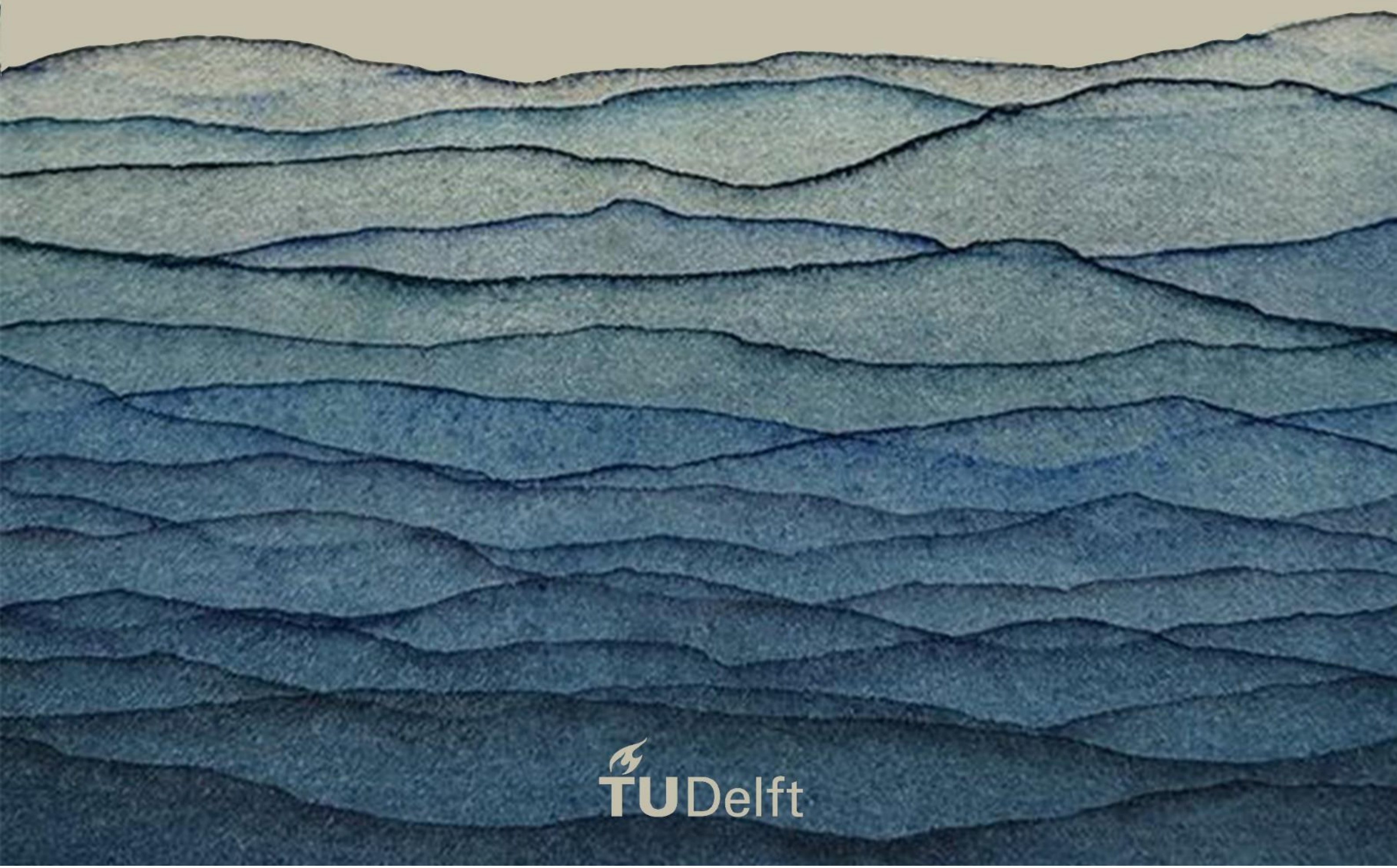
Life Cycle Assessment and Life Cycle Costing on Brine Effluent Treatment

A Case Study of the Zero Brine Project in the Netherlands

Kallirroi Panteleaki-Tourkodimitri



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Author: Kallirroi Panteleaki-Tourkodimitri

Student ID: 4754093

Graduation Committee

Chairman: Prof. dr. C.P. (Cees) van Beers

First supervisor: Dr.ir. G. (Gijsbert) Korevaar

Second supervisor: Ir. M.W. (Marcel) Ludema

Daily supervisor: Dr. ir. G.A. (Georgios) Tsalidis,

Date: December 2019

Contact: kallirroipantel@gmail.com

Preface

The last two years have been a challenge but also a great opportunity for me. From this journey, I gained invaluable experience and noteworthy knowledge which evolved me both personally and professionally. This work constitutes the conclusion of my Master of Science in Management of Technology program at the Delft University of Technology.

The achievement of this milestone would be impossible without the support and guidance of certain people. First of all, I would like to express my appreciation to my supervisor, Dr. Ir. G. Korevaar for his thoughtful insights and the intriguing conversations we had. Furthermore, I would like to express my sincerest thanks to the other members of the graduation committee Ir. Marcel Ludema and Professor dr. C.P. van Beers for their decisive comments and feedback. Special appreciation goes to my daily supervisor George for making himself available to provide explanations to my inquiries and his continuous encouragement.

Following, I would like to express my gratitude and love to all the people that I met these two years and become my family in Delft. Without them, this journey would have been incomplete. Cheers to those that we lived and those that will come.

Last but not least, I would also like to thank my friends in Greece for standing by me even though we were apart. Above all, I want to thank my parents and my brother for their support from the very first beginning and their encouragement every step of the way. I am thankful for having you.

Yours sincerely,

Kallirroi Panteleaki - Tourkodimitri

Executive Summary

The water crisis is one of the most important global risks influencing humanity. Urbanization as well as economic, social and technological evolution have led to water overconsumption across the world and thus to water scarcity. Industry comprises one of the main water consumers along with agriculture and municipalities. At the same time, industry constitutes a significant water polluter since a large amount of its wastewater does not receive treatment prior to its disposal to the environment. One of the greatest sources of wastewater is brine effluent, a hypersaline concentrate created during the water treatment in the industries. Brine concentrate is linked with numerous negative environmental impacts such as the pollution of groundwater or the alteration of water's properties. Furthermore, the treatment of brine before its disposal is costly and requires enhanced management solutions.

In an effort to tackle the challenges that brine effluent imposes, both in terms of management and costs, the process industry should shift to technical solutions that foster sustainable development. There are three dimensions with respect to sustainability; the environmental, the economic and the social. Life Cycle Assessment (LCA) as well as Life Cycle Costing (LCC), both of which are the main axis of this thesis, are tools for identifying and analyzing environmental and economic impacts respectively.

The object of this thesis is the Zero Brine (ZB) project which promotes a closed-loop approach to address the complex brine effluents by eliminating them, mitigating the effects of industrial processes while recovering materials such as water, energy, minerals, magnesium, and salts. This research is focusing on the Dutch case study where the assessment of a demineralized water production system before and after the implementation of ZB applications is taking place. The evaluation of sustainability performance comprises one of the main goals of this project. Thus, this thesis aims to assess the environmental and economic sustainability of the ZB project by implementing the LCA and LCC techniques. Given that, the following research question was developed:

"Which is the most efficient approach to identify the environmental and economic performance of the Dutch Zero Brine case study, in terms of environmental and economic assessment techniques?"

To that end, the parallel implementation of LCA and LCC was performed. Furthermore, the three types of LCC; conventional (cLCC), environmental (eLCC) and societal (sLCC), were also included in the analysis. The cLCC included the internal costs of the company; capital and operational expenses. Regarding eLCC and sLCC, they also incorporated the internal costs. Furthermore, the costs that are anticipated to be internalized in the decision-relevant future were included in the eLCC while the monetized environmental impacts were incorporated in sLCC. For the monetization of the external costs, LCA-based economic weighting factors were used. In light of this approach, all three types of LCC were utilized to perform the economic analysis of the system.

To support Evide's decision-making and strategic planning, the environmental and economic performance of the demineralized water plant before and after the application of the ZB systems was performed. The environmental impacts were quantified by utilizing the SimaPro software and applying the ReCiPe impact assessment method. In addition, the appropriate equations for the estimation of internal, transfer and external costs as well as for the Net Present Value (NPV) of the analyzed system were employed.

The results of the analysis showed that the implementation of the ZB system has ambiguous results concerning environmental performance. On the one hand, the majority of the environmental impacts were decreased by 15% to 22%. Specifically, ionizing radiation, human toxicity, marine eutrophication, freshwater, and marine

ecotoxicity categories showed better environmental performance after the implementation of ZB design. On the other hand, global warming, acidification as well as particulate matter formation categories were sharply increased by more than 100%. Regarding DWP itself, its performance showed an increase of 77% due to the fact that ZB applications reduce its requirements of sodium chloride, water, and electricity. However, this increase does not counterbalance the environmental damages of the ZB system in global warming, particulate matter formation, and acidification categories. ZB applications incorporate two different systems; Site I and Site II. Through contribution analysis, it was found that Site's I contribution is nearly zero while Site II is responsible for almost 50% of the environmental impacts. The main reason behind this is the large requirements of electricity and raw materials in order to treat the brine stream that derives from the demineralized water plant. Specifically, one particular process in Site II; Total Organic Carbon (TOC) removal, contributes to its environmental damages by more than 94%.

From the economic assessment results, it was concluded that the application of ZB design is not financially viable since it degrades the economic performance of the current production scheme. By estimating the NPV after the implementation of ZB applications, it was observed that it is negative thus rendering the project unsustainable. Even though, Site I & II increase the benefits of demineralized water plant by 7%, the costs for their operation are significant high. By examining the costs, it was observed that Site II is also accountable for the poor economic performance. The large quantities of consumables required for the TOC removal process are also the main cost contributor.

Overall, with the implementation of ZB systems; namely Site I and Site II, the environmental performance slightly increases in the majority of the impacts categories, however, it also sharply decreases in the remaining ones. Furthermore, the economic performance of the DWP's production scheme is not enhanced, rather it gets worse. To enhance the environmental and economic performance of ZB applications, more research required to tackle the abovementioned issues and to render ZB project a sustainable, industrially applicable solution for the treatment of brine and the recovery of valuable resources.

Abstract

In an attempt to address the issues that brine discharges impose, the Zero Brine project, a European Union-funded project under the Horizon 2020 program was developed. Zero Brine project's objective is the demonstration of novel, economically sustainable and industrially applicable solutions for the recovery of valuable resources while at the same time reducing the environmental footprint of brine effluent. The development of pilot plants in the Netherlands, Poland, Spain, and Turkey is taking place with the purpose of attaining the objectives of the ZB project. This study is focusing only on the Netherlands where the assessment of a demineralized water production system before and after the implementation of ZB applications is taking place. Therefore, this master thesis targets to evaluate the environmental and economic sustainability of the ZB project by implementing the LCA and LCC techniques

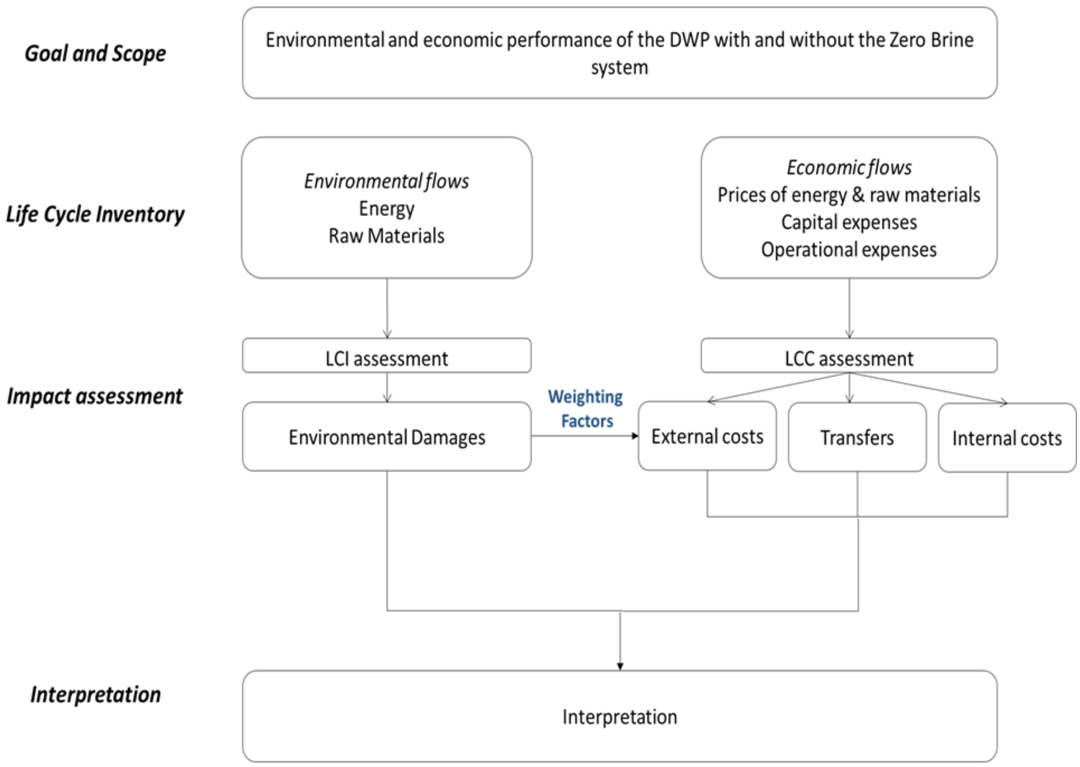
In regards to the Chemical Process industry and in particular to the water supply industry sector, there is no research so far that employs both LCA and LCC with the purpose of assessing the ecological footprint as well as estimating the related costs. Therefore, this thesis is partially conducted to contribute in that respect. Moreover, the three types of LCC are applied. Despite the fact that LCC has been gaining ground in recent literature, cLCC, eLCC, and sLCC are not developing at the same pace with sLCC lagging behind the other two. Thus, this report also aims to provide a new paradigm on how to conduct an sLCC and how to integrate cLCC, eLCC and sLCC into one analysis. Given that, this master thesis aims to elucidate the above-mentioned issues by developing the following research question:

"Which is the most efficient approach to identify the environmental and economic performance of the Dutch Zero Brine case study, in terms of environmental and economic assessment techniques?"

The answer to this question comes as a product of the following research scheme. Firstly, a literature review is performed with respect to the brine effluent and ZB project. In order to provide a background for theory on brine effluent, its characteristics along with its disposal methods and treatment technologies are reviewed. Furthermore, this review is followed by the provision of a complete overview of the ZB case study. The process flow diagrams for the current demineralized water plant (DWP) and ZB applications, namely Site I & II, are described. In particular, during the production of demineralized water, two streams of brine are generated, Ion Exchange (IEX) and Reverse osmosis (RO) brine which are treated from Site I & II, respectively. Following, the description of the ZB technologies for the treatment of brine effluents is provided. In addition, the contribution of the recovered resources, namely water, energy, sodium hydroxide, calcium hydroxide, and sodium bicarbonate is outlined.

Subsequently, a thorough literature review is conducted to comprehend the key theories, principles, and applications of LCA and LCC. Regarding LCA, its structure is provided based on the ISO 14040 and 14044 standards and consists of the following steps: 1) goal and scope definition, 2) inventory analysis 3) impact assessment and 4) interpretation of results. Next, the cLCC, eLCC and sLCC are reviewed and their key differences are detected. Furthermore, the identified ways in the literature to integrate the LCA and LCC tools are presented, specifically the parallel application of both LCA and LCC, the integration of cost aspects in LCA and the LCC as a leading concept. Moreover, the methods and approaches to quantify non-monetary units derived from LCA are presented and an appropriate weighting set for the ZB case study to monetize the environmental impacts is selected. Overall, the report's structure is presented. Specifically, the costs included in each type of LCC are determined, cLCC includes internal costs, eLCC includes internal costs as well as transfers (environmental taxes & subsidies) and sLCC take into account internal and external costs with the latter ones to be calculated with the help of the selected weighting factors. Additionally, the parallel implementation of

LCA and LCC is selected to integrate these two tools by using the LCA's structure to combine the environmental impact assessment and internal and external cost estimations as shown in the following figure.



To identify the environmental and economic performance of DWP before and after the application of ZB systems, the system boundaries, the allocation and the functional unit of the system are defined, the data required for the analysis are gathered and the necessary assumptions are made for the upscaling of Site I & II. The environmental impacts are quantified by utilizing the SimaPro software and applying the ReCiPe impact assessment method. In addition, the appropriate equations for the estimation of internal, transfer and external costs as well as for the Net Present Value (NPV) of the analyzed system are presented.

The results for the environmental and economic analysis of the Dutch ZB case were presented. Regarding the environmental analysis, it is concluded that after the implementation of the ZB system the environmental performance of the post-ZB system varies across the analyzed impact categories for the production of 1000 m^3 demi water. On the one hand, the majority of the environmental impacts were decreased by 15% to 22%. Specifically, ionizing radiation, human toxicity, marine eutrophication, freshwater, and marine ecotoxicity categories show better environmental performance after the implementation of ZB design. However, only freshwater ecotoxicity showed a significant decrease of 22%, with the other categories to demonstrate a decrease lower than 20%. On the other hand, global warming, acidification as well as particulate matter formation categories were sharply increased in the post – ZB system. In particular, the global warming impact category showed an increase of 114% and ionizing radiation and acidification were risen by thirteen and fifteen times, respectively. The performance of DWP itself after the application of the ZB system is increased in all the examined impact categories due to the reduction in the quantities of sodium chloride, water, and electricity. However, the categories that showed a significant decrease greater than 20% were the ionizing radiation, human toxicity marine eutrophication and ecotoxicity categories.

For the economic analysis, a full cost calculation was applied for the post – ZB system. The internal, transfer and external costs per functional unit were calculated for the cLLC, eLCC, sLCC. Overall the costs for cLCC, eLCC and sLCC for the production of 1000 m^3 demi water were equal to 3768.17€, 3826.55€ and 4519.17 €, accordingly. Furthermore, it was observed that Site I & II are responsible for almost 67% of the total costs in all three types of LCC compared to DWP that is responsible for 33%. Finally, the NPV rule was applied to determine the viability of the post-ZB system with and without the transfer and external costs. It is concluded that the post – ZB is not financially viable for implementation even without the inclusion of transfer and external costs.

Regarding the LCA, contribution analysis of the post-ZB sub system's and the processes of DWP and Site II was provided to determine the "hot-spots" of the system. Firstly, the post-ZB system was divided into three major sub-systems; the DWP (after the implementation of ZB design), the Site I and Site II. The percentage contribution of each sub-system to the total environmental impacts engaged in the production of 1000 m^3 demi water was obtained. It was concluded that both the DWP and Site II have the greatest impact on the environment with 50.6% and 48.5%, respectively. Site I had nearly zero environmental burdens compared to DWP and Site II.

Following, since Site II and DWP had the highest environmental impacts, process-specific contribution analysis was used to reveal which are the "hot-spots" in their processes. Regarding the DWP, the data provided by the company concerned the overall consumption of consumables and energy in the plant and were not process-specific. It was noticed that electricity and stream are mainly responsible for the environmental burdens across all impacts categories comprising more than 50% of the overall burdens across all impacts categories. As far as Site II is concerned, it was observed that the TOC removal process had the greatest contribution to all the impact categories, accounting for 94% to 98% of the overall impact. In particular, the underlying reason behind the bad environmental performance of the TOC removal process found to be the large quantities of sodium hydroxide and sulphuric acid employed. The environmental impact of sodium hydroxide was the largest contributor in all categories except acidification and particulate matter formation, ranging from 66% to 87%. In acidification and particulate matter formation, sulphuric acid comprised 67% and 79% of the overall environmental damages.

Concerning the LCC, the results obtained from the cLCC, eLCC and sLCC were analyzed. For the cLCC, the internal costs and benefits that occur in a year after the system's implementation were shown with each sub system's costs and benefits to be presented. It was observed that the costs for raw materials of Site II represent 96.4% of the overall cost for raw materials and 41% of the total internal costs. To that end, it is concluded that the bad economic performance of the post-ZB system is assigned to the Site II consumables. The large quantities of sodium hydroxide and sulfuric acid were again behind the poor economic performance of Site II.

For the total economic performance of the post-ZB system, it was observed that it is the implementation of Site I & II rendered the project economical unsustainable. The contribution of Capex, Opex, environmental and social costs of DWP and Site I & II to the economic performance of the post-ZB system were presented. It was observed that operational costs comprise the largest expense for the company. Both DWP and Site I & II operational expenses contribute 66% to the total costs. However, the operational costs of Site I & II are almost twice the DWP's costs. The capital costs along with the external costs of Site I & II contribute by 14% and 13%, respectively. Furthermore, the transfer costs along with the external costs of DWP have a low percentage of 1% and 3%, accordingly. Finally, sensitivity analysis was carried out for the assumptions that were made regarding the upscaling of Site I & II, the emissions attributed to the company, the selected discount rates, and the economic data used.

Concerning the incorporation of the three types of LCC in the ZB case study, several conclusions are also drawn. Despite the estimation of internal, transfer and external costs for the calculation of NPV, valuable insights are given from each type of LCC. With respect to cLCC, it is concluded that it could be excluded for the analysis since the internal costs captured in it are included also in eLCC and sLCC. eLCC except for the internal costs also includes the taxes for electricity and CO_2 emissions which amount to 0.55 M € per year. On the other hand, sLCC does not include transfer costs but incorporates the external costs by monetizing the environmental emissions resulted from the LCA analysis. These costs add up to 7.29 M € per year. By analyzing the external cost of each impact category, it is noticed that particulate matter formation, global warming, and acidification contribute the most in external cost estimation.

Finally, the research contribution and the limitations of the study along with recommendations for future research are present. The filling of the identified knowledge gaps as well as the identification of the "hot-spots" of the processes of ZB contribute to the scientific relevance of this project. Furthermore, the limitations of this work are discussed based on the assumptions that are made throughout the analysis. Additionally, recommendations for future research are given to ZB stakeholders as well as to LCA and LCC practitioners.

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List of Abbreviations

ALCA	Attributional Life Cycle Assessment
CE	Circular Economy
CLCA	Consequential Life Cycle Assessment
cLCC	Conventional Life Cycle Costing
Demi water	Demineralized water
DW	Drinking Water
DWP	Demineralized Water Plant
ED	Electrodialysis
EFC	Eutectic Freeze Crystallization
eLCC	Environmental Life Cycle Costing
EU	European Union
GHG	Greenhouse Gases
IEX	Ion Exchange
LCA	Life Cycle Assessment
LCC	Life Cycle Costing
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
MBR	Membrane Bioreactor
MD	Membrane Distillation
MF-PFR	Multiple Feed – Plug Flow Reactor
NF	Nanofiltration
PRO	Pressure – Retarded Osmosis
RED	Reverse Electrodialysis
RO	Reverse Osmosis
sLCC	Societal Life Cycle Costing
TOC	Total Organic Carbon
WTA	Willingness to Accept
WTP	Willingness to Pay
WWTP	Wastewater Treatment Plant
ZB	Zero Brine

Chapter 1

Introduction

1.1. Background

Water is the key element for preserving life on our planet and it constitutes an essential part of the development of every society as it is the driving force for both primary and secondary production sectors. However, the economic, social and technological evolution was based on the overexploitation of the natural environment, causing a number of adverse effects. Furthermore, urbanization increased the demand for water locally, requiring the transfer of ever-larger quantities of water from distant regions. The aforementioned burdened the water systems drastically and led to water scarcity. According to the World Economic Forum (2019), the water crisis is the fourth most important global risk influencing humanity, after weapons of mass destruction, failure of climate-change mitigation and adaptation, and extreme weather events.

Water scarcity can be defined as a lack of water due to physical shortage (physical scarcity) or due to insufficient infrastructure or failure of institutions to ensure a regular supply (economic scarcity) (UNESCO, 2019). In accordance with recent statistics, 40% of the global population encounters water scarcity and it is projected to rise up to 60% by 2025 (Schewe et al., 2013). Specifically, according to the United Nations (2018), around 1.2 billion people live in areas of physical scarcity and another 1.6 billion people face economic water shortage. Furthermore, in the last century, the use of water has increased by more than twice the growth of the population (Damania et al., 2017), resulting in the decrease of the world's freshwater reserves (FAO, 2017).

The global water consumption corresponds to 4001 km^3/year and the three main sectors that withdraw water are agriculture, industries, and municipalities which account for 70%, 19%, and 11% respectively. Although agriculture comes first to global water usage, it is remarkable that from a total of 334 km^3/year of water consumption in Europe, the industry's use amounts to 54% compared to agriculture that holds 25%. It is obvious that the dependence of the industry on the water makes it indispensable for its survival. In particular, in the Netherlands, as it is the main focus of this research, the industrial use of water amounts to 9.446 km^3/year compared to agriculture that consumes 0.0602 km^3/year (FAO, 2017).

However, the process industry can be characterized as a significant water polluter since 40% of its wastewater does not receive treatment prior to its disposal into the environment ("Water use in industry", 2016). The water that is processed in the industries is converted to brine effluent being one of the greatest sources of wastewater. Brine is a typically hypersaline concentrate discharge that requires disposal which is both costly and related to negative environmental impacts (Jones, Qadir, van Vliet, Smakhtin & Kang, 2019). Such impacts could be the alteration of the receiving water's physicochemical properties (Roberts, Johnston & Knott, 2010) or the pollution of the groundwater and soil (Mohamed, Maraqa & Al Handhaly, 2005). Current global brine production stands at 51.7 billion km^3/year that constitutes a 50% increase compared to earlier assessments (Jones et al., 2019). In view of the fact that the supply of quality water is declining, there is an increasing demand for wastewater reuse and recycling from the effluent. Furthermore, a number of scholars point out that brine could be a possible source of valuable minerals and magnesium compounds (Attia, Jawad & Al-Saffar, 2015;

Loganathan, Naidu & Vigneswaran, 2017). The recovery of these products, such as sodium chloride, calcium, potassium, and magnesium could provide a new source of raw materials, diminish the environmental effects of discharged brine effluents, enhance the recovery of freshwater and decrease the total cost of desalinated water (Loganathan et al., 2017). Consequently, there is an increased need for enhanced management solutions.

The consequences of current brine management techniques are in direct conflict with sustainable development that constitutes the main objective of the United Nations and the international community. Sustainable development is referring to the *"development that meets the needs of the present without compromising the ability of future generations to meet their own needs"* (Brundtland, 1987). In particular, these techniques fly in the face of sustainable development goals that aim to attain a better and more sustainable future by tackling global challenges, such as water scarcity and environmental degradation (United Nations, 2018). Furthermore, the concept of the Circular Economy (CE) is suggested to alter the ongoing ways of production and consumption that place a substantial burden on earth and its environmental ability (Walmsley, Ong, Klemeš, Tan & Varbanov, 2019). The need to move towards CE in the industry has been also acknowledged by the Dutch government (The Ministry of Infrastructure and the Environment and the Ministry of Economic Affairs, 2016).

