rock, where the recovery with LSW increased by 25% compared to HSW (Morrow and Buckley, 2011). Lager et al. (2007) found that the average relative increase in recovery rate was 14% for LSW compared to HSW for 18 reservoirs.

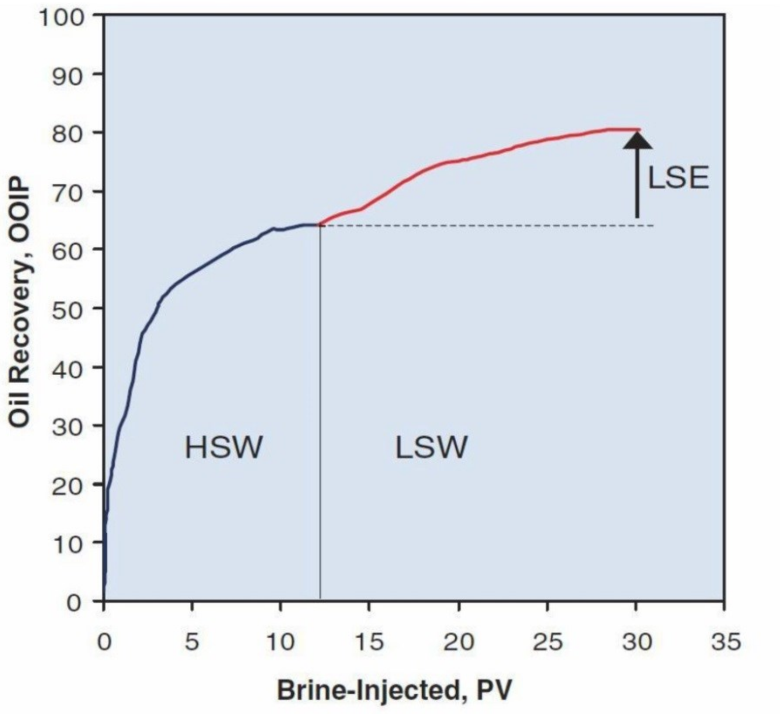

Figure 4: Waterflood recovery efficiency versus PV, showing LSE for LSW at residual oil saturation (Morrow and Buckley, 2011).

Webb et al. (2004) were the first to prove that LSW result in increased oil recovery rates in a field test. They did a so-called log-inject-log test, which is based on running several passes of pulsed neutron capture logs after injecting different brines. These results corresponded to the results from previous laboratory tests and showed that LSW resulted in 25-50% reduction in residual oil saturation.

So far, LSW has mainly been studied for sandstone reservoirs, both in the laboratory and in the field. However, some scientists have recently started to investigate whether or not LSE can be found for carbonate rocks as well. In a study done by Zahid et al. (2012) a substantial increase in the oil recovery was demonstrated for diluted versions of seawater at 90°C in carbonate rocks.

Although laboratory and field tests demonstrate the effect of LSW, one has not been able to find a consistent explanation for the phenomena. The use of various materials may be one explanation, as well as the complexity of the minerals, crude oils and aqueous compositions that can be found in a reservoir. It is also possible that more than one mechanism cause LSE (Morrow and Buckley, 2011).

Challenges for the future involve understanding the factors that determine the crude oil recovery rates for different combinations of crude oil, brine and rock. It is also necessary to identify necessary conditions for LSE to occur and to understand why there in some cases are little or no LSE (Morrow and Buckley, 2011).

In terms of desalination of the seawater, there are several technologies available. The most common commercial desalination technologies are reverse osmosis (RO), multistage flash distillation (MSF) and multiple effect distillation (MED) (Fritzmann et al., 2007). In this thesis RO is used for desalination in accordance with a case study done by Statoil (Hegre, 2008).

2.4.1 Reverse osmosis (RO)

RO is a membrane based desalination technology. Semi-permeable membranes are used to separate water from dissolved matter. A pressure difference is applied across the membrane, forcing the water, also called permeate, through the membrane (Fritzmann et al., 2007).

Reverse osmosis fundamentals

Osmosis takes place when two aqueous solutions with different concentrations are separated by a semi-permeable membrane. The water will then flow from the solution with the lowest concentration to the solution with the highest concentration. This will occur until the concentrations in both solutions are equal. Osmosis will occur as long as the pressure difference Δp is smaller than the osmotic pressure difference $\Delta\Pi$, which depends on the difference in concentrations in the two solutions (Fritzmann et al., 2007).

Reverse osmosis takes place when Δp is larger than ΔΠ. Then, the water will flow from the solution with the highest concentration to the solution with the lowest concentration (Fritzmann et al., 2007). Figure 5 illustrates the principle of reverse osmosis.

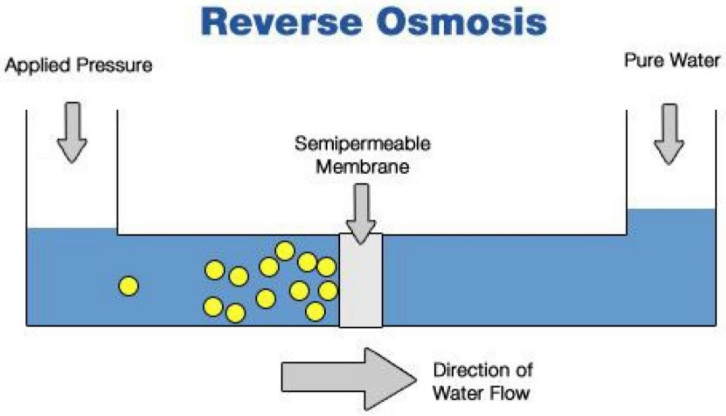

Figure 5: The principle of reverse osmosis (ROSHE, 2013).

The osmotic pressure is defined as following (Fritzmann et al., 2007):

$$\pi = -\frac{R \cdot T}{V_b} \ln(x_W) \tag{2.6}$$

Where π is the osmotic pressure, R is the ideal gas constant, T is temperature, V_b is the molar volume of water and x_W is the mole fraction of water.

System and components

Key properties for RO membranes are high flux and high rejection. As the flux is inversely proportional to the thickness of the membrane, very thin membranes are required. They consist of two layers; one active non-porous layer and one porous supporting layer. The supporting layer prevents ripping, while the active layer prevents the transport of mass through the membrane. This type of membrane is called an asymmetric membrane (Fritzmann et al., 2007).

In the 1970s, cellulose-acetate (CA) membranes were the first commercially available RO membranes on the market. They are still in use today, but composite membranes are more and more taking over. Composite membranes consist of one active layer made of polyamide and one porous supporting layer which can be made of different materials. Composite membranes are preferred over CA membranes as they are more stable chemically and physically, they resist bacterial degradation and do not hydrolyse. Composite membranes will however have a larger tendency for fouling compared to CA membranes as they are less hydrophilic (Fritzmann et al., 2007).

Today, most membranes are installed in a spiral wound module, illustrated in Figure 6. Fritzmann et al. (2007) describe how a spiral wound module is made up of multiple sheet membranes. These are glued together with a permeate spacer in between. This is done to form membrane pockets, which are attached to the perforated central tube. Feed spacers are placed between each membrane pocket to create alternating feed and permeate channels. These are then rolled around the tube and the feed is forced through the membrane. The produced permeate is collected in the perforated central tube. The spiral wound module is cheap and easy to manufacture and has a high packing density. It also has a good balance as regards permeability. Drawbacks are that it is difficult to clean and that it is susceptible to fouling (Fritzmann et al., 2007).

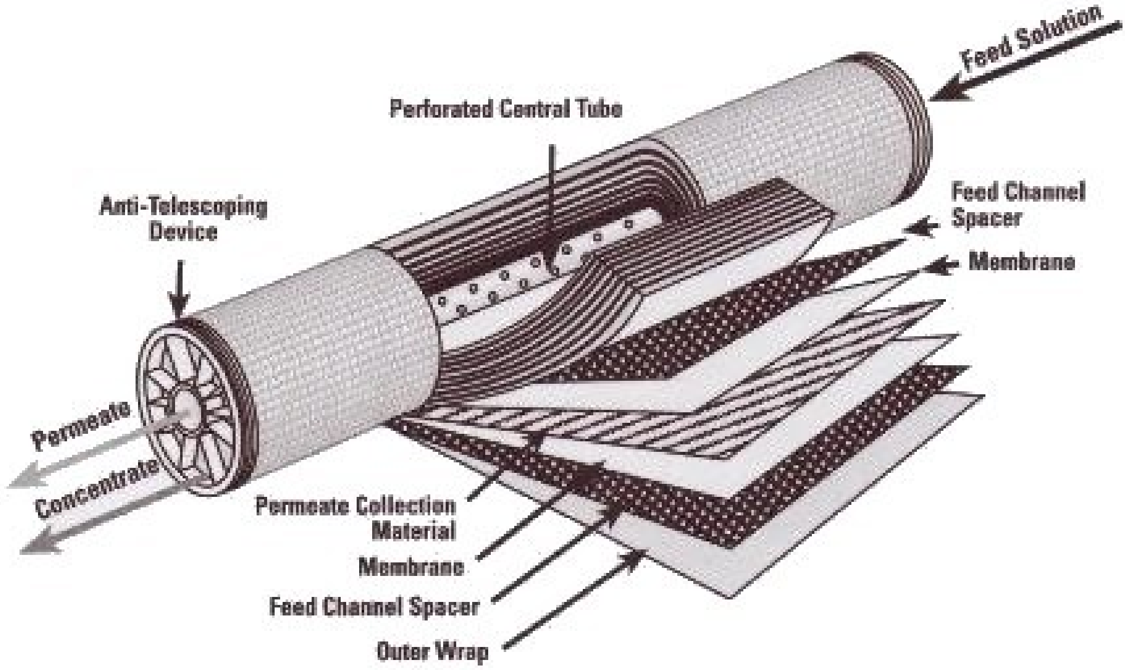

Figure 6: A spiral wound module (NEOROPurifiers.com, 2013)

A membrane will typically have a shorter lifetime than the RO unit itself. This has to do with mechanisms and factors which can lead to reduced performance, such as membrane deterioration due to usage of oxidants in the pre-treatment phase. Membranes are also sensitive to very high or very low pH values, thus pH adjustment is important in order to ensure stable operation (Fritzmann et al., 2007).

Blocking of the membrane can also occur as a result of build-up of dissolved or biologic matter on the membrane surface. Fritzmann et al. (2007) separate between two blocking mechanisms; scaling and fouling. Scaling is the accumulation of inorganic material on the membrane surface. Substances that are important contributors to scaling are $CaCO_3$, $CaSO_4$, $BaSO_4$ and silica. Scaling can be prevented or reduced by pre-treatment using anti-scalent chemicals and pH adjustment. Fouling can be transport of particulate matter to the membrane surface or it can be biological growth on the surface. Particulate fowling can be reduced by mechanical pre-treatment with sand filters, while biological fouling can be reduced by chlorination as a pre-treatment of the feed water (Fritzmann et al., 2007).

2.4.2 Multistage flash distillation (MSF)

Distillation is a desalination method based on phase separation, where freshwater is produced by condensing vaporised saline water (UNEP, 1997). A simplified illustration of a multistage flash unit is shown in Figure 7. Seawater flows through the heat exchangers and into the brine heater where it is heated by steam. At the brine heater outlet the sea water is overheated relative to the pressure and temperature in the first evaporating chamber, resulting in an instant release of heat and vapour, a so-called "flash". This vapour is then condensed and forms freshwater at the top section of the chamber. This is repeated throughout the evaporation chambers and at the last and coldest chamber the brine reject is removed and the distillate is extracted (Sidem, 2013b).

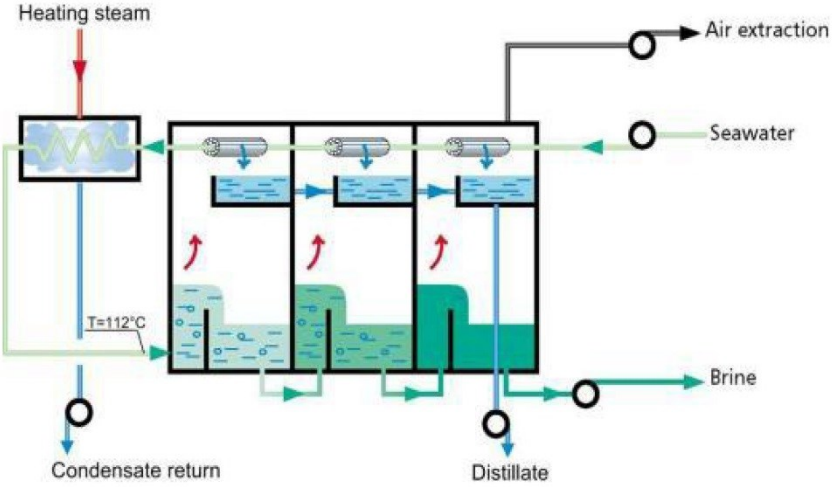

Figure 7: Multistage flash "once-through" desalination process (Sidem, 2013b).

Figure 7 shows an MSF unit where the brine only passes through once. There are also MSF units where some of the brine is recycled and mixed with the incoming feed water, so-called brine recycle plants. It is recommended that these should be used for larger plants where the seawater temperature varies through the year (Sidem, 2013b).

2.4.3 Multiple effect distillation (MED)

A simplified illustration of a multiple effect distillation unit is shown in Figure 8. The unit consist of coherent cells, also called effects, with decreasing temperature and pressure. In each cell there are a bundle of tubes containing heating steam. Seawater is sprayed on each of these bundles of tubes, cooling them and condensing the steam inside of them, resulting in the production of freshwater. The seawater at the bottom of each effect is heated as a result of the condensing and some of it will therefore evaporate. The warmed seawater is used in the next effect as a heating medium, although the temperature will be lower than in the previous cell. This is repeated until the last cell, where the produced steam is condensed in a heat exchanger cooled by seawater. Some of the remaining seawater is used as a spray on the different cells, while the rest is rejected to the sea.

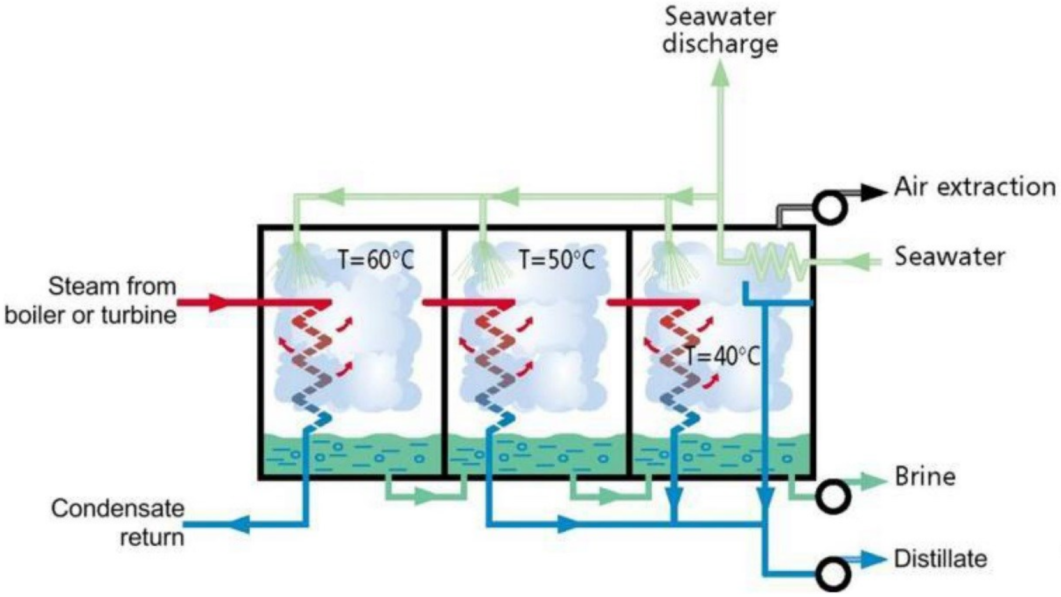

Figure 8: Multiple effect distillation process (Sidem, 2013a)

2.5 Literature review

Literature search shows that that there are few LCA studies on oil production offshore. The reason for this is probably that life cycle inventory databases, such as Ecoinvent, contain sufficient information on oil production. A life cycle inventory database contains compilations of inventories for a variety of processes, and can thus be seen as a collection of interlinked LCA studies (Strømman, 2010).

The Ecoinvent database contains information on the environmental impacts associated with recovery of crude oil. The inventories may be global, region-specific or country-specific and include data on processes such as oil field exploration, production of crude oil, transportation over long distances, refining of oil, distribution and use of petroleum products. Associated pollutants and wastes are also included, as well as energy and material requirements, infrastructure, facilities and transport services (Dones et al., 2007).

The inventories are separated into different process stages, making it possible to investigate the associated environmental impacts at different levels of oil production. E.g. one process include the production of crude oil offshore, another include the processes up until refining of the oil, while a third process contain information on all processes up until the use of a petroleum product (Dones et al., 2007).

For oil production offshore, there are three processes in Ecoinvent, one for Norway, one for Great Britain and one for the Netherlands (Ecoinvent, 2013). The Norwegian process is assumed to be valid for production in mature fields in the North Sea and includes exploration and production of petroleum on the Norwegian Continental Shelf. It is based on emissions and discharges from 49 producing fields and 178 exploration and injection wells (amee, 2013).

A life cycle impact assessment (LCIA) with the ReCiPe method based on the inventory of oil production offshore for Norway, results in a global warming potential (GWP) of 54.8 kg of CO_2 equivalents per standard cubic meter (Sm^3) of extracted crude oil. If long-distance transport is included the GWP is 60.9 kg of CO_2 equivalents.

In terms of EOR, few LCA studies are done and mostly on CO_2 capture and storage (CCS). Hertwich et al. (2008) conducted an LCA of CCS at Tjeldbergodden for EOR at the Draugen field in the Norwegian Sea. The results gave a GWP of 128 kg of CO_2 equivalents for EOR with diesel combustion for power generation. The majority of the greenhouse gas emissions, about 88%, were associated with the power generation offshore. Diesel combustion was also the main contributor to the emissions of nitrous oxides and sulphur dioxide.

Several LCA studies are done on reverse osmosis for desalination of water. Muñoz and Fernández-Alba (2008) conducted an LCA of desalination of seawater and brackish water by RO. Their results gave desalination of seawater by RO a GWP value of 1.9 kg of CO_2 equivalents per m^3 of desalinated water. It was concluded that the electricity consumption for the RO process is a key environmental aspect as it was responsible for more than 95% of the contribution in all impact categories but one, where the contribution was 91%.

A comparative LCA of different desalination technologies by Raluy et al. (2006) resulted in a GWP of 1.78 kg of CO_2 equivalents per m^3 of desalinated water from RO. Also here the energy consumption was associated with more than 95% of the environmental load. Zhou et al. (2011) investigated to what extent environmental impacts of desalination by RO varied when different LCIA methods were used. Their results gave a GWP of 1.58 kg of CO_2 equivalents per m^3 of desalinated water for all LCIA methods.

R. G. Raluy Rivera (2009) did a doctoral thesis on water production systems, where LCAs of different desalination technologies were conducted. For desalination of seawater by RO a GWP of 2.121 kg of CO_2 per m^3 of desalinated water was calculated. The electricity consumption was identified as being the largest contributor to environmental impacts. This thesis was based on inventories from a previous study done by R. G. Raluy Rivera (2003) and several other LCA studies have also used these inventories (Muñoz and Fernández-Alba, 2008, Raluy et al., 2004, Raluy et al., 2005a, Raluy et al., 2005b, Raluy et al., 2005c).

The aforementioned studies all have in common the operation phase as main contributor to environmental impacts. The LCA studies on RO use electricity mixes with different compositions of energy sources, while CCS for EOR use diesel consumption as energy input into the operation phase. Nevertheless, they all identify energy consumption as the key issue in terms of environmental impacts. Thus, substantial environmental benefit could be achieved by reducing the energy requirement or by using renewable energy sources for power generation for the described technologies.

3 Methodology

This chapter presents the methodology of LCA. The theoretical framework and its different phases are described and the basic mathematics is derived. The tools and methods used in this thesis are also presented.

3.1 Theoretical framework

The principles and framework of LCA is defined by the International Organization of Standardization in the international standard ISO 14040. Here, LCA is defined as a method that "addresses the environmental aspects and potential environmental impacts (e.g. use of resources and the environmental consequences of releases) throughout a product's life cycle from raw material acquisition through production, use, end-of-life treatment, recycling and final disposal (i.e. cradle-to-grave)" (ISO, 2006, p. V). LCA is an holistic approach, which means that one looks at the whole system as well as the interdependence of its parts (Guinée, 2002).

The LCA framework is made up of four phases, which are all described in ISO14040 (2006). Figure 9 illustrates the four different phases and their interactions.

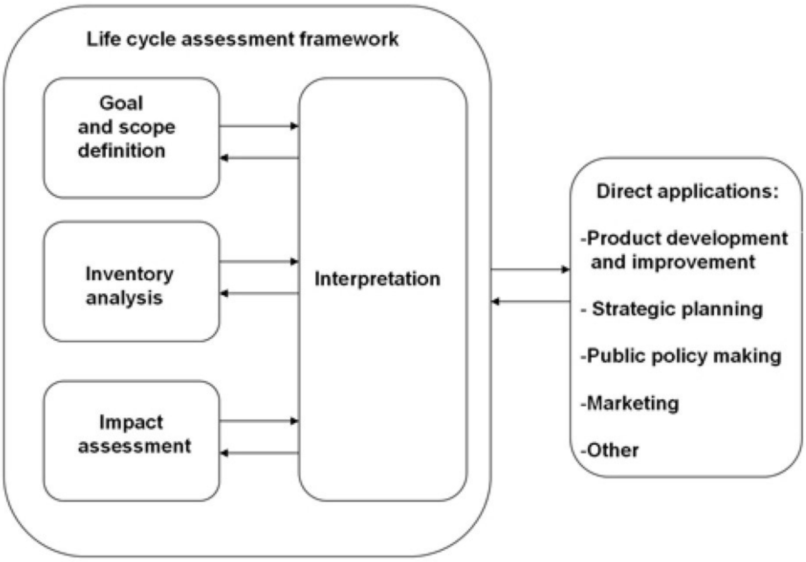

Figure 9: The LCA framework (ISO, 2006).

The goal and scope definition phase is where the aim of the study and the system boundaries are established. The level of sophistication and the width and depth of the study is chosen based on this (ISO, 2006). An important part of this phase is to define the functional unit. The functional unit is a quantitative measure of the function that is to be delivered and it should reflect the function of the product (Strømman, 2011).

The life cycle inventory analysis (LCI) involves collecting data on the relevant input and output flows of the system. These flows are typically energy and material requirements, wastes, emissions and other environmental aspects. This is a time-consuming and resource-intensive process, and constraints should therefore be made when defining the scope of the study. To ease the data collection it is possible to use existing inventory databases, such as Ecoinvent (ISO, 2006).

The LCI phase also comprises calculation procedures where the input and output flows of the system are quantified. This includes validation of data and connecting it to unit processes and to the reference flow of the functional unit. The result of the LCI phase is usually an inventory table which contains information on material and energy use, as well as emissions, aggregated over the life cycle of the product (ISO, 2006).