In order to move forward towards CE and sustainability, three pillars of sustainable development have to be taken into consideration; the environmental, the economic and the social. Life Cycle Assessment (LCA) as well as Life Cycle Costing (LCC), both of which will be the main axis of this thesis, are tools for identifying and analyzing environmental and economic impacts respectively. Specifically, LCA is a standard methodology to evaluate the environmental impacts of products and services across the life cycle of a product (ISO, 2006) and LCC is a compilation and assessment of all costs related to a product, over its entire life cycle (Hunkeler, Lichtenvort & Rebitzer, 2008). The implementation of LCA together with LCC is of essential importance as it allows the researchers to broaden the influence and significance of LCA for decision making, to draw the important relationships and trade-offs between the environmental and economic performance, and to take into consideration LCA's scope and findings for the economic performance of companies (Norris, 2001). Finally, the significance of incorporating LCA and LCC into an analysis stems from the fact that it offers the starting point for sustainability incorporating the emerging future trends and the costs.

1.2. Research Problem

In an effort to tackle the challenges that brine discharges impose, both in terms of management and costs, the Dutch chemical process industry and in particular the water industry sector should shift to solutions which promote sustainability as it has been highlighted by researches, practitioners and policy agents (Singh, Murty, Gupta & Dikshit, 2009; Bakshi, 2011; Angelakoglou & Gaidajis, 2015; A, Pati & Padhi, 2019). There are considerable indications that the greening of economies promotes environmental, economic, and social sustainability in the long – term. However, it is of crucial importance for the companies to take into account both the environmental impacts as well the economic performance of the product or system that is under study.

The research problem of this master thesis concerns the assessment of the environmental and economic performance of a demineralized water (demi water) production system before and after the implementation of CE solutions which aim to achieve zero brine discharges and recover valuable minerals and magnesium compounds. In this regard, the identification of the trade-offs or win-win situation between environmental performance and cost minimization could possibly enhance the performance of water production systems throughout their life cycle. A scientific way to measure and then assess the various implications that come from the employment of CE solutions is the implementation of LCA and LCC techniques. Special attention will be given to LCC as this study will incorporate all three types of LCC (Hunkeler et al., 2008); conventional (cLCC), environmental (eLCC) and societal (sLCC). Hence, the external costs that will be determined by the identified environmental and social impacts will be measured and included in the analyzed system.

Concerning the application of LCA in the water industry, the number of LCA studies have sharply increased in the 1990s and 2000s. LCA applications in the aforementioned sector are mainly distinguished in two categories; wastewater treatment and water supply systems (Friedrich, Pillay & Buckley, 2010). Numerous researchers have performed the LCA of water supply systems (Sombekke et al., 1997, Raluy et al., 2002, Bonton et al, 2012,) or wastewater treatment plants (Kalbar et al., 2012; Lorenzo-Toja et al., 2016). The abovementioned studies focused on the comparison of various drinking water supply systems and the technologies that are included in the analyzed system. Furthermore, LCA was also employed in many studies in order to investigate the solutions to alleviate the environmental burdens by decreasing the salinity of feed water (Muñoz & Fernández-Alba, 2008), by utilizing efficient pretreatment (Beery & Repke, 2010) or by engaging cleaner energy sources (El-Nashar, 2001). However, the majority of the prior scientific papers only emphasized on the design and implementation phases to tackle the energy shortage issue (Zhou, Chang & Fane, 2013). Several LCA studies made the assumption that the brine was completely diluted prior to its disposal and had insignificant implications on the aquatic ecosystem (Raluy, Serra & Uche, 2005; Raluy, Serra, Uche & Valero, 2006; Zhou, Chang & Fane, 2011).

Nevertheless, from the above-mentioned studies, only Lorenzo-Toja et al. (2016) have included the LCC into their analysis to evaluate the economic feasibility of 22 wastewater treatment plants in Spain. It is remarkable that only a few studies concerning water industry have incorporated both the LCA and LCC and they are assessing only wastewater treatment systems (Lim, Park & Park, 2008; Kalbar, Karmakar & Asolekar, 2012; Kalbar, Karmakar & Asolekar, 2016; Resende, Nolasco & Pacca, 2019). Consequently, there exists a gap in the literature concerning the analysis of water supply systems by employing both the LCA and LCC and in particular assessing the process industry that produces demineralized water in the Netherlands.

Problem statement 1. The assessment of water supply systems by employing both the LCA and LCC is lacking in the literature.

Regarding LCC, the SETAC-Europe Working Group on Life Cycle Costing has classified LCC into conventional, environmental, and societal on the basis of the cost category and scope of assessment. In the literature, cLCC has widely applied to many sectors with various standards concerning its harmonization to be available (Hunkeler et al., 2008). Moreover, various methods for carrying out cLCC have been proposed (Dhillon 1989; Ellram 1993, 1994, 1995; Fuller and Petersen 1996). With respect to eLCC, its first appearance was in 2005 when Carlsson Reich (2005) used the term eLCC for the economic assessment of municipal waste management systems by combining the LCA and LCC techniques. Since then, it has become known and progressively acknowledged. In 2011, SETAC has eventually proposed a methodological approach for eLCC (Swarr et al., 2011). As a result of the fact that eLCC combines the costs that are associated with the environment together with the economic performance of a product or service, it has been broadly studied and applied in many industries (Taelman, Tonini, Wandl & Dewulf, 2018). At the same time, sLCC is not being entirely defined by the scientific community (Hunkeler et al., 2008). A very limited number of papers have employed sLCC. The majority of them have incorporated sLCC into their study as a part of a Life Cycle Sustainability Assessment (LCSA) (Muiña, González-Sánchez, Anna Maria Ferrari & Davide Settembre-Blundo, 2018) or combined with eLCC (Martinez-Sanchez, Tonini, Møller & Astrup, 2016; Edwards, Burn, Crossin & Othman, 2018). So far, only one paper concerning energy production has focused on sLCC as a stand-alone method (Weldu, 2018). It is clear that the sLCC is still in an initial phase and additional research is required to enhance the clarity and reliability of this method specifically for application in practice.

Problem statement 2. sLCC is still in an early stage in terms of methodology and more case studies are required to advance the research area.

As a result of the immaturity of sLCC, few studies have included all the three types of LCC into their analysis (Martinez-Sanchez, Kromann & Astrup, 2015; Martinez-Sanchez, Tonini, Møller & Astrup, 2016; Edwards, Burn, Crossin & Othman, 2018). There is no consensus on how to incorporate all three types in one study. Moreover, none of the above literature so far concerning sLCC or the incorporation of cLCC, eLCC and sLCC deal with the water industry. They are addressing the energy production sector (Weldu, 2018) and waste management systems (Martinez-Sanchez, Kromann & Astrup, 2015; Martinez-Sanchez, Tonini, Møller & Astrup, 2016; Edwards, Burn, Crossin & Othman, 2018).

Problem statement 3. The incorporation of the three different types of LCC; cLCC, eLCC, sLCC, into one study is lacking from the literature.

1.3. Research Object: The Zero Brine (ZB) Project

Under the Horizon 2020 program, the ZB project's main objective is the demonstration of novel, economically sustainable and industrially applicable solutions for the recovery of valuable resources. These materials will be recovered from the process industry wastewater brine while at the same time eliminating it. ZB promotes a closed-loop approach to address the complex brine effluents by eliminating them, mitigating the effects of industrial processes while at the same time recovering materials such as water, energy, minerals, magnesium and salts that it is possible to be recycled in the same (internal valorization) as well as different process industries (external valorization) (Xevgenos, Kutka, & Wabitsch, 2019).

The development of demonstration plants in the Netherlands, Poland, Spain, and Turkey is taking place with the purpose of attaining the objectives of the ZB project. This thesis will focus only on the Netherlands, in the multi-company site of the energy port and petrochemical cluster in the Botlek area of Rotterdam Port, where the goal is to re-define the supply chain of water and minerals. One of the industries that participate in the project is Evides. Evides is producing pure demi water, that is purchased by the Chlor-alkali industry in the Botlek area, and brine as a waste. There are two sources of brine in the demi water plant (DWP), the ion exchange softening (IEX) unit and reverse osmosis (RO) unit. The ZB project aims at treating brine from Evides to recover distilled water, salts and magnesium. For this purpose, the demonstration plant will consist of two sites, with each site processing brine from one of the two brine sources in the DWP. The case study is described in more detail in section 2.2.

The integration of several existing and innovative technologies aiming at the recovery of high quality and sufficient purity end-products is a key concept in the ZB project. However, apart from technical feasibility, the evaluation of sustainability performance comprise one of the main goals of this project. Sustainability performance is of essential importance and all the three dimensions of it; economic, environmental and social will be addressed under the ZB project requirements. Aiming to identify the sustainability of the suggested processes, the environmental and economic impacts are modeled with the LCA and LCC techniques.

1.4. Research Objective and Questions

On the basis of the above mentioned, this thesis aims to 1) tackle the knowledge gaps identified in the literature research by 2) applying the LCA and LCC techniques in the ZB project. Therefore, the underlying objective of this thesis is to assess the environmental and economic sustainability of ZB applications by employing the LCA and LCC methodology while taking into consideration all the three types of LCC. In particular, concerning LCA, the environmental impacts of the Evide's current production scheme will be measured and compared with the ZB applications to assess how environmentally sustainable are the proposed applications. Furthermore, regarding economic sustainability, cLCC, eLCC and sLCC will be applied both to DWP and to ZB system. Besides the comparison between Evide's economic performance with (post-ZB system) and without (pre-ZB system) the ZB applications, the inclusion of all types of LCC will allow the identification of the most appropriate type for this case study as well as it will contribute to the development of LCC methodology. Moreover, the trade-offs or win-win situation between environmental performance and cost minimization will be acknowledged.

In regards to the above, this report aims to elucidate the above-mentioned issues by developing the following research question:

"**Which is the most feasible approach to identify the environmental and economic performance of the Dutch Zero Brine case study, in terms of environmental and economic assessment techniques?"**

Aiming to answer the above main research question of the project, four sub-questions (SQs) have been developed:

SQ1: How to perform an environmental and economic assessment in terms of LCA and LCC techniques?

SQ2: Which are the key differences among the conventional, environmental and societal LCC?

SQ3: Which LCC type is the most appropriate for the Dutch Zero Brine case study?

SQ4: Could the Zero Brine applications enhance the environmental and economic performance of the analyzed Demineralized Water Plant's production scheme?

1.5. Research Methodology

1.5.1. Research Approach

In order to answer the above questions qualitative as well as quantitative research are going to be followed. Qualitative research is conducted with the aim of answering the SQs 1-2. The fundamental principles and the state-of-art about LCA and LCC techniques along with the approaches that could be used to monetize the environmental and social impacts are reviewed. Furthermore, the key differences between the three types of LCC are reviewed to apply them in practice. This comprehensive literature research is expected to assess what is already known in the field and identify best practices.

Subsequently, quantitative research is followed to answer SQs 4-5. This step aims to evaluate the environmental and economic sustainability of ZB applications. Moreover, the most appropriate LCC type is determined to find a feasible approach for improved environmental performance and cost minimization of the ZB project.

1.5.2. Research Methods

For the qualitative research, textual sources are investigated by conducting document analysis and literature review throughout the thesis and most specifically in the initial phase. Journal articles, book chapters, reports, conference papers, MSc and Ph.D. thesis are studied. The main search databases that are used for this purpose are Science Direct, Research Gate, Scopus, and Google Scholar. Different keywords such as " "Integrating LCA & LCC", "LCA & LCC in the water industry", "monetization of external costs" are utilized to identify the information needed.

With respect to quantitative research, the LCA is performed by using the SimaPro software to measure the related environmental impacts. The assessment of the environmental performance is conducted under the standards of ISO 14040 and ISO 14044 and the impact assessment method that is utilized is ReCiPe. Concerning the modeling of DWP and ZB systems, the required data for material and energy inputs and outputs were collected from Evides, ZB Grant Agreement document as well as upon communication with the consortium partners that are providing the technologies.

The LCC analysis is carried out by estimating the internal, transfer and external costs. The economic assessment is conducted by measuring the Net Present Value (NPV) of the detected costs under a specific time horizon. The economic data were collected from Evides annual report, literature and online databases.

1.6. Research Relevance

1.6.1. Scientific & Societal Relevance

The research carried out within the context of this thesis aims to fulfill the knowledge gaps identified in section 1.2 and contribute to the development of a theoretical and practical knowledge base. In regards to the chemical process industry and especially the water supply industry sector, there is not yet research that employs both LCA and LCC with the purpose of assessing the ecological footprint as well as estimating the related costs. Therefore, the chemical process industry that is under study – that produces demi water for usage in the Chlor-alkali industry in the Netherlands and discharge brine – calls for further research to evaluate its performance in terms of environmental and economic efficiency.

In this thesis, this will be addressed by applying the three types of LCC. Despite the fact that LCC has been gaining ground in recent literature, cLCC, eLCC, and sLCC are not developing at the same pace with sLCC lagging behind the other two. To enhance the reliability of sLCC, it is imperative to enrich the literature with more case studies and methodologies. Furthermore, the incorporation of all three types of LCC into one study is lacking from the literature. The goal of this work is to make a scientific contribution by applying the LCA and LCC tools in a country and industry that have been neglected the literature. Finally, by paying special attention to LCC, this research will provide a new paradigm on how to conduct an sLCC and how to integrate cLCC, eLCC and sLCC into one analysis. On the whole, this study aims to provide insights into the sustainability of both water and brine treatment systems.

Societally, the contribution of this research is two-fold. On the one hand, the results of this thesis could provide incentives for more companies in the industry to pursue the life cycle thinking along with their processes. Following the implementation of LCA and LCC methodology, the identification of critical issues along with the provision of suggestions could improve the environmental and economic sustainability of the concerned companies. The outcome of this research could motivate companies to follow life cycle thinking techniques to determine which actions should be taken in an effort to be environmentally and economically sustainable. On the other hand, this project could assist businesses in the decision-making process of pursuing alternative sustainable technologies concerning the treatment of brine effluent. The evaluation of ZB technologies is expected to assist in strategic planning and decision-making processes for the elimination of brine discharges.

1.6.2. Relevance to Management of Technology (MoT)

In the context of the MoT master's program, students should follow three main axes when conducting their thesis project. These indicators were defined in the following way:

i. "The work reports on a scientific study in a technological context"

This project constitutes a case-specific study concerning the ZB project. The last referred is a project which incorporates innovative technologies aiming to treat industrial brine effluents by means of collaboration among organizations, researchers and public actors with the purpose of alleviating the environmental, economic and social burdens of industrial discharges. Accordingly, this thesis is founded on a technological context given that its main objective is to evaluate the sustainability of the suggested technological solutions both environmentally and economically. Furthermore, the assessment of the system's sustainability performance is one of the ZB project objectives. Concerning the scientific relevance of this study, it has been elaborated in section 1.6.1.

ii. "The work shows an understanding of technology as a corporate resource or is done from a corporate perspective"

The viewpoint of this thesis is that of a company. Specifically, the evaluation of the economic and environmental sustainability of ZB systems will provide incentives and recommendations not only to the concerned company but also to other companies of the industry so as to engage in sustainable innovations and business models. This research can facilitate the implementation of sustainable techniques and enable companies to successfully carry out these activities.

iii. "Students used scientific methods and techniques to analyze a problem as put forward in the MoT curriculum"

The courses of the MoT curriculum and the related concepts employed in this thesis are as follows:

- *Financial Management – MOT1461*: Fundamental financial concepts and tools
- *Inter- and Intra-organizational Decision- making – MOT1451*: Complex decision-making
- *Research Methods – MOT2312:* Design and execution of research
- *Technology Dynamics - MOT1412:* Incorporation of values in order to steer innovation into a societally responsible direction
- *Technology, Strategy & Entrepreneurship -MOT1435:* Patterns of technological innovations and related technology strategies

1.7. Report Structure

This report is structured based on the following logical sequence, first the research problem is defined, the appropriate literature is reviewed, the relevant literature findings are applied, the results are discussed and then the conclusions are drawn and the recommendations for future research are given. The full structure of this thesis is visualized in the research flow diagram presented in Figure 1 which supports the structure of this work and assists in balancing the research load of the sub-questions.

Following this introduction, Chapter 2 offers an overview of brine effluent challenges and opportunities along with the technologies that are used for its treatment. Furthermore, the ZB case study is thoroughly analyzed by addressing its key concepts and technological implications. Next, in Chapter 3 a thorough literature review is carried out with the purpose of understanding the basic notions employed in this report. The key theories, principles, and applications of both LCA and LCC are reviewed. Furthermore, the methods and approaches for monetizing the external costs are also examined. Subsequently, the followed methodology is described in Chapter 4. In the beginning, the goal and scope, the system boundaries and the functional unit of the environmental and economic assessment are presented. Then, the inventory list is formulated consisting of the data concerning the used raw materials and energy. Subsequently, the outcomes of the analysis are presented in Chapter 5. Then, the results are interpreted in Chapter 6. Contribution along with sensitivity analysis takes place. Thereinafter, Chapter 7 answers the raised research questions and offers the conclusions of this report. Finally, the researcher reflects on how the findings of this research contribute to academia and in practice, and possible agendas are proposed for future research.

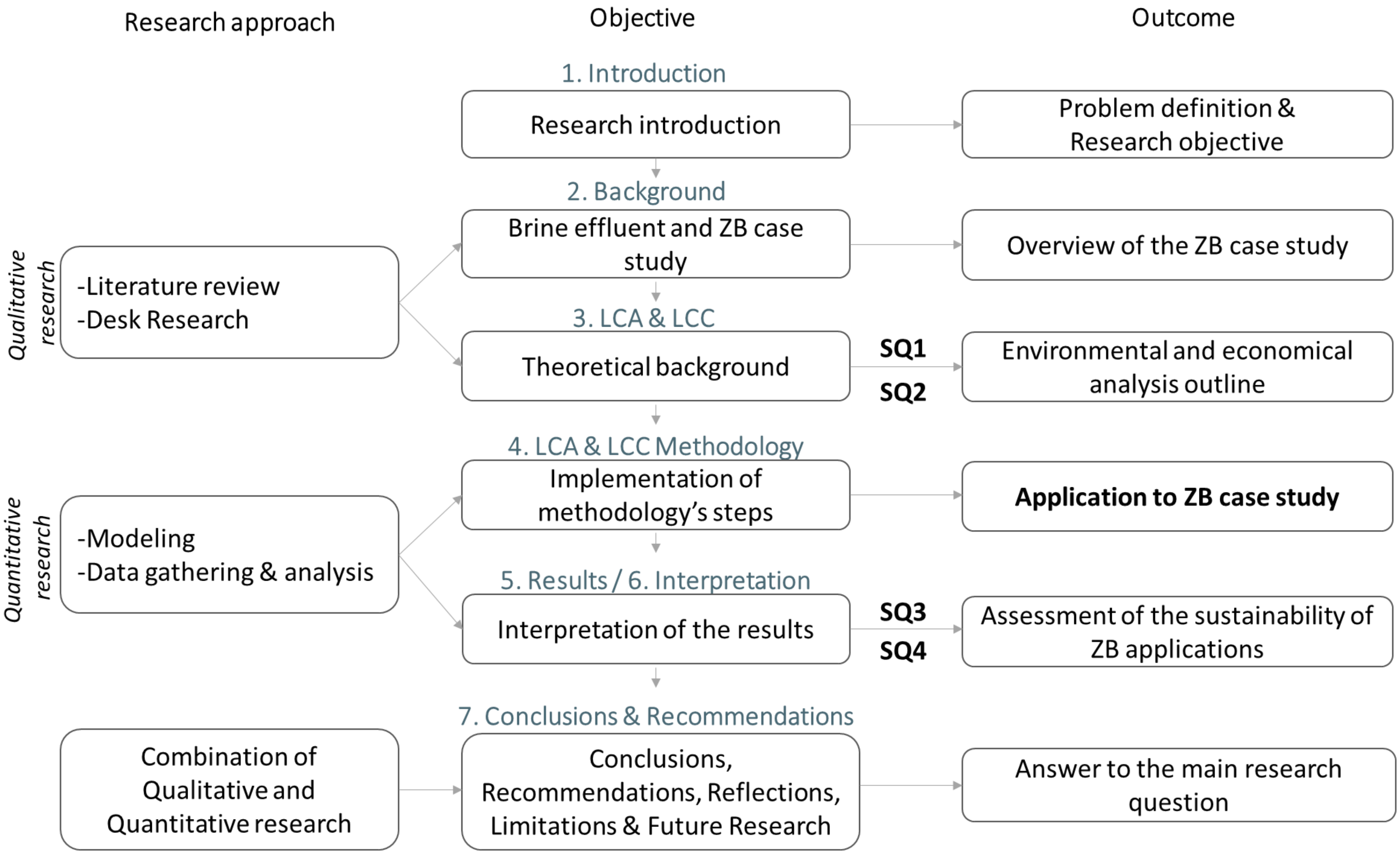

Figure 1. Research Flow Diagram

Chapter 2

Background

In the present chapter, the outcomes of the literature review and the description of the ZB case study are provided. The aim of this section is to provide the theoretical background on brine effluents and a complete overview of the ZB case study. This chapter is organized as follows. In the first section, Section 2.1, the characteristics of brine effluent are presented along with its disposal methods and treatment technologies. Furthermore, the importance of recovering materials from brine is highlighted. Next, Section 2.2 outlines the production schemes of DWP and ZB applications. Following, the technologies employed in the ZB project are analyzed. Finally, the contribution of targeted recovered resources is explained.

2.1. Brine Effluent

In the early '70s, it was the first time that brine effluent was acknowledged as an environmental problem when increase salinity was observed in Colorado River caused by power plant releases (Seigworth, Ludlum & Reahl, 1995). Nowadays, brine discharges are produced from many processes in the industry, such as desalination, IEX regeneration, solar ponds and ballast water (Ariono, Purwasasmita & Wenten, 2016). However, due to the fact that the chemical process industry is mainly producing brine effluent from desalination and IEX regeneration, these two sources of brine effluent will be analyzed.

The elevated levels of salts and pollutant substances of brine discharges constitute an important burden on the environment (Ariono et al., 2016). These pollutants differ in regard to the source. Concerning the desalination plants, effluents have increased salinity compared to seawaters and the various chemicals that are employed throughout the process, contaminate the desalination concentrates (Kress, 2019). Meanwhile, brine effluents that derive from IEX regeneration are tainted with resinous substances, multivalent ions of softeners or organic pollutants (Kabsch-Korbutowicz, Wisniewski, Łakomska & Urbanowska, 2011). As a consequence, in the absence of further treatment, the brine effluents are usually discharged directly causing adverse effects on the environment.

The chemical process sector is the main brine producer in Europe. Brine discharges are problematical as they induce environmental damage while at the same time contain critical raw materials. Regulations have been developed to prohibit European companies from discarding industrial brine on surface water (US EPA, 2014). Moreover, brine's substances are of essential importance as the European Union (EU) has identified some of them as Critical Raw Materials (European Commission, 2014) and are prioritized in its Circular Economy Package (Bourguignon, 2016).

2.1.1. Environmental Impacts

Many researchers have studied the impact of brine disposal. The main environmental challenges regarding desalination processes are the disposal of brine effluents to the aquatic surroundings as well as the energy requirement. According to Roberts et al. (2010), these effluents could induce physiochemical and ecological consequences. To begin with, the physicochemical characteristics of the receiving water could be modified as a result of effluent's high concentration of salt, temperature, and components. Accordingly, this modification could result in negative impacts on the marine environment. For instance, increased salinity and temperature may alter the osmotic equilibrium between the water and the organisms which could be harmful to them.

Moreover, the presence of toxic elements like heavy metals might also damage the organisms causing water eutrophication (Von Medeazza, 2005).

Furthermore, anti-biofouling and anti-scaling agents in brine, which are used in desalination plants, constitute a severe environmental problem due to high toxicity and low degradability, respectively (Hoepner & Lattemann, 2003). Meanwhile, a study has indicated that improper discharge of brine effluent, also from desalination plants, could pollute the groundwater and soil resulting in a reduction in plant and soil productivity (Mohamed et al., 2005).

As far as the IEX brine from regeneration is concerned, disposal wells are required as the containing organic matter is unlikely to be handled in conventional sewage systems. However, the scaling tendencies in the pipeline system have to be taken into account during the release of the brine in the disposal well. Considering that, the anti-scaling reagent needs to be included throughout the disposal which is costly and raises the total cost of disposal (Barranco, Balbuena, Garcia & Fernández, 2001). Moreover, the increased salinity poses difficulties to organic activities, hence microorganisms that are persistent to salt are necessary to absorb the organic matter in the spent brine (Ariono et al., 2016).

2.1.2. Current Brine Disposal Methods

Brine disposal is distinguished in the following four disposal options; direct disposal, evaporation-treated disposal, energy-intensive disposal, and land applications (Ariono et al., 2016, Panagopoulos et al., 2019). As illustrated in Table 1, it stands to reason that direct disposal of brine comprises the most economical option as it acquires low investment, operating, and maintenance expenses. However, with the purpose of alleviating the potential environmental consequences of direct disposal, a number of further treatments might be required including blending, mixing zone, usage of non-toxic chemicals, pH adjustment, and diffusers (Panagopoulos, Haralambous & Loizidou, 2019). Sewer discharge also requires lower expenses and energy in the case that a wastewater treatment system exists near to the factory. Nevertheless, when a large amount of brine is released, the organic matter might not be handled entirely as the composition of brine could decrease the ability of organic processes (Xu et al., 2013). Deep-well injection is proper for inland desalination plants for high volumes of discharged brines. In spite of this, deep-well injection requires large capital costs, it might cause contamination to the groundwater and it might set off earthquakes (Ahmad & Baddour, 2014).

Table 1. Brine Disposal Options (Ariono et al., 2016; Panagopoulos et al., 2019)

Methods	Options	Main environmental problems
Direct disposal	Surface discharge Sewer discharge Deep-well injection	Marine environment pollution Performance of biological treatment Groundwater pollution and soil salinization
Evaporation – processed disposal	Evaporation pond Solar evaporation Wind-aided intensification (WAIV) technology	Pollution of groundwater aquifers
Energy-intensive disposal (Zero Liquid Discharge)	Brine concentrator Crystallizer Spray dryer	Intensive energy consumption
Land applications	Irrigation of halophilic crops and weeds	Soil salinization

Considering the drawbacks of direct disposal of brine effluents, evaporation – treated and energy-intensive disposal methods are coming forth. Evaporation ponds are rather simple to build and requiring low maintenance and operation cost. However, a relatively big area of land is needed leading to very large capital costs (Panagopoulos et al., 2019). Furthermore, they are very dependent on solar energy, thus relies much on weather conditions. In order to address the abovementioned issues, WAIV technology has emerged to exploit the wind energy by evaporating moistened surfaces that have a high packing density per footprint (Gilron, Folkman, Savliev, Waisman, Kedem, 2003).

Meanwhile, the zero liquid discharge (ZLD) technology has been developed with the purpose of providing freshwater of high quality while eliminating the liquid effluents (Panagopoulos, 2019). For the recovery of solid components from brine, a brine concentrator and a crystallizer or spray dryer are employed (Brandhuber, Cerone, Kwan, Moore & Vieira, 2007). Nevertheless, as a result of intensive energy use, ZLD method is described as a capital intensive technology and it has not yet extensively implemented.

Apart from the mentioned methods, land applications have also been established despite the fact that its implementation is severely restricted. This method is dependent only on land resources, microclimate, crop resistance to salinity and the location of the underground water (Panagopoulos et al., 2019). In addition, there is a small proportion of plants that could flourish in high salinity water conditions.

2.1.3. Brine Treatment Methods

2.1.3.1. Brine from Desalination Process

As mentioned above, brine effluents have negative impacts on the environment and thus some advanced treatments have been developed to mitigate them. The purposes of these technologies, as summarized in Table 2, are the recovery of water and salt, the generation of salt, the recovery of valuable metals, and further profitable applications of the brine. During the desalination process, the feed water is divided into two distinct flows – a freshwater stream and a concentrate waste stream (Wenten, Ariono, Purwasasmita, Khoirudin, 2017). For that purpose, water recovery is held synchronously with salt production in a plant with dual function (Wenten, 2016). The more water that is recovered, the more brine is produced, which is suitable for the production of salt.

Potential treatments for water recovery and salt production are electrodialysis (ED) as well as electrodialysis reversal (EDR). ED is employed as a pre-concentrator that is established prior to the evaporator (Korngold, Aronov & Daltrophe, 2009). A study concerning the implementation of ED in salt production in a seawater reverse osmosis plant (SWRO) has demonstrated that the use of brine discharges causes a 20% cost reduction in the salt production process compared to the use of seawater (Tanaka, Ehara, Itoi & Goto, 2003). Meanwhile, in order to restrain fouling as a result of the precipitation of organic or inorganic components on the IEX membrane, EDR has been suggested. During EDR, the foulant is withdrawn from the membrane by inverting the polarity. As a result, the level of salt saturation and the total water recovery are enhanced along with RO (Turek, Was, Dydo, 2009). Nevertheless, ED is yet subject to additional advancement to lower energy consumption.

Table 2. Brine treatment technologies – Desalination process

Purpose	Technologies	Reference
Water recovery	Capacitive de-ionization (CDI) Electrodialysis (ED) Electrodialysis reversal (EDR) Forward osmosis (FO) Membrane distillation (MD)	Mericq et al. (2010) Korngold et al. (2009) Turek et al. (2009) Ng et al. (2008) Subramani et al. (2012)
Water and salt recovery	Electrodialysis (ED) Membrane distillation (MD) Membrane Crystallization (MCr)	Tanaka et al. (2003) Ji et al. (2010) Ji et al. (2010)
Energy generation and brine dilution	Reversed electrodialysis (RED) Pressure retarded osmosis (PRO)	Tufa et al. (2014) Kim, Park, Snyder & Kim (2013)

Another possible treatment for water recovery and salt production is membrane distillation (MD). Throughout MD, water with high purity is recovered at the same time with a solution that contains high solute concentration (Subramani, DeCarolis, Pearce & Jacangelo, 2012). Nonetheless, the main issue concerning MD is the fouling as a result of organic or inorganic components that result in lower membrane permeability (Mericq, Laborie & Cabassud, 2010), or even to abiding damage of the membrane (Gryta, 2008). An additional issue of MD constitutes the wetting phenomenon. During this phenomenon, mass transfer resistance is enhanced and in order to repress it numerous alterations to the membrane's surface have been proposed (Himma, Anisah, Prasetya & Wenten, 2016).

Pressure-retarded osmosis (PRO), as well as reverse electrodialysis (RED), could be used for power production by taking advantage of the salinity gradient between brine effluent and freshwater (Wenten, Khoiruddin, Aryanti, Hakim, 2016). Regarding PRO, a semi-permeable membrane enables the water transfer from a solution with a lower salt concentration into one draw solution with high concentration (Lee, Baker & Lonsdale, 1981). A hydro – turbine is utilized in order to create energy, which is converted from kinetic to electric energy. During RED, several anion and cation exchange membranes are jointly piled up in a changing pattern between anode and cathode to enable the discerning transfer of salt ions (Długołęcki, Gambier, Nijmeijer & Wessling, (2009). Therefore, electrical potential is produced from chemical potential by transferring ions from the high salinity solution to the low salinity solution.

In order to accomplish the objective of ZLD and the twofold objective of concurrent desalination and salt production, numerous unified desalination processes have been suggested. The abovementioned have zero or near zero liquid discharge, enhanced water recovery and lucrative salt production that could be employed to counterbalance the total freshwater production cost.

2.1.3.2. Spent Brine Regenerant

Brine reuse consists one of the most auspicious options for the treatment of spent brine regenerant from ion-exchange processes. The technologies that could be applied for this objective are shown in Table 3.

Chemical precipitation has considerable advantages, such as the decrease of regeneration and brine disposal expenses by 90% compared to waste hauling and the reduction of water's cost for regeneration. However, the cost of the chemicals used is greater than sewer disposal owing to the alkaline solution that is used (Michaud, 1994). During chemical precipitation, precipitation, neutralization by using acid and salt production are taking place (Michaud, 2010). Generally, in ion-exchange softening plants, chemical precipitation is applied due to the acceptable overall cost for brine discharges.

Another alternative is nanofiltration (NF), which is a membrane-based technology with the ability to eliminate the organic components. The aforementioned is based on the fact that the rejection towards bivalent and multivalent ions is high compared to the rejection towards monovalent ions which is low to moderate (Cartier, Theoleyre, Decloux, 1997). However, further acidification or anti-scaling injection is needed due to the high concentration of brine.

Table 3. Brine treatment technologies – Spent brine regenerant

Purpose	Technologies	Reference
Water reuse	Reverse osmosis (RO) Membrane distillation (MD)	Ghasemipanah (2013) Gryta, Karakulski, Tomaszewska & Morawski (2005)
Brine reuse	Chemical precipitation Nanofiltration (NF) Biological treatment Membrane bioreactor (MBR)	Michaud (1994) Wadley, Brouckaert, Baddock & Buckley (1995) Hiremath et al. (2006) McAdam, Pawlett & Judd (2010)

Membrane bioreactor (MBR) process is a mixture of standard activated sludge and membrane separation process. Although during the MBR process the organic substances are being removed, the fouling of membrane is a major problem that hinders the method. Many approaches have been employed to reduce membrane fouling (Aryanti, Yustiana, Purnama, Wenten, 2015), as without fouling, MBR is an auspicious option for spent brine, especially for the brine that is highly burdened with organic substances.

2.1.4. Recovery of Minerals

Brine effluents contain substances of great value that can be recovered for beneficial usage. These elements are the following: boron (B), bromine (Br), calcium (Ca), chlorine (Cl), iodine (I), lithium (Li), magnesium (Mg), potassium (K), sodium (Na), strontium (Sr) and sulfate (SO_4) (Attia, Jawad & Al-Saffar, 2015). The recovery of the aforementioned elements could offer an advanced solution to the growing problem of brine disposal. The extracted elements could convert in quality minerals that could substitute or decrease the current overconsumption of non-renewable minerals employed in various industries.

Therefore, the extraction of minerals from the high salinity effluent has two major advantages:

i. Environmentally, by discharging less concentrated brine, thus minimizing the adverse effects on the aquatic environment and approaching ZLD. Additionally, decreasing the depletion of natural assets and the usage of water and energy (Perers & Pintó, 2008, Meneses, Pasqualino, Céspedes-Sánchez, Castells, 2010).

ii. Economically, by turning minerals into commercially marketable products. Additionally, increasing the amount of recovered water and lowering the cost of desalinated water (Kim, 2011, Morillo et al., 2014).

The main technologies that are used for the recovery of minerals are solar evaporation ponds, WAIV, ED, IEX, eutectic freezing crystallization (EFC), membrane separation, NF, evaporation, crystallization and chemical precipitation (Attia, Jawad & Al-Saffar, 2015) that have been already addressed above.

2.2. Case Study: Zero Brine Project

ZB project is financed by the European Commission under the Horizon 2020 program and its main objective is the demonstration of novel, economically sustainable and industrially applicable solutions for the recovery of valuable resources while at the same time reducing the environmental footprint of brine discharges. The demonstration is applied in the DWP in the Botlek area of Rotterdam which is owned by Evides. Evides manufactures pure demi water which is used by the Chlor-alkali industry and brine as a waste product. Evides currently discharges the brine effluent to the seawater nearby, but it considers improving its environmental performance by recovering and reusing the sodium salt. The demonstration plant consists of Site I and Site II, with each site processing brine from one of the two brine sources in the DWP. This part explains in more detail the DWP set – up as well as the ZB applications (Zero Brine, 2017).

2.2.1 Demineralized Water Plant

The Evide's DWP at Botlek area is supplied with water from the Brielse Lake, which is one of the branches of the river Maas. The DWP produces high-quality demi water by using several purification techniques and supplies numerous organizations in the Botlek area. The DWP, as shown in Figure 2, includes dissolved air flotation and filtration to remove suspended matters from the intake lake water and cationic IEX columns for removing divalent cations and softening the water. Next, an RO unit is involved to remove monovalent salts, and mixed bed IEX columns to polish the RO permeate, namely to remove traces of salts and charged organics, and make the permeate ready to be used by the companies.

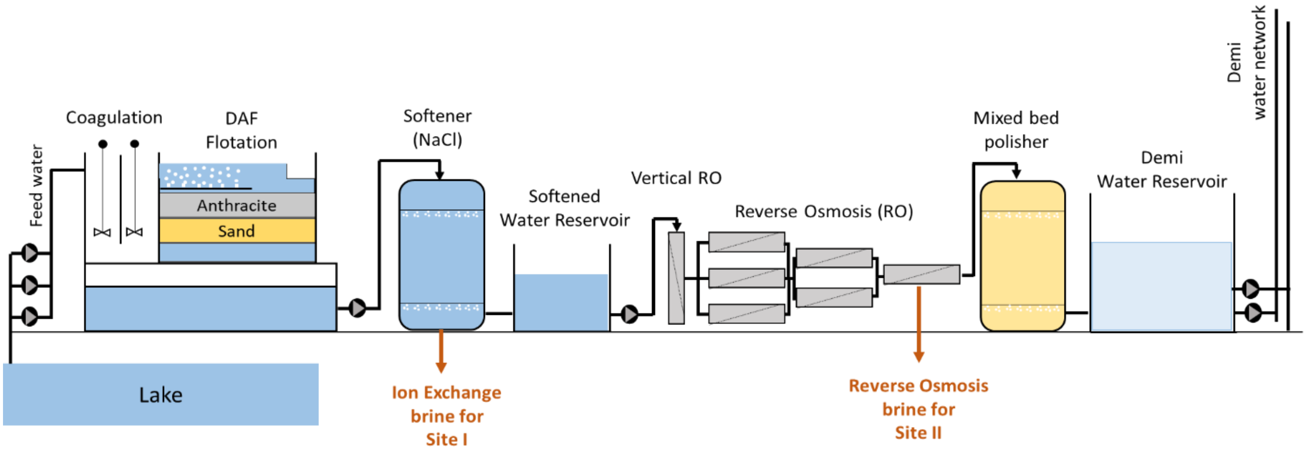

Figure 2. DWP

Both the IEX and RO units produce brine which is processed by the Zero Brine system due to the resin's regeneration of the IEX softener and extraction of salts during normal operation. For the regeneration of the IEX resins, high-purity salt is necessary. Currently, brine is disposed into the sea, however, the installation of the ZB system results in processing the brine to produce distilled water, magnesium, calcium and sodium salts. The recovered sodium chloride will be recycled in the DWP, recovered water will decrease the water pumped from the lake, whereas the remaining salts will be sold externally and replace existing products in the market. The chemical composition of IEX and RO brines are shown in Table 4a and 4b respectively.

Table 4. Characterization of brine generated from a) Ion exchange for Site I and b) Reverse osmosis for Site II (Zero Brine, 2017)

(a)

Materials	Concentration (mg/L)
Sodium (Na^+)	4000
Magnesium (Mg^{2+})	1000
Calcium (Ca^{2+})	8000
Chloride (Cl^-)	23236

(b)

Materials	Concentration (mg/L)
Sodium (Na^+)	1000
Bicarbonate (HCO^{3-})	1067
Chloride (Cl^-)	600
Silica (H_4SiO_4)	52
TOC (Total Organic Carbon)	16.5
Sulfate (SO_4^{2-})	387

2.2.2. Site I

Site I offers a design to face the spent regenerant disposal challenge in the DWP of Evides. The set-up of Site I is displayed in Figure 3. The spent regenerant from the IEX columns passes through a NF, which separates the spent regenerant into a permeate and a concentrate. NF permeate contains mainly water and monovalent ions such as Na^+ and Cl^-. While the NF permeate streams directly toward an evaporator, the NF concentrate passes through two crystallization units wherein the magnesium and calcium are removed from the concentrate. The evaporation unit is the last step of Site I, which receives the monovalent rich NF permeate and the effluent of the crystallization unit. The effluent of the evaporator consists of purified water and a flow with a high NaCl concentration.

The regeneration of the resins is taking place every 18 hours, which leads to an IEX brine flow of 106 m^3/day (per unit, in total 8 units). The demo plant has the ability to treat 24 m^3/day, hence accounting for 5% of the total brine flow.

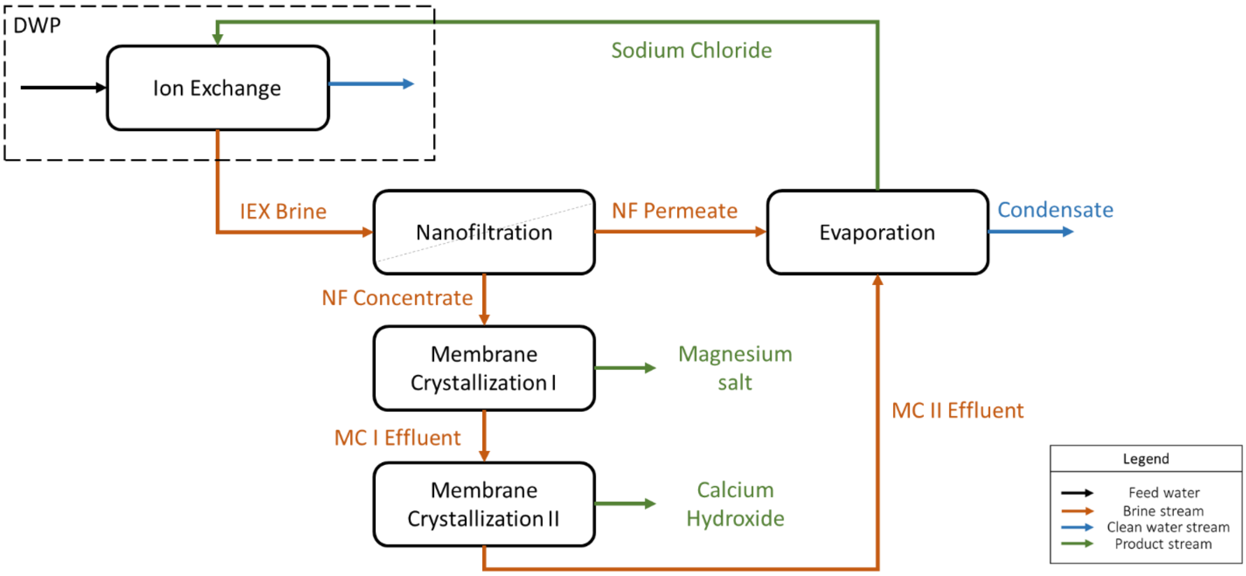

Figure 3. Process flow diagram for Site I

2.2.2 Site II

In the DWP, the IEX effluent (softened water) is used as the feed water for the RO unit. The concentrate stream produced in the last stage of RO contains salts and organic matters and is discharged to the sea in the current DWP. Although the concentration of salts and organics in the RO concentrate stream is lower than that of seawater, it could have negative environmental effects on the sea ecosystem due to the use of chemicals added in the RO pretreatment. These added chemicals are rejected by the membrane and therefore will be present at a higher concentration, than the concentration they are added, in the concentrate stream.

Site II is an innovative design that aims to treat the RO concentrate of DWP. To this aim, an anionic IEX is used to remove the anions and charged organic matters from the RO concentrate. The spent regenerant of anionic IEX is fed to the Total Organic Carbon (TOC) removal unit to remove the organics and then to a NF unit. The NF concentrate contains a high concentration of multivalent anions such as sulfate and monovalent ions and thus is sent to a Eutectic Freeze Crystallization (EFC) unit, wherein the concentrate is separated into ice and salts. The effluent of the anionic IEX column passes an RO unit, which is operated with a recovery of around 85%. The permeate of the RO unit can be used as the process water and the concentrate is sent to the evaporator, wherein it will be separated into pure water (condensate) and salts. Monovalent-rich (mostly NaCl) permeate of NF could be mixed with the condensate of the evaporator and be used for regeneration of the anionic IEX column.

The current RO unit of DWP is producing 250 m^3/h of brine effluent. Site II will have the ability to treat about 1 m^3/h of the brine, thus accounting for 0.4% of the total brine flow. The evaporator unit that will be demonstrated in Site I will be used also in Site II.

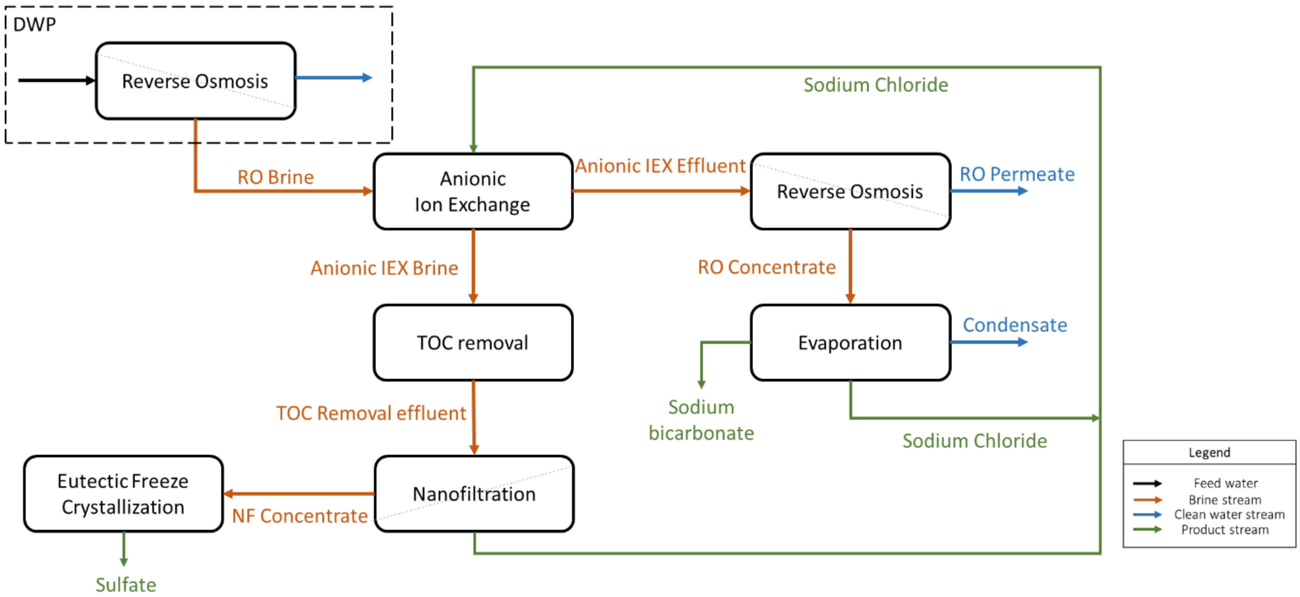

Figure 4. Process flow diagram for Site II

2.2.3 Zero Brine Technologies

In section 2.1, the technologies that are used for the treatment of brine were reviewed. However, in this section, the technologies that are employed in the Zero Brine project will be analyzed in more detail.

2.2.3.1 Anionic Ion Exchange

Anionic IEX is a purification process by which soluble negatively charged (anionic) ions, or contaminants, are separated from a solution by being exchanged with another negatively charged ion. Such negatively charged ions or contaminants may be sulfates (SO_4^{2-}) or TOC or naturally occurring organic matter, which is typically negatively charged, as in the case of the RO brine treated at Site II.

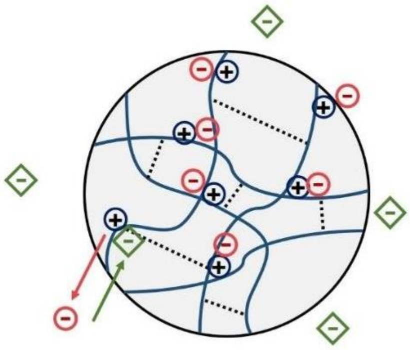

Figure 5. Anion exchange resin bead ("Lennetech", 2017)

The exchange media, or ion exchanger, is usually a porous resin bead with functional groups. Inside these beads, there exists invisible water that is calculated either as "humidity" or "moisture content". The resin is structured as a polymer whereon a fixed charged ion is attached. This ion is fixed and thus it cannot be removed or displaced. In order to maintain the electrical neutrality, every ion within the bead has to be neutralized with an anion outside the bead. This is what is called IEX. Figure 5 demonstrates how an anion exchange resin bead resembles. The polymeric skeleton of the bead is depicted with the blue curved lines (Zero Brine, 2018).

In industry, resins beads are utilized in columns. The treatment of the desired solution occurs through its flow in the resin. In Figure 6, the fresh resin is shown, which becomes gradually loaded with feed solution's ions. Eventually, the treatment stops as the pure solution is leaked with the "orange" ions to be released. As soon as the resins are depleted, it is possible to bring them back to the clean state and begin again. However, during the regeneration step brine is produced, which is the principal disadvantage of ion exchange ("Lennetech", 2017).

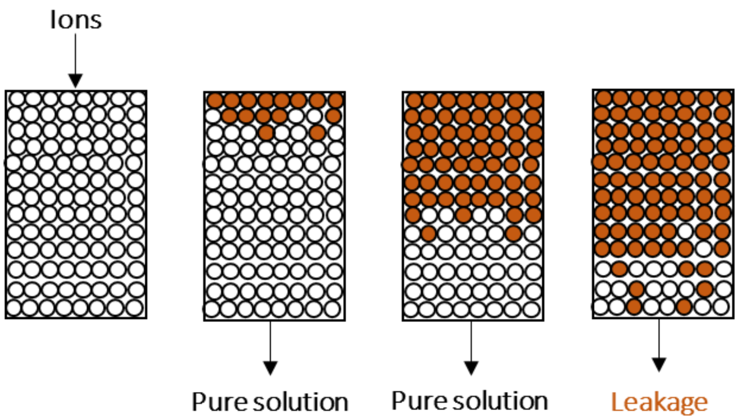

Figure 6. Column operation ("Lennetech", 2017)

2.2.3.2 Evaporation

The evaporation device, as depicted in Figure 7, is comprised of two sequential effects that function at decreasing levels of pressure. The brine effluent is vaporized in both the evaporator effects and subsequently two flows are generated. Firstly, a water vapor stream, which is afterward concentrated and recovered as clean water and secondly, a stream with a high concentration of brine, which is subjected to further treatment. Regarding the water vapor stream that is derived from the first effect, it is employed to heat the brine that is generated. This heating steam is used to evaporate the brine in the second effect and thus energy recovery is accomplished. The water vapor stream generated by the second effect is intended to be utilized for pre-heating objectives. Its heat energy is transferred to brine feed and hence the recovery of clean water and thermal energy is achieved to the greatest extent feasible (Xevgenos, Moustakas, Malamis & Loizidou, 2016).

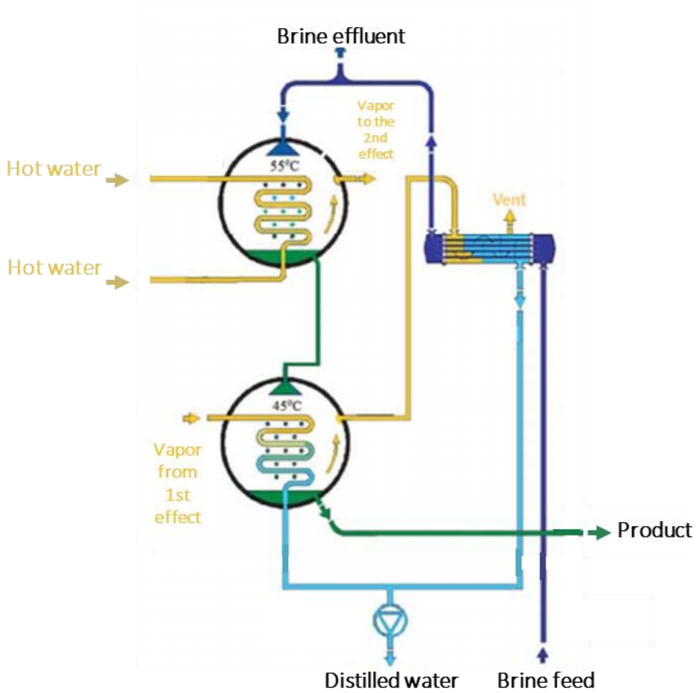

Figure 7. Process flow diagram of evaporation unit ("Solbrine", 2017)

2.2.3.3 Eutectic Freeze Crystallization

EFC principle is based on the eutectic point. Concerning the phase diagram of a salt and water system, the eutectic point constitutes a typical point. In a binary solution, the coexistence of three phases is taking place at the eutectic point, namely saturated solution, salt, and ice. Through decreasing the temperature of brine to the eutectic point temperature, the crystallization of salt and ice occurs. As a result, the separation of these two is possible owing to the different densities, specifically the salt will go down and the ice will be uplifted in the top of the crystallizer (Fernández-Torres, Ruiz-Beviá, Rodríguez-Pascual & Von Blottnitz, 2012). Figure 8 outlines an EFC in a simplified form for a binary system. It is noteworthy that EFC might be employed to solutions that comprise several ions such as wastewater. In the case of this situation, multiple EFC units are organized sequentially.