In the life cycle impact assessment (LCIA) phase, the goal is to use the results obtained from the inventory analysis to evaluate the significance of potential environmental impacts. This is done by associating the data from the inventory table with different environmental impact categories and category indicators. Environmental impact categories are categories that represent specific environmental issues of concern, such as climate change and acidification, and category indicators are quantifiable representations of these (ISO, 2006).

The next phase of the LCIA is the classification phase, where the results from the LCI are aggregated and converted to equivalent amounts of a reference compound, before they are divided into the different environmental impact categories. The reference compound can be different from category to category, e.g. CO_2 for climate change and SO_2 for acidification. After the classification, the category indicator results are calculated in what is called the characterisation phase (ISO, 2006).

The LCIA phase also includes some optional steps. Normalisation is where the category indicator results are applied in order to find a relation between the functional unit and the total extent of the problem, e.g. comparing environmental impacts from different EOR technologies. Weighting involves comparing the relative importance of the different environmental impact categories. This step introduces subjectivity into the LCIA phase, and many scientists therefore choose not to include it. It is difficult to agree upon which environmental impact category is more severe. Other issues, such as choice of approach and interpretation of environmental impact categories may also result in subjectivity. Transparency is thus critical to make sure that all the assumptions are clear (ISO, 2006).

The interpretation phase of an LCA is where the results from the other phases are discussed and analysed by decomposing them among the contributing processes and/or substances. It usually includes a discussion of the reliability of the results, and sensitivity and uncertainty analyses may be performed. This phase is typically rounded off with a final conclusion which in some cases includes recommendations for the future (ISO, 2006).

3.2 Basic mathematics

Table 3-1 presents the nomenclature used in the basic mathematics of LCA, based on Strømman (2010) . Usually, a mathematical modelling tool, such as MatLab is used when performing an LCA. It is also possible to use LCA software, e.g. Arda.

Table 3-1: Nomenclature for the different vectors and matrices used in the basic mathematics of LCA (Strømman, 2010).

Sets	**pro**		**Processes**
	str		**Stressors**
	imp		**Impact categories**
Matrices and variables	A	$pro \times pro$	Matrix of inter process requirements
	y	$pro \times 1$	Vector of external demand of processes
	x	$pro \times 1$	Vector of outputs for a given external demand
	L	$pro \times pro$	The Leontief inverse, matrix of outputs per unit of external demand
	S	$str \times pro$	Matrix of stressor intensities per unit output
	e	$str \times 1$	Vector of stressors generated for a given external demand
	E	$str \times pro$	Matrix of stressors generated from each process for a given external demand
	C	$imp \times str$	Characterization matrix
	d	$imp \times 1$	Vector of impacts generated for a given external demand
	D_{pro}	$imp \times pro$	Matrix of impacts generated from each process for a given external demand
	D_{str}	$imp \times str$	Matrix of impacts generated from each stressor for a given external demand

LCA is based on the so-called open Leontief model, where the interdependence between the processes is assumed to be linear. Information on requirements of inputs to production is gathered for each process, thus forming a basis for establishing the *A* matrix. The *A* matrix contains the recipe of what and how much we need of inputs to the production. The columns in the *A* matrix represent the input that is required to produce one unit of output for the respective process, for a given external demand *y*. The *x* vector represents the required output for a given external demand (Strømman, 2010):

$$Ax + y = x \tag{3.1}$$

Rearranging yields,

$$x = Ax + y \Leftrightarrow (I - A)x = y \Leftrightarrow x = (I - A)^{-1} y \tag{3.2}$$

The Leontief matrix, *L*, represents outputs per unit of external demand, and is defined as follows (Strømman, 2010):

$$L = (I - A)^{-1} \Rightarrow x = Ly \Rightarrow L = (I - A)^{-1} \Rightarrow x = Ly \tag{3.3}$$

Equations (3.1) - (3.3) represent the central elements of the open Leontief model. The next step in an LCA is the contribution analysis and this is where one calculates emissions and environmental loads for a given external demand.

Contribution analysis

In LCA, it is normal to use the term *stressor* rather than *emission*, as environmental loads can have other forms than what is normally associated with the term emissions, such as mineral depletion and water use. The stressor intensity matrix, S, contains stressor intensities per unit output. The e vector represents the stressors that are generated for a given external demand and can be calculated from (Strømman, 2010),

$$e = Sx = SLy = S(I - A)^{-1} y \tag{3.4}$$

In order to find the stressors generated from each process for a given external demand, the E matrix has to be established (Strømman, 2010):

$$E = S\hat{x} \tag{3.5}$$

The characterization matrix, C, is used to convert the emissions from the different substances into equivalents. This means that substances with the same type of environmental impact are grouped together and converted into the same compound, e.g. all substances that contribute to climate change are converted to CO_2 equivalents. This allows us to calculate the vector of total environmental impacts, d, for a given external demand (Strømman, 2010):

$$d = Ce = CSx = CSLy = CS(I - A)^{-1} y \tag{3.6}$$

From d, we can find the D_{pro} matrix, which gives the environmental impacts generated from each process for a given external demand (Strømman, 2010):

$$D_{pro} = CE = CS\hat{x} \tag{3.7}$$

The D_{str} matrix represents the environmental impacts generated from each stressor for a given external demand (Strømman, 2010):

$$D_{str} = C\hat{e} \tag{3.8}$$

Structural path analysis

Structural path analysis involves the decomposition of a complex network system into individual paths. This can be used to locate key paths which have a significant contribution to environmental impacts (Suh and Heijungs, 2007). This means that it is possible to identify which background processes contribute to environmental impacts due to the demand from a foreground process. For insight into the mathematics behind structural path analysis, see Suh and Heijungs (2007).

In this thesis, structural path analysis is used to identify the background processes which contribute to environmental impacts in order to understand how and why the different foreground processes contribute to environmental impacts.

3.3 Software and tools

As mentioned previously, subjectivity may be an issue associated with LCA. One reason is that there are different impact categories, category indicators and characterization models to choose from. In this thesis, the LCIA phase is done by using the ReCiPe method (Goedkoop et al., 2013). For this method one can choose between a midpoint-oriented and an endpoint-oriented approach. The midpoint approach is proposed as a baseline method for characterisation by Guinée (2002) and it describes the *potential* for damage. E.g. for the impact category climate change, the characterization factor is GWP and it represents the potential damage that can be done by contributing to climate change. The endpoint approach measures the effects and damages from a given impact. One example of an endpoint indicator is disability-adjusted life years (DALY), which represents damage to human health (Goedkoop et al., 2013).

There are 18 midpoint impact categories and three different perspectives in the ReCiPe method. The three perspectives are a result of different cultural perspectives, subjective choices and uncertainties, and are called individualist (I), hierarchist (H) and egalitarian (E) (Goedkoop et al., 2013). The individualist perspective includes substances that have undisputable impacts on the environment. This perspective has the shortest time horizon with 100 years or less. The hierarchist perspective includes substances for which there is consensus regarding their impacts on the environment. It has a long time horizon. The egalitarian perspective is the most conservative and includes substances for which there is just an indication regarding their impact on the environment. This perspective has an extremely long time horizon (Strømman, 2011). The hierarchist perspective is the default ReCiPe method and will therefore be used in this thesis. The 18 impact categories, their units and characterisation factors are displayed in Table 3-2.

Table 3-2: Overview of the midpoint categories, units and characterization factors (Goedkoop et al., 2013).

Impact category Name	Unit	Characterisation factor Name
Climate change	kg CO_2 eq	Global warming potential (GWP)
Ozone depletion	kg CFC-11 eq	Ozone depletion potential (ODP)
Terrestrial acidification	kg SO_2 eq	Terrestrial acidification potential (TAP)
Freshwater eutrophication	kg P eq	Freshwater eutrophication potential (FEP)
Marine eutrophication	kg N eq	Marine eutrophication potential (MEP)
Human toxicity	kg 1,4-DCB eq	Human toxicity potential (HTP)
Photochemical oxidant formation	kg NMVOC eq	Photochemical oxidant formation potential (POFP)
Particulate matter formation	kg PM eq	Particulate matter formation potential (PMFP)
Terrestrial ecotoxicity	kg 1,4-DCB eq	Terrestrial ecotoxicity potential (TETP)
Freshwater ecotoxicity	kg 1,4-DCB eq	Freshwater ecotoxicity potential (FETP)
Marine ecotoxicity	kg 1,4-DCB eq	Marine ecotoxicity potential (METP)
Ionizing radiation	kg U^{235} eq	Ionizing radiation potential (IRP)
Agricultural land occupation	m^2yr	Agricultural land occupation potential (ALOP)
Urban land occupation	m^2yr	Urban land occupation potential (ULOP)
Natural land transformation	m^2	Natural land transformation potential (NLTP)
Water depletion	m^3	Water depletion potential (WPD)
Metal depletion	kg Fe eq	Mineral resource depletion potential (MDP)
Fossil depletion	kg oil eq	Fossil resource depletion potential (FDP)

The Ecoinvent v2.2 database is used for the LCI as it is the most complete LCA database for European purposes, even though it is somewhat fragmented in construction (Strømman, 2011). It includes more than 4000 datasets on a variety of areas and it is the world's leading database on LCI data (Ecoinvent, 2013).

The LCIA results are calculated by using the software Arda 16.1, which is developed by the Industrial Ecology program at the Norwegian University of Science and Technology. Arda is run through Matlab and is directly linked to the Ecoinvent database. It uses the ReCiPe framework to perform a contribution analysis.

4 Life Cycle Inventory Analysis

This chapter describes the system used in this thesis and its delimitations. An illustration of the system is given, as well as a flow sheet. It also presents the assumptions and the LCI data found for each component of the system.

4.1 System description

Figure 10 illustrates the unit block scheme for desalination of seawater by using RO. The system is generic and based on descriptions from Rivera (2009) and Hegre (2008). Seawater is first pumped into the system and treated with different chemicals in order to eliminate any biological activity, as well as other substances that could potentially reduce the lifetime of the RO membranes. The seawater also goes through mechanical pre-treatment, where filters remove turbidity suspended particles from the water. The high pressure pumps make sure that the pressure is sufficiently high so that the RO process takes place, as described in chapter 2.4.1.

The RO membrane package system desalinates the seawater and is attached to a clean-in-place (CIP) system that is used for in-situ chemical cleaning of the RO membranes. The desalinated water is chemically treated, and includes disinfection and mineralizing. The desalinated water is then pumped into the petroleum reservoir.

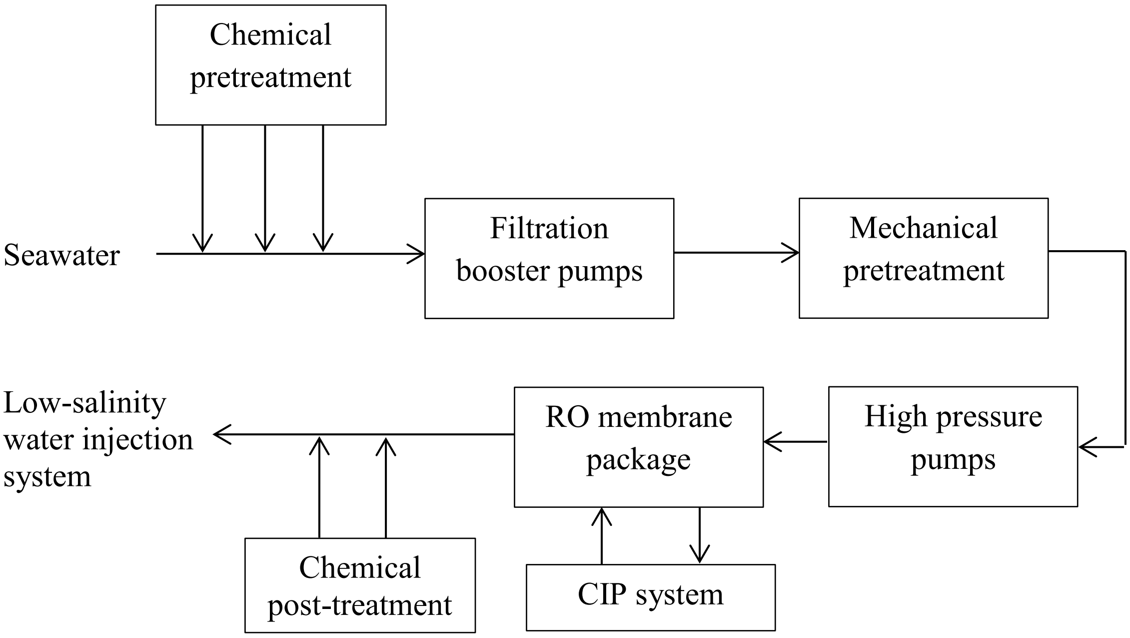

Figure 10: Unit block scheme for desalination of seawater by RO (Hegre, 2008) (Rivera, 2009).

4.2 Life cycle inventories

The inventories are given for each foreground process. The complete inventory lists are given in Table A-1 and Table A-2, Appendix A.

4.2.1 System flow chart and delimitations

Figure 11 illustrates the flow chart for the LSW system. It includes the operation phase, extracting and processing of raw materials, as well as manufacturing of the RO module and other components, such as pumps and a CIP system. It also includes production of chemicals for chemical treatment, and raw materials for the mechanical pre-treatment system. For the disposal phase, it is assumed that all materials are landfilled, as is done in previous LCA studies on RO (Muñoz and Fernández-Alba, 2008) (Rivera, 2009). Transportation is included in the inventories for the different components and phases. The dotted line represents the foreground system boundary.

The functional unit is 1 Sm^3 of crude oil recovered by LSW, and the LCA only takes into account the equipment and additional steps required for *enhanced* oil recovery. Thus already existing parts, such as well pipes, pipelines and equipment on the platform are *not* included in this study. It is assumed that the LSW system is installed on a platform in the North Sea and European conditions are therefore used, when possible, in all the inventories.

The RO plant is assumed to have a lifetime of 25 years, with the exception of the membranes which have a lifetime of 5 years (Rivera, 2009). This means that they will be replaced four times over the system's lifetime.

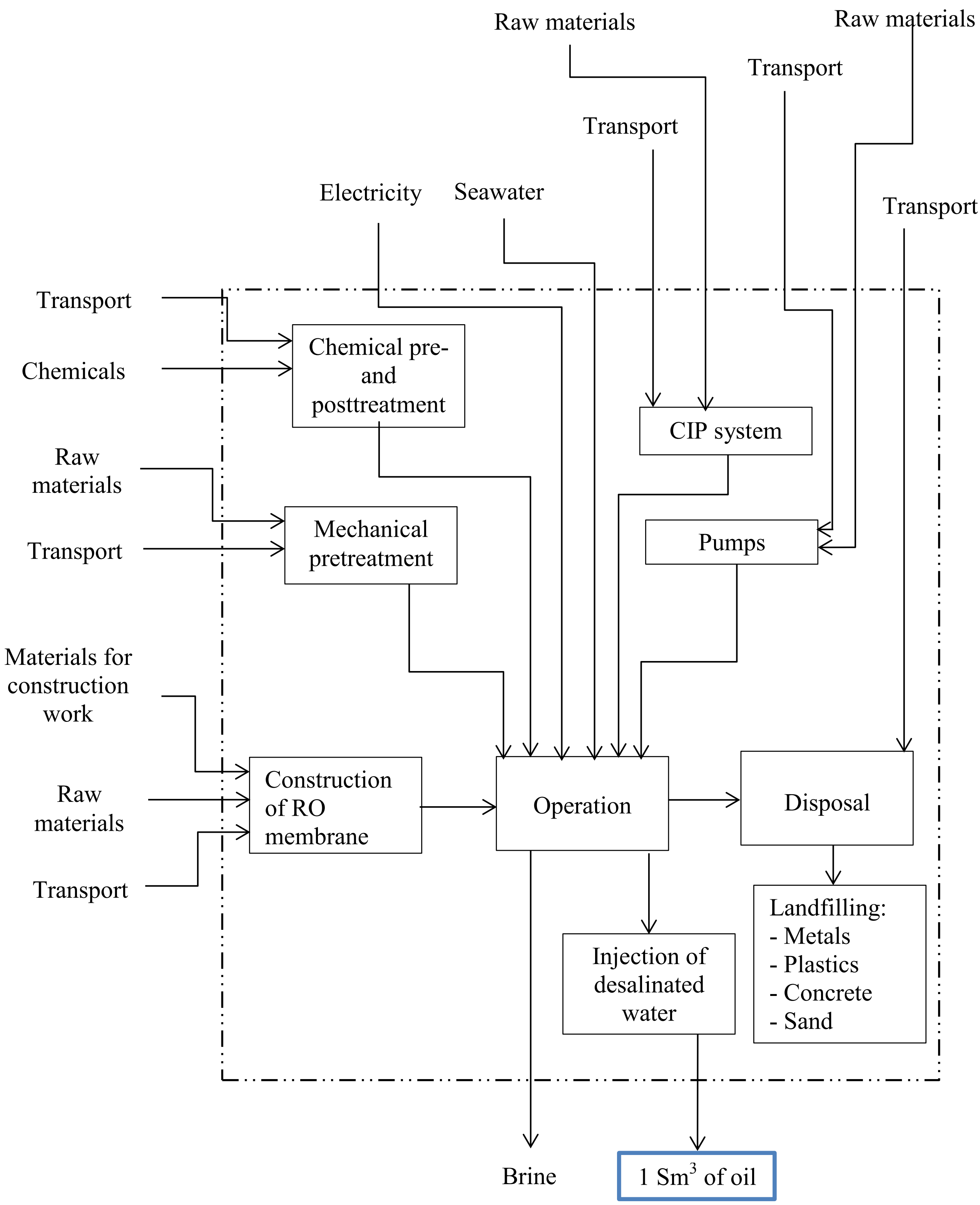

Figure 11: System flow chart for the low-salinity waterflooding system.

4.2.2 Injection of desalinated water

In this thesis, it is chosen to use a RO plant similar to what is described by Rivera (2009). The plant has a capacity of injecting 46 000 m^3 of desalinated water into the reservoir every day. The recovery rate is set to 45% (Hegre, 2008), resulting in a seawater inflow of 102 222 m^3 per day. The reject water, also called brine, has a flow rate of 56 222 m^3 per day. It is assumed that the seawater has 35 000 ppm of total dissolved solids (TDS), while the permeate has a TDS concentration of 400 ppm, resulting in a salt rejection rate of 99.57% for the reverse osmosis plant. The brine has a TDS concentration of about 63 600 ppm (Rivera, 2009).

The environmental impacts resulting from the discharge of brine into the sea are not taken into account. This is due to a lack of complete data regarding brine composition for desalination of seawater. TDS is therefore the only indicator used to characterize the produced brine (Muñoz and Fernández-Alba, 2008).

In terms of desalinated water required for EOR, there are little data on the amount of water required per amount of recovered crude oil. Thomas (2008) state that the water requirement can vary from 5 m^3 water per Sm^3 of crude oil for steam injection, up to over 300 m^3 of water for micellar processes. Thakur and Satter (1998) give a water-to-oil ratio of 5.6-70.8, depending on multiple variables. In this thesis, it is chosen to use a ratio of 50 m^3 of desalinated water per Sm^3 of crude oil recovered, as this is a conservative approach. The data for injection of desalinated water are given in Table 4-1.

Table 4-1: Data for injection of desalinated water.

Input [per Sm^3 of recovered crude oil]	**Value**	**Unit**
Desalinated water	50	m^3
Output		
Recovered crude oil	1	Sm^3

4.2.3 Operation

The operation phase includes inputs of components, electricity, seawater and disposal. The disposal phase is for simplicity modelled as input into the operation phase in Arda, although in reality the components from the operation phase are inputs into the disposal phase. The inputs into the operation phase are modelled on a "per m^3 of desalinated water"-basis.

In order to desalinate 1 m^3 of seawater, 4 kWh are required (Rivera, 2009). The power generation on a platform is usually done by dual diesel/gas turbines that use diesel or natural gas directly from the petroleum reservoir. The amount of energy generated from natural gas is significantly larger than the amount of energy generated from diesel (Shell, 2012). It is therefore assumed that all of the electricity used for desalination is generated by burning natural gas in a turbine.

As mentioned in chapter 4.2.2, the recovery rate of the reverse osmosis plant is 45%, resulting in a daily inflow of 102 222 m^3 of seawater, thus 2.22 m^3 of seawater is required per m^3 of desalinated water.

For the different components and for the disposal phase, the total amount of raw materials and transport is given in the background to foreground matrix in Arda. The data is then connected to the foreground matrix by multiplying it with a number that convert the numbers from total amount to amount per m^3 of desalinated water. For this case, this number is

$$\frac{1}{\text{amount of desalinated water during the lifetime of the system}} = \frac{1}{46000\, m^3/d \cdot 365\, d/y \cdot 25y} = 2.38 \cdot 10^{-9} / m^3 \qquad (4.1)$$

Data for the operation phase is given in Table 4-2.

Table 4-2: Data for operation of the desalination plant.

Input [per m^3 of desalinated water]	**Value**	**Unit**
Pumps	$2{,}38 \cdot 10^{-9}$	p
CIP system	$2{,}38 \cdot 10^{-9}$	p
Chemical treatment	$2{,}38 \cdot 10^{-9}$	kg
Mechanical pre-treatment	$2{,}38 \cdot 10^{-9}$	p
Construction of RO membrane package	$2{,}38 \cdot 10^{-9}$	p
Disposal	$2{,}38 \cdot 10^{-9}$	kg
Output		
Desalinated water	1	m^3

4.2.4 Pumps

The pumping system consist of seawater intake pumps, filtration booster pumps, high pressure pumps and pumps that inject the desalinated water into the reservoir (Hegre, 2008). The pumps are made of stainless steel and the material requirement is 230 tonnes (Rivera, 2009).

According to the European scenario in the Ecoinvent database, metals are transported 200 km by train and 100 km by lorry. The corresponding processes in Ecoinvent include production, operation, maintenance and disposal of trains and lorries, as well as construction, maintenance and disposal of railway tracks and roads (Frischknecht et al., 2007). It is assumed that the platform is located 100 km from shore, and the transport offshore is assumed to be by barges.

The data for the pumps are given in Table 4-3.

Table 4-3: Data for the pumps.

Input	Value [total over lifetime]	Process in Ecoinvent
Stainless steel	230 000	chromium steel18/8, at plant/RER/kg
Transport, by rail	46 000	transport, freight, rail/RER/tkm
Transport, by road	23 000	transport, lorry >16t, fleet average/RER/tkm
Transport, by sea	23 000	Transport, barge/RER/tkm

4.2.5 CIP system

The CIP system is designed to clean an RO unit without the need for disassembly. This ensures maximum performance of the RO unit and increases its lifetime. The CIP system consist of a centrifugal pump, a linear polyethylene chemical tank, recirculation valve, tank drain valve, polyvinyl chloride (PVC) piping, steel frame, filters and different equipment for measuring temperature, flow and pressure. The chosen model is one that is used for heavy duty, industrial and large system applications (Siemens, 2013).

The weight of a CIP system is given as 3 447 kg (Siemens, 2013). There is not any information on the weight of the different components, so crude assumptions have had to be made. It is assumed that steel constitutes 75% of the system's weight and that it includes frame, pumps, valves, filters and electrical equipment. It is further assumed that the chemical tank of linear polyethylene represent 20% of the system's weight, while PVC piping makes up the last 5%.