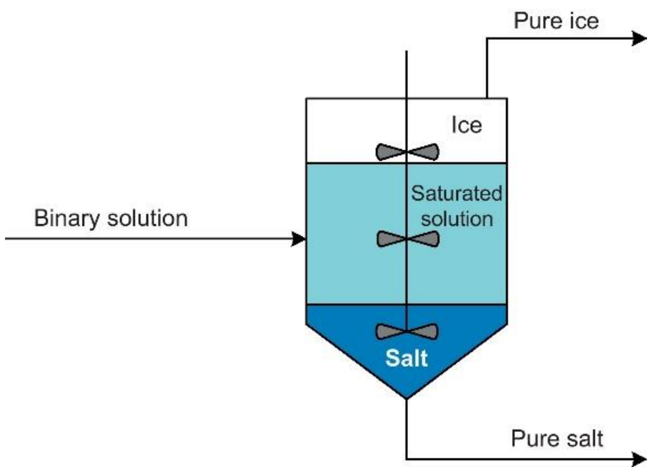

Figure 8. Eutectic freeze crystallizer (Fernández-Torres et al., 2012)

2.2.3.4 Nanofiltration

NF is a membrane filtration process, similar to the industrially applied reverse osmosis, however with a coarser membrane. That is, the pores of the membranes are wider than RO, allowing hydrated monovalent ions to pass through it. NF is often applied to remove multivalent ions such as calcium and magnesium, to soften water, or for the removal of heavy metals. Therefore, NF membranes can be used to separate calcium (Ca^{2+}), magnesium (Mg^{2+}) and sulfate (SO_4^{2-}) ions from a sodium chloride rich and purified stream (Zero Brine, 2018)

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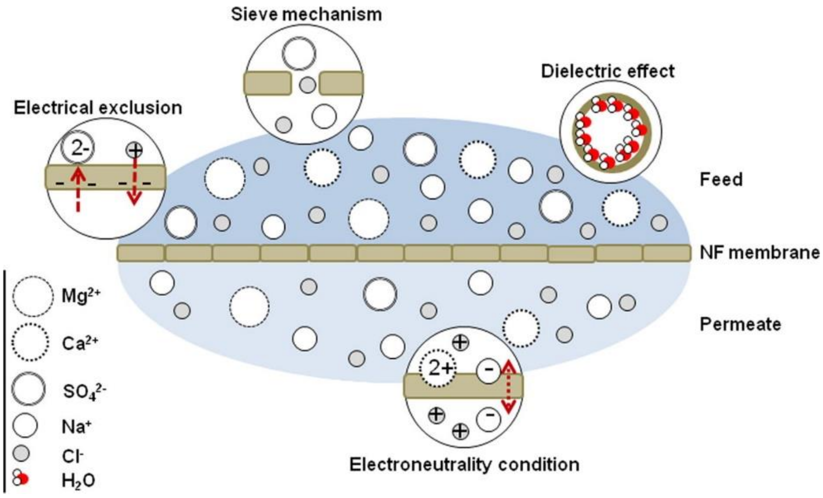

Figure 9. Rejection mechanisms of NF membranes (Zero Brine, 2018)

For the minimization of fouling, NF needs a stream tangential to the membrane. Therefore, a feed stream will be separated into a permeate stream with mostly monovalent ions and a concentrate stream with mostly multivalent ions. NF membranes usually attain a multivalent ion rejection between 75-99% and monovalent ion rejection between 30-50%, depending on the chemistry of the membrane active layer. As the separation is not perfect, both permeate stream will contain multivalent ions and concentrate stream will contain monovalent ions. Therefore it is important to consider the design of the NF on a system level, as well as the process parameters such as pH and temperature and solution chemistry, which can influence the passage of mono- and multivalent ions through the membranes. Furthermore, the interaction of different concentrations of ions in the feed will have an effect on the passage of specific ions. NF membrane performances, in fact, are described using three adaptable parameters: average pore radius, volumetric charge density and effective membrane thickness (Nicolini, Borges & Ferraz, 2016). These mechanisms are displayed in Figure 9.

2.2.3.5 Membrane Crystallization

The Crystallization physical phenomenon includes the formation of particles in vapors, the solidification of melted mixtures or the species precipitation in aqueous solutions. In particular, reactive crystallization can be used to produce an insoluble salt by the reaction of two ions when the species are dissolved in an electrolyte solution. In Site I, this is the case of magnesium hydroxide precipitation from electrolytic solution by means of a reaction with an alkaline as sodium hydroxide. In particular, for the Zero Brine project, a Multiple Feed – Plug Flow Reactor (MF-PFR) is employed, as shown in Figure 10. A PFR is a process in which, ideally, the solution containing the reactants moves with a piston flow, i.e. with a speed without radial gradients; the reaction occurs along the entire length of the reactor and at the steady-state, is not a function of time but only of space. Passing from ideality to reality, the piston flow hypothesis becomes only an approximation. This traditional reactor was slightly modified introducing more than one feed for the alkaline solution, in order to have a better supersaturation distribution along the reactor (Zero Brine, 2018). This configuration is called “Multiple Feed – Plug Flow Reactor”.

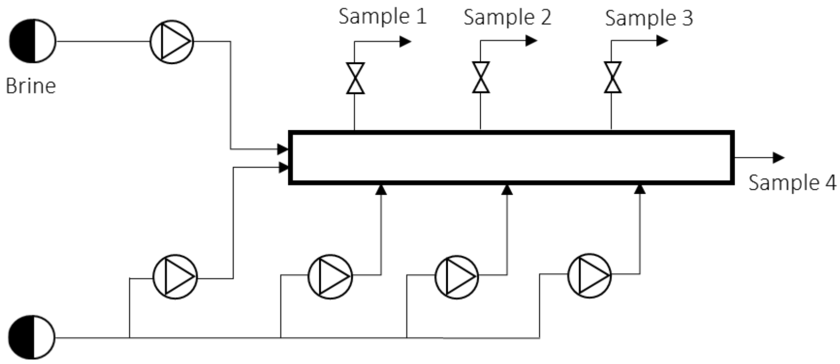

Figure 10. Simplified scheme of MF-PFR (Zero Brine, 2018)

2.2.3.6 Reverse Osmosis

As it is suggested by its name, RO is the contrary of osmosis. During osmosis, the osmotic pressure that occurs naturally moves the solution unpromptedly across a semi-permeable membrane towards higher solute ion concentration. In this way, the ion concentration or the chemical potential are balanced at both sides of the membrane. As displayed in Figure 11, this operation is continued so that the osmotic pressure and the pressure of the high solute ion concentration to be equalized. Consequently, during RO, pressure is applied to exceed the osmotic pressure and as a result, the solution is demineralized or deionized while it is pushed across the semi-permeable membrane (Zero Brine, 2018).

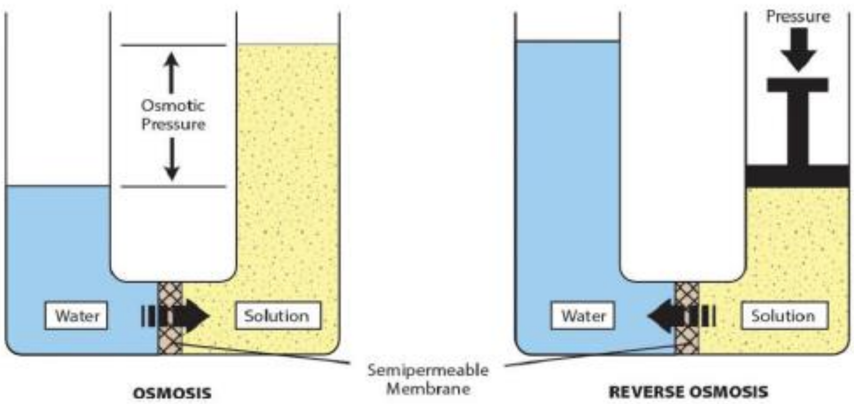

Figure 11. Principle of operation of osmosis and reverse osmosis (Zero Brine, 2018)

The semi-permeable RO membrane is typically made of ion chain polymer molecules with functional groups that interact with water allowing water molecules to diffuse through it, however, most contaminants in the water do not have the same ability and are therefore left behind. The feed water has a tangential flow to the RO membrane. Most of the water follows the membrane surface while the remaining get through the membrane. Therefore, two streams are derived, the concentrate that has a high concentration of small particles and dissolved ions and the permeate, clean water with a low concentration of ions. The recovery of permeate from feed water is desirably kept as high as possible, however increasing this would increase the concentration of ions on the concentrate side of the membrane and this would require increasingly higher pressures, and thus energy, to drive the process, as well as making the membranes more subject to fouling and scaling, shortening their lifetime.

2.2.3.7 Total Organic Carbon Removal

For the removal of TOC, an absorbent concentrates organic contaminants on its surface and regenerates while they are electrochemically oxidized. Arvia's proprietary adsorbent material, Nyex, removes the organics without generating sludge or secondary by-products ("Our Technology Nyex Treatment Systems", n.d.). Figure 12 shows the process of TOC removal. The contaminated water to be treated is injected into the adsorption zones and air is injected at the bottom of the cell in order to fluidize the adsorbent which is mixed with the incoming effluent. At the top of the adsorption zones, the treated water and loaded adsorbent flow into the settlement and regeneration zone, where a bed is formed after the rapid settlement of the adsorbent. The treated water overflows at the top of the unit. The adsorbent bed moves slowly downwards due to gravity and a direct current is applied across the bed. Once the adsorbent reaches the bottom of the electrochemical cell it is fully regenerated by anodic oxidation and ready for reuse.

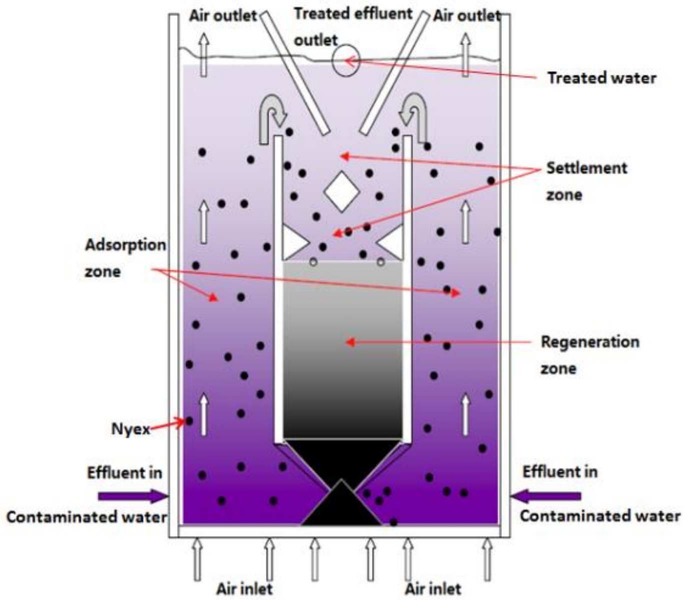

Figure 12. Arvia's continuous adsorption and electrochemical regeneration process (Liu, 2015)

2.2.4 Recovery of resources

The ZB project is demonstrating new ways of raw material production through the recovery of resources from brines generated by the chemical process industry. These resources include water, energy (through waste heat recovery), minerals, magnesium, sodium chloride, carbonates, and other salts. It is noteworthy that certain of these materials are currently imported to the EU. This is particularly relevant for magnesium, identified as Critical Raw Material, as 96% of magnesium is imported and 86% of this amount is imported from China (Zero Brine, 2017). Brine effluents contain also other materials that were identified as economically important but not included among the critical raw materials such as industrial minerals such as sodium chloride and calcium. These are used in numerous applications, with chemical industry accounting for the largest share, indicatively the chemical industry accounts for 62% of NaCl consumption in Europe.

2.2.4.1 Water

Water is the most used carrier of materials and, at the same time, a precious resource itself. However, water becomes more and more a scarce resource as a result of urbanization and increased competition between various users and economic sectors. The large-scale demonstration of Zero Brine in Evides will increase the water recovery from the 30-70% range currently to approximately 100% (Zero Brine, 2017). Thus in Zero Brine project water is addressed throughout its value chain i.e. as a resource, as a productive input and as a waste stream.

2.2.4.2 Raw Materials

The importance of improving resource recovery as well as decreasing import dependency has been recognized by the EU in the Communication on CE and to this end, the market of secondary raw materials has been recognized by the European Commission as a particularly important market. In DWP, the chemicals needed to regenerate the softening units will be recovered from the brine effluent cutting down the input of the raw materials by more than 70% (Zero Brine, 2017).

As already mentioned, the recovery of magnesium is targeted. In order to produce magnesium hydroxide, currently, either brucite or magnesite is used as raw material. Most of the main suppliers of these raw materials comprise non-EU enterprises. All the European manufacturers of magnesium hydroxide except one are being supplied with their raw materials from non-EU suppliers mentioned above. This reduces drastically the competitiveness of their products and their dependence on the raw material. By sourcing the raw materials from the industry brine effluents, not only sustainability is achieved, but also business opportunities across sectors and significant improvement of the competitiveness of this key sector for the EU, with the potential to cut down costs by approximately 64%.

Moreover, until today, Evides uses a 9% w/w solution of NaCl to regenerate the softening stage used as a first step to treat lake water into high-quality demi water. The amount of water produced, approximately 33,000 m^3/day covers the needs of the industrial cluster of Rotterdam Port. The salt used for the regeneration is up to 2,000 tons/year and is produced approximately 300 km away from the Botlek area through solution mining. Through the application of ZB systems, the salt consumption for regeneration is expected to reduce up to 75%, namely 2,000 tons of NaCl. As a result, the cost of purchasing the NaCl will also be reduced (Zero Brine, 2017).

2.2.4.3 Energy

According to SPIRE Roadmap, it is estimated that 20-50% of the energy used in industrial purposes is lost in the form of exhaust gases, cooling water and heat losses from equipment and products. ZB suggests ways of re-utilizing waste heat streams as a resource. In Rotterdam Port, the energy that goes wasted has been estimated at 5.5€ billion per year (Zero Brine, 2017). It is often that waste heat is under-exploited since the greatest change for mitigating energy usage lies through solutions that are most frequently cross-sectorial. At the same time, the main challenge for resource recovery from brine is that the concentration of valuables is often too low and the amount of water too high to justify recovery and reuse from an economic point of view.

2.3. Conclusions of Chapter 2

This chapter has established the required theoretical basis for this research. Firstly, a literature review about the brine effluent was performed to highlight the main challenges that this effluent imposes along with the treatment methods that have been established for mitigating them. It is concluded that even though numerous technologies have been employed for the treatment of brine effluent, they still subject to improvements in terms of environmental and economic efficiency. Furthermore, the recovery of valuable resources from brine discharges proved to be an advanced solution to the problem of brine disposal both environmentally and economically. Through the abovementioned, since brine effluent is a main underlying theme of this research, the basis for the importance and the relevance of the ZB case study was established. Following, a description of the ZB case study was provided. It was highlighted how the implementation of the ZB applications, namely Site I & II, will be able to treat the brine effluent derived from DWP by eliminating it and at the same time recovering valuable resources. Finally, the contribution of the recovered resources, namely water, raw materials, and energy is presented particularly for the ZB case study pointing out the added value that the ZB project could bring and providing the breeding ground for the environmental and economic analysis.

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Chapter 3

Life Cycle Assessment and Life Cycle Costing

In this chapter, a thorough literature review is carried out with the purpose of understanding the basic notions employed in this report. The key theories, principles, and applications of both LCA and LCC are reviewed in Sections 3.1 and 3.2, respectively. Specifically, the structure of the LCA is described according to ISO 14040 and 14044 standards. Moreover, regarding the LCC, the cLCC, eLCC and sLCC are reviewed and their key differences are identified. Following in Section 3.3, ways of integrating LCA and LCC are presented. In Section 3.4., the methods and approaches for monetizing the external costs are also examined. Finally, Section 3.4 reviews the critical aspects found in the literature.

3.1. Life Cycle Assessment

LCA is a technique used to assess the overall environmental impacts of products and services across the life cycle of a product, namely from extraction, manufacturing, operating to end use (Guinée, 2002). LCA method has been standardized by the ISO which gives the following definition “LCA is compilation and evaluation of the inputs, outputs and potential environmental impacts of a product system throughout its lifecycle" (ISO, 2006).

Generally, LCA can be distinguished in two types; consequential and attributional. On the one hand, consequential LCA (CLCA) aims to depict in which way environmentally-related flows will alter, in reply to possible decisions. On the other hand, attributional LCA (ALCA) is focused on environmentally related physical flows which are inputs or outputs to a system (Heimersson, Svanström & Ekvall, 2019). These two types have two main differences. Primarily, CLCA includes activities that are beyond the system boundaries and are anticipated to be influenced by variation in demand. Secondly, CLCA uses the system expansion method by avoiding allocation that is used in ALCA. During system expansion, the avoided products are defined and thus the equivalent environmental effected is not assessed (Heijungs & Guinée, 2015). It becomes apparent that though CLCA is more complicated as it incorporates more concepts, it seems to be better than ALCA. However, ALCA could be adequately employed for the identification of hotspots and the comparison of different systems. ALCA was a more viable option in the context of this report due to time constraints.

3.1.1. Structure

It is widely recognized that LCA consists of the four following phases as outlined in Figure 13; goal and scope definition, inventory analysis, impact assessment, and interpretation.

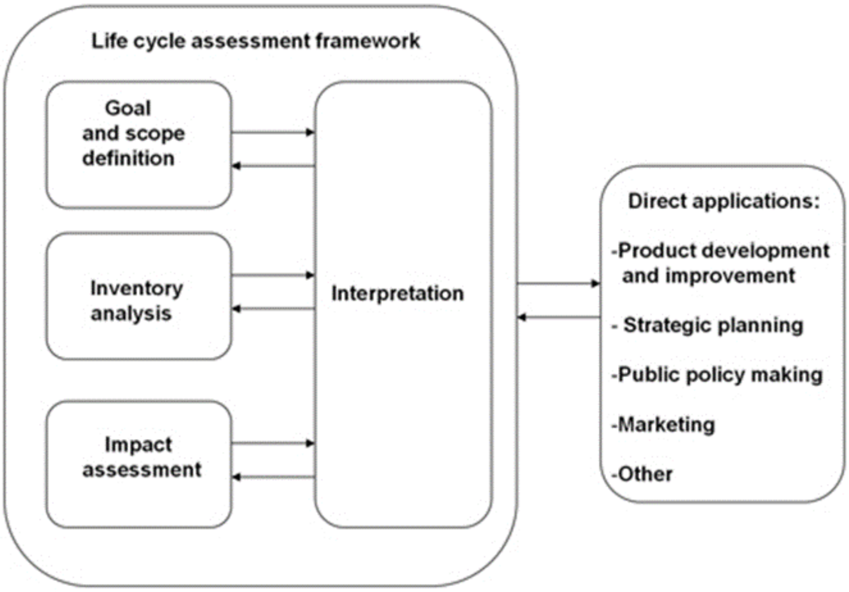

Figure 13. Life Cycle assessment framework – The four phases of an LCA (ISO, 1997)

3.1.1.1. Goal and Scope Definition

The main purpose of this phase is the identification of the entire structure of the LCA. Regarding the goal definition, the objective of the research, the intended use and the target group are mentioned. Furthermore, the scope of the study is stated with regard to temporal, geographical and technology coverage along with the system boundaries. The temporal and geographical coverage demonstrate the aptness of the gathered data with respect to time and geography, respectively. The technology coverage takes into consideration the addressed technologies and process stages. Lastly, the reference flow, the function, and the functional unit are identified. The reference flow measures the outcome of the system under study, the function is the operation of the system and the functional unit is the quantified definition of the function of a product that evaluates the system's performance (Heijungs & Guinée, 2015).

3.1.1.2. Life Cycle Inventory (LCI)

During this phase, the required data are collected to set up the inventory list and hence build the system in the SimaPro software ("SimaPro", 2019). The collected data concern materials, technologies, transportation, and power consumption. Accordingly, the elementary flows, the inflows, and outflows of the various processes to the environment from raw materials' production to the waste disposal are included in the set of data in the inventory table (Heijungs & Guinée, 2015). The referred data are collected from organizations, public institutions, literature or from databases included in the software. Subsequently, the inventory list is modeled by choosing the relevant unit processes from the databases. Overall, the environmental emissions are identified and quantified, compiling an inventory. LCA requires a considerable quantity of data, thus the quality of data is a critical phase. Multifunctionality is defined as a process that performs more than one function thus producing more than one product. Concerning multifunctional processes, allocation is applied. Various methods of allocation exist, namely mass, economic, energy, etc. Generally, LCI requires a great deal of time and effort.

3.1.1.3. Life Cycle Impact Assessment (LCIA)

LCIA translates the results from the inventory analysis to environmental impacts. This phase is conducted in five steps; selection, classification, characterization, normalization, and weighting (Heijungs & Guinée, 2015), as detailed below. Normalization and weighting are optional steps.

i. Selection

In this step, the impact categories, the category indicators as well as the evaluation models are selected. The impact categories concern the depletion of resources, the impact on human health along with the impact on ecosystems. Concerning these categories, several models are accessible such as CML-IA baseline (CML), ReCipe, etc. Additionally, the category indicators such as global warming potential, eutrophication, acidification, etc. are chosen based on the goal and the scope of the research.

ii. Classification

During classification, the elementary flows identified are allocated to the associated impact categories. This step is executed directly by SimaPro software thus no further details are given.

iii. Characterization

The values derived from the classification are converted to a common indicator score by multiplying them with a characterization factor. For instance, the characterization factors of CO_2 and CH_4 could be 1 and 25 respectively. If during the process 1kg of CH_4 is utilized, this will have as result 25kg of CO_2. The characterization factors differ with respect to the evaluation methods utilized.

iv. Normalization

Normalization helps the analysis by checking the discrepancies and the relevance of the impact results. During this step, the obtained results are compared to reference data. This data could be based on a community, person or other systems over a given period of time.

v. Weighting

Weighting is performed to evaluate the results within impact categories. Weighting factors are multiplied with the acquired results. These factors relying on the relevant significant among the various impact categories and on value choices.

3.1.1.4. Interpretation

The last phase is the interpretation of the results, which provide an evaluation of the environmental effects. The assumptions, as well as the uncertainties, are analyzed. Accordingly, this will result in the overall conclusions. A comparative analysis could be made among other systems and recommendations could be given.

3.2. Life Cycle Costing

According to ISO 15686, traditional LCC is a technique that provides the opportunity of assessing the costs related to a product over a certain period, considering all the appropriate economic factors with regard to initial and future operational costs (ISO, 2008). According to literature, LCC is also known under various other terms such as Full Cost Accounting (FCA), Total Cost of Ownership (TCO) and Total Cost Assessment (TCA) (Klöpffer, 2003, Hunkeler & Rebitzer, 2003). However, owing to the lack of recognized standards, the disparities between these terms continue to be a subjective opinion built on experience, field of research and economic perspective (Rödger, Kjær & Pagoropoulos, 2018).

LCC is a multipurpose tool that could be employed for a wide array of applications and at various phases of the life cycle in order to assist decision – making. It could be employed for an absolute as well as a relative analysis, namely to help the budgetary process or to assess alternative technologies, respectively. Furthermore, LCC could be employed as a tool for planning, optimization, hotspot identification or as part of a life cycle sustainability assessment of a specific product, or to evaluate investment decisions. The conceptual framework of the LCC is shown in Figure 14.

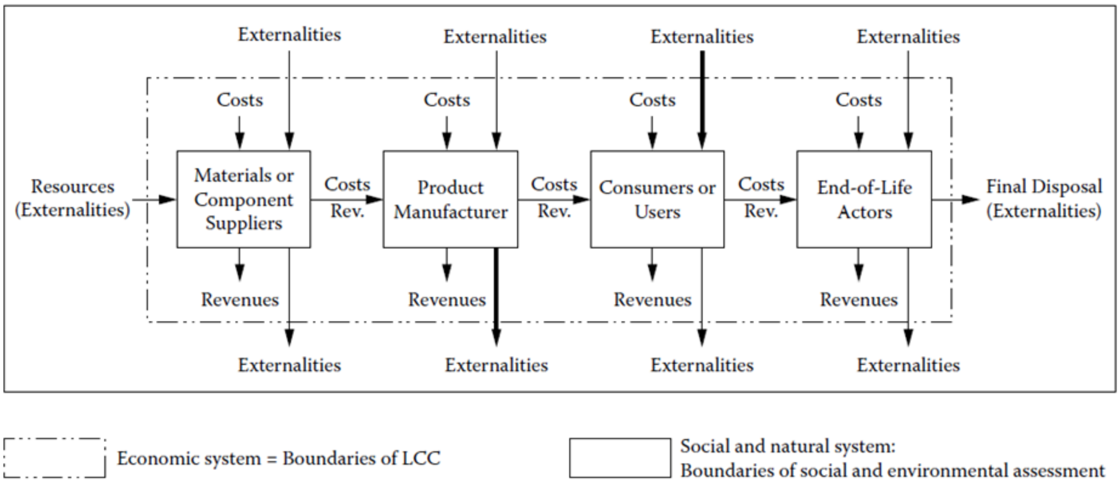

Figure 14. Conceptual framework of LCC (Hunkeler et al., 2008)

Apart from traditional LCC, the SETAC European working group has distinguished LCC into three variants; conventional, environmental and societal (Hunkeler et al., 2008, Swarr et al. 2011). In Figure 15, the main differences between cLCC, eLCC and sLCC are shown.

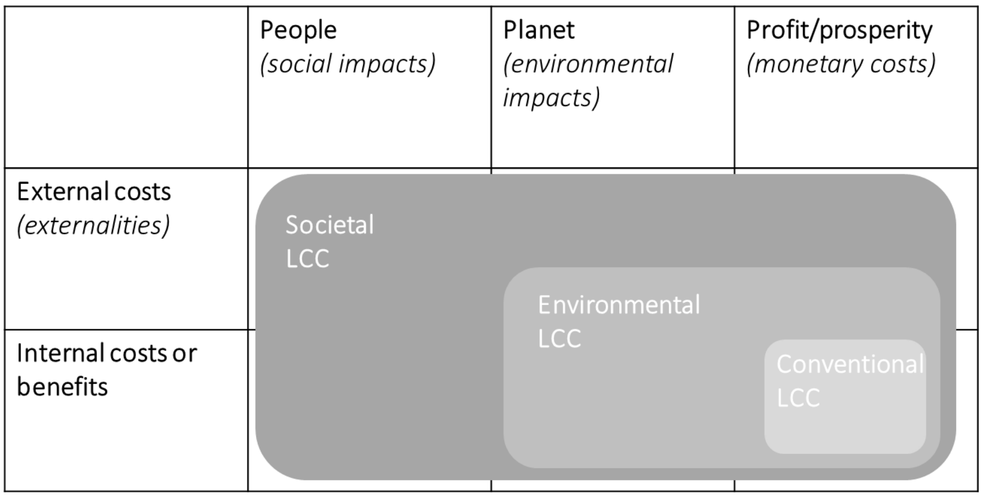

Figure 15. Main differences between the three types of LCC (Rödger, Kjær & Pagoropoulos, 2018)

3.2.1. Conventional

cLCC, also known as financial LCC, comprises to a great extent the traditional and common practice of organizations and governmental authorities. It is founded on strictly economic analysis, taking into consideration many phases of a life cycle of a product or a system. cLCC is a quasi – dynamic method and usually incorporates product's or service's costs which are covered by a specific actor and is generally introduced from the point of view of the manufacturer or end-user alone. The focal point of the analysis is real, internal costs with external costs to be ignored. However, it is often that cLCC does not take into account the whole life cycle; excluding some stages such as End-of-Life (EoL) operations. Hence, the scope's comprehensiveness of cLCC is inferior to systematic environmental analyses like LCA. For example, from a manufacturer's perspective, the costs related to the production of a passenger car, such as raw materials and transportation costs will be included in cLCC.

3.2.2. Environmental

eLCC was established to assist LCA in the sense of economic dimension's coverage while identifying hot-spots concerning both cost and environmental effects. Contrary to cLCC, eLCC takes the viewpoint of a functional unit rather than of a single actor, as it is aligned with LCA based on ISO 14040 and 14044 standards. LCA and eLCC could be seen as complementary in the sense all costs are included as directly borne throughout the life cycle. Moreover, eLCC is a steady-state analysis and considers the complete life cycle of a system or a product by incorporating all the actors in the life cycle. Considering costs, except for internal costs eLCC also incorporates external costs that are supposed to be internalized in the immediate future. In regards to the abovementioned example, the passenger car, the anticipated taxes on pollution from fuel usage could be incorporated in the operating costs of eLCC. However, a particular focus should be given to prevent double-counting, which is the inclusion of external costs from environmental impacts that are already calculated in LCA. eLCC goes a step further from cLCC by incorporating all the costs borne over the life cycle, both the internal costs and costs that are expected to be internalized within a short period of time.