As for the pumps, the European scenario in the Ecoinvent database states that metals are transported 200 km by train and 100 km by lorry. The same transport distances and means are also true for plastics (Frischknecht et al., 2007). The transport offshore is also here assumed to be by barges for 100 km.

Data for the CIP system are given in Table 4-4.

Table 4-4: Data for the CIP system.

Input	Value [total over lifetime]	Process in Ecoinvent
Linear polyethylene	689,40	polyethylene, LLDPE, granulate, at plant/ RER/ kg
Polyvinylchloride	172,35	polyvinylchloride, at regional storage/ RER/ kg
Low-alloy steel	2 585,25	steel, low-alloyed, at plant/ RER/ kg
Transport, by rail	689,40	transport, freight, rail/RER/tkm
Transport, by road	344,70	transport, lorry >16t, fleet average/RER/tkm
Transport, by sea	344,70	Transport, barge/RER/tkm

4.2.6 Chemical treatment

Chemical treatment is an important process as it eliminates any biological activity in the seawater, thus maximizing the system's lifetime. There is chemical treatment both before and after desalination. The pre-treatment includes acidification of the seawater in order to prevent precipitation of calcium carbonate on the membranes and to lower the pH value. Chlorination is also done to eliminate organic substances and bacterial growth, as well as injection of antifouling substances. The post-treatment process includes disinfection with chlorine gas and usage of sodium hypochlorite to mineralize the desalinated water and to adjust the pH value (Rivera, 2009).

Table 4-5 contains data for the chemical treatment process; type of chemical, amount and transport distances. Detailed calculations of the different amounts of chemicals can be found in Appendix B.

The sulphuric acid is used for acidification in the pre-treatment. It is assumed to have a purity of 96% and a concentration of 20 ppm is required. Sodium hypochlorite is used in both pre- and post-treatment to adjust the pH value. A concentration of 1 ppm is required for the pre-treatment, while 0.5 ppm is required for the post-treatment. Sodium bisulphate is used to lower the pH value for effective chlorination. This chemical is not in Ecoinvent, so sodium sulphate is used as a substitute. It is assumed that a concentration of 15 ppm is required (Rivera, 2009).

Iron(III) chloride is used in the pre-treatment to precipitate phosphates. However, this chemical is not found in Ecoinvent and aluminium sulphate is therefore used as a substitute, with an assumed purity of 40% and a concentration of 15 ppm. Calcium hydroxide is used as an antifouling agent in the post-treatment process, and a concentration of 0.5 ppm is assumed, as well as a purity of 20%. This cannot be found in Ecoinvent, and lime is thus used as a substitute. Sodium hexametaphosphate is used as antifouling in the pre-treatment process and phosphoric acid is used as a substitute. The required concentration is assumed to be 1 ppm (Rivera, 2009). The chlorine gas is used to disinfect the water and is used in both the pre- and post-treatment process (Muñoz and Fernández-Alba, 2008).

The Ecoinvent scenario for Europe assumes that chemicals are transported 600 km by train and 100 km by lorry (Frischknecht et al., 2007). The transport to the platform is assumed to be done by barges over a distance of 100 km.

Table 4-5: Data for chemical treatment.

Input	Value [total over lifetime]	Process in Ecoinvent
Sulphuric acid	19 432 828	sulphuric acid, liquid, at plant/ RER/ kg
Sodium hypochlorite	1 142 651	sodium hypochlorite, 15% in H2O, at plant/ RER/ kg
Sodium bisulfate	13 991 636	sodium sulphate, from Mannheim process, at plant/ RER/ kg
Iron(III) chloride	34 979 091	aluminium sulphate, powder, at plant/ RER/ kg
Calcium hydroxide	1 049 375	lime, hydrated, loose, at plant/ CH/ kg
Sodium hexametaphosphate	932 776	phosphoric acid, industrial grade, 85% in H2O, at plant/ RER/ kg
Chlorine gas	1 149	chlorine, gaseous, membrane cell, at plant/ RER/ kg
Transport, by rail	7 152 951	transport, freight, rail/RER/tkm
Transport, by road	42 917 703	transport, lorry >16t, fleet average/RER/tkm
Transport, by sea	7 152 951	Transport, barge/RER/tkm

4.2.7 Mechanical pre-treatment

The purpose of mechanical pre-treatment is to remove turbidity suspended particles from the inflow of seawater. In this case it is done by using sand filters. It is assumed that the RO plant has five horizontal filters made of carbon steel with a diameter of 4 meters and a length of 12.5 meters. Carbon steel is not found in the Ecoinvent database and low-alloy steel is therefore used as a substitute. The total amount of carbon steel needed is 1 217 500 kg. Each filter contains 200 000 kg of silica sand, resulting in a total amount of 1 000 000 kg (Rivera, 2009).

As for the previous inventories of components, metals are assumed to be transported 200 km by train and 100 km by lorry for the European scenario in Ecoinvent, while sand is assumed to be transported 50 km by lorry (Frischknecht et al., 2007). Offshore transport is assumed to be done by barges over a distance of 100 km.

Data for the mechanical pre-treatment is given in Table 4-6.

Table 4-6: Data for mechanical pre-treatment.

Input	Value [total over lifetime]	Process in Ecoinvent
Carbon steel	1 217 500	steel, low-alloyed, at plant/ RER/ kg
Silica sand	1 000 000	silica sand, at plant/ DE/ kg
Transport, by rail	243 500	transport, freight, rail/RER/tkm
Transport, by road	71 750	transport, lorry >16t, fleet average/RER/tkm
Transport, by sea	221 750	Transport, barge/RER/tkm

4.2.8 Construction of the reverse osmosis module

It is assumed that modifications on the platform, in terms of construction work, are required in order to implement the RO plant. It is therefore assumed that the amount of construction materials such as metals and concrete are the same for this plant as for a land based plant of the same size.

For the construction work of the plant, 10 000 kg of cast iron and 100 000 kg of steel are required. 2 000 000 kg of concrete and 750 000 kg of reinforced concrete is also required, but as reinforced concrete is not found in Ecoinvent, these are combined and modelled as regular concrete (Rivera, 2009).

The membranes in the RO unit are spiral wound membranes, made up of aromatic polyamide. This material is not found in Ecoinvent, thus nylon is used as a substitute. The membranes are assumed to have a lifetime of 5 years. The RO system also contains 8 pressurized cylindrical tubes that each contains 7 membranes. Each tube has 82 pressure pipes or lines. The tubes are made of plastic fiberglass reinforced with epoxy resin (Rivera, 2009). As an approximation, this is modelled as epoxy resin in the inventory.

The number of membranes used over the system's lifetime is calculated as follows:

$$\text{Number of membranes} = \frac{25\text{ years}}{5\text{ years}}\text{ substitutions} \cdot 7\,\frac{\text{membranes}}{\text{tubes}} \cdot 82\,\frac{\text{tubes}}{\text{line}} \cdot 8\text{ lines} = 22\,960\text{ membranes} \qquad (4.2)$$

Each membrane has a weight of 20 kg, resulting in a total requirement of 459 200 kg of aromatic polyamide. The tubes require a total amount of 22 960 kg of plastic fiberglass reinforced with epoxy resin (Rivera, 2009).

The European scenario in the Ecoinvent database assumes that metals are transported 200 km by train and 100 km by lorry. The same transport distances and means are also true for plastics. Concrete is assumed to be transported 50 km by lorry (Frischknecht et al., 2007). As for the other components, offshore transport is here assumed to be done by barge tankers over a distance of 100 km.

Data for the construction of the RO unit is displayed in Table 4-7.

Table 4-7: Data for the construction of the reverse osmosis system.

Input	Value [total over lifetime]	Process in Ecoinvent
Cast iron	10 000	cast iron, at plant/ RER/ kg
Steel	100 000	steel, low-alloyed, at plant/ RER/ kg
Concrete	2 750 000	concrete block, at plant/ DE/ kg
Epoxy resin	22 960	epoxy resin, liquid, at plant/ RER/ kg
Aromatic polyamide	459 200	nylon 66, glass-filled, at plant/ RER/ kg
Transport, by rail	118 432	transport, freight, rail/RER/tkm
Transport, by road	196 716	transport, lorry >16t, fleet average/RER/tkm
Transport, by sea	334 216	Transport, barge/RER/tkm

4.2.9 Disposal

In this thesis, landfilling is assumed as end-of-life scenario, in accordance with other life cycle assessments of reverse osmosis (Rivera, 2009) (Muñoz and Fernández-Alba, 2008) (Raluy et al., 2006). This is a worst-case scenario and it is likely that some of the materials will be recycled.

The different metals from the system; stainless steel, low-alloy steel and cast iron are modelled as steel in terms of demolition, and are assumed to be sent to inert material landfill. Epoxy resin and nylon are modelled as a mixture of plastics and are sent to sanitary landfill. Landfilling of sand is not found in Ecoinvent, and the disposal of silica sand is thus modelled as disposal of concrete gravel to final disposal. The concrete is sent to inert material landfill, while polyethylene and PVC are sent to sanitary landfill.

The materials are transported from the platform to shore by barges and the distance is assumed to be 100 km. The transport to the landfill is done by lorry in accordance with the Ecoinvent scenario for Europe. The distance to inert material landfill is assumed to be 15 km, while the distance to sanitary landfill is assumed to be 10 km (Frischknecht et al., 2007). The distance to final disposal for concrete gravel is assumed to be 50 km.

Data for the disposal phase is given in Table 4-8.

Table 4-8: Data for the disposal phase.

Input	Value [total over lifetime]	Process in Ecoinvent
Disposal of metals	1 560 085	disposal, steel, 0% water, to inert material landfill/ CH/ kg
Disposal of epoxy resin and nylon	482 160	disposal, plastics, mixture, 15.3% water, to sanitary landfill/ CH/ kg
Disposal of silica sand	1 000 000	disposal, building, concrete gravel, to final disposal/ CH/ kg
Disposal of concrete	2 750 000	disposal, concrete, 5% water, to inert material landfill/ CH/ kg
Disposal of polyethylene	689	disposal, polyethylene, 0.4% water, to sanitary landfill/ CH/ kg
Disposal of PVC	172	disposal, polyvinylchloride, 0.2% water, to sanitary landfill/ CH/ kg
Transport, by road	119 481	transport, lorry >16t, fleet average/RER/tkm
Transport, by sea	579 311	Transport, barge/RER/tkm

5 Life Cycle Impact Assessment

This chapter presents the results from the impact assessment of the LSW system. The total environmental impacts are given and they are also distributed across the different foreground processes of the system. As mentioned in chapter 3, the mid-point oriented approach is used in this thesis, meaning that “environmental impacts” refer to potentials for damage.

The impact assessment is done by using the ReCiPe method with a hierarchist perspective; see chapter 3.3 for a brief description.

5.1 Total environmental impacts

Table 5-1 shows the total environmental impacts from the recovery of 1 Sm^3 of crude oil by using LSW. The results are also normalised with respect to the ReCiPe normalisation factors for Europe, which can be downloaded from www.lcia-recipe.net.

Table 5-1: Total environmental impacts for 1 Sm^3 of crude oil produced with LSW.

Impact category	Value	Unit	Normalised
Agricultural land occupation	1,33E-01	m^{2}yr	2,95E-05
Climate change	1,51E+02	kg CO_2 eq	1,34E-02
Fossil depletion	6,26E+01	kg oil eq	4,03E-02
Freshwater ecotoxicity	9,65E-02	kg 1,4-DCB eq	8,87E-03
Freshwater eutrophication	3,96E-03	kg P eq	9,54E-03
Human toxicity	3,59E+00	kg 1,4-DCB eq	6,06E-03
Ionizing radiation	1,89E+00	kg U^{235} eq	3,02E-04
Marine ecotoxicity	1,34E-01	kg 1,4-DCB eq	1,58E-02
Marine eutrophication	4,86E-02	kg N eq	4,81E-03
Metal depletion	1,42E+00	kg Fe eq	2,00E-03
Natural land transformation	3,02E-02	m^2	1,87E-01
Ozone depletion	1,88E-05	kg CFC-11 eq	8,53E-04
Particulate matter formation	1,11E-01	kg PM eq	7,44E-03
Photochemical oxidant formation	4,07E-01	kg NMVOC eq	7,66E-03
Terrestrial acidification	3,29E-01	kg SO_2 eq	9,58E-03
Terrestrial ecotoxicity	3,05E-03	kg 1,4-DCB eq	3,71E-04
Urban land occupation	1,30E-01	m^{2}yr	3,19E-04
Water depletion	2,45E-02	m^3	0

From Table 5-1 it can be seen that the highest impacts after normalisation can be found for natural land transformation, climate change, marine ecotoxicity, fossil depletion and terrestrial acidification, indicating that these are the most important impact categories. The recovery of 1 Sm^3 of crude oil contributes to natural land transformation with 0.03 m^2. The contribution to climate change is 150.56 kg of CO_2 equivalents and for fossil resource depletion it is 62.64 kg of oil equivalents. Note that the fossil depletion potential *does not* include the depletion of crude oil from the recovery process itself. The recovery of 1 Sm^3 of crude oil also contribute to terrestrial acidification with 0.33 kg of SO_2 equivalents and to marine ecotoxicity with 0.134 kg of 1,4-DCB equivalents. The normalisation factor for water depletion was 0, therefore this is excluded and the normalised value is given as 0.

5.2 Environmental impacts distributed across the foreground processes

Figure 12 illustrates the relative contribution of each foreground process to the total impacts for all of the impact categories. The absolute values can be found in Table C-1, Appendix C.

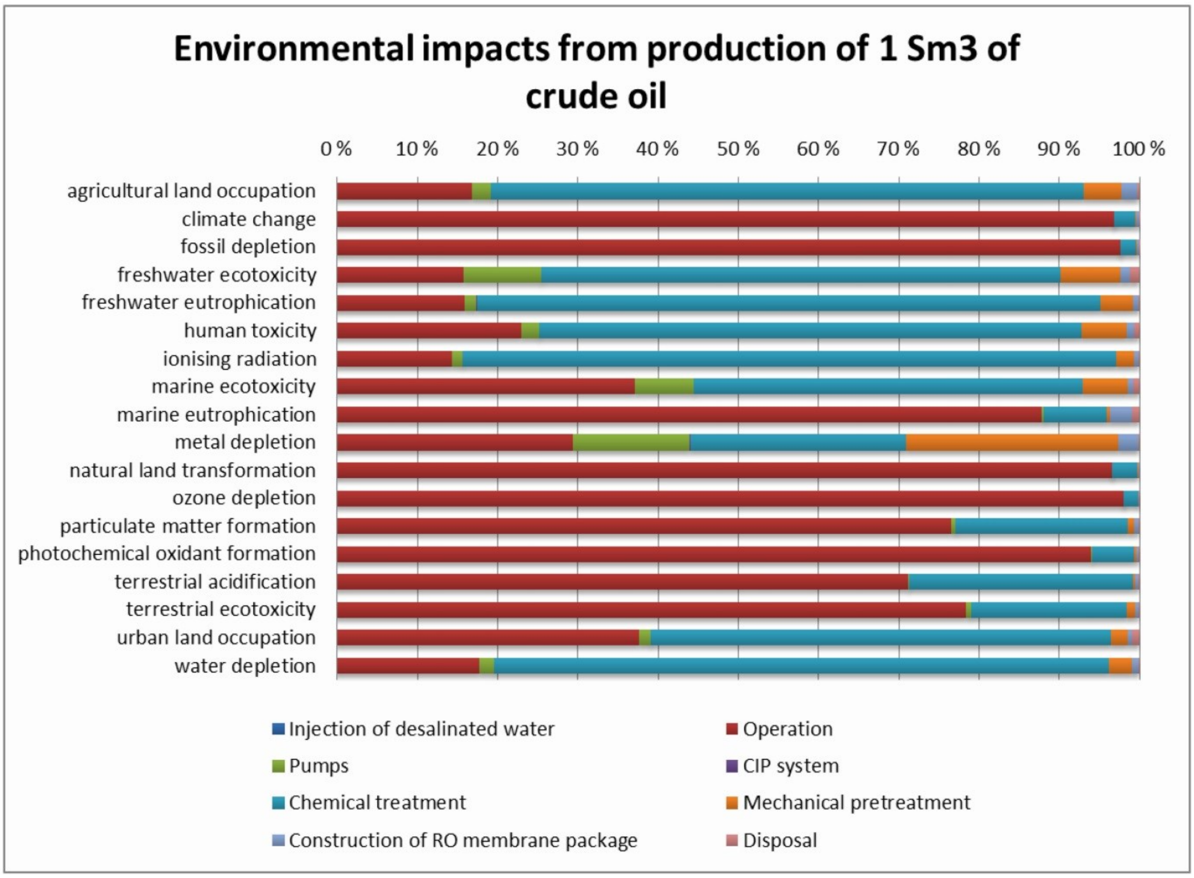

Figure 12: Environmental impacts from recovery of 1 Sm^3 of crude oil distributed across the foreground processes.

From the figure it can be observed that the operation phase is the process with the largest contribution in 10 out of 18 impact categories and ranges from 14.3% for ionizing radiation to 98% for ozone depletion. The chemical treatment process is the largest contributor in the remaining 8 impact categories and has a range of contribution from 1.8% for ozone depletion to 81.5% for ionising radiation.

After the operation and chemical treatment phases, mechanical pre-treatment and pumps contribute the most to environmental impacts. Mechanical pre-treatment has contributions ranging from 0.06% for ozone depletion to 26.4% for metal depletion. For the pumps, contribution ranges from 0.04%, for both ozone depletion and natural land transformation, to 14.6% for metal depletion.

From Figure 12 we see that the construction of the RO membrane package and the disposal phase contribute to a lesser extent than the processes mentioned in the previous paragraphs. The construction of the RO membrane package contributes in the range of 0.02% for ozone depletion to 2.7% for marine eutrophication. The disposal phases contributes from -0.08% for natural land transformation to 1.1% for freshwater ecotoxicity. The injection of desalinated water has no environmental impacts, while the contributions from the CIP system are negligible for most of the impact categories, with its highest contribution of 0.06% to metal depletion.

Structural path analysis show that most of the environmental impacts from the operation phase is associated with the combustion of natural gas in the gas turbine. Other background processes that contribute are the transportation of the natural gas, the use of copper in the turbine, the exploration and production wells and processes linked to the onshore and offshore natural gas production sites.

For the chemical process, it can be seen from the structural path analysis that most environmental impacts originate from the wastes and energy consumption for the production of aluminium sulphate. The production of phosphoric and sulphuric acids is also important contributors to environmental impacts.

For the pumps the main contributors to environmental impacts are the wastes and the extraction of raw metals for the manufacturing of stainless steel. Likewise, for the mechanical treatment the environmental impacts are associated with the manufacturing of low-alloy steel, such as wastes and extraction of raw materials.

5.3 Stressors for selected impact categories

In the previous paragraph we looked at the contributions of the different foreground processes to the environmental impacts. To better understand the environmental impacts of a system it may also be useful to look at the different stressors which contribute to the respective impact categories.

Stressors are substances or compounds which exerts a strain on the environment (Strømman, 2010). Based on the normalised total environmental impacts, it is chosen to look at stressors contributing to climate change, fossil depletion, marine ecotoxicity, natural land transformation and terrestrial acidification.

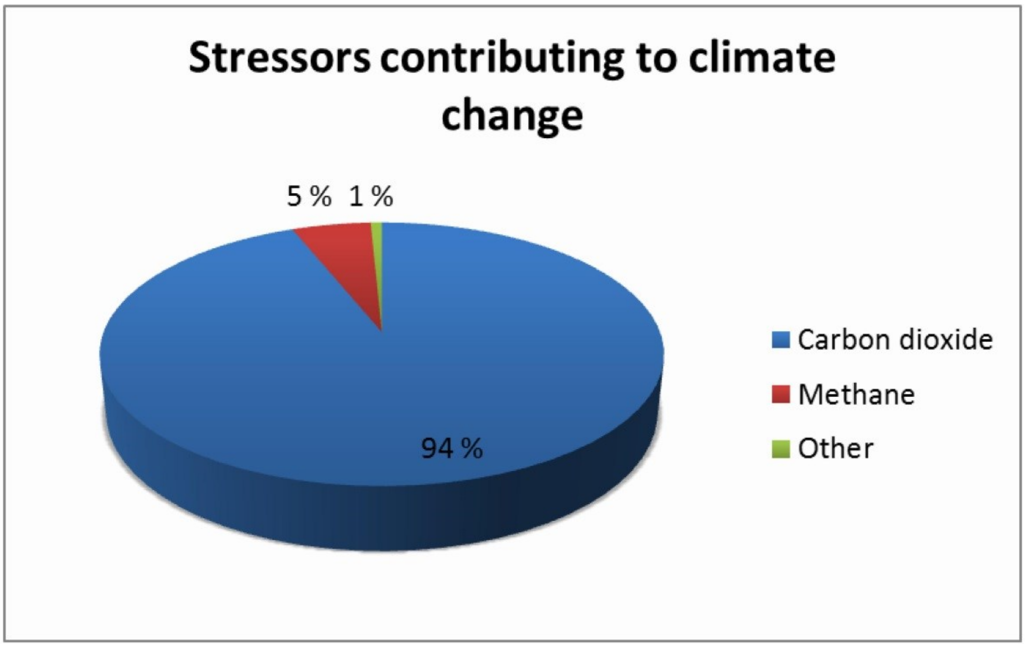

Figure 13: Distribution of stressors contributing to climate change.

Figure 13 displays the main stressors that contribute to climate change. We see that 94% of the stressors are CO_2 emissions to air from fossil energy sources, while methane represents 5% of the emissions. Absolute values can be found in Table C-2, Appendix C.

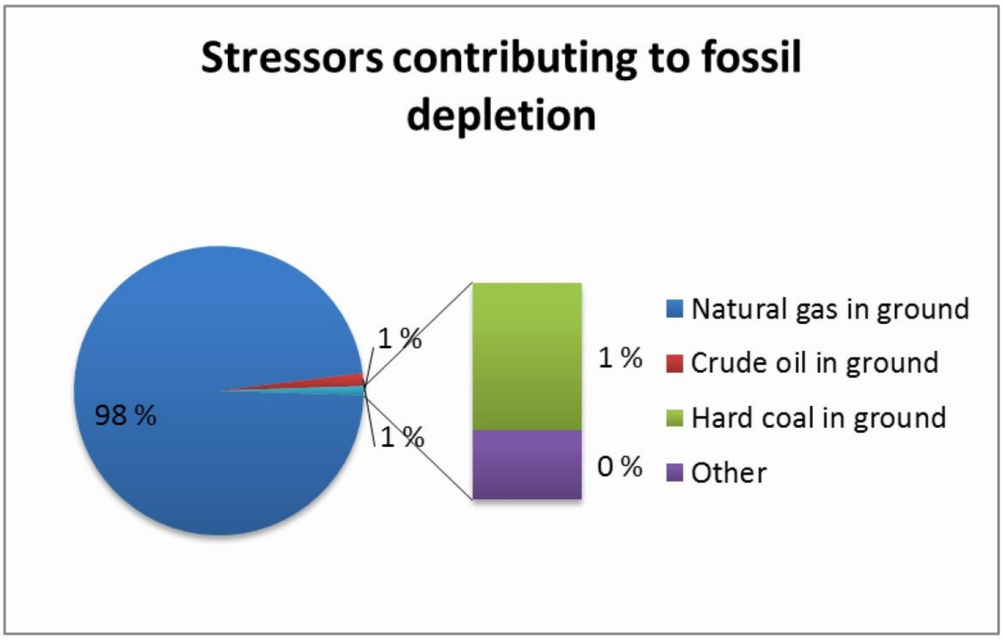

Figure 14: Distribution of stressors contributing to fossil depletion.