3.2.3. Societal

The main objective of sLCC is to assist decision-making processes from a societal perspective incorporating governments and public agencies. It encapsulates externalities costs, namely internalized environmental and social impacts in monetary terms by allocating a monetary value on them (Martinez-Sanchez, Kromann, & Astrup, 2015; Rödger, Kjær & Pagoropoulos, 2018). sLCC includes all of the eLCC as well as further monetized environmental and social impacts, such as human well-being, job quality, etc. Consequently, sLCC goes beyond eLCC by taking into account all the external that could be monetized or even those that there is difficulty in monetizing them and might be regarded qualitatively. However, it is obvious that there are too many externalities that could be quantified. As is the case for cLCC, the method is quasi-dynamic. Compared to eLCC, subsidies and taxes have no net cost effect and hence are not included in sLCC.

Table 5. Comparison of the three types of LCC (Hunkeler et al., 2008; Rödger, Kjær & Pagoropoulos, 2018)

	cLCC	eLCC	sLCC
Objective	Assessment of the total life cycle costs borne by one main actor in the product's life cycle	Assessment of the total life cycle costs borne by all stakeholders related to product's life cycle	Assessment of the total life cycle costs borne by anyone in the society
Actors	Mainly one actor; either the manufacturer or the user or the consumer	Stakeholders related to product's life cycle;	Society
Life cycle	Economic lifetime, usually excluding EoL	Total life cycle	Total life cycle
Reference unit	Product or project	Functional unit	Functional unit
Types of costs	Internal costs of one stakeholder, focusing mainly on acquisition and ownership costs	Internal costs of stakeholders connected to the life cycle, plus external costs and benefits expected to be internalized in the near future	Internal costs of all actors plus external costs, i.e. impacts that production or consumption have on third parties
Cost model	Generally quasi-dynamic	Steady-state	Generally quasi-dynamic
Discounting of results	Consistent; discount factors between 5-10%	No. Discounting the results of the LCC would make the analysis inconsistent with the steady-state assumption of LCA	Consistent; usually low discount factors (<3%)
Discounting of cash flows for calculation	Recommended	Recommended	Recommended
Consistency with LCA	No	Yes, but with a risk of double counting the monetarized environmental impacts	No, due to risk of double counting and inconsistencies with the quasi-dynamic approach in sLCC
Standards	Multiple standards, including ISO 15663, IEC 60300-3-3, BS 3843, AS/NZS 4536, ISO 15686	None, but follows the LCA standards ISO 14040 /14044	Currently no standards

3.3. Integrating LCA and LCC

Notwithstanding the extensive literature research on LCA and LCC as standalone applications, integrating LCA and LCC remains an issue. Swarr et al. (2011) along with ISO (2006) have provided some recommendations on how to incorporate there two techniques into one analysis, as shown in Figure 16.

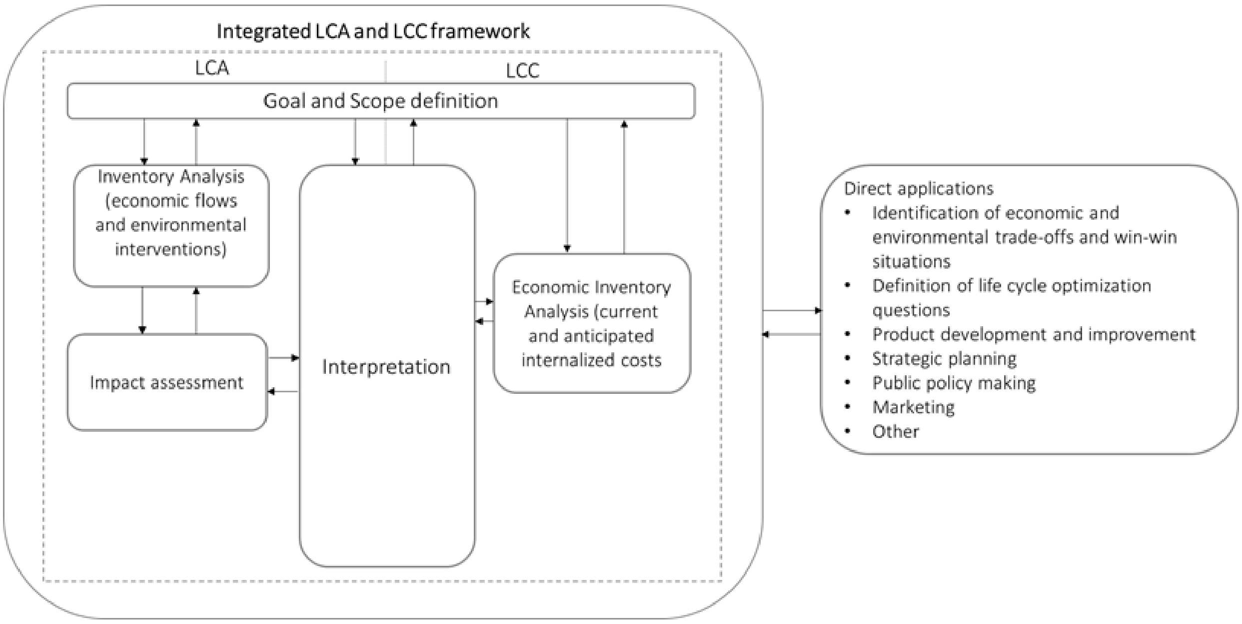

Figure 16. Integrated LCA and LCC framework (ISO, 2006; Swarr et al., 2011)

Nevertheless, the majority of the research concerning the integration of LCA and LCC is centered on one assessment technique. According to Bierer, Götze, Meynerts & Sygulla (2015), the studies are distinguished in the following three categories:

i. Parallel application of both LCA and LCC

The aim of this study's type is the merge of the outcomes of the two techniques. The system or product under study has similar system boundaries, time frames, and functional unit.

ii. Integration of cost aspects in LCA

During this type of study, the cost aspects are combined with LCA research without however to clearly establishing the required approach for the costing.

iii. LCC as leading concept

The underlying concepts of LCC are employed for the cost calculation of a system or a product. This kind of research is mainly focused on the economic side, disregarding the environmental burdens and hence resulting in inaccurate interpretation of the results.

Furthermore, the already designed methods for the integration of LCA and LCC are concerning particular industrial applications. The vast majority of the scientific work focus on the building, water and waste sectors (Petit-Boix et al., 2017). On the other hand, there is a limited number of literature concerning the chemical industry and the production systems (Auer, Bey & Schäfer, 2017). Regarding the chemical process industry, as already mentioned, there exists a gap in the literature concerning the analysis of water supply systems by employing both the LCA and LCC. In section 3.5, the literature findings are presented.

3.4. Monetization of external costs

3.4.1. Approaches and Methods

There some non-market goods that it is very difficult to assign to them an objective market price. Such goods or services could be clustered in the external costs group. Without the existence of economic value for them, it is essential to apply economic valuation methods to establish their value. eLCC, as well as sLCC, could utilize the monetary valuation methods to estimate the selected external costs. Even though that the monetization of external costs is beneficial for the LCC, it can also be the case for the LCA, since it enables comparisons across the analyzed impact categories, on midpoint level. With the purpose of monetizing the environmental impacts, several methods have been proposed as shown in Figure 17, while in Table 6 a brief explanation of the different approaches is given.

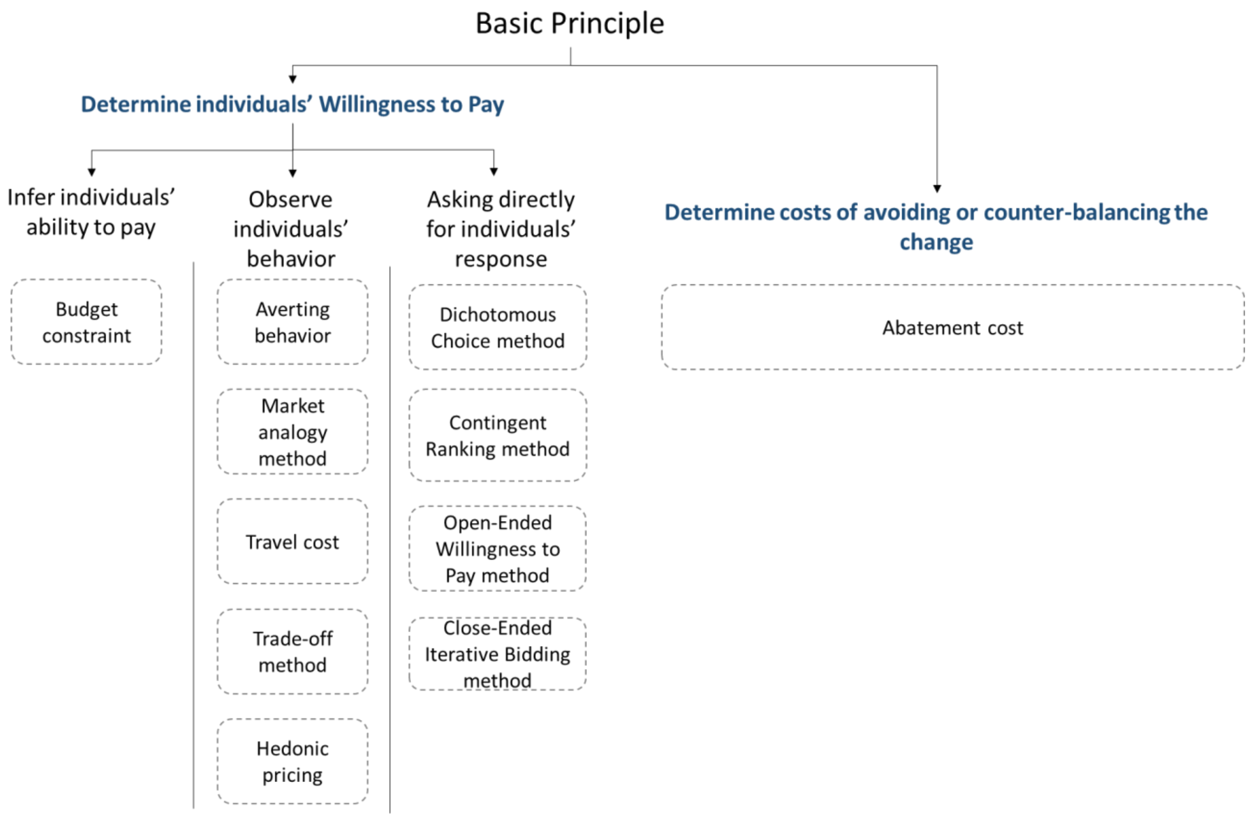

Figure 17. Determing costs – approaches and methods (Boardman et al., 2010)

The majority of the valuation methods attempt to identify the individuals' Willingness to Pay (WTP) for a specific advantage or conversely the individuals' Willingness to Accept (WTA) a compensation for a specific disadvantage. However, a different method assesses the external costs by determining the costs of avoiding or counter-balancing the change. There are various approaches to identify the WTP, with each approach to provide different and numerous methods.

Table 6. Determining costs—approaches' description (Pizzol et al. 2014; Boardman et al. 2010)

Approach	Description	Indicative examples	Main weakness	Methods
Determine costs of avoiding or counter-balancing the change	Equalize the value of external costs with the cost of actions required to mitigate it	Assess the cost of GHG emissions by evaluating the costs for decarbonization	Does not value utility losses, and hence does not express individuals' attitudes, but rather external targets	Abatement cost
Asking directly for individuals' response	Elicit individual's WTP for variations in the quality or quantity of a good	Inquire the WTP of several people for the protection of a national park	Limitations due to possible errors in the survey, i.e. sample's size and representativeness	Contingent ranking method, Close-ended iterative bidding, Dichotomous choice method, Open-ended willingness to pay method
Observe individuals' behavior	The value of a non-market good might be reflected in the substitute market for a relevant good	Evaluate the benefit of newer catalysts in cars by evaluating its impact on healthcare costs for respiratory diseases	Assumption that individuals take decisions fully informed; not the case in practice	Averting behavior, Hedonic pricing, Market analogy method, Trade-off method, Travel cost,
Infer individual's ability to pay	Determine individual's WTP for an additional year of life of absolute well-being	Cost of a statistical life	Applicable only to the value of human well-being	Budget constraint

However, due to the fact that there are many uncertainties engaged in the monetization of environmental and social impacts, the results might differ with regards to the applied methodology. There is the risk of trivializing a difficult issue by displaying a financial unit that could be considered as something very certain and definite. Market prices are factual in comparison to monetized environmental and social impacts that at all times are dependent on perceptions and value judgments. For this reason, methodological choices must always take into consideration the goal and scope of the analysis as well as the targeted audience.

3.4.2. LCIA Methods for monetization

Generally, economic weighting sets are employed to translate the environmental impacts derived from LCIA into monetary units (Huysegoms et al., 2018). Furthermore, it is argued that the obtained monetized environmental impacts could be used together with LCC to estimate the total external costs of a product or system (Ahlroth et al., 2011). On the basis of the methods and approaches described in the previous section, several LCIA methods have been established to convert the identified impacts to monetary values. Table 7 displays the found LCIA methods, their associated valuation method and the impact categories that could be monetized. The monetary weighting factors of each LCIA are presented in Appendix A.

Table 7. LCIA models for weighting environmental effects in monetary terms

LCIA Model	Valuation Method	Impact categories that could be monetized	Reference
EPS	Market Prices Contingent Valuation Abatement Costs	Human Health Ecosystems	Steen (1999b)
ReCiPe	Market Prices	Human Health Ecosystems	RIVM (2018)
Ecovalue08	Market Prices Contingent Valuation	Abiotic resources Acidification Global Warming Eutrophication	Ahlroth & Finnveden (2011)
Stepwise 2006	Budget Constraint	Human Health Ecosystems Abiotic resources Acidification Global Warming Eutrophication Ozone Depletion	Weidema (2009)
LIME	Choice Experiment	Human Health Ecosystems	Itsubo et al. (2004)
Ecotax02	Averting Behavior	Abiotic resources Acidification Global Warming Eutrophication Ozone Depletion	Eldh & Johansson (2006)

According to Diafonidis (2019) which examined all the above-mentioned weighting sets, it was concluded that the implemented weighting sets are incomplete due to the following reasons. Firstly, the identified weighting sets do not incorporate factors for the monetization of all LCA midpoint and endpoint indicators, signifying the inconsistency of these factors in terms of applying them in all impact categories. In addition, cultural, social and economic differences among the studies cause the variation of the results. Therefore, the generalizability and representativeness of the weighting factors are heavily influenced by their different socio-cultural orientation.

In order to overcome this problem, the abovementioned LCIA models will not be used in this study. As this report deals with the Dutch chemical process industry, it is sound to employ weighting factors that are meant for the Netherlands. Dutch Ministry of Infrastructure and Environment commissioned CE Delft to prepare the Environmental Prices Handbook (De Bruyn et al., 2018). This Handbook presents sets of environmental prices and weighting factors for use in the Netherlands as indices in economic and environmental analysis. Except from the location, the environmental prices presented in the Handbook are average values for emissions in 2015, compared to mentioned LCIA models that use rather old data for the emissions (see Appendix A). Consequently, this research is using the weighting factors provided from De Bruyn et al. (2018), that are especially suitable for usage in LCA according to the ReCiPe methodology, in order to monetize the external costs. The following procedure is presented in Chapter 4.

3.5. Literature findings

3.5.1. LCA

Concerning the use of LCA in the water industry, the number of LCA studies have sharply increased in the 1990s and 2000s. LCA applications in the aforementioned sector are mainly distinguished in two categories; wastewater treatment and water supply systems (Friedrich, Pillay & Buckley, 2010). Numerous researchers have performed the LCA of water supply systems, as shown in Table 8 (Sombekke et al., 1997, Raluy et al., 2005, Bonton et al, 2012,) or wastewater treatment plants, as displayed in Table 9 (Kalbar et al., 2012; Lorenzo-Toja et al., 2016). The abovementioned studies focused on the comparison of various drinking water (DW) supply systems and the technologies that are included in the analyzed system, especially membrane processes. Furthermore, LCA was also employed in many studies in order to investigate the solutions to alleviate the environmental burdens by decreasing the salinity of feed water (Muñoz & Fernández-Alba, 2008), by utilizing efficient pretreatment (Beery & Repke, 2010) or by engaging cleaner energy sources (El-Nashar, 2001). However, the majority of the prior scientific papers only emphasized on the planning and operational stages to tackle the energy shortage concern (Zhou, Chang & Fane, 2013). Several LCA studies assumed that the brine was completely diluted before the disposal and caused insignificant impacts on the aquatic ecosystem (Raluy, Serra & Uche, 2005; Raluy, Serra, Uche & Valero, 2006; Zhou, Chang & Fane, 2011). Not including the impacts resulted from brine disposal process might lead to biased results and actually contradicts the ‘cradle-to-grave’ nature of LCA.

Table 8. LCA studies on water production

Study Objective	Functional Unit	Result	Reference
Conventional treatment vs nanofiltration	1 m^3 of DW	Minor differences between treatment methods; significant impacts of Granular Activated Carbon (GAC) and energy	Sombekke et al. (1997)
Conventional treatment vs RO	1 m^3 of DW	Minor differences between treatment methods; significant impacts of GAC, chemical and conventional energy	Mohapatra, Siebel, Gijzen, Van der Hoek, & Groot (2002)
Desalination vs big hydraulic infrastructure	25 000 hm^3 of DW	Slightly higher impacts for desalination; significant impacts of energy; minor impacts of construction	Raluy et al. (2005)
Conventional treatment vs ultrafiltration	1 m^3 of DW	Significant impacts of energy (80%); minor impacts of construction (< 15%)	Friedrich et al. (2010)

Conventional treatment vs nanofiltration	1 m^3 of DW	Greater impacts for conventional system; significant impacts of GAC and chemicals	Bonton et al. (2012)

Table 9. LCA studies on wastewater treatment

Study Objective	Functional Unit	Result	Reference
Evaluation of 13 WWTPs	1 m^3 of water	Eutrophication and terrestrial ecotoxicity were identified as the main impact categories	Gallego, Hospido, Moreira & Feijoo, (2008)
Comparison of different scenarios for wastewater reuse	1m^3 water for irrigation in agriculture	None of the examined scenarios is simultaneously the best choice under ecotoxicity, human toxicity, and global warming potential.	Muñoz & Fernández-Alba (2008)
Comparison of SWRO pre-treatment methods	1 m^3 of water	Membrane pre-treatment is somewhat less preferable regarding the environmental and societal aspects, due to its higher energy	Beery & Repke (2010)
Assessment of different brine final disposal alternatives from a desalination plant	1 m^3 of DW	No universal solution. • Direct disposal when no sensitive species are present • Diluting brine with WWTP when a WWTP is located • Brine dilution with when sensitive species are present.	Meneses, Pasqualino, Céspedes-Sánchez & Castells (2010)
Comparison of EFC vs EC for the treatment of saline water	40 ton of Na_2SO_4 90 ton of ice & liquid water	EFC is preferred to EC as it uses 6–7 times less non-renewable energy	Fernández-Torres, Randall, Melamu & Von Blottnitz (2012)

Nevertheless, from the above-mentioned studies, only Lorenzo-Toja et al. (2016) have included the LCC into their analysis in order to assess the economic feasibility of 22 wastewater treatment plants in Spain. It is remarkable that only a few studies concerning water industry have incorporated both the LCA and LCC and they are assessing only wastewater treatment systems (Lim, Park & Park, 2008; Kalbar, Karmakar & Asolekar, 2012; Kalbar, Karmakar & Asolekar, 2016; Resende, Nolasco & Pacca, 2019).

3.5.2. LCC: conventional, environmental, societal

As stated above, the number of publications regarding the incorporation of both LCA and LCC in one study is very limited in the chemical process industry. Nevertheless, these studies are included only the cLCC, without taking into consideration the monetized environmental and social impacts. Heretofore, there is a lack of consensus on which is the most useful and sound way for the alignment of LCA and LCC analysis with the purpose of converting the detected environmental impacts into external costs (Ciroth et al., 2011; Neugebauer et al., 2016). To tackle this issue, several tools that support decision-making processes have been proposed to monetize the environmental impacts derived from LCA, namely cost-benefit and eco-efficiency analysis along with economic evaluation methods (Reich, 2005; Hunkeler et al., 2008 Rödger, Kjær & Pagoropoulos, 2018).

When eLCC is performed without sLCC, the valuation methods and their derived LCIA weighting models described in section 3.4 are applied to translate the LCA results into monetary terms. For example, Reich (2005) uses the EcoTax02 and EPS models and Huysegom et al. (2018) used Stepwise and Ecovalue08. The monetized results are presented as eLCC. However, it is remarkable that in the studies that incorporate both eLCC and sLCC, the monetized LCA results are presented as sLCC. For example, Weldu (2018) used Stepwise for sLCC. Consequently, the approach of this research should be critically considered regarding the eLCC and sLCC.

Generally, the literature that incorporates all three types of LCC is severely restricted. The majority has incorporated sLCC as a part of a Life Cycle Sustainability Assessment (LCSA) (Muiña, González-Sánchez, Anna Maria Ferrari & Davide Settembre-Blundo, 2018) or combined with eLCC (Martinez-Sanchez, Tonini, Møller & Astrup, 2016; Edwards, Burn, Crossin & Othman, 2018). Only one paper concerning energy production has focused on sLCC as a stand-alone method (Weldu, 2018).

Martinez-Sanchez, Kromann, & Astrup (2015) provided a cost model for the economic evaluation of waste management systems. During this study, the costs were into three categories; budget costs, transfers, and externality costs. For implementing the three different types of LCC, budget costs were included in all three types, transfers in cLCC and eLCC, and externality costs only in sLCC. In cLCC and eLCC, budget costs were calculated in factor prices, while in sLCC were calculated in accounting prices (also called shadow prices). In order to convert factor prices into accounting prices, the Net Tax Factor (NTF) proposed by the Danish Ministry of Finances was implemented. eLCC and sLCC used the environmental impacts from the LCA in order to calculate the final results and LCA was presented as complementary to eLCC. Moreover, Martinez-Sanchez et al. (2016) and Edwards et al. (2018) followed the same rationale for the life cycle costing of food waste management systems, with the only difference to be that Edwards et al. did not use the NTF but gathered the required accounting prices from literature review.

Weldu (2018) applied sLCC as a stand-alone method. In order to calculate the sLCC, the results of Weldu & Assefa (2016) were utilized concerning climate change, human health and the ecosystem's impacts on the environment. Subsequently, these environmental impacts were translated to monetary terms by applying the Stepwise2006 model by Weidema (2009). Furthermore, Muiña, González-Sánchez, Anna Maria Ferrari & Davide Settembre-Blundo (2018) were implied in their study that sLCC is the sum the eLCC and cLCC without further elaborating on that.

Finally, Hunkeler et al. (2018) in their book about eLCC, have identified economic and social impacts that are relevant when performing a sLCC (see Appendix B). These impacts were examined in order to be incorporated into this study. However, the social impacts indicated were either beyond the scope of the ZB case study or their monetization was difficult. Concerning the relevant economic impacts, such as employment, they are not included in sLCC to avoid double counting since they are incorporated into cLCC.

3.5.3. Critical aspects of the existing literature

Primarily, there is a misunderstanding regarding what constitutes an eLCC and what an sLCC. Due to the immaturity of sLCC as well as to the absence of a standardized method to convert environmental damages into external costs, the monetized LCA results are presented as eLCC in some studies and as sLCC in others. sLCC has been characterized as a “welfare-economic’’ assessment (Hunkeler et al., 2008 Martinez-Sanchez, Kromann, & Astrup, 2015) as indicates how a trade-off between the welfare effects of market effects and non-market effects can be made. In welfare economics relative changes caused by environmental and social impacts are directly linked to the concept of external costs. Since the valuation methods that derive from WTP and WTA principles are based on welfare economics, it is sound that the conversion of environmental damages into external costs through these methods to constitute the sLCC and not the eLCC.

Secondly, the boundaries of the system in terms of environmental, economic and social assessments are not always in accordance. For instance, emissions concerning the production and disposal that are included in economic evaluations often neglected in LCAs studies (Carlsson Reich, 2005). Moreover, in welfare cost assessments, the scope of the studies usually concerns national geographical boundaries (Møller et al., 2014) while in LCA global boundaries are commonly implemented (ISO, 2006). As a consequence, the application of cost assessment outcomes from one research as base data in other studies might result in imprecisions and biased results, such as the transfers that might differ between European countries.

Finally, the monetization of environmental burdens is a critical issue. Concerning the anticipated transfers, various studied have incorporated them in cLCC and eLCC (Kim et al, 2011; Zhang, 2013). However, several approaches have been employed. It is of crucial importance to transparently report the selected procedure because the results are highly influenced by the assumption and valuation principles employed. Furthermore, Carlsson Reich (2005) has highlighted that even though the market price of resources moderately reflect resource scarcity, it is vague how large is the portion of the price that is associated with the scarcity itself. It is possible that present market prices are dependent on short-term resource availability rather than long-term abiotic resource depletion. Therefore, market prices might not be taken completely into account for the relevant future impacts triggered by present resource use. In eLCC, short- and long-term resource features are incorporated either in the economic analysis (market price) or in the environmental analysis (resource depletion in the LCA). However, in sLCC, future impacts are not assessed unless empirical researches assess the external costs involved. Likewise, there are several environmental emissions whose shadow prices have not been assessed yet. At last, the discount on future financial costs is also of crucial importance when LCC is performed together with LCA. Despite the fact that Hunkeler et al. (2008) support that discounting is inconsistent with eLCC, most of the examined research discounted the costs to adequately perform the cost allocation.

3.6. Conclusions of Chapter 3

The main theories and principles of LCA and LCC were examined. Furthermore, the methods for monetizing the environmental and social impacts were also reviewed. In line with the literature findings, a generic approach to integrate these two life cycle assessment techniques is by employing the structure of LCA to combine the environmental impact assessment and internal and external cost estimations. Concerning the three types of LCC, after thorough research, it is decided that cLCC will include internal costs, eLCC will include internal costs as well as transfers (environmental taxes & subsidies) and sLCC will take into account internal and external costs. The latter will be calculated with the help of the weighting factors provided by De Bruyn et al. (2018) that concern average values for emissions in 2015 for the Netherlands.

External costs usually include both environmental and social impacts. However, in this report, only environmental impacts are included in the economic analysis since the reviewed social impacts were either out either beyond the scope of the ZB case study or their monetization was difficult. LCC's structure is displayed in Figure 18.