Figure 14 shows the stressors which contribute to fossil depletion. We see that natural gas from the ground is the stressor with the highest contribution of 98%. Crude oil from the ground contributes with 1% and hard coal has a contribution of 1%. Absolute values can be found in Table C-3, Appendix C.

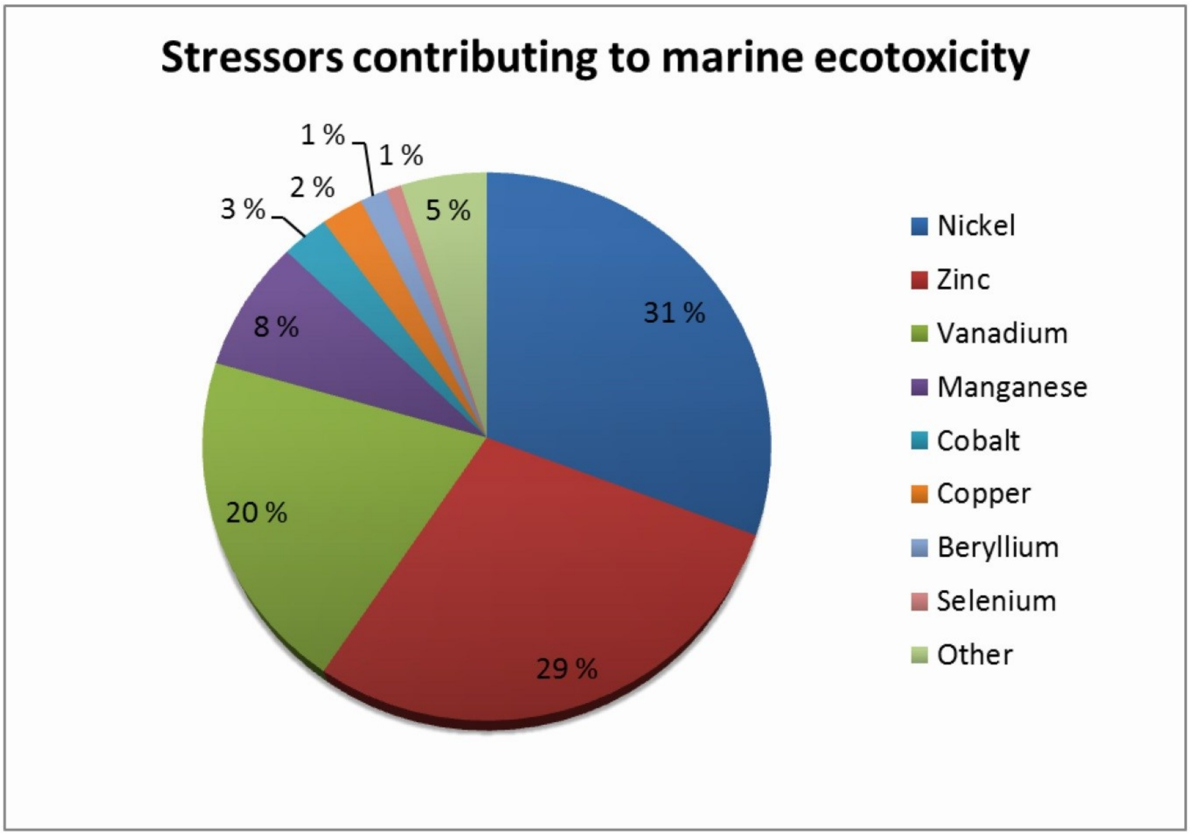

Figure 15: Distribution of stressors contributing to marine ecotoxicity.

Figure 15 shows the stressors which contribute to marine ecotoxicity and we see that nickel contribute to 31%. Zinc in its various forms emitted to water contributes to 29%, while emissions of vanadium to air and water contribute to 20%. Manganese has an 8% contribution while cobalt contributes with 3%. Other contributing substances are chromium, beryllium, copper and selenium. Absolute values can be found in Table C-4, Appendix C.

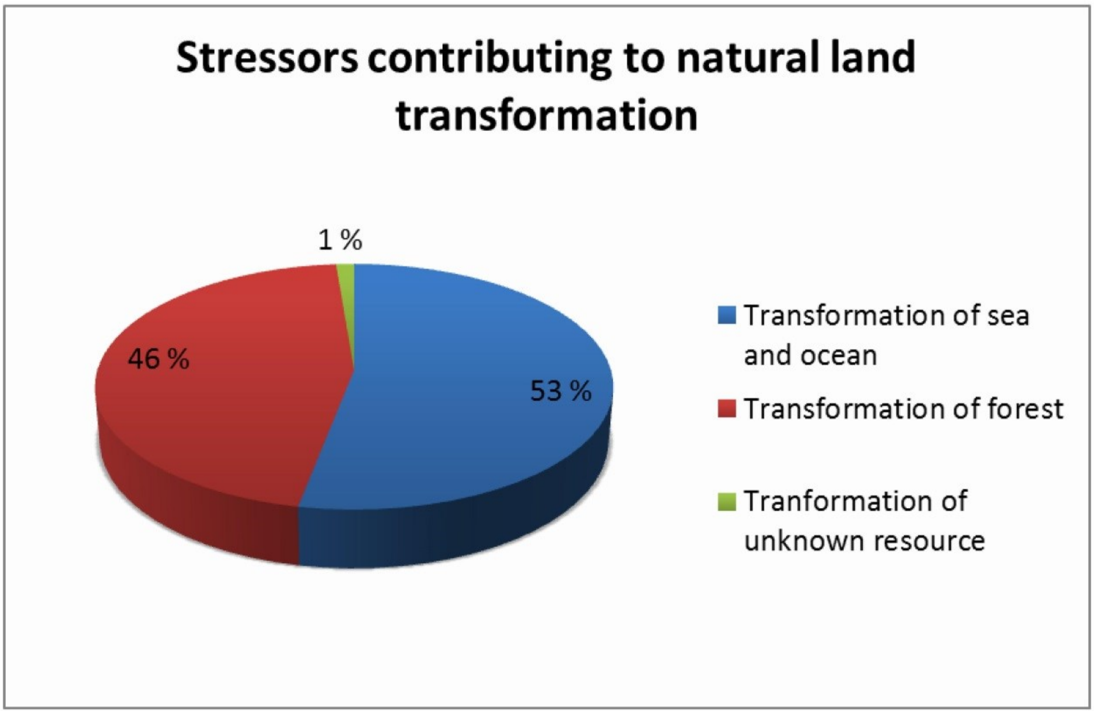

Figure 16: Distribution of stressors contributing to natural land transformation.

Figure 16 displays the stressors that contribute to natural land transformation. 53% can be ascribed to transformation of sea and ocean. 46% can be ascribed to transformation of forest, while transformation of unknown resource contributes to 1%. Absolute values can be found in Table C-5, Appendix C.

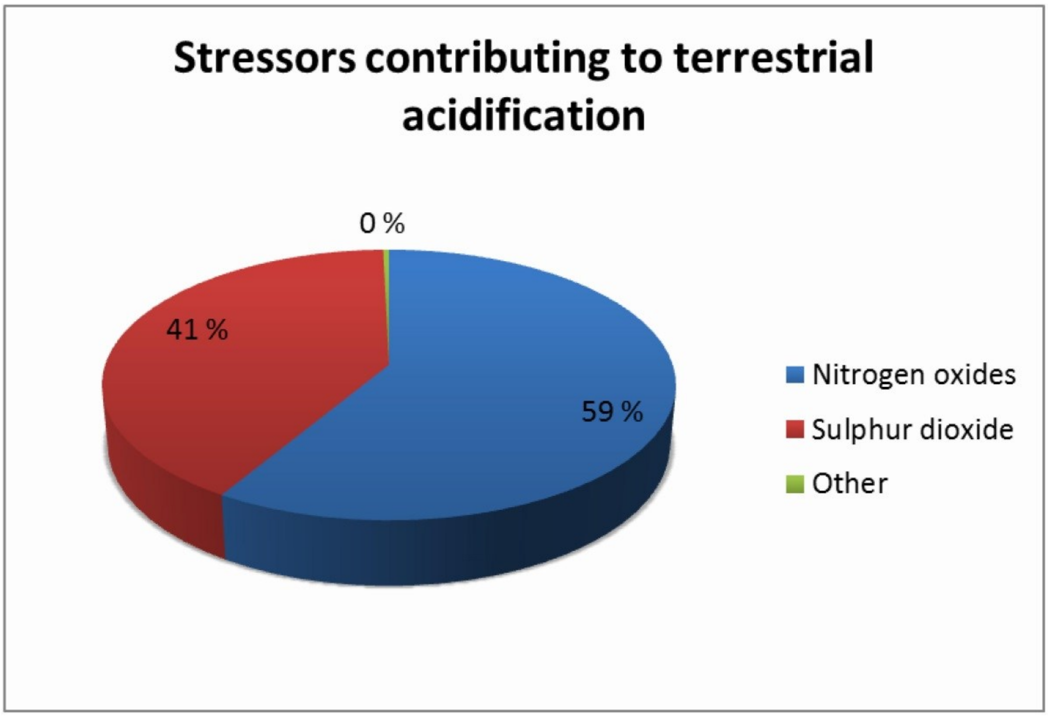

Figure 17: Distribution of stressors contributing to terrestrial acidification.

Figure 17 shows the stressors which contribute to terrestrial acidification. It can be seen that 59% of the stressors are nitrogen oxides, while 41% are sulphur dioxide. Absolute values can be found in Table C-6, Appendix C.

6 Sensitivity Analysis

This chapter contains the sensitivity analysis, where the robustness of the results from the LCIA is investigated. Different assumptions made in the inventory analysis are tested for some of the most relevant environmental impact categories. Alternative scenarios for power supply are also investigated.

6.1 Electrification of the platform

There is currently an on-going debate in Norway on whether or not to electrify the Norwegian continental shelf in the North Sea. Today, near all of the platforms in the North Sea uses gas turbines for power generation and for the operation of compressors and pumps. About 26% of the total Norwegian emissions of greenhouse gases are emitted by the oil and gas industry and the use of gas turbines is a major contributor to this (SSB, 2013). Miljøverndepartementet (2006) state that CO_2 emissions can be reduced by 9.4 million tonnes per year if all the gas turbines on the Norwegian continental shelf were replaced by power produced onshore. However, a key issue would then be to generate power using renewable energy sources or using gas power plants with carbon capture and storage.

From the impact assessment it was clear that the operation phase is a large contributor to environmental impacts. This is due to the energy requirement which is met by generating power from a gas turbine. As electrification is a topical option to the traditional power generation on platforms, it is chosen to do a sensitivity analysis were the platform is modelled as being supplied with power from the onshore electricity grid.

It is chosen to use the system described in Rasmus Nikolai Nes' master thesis (2012), which uses high voltage direct current (HVDC) cables to transport power offshore. HVDC cables are chosen over high voltage alternating current (HVAC) cables as HVAC cables have limited transmission capacities when the distance is larger than about 70 km. This has to do with the HVAC cable's large capacitance which results in high reactive currents. A high reactive current result in power losses in the cable, thus reduce the cable's ability to transmit active power (Hernando et al., 2011).

The system described by Nes (2012) includes export cables, which are 450 kV HVDC cables that transmit power, as well as offshore and onshore substations, containing electrical equipment and the power converter. The structure of the substation consists of topside and foundation and is required to support the substation offshore. DC breakers and switchgears are also included in the model. The HVDC cables have a capacity of 700 MW. These components were included in the model as an addition to the already existing foreground processes. It was assumed a lifetime of 25 years for the HVDC cables, substations and all of the equipment.

It was chosen to make three different scenarios with three different electricity mixes; European, Nordel and Norwegian, as a replacement for the gas turbine. The Norwegian electricity mix was chosen as Norway dominates the oil production in the North Sea. The results will be displayed for five of the environmental impact categories; climate change, fossil depletion, marine ecotoxicity, natural land transformation and terrestrial acidification. The functional unit is still 1 Sm^3 of crude oil recovered from the petroleum reservoir. Thus, it is assumed that the HVDC cables only transmit power to this one platform. The complete inventory list for the *additional* processes required for electrification of the platform is given in Table D-1, Appendix D.

6.1.1 Scenario 1 - European electricity mix

In the first scenario, a European production mix was chosen in accordance with Itten et al. (2012). Table 6-1 shows the total environmental impacts for the selected impact categories. Environmental impacts for all impact categories can be found in Table D-2, Appendix D.

Table 6-1: Selected total environmental impacts for the recovery of 1 Sm^3 of crude oil on an electrified platform supplied by the European electricity mix.

Impact category	Value	Unit
Climate change	1,10E+02	kg CO_2 eq
Fossil depletion	3,23E+01	kg oil eq
Marine ecotoxicity	1,58E+00	kg 1-4-DCB eq
Natural land transformation	1,55E-02	m^2
Terrestrial acidification	5,40E-01	kg SO_2 eq

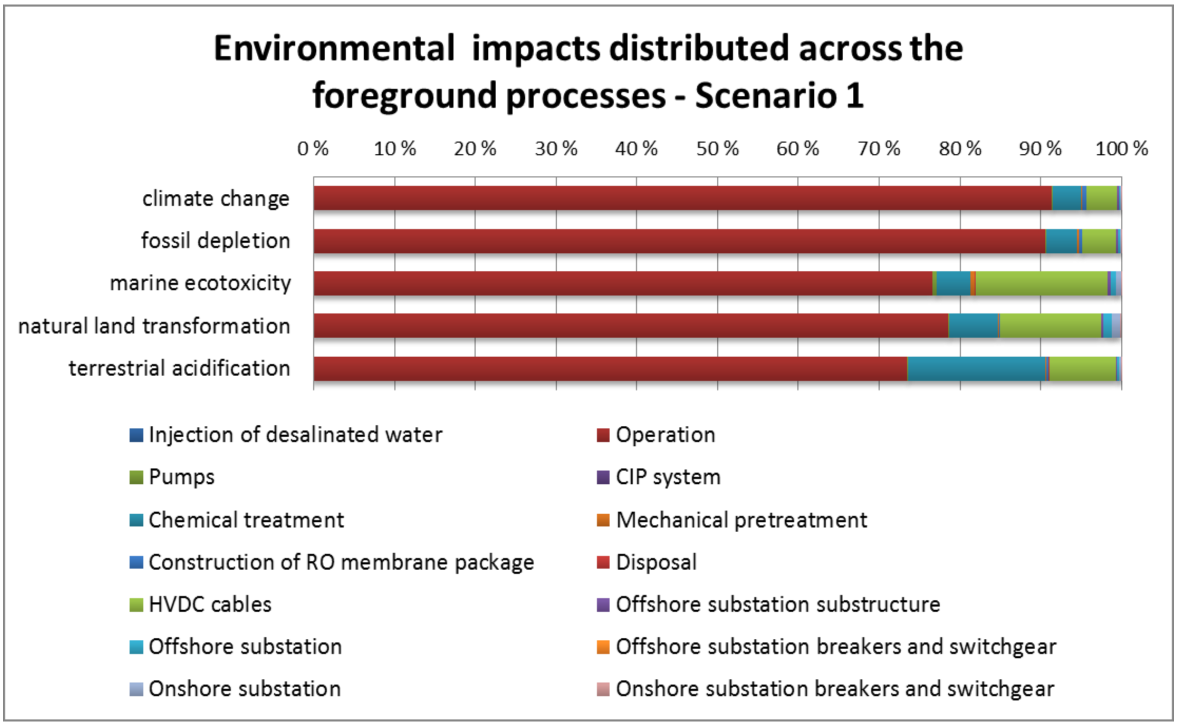

Figure 18: Environmental impacts distributed across the foreground processes for scenario 1.

Figure 18 gives the environmental impacts distributed across the foreground processes. The operation phase is the largest contributor in all five of the impact categories, and has a contribution ranging from 73.5% for terrestrial acidification to 91.3% for climate change. The chemical treatment phase contributes in the range of 3.5% for fossil depletion to 17% to terrestrial acidification. The manufacturing of the HVDC cables have notable contributions to marine ecotoxicity (16.4%) and to natural land transformation (12.5%). The other foreground processes have only minor contributions.

6.1.2 Scenario 2 - Nordel electricity mix

In the second scenario, a Nordel production mix was selected as recommended by Itten et al. (2012). Table 6-2 displays the total environmental impacts for the five impact categories. Total environmental impacts for all impact categories can be found in Table D-2, Appendix D.

Table 6-2: Selected total environmental impacts for the recovery of 1 Sm^3 of crude oil on an electrified platform supplied by the Nordel electricity mix.

Impact category	Value	Unit
Climate change	4,36E+01	kg CO_2 eq
Fossil depletion	1,18E+01	kg oil eq
Marine ecotoxicity	5,21E-01	kg 1-4-DCB eq
Natural land transformation	8,54E-03	m^2
Terrestrial acidification	2,30E-01	kg SO_2 eq

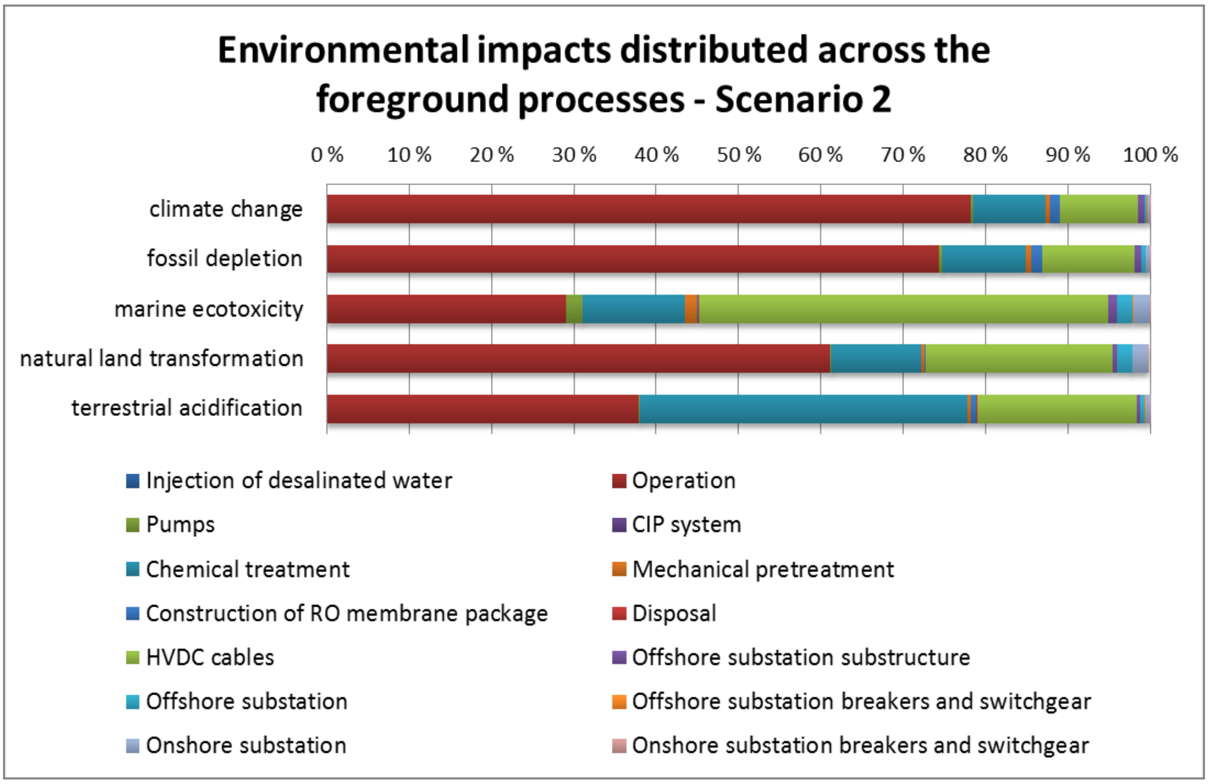

Figure 19: Environmental impacts distributed across the foreground processes for scenario 2.

Figure 19 shows the environmental impacts distributed across the foreground processes of the system. We see that the operation phase is the largest contributor to climate change (78.2%), fossil depletion (74.4%) and natural land transformation (61.5%). The HVDC cable is the largest contributor to marine ecotoxicity with 49.6%, while the chemical treatment process is the largest contributor to terrestrial acidification with 39.8%. The contributions of the other foreground processes are minor.

6.1.3 Scenario 3 - Norwegian electricity mix

For the third scenario it is chosen to use the Norwegian electricity supply mix. Itten et al. (2012) state that the supply mix should be chosen when electricity from a specific country is modelled. The reason for this is that the supply mix includes both domestic production and imports. The total environmental impacts for five of the impact categories are shown in Table 6-3. Total environmental impacts for all impact categories can be found in Table D-2, Appendix D.

Table 6-3: Selected total environmental impacts for the recovery of 1 Sm^3 of crude oil on an electrified platform supplied by the Norwegian electricity mix.

Impact category	Value	Unit
Climate change	1,68E+01	kg CO_2 eq
Fossil depletion	4,91E+00	kg oil eq
Marine ecotoxicity	4,26E-01	kg 1-4-DCB eq
Natural land transformation	5,78E-03	m^2
Terrestrial acidification	1,64E-01	kg SO_2 eq

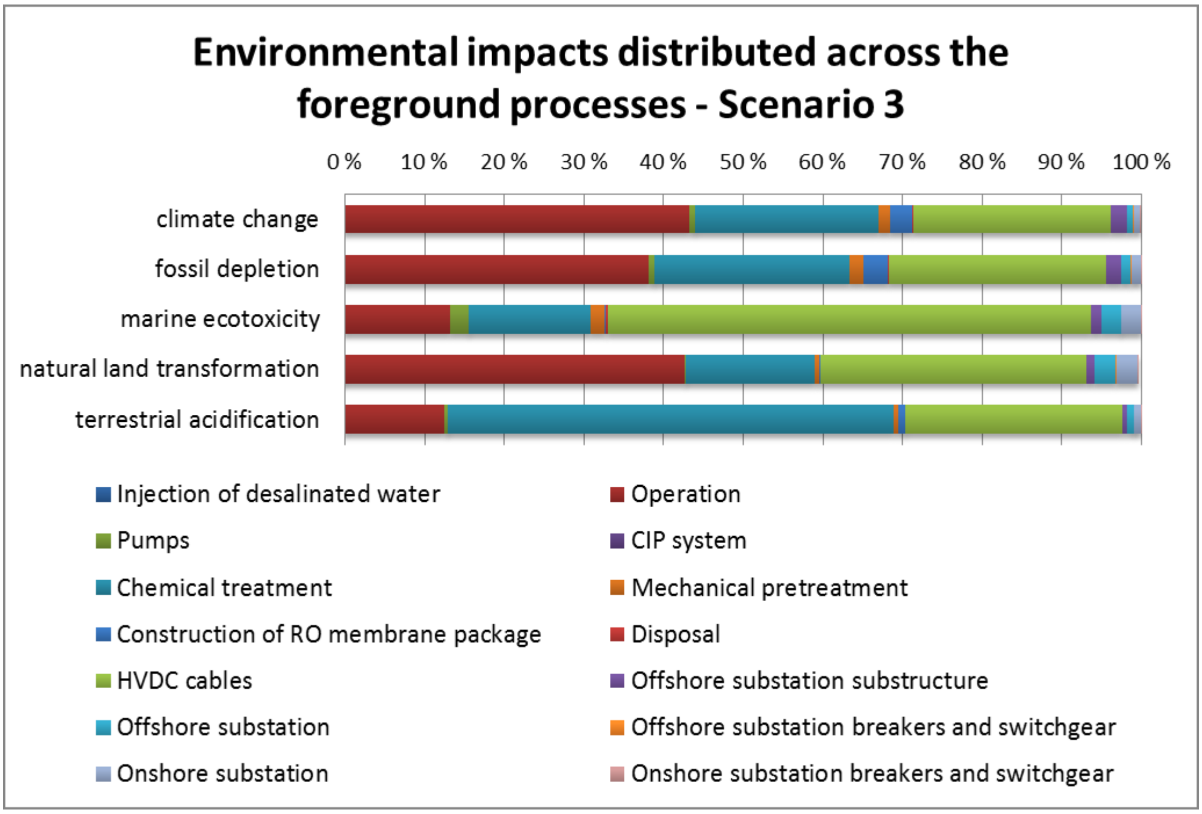

Figure 20: Environmental impacts distributed across the foreground processes for scenario 3.

Figure 20 gives the environmental impacts distributed across the foreground processes for five of the impact categories. It can be seen that the operation phase is the largest contributor in three of the five impact categories, and ranges from 12.5% for terrestrial acidification to 43.3% for climate change. The HVDC cable is the largest contributor to marine ecotoxicity (60.7%), and is also a substantial contributor to fossil depletion, natural land transformation, climate change and terrestrial acidification, with 27.3%, 33.7%, 24.8% and 27.2%, respectively. The chemical treatment process is the largest contributor to terrestrial acidification with 56% and has a 22.9% contribution to climate change and 24.5% contribution to fossil depletion. Impacts from the other processes are of lesser importance.

6.1.4 Comparison of the results from scenarios 1-3

Table 6-4 gives the total environmental impacts for the five selected impact categories. It compares the results from the impact assessment with the results from scenarios 1-3. Figure 21 shows this graphically and is normalised with respect to the case with the highest environmental impact in the respective categories.

Table 6-4: Total environmental impacts for the five selected impact categories, given for the original case and for the different electricity mixes.

Environmental impacts	Gas turbine	European electricity mix	Nordel electricity mix	Norwegian electricity mix	Unit
Climate change	1,51E+02	1,10E+02	4,36E+01	1,68E+01	kg CO_2 eq
Fossil depletion	6,26E+01	3,23E+01	1,18E+01	4,91E+00	kg oil eq
Marine ecotoxicity	1,34E-01	1,58E+00	5,21E-01	4,26E-01	kg 1,4-DCB eq
Natural land transformation	3,02E-02	1,55E-02	8,54E-03	5,78E-03	m^2
Terrestrial acidification	3,29E-01	5,40E-01	2,30E-01	1,64E-01	kg SO_2 eq

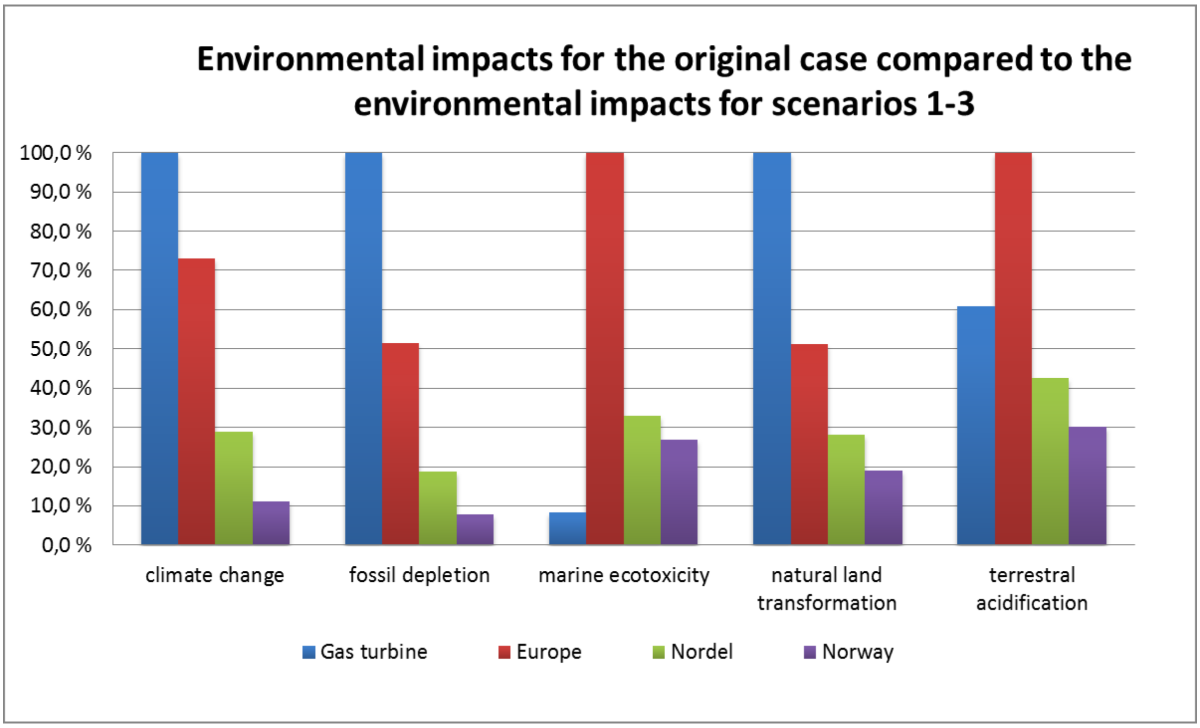

Figure 21: Comparison of environmental impacts for the different scenarios.

We see from Figure 21 that the GWP is highest for the original case, 150.56 kg of CO_2 equivalents, where the electricity is provided by burning natural gas in a gas turbine. The second highest GWP, 109.92 kg of CO_2 equivalents, is associated with scenario 1, which compared to the original case has a 27% lower contribution to climate change. Scenario 2 gives an even lower contribution to climate change, 43.64 kg of CO_2 equivalents, which is only 29% of the contribution of the original case. Scenario 3 results in the lowest GWP, with only 16.78 kg CO_2 equivalents per Sm^3 of crude oil recovered.

These differences can be explained by the various compositions of the electricity mixes used in the different cases. For the original case, natural gas is burnt in a gas turbine, thus resulting in high greenhouse gas emissions. The average electricity production in Europe consist of 43.3% thermal power, 40.3% nuclear power and 16.4% hydro power (Raluy et al., 2005a). A large fraction is thereby energy from non-fossil sources, meaning that the associated greenhouse gas emissions are lower, compared to the burning of only fossil fuels.

Nordel is a former transmission system operator (TSO) association between the Nordic countries; Norway, Sweden, Denmark, Finland and Iceland. Today, this has been replaced by the Nordic region in the European Network of Transmission System Operators for Electricity (ENTSO-E) (ENTSO-E, 2012). The typical electricity mix in the Nordel region consist of 58% hydro power, 20% nuclear power, 19% thermal power and 3% wind power (Nordel, 2009).

The Norwegian power generation differs greatly from power generation elsewhere in Europe, as 99% of the electricity is generated by hydro power (SFFE, 2011). It is chosen to use the supply mix in Ecoinvent, as recommended by Frischknecht et al. (2007). This electricity mix consist of 96.2% hydro power, 0.85% power from renewables, 0.08% power from waste, 0.1 % power from other sources and 2% imports from Sweden, Denmark, Finland, Netherlands and the Russian Federation.

From Figures 12, 18, 19 and 20, we also see that the relative contribution from the operation phase is different for all the cases. For the original case, the operation phase contributes to climate change with more than 96%. In scenario 1, the operation phase has a contribution of over 90%, for scenario 2 the fraction is below 80%. In scenario 3, the contribution has fallen to less than 45%. This means that not only is the absolute GWP different in all the cases, the distribution of impacts from the foreground processes is also very different. We see that the more renewable energy that is included in the electricity mix, the less is the GWP affected by the operation phase.

The fossil depletion potential is also highest for the original case, where the gas turbine is used to generate power. The fossil depletion potential here is 62.64 kg of oil equivalents. If the gas turbine is replaced by HVDC cables, the fossil depletion potential will be 32.32 kg of oil equivalents for Scenario 1, which is a reduction of 48%. If the Nordel electricity mix is used in scenario 2, it will result in a reduction of 81.1% compared to the original case, as the fossil depletion potential is 11.84 kg of oil equivalents. Scenario 3 results in the lowest fossil depletion potential, 4.91 kg of oil equivalents, corresponding to a reduction of 92.2% compared to the usage of gas turbines.

Figure 21 shows the same development for fossil depletion as for climate change; the original case has the highest contributions for the two impact categories, while the first scenario has the second highest contributions, and then follows scenario 2. Scenario 3 has the lowest contributions to both impact categories. This can be explained by the different compositions of the electricity mixes, as described previously. The original case gets all of its energy from burning of natural gas; thus resulting in large fossil fuel consumption. 43.3% of the European electricity mix is generated by thermal power; hence lower fossil fuel consumption is required. For the Nordel electricity mix, 19% of the electricity is generated by thermal power, resulting in an even lower consumption of fossil fuels. For the Norwegian electricity mix, structural path analysis reveals that only imports of electricity are responsible for the fossil consumption in the operation phase. Figure 20 shows that also the manufacturing of the HVDC cables and the chemical treatment process are major contributors to fossil fuel consumptions for this scenario.

Unlike the results for climate change and fossil depletion, we see that the original case has the lowest marine ecotoxicity potential, with 0.13 kg of 1,4-DCB equivalents per Sm^3 of crude oil recovered. The highest marine ecotoxicity potential, 1.58 kg of 1,4-DCB equivalents, is found for scenario 1. This is almost 12 times higher than the potential for the original case. Scenario 2 result in a marine ecotoxicity potential of 0.52 kg of 1,4-DCB equivalents, which is about four times higher than for the original case. The marine ecotoxicity potential for scenario 3 is 0.43 kg of 1,4-DCB equivalents, more than three times higher than for the original case. We also see that the marine ecotoxicity potential for scenario 1 is a lot higher than for the other cases; three times higher than the potential for scenario 2, which has the second largest marine ecotoxicity potential.

For the original case, the chemical treatment process is the largest contributor to marine ecotoxicity, followed by the operation phase. This differs from scenario 1, where the operation phase is the largest contributor, followed by the manufacturing of the HDVC cable. From the structural path analysis, we can see that the disposal from lignite mining, associated with thermal power production, gives the largest contribution to marine ecotoxicity for this scenario. Also, the use of copper in the HVDC cables is a major contributor.

Figure 19 shows that the manufacturing of the HVDC cables is the largest contributor to marine ecotoxicity for scenario 2, while the operation phase is the second largest contributor. The structural path analysis shows that the usage of copper is responsible for a large fraction of the marine ecotoxicity potential, as well as steel and aluminium manufacturing. The distribution of the environmental impacts for third scenario can be found in Figure 20. For this case the manufacturing of the HVDC cables is the largest contributor to marine ecotoxicity, while the operation phase and the chemical treatment phase are the second and third largest, respectively. From the structural path analysis we can see that the usage of copper in the HVDC cables is the main contributor to marine ecotoxicity for scenario 3.

Similar to the results for climate change and fossil depletion, the original case has the highest natural land transformation potential with 0.03 m^2. This is almost twice the value which can be found for scenario 1, which has a natural land transformation potential of 0.016 m^2. Scenario 2 has a natural land transformation potential of 0.009 m^2, only 28.3% of the value for the original case. The natural land transformation potential for scenario 3 is only 19.1% of the original case with a value of 0.006 m^2.

From Figure 18, we see that 78.8% of the natural land transformation potential for scenario 1 is attributed to the operation phase, while 12.5% is attributed to the HVDC cables. The interpretation of the structural path analysis shows that the transportation of HVDC cables by barge and the electricity generation by thermal power are the main contributors to the natural land transformation potential. For scenario 2, the operation phase is the main contributor to natural land transformation with 61.5%, followed by the manufacturing of the HVDC cables with 22.8%. The structural path analysis reveals that transportation of HVDC cables by barge are a contributor to this scenario as well. Also, electricity generated by hydropower and burning of crude oil, hard coal and softwood are major contributors. The same structural path analysis results can be seen for the third scenario.

From Figure 21 it can be seen that scenario 1 has the highest terrestrial acidification potential with 0.54 kg of SO_2 equivalents for each Sm^3 of crude oil extracted. This is 61% higher than the terrestrial acidification potential for the original case, which is 0.33 kg of SO_2 equivalents. Scenario 2 has the second lowest contribution to terrestrial acidification with 0.23 kg of SO_2 equivalents, about 30% lower than the value for the original case. Scenario 3 has the smallest contribution to terrestrial acidification with 0.16 kg of SO_2 equivalents, which is about half of the contribution from the original case.

From the structural path analysis it can be seen that for scenario 1, the thermal power production is the main contributor to terrestrial acidification as it used fossil fuels such as hard coal, lignite and heavy fuel oil. For scenario 2, the manufacturing of chemicals such as aluminium sulphate and phosphoric acid are major contributors to terrestrial acidification. Also the thermal power generation is a large contributor as it is associated with combustion of hard coal and peat. For scenario 3, the manufacturing of sulphuric acid and aluminium sulphate are the main contributors, followed by transportation and the use of copper in the manufacturing of the HVDC cables.

6.1.5 The importance of the HVDC cable length

In scenarios 1-3, the assumption was made that the required HVDC cable length is 100 km. In order to see how much this assumption affects the results, two more simulations were done; one where the HVDC cable length was halved to 50 km and one where the HVDC cable length was doubled to 200 km. It was chosen to do this for scenario 3, as the HVDC cable has the largest contributions for this scenario. Figure 22 shows the results graphically. Total environmental impacts for the different HVDC cable lengths can be found in Table D-3, Appendix D.

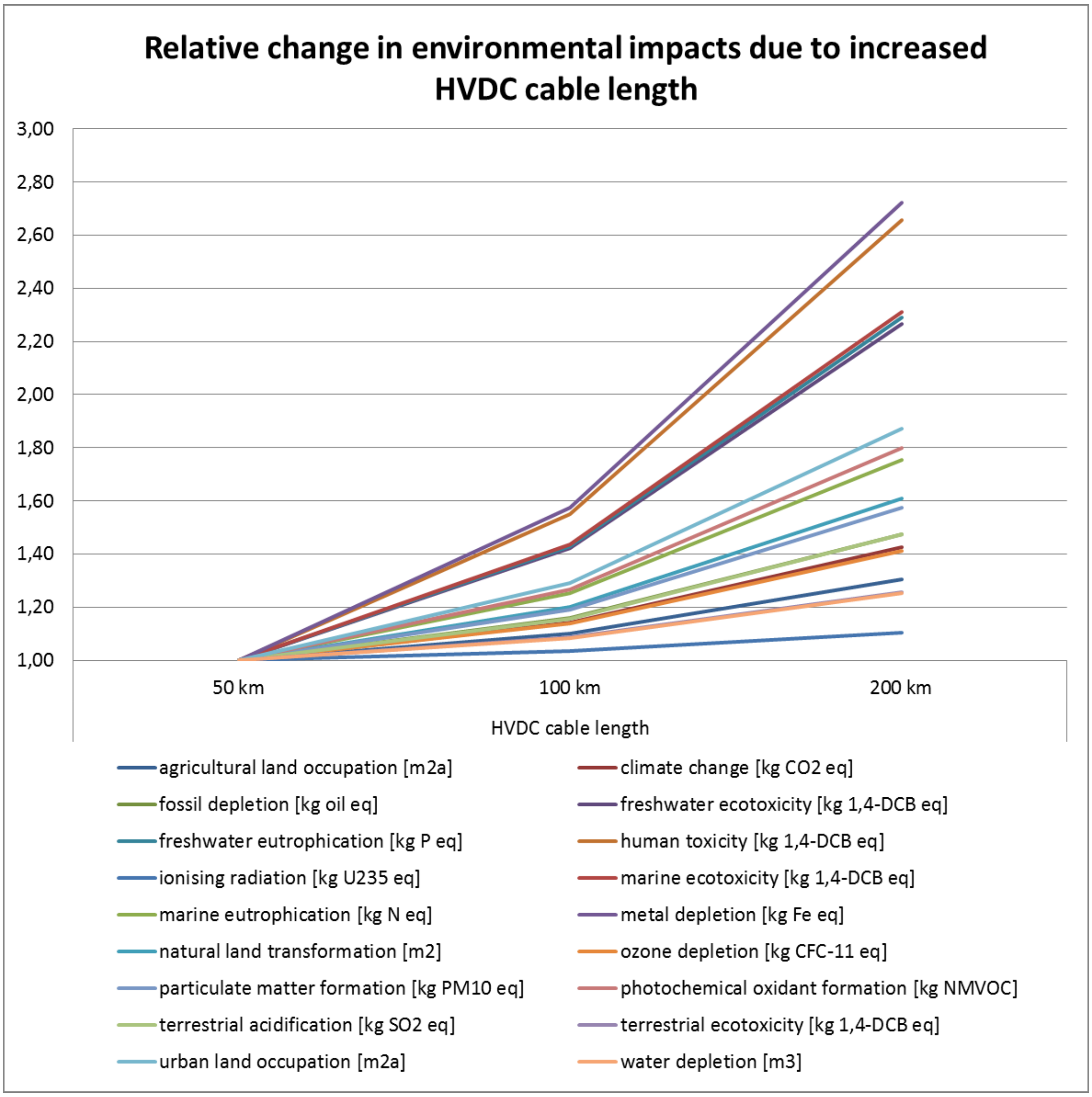

Figure 22: Relative change in environmental impacts due to increased HVDC cable length for scenario 3.

We see from Figure 22 that an increase in HVDC cable length has the highest impact on metal depletion. An increase from 50 km to 100 km results in an increase of 57%, while an increase from 100 km to 200 km results in an increase of 73% for this environmental impact caategory. The second highest impact can be found for human toxicity, a doubling from 50 km to 100 km results in an increase of about 55%, while a further doubling to 200 km result in an increase of about 71%. An increase from 50 km to 100 km gives an increase of about 4% for marine ecotoxicity and freshwater ecotoxicity, while a doubling to 200 km gives an increase of about 60%.

6.2 Electricity requirement

In the inventory analysis it was chosen to use the same electricity requirement as found in previous studies; 4 kWh per m^3 of desalinated water (Muñoz and Fernández-Alba, 2008, Rivera, 2009, Raluy et al., 2004). The reverse osmosis facilities used in these studies differs from the facility used in this thesis as they are land-based plants. The facility used in this thesis is modelled as an offshore plant integrated on a platform. It is conceivable that the offshore reverse osmosis facility has a somewhat larger energy requirement than the land-based facilities, and it is therefore modelled increases in the energy requirement to see how this affects the total environmental impacts of the system. It is chosen to increase the energy requirement by 20%, 50% and 80%, corresponding to energy requirements of 4.8 kWh, 6 kWh and 7.2 kWh per m^3 desalinated water.

Figure 23 gives the relative change in environmental impacts as a result of increased electricity requirement. As the relative increase for some of the impact categories are identical, it is difficult to see all the different impact categories in the figure. Reference is thus made to the absolute values given in Table D-4, Appendix D.

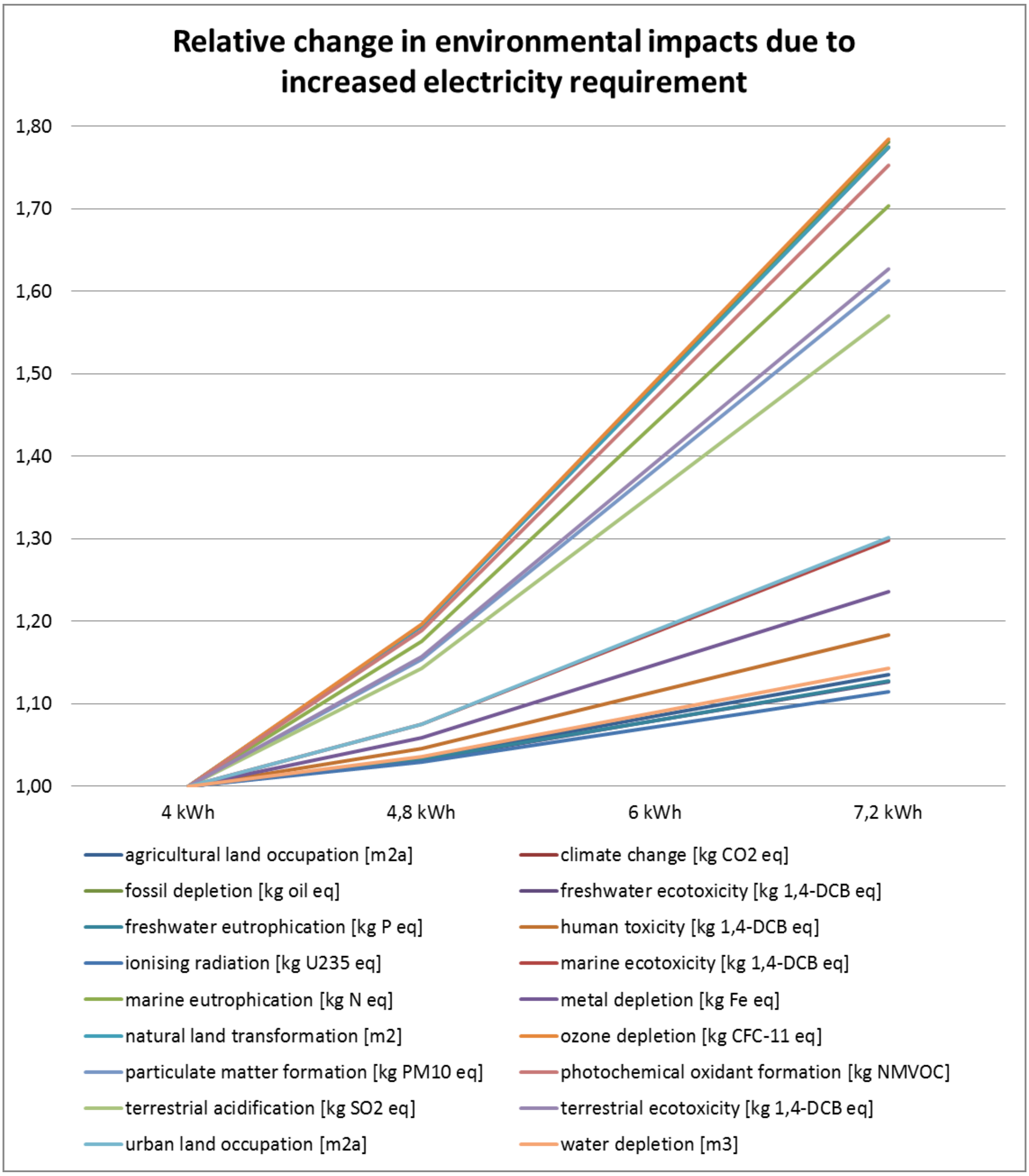

Figure 23: Relative change in environmental impacts due to increased electricity requirement.