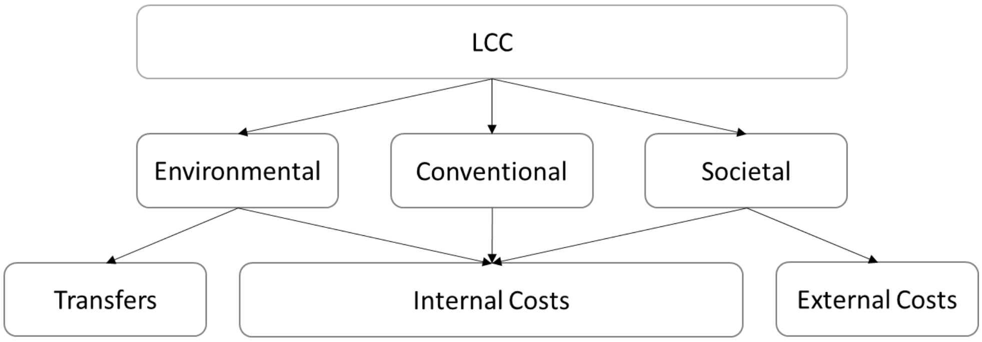

Figure 18. LCC's structure

Figure 19 depicts the employed approach of this report concerning the incorporation of LCA and LCC analysis. Based on ISO 14040 and 14044, LCA and LCC tools are parallel implemented. First of all, the goal and scope of the study are established to be coherent with both parts of the assessment. Following, after defining the system boundaries, the LCI is compiled by the environmental and economic flows. Next, the system's environmental damages are quantified by employing an appropriate LCA software and choosing an impact assessment method. Then, the identified environmental impacts can be expressed in monetary terms. For that purpose, internal, transfer and external costs are gathered and estimated for a certain time period. The estimation of the anticipated future cash flows was performed by choosing a suitable discount rate. Lastly, the interpretation along with the discussion of the results is conducted.

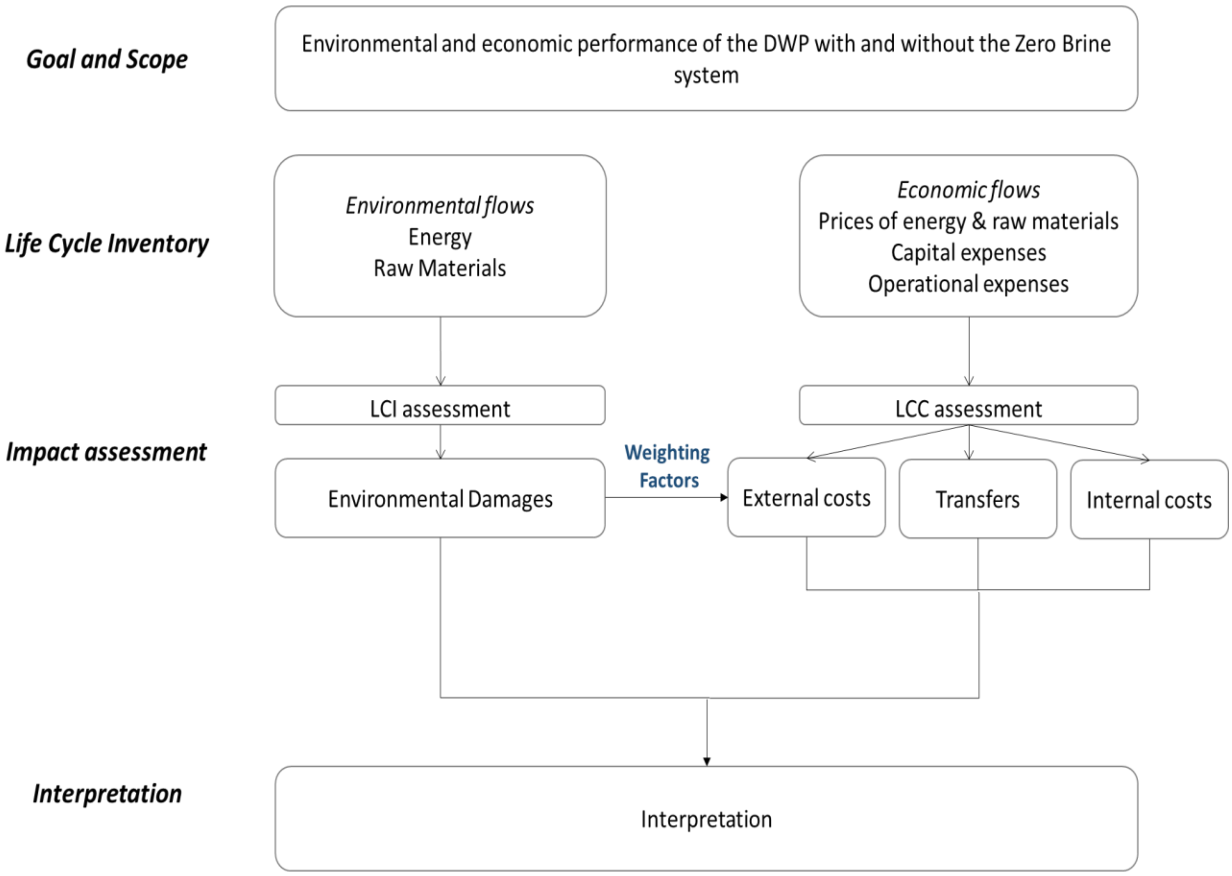

Figure 19. LCA & LCC analysis structure

Chapter 4

LCA and LCC Methodology

In this section, the developed LCA and LCC structure is applied to the ZB case study. The applied methodology of LCA and LCC is presented based on the ISO 14040 and ISO 14044 standards. In Section 4.1, the goal and scope of this study are presented, including the system boundaries, the functional unit, the allocation, and the impact categories. Following, in Section 4.2, the LCI is provided with the environmental and economic flows. Furthermore, the mathematical formulas for the economic analysis are also provided. Finally, in section 4.3 the assumptions that were made are outlined.

4.1. Goal and Scope

4.1.1. Goal Definition

The goal of the study is to investigate the environmental and economic performance of ZB applications by implementing the LCA and LCC tools in an integrated way. Specifically, the environmental and economic impact of DWP with and without the ZB system are examined. Along with LCA, the three types of LCC are applied, namely cLCC, eLCC, and sLCC. For that reason, the external costs are quantified and the internal and transfer costs are calculated. In order to quantify the external costs, weighting factors are used to translate the LCA results into monetary terms for the sLCC. By comparing the pre- and post- ZB system, we could indicate the performance of ZB design in terms of environmental and economic efficiency. Furthermore, an additional objective is the identification of the analyzed processes "hot-spots" and the suggestion of improvements.

4.1.2. System Boundaries

By outlining the boundaries of the system, an overview of the time and spatial limits of the assessment is provided. For the present study, considering the analyzed systems, it was decided to use a "cradle – to – gate" approach for the LCA. The modeling of the energy and mass flows involves raw material production as well as the manufacturing stage. This selection does not differ for the LCC study where any relevant phase of an activity's or product's life cycle should be incorporated in the economic assessment. Therefore, also for the economic assessment, the "cradle-to-gate" system boundaries will be undertaken. Overall, the use and disposal phase of the products are omitted in this report given that they are not included in the selected "cradle – to – gate" orientation.

By defining the system boundaries, the processes that will be included or excluded from the system analysis are determined. Figures 20 and 21 show the process flow diagrams for all the processes included in LCA and LCC. Figure 20 illustrates the pre –ZB system, namely DWP's current operation. The system boundaries start with the inflows of raw materials and energy needed for the operation of DWP. The water from the Brielse Lake is pumped to the DWP and processed in order to produce ultra-pure demi water and brine. The lake water is processed firstly with coagulation, floatation and filtration processes for the removal of suspended matters. Subsequently, the IEX unit removes the divalent cations and softening the water. Then, the RO is involved to remove monovalent salts and lastly, the mixed bed polisher removes traces of salts and charged organics and makes the water ready to be supplied to the water network. Both the IEX and RO units produce brine. The infrastructure costs for the illustrated technological units are also included in the system boundaries.

Figure 21 demonstrates the post- ZB system that consists of the DWP, Site I and Site II. The raw materials and energy required for the production of the demineralized water and the treatment of brine are the inflows of the system. The DWP includes the same processes described above. Site I and Site II treat the IEX and RO brine that derive from the DWP. Concerning Site I, the proposed brine treatment system includes the following components: i) NF, ii) Evaporator and iii) MC units. Combining NF with evaporation for treatment of IEX regenerate comprises an integrated resource recovery system in which NF serves as the purification/separation step and evaporation as the concentration step. During the operation of Site I, magnesium and calcium hydroxide are produced as well as sodium chloride and water. Furthermore, the RO brine is treated in Site II. During its operation, i) IEX, ii) TOC removal, iii) NF, iv) RO and v) evaporation technologies are employed. During the treatment of RO brine, sodium bicarbonate, sodium chloride, and water are produced. Overall, the recovered sodium chloride will be reused internally in the DWP and in Site's II IEX unit for the regeneration of resins, the recovered water will replace water pumped from Brielse Lake, while the rest salts are expected to be sold externally and replace existing products in the market. As also mentioned in the pre –ZB system, infrastructure costs are also included in the post –ZB system analysis.

The system boundaries are the same for both LCA and LCC. For this reason, the economic inflows (benefits) and outflows (internal, transfer and external costs) define the main characteristics of the LCC analysis. Nevertheless, as previously noted, LCC is usually performed from the viewpoint of a specific economic decision-maker and is classified by the position of this actor in the supply chain of the product. Regarding the economic perspective of this study, it should be stated that the economic evaluation is conducted from the manufacturer's perspective (Evides). LCC analysis is focused on all those life cycle phases (or parts of it) where monetary effects occur that are relevant for the respective decision-maker. Hence, the scope of the LCC analysis in this study is expanded in order to include the required capital investments for the analyzed systems.

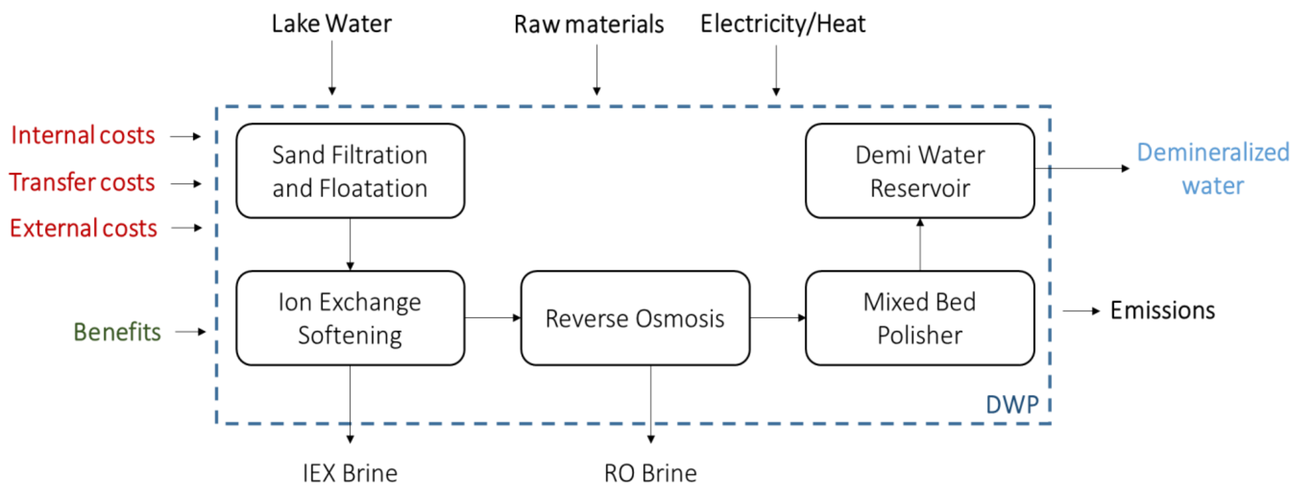

Figure 20. System boundaries of pre-ZB system

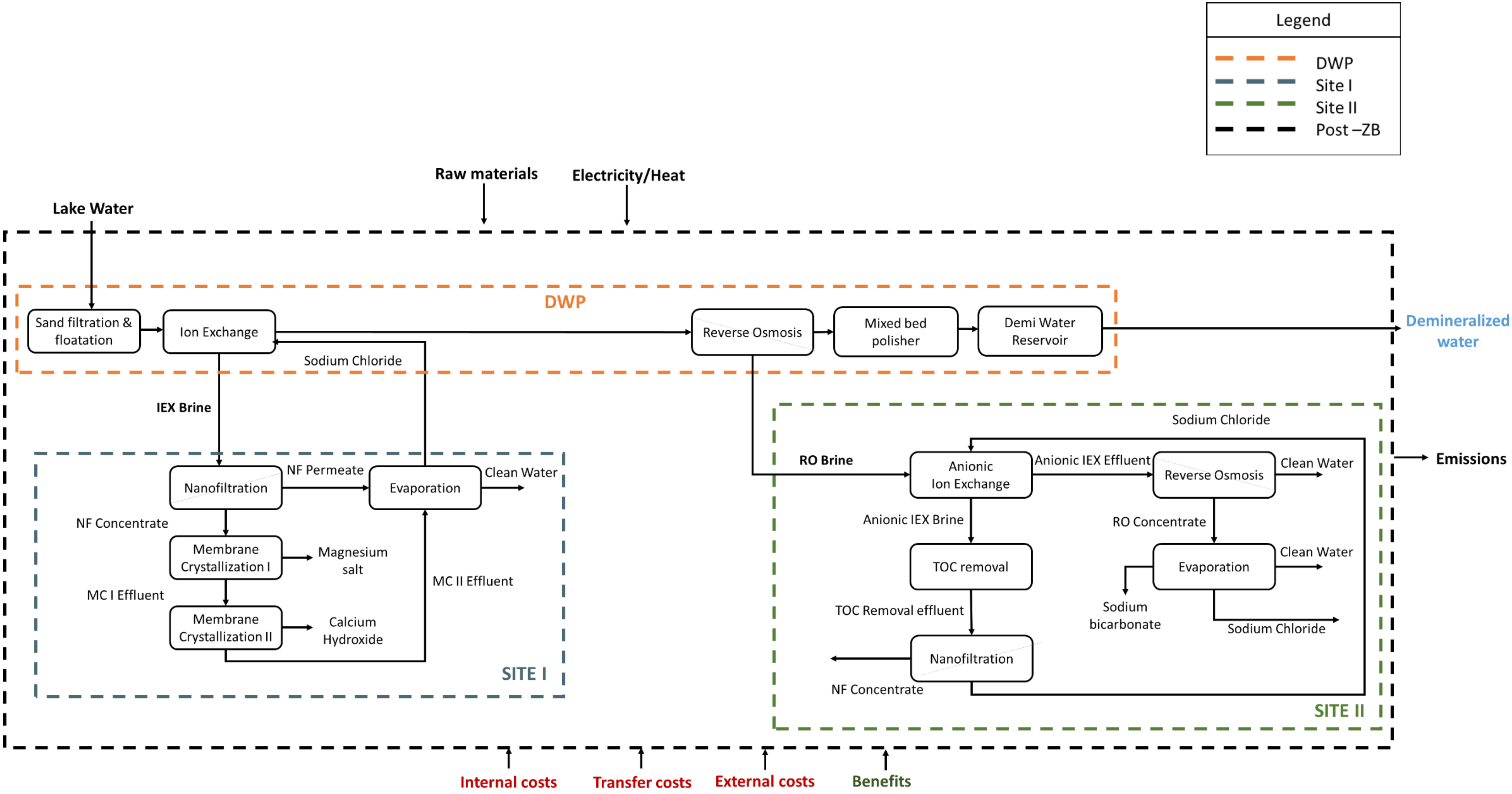

Figure 21. System boundaries of post-ZB system

4.1.3. Allocation

According to ISO 14040 and 14044, allocation in LCA should be avoided wherever possible. However, allocation is required when there are multifunctional processes. Concerning the pre-ZB system, as depicted in figure 20, all the processes are mono-functional so there is no need for allocation. Concerning the post –ZB system, mass, and economic allocation were performed since most of Site's I & II are multifunctional as shown in figure 21. The combination of mass and economic allocation derives from the fact that even though there are processes that are multifunctional, some of the outflows have not economic value in order to apply economic allocation. For example, the outflows of Membrane Crystallization I in Site I are magnesium hydroxide and a brine stream that is treated as waste and economic allocation cannot be applied. Hence mass allocation is performed. However, in the evaporation process in Site I economic allocation is performed since clean water and sodium chloride with high quality are produced. In Appendix C, the allocation factors along with their calculation processes are presented. Economic allocation was based on the prices presented in section 4.2.1.6.

4.1.4. Functional Unit

The functional unit is the quantified definition of the function of a product. In order to be able to compare the pre- and post – ZB system, the same functional unit should be employed. Part of defining a functional unit is the definition of a reference flow, which is the measure of product components and materials needed to fulfill the function, as defined by the functional unit. Furthermore, the data used both in LCA and LCC must be calculated in accordance with the reference flow. In this context, in accordance with the goal of this study, the functional unit selected for this project is 1000 m^3 demi water for both environmental and economic assessment. Accordingly, the function of the system is the production of demi water and the reference flows is the production of 1000 m^3 demi water by the DWP with and without the application of ZB systems. In Table 10 the abovementioned choices are presented.

Table 10. Function, functional unit, and reference flows

	LCA/LCC
Function	Production of demi water
Functional Unit	1000 m^3 demi water
Reference flows	Production of 1000 m^3 demi water by pre –ZB system Production of 1000 m^3 demi water by post –ZB system

4.1.5. Impact Categories

For the identification and quantification of the environmental impacts, the SimaPro 8.5.2.0 software was utilized. Furthermore, the ReCiPe 2016 v1.1 Midpoint method (H) was applied (Goedkoop et al., 2013). This specific impact assessment method was selected as a consequence of its global scope and recent introduction in the LCA field. Moreover, this method will allow us to apply the weighting factors found in the literature since they are especially suitable for usage in LCAs according to the ReCiPe midpoint methodology under the hierarchist perspective. The latter is based on the scientific consensus with regard to the time frame and plausibility of impact mechanisms.

All the midpoint impact categories Recipe were considered. However, in order to make the analysis simple and clear, the impact categories with a low contribution to the environmental impacts are neglected. The descriptions of the nine midpoint indicators that are covered are listed below (Goedkoop et al., 2013):

- **Particulate matter formation:** Particulate Matter is a complex mixture of extremely small particles, also called particulate pollution. Particle pollution can be made up of a number of components, including acids (such as nitrates and sulfates), organic chemicals, metals, and soil or dust particles. It is measured in PM_{10} equivalents, i.e. particles with a size of 10 μm.
- **Freshwater ecotoxicity:** the impacts of toxic substances on freshwater aquatic ecosystems, measures the emissions of toxic substances into the air, water, and soil. 1,4-dichlorophenoxy (1,4-DB) is used as a reference chemical.
- **Global Warming Potential**: the impact of human emissions on the radiative forcing of the atmosphere, measures the emissions of greenhouse gases into the air. Resulting from an increase in diseases and death caused by climate change. Factors are expressed as Global Warming Potential over the time horizon of different years, measured in the reference unit, kg CO_2 equivalent.
- **Human toxicity:** Human toxicity characterization provides relative comparisons of a large number of chemicals that may have the potential to contribute to cancer or other negative human health effects. 1,4-dichlorophenoxy (1,4-DB) is used as a reference chemical in this impact category.
- **Ionizing radiation:** is linked to the emissions of radionuclides throughout a product life cycle. The category takes into account the radiation types α-, β-, γ-rays, and neutrons. The unit the impact is given is kg of uranium-235 (U^{235})
- **Marine ecotoxicity** refers to the impacts of toxic substances on the sediment of seawater ecosystems.
- **Marine eutrophication**: refers to the emissions of nutrients in the air, water, and soil. Due to the enrichment by macronutrients (mostly nitrogen and phosphorus), an undesirable shift in species composition and biomass growth may occur in aquatic ecosystems.
- **Terrestrial acidification:** the impact of acidifying pollutants on soil, groundwater, surface waters, biological organisms, ecosystems, and materials (buildings). Characterization factors for acidification on both the global and the European scale are defined for SO_2, NOx, and NH_3.

4.1.6. Upscaling of Site I & II

Since Site I & II are pilot systems that treat the 5% of IEX brine and 0.4 % of RO brine respectively, they cannot be compared with DWP due to different scales. As a result, in the context of this thesis, the Site I & Site II pilot systems have been upscaled to perform the environmental and economic analysis. For the upscaled systems, various assumptions have been made. To begin with, as it is mentioned in section 2.2, the DWP is currently producing 100 m^3/day of IEX brine and 2000 m^3/day of RO brine. The upscaled Site I & II will be able to treat all the brine produced, so the inputs for the upscaled Sites I & II will be 100 m^3/day of IEX brine and 2000 m^3/day of RO brine, accordingly. Furthermore, in order to calculate the outflows of each process in both Sites, it is assumed that the efficiency of each equipment remains the same. Consequently, the inflows and outflows of Site I & II were increased by an upscaling factor of 12.5 and 250, respectively. Overall, the systems were upscaled linearly with the abovementioned mentioned factors. However, various variables could influence the procedure in scale-up processes. For example, in the design of a chemical plant, the application of continuous processing could reduce the quantities of energy and raw materials required as inputs for the processes. For this reason, sensitivity analysis was carried out in section 6.1.3 to tackle this issue.

4.2. Life Cycle Inventory

4.2.1. LCA

The required data for the LCA are gathered in this step. The inflows and outflows of pre- and post-ZB systems are presented. Due to confidentiality issues, they have been removed from this version.

4.2.2. LCC

In order to evaluate the economic performance of the post-ZB system, a full cost calculation was performed by employing the LCC technique. This report applied a model for cLCC, eLCC and sLCC to calculate the internal costs as well as to determine the transfers and external costs per functional unit of 1000 m^3 demi water. During the LCC analysis, the system specifications, as well as the time horizon, are the same as the LCA analysis.

4.2.2.1. cLCC

For the cLCC, the internal costs which are divided into operational (Opex) and capital expenses (Capex), are estimated. The capital costs for the DWP have been estimated based on the Evides annual report (Evides, 2018). Regarding the upscaled Site I & II, the required capital for the equipment was based on estimations made on the Grant Agreement of the ZB project (Zero Brine, 2017).

The operating costs consist of maintenance, labor, personnel, raw materials and energy. The maintenance cost (M_c) is calculated as the 3% of the Capex (on an annual basis), whereas the labor cost for (L_c) is estimated as 20% of the personnel cost (P_c) (Trieb, F., Moser, M., & Fichter, T., 2012; Micari et al., 2019). The personnel cost is quantified by multiplying the average cost of the personnel ($\bar{C}_P$) with the number of employees (N_P). The raw material's cost (R_c) is given by the sum of each material's quantity (Q_R) multiplied its specific cost (C_R). Finally, the electric energy cost (E_c) is calculated by multiplying the specific electric consumption (Q_E) with the specific electric energy cost (C_E). The total operating cost is given by the sum of all the described operating cost terms. The abovementioned costs calculations are described from the following equations (see equation 4.1 – 4.6):

$$M_c = 0{,}03 * Capex \qquad (4.1)$$

$$L_c = 0{,}20 * P_c \qquad (4.2)$$

$$P_c = N_p * \bar{C}_P \qquad (4.3)$$

$$R_c = \sum(Q_R * C_R) \qquad (4.4)$$

$$E_c = Q_E * C_E \qquad (4.5)$$

$$Opex = \sum(M_c + L_c + P_c + R_c + E_c) \qquad (4.6)$$

Subsequently, the Present Value (PV) approach is applied. It is assumed that all money flows occur by the end of each year. However, the required Capex takes place on the first day of the first year of the analysis. Hence, the discounting of all future expenses takes place on the first day (day 1). As shown in the equation (4.7), the discounted future Opex are added up with the Capex to calculate the internal costs. All the expenses that took place before day 1 are not relevant and thus are not examined in the analysis.

$$PV_{internal\ costs} = Capex + \sum_{t=0}^{N} \frac{Opex}{(1+i)^t} \qquad (4.7)$$

Where

N the time horizon

i the nominal discount rate

The nominal discount rate (i) is given by equation (4.8):

$$i = (1 + r) * (1 + \pi) - 1 \qquad (4.8)$$

Where

r the real discount rate

π the Inflation rate

The required prices for the estimation of the internal costs of the post-Zero Brine system are provided in section 4.2.1.6. Furthermore, the choice of the real and nominal discount rate as well as of the inflation rate is presented in section 4.2.1.5.

4.2.2.2. eLCC

Concerning the eLCC, it includes the internal costs calculated in cLCC as well as the transfers. Transfer costs represent income redistribution between stakeholders with no re-allocation of resources (Huppes et al., 2008; Martinez- Sanchez et al., 2015, Edwards et al., 2018). They are typically in the form of taxes or subsidies. Transfer costs considered in the study are energy and carbon taxes that are calculated based on the electrical consumption and the results from LCA for the global warming potential, respectively.

Energy tax is a government tax imposed on all companies in the Netherlands that consume energy ("Business.gov.nl", 2019). The amount of energy tax owed depends on the quantity of energy used. Energy tax is collected by the supplier, who pays it to the Dutch Tax and Customs Administration. The rates for 2019 are shown in Table 14. The electricity tax is assumed that will be applied also to the ZB system when it will be constructed.

Table 11. Electricity rates for 2019 ("Tabellen tarieven milieubelastingen", 2019)

Electricity consumption	0 to 10,000 kWh	10,001 to 50,000 kWh	50,001 to 10 million kWh	More than 10 million kWh
Tax (€/kWh)	0.09863	0.05337	0.01421	0.00058

Carbon taxation is not currently applied in the Netherlands. However, the Dutch government plans to introduce a carbon levy in the industry as stated in the national Climate agreement. The carbon tax will start at 30 €/ton CO_2 in 2021 and will rise to 125 – 150 €/ton CO_2 in 2030 ("Climate deal makes halving carbon emissions feasible and affordable", 2019). This tax will be included in the eLCC, starting at 30 €/ton in year 1 and then it will be increased in year 12 to 130 €/ton. The following equations (4.9 – 4.11) will be used for the calculation of transfer costs (€).

$$Electricity\ tax = Q_E * T_E \qquad (4.9)$$

$$Carbon\ tax = GW * T_C \qquad (4.10)$$

$$Transfer\ costs = Electricity\ tax + Carbon\ tax \qquad (4.11)$$

Where

Q_E the electricity consumption (kWh)

T_E the electricity tax rate (€/kWh)

GW global warming (kg CO_2 eq.)

T_C the carbon tax rate (€/kg CO_2)

Furthermore, the PV of the transfer costs is calculated as the internal costs and is shown in equation 4.12.

$$PV_{transfer\ costs} = \sum_{t=0}^{N} \frac{Transfer\ costs}{(1+i)^t} \qquad (4.12)$$

4.2.2.3. sLCC

For the sLCC, the internal costs along with the external costs were taken into consideration. For the calculation of the external costs, the results from the LCA for the nine midpoint impact categories that analyzed in section 4.1.5, were monetized. In order to translate the environmental impacts into monetary terms, the weighting factors provided by CE Delft were utilized (De Bruyn et al., 2018). The referred weighting factors along with the impact categories are displayed in Table 15.

Table 12. Weighting factors for monetizing the LCA midpoint impact categories (De Bruyn et al., 2018).

Impact Category	Weighting factor	Unit
Global Warming Potential	0.057	€/kg CO_2 – eq.
Ionizing radiation	0.0473	€/kg kBq U_{235}-eq.
Particulate matter formation	69	€/ kg PM_{10} – eq.
Acidification	5.4	€/ kg SO_2-eq.
Freshwater eutrophication	1.9	€/ kg P- eq.
Marine eutrophication	3.11	€/ kg N
Freshwater ecotoxicity	0.0369	€/ kg 1,4 DB – eq.
Marine ecotoxicity	0.00756	€/ kg 1,4 DB – eq.
Human toxicity	0.214	€/ kg 1,4 DB – eq.