From Figure 23 and Table D-4, Appendix D, it can be seen that fossil depletion and ozone depletion have identical relative changes for all the energy requirements. An increase in energy requirement of 20% from 4 kWh to 4.8 kWh result in an increase of 20% in fossil depletion potential and ozone depletion potential. Similarly, for a 50% increase in energy requirement, the fossil and ozone depletion potentials increase 49%, while an energy requirement increase of 80% result in 78% increase in the fossil and ozone depletion potentials. Climate change and natural land transformation potentials have near identical increases; 19%, 48% and 77% respectively, while the photochemical oxidant formation potential have increases of 19%, 47% and 75%. Other impact categories, such as marine eutrophication, terrestrial ecotixicity, particulate matter formation and terrestrial acidification, is also largely impacted by an increase in energy requirement. The other impact categories are less impacted, as can be seen from the figure, and increase in energy requirement of 80% will only result in increases in environmental impacts of 30% or less.

Figure 23 shows that an increase in energy requirement will affect fossil depletion and ozone depletion the most, followed by climate change and natural land formation. The potentials of these environmental impact categories will increase in the same proportion as the energy requirement. This because the operation phase is the main contributor with over 95% in all of these categories and the energy requirement is a direct input into the operation phase. Also other impact categories that are dominated by the operation phase, such as photochemical formation, marine eutrophication, particulate matter formation and terrestrial acidification and ecotoxicity, are largely impacted by an increase in energy requirement.

6.3 Lifetime of the system

In the inventory analysis it is assumed a lifetime of 25 years for the system. It is decided to test this assumption and to see how the environmental impacts are affected if the lifetime is reduced. Three new simulations will be done; one with a lifetime of 20 years, one with a lifetime of 15 years and one with a lifetime of 10 years. Figure 24 displays the results from the new simulations. The total environmental impacts for all the lifetime scenarios are given in Table D-5, Appendix D.

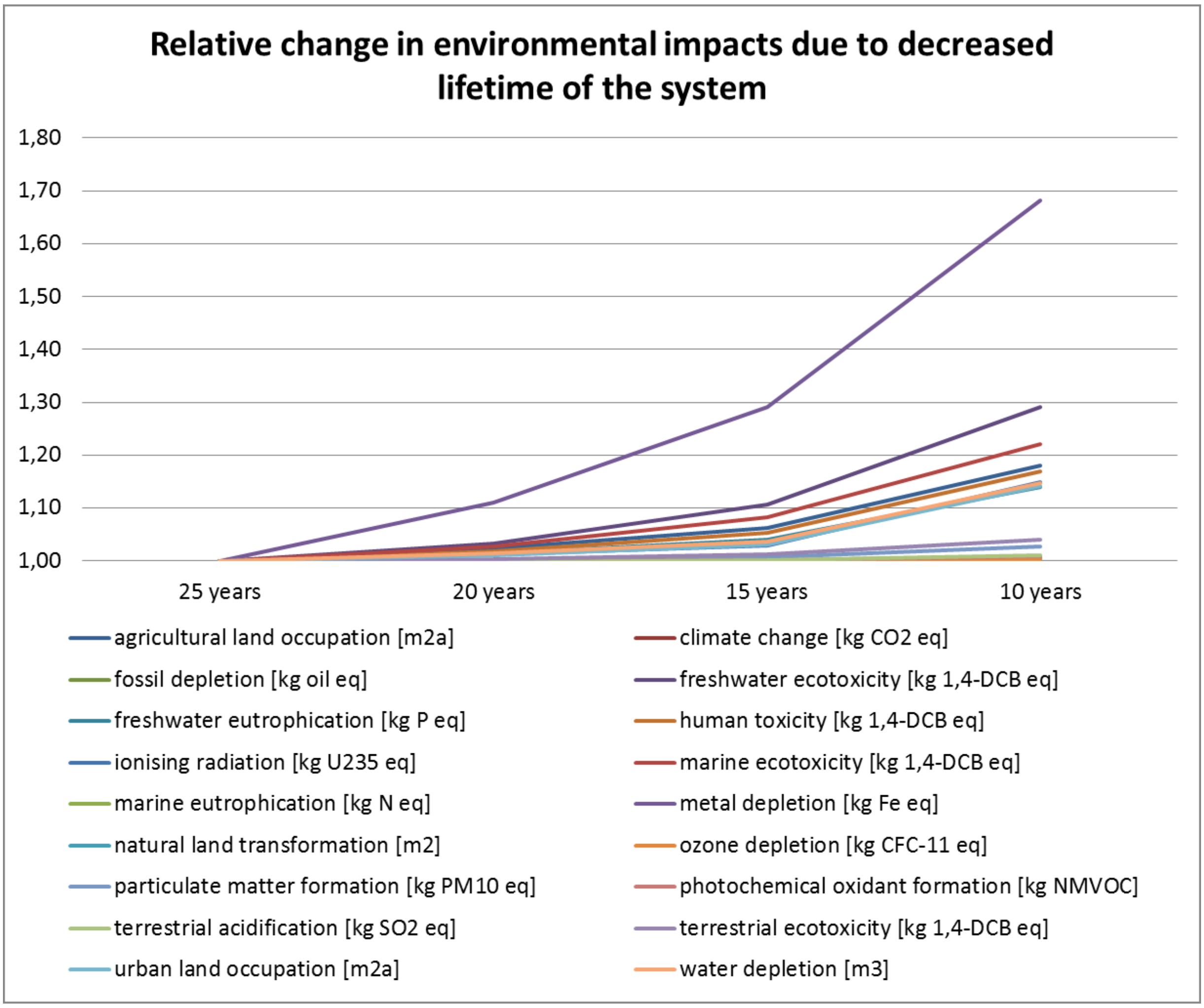

Figure 24: Relative change in environmental impacts due to decreased system lifetime.

From the Figure 24 we can see that metal depletion is the impact category that is most affected by a decrease in system lifetime. A 20% reduction in the system's lifetime, from 25 years to 20 years, result in an increase in the metal depletion potential of 11%, while a 40% reduction, from 25 years to 15 years, result in an increase of 29 %. Reducing the lifetime of the system with 60%, from 25 years to 10 years, will increase the metal depletion potential with 68%. The freshwater ecotoxicity potential have somewhat smaller increases; 3%, 11% and 29%, while the marine ecotoxicity potential have increases of 3%, 8% and 22%. Other impact categories, such as agricultural land occupation, human toxicity, water depletion and ionising radiation, are also affected by a decrease in the system's lifetime. The other impact categories are affected to a lesser degree.

When the system's lifetime is decreased we can see from Figure 24 that metal depletion is the impact category that is most affected, followed by freshwater ecotoxicity and marine ecotoxicity. Other impact categories, such as climate change, fossil depletion and ozone depletion, have no change at all for a 60% decrease in system lifetime. Common to the impact categories that are most affected is that the manufacturing of pumps and/or the mechanical pre-treatment phase is a notable contributor to environmental impacts. The most likely explanation for this is that the inputs to the manufacturing of pumps and to the mechanical pre-treatment phase are mostly amounts of metals which are independent of the operation time, i.e. the lifetime of the system. The main inputs into the impact categories that are not affected by a decrease in the system's lifetime are parameters that are dependent on the system's lifetime i.e. they decrease or increase proportionally with the lifetime.

7 Discussion

7.1 Interpretation of results

The results from the LCA of LSW show that the operation phase is the largest contributor to environmental impacts in 10 out of 18 impact categories, while the chemical treatment process is the largest contributor in the other eight impact categories. The mechanical pre-treatment process and the manufacturing of pumps also contributes to a varying extent, while the environmental impacts from the construction of the RO membrane package, the CIP system, the disposal phase and the injection of desalinated water are negligible.

Normalisation of the results with respect to the average European normalisation values indicate that climate change, fossil depletion, marine ecotoxicity, natural land transformation and terrestrial acidification are the most important environmental impacts associated with the system described in this thesis. It is therefore chosen to discuss the contributions from the different parts and components of the system to these impact categories.

The GWP of the recovery of 1 Sm^3 of crude oil using LSW is 150.56 kg of CO_2 equivalents. Figure 12 shows that the operation phase has a contribution of 96.9% to climate change. Figure 13 shows that 94% of the stressors are CO_2 emissions to air and 5% are methane emissions. Structural path analysis reveals that the majority of the stressors from the operation phase are caused by the burning of natural gas in gas turbines in order to generate electricity. When natural gas is burned, methane may be the emitted into the air if the natural gas is not burned completely (EPA, 2013). This is thus the likely explanation as to why the operation phase emits methane.

The fossil depletion potential for the system is 62.64 kg oil equivalents per Sm^3 of crude oil recovered. Figure 12 shows that 97.6% of the fossil depletion potential is due to the operation phase and Figure 14 shows that natural gas from the ground is the greatest stressor with a contribution of 98%. From the structural path analysis it can be seen that most of the stressors are associated with the background processes *natural gas at production onshore* and *natural gas at production offshore.* These processes include exploration and production of natural gas, as well as well testing. It is likely that the stressors *hard coal in ground* and *crude oil in ground* (see Figure 14) are associated with these processes.

For marine ecotoxicity, the potential for recovery of 1 Sm^3 of crude oil is 0.134 kg of 1,4-DCB equivalents. The chemical treatment process contributes to 48.5% of the marine ecotoxicity potential, as can be seen from Figure 15, while the operation phase has a contribution of 37.2%. The main stressors are nickel, zinc and vanadium, see Figure 15. Nickel is used in the industry to make stainless steel and other metallurgical alloys (Tilset, 2013), while zinc is often used for corrosion protection (Mostad, 2013). Vanadium is also primarily used in metallurgical alloys (Pedersen, 2013). The structural path analysis shows that the production of aluminium sulphate and phosphoric acid, as well as the exploration and production of natural gas are the main contributors. Also, the processes associated with the manufacturing of stainless steel and low-alloy steel for pumps and mechanical pre-treatment filters are important background processes which contribute to marine ecotoxicity.

The natural land transformation potential is 0.03 m^2 and the largest contributing foreground process is the operation phase (96.8%). The main stressors are transformation of sea and ocean and the transformation of forest, as can be seen from Figure 16. The structural path analysis tells us that the all of the stressors resulting in natural land transformation are connected to the wells for exploration and production of natural gas, as well as the pipelines for transportation of the gas.

From Table 5-1 we see that the terrestrial acidification potential for the system is 0.33 kg of SO_2 equivalents for each Sm^3 of crude oil recovered by LSW. From Figure 12 we can observe that the operation phase is the largest contributor to terrestrial acidification with 71.2%, followed by the chemical treatment process with 27.8%. From the structural path analysis it can be seen that the terrestrial acidification is mainly associated with the use of natural gas as a fuel in the gas turbine. Also, the manufacturing of sulphuric acid and aluminium sulphate contribute to this.

As mentioned in chapter 5.2, the production of aluminium sulphate and phosphoric acid are two of the largest contributions to environmental impacts for the chemical treatment process. These chemicals are used as a substitute for iron(III) chloride and sodium hexametaphosphate, respectively. These could not be found in the Ecoinvent database and it was decided to use aluminium sulphate and phosphoric acid as these are similar chemicals with similar production processes (Rivera, 2009). It can however not be ruled out that the environmental impacts perhaps would have been different if the production processes of iron(III) chloride and sodium hexametaphosphate would have existed in Ecoinvent. This is something we cannot know for sure, but as the production processes of these chemicals are similar, it is feasible that the results are approximate even including these substitutions.

In chapter 6, a sensitivity analysis was performed. A sensitivity analysis can be used to identify critical parameters that will affect the behaviour of the system, and parameters that can be neglected. It can also be used to validate a model and to test the robustness of the results (Smith et al., 2008). It was decided to model the platform as supplied with electricity from shore and three scenarios were made with three different electricity mixes; European, Nordic and Norwegian. It was chosen to look at the Norwegian electricity mix as this is a likely scenario for the future, in that Norway exports 90% of all the produced oil in the North Sea (Hagland, 2000). The issues concerning electrification on the shelf will therefore be discussed from a Norwegian point of view. It was also decided to test the results for an increase in energy requirement and a decrease in the system's lifetime.

The sensitivity analysis shows that by electrifying the platform, substantial reductions in contributions to climate change, fossil depletion and natural land transformation is achieved for all scenarios. The largest reductions is observed for the scenario using the Norwegian electricity mix, as a result of its large share of hydro power.

Despite good results for three of the impact categories, electrification of the platform results in an increase in the marine ecotoxicity and terrestrial acidification potentials. For marine ecotoxicity, the European electricity mix scenario has a significantly higher contribution than the other scenarios, much due to thermal power generation in Europe. The terrestrial acidification potential is highest for scenario 1, followed by scenario 2. It has its third highest value for the original case and its lowest value for scenario 3. The contributions to terrestrial acidification are mainly due to thermal power production and the manufacturing of different chemicals.

As discussed in chapter 6.1 there are on-going discussions in Norway today on whether or not to electrify the Norwegian continental shelf. Non-profit environmental organisations such as Bellona and Zero argue that electrification of the shelf will result in substantial reductions in emissions of greenhouse gases from the Norwegian oil and gas industry. This is however provided that renewable energy sources are used to generate the power. Important assumptions are that offshore wind power and gas power plants with CCS are developed and implemented in the offshore power grid (Bellona, 2008, Lundberg and Kaski, 2011).

Those against electrification of the shelf argue that there is not enough power on the main land to supply the offshore industry. The mid-parts of Norway already experience power deficit certain parts of the year, resulting in the need to import power from neighbouring countries such as Sweden, Denmark and the Netherlands. This electricity is produced by nuclear power, gas and coal. What the opponents fear is that as more of the Norwegian power is transmitted offshore, more power from abroad has to be imported.

This imported power is typically generated by the use of fossil fuels, thus will the reduction in greenhouse gases from the offshore industry be eliminated by the increased emissions of greenhouse gases emitted onshore. This will also mean that less electricity produced by hydro power from Norway is exported to Europe and that coal power plants on the continent may have to replace this power (Tomasgard, 2011). This means that scenario 3 is not realistic at this point in time; one would probably not be able to supply the platform with power mainly produced by hydropower, and the associated environmental impacts would then rise.

The electrification of the shelf may also result in local impacts such as interventions in nature, increased transmission losses and network tariffs, reduced security of supply and power deficit on the main land. It is also extremely expensive and includes large interventions on the already existing platforms (Tomasgard, 2011).

Another argument against electrification of the Norwegian continental shelf is that the emission quotas saved in Norway can be bought somewhere else through the EU's Emissions Trading Scheme (ETS). This means that the increase in European emissions remains the same (NTB, 2012). This leads us to the question on whether we should focus on local or global measures to mitigate climate change.

The sensitivity analysis also shows that increasing the electricity requirement affects the impact categories where the operation phase is the main contributor, such as climate change, fossil depletion and ozone depletion. It can be seen that an increase in energy requirement result in a corresponding increase in the environmental impact potentials for these impact categories.

A reduction in the system's lifetime affects the impact categories where manufacturing of pumps and/or the mechanical pre-treatment phase are notable contributors. This includes metal depletion, freshwater ecotoxicity and marine ecotoxicity. Climate change, fossil depletion and ozone depletion on the other hand, are not affected as their main inputs change proportionally with a change in the system's lifetime.

7.1.1 Comparison of total impacts with results from previous studies

Previously, there have been conducted several LCA studies on desalination of water by the use of RO (Muñoz and Fernández-Alba, 2008, Rivera, 2009, Raluy et al., 2004). Common to these studies is that the desalination plants are located onshore and are used for purposes such as drinking water and irrigation in agriculture.

In their study, Muñoz and Fernández-Alba (2008) investigated whether or not, and to what extent, RO of brackish water results in reduced environmental impacts compared to desalination of seawater. The study was based on two different water production plants where one desalinated seawater and the other desalinated brackish water.

Table 7-1 shows the results from this thesis compared to the results from Muñoz and Fernández-Alba's LCA study (2008). The functional unit used in the previous studies is m^3 of desalinated water, and the results calculated in this thesis are therefore given per m^3 of desalinated water.

Table 7-1: A selection of environmental impacts from this thesis and from Muñoz and Fernández-Alba (2008) for 1 m^3 of desalinated water.

Characterisation factor	Results from this thesis	Muñoz and Fernández-Alba (2008)
Global warming potential	3.0123 kg CO_2 eq	1.9 kg CO_2 eq
Human toxicity potential	0.0705 kg 1,4-DCB eq	0.65 kg 1,4-DCB eq
Photochemical oxidant formation potential	0.0082 kg NMVOC eq	0.001 kg C_2H_4 eq
Acidification potential	0.0066 kg SO_2 eq	0.027 SO_2 eq

As can be seen from the table, the same impact categories are used, however different LCIA methods have been chosen. In this thesis, the ReCiPe method is chosen, while Muñoz and Fernández-Alba (2008) have chosen to use different methods for each impact category. This makes it difficult to compare the results as the different LCIA methods are based on different characterization models and category indicators. This can be seen for the impact category photochemical oxidant formation, where the reference compound in the ReCiPe method is

NMVOC equivalents, while Muñoz and Fernández-Alba (2008) use a method where C_2H_4 equivalents is the reference compound, making it difficult to compare them.

Toxicity is a weak spot in LCA due to large uncertainties and omissions regarding the effect on chemicals on human and on ecosystems (3R, 2013). This may lead to various results when assessing the toxicity of a product using different LCIA methodologies.

We are, however, more familiar with the GWP values of different substances, and it is therefore useful to compare these values, despite the different LCIA methodologies. The GWP is calculated in the impact assessment to be 3 kg of CO_2 equivalents per m^3 of desalinated water. The study of Muñoz and Fernández-Alba (2008) resulted in a GWP of 1.9 kg of CO_2 equivalents. This means that the GWP obtained in this thesis is about 60% higher. The reason for this is likely the choice of electricity mix as the energy requirements are the same; 4 kWh per m^3 of desalinated water.

In this thesis, the energy demand is modelled as electricity produced by gas turbines, as this will be the case on a platform, while Muñoz and Fernández-Alba (2008) have chosen to use the Spanish electricity consumption mix. This electricity mix consist of about 58% of energy from fossil fuels, 9% of energy from hydro power, 18% of energy from nuclear power and 12% of energy from other renewable energy sources. The rest of the electricity is imported from France and Portugal (Itten et al., 2012). This means that the electricity mix used in the study by Muñoz and Fernández-Alba (2008) emit less greenhouse gases through generation as 42% of the energy come from energy sources other than fossil fuels. In this thesis, the electricity is entirely generated by natural gas, thus emitting more substances that contribute to climate change.

It can also be seen that the acidification potential in this thesis is about 25% of the acidification potential found by Muñoz and Fernández-Alba (2008). This may be due to variations in NO_X and SO_2 emissions, but it could also have to do with the choice of LCIA method and/or with the choice of electricity generation.

Table 7-2 shows the GWP values from this thesis and from the studies of Raluy et al. (2004) and Rivera (2009).

Table 7-2: GWP values from this thesis, Raluy et al. (2004) and Rivera (2009) per m^3 desalinated water.

Reference	GWP in kg CO_2 eq	Energy source
This thesis	3.0123	Gas turbine, natural gas
Rivera (2009)	1.958	Spanish electricity mix
Raluy et al. (2004)	1.78	European electricity mix
	2.79	Steam cycle, natural gas
	2.13	Internal combustion engine, natural gas
	1.75	Combined cycle, natural gas

For all these studies the same energy requirement is used; 4 kWh per m^3 of desalinated water. It can be seen that the GWP obtained in this thesis is about 54% higher than the GWP from (Rivera, 2009). Also for this case it is probable that the difference is caused by different energy inputs into the models. Rivera (2009) have chosen to use a Spanish electricity mix. Equivalent to the case described in Muñoz and Fernández-Alba (2008), the electricity mix consist of 58% fossil fuels, 18% nuclear power, 12% renewable sources and 9% hydro power. In this thesis, the electricity is generated by the use of natural gas which is a fossil fuel. This means that it will emit more substances that contribute to climate change, than the case described in (Rivera, 2009) where 42% of the energy does not originate from fossil fuels.

Raluy et al. (2004) conducts an LCA of desalination processes integrated with different energy production systems. For RO, three different cases have been considered. The first is a conventional steam cycle with an electrical efficiency of 34%, the second is an internal combustion engine with an electrical efficiency of 45% and the third is a combined cycle with an electrical efficiency of 55%. All the energy systems use natural gas as fuel. The results are compared to an RO unit supplied by an average European electricity mix. This study is very interesting as it corresponds to the model in this thesis in that natural gas is used as fuel for power generation.

The resulting GWP for RO supplied by the European electricity mix is 1.78 kg of CO_2 equivalents (Raluy et al., 2004), meaning that the GWP obtained in this thesis is 69% higher. As the European electricity mix is made up of 43.3% thermal power, 40.3% nuclear power and 16.4% hydro power (Raluy et al., 2004) it will emit less greenhouse gases than the system defined in this thesis.

The case with the conventional steam cycle use natural gas as an energy source, making it very similar to the system in this thesis where gas turbines burn natural gas to generate power. The GWP obtained from Raluy et al. (2004) for this case is 2.79 kg of CO_2 equivalents, 7.4% lower than the GWP calculated in this thesis. This difference is probably a result of using different LCIA frameworks and inputs. E. g. Raluy et al. (2004) assumes 8000 operating hours per year, but in this thesis the assumption is 8760 operating hours.

For the second and third case described by Raluy et al. (2004) the GWP values are 2.13 and 1.75 kg of CO_2 equivalents, respectively. This decline may be explained by the increase in system efficiency.

Grandell et al. (2011) look at the profitability of Norwegian oil and gas fields from 1991 to 2008 by using the profitability measure Energy Return on Investment (EROI). They state that on a global scale, few resources can show such favourable EROI values as can be found in the Norwegian oil and gas industry. This profitability is however declining as the fields become older, and Grandell et al. (2011) anticipate that new energy-intensive production techniques will become more important in the future. LSW is such a technique and there is already a planned pilot project were this technology will be implemented in oil production offshore

(Hegre, 2008). It will therefore be important to have a basic understanding of how these projects may impact the environment, and the results from this thesis contribute to this.

To the author's knowledge, there has not been published any LCAs on LSW for EOR. This thesis has identified and assessed life cycle environmental impacts associated with desalinated water for EOR, by means of RO. By doing this, the data foundation regarding environmental impacts for EOR methods have been improved and extended, thus making it possible to anticipate environmental loads associated with future LSW for EOR projects. This thesis also includes several environmental impacts that have not been included in previous LCA studies on RO, resulting in a more detailed knowledge on the environmental impacts associated with this technology. It was also decided to include an alternative scenario where the platform was electrified and connected to main land in Norway by HVDC cables. There has not been published any LCA studies which assesses the environmental impacts from LSW on an electrified platform, and this thesis therefore contributes with new knowledge in this area as well.

7.1.2 Comparison with other fuel production methods

The Ecoinvent database includes information on the environmental impacts associated with the recovery of crude oil, as described in chapter 2.5. By performing an LCIA with this data it is possible to calculate the environmental impacts associated with conventional oil production offshore; oil production without EOR methods. As Norway exports 90% of all the oil produced in the North Sea (Hagland, 2000), it is chosen to use the inventory of oil production offshore for Norway in the LCIA. Applying the ReCiPe framework result in a GWP of 54.82 kg of CO_2 equivalents per Sm^3 of crude oil recovered.

The GWP associated with the recovery of 1 Sm^3 of crude oil by LSW is 150.62 kg of CO_2 equivalents, nearly three times higher than for conventional oil production. Emissions of greenhouse gases from the oil industry are mainly caused by the burning of natural gas in turbines for power generation and for the operation of pumps and compressors. Flaring is also an important source of emissions (Miljøstatus, 2012). Emissions of greenhouse gases from the low-salinity system are mainly associated with gas turbines fuelled by natural gas. Nevertheless, the increase in GWP value for LSW is expected. LSW is an *enhanced* oil recovery process, which means that more energy is required for recovering crude oil compared to primary and secondary oil recovery.

As oil prices are high and the demand for energy is growing in North America, the interest in unconventional fossil reserves increase. The oil sands in Canada are such reserves and they are second only to Saudi Arabian oil sands in size. There are two main methods for recovery of oil sands; surface mining and in-situ extraction. For reservoirs shallower than 75 m, surface mining is employed. In-situ extraction is applied for deeper reservoirs. The bitumen is heated or diluted and pumped to the surface where it is upgraded, i.e. made lighter and sweeter. This upgraded bitumen is called synthetic crude oil (SCO) and it is this substance that is refined into gasoline and other fuels (Alex et al., 2009).

The oil sand industry is growing as production and investments increase. It is estimated that it could supply 16% of the oil demand in North America by 2030, with a production of 5 million barrels per day (Alex et al., 2009). Despite the economic benefits gained from this technology, much controversy surrounds it due to its impacts on the environment, and especially its high greenhouse gas emissions. Alex et al. (2009) review thirteen studies that estimate greenhouse gas emissions from oil sands operations. The studies are divided into two categories: “well-to-refinery entrance gate” (WTR) and “well-to-wheel” (WTW). The WTR studies focus on the bitumen extraction and the SCO production, and these results are used here, as this process chain is the most similar to the process chain described in this thesis.

Each study reviewed by Alex et al. (2009) used a different combination of fuels and pathways, research methods, levels of detail, purposes and assumptions. The system boundaries also varied from study to study. Alex et al. (2009) therefore decided to set a boundary in order to compare the different WTR results, which included the operation of the projects, transportation during extraction and the production of energy inputs, such as fossil fuels and electricity. The results for surface mining and upgrading were reported to be in the range of 62 to 164 kg of CO_2 equivalents per barrel, which is 390 to 1031.5 kg of CO_2 equivalents per Sm^3 of SCO recovered. For in-situ extraction and upgrading the results varied from 99 to 176 kg of CO_2 equivalents per barrel or 622.7 to 1107 kg of CO_2 equivalents per Sm^3 SCO. These results have a wide range and Alex et al. (2009) emphasise that even if some of the variation may be the result of different projects using different technologies, one cannot definitely conclude that some projects contribute more to climate change than others based on these studies.

Table 7-3 gives the GWP values associated with conventional oil recovery, LSW and production of oil from oil sands, given for 1 Sm^3 of crude oil recovered.

Table 7-3: GWP values for conventional oil recovery, LSW and oil sand production for 1 Sm^3 of crude oil recovered.

Reference	**GWP in kg CO_2 eq**	**Technology**
Ecoinvent (2013)	54.82	Conventional oil production
This thesis	150.62	EOR, LSW
Alex et al. (2009)		Canadian oil sands:
	390-1031.5	- Surface mining and upgrading
	622.7-1107	- In-situ extraction and upgrading

It can be seen from Table 7-3 that even though the GWP for oil sands production has a wide range, it is significantly higher than the GWP values for conventional oil recovery and for LSW. The GWP for the surface mining process is between 2.6 and 6.9 times larger than the GWP for LSW. For in-situ extraction the GWP is between 4.1 and 7.4 times larger. For the conventional oil recovery modelled in Ecoinvent (2013) the difference is even bigger; 7.1-18.8 and 11.3-20.2 times larger for surface mining and in-situ extraction, respectively.

In addition to the high emissions of greenhouse gases, oil production from oil sands has also been reported to release toxins into the environment. Kelly et al. (2010) collected samples from around the Athabasca Delta, Lake Athabasca and the Athabasca River and its tributaries in Canada. They found thirteen toxins classified as so-called *priority pollutants* by the US Environmental Protection Agency, among them mercury, cadmium, copper, lead and zinc.

From this it can be concluded that the contributions to climate change for LSW is significantly lower than the contributions from oil sands production, regardless of the wide range of the results. This means that even though the greenhouse gas emissions from conventional oil production is lower than the emissions from LSW, it is still recommended to implement LSW rather than producing oil from unconventional fossil reserves such as oil sands. The possible contribution from oil sands production to the different toxicity categories strengthens this conclusion.

7.2 Data quality and uncertainty

There are several sources of uncertainty in an LCA. One is that the data in the LCI databases may be inaccurate or out of date due to advancements of technology or changes in the environment. Other uncertainties are related to poor data quality and non-transparent assumptions in the inventories. There have been discussions on whether or not the data that is collected in the inventory analysis can be related to actual environmental impacts, due to the simplifications made in the data collection phase. The inventory contains quantified inputs of materials and energy and outputs of emissions and waste, but information on spatial and temporal dimensions, dose-response relationships and thresholds are missing. This information is left out due to the vast amount of data that would be required in order to include these relationships in the inventories (Ross et al., 2002).

Assumptions made in the model are also sources of uncertainties. It would be too time-consuming to collect data for every single input and environmental impact for all the processes of a product’s lifetime and in many cases it would not be possible, thus some assumptions have to be made. These are often based on a subjective system boundary, which will result in uncertainties that change as the system boundary changes.

Characterisation models are also a source of uncertainty as our knowledge on the environmental mechanisms involved in issues such as climate change and human toxicity are incomplete. In ReCiPe there are three different perspectives that include different uncertainties and decisions on system boundary. In this thesis, the hierarchist perspective is chosen and this is based on the general consensus with regards to policy principles and time frame. Another source of uncertainties in the ReCiPe method is that there are some links between midpoint and endpoint categories that have not been established (Goedkoop et al., 2013).

In this thesis, all the data for the construction of the RO unit, the pumps, mechanical and chemical treatment, operation and disposal are found in Rivera (2009). The data is based on a real RO desalination plant in Chile and it can therefore be assumed that these numbers are reasonable. There is, however, one big difference in the desalination facility described by Rivera (2009) and the desalination facility described in this thesis; the facility in this thesis is located offshore on a platform, while the other is located onshore in Chile. In this thesis it is assumed that the RO plant is the same as the ones used onshore, only it is implemented inside a platform in the North Sea. There is uncertainty regarding the size of the offshore plant, is the described plant too big to fit or should it be even bigger? This would again affect the RO plant's desalination capacity. However, the recovery factor of the RO plant is given for a planned LSW pilot project by Statoil, which means that this number is associated with little uncertainty. The implication of this is that the environmental impacts are the same per m^3 of desalinated water, independent of the plant's capacity.

The LCIA shows that the operation phase and chemical treatment phase contribute the most to environmental impacts. The amounts of chemicals required was described in detail by Rivera (2009). They are obtained from a real RO plant and it is therefore conceivable that these are valid. The energy requirement in the operation phase is also obtained from a real RO plant, but it was chosen to do a sensitivity analysis on this in chapter 6.

Most of the LCA studies found on RO use the inventory data from Rivera (2009) as a basis (Raluy et al., 2005c, Raluy et al., 2004, Raluy et al., 2005b, Raluy et al., 2005a, Raluy et al., 2006, Muñoz and Fernández-Alba, 2008). This could mean that the data seems so comprehensive and reliable that other researchers have deemed it unnecessary to collect new inventory data, but if there happen to be larger uncertainties in the data foundation it will reflect on the results of these studies, making them less reliable.

The CIP system was modelled based on crude assumptions, but the LCIA showed that the contribution from this component was negligible for all impact categories. The transport and disposal was based on the European scenario in the Ecoinvent database, but had little to no impact according to the LCIA.

The assumption with the biggest uncertainty in this thesis is the amount of desalinated water required per Sm^3 of recovered crude oil. The water-to-oil ratio required can according to Thakur and Satter (1998) range from 5.6-70.8, depending on a number of variables related to the properties of the petroleum reservoir and its content. Other sources of information on this issue were not found. It was decided to use a large ratio; 50 m^3 of desalinated water per Sm^3 of crude oil recovered. This is a conservative approach, in many cases the environmental impacts will be lower than what is modelled here, but in a few cases they may be higher. However, this does not affect the environmental impacts associated with 1 m^3 of desalinated water. If the water-to-oil ratio is halved, then the environmental impacts are halved. Likewise, if the water-to-oil ratio is increased by 20%, so are the environmental impacts. It was therefore decided to not do a sensitivity analysis on this assumption.

The environmental impacts associated with desalination of water are considered to be reliable, but more information is required in order to assess the actual environmental impacts associated with the recovery of crude oil from a specific LSW project. In order to do this, one has to know the specific water-to-oil ratio required in each case. One also needs to quantify raw materials, energy inputs, emissions and wastes associated with a project. As all petroleum reservoirs have different properties and characteristics, it is not possible to create a generic model that fits all.

8 Conclusion and Further Work

The LCA of desalinated water for EOR shows that the operation phase is the largest contributor to environmental impacts due to the generation of power by natural gas-driven turbines on the platform. The chemical treatment process is also a major contributor to environmental impacts, caused by energy inputs and wastes from chemical manufacturing. The highest impacts after normalisation are found for natural land transformation, climate change, marine ecotoxicity, fossil depletion and terrestrial acidification, indicating that these are the most important impact categories.

From the sensitivity analysis it can be seen that an electrification of the platform will result in substantial environmental benefits in terms of reduced emissions of greenhouse gases, depending on the choice of electricity mix. However, there are issues that need to be addressed before this should be implemented on a large scale, such as implementation of CCS in onshore power plants, the use of offshore wind power and the influence of ETS. The big question is whether one should focus on local or global measures to mitigate climate change.

Compared to the existing literature, the results from this thesis are as anticipated; a somewhat higher contribution to climate change due to the use of gas turbines for power generation, whereas other LCA studies use electricity mixes with a larger fraction of non-fossil fuel power generation.

The results from this thesis were also compared to the environmental impacts associated with conventional oil recovery and oil sands production. It is found that the greenhouse gas emissions from LSW are about three times higher than the emissions from conventional oil recovery. For oil sands production the results are divergent, but nevertheless the emissions are substantially higher than for LSW. It is recommended to implement EOR methods such as LSW, rather than producing oil from unconventional fossil reserves such as oil sands.

This thesis is based on data from only a few sources. The defined system is generic and in some cases crude assumptions is made, but from the results it can be seen that this had little impact. The environmental impacts associated with *desalination of water* are deemed reliable. However, great uncertainty is linked to the required amount of water per Sm^3 of recovered crude oil, as a result of little to no data on this issue. This lead to uncertainty regarding the calculated environmental impacts associated with *the recovery of crude oil*.

The results from this thesis create a basis for future research, as it is an analysis based on a generic case. In order to calculate the environmental impacts from one specific oil field or LSW project, it is necessary to quantify material and energy inputs, emissions and wastes, as well as the exact water-to-oil ratio which is required. In order to achieve this, extensive research needs to be conducted in order to map and identify key parameters and properties of the petroleum reservoir in question.

It could also be useful to conduct an LCA on different desalination technologies for LSW in order to assess and compare the environmental impacts associated with each of these technologies. Challenges for the future also involve determining the factors that influence crude oil recovery rates in relation to LSW. It is important to understand why the effect of LSW arises and under which circumstances. Extensive research will be needed in order to address these issues and studies should also be conducted on different types of reservoir rocks in order to map the potential for LSW in different reservoirs.

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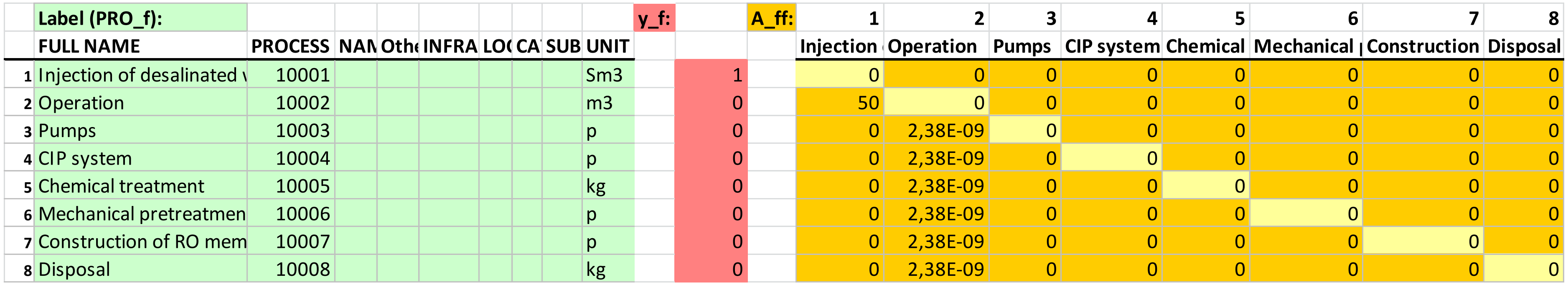
Appendix A: Complete inventory

Figure A-1 gives the foreground matrix and external demand as it is created for use in Arda.



	Label (PRO_f):									y_f:		A_ff:	1	2	3	4	5	6	7	8
	FULL NAME	PROCESS	NAI	Oth	INFRA	LO(	CA	SUB	UNIT				Injection	Operation	Pumps	CIP system	Chemical	Mechanical	Construction	Disposal
1	Injection of desalinated v	10001							Sm3		1		0	0	0	0	0	0	0	0
2	Operation	10002							m3		0		50	0	0	0	0	0	0	0
3	Pumps	10003							p		0		0	2,38E-09	0	0	0	0	0	0
4	CIP system	10004							p		0		0	2,38E-09	0	0	0	0	0	0
5	Chemical treatment	10005							kg		0		0	2,38E-09	0	0	0	0	0	0
6	Mechanical pretreatmen	10006							p		0		0	2,38E-09	0	0	0	0	0	0
7	Construction of RO mem	10007							p		0		0	2,38E-09	0	0	0	0	0	0
8	Disposal	10008							kg		0		0	2,38E-09	0	0	0	0	0	0

Figure A-1: Foreground matrix, A_{ff}, and external demand, y_f.

Table A-1: Complete inventory list for the recovery of 1 Sm^3 of crude oil by LSW.

Process	Value	Unit
Operation		
electricity, natural gas, at turbine, 10MW/ GLO/ kWh	4	kWh
Pumps		
chromium steel 18/8, at plant/ RER/ kg	230 000	kg
transport, freight, rail/ RER/ tkm	46 000	tkm
transport, lorry >16t, fleet average/ RER/ tkm	23 000	tkm
transport, barge/ RER/ tkm	23 000	tkm
CIP system		
polyethylene, LLDPE, granulate, at plant/ RER/ kg	689,40	kg
polyvinylchloride, at regional storage/ RER/ kg	172,35	kg
steel, low-alloyed, at plant/ RER/ kg	2 585,25	kg
transport, freight, rail/ RER/ tkm	689,40	tkm
transport, lorry >16t, fleet average/ RER/ tkm	344,70	tkm
transport, barge/ RER/ tkm	344,70	tkm
Chemical treatment		
sodium hypochlorite, 15% in H2O, at plant/ RER/ kg	1 142 650,75	kg
sodium sulphate, from Mannheim process, at plant/ RER/ kg	13 991 636,52	kg
aluminium sulphate, powder, at plant/ RER/ kg	34 979 090,63	kg
lime, hydrated, loose, at plant/ CH/ kg	1 049 375	kg
phosphoric acid, industrial grade, 85% in H2O, at plant/ RER/ kg	932 775,75	kg
chlorine, gaseous, membrane cell, at plant/ RER/ kg	1 148,85	kg
transport, lorry >16t, fleet average/ RER/ tkm	7 152 950,54	tkm
transport, freight, rail/ RER/ tkm	42 917 703,21	tkm
transport, barge/ RER/ tkm	7 152 950,54	tkm
Mechanical pretreatment		
steel, low-alloyed, at plant/ RER/ kg	1 217 500	kg
silica sand, at plant/ DE/ kg	1 000 000	kg
transport, freight, rail/ RER/ tkm	243 500	tkm
transport, lorry >16t, fleet average/ RER/ tkm	71 750	tkm
transport, barge/ RER/ tkm	221 750	tkm
Construction of RO membrane package		
cast iron, at plant/ RER/ kg	10 000	kg
steel, low-alloyed, at plant/ RER/ kg	100 000	kg
concrete block, at plant/ DE/ kg	2 750 000	kg
epoxy resin, liquid, at plant/ RER/ kg	22 960	kg
nylon 66, glass-filled, at plant/ RER/ kg	459 200	kg
transport, freight, rail/ RER/ tkm	118 432	tkm
transport, lorry >16t, fleet average/ RER/ tkm	196 716	tkm
transport, barge/ RER/ tkm	334 216	tkm

Disposal		
disposal, steel, 0% water, to inert material landfill/ CH/ kg	1 560 085,25	kg
disposal, plastics, mixture, 15.3% water, to sanitary landfill/ CH/ kg	482 160	kg
disposal, building, concrete gravel, to final disposal/ CH/ kg	1 000 000	kg
disposal, concrete, 5% water, to inert material landfill/ CH/ kg	2 750 000	kg
disposal, polyethylene, 0.4% water, to sanitary landfill/ CH/ kg	689,40	kg
disposal, polyvinylchloride, 0.2% water, to sanitary landfill/ CH/ kg	172,35	kg
transport, lorry >16t, fleet average/ RER/ tkm	119 481,50	tkm
transport, barge/ RER/ tkm	579 310,70	tkm

Table A-2: The stressor associated with the recovery of 1 Sm^3 of crude oil by LSW.

Stressor	Value	Unit
Operation		
Water, salt, ocean/ resource/ in water	2,222	m^3

Appendix B: Chemicals

The total amount of chemicals used in the chemical treatment phase is calculated as follows:

Sulphuric acid (20 ppm, 96% purity):

$$102\,222\ \mathrm{m^3/d} \cdot \left(\frac{20\times10^{-3}\ \mathrm{kg/m^3}}{0.96} \right) \cdot 365\ \mathrm{d/y} \cdot 25\ \mathrm{y} = 19\,432\,828.13\ \mathrm{kg} \tag{1}$$

Sodium hypochlorite (1 ppm for pretreatment, 0.5 ppm for post-treatment):

$$\begin{aligned}&\left(102\,222\ \mathrm{m^3/d} \cdot 1\cdot10^{-3}\ \mathrm{kg/m^3} + 46\,000\ \mathrm{m^3/d} \cdot 0.5\cdot10^{-3}\ \mathrm{kg/m^3}\right)\cdot\\ &365\ \mathrm{d/y} \cdot 25\ \mathrm{y} = 1\,142\,650.75\ \mathrm{kg}\end{aligned} \tag{2}$$

Sodium bisulphate (15 ppm):

$$102\,222\ \mathrm{m^3/d} \cdot 15\cdot10^{-3}\ \mathrm{kg/m^3} \cdot 365\ \mathrm{d/y} \cdot 25\ \mathrm{y} = 13\,991\,636.25\ \mathrm{kg} \tag{3}$$

Iron(III) chloride (15 ppm, 40% purity):

$$102\,222\ \mathrm{m^3/d} \cdot \left(\frac{15\cdot10^{-3}\ \mathrm{kg/m^3}}{0.4} \right) \cdot 365\ \mathrm{d/y} \cdot 25\ \mathrm{y} = 34\,979\,090.63\ \mathrm{kg} \tag{4}$$

Calcium sulphate (0.5 ppm, 20% purity):

$$46\,000\ m^3/d \cdot \left(\frac{0.5\cdot10^{-3}\ kg/m^3}{0.4} \right) \cdot 365\ \mathrm{d/y} \cdot 25\ \mathrm{y} = 1\,049\,375\ \mathrm{kg} \tag{5}$$

Sodium hexametaphosphate (1 ppm):

$$102\,222\ \mathrm{m^3/d} \cdot 1 \cdot 10^{-3}\ \mathrm{kg/m^3} \cdot 365\ \mathrm{d/y} \cdot 25\ \mathrm{y} = 932775.75\ \mathrm{kg} \qquad (6)$$

Chlorine gas:
$3 \cdot 10^{-6}$ kg chlorine/m^3 desalinated water

$$3 \cdot 10^{-6}\ \mathrm{kg/m^3} \cdot 46\,000\ \mathrm{m^3/d} \cdot 365\ \mathrm{d/y} \cdot 25\ \mathrm{y} = 1\,148.85\ \mathrm{kg} \qquad (7)$$

Appendix C: Impact assessment

Table C-1: Environmental impacts distributed across the foreground processes for 1 Sm^3 of crude oil recovered by LSW.

Impact category	Injection of desalinated water	Operation	Pumps	CIP system	Chemical treatment	Mechanical pre-treatment	Construction of RO package system	Disposal	Unit
Agricultural land occupation	0	2,25E-02	3,17E-03	1,33E-05	9,85E-02	6,27E-03	2,66E-03	1,96E-04	m^2a
Climate change	0	1,46E+02	1,24E-01	7,32E-04	3,85E+00	2,55E-01	4,70E-01	1,56E-02	kg CO_2 eq
Fossil depletion	0	6,12E+01	3,71E-02	3,37E-04	1,20E+00	8,33E-02	1,51E-01	5,27E-03	kg oil eq
Freshwater ecotoxicity	0	1,53E-02	9,36E-03	1,53E-05	6,24E-02	7,17E-03	1,21E-03	1,06E-03	kg 1,4-DCB eq
Freshwater eutrophication	0	6,31E-04	5,94E-05	3,42E-07	3,08E-03	1,59E-04	2,88E-05	1,06E-06	kg P eq
Human toxicity	0	8,24E-01	8,24E-02	4,38E-04	2,42E+00	2,04E-01	3,34E-02	2,00E-02	kg 1,4-DCB eq
Ionizing radiation	0	2,70E+01	2,56E-02	9,21E-05	1,54E+00	4,33E-02	9,80E-03	1,08E-03	kg U^{235} eq
Marine ecotoxicity	0	4,99E-02	9,77E-03	1,58E-05	6,50E-02	7,41E-03	1,03E-03	8,96E-04	kg 1,4-DCB eq
Marine eutrophication	0	4,28E-02	9,11E-05	4,49E-07	3,84E-03	1,88E-04	1,33E-03	4,25E-04	kg N eq
Metal depletion	0	4,19E-01	2,08E-01	7,99E-04	3,84E-01	3,76E-01	3,62E-02	4,21E-04	kg Fe eq
Natural land transformation	0	2,93E-02	1,26E-05	7,25E-08	9,37E-04	3,55E-05	1,16E-05	-2,4E-05	m^2
Ozone depletion	0	1,84E-05	6,60E-09	2,82E-11	3,43E-07	1,16E-08	3,47E-09	2,28E-09	kg CFC-11 eq

Particulate matter formation	0	8,50E-02	6,00E-04	2,15E-06	2,38E-02	9,12E-04	6,31E-04	4,23E-05	kg PM-10 eq
Photochemical oxidant formation	0	3,83E-01	4,24E-04	2,66E-06	2,15E-02	9,01E-04	1,25E-03	1,26E-04	kg NMVOC eq
Terrestrial acidification	0	2,35E-01	5,93E-04	2,51E-06	9,16E-02	9,33E-04	1,54E-03	7,34E-05	kg SO_2 eq
Terrestrial ecotoxicity	0	2,39E-03	2,14E-05	7,37E-08	5,89E-04	3,45E-05	1,01E-05	3,45E-06	kg 1,4-DCB eq
Urban land occupation	0	4,89E-02	1,90E-03	5,71E-06	7,44E-02	2,730E-03	7,81E-04	1,04E-03	m^2a
Water depletion	0	4,36E-03	4,37E-04	1,64E-06	1,88E-02	7,25E-04	1,90E-04	1,54E-05	m^3

Table C-2: Stressors contributing to climate change for 1 Sm^3 of crude oil recovered by LSW.