In order to calculate the external cost for each impact category (y) and the PV of all costs, the following equations (see 4.13-4.14) are used:

$$External\ cost(y) = \frac{Impact\ (y)}{FU} * Weighting\ factor(y) \qquad (4.13)$$

In order to calculate the PV of the external costs, it was assumed that the amount of environmental impacts remains stable every year. Equation (4.14) provides the annuity for the calculation of the external cost's PV.

$$PV_{external\ costs} = AP * \frac{\sum_{y=1}^{10} external\ cost(y)}{i} * \left(1 - \frac{1}{(1+i)^N}\right) \qquad (4.14)$$

Where

AP the annual production of demi water

4.2.2.4. Total Economic Performance

The transfer, as well as the external costs, are added up with the internal costs to obtain the total economic performance of the analyzed systems. The PV of the total costs for the selected time horizon is presented in equation (4.15):

$$PV_{total\ costs} = PV_{internal\ costs} + PV_{transfer\ costs} + PV_{external\ costs} \quad (4.15)$$

Furthermore, in order to estimate the net benefits of the post-ZB system, the Net Present Value (NPV) rule has been implemented. The time value of money flows that occur at various points in time is calculated by the NPV. In this study, the NPV is given by equation 4.16. As such, the NPV is calculated by subtracting the PV of the total costs from the PV of the total benefits. The prices for the calculation of the total benefits are displayed in section 4.2.1.6.

$$NPV = PV_{total\ benefits} - PV_{total\ costs} \quad (4.16)$$

4.2.2.5. Time Horizon and Discount Rate

The selection of the time horizon and the discount rate is of crucial importance in the economic analysis of a project or system since they highly influence the results. European Commission (2014) has presented some directions regarding the economic analysis of environmental projects. In accordance with these guidelines, a time horizon of twenty years was selected for the environmental and economic analysis of the post-ZB system. As far as the nominal discount rate is concerned, it takes into account the alterations in the purchase power of the Netherlands and thus it is adapted to inflation rates. The selected inflation rate is 1.6% portraying the Dutch average rate for 2018 (Statista, 2019). Furthermore, the nominal discount rate is estimated based on the provided equation (4.8).

Regarding the real discount rate, two different discount rates were applied. For the calculation of the internal costs, companies should choose a conservative discount rate which usually varies from 0% to 15%. Generally, the discount rate is slightly higher than the local inflation rate. However, after the financial crisis in 2008 and the worldwide public debt problem, a lower discount factor is more likely for private companies, thus a modest discount rate of 4% was selected based on the guidelines of the European Commission (Rödger, Kjær & Pagoropoulos, 2018). Furthermore, for the estimation of transfer costs, the same discount rate was applied (Hunkeler, 2008; Martinez-Sanchez, Tonini, Møller & Astrup, 2016; Edwards, Burn, Crossin & Othman, 2018). Finally, for environmental impacts monetized in the sLCC, the discount factor depends on the time horizon of the impacts under study. In this report, a low discount factor was selected, namely 0.001%. The selection of this rate prevents the transfer of the environment damages to future generations. In this way, the involved stakeholders are incentivized to take action in order to reduce the environmental costs.

4.2.2.6. Economic Data

One of the main challenges regarding LCC analysis is the quality of the input data. The availability, the accessibility as well as the reliability of the economic data are highly uncertain when an economic analysis is performed. Even though the LCC analysis in this report was performed from the perspective of a company, the required economic inputs were not provided by Evides since some data may be business sensitive. Hence, the required inputs were collected from various journal articles, reports, websites, and databases. In Table 16, the applied economic values are presented to transparently show the choices that were made.

Table 13. Economic data for the LCC analysis

	Unit Cost	Reference
Benefits		
Demi Water	2 – 3 €/kg	Zero Brine (2017), Global CCS Institute (2018)
Magnesium hydroxide	1.47 – 1.68 €/kg	Zero Brine (2017)
Calcium hydroxide	0.091 – 0.27 €/kg	Mastali, Abdollahnejad and Pacheco-Torgal, (2018), Kemcore.com (2019)
Sodium bicarbonate	0.20 – 0.25 €/kg	Kemcore.com (2019)
Internal Costs		
Consumables		
Sodium Chloride	0.07 – 0.2 €/kg	Zero Brine (2017)
Sodium Hydroxide	0.36 – 0.6 €/kg	IHS Markit (2018)
Hydrochloric acid	0.15 – 0.32 €/kg	ICIS (2019)
Iron(III) Chloride	0.14 - 0.6 €/kg	Kemcore.com (2019)
Polyacrylamide	4.51 – 4.89 €/kg	Wasseraufbereitung.de (2019)
Antiscalant (Vitec 3000)	8.13 – 8.56 €/kg	Wasseraufbereitung.de (2019)
Antiscalant (Vitec 4000)	13.75 – 14.47 €/kg	Wasseraufbereitung.de (2019)
Sulfuric acid	0.17 – 0.36 €/kg	Kemcore.com (2019)
Energy		
Electricity	0.0679 €/kWh	Eurostat (2019)
Steam (200°C, 18 bar)	0.02 – 0.03 €/kg	Global CCS Institute (2018)
Maintenance	% of Capex	Trieb et al. (2012),Micari et al. (2019)
Labor	% of Personnel Cost	Micari et al. (2019)
Personnel	81.700 €/year/employee	Evides Waterbedrijf (2018)

As illustrated in Table 16, various sources were employed to gather the required inputs for economic analysis. For the identification of the raw material's prices, the following procedure was followed. Firstly, the ZB Grant Agreement (Zero Brine, 2017), scientific papers (Mastali, Abdollahnejad, and Pacheco-Torgal, 2018) and available market reports (Global CCS Institute, 2018, IHS Markit, 2018, ICIS 2019) for the abovementioned commodities were examined to find the relevant market prices. Following, since some of the prices were not found in the previous step, websites such as Kemcore.com and Wasseraufbereitung.de were utilized. These websites sell chemicals in bulk quantities from multiple manufacturers. Hence, the minimum and maximum price given from the different providers for a specific commodity were selected. Overall, in this report, the average unit cost was selected for the economic analysis.

Concerning the energy costs, the electricity price was adapted from Eurostat for Dutch non-households consumers for the first semester of 2019 (Eurostat, 2019). The electricity price does not include taxes and levies since they are calculated in the eLCC for the electricity taxes. Furthermore, the average personnel cost was used based on the annual report of Evides (Evides Waterbedrijf, 2018). The maintenance and labor costs were calculated as a percentage of capital and personnel costs, respectively. As far as the capital costs are concerned, they amount to 1,03M € and 6M € for the DWP and Site I & II, accordingly. For the DWP it was assumed that since the water produced from the DWP accounts for 10% of the total industrial water production of Evides (Evides, 2018), the Capex of DWP are 10% of the capital costs for the industrial water production. Regarding Site I & II, the capex were based on preliminary estimations stated on ZB Grant Agreement (Zero Brine, 2017). Even though these appraisals are not completely accurate, it was difficult to gather the Capex for each technology in the ZB project since some of these technologies do not exist in the market. In section 6.3.4, the contribution of each type of cost to the total economic performance of the company is presented.

4.3. Assumptions

4.3.1. Environmental Analysis

Concerning the post-Zero Brine system, it was assumed that the sodium chloride produced from Site I & II has a two-fold purpose. On the one hand, 80% of the sodium chloride produced is used for the reduction of the quantity of NaCl in the DWP and the remaining 20% is used for the regeneration of the anionic IEX of Site II. Furthermore, the water produced for Site I & II is assumed to replace the 4% of the water lake fed in DWP. However, due to the fact that the produced water is distilled, there is no need for dissolved air flotation and filtration processes and thus the electricity required for this process is reduced. This reduction is based on Sharaai, Mahmood & Sulaiman (2009) that calculated that dissolved air flotation and filtration processes require 446.17 kWh per 1000 m^3 treated water. Moreover, the EFC process in Site II has not included either in environmental or economic analysis since the equipment is not ready yet and there is no information regarding its outflows and electricity consumption. Finally, one limit of the Life Cycle Impact Assessment models is their limit to assess the effect of brine, and even saline, effluents on the environment. It is expected that brine disposal results in environmental burdens when disposed in surface water, but when this is modeled in Simapro software, no effects are identified in ecotoxicity impact indicators. Therefore, it is expected that the freshwater and marine ecotoxicity are underestimated in the pre-ZB system. The assumptions about the upscaled Site I & II were stated in section 4.1.6.

4.3.2. Economic Analysis

For the calculation of Capex for the DWP, as it was already mentioned in the previous section, it was assumed that since the water produced from the DWP accounts for the 10% of the total industrial water production of Evides (Evides, 2018), the Capex of DWP are the 10% of the capital costs for the industrial water production. For Site I & II, the Capex was obtained based on estimations on Grant Agreement (Zero Brine, 2018). Regarding the Opex, it was assumed that both pre- and post-ZB systems operate 5 days per week and 8 hours per day. Furthermore, due to the lack of data about the cost of the lake's water, it is supposed that Evides is not paying for it. Moreover, the employees for Site I & II are assumed to be 40 and the average wage is expected to be the same as DWP. Moreover, it is assumed that all technologies will operate for the selected time horizon (20 years) without the need for replacement. This was based on the estimations obtained of ZB technology providers which stated that the lifetime of all employed technologies will be at least 20 years. Finally, it was assumed that the environmental impacts will be the same every year for the given production level and that both the inflation and the interest rate will remain stable for the selected time horizon.

4.4. Conclusions of Chapter 4

This chapter described the application of the LCA and LCC methodology to the ZB case study. Firstly, the goal of this case study is two-fold: 1) to investigate the environmental and economic performance of DWP with and without the application of the ZB system and 2) to identify the "hot-spots" of the analyzed processes. Therefore, the LCA and LCC analysis was applied in a way to attain these targets in the results of this report. Subsequently, the system boundaries were defined concerning the pre-ZB system (DWP before the application of ZB design) and post –ZB system (DWP after the application of ZB design, Site I and Site II). Consequently, a "cradle-to-gate" approach was selected for both LCA and LCC analysis, including two main life cycle stages, the procurement, and manufacturing. However, the scope of LCC analysis was expanded to include also the required capital investments since the economic evaluation is conducted from the manufacturer's perspective (Evides). Following, the functional unit selected for this project is 1000 m^3 demi water for both environmental and economic assessments. Concerning the allocation, no allocation was performed to the DWP since all the processes were mono-functional compared to Site I & II that mass and economic allocation were applied. Finally, it was decided to linearly upscale the Site I & II by two upscaling factors; namely 12.5 and 250, in order to be able to compare them with the DWP and to better reflect the conditions in a full-scale implementation.

Next, the LCI was addressed. The foreground data used for the modeling of the environmental impacts through the Simapro software were outlined. Furthermore, the data, as well as the mathematical formulas for the cost calculations were presented. The provided data and equations concern the cLCC, the eLCC, the sLCC along with the total economic performance of the system. Following, a time horizon of 20 years for the case study was selected based on the European Commisssion (2014) guidelines and the lifespan of the technologies involved in the project. Moreover, an inflation rate of 1.6% was chosen which reflects the average rate for the Netherlands for 2018 and two real discount rates were selected for the internal and external costs. Regarding the internal costs, a modest discount rate of 4% was used. For the external costs, a low discount rate of 0.001% was used to prevent the transfer of the environment damages to future generations. Finally, all the assumptions made for environmental and economic analysis were explicitly presented to make the analysis transparent.

Chapter 5

Impact Assessment

This chapter summarizes the results of the environmental and economic analysis of pre- and post-ZB systems. In Section 5.1, the outcomes of the environmental analysis in the selected midpoint categories are presented. The pre- and post-ZB are compared in terms of environmental performance and also the DWP itself before and after the application of ZB systems. Following, in Section 5.2 the results from the three types of LCC; cLCC, eLCC, sLCC are displayed per functional unit. Lastly, the overall economic performance of the system for the selected time horizon is provided.

5.1. LCA

The characterized values of the different impact categories for pre- and post-ZB systems are presented in Appendix D. By analyzing the environmental data in SimaPro, a certain amount of a substance is given for each impact category. Figures 22,23 show the environmental performance of the analyzed systems across all the examined impact categories. When the ZB system is implemented, the environmental performance of the post-ZB system varies across the considered categories. On the one hand, the majority of the environmental impacts were decreased by 15% to 22%. Specifically, ionizing radiation, human toxicity, marine eutrophication, freshwater, and marine ecotoxicity categories show better environmental performance after the implementation of ZB design. On the other hand, global warming, acidification as well as particulate matter formation categories were sharply increased in the post – ZB system. In particular, the global warming impact category showed an increase of 114% and ionizing radiation and acidification were risen by thirteen and fifteen times, respectively.

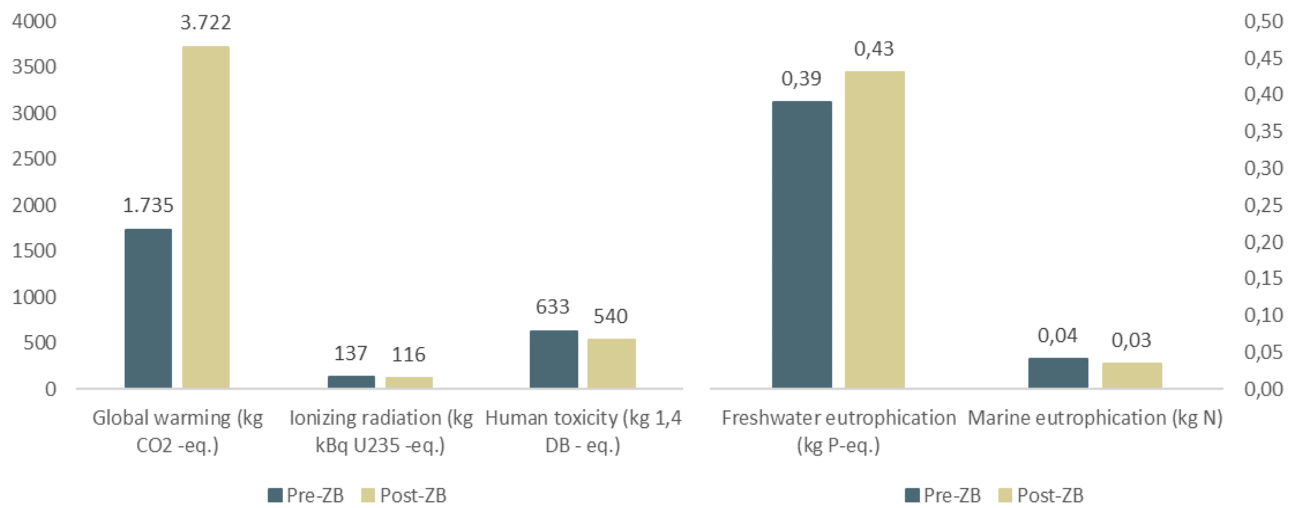

Figure 22. LCIA results for 1000 m^3 demi water (Global warming, ionizing radiation, human toxicity, freshwater and marine eutrophication)

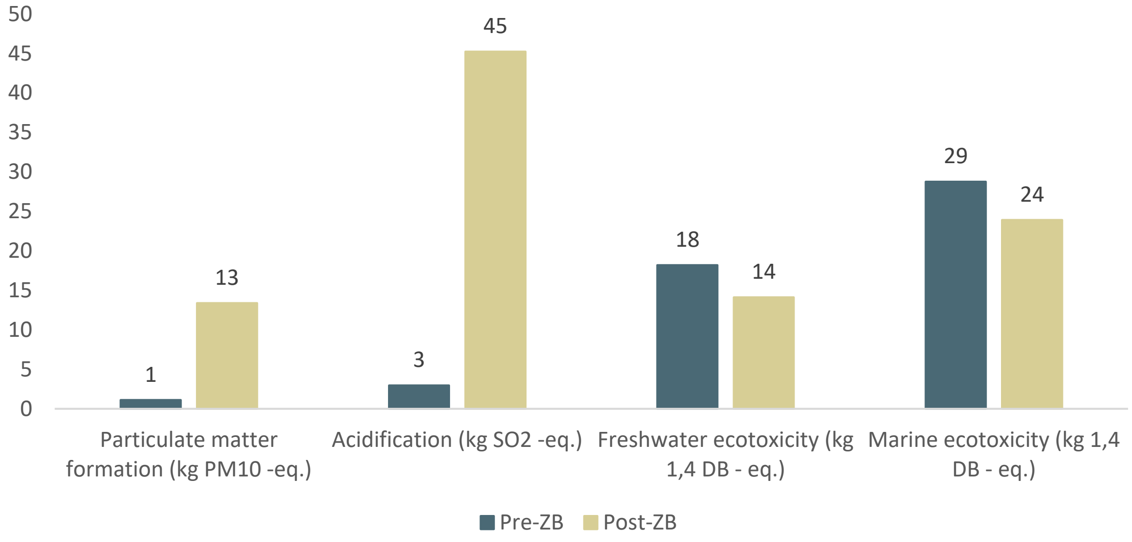

Figure 24. LCIA results for 1000 m^3 demi water (Particulate matter formation, acidification, freshwater and marine ecotoxicity)

Since the environmental performance of the post – ZB differs among the impact categories, Site I & II as well as the DWP after the implementation of the ZB project were analyzed individually to find its subsystem's contribution to each impact category. In Figure 24, the environmental impacts of the referred systems with respect to the total environmental performance of each impact category are presented. It can be observed that Site I has a small contribution to the environmental burdens with nearly zero percent to all impact categories except from global warming and marine eutrophication that contribute 1% to the overall environmental impact. Moreover, Site II shows a considerable contribution to particulate matter formation as well as acidification up to 92% and 94% respectively. However, on the other hand, DWP has the largest contribution in the majority of the categories. Specifically, it contributes to ionizing radiation, human toxicity, freshwater and marine ecotoxicity categories with more than 90%. Furthermore, in freshwater and marine eutrophication categories, it constitutes 75% and 77% respectively. Finally, DWP is responsible for 44% of the global warming impact. However, this is a small difference in comparison to Site II that yields almost 54% in the referred category.

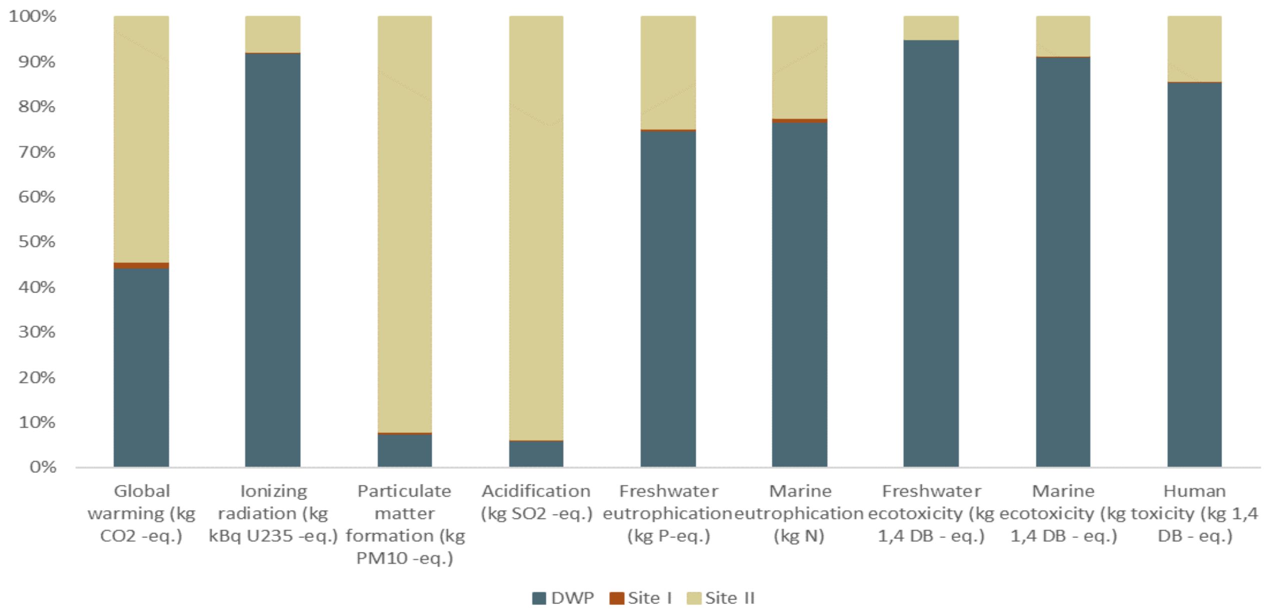

Figure 23. Contribution of DWP, Site I & Site II to post-ZB environmental performance for 1000m^3 demi water

In Figure 25, 26, the environmental burdens of DWP before and after the application of the ZB design are exhibited. It is noticed that regarding the DWP, the post-ZB system outperforms the pre-ZB system across all impact categories. Specifically, the environmental burdens are decreased by 5% to 34%. Upon initial inspection, this is reasonable since the quantities of sodium chloride, water and electricity are reduced due to the implementation of ZB design. A more detailed analysis is presented in the result's interpretation in Chapter 6.

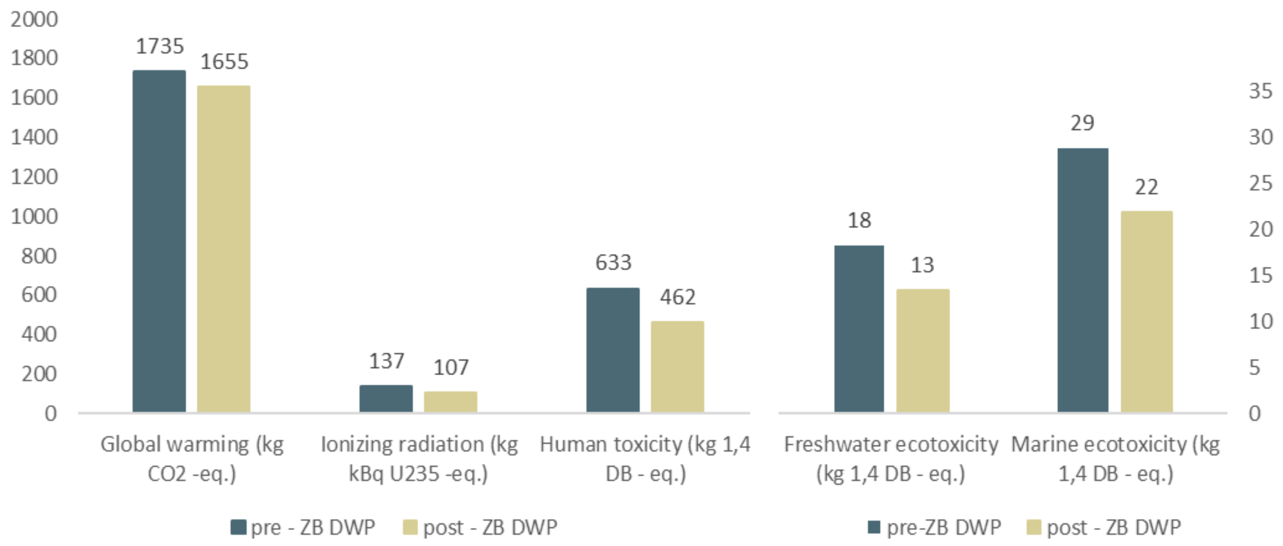

Figure 25. LCIA results of pre- and post-ZB DWP for 1000 m^3 demi water (Global warming, ionizing radiation, human toxicity, freshwater and marine ecotoxicity)

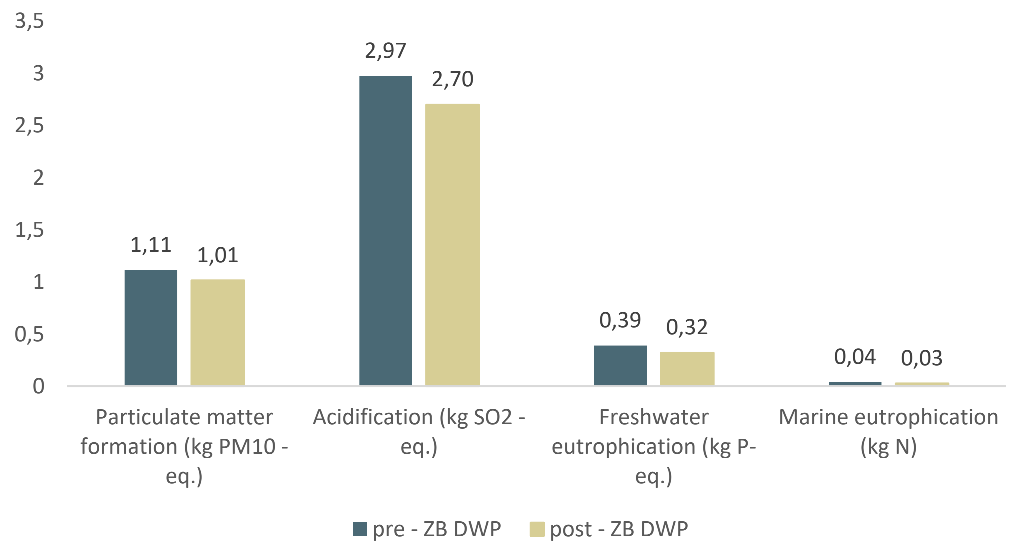

Figure 26. LCIA results of pre- and post-ZB DWP for 1000 m^3 demi water (Particulate matter formation, acidification, freshwater and marine eutrophication)

Overall, the implementation of ZB design has slightly enhanced the environmental performance of the post – ZB system in the majority of impacts category. Moreover, the performance of DWP itself after the application of the ZB system is increased in all the examined impact categories. However, it is important to mention that the particulate matter formation, acidification, and global warming potential have shown a steep increase due to the environmental performance of Site II. In the next section, we will investigate the underlying reasons behind this increase through contribution analysis.

The total environmental impacts of the production of 1000m^3 demi water have been presented so far. However, the environmental burdens identified in the LCIA are derived from the direct emissions from on-site sources, indirect emissions associated with off-site energy production and other indirect emissions such as the production of chemicals, material, and fuels. Particularly, in the post-ZB system, all the emissions are indirect and come from the involved actors in the supply chain. LCA usually includes supply chain emissions associated with material and chemical production. From an LCA perspective, the identified environmental impacts for the production of demi water are usually attributed to all of the involved supply chain actors. Nevertheless, in this study, it is assumed that the environmental consequences assigned to the company for the production of 1000 m3 demi water amount to 50% of the identified burdens. In Table 17, the calculated emissions attributed to the manufacturer, namely Evides are shown. For the economic analysis, the values presented in Table 17 are used.

Table 14. Environmental impacts attributed to the manufacturer (50% of identified burdens) of pre and post-ZB systems per 1000 m^3 demi water

Impact Category	Unit	Pre-ZB system	Post-ZB system
Global Warming	kg CO_2 – eq.	867.67	1860.83
Ionizing radiation	kg kBq U_{235}-eq.	69.69	58.07
Particulate matter formation	kg PM_{10} – eq.	0.56	6.69
Acidification	kg SO_2-eq.	1.49	22.61
Freshwater eutrophication	kg P- eq.	0.19	0.22
Marine eutrophication	kg N	0.02	0.02
Freshwater ecotoxicity	kg 1,4 DB – eq.	9.09	7.05
Marine ecotoxicity	kg 1,4 DB – eq.	14.38	11.96
Human toxicity	kg 1,4 DB – eq.	316.61	269.82

5.2. LCC

For the economic analysis, a full cost calculation is applied for the post – ZB system. The internal, transfer and external costs are calculated for the cLLC, eLCC, sLCC as well as the estimation of NPV. Furthermore, in Section 5.2.4, the NPV for the pre-ZB system is also presented in order to get a full overview of the changes in terms of costs that the implementation of ZB design has incurred.