Stressors contributing to climate change	
Stressor	**Value [CO_2 eq]**
Carbon dioxide, fossil/ air/ high population density	134,13
Methane, fossil/ air/ low population density	8,01
Carbon dioxide, fossil/ air/ low population density	6,24
Carbon dioxide, fossil/ air/ unspecified	1,05
Other	1,13

Table C-3: Stressors contributing to fossil depletion for 1 Sm^3 of crude oil recovered by LSW.

Stressors contributing to fossil depletion	
Stressor	**Value [oil eq]**
Gas, natural, in ground/ resource/ in ground	61,11
Oil, crude, in ground/ resource/ in ground	0,90
Coal, hard, unspecified, in ground/ resource/ in ground	0,43
Other	0,21

Table C-4: Stressors contributing to marine ecotoxicity for 1 Sm^3 of crude oil recovered by LSW.

Stressors contributing to marine ecotoxicity	
Stressor	**Value [kg 1,4-DCB eq]**
Nickel, ion/ water/ ground-, long-term	4,12E-02
Zinc, ion/ water/ ocean	3,47E-02
Vanadium, ion/ water/ ground-, long-term	2,60E-02
Manganese/ water/ ground-, long-term	1,04E-02
Cobalt/ water/ ground-, long-term	3,71E-03
Zinc, ion/ water/ ground-, long-term	3,71E-03
Copper/ air/ low population density	2,28E-03
Beryllium/ water/ ground-, long-term	2,14E-03
Selenium/ water/ ground-, long-term	1,19E-03
Copper/ air/ high population density	1,02E-03
Vanadium/ air/ high population density	8,20E-04
Other	6,79E-03

Table C-5: Stressors contributing to natural land transformation for 1 Sm^3 of crude oil recovered by LSW.

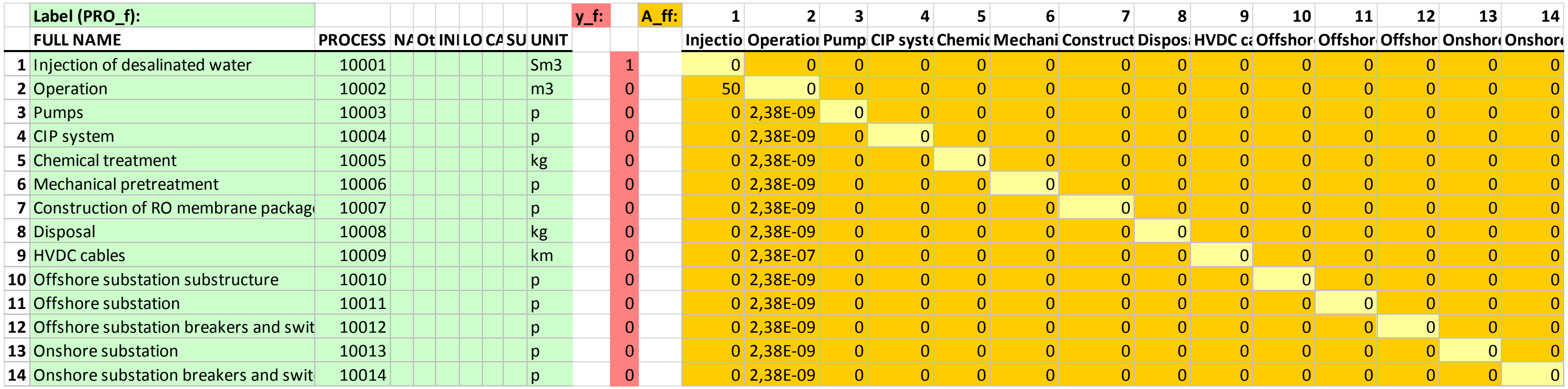
Stressors contributing to natural land transformation	
Stressor	**Value [m^2]**
Transformation, from sea and ocean/ resource/ land	1,67E-02
Transformation, from forest/ resource/ land	1,33E-02
Transformation, from forest, extensive/ resource/ land	1,07E-03
Transformation, from unknown/ resource/ land	4,18E-04
Other	-1,32E-03

Table C-6: Stressors contributing to terrestrial acidification for 1 Sm^3 of crude oil recovered by LSW.

Stressors contributing to terrestrial acidification	
Stressor	**Value [kg SO_2]**
Nitrogen oxides/ air/ high population density	1,78E-01
Sulphur dioxide/ air/ high population density	7,94E-02
Sulphur dioxide/ air/ low population density	5,57E-02
Nitrogen oxides/ air/ low population density	1,09E-02
Nitrogen oxides/ air/ unspecified	3,79E-03
Other	1,23E-03

Appendix D: Sensitivity analysis

Figure D-1 gives the foreground matrix and external demand as it is given for Arda for the alternative scenario where the platform is supplied by power produced onshore. The values are the same for all three scenarios.



	Label (PRO_f):									y_f:		A_ff:	1	2	3	4	5	6	7	8	9	10	11	12	13	14
	FULL NAME	PROCESS	NA	Ot	INI	LO	CA	SU	UNIT				Injectio	Operatio	Pump	CIP syst	Chemic	Mechani	Construct	Dispos	HVDC ca	Offshor	Offshor	Offshor	Onshor	Onshor
1	Injection of desalinated water	10001							Sm3		1		0	0	0	0	0	0	0	0	0	0	0	0	0	0
2	Operation	10002							m3		0		50	0	0	0	0	0	0	0	0	0	0	0	0	0
3	Pumps	10003							p		0		0	2,38E-09	0	0	0	0	0	0	0	0	0	0	0	0
4	CIP system	10004							p		0		0	2,38E-09	0	0	0	0	0	0	0	0	0	0	0	0
5	Chemical treatment	10005							kg		0		0	2,38E-09	0	0	0	0	0	0	0	0	0	0	0	0
6	Mechanical pretreatment	10006							p		0		0	2,38E-09	0	0	0	0	0	0	0	0	0	0	0	0
7	Construction of RO membrane packag	10007							p		0		0	2,38E-09	0	0	0	0	0	0	0	0	0	0	0	0
8	Disposal	10008							kg		0		0	2,38E-09	0	0	0	0	0	0	0	0	0	0	0	0
9	HVDC cables	10009							km		0		0	2,38E-07	0	0	0	0	0	0	0	0	0	0	0	0
10	Offshore substation substructure	10010							p		0		0	2,38E-09	0	0	0	0	0	0	0	0	0	0	0	0
11	Offshore substation	10011							p		0		0	2,38E-09	0	0	0	0	0	0	0	0	0	0	0	0
12	Offshore substation breakers and swit	10012							p		0		0	2,38E-09	0	0	0	0	0	0	0	0	0	0	0	0
13	Onshore substation	10013							p		0		0	2,38E-09	0	0	0	0	0	0	0	0	0	0	0	0
14	Onshore substation breakers and swit	10014							p		0		0	2,38E-09	0	0	0	0	0	0	0	0	0	0	0	0

Figure D-1: Foreground matrix, A_{ff}, and external demand, y_f, for the alternative scenario where the platform is supplied by power produced onshore

Table D-1: Complete inventory for the *additional* processes required for electrification of the platform. The inventory lists in Table A-1 and Table A-2 were also included for the three scenarios.

Process	Value	Unit
HVDC cables		
electricity, production mix RER/ RER/ kWh	86 469	kWh
copper, at regional storage/ RER/ kg	13 000	kg
lead, at regional storage/ RER/ kg	23 000	kg
steel, low-alloyed, at plant/ RER/ kg	65 480	kg
zinc coating, pieces/ RER/ m2	512	m^2
natural gas, burned in industrial furnace >100kW/ RER/ MJ	59 293	MJ
kraft paper, unbleached, at plant/ RER/ kg	5 500	kg
polypropylene, granulate, at plant/ RER/ kg	3 000	kg
transport, lorry >32t, EURO5/ RER/ tkm	31 000	tkm
transport, transoceanic freight ship/ OCE/ tkm	73 100	tkm
transport, barge/ RER/ tkm	2 680 000	tkm
Offshore substation substructure		
gravel, unspecified, at mine/ CH/ kg	5 151 000	kg
concrete, normal, at plant/ CH/ m3	2 425	m3
reinforcing steel, at plant/ RER/ kg	560 000	kg
steel, low-alloyed, at plant/ RER/ kg	630 000	kg
transport, lorry >32t, EURO5/ RER/ tkm	2 432 200	tkm
transport, barge/ RER/ tkm	22 600	tkm
transport, transoceanic freight ship/ OCE/ tkm	129 000	tkm
Offshore substation		
lubricating oil, at plant/ RER/ kg	85 390	kg
electricity, production mix RER/ RER/ kWh	406 621	kWh
aluminium, production mix, cast alloy, at plant/ RER/ kg	2 310	kg
copper, at regional storage/ RER/ kg	54 162	kg
steel, low-alloyed, at plant/ RER/ kg	216 447	kg
natural gas, burned in industrial furnace >100kW/ RER/ MJ	406 621	MJ
alkyd paint, white, 60% in H2O, at plant/ RER/ kg	2 982	kg
kraft paper, unbleached, at plant/ RER/ kg	8 810	kg
epoxy resin insulator (Al2O3), at plant/ RER/ kg	219	kg
transport, lorry >32t, EURO5/ RER/ tkm	78 849	tkm
sanitary ceramics, at regional storage/ CH/ kg	45,18	kg
sawn timber, softwood, planed, air dried, at plant/ RER/ m3	3 592	m^3

Offshore substation breakers and switchgear

sulphur hexafluoride, liquid, at plant/ RER/ kg	534	kg
electricity, production mix RER/ RER/ kWh	7 578	kWh
aluminium, production mix, at plant/ RER/ kg	9 746	kg
cast iron, at plant/ RER/ kg	108	kg
chromium steel 18/8, at plant/ RER/ kg	435	kg
copper, at regional storage/ RER/ kg	1 146	kg
nickel, 99.5%, at plant/ GLO/ kg	3	kg
silver, at regional storage/ RER/ kg	1	kg
steel, low-alloyed, at plant/ RER/ kg	2 885	kg
natural gas, burned in industrial furnace >100kW/ RER/ MJ	7 578	MJ
polyester resin, unsaturated, at plant/ RER/ kg	80	kg
kraft paper, unbleached, at plant/ RER/ kg	10	kg
epoxy resin insulator (Al2O3), at plant/ RER/ kg	1 096	kg
polycarbonate, at plant/ RER/ kg	5	kg
polyethylene, HDPE, granulate, at plant/ RER/ kg	22	kg
synthetic rubber, at plant/ RER/ kg	65	kg
transport, lorry >32t, EURO5/ RER/ tkm	3 804	tkm
sawn timber, softwood, planed, air dried, at plant/ RER/ m3	2,69	m3
sanitary ceramics, at regional storage/ CH/ kg	1 500	kg
transport, barge/ RER/ tkm	41 300	tkm
transport, transoceanic freight ship/ OCE/ tkm	180 000	tkm

Onshore substation

lubricating oil, at plant/ RER/ kg	85 390	kg
electricity, production mix RER/ RER/ kWh	406 621	kWh
aluminium, production mix, cast alloy, at plant/ RER/ kg	2 310	kg
copper, at regional storage/ RER/ kg	54 162	kg
steel, low-alloyed, at plant/ RER/ kg	216 447	kg
natural gas, burned in industrial furnace >100kW/ RER/ MJ	406 621	MJ
alkyd paint, white, 60% in H2O, at plant/ RER/ kg	2 982	kg
kraft paper, unbleached, at plant/ RER/ kg	8 810	kg
epoxy resin insulator (Al2O3), at plant/ RER/ kg	219	kg
transport, lorry >32t, EURO5/ RER/ tkm	78 849	tkm
sanitary ceramics, at regional storage/ CH/ kg	45,18	kg
sawn timber, softwood, planed, air dried, at plant/ RER/ m3	3 592	m^3

Onshore substation breakers and switchgear		
sulphur hexafluoride, liquid, at plant/ RER/ kg	534	kg
electricity, production mix RER/ RER/ kWh	7 578	kWh
aluminium, production mix, at plant/ RER/ kg	9 746	kg
cast iron, at plant/ RER/ kg	108	kg
chromium steel 18/8, at plant/ RER/ kg	435	kg
copper, at regional storage/ RER/ kg	1 146	kg
nickel, 99.5%, at plant/ GLO/ kg	3	kg
silver, at regional storage/ RER/ kg	1	kg
steel, low-alloyed, at plant/ RER/ kg	2 885	kg
natural gas, burned in industrial furnace >100kW/ RER/ MJ	7 578	MJ
polyester resin, unsaturated, at plant/ RER/ kg	80	kg
kraft paper, unbleached, at plant/ RER/ kg	10	kg
epoxy resin insulator (Al2O3), at plant/ RER/ kg	1 096	kg
polycarbonate, at plant/ RER/ kg	5	kg
polyethylene, HDPE, granulate, at plant/ RER/ kg	22	kg
synthetic rubber, at plant/ RER/ kg	65	kg
transport, lorry >32t, EURO5/ RER/ tkm	3 804	tkm
sawn timber, softwood, planed, air dried, at plant/ RER/ m3	2,69	m3
sanitary ceramics, at regional storage/ CH/ kg	1 500	kg
transport, barge/ RER/ tkm	41 300	tkm
transport, transoceanic freight ship/ OCE/ tkm	180 000	tkm
Operation, scenario 1		
electricity, medium voltage, production RER, at grid/ RER/ kWh	4	kWh
Operation, scenario 2		
electricity, medium voltage, production NORDEL, at grid/ NORDEL/ kWh	4	kWh
Operation, scenario 3		
electricity, medium voltage, at grid/ NO/ kWh	4	kWh

Table D-2: Total environmental impacts for the three scenarios for electrification of the platform.

Environmental impacts	**Scenario**			**Unit**
	1	**2**	**3**	
Agricultural land occupation	4,77E+00	6,52E+00	3,67E+00	m^2yr
Climate change	1,10E+02	4,36E+01	1,68E+01	kg CO_2 eq
Fossil depletion	3,23E+01	1,18E+01	4,91E+00	kg oil eq
Freshwater ecotoxicity	1,54E+00	4,68E-01	3,84E-01	kg 1,4-DCB eq
Freshwater eutrophication	9,95E-02	2,09E-02	1,67E-02	kg P eq
Human toxicity	8,22E+01	3,61E+01	2,85E+01	kg 1,4-DCB eq
Ionizing radiation	7,98E+01	6,66E+01	1,44E+01	kg U^{235} eq
Marine ecotoxicity	1,58E+00	5,21E-01	4,26E-01	kg 1,4-DCB eq
Marine eutrophication	1,15E-01	2,77E-02	1,70E-02	kg N eq
Metal depletion	1,02E+01	1,01E+01	9,91E+00	kg Fe eq
Natural land transformation	1,55E-02	8,54E-03	5,78E-03	m^2
Ozone depletion	5,45E-06	3,22E-06	1,25E-06	kg CFC-11 eq
Particulate matter formation	1,75E-01	9,81E-02	5,69E-02	kg PM eq
Photochemical oxidant formation	2,68E-01	1,31E-01	7,59E-02	kg NMVOC eq
Terrestrial acidification	5,40E-01	2,30E-01	1,64E-01	kg SO_2 eq
Terrestrial ecotoxicity	1,32E-02	3,59E-02	8,00E-03	kg 1,4-DCB eq
Urban land occupation	5,89E-01	4,35E-01	2,98E-01	m^2yr
Water depletion	8,13E-01	5,22E-01	1,30E-01	m^3

Table D-3: Total environmental impacts for the three different HVDC cable lengths.

Environmental impacts	**HVDC cable length**			**Unit**
	50 km	**100 km**	**200 km**	
Agricultural land occupation	3,33E+00	3,67E+00	4,34E+00	m^2yr
Climate change	1,47E+01	1,68E+01	2,10E+01	kg CO_2 eq
Fossil depletion	4,24E+00	4,91E+00	6,25E+00	kg oil eq
Freshwater ecotoxicity	2,70E-01	3,84E-01	6,12E-01	kg 1,4-DCB eq
Freshwater eutrophication	1,17E-02	1,67E-02	2,68E-02	kg P eq
Human toxicity	1,84E+01	2,85E+01	4,89E+01	kg 1,4-DCB eq
Ionizing radiation	1,39E+01	1,44E+01	1,54E+01	kg U^{235} eq
Marine ecotoxicity	2,97E-01	4,26E-01	6,86E-01	kg 1,4-DCB eq
Marine eutrophication	1,36E-02	1,70E-02	2,38E-02	kg N eq
Metal depletion	6,30E+00	9,91E+00	1,72E+01	kg Fe eq
Natural land transformation	4,81E-03	5,78E-03	7,73E-03	m^2
Ozone depletion	1,10E-06	1,25E-06	1,56E-06	kg CFC-11 eq
Particulate matter formation	5,01E-02	5,69E-02	7,89E-02	kg PM eq
Photochemical oxidant formation	6,00E-02	7,59E-02	1,08E-01	kg NMVOC eq
Terrestrial acidification	1,41E-01	1,64E-01	2,08E-01	kg SO_2 eq
Terrestrial ecotoxicity	7,37E-03	8,00E-03	9,25E-03	kg 1,4-DCB eq
Urban land occupation	2,31E-01	2,98E-01	4,33E-01	m^2yr
Water depletion	1,20E-01	1,30E-01	1,50E-01	m^3

Table D-4: Total environmental impacts for different energy requirements.

Environmental impacts	Energy requirement				Unit
	4 kWh	**4,8 kWh**	**6 kWh**	**7,2 kWh**	
Agricultural land occupation	1,33E-01	1,38E-01	1,45E-01	1,51E-01	m^2yr
Climate change	1,51E+02	1,80E+02	2,23E+02	2,67E+02	kg CO_2 eq
Fossil depletion	6,26E+01	7,49E+01	9,32E+01	1,12E+02	kg oil eq
Freshwater ecotoxicity	9,65E-02	9,95E-02	1,04E-01	1,09E-01	kg 1,4-DCB eq
Freshwater eutrophication	3,96E-03	4,08E-03	4,27E-03	4,46E-03	kg P eq
Human toxicity	3,59E+00	3,75E+00	4,00E+00	4,25E+00	kg 1,4-DCB eq
Ionizing radiation	1,89E+00	1,94E+00	2,02E+00	2,10E+00	kg U^{235} eq
Marine ecotoxicity	1,34E-01	1,44E-01	1,59E-01	1,74E-01	kg 1,4-DCB eq
Marine eutrophication	4,86E-02	5,72E-02	7,00E-02	8,29E-02	kg N eq
Metal depletion	1,42E+00	1,51E+00	1,63E+00	1,76E+00	kg Fe eq
Natural land transformation	3,02E-02	3,61E-02	4,49E-02	5,36E-02	m^2
Ozone depletion	1,88E-05	2,25E-05	2,80E-05	3,35E-05	kg CFC-11 eq
Particulate matter formation	1,11E-01	1,28E-01	1,53E-01	1,79E-01	kg PM eq
Photochemical oxidant formation	4,07E-01	4,84E-01	5,99E-01	7,14E-01	kg NMVOC eq
Terrestrial acidification	3,29E-01	3,76E-01	4,47E-01	5,17E-01	kg SO_2 eq
Terrestrial ecotoxicity	3,05E-03	3,52E-03	4,24E-03	4,96E-03	kg 1,4-DCB eq
Urban land occupation	1,30E-01	1,40E-01	1,54E-01	1,69E-01	m^2yr
Water depletion	2,45E-02	2,54E-02	2,67E-02	2,08E-02	m^3

Table D-5 : Total environmental impacts for different system lifetimes.

Environmental impacts	System lifetime				Unit
	10 years	15 years	20 years	25 years	
Agricultural land occupation	1,57E-01	1,42E-01	1,37E-01	1,33E-01	m^{2}yr
Climate change	1,51E+02	1,51E+02	1,50E+02	1,51E+02	kg CO_2 eq
Fossil depletion	6,28E+01	6,26E+01	6,26E+01	6,26E+01	kg oil eq
Freshwater ecotoxicity	1,25E-01	1,07E-01	9,96E-02	9,65E-02	kg 1,4-DCB eq
Freshwater eutrophication	4,51E-03	4,11E-03	4,02E-03	3,96E-03	kg P eq
Human toxicity	4,19E+00	3,78E+00	3,65E+00	3,59E+00	kg 1,4-DCB eq
Ionizing radiation	2,17E+00	1,94E+00	1,91E+00	1,89E+00	kg U^{235} eq
Marine ecotoxicity	1,64E-01	1,45E-01	1,38E-01	1,34E-01	kg 1,4-DCB eq
Marine eutrophication	4,80E-02	4,73E-02	4,71E-02	4,86E-02	kg N eq
Metal depletion	2,40E+00	1,84E+00	1,58E+00	1,42E+00	kg Fe eq
Natural land transformation	3,04E-02	3,03E-02	3,02E-02	3,02E-02	m^2
Ozone depletion	1,88E-05	1,88E-05	1,88E-05	1,88E-05	kg CFC-11 eq
Particulate matter formation	1,14E-01	1,12E-01	1,11E-05	1,11E-01	kg PM eq
Photochemical oxidant formation	4,11E-01	4,08E-01	4,07E-01	4,07E-01	kg NMVOC eq
Terrestrial acidification	3,33E-01	3,30E-01	3,29E-01	3,29E-01	kg SO_2 eq
Terrestrial ecotoxicity	3,17E-03	3,08E-03	3,06E-03	3,05E-03	kg 1,4-DCB eq
Urban land occupation	1,48E-01	1,34E-01	1,31E-01	1,30E-01	m^{2}yr
Water depletion	2,81E-02	2,54E-02	2,49E-02	2,45E-02	m^3