5.2.1. cLCC

For the identification of internal costs, we conducted an estimation of the relevant costs of the post-ZB system per functional unit. As such, the mathematical formulas (4.1) – (4.6) and the information provided in the previous chapter were employed. Table 18 presented the internal cost for the production of 1000 m^3 of demi water.

Table 15. Internal costs of the post-ZB system per 1000 m^3 demi water

Post-ZB system	Internal Costs (€)
Capital Costs	724.23
Operational Costs	
Raw materials	
Sodium Chloride (NaCl)	15.76
Sodium Hydroxide (NaOH)	638.81
Hydrochloric acid (HCl)	9.18
Iron(III) Chloride (FeCl3)	4.81
Polyacrylamide	1.08
Antiscalant (Vitec 3000)	10.88
Antiscalant (Vitec 4000)	459.69
Sulfuric acid (H2SO4)	499.02
Energy	
Electricity	186.34
Stream	71.11
Maintenance	36.2
Labor	185.16
Personnel	925.83
Total Costs	**3768.17**

As it can be derived from table 18, the capital and personnel costs contribute the most to the total costs, with 19% and 25% respectively. Concerning the raw materials, the production system is highly dependent on resource consumption. It is noteworthy that sodium hydroxide, antiscalant (Vitec 4000) and sulphuric acid have very high costs with respect to the other materials and comprise 98% of the raw material's costs. The requirements in those materials comprise a considerable expense, ranging between 12% and 17% of the overall costs. An additional important cost concerns electricity production, since high consumption of energy results in high costs. Particularly, 6% of total costs are due to electricity compared to steam consumption that accounts only for 2%. Finally, approximately 30% of the total costs are attributed to maintenance as well as labor costs.

5.2.2. eLCC

The eLCC consists of internal and transfer costs. The internal costs are the same as calculated for the cLCC. Regarding the transfers, Table 19 shows the calculated tax expenses in terms of electricity and carbon emissions per functional unit as derived from equations (4.9) – (4.11).

Table 16.Transfer Costs of the post-ZB system for 1000m^3 demi water

Post-ZB system	Transfer Costs (€)
Electricity tax	1.28
Carbon Tax	57.10
Total Costs	***58.38***

It is found that carbon tax comprises almost 100% of the transfer costs and electricity tax is significantly lower in regards to the carbon tax. Furthermore, the total environmental costs of the post-ZB system amount to 3826.55 €/1000 m^3 demi water produced.

5.2.3. sLCC

The sLCC analysis incorporates the internal as well as the external costs. As already mentioned, the internal expenses have been estimated for the cLCC. For the calculation of external costs, the environmental burdens presented in Table 17 are used. As stated above, in this study it is assumed that 50% of the identified environmental impacts are attributed to the company (manufacturer). Hence, the monetized environmental impacts derived from equation (4.13) across the nine impact categories can be seen in Table 20.

Table 17. External costs of the post-ZB system for 1000m^3 demi water

Post-ZB system	External costs (€)
Global Warming	106.07
Ionizing radiation	2.74
Particulate matter formation	461.58
Acidification	122.10
Freshwater eutrophication	0.41
Marine eutrophication	0.05
Freshwater ecotoxicity	0.26
Marine ecotoxicity	0.09
Human toxicity	57.74
Total Costs	**751.05**

The magnitude of the external costs depends on the calculated impacts as well as on weighting factors applied. Particulate matter formation has the highest cost, comprising 58% of the overall expenses. Following, global warming along with acidification accounts for 16% and 15%, respectively. On the contrary, eutrophication and ecotoxicity categories contribute nearly zero to the total costs. Furthermore, 10% of the costs are due to human toxicity. Finally, the overall social costs for the production of 1000m^3 demi water are 4519.17 €.

Overall the internal, transfer and external costs of Site I & II and DWP are depicted in Figure 27. Regarding the internal costs, the ZB design which involves Site I & II contributes almost two times more than DWP, namely 67% compared to 33%. Furthermore, Site I & II have significant higher external costs, comprising 82% of the overall external costs. Nonetheless, DWP has slightly higher transfer costs than the ones of Site I & II. The transfer costs for DWP and Site I & II contribute 57% and 43% respectively.

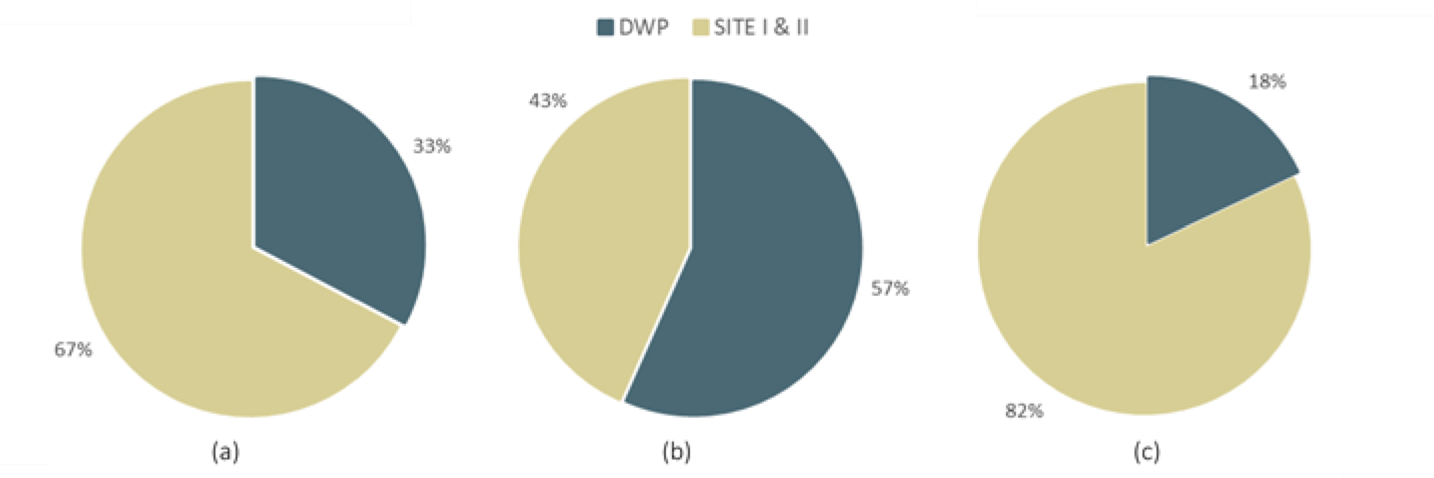

Figure 27. Post-ZB's sub-system comparison in terms of (a) internal, (b) transfer and (c) external costs

5.2.4. Total Economic Performance

The estimation of the relevant costs for the cLCC, eLCC and sLCC, are depicted in Figure 28. The costs for the production of 1000m^3 demi water for the post-ZB system are delineated in terms of internal, environmental and social costs. It is apparent that the internal costs are the highest, followed by the external and then the transfer costs. Furthermore, it is shown that the costs derived from cLCC, eLCC and sLCC are equal to 3768.17€, 3826.55€ and 4519.17 €, respectively.

Figure 28. Post – ZB's sub-system comparison of in terms of (a) cLLC, (b) eLCC and (c) sLCC

Furthermore, in Figure 29 the cLCC, eLCC, and sLCC for the DWP and Site I & II are presented. As stated previously, the eLCC is the summation of the internal and the transfer costs while the sLCC includes both internal and external costs. It is noticed that Site I & II contribute more than DWP in all three types of LCC with a range of 67% - 69% of the overall costs. Even though, as it was presented in Figure 27, DWP has higher transfer costs than Site I & II, the environmental costs of ZB design are higher than DWP due to the high contribution of the internal costs to eLCC.

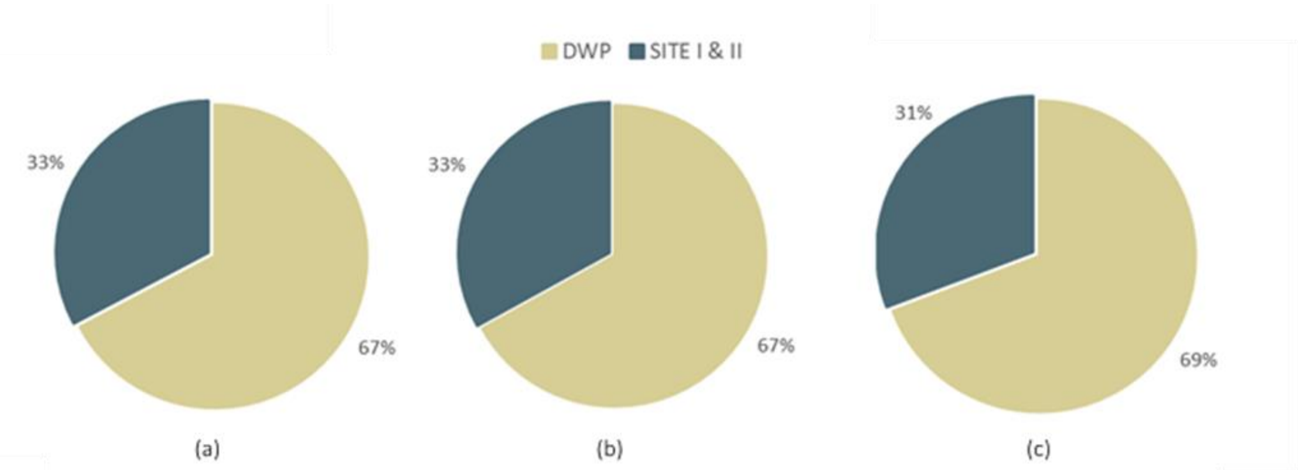

Figure 29. cLCC, eLCC and sLCC of post-ZB system for 1000 m^3 demi water

Taking into consideration the time value of money and the time period of 20 years set for this project, the NPV rule is applied to determine the viability of the post-ZB system with and without the transfer and external costs. Table 21 shows the NPV for a 20 years' time horizon. In accordance with the NPV rule, the post – ZB is not financially viable for implementation. Since the DWP before the implementation of the ZB project (pre-ZB system) has positive NPV, it can be concluded that the economic performance of Site I & II is responsible for the negative NPV. The reasons for this result will be thoroughly explained in the next chapter.

Table 18. NPV for the pre- and post- ZB systems for 20 years' time horizon

Costs	NPV (M €)	Pre-ZB system	Post-ZB system
	Total PV Benefits	345.97	309.64
Internal	Total PV Costs	128.42	404.34
	NPV	**217.54**	**-94.67**
Internal	PV Internal	128.42	404.34
Transfer	PV Transfer	3.53	7.90
External	PV External	32.64	172.6
	Total PV Costs	164.59	591.27
	NPV	**181.38**	**-275.20**

As described above, a considerable amount of the environmental burdens occur off-site due to consumables and energy production. These processes involve the activities of other actors involved in the chemical process industry and not the company itself. During the NPV estimation, it was assumed that 50% of the environmental impacts and hence the external costs are attributed to the company which is the one responsible for the acquisition of these resources. However, there is a very fine line between who is to pay for those costs and to which extent; the company itself or other involved actors in the supply chain. This is a controversial matter that we will try to approach in the next chapter.

5.3. Conclusions of Chapter 5

The results for the environmental and economic analysis of the Dutch ZB case were presented. Regarding the environmental analysis, it is concluded that after the implementation of the ZB system the environmental performance of the post-ZB system varies across the analyzed impact categories for the production of 1000 m^3 demi water. On the one hand, the majority of the environmental impacts were decreased by 15% to 22%. Specifically, ionizing radiation, human toxicity, marine eutrophication, freshwater, and marine ecotoxicity categories show better environmental performance after the implementation of ZB design. However, only freshwater ecotoxicity showed a significant decrease of 22%, with the other categories to demonstrate a decrease lower than 20%. On the other hand, global warming, acidification as well as particulate matter formation categories were sharply increased in the post – ZB system. In particular, the global warming impact category showed an increase of 114% and ionizing radiation and acidification were risen by thirteen and fifteen times, respectively.

Following, the contribution of each sub-system (DWP after the implementation of ZB design, Site I and Site II) in the post-ZB was presented. It was observed that the contribution of Site I was nearly zero across all the impact categories. Site II demonstrated a considerable contribution to particulate matter formation, acidification and global warming categories of 92%, 94%, and 54%, accordingly. In the remaining categories, DWP had the highest contribution with more than 90% in ionizing radiation, human toxicity, freshwater, and marine ecotoxicity categories and almost 75% in freshwater and marine eutrophication impact categories.

The performance of DWP itself after the application of the ZB system is increased in all the examined impact categories due to the reduction in the quantities of sodium chloride, water, and electricity. However, the categories that showed a significant decrease greater than 20% were the ionizing radiation, human toxicity marine eutrophication and ecotoxicity categories.

For the economic analysis, a full cost calculation was applied for the post – ZB system. The internal, transfer and external costs per functional unit were calculated for the cLLC, eLCC, sLCC. Overall the costs for cLCC, eLCC and sLCC for the production of 1000 m^3 demi water were equal to 3768.17€, 3826.55€ and 4519.17 €, accordingly. Furthermore, it was observed that Site I & II are responsible for almost 67% of the total costs in all three types of LCC compared to DWP that is responsible for 33%. Finally, the NPV rule was applied to determine the viability of the post-ZB system with and without the transfer and external costs. It is concluded that the post – ZB is not financially viable for implementation even without the inclusion of transfer and external costs. By estimating also the NPV for the DWP before the implementation of the ZB project (pre-ZB system) and identifying that has positive NPV, it was concluded that the economic performance of Site I & II is responsible for the negative NPV of post-ZB system.

For both the environmental and economic assessments, it was assumed that 50% of the environmental impacts and hence the external costs are attributed to the company which is the one responsible for the acquisition of these resources.

Chapter 6

Results Interpretation

The interpretation of results obtained from LCA and LCC analysis is presented. In Section 6.1, the contribution analysis of the post-ZB sub system's as well as DWP and Site II processes is presented. Following, in Section 6.2, the results obtained from the cLCC, eLCC and sLCC are analyzed. Finally, in Section 6.3, sensitivity analysis is carried out for the various assumptions that were made.

6.1. LCA

6.1.1. Contribution Analysis

During contribution analysis, the LCA's processes that contribute the most are identified. Thereby, it becomes apparent the extent to which a specific process contributes to the life cycle. From this point of view, possibilities for improvements could be identified. In this section, the LCA's results are analyzed to determine the "hot-spots" of the system.

6.1.1.1. Post-ZB sub-systems

The post-ZB system is divided into three major subsystems; the DWP, the Site I and Site II. The DWP is responsible for the production of demi water. However, along with the production of demi water, IEX and RO brine flows are generated. The Site I & II are responsible for the treatment of those two streams of brine that leads to the production of valuable co-products. Figure 30 depicts the percentage contribution of the total environmental impacts produced from the sub-systems engaged in the production of 1000 m^3 demi water.

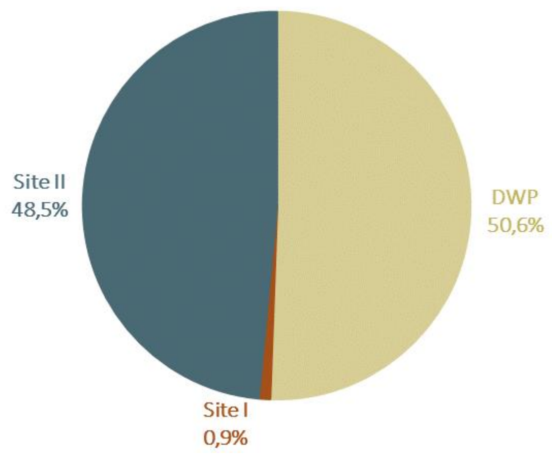

Figure 30. Contribution of post-ZB sub-systems to the total environmental impacts per 1000 m^3 demi water

Taking the impact assessment results of the previous consideration, it can be concluded that the DWP, as well as Site II, have the greatest impact on the environment. Site II treats the RO brine produced by DWP and generates sodium chloride and clean water for internal usage in the DWP and sodium bicarbonate for external use. It consists of various processes; namely Anionic IEX, RO, TOC removal, Evaporation, and NF. Electricity consumption, supplied from the national power system along with the large quantities of raw materials are the most important factors which are directly linked to the identified contribution of Site II.

Site I has nearly zero environmental burdens compared to DWP and Site II. In this phase, the treatment of IEX occurs which results in the production of sodium chloride and clean water for internal valorization and magnesium and calcium hydroxide for external valorization. It includes NF, MC and Evaporation processes. As mentioned also for Site II, the quantities of raw materials and electricity consumption are responsible for environmental damages. However, as it is observed, Site I is negligible to Site II. The reason behind this may be the quantities of brine treated in the Sites. During the production of 1000 m^3 demi water, 94kg of IEX brine are produced in comparison to RO brine that amounts to 156kg. Furthermore, the brine composition highly influences the raw materials and energy requirements.

Finally, DWP is responsible for almost half of the environmental burdens of the post – ZB system. The underlying reason is its energy and steam consumption. However, with the application of ZB design, the overall environmental performance of the DWP itself was improved as highlighted earlier. In the next section, the process-specific contribution analysis will be used to identify the "hot-spots" of the DWP and Site II.

6.1.1.2. Process-specific

This analysis will help to first identify and then suggest which processes of the system could be improved. However, as presented in the previous section Site I has nearly zero contribution to the environmental performance and thus is left out of the process-specific analysis.

Regarding the DWP, the data provided by the company concern the overall consumption of consumables and energy in the plant and are not process-specific. However, a conclusion could be drawn regarding consumables and energy. Figure 31 shows the contribution of each raw material across the impact categories.

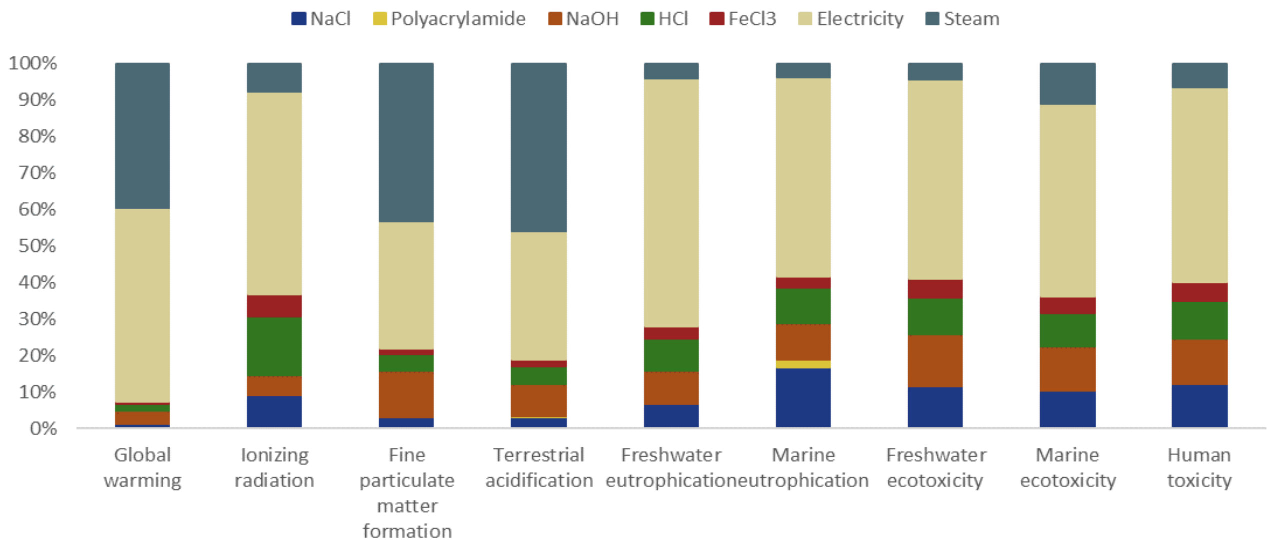

Figure 31. DWP contribution for the production of 1000m^3 demi water (post –ZB system)

First of all, it has to be noted that the contribution of sodium chloride across all impact categories has been significantly decreased in the post –ZB system since the 75% of NaCl purchased from Evides have been replaced with sodium chloride produced in Site I & II. Furthermore, as it is noticed, electricity and stream are mainly responsible for the environmental burdens across all impacts categories comprising more than 50% of the overall burdens across all impacts categories. Even though the environmental performance of DWP was enhanced after the application of ZB design, still improvements could be made in terms of energy efficiency. In order to identify the most energy-intensive process of DWP in terms of environmental impacts, each process's energy consumption must explicitly be defined which is something that is missing from the data collected. The aforementioned fact constitutes a limitation for this study.

As far as Site II is concerned, Figure 32 shows which foreground processes are the major contributors to the overall impact across the analyzed categories. In particular, as shown in section 5.1, Site II is mainly responsible for the environmental impacts of the post – ZB system for the particulate matter formation, global warming as well as acidification categories. The TOC removal process has the greatest contribution not only to the last referred categories but to all indicators, accounting for 94% to 98% of the overall impact. Hence, it is concluded that the bad environmental performance of Site II is attributed to the TOC removal process.

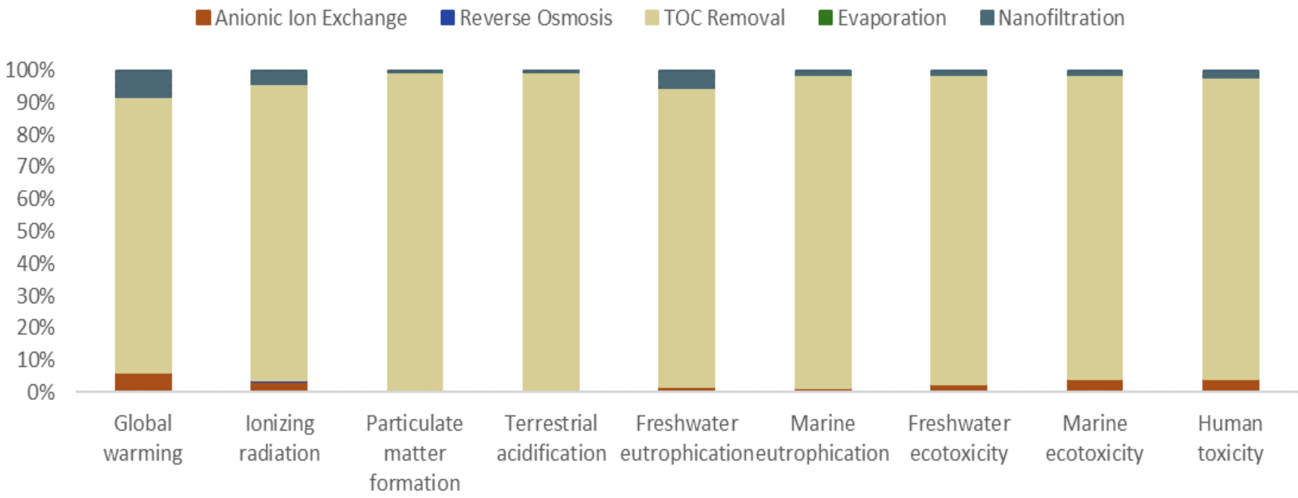

Figure 32. Site II process contribution for the production of $1000m^3$ demi water

The dominant reason for the TOC removal process being the largest contributor is the large quantities of sodium hydroxide and sulphuric acid employed. For the treatment of 156kg RO brine generated by the production of 1000 m^3 demi water, 1.9 and 1.3 tons of sulphuric acid and sodium hydroxide are required, respectively. These quantities are translated to large energy requirements for their production. As shown in Figure 33, the environmental impact of sodium hydroxide is the largest contributor in all categories except acidification and particulate matter formation, ranging from 66% to 87%. In acidification and particulate matter formation, sulphuric acid comprises 67% and 79% of the overall environmental damages. This is reasonable since their characterization factors are defined for sulfates as well as SO_2. Finally, the electricity used for the TOC removal process has lower contribution than sodium hydroxide in all categories.

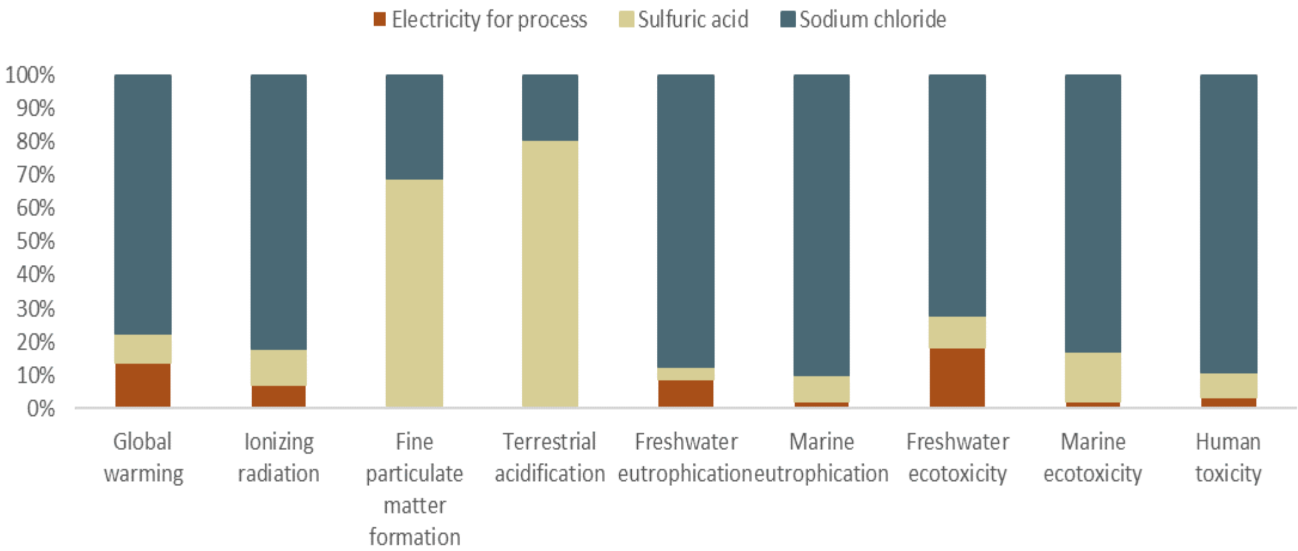

Figure 33. TOC removal process contribution

Overall, with the implementation of ZB systems; namely Site I and Site II, the environmental performance of DWP itself is enhanced. As presented in section 6.1.1.2, further improvements could be made in terms of DWP's energy consumption to enhance more its environmental efficiency. Furthermore, the total performance of the post – ZB system varies across the impact categories. The operation of Site II and more specifically the TOC removal process is the major contributor across the analyzed categories. Particulate matter formation, global warming, and acidification impacts increased dramatically and hence actions are required to improve Site II's environmental efficiency.

6.2. LCC

6.2.1. cLCC

Taking into consideration the economic results of Chapter 5, it is concluded that the post-ZB system is not financially viable. By estimating the NPV for the DWP before the application of post-ZB design, it is observed that it is the implementation of Site I & II that renders the project economical unsustainable. Even though Site I & II contribute 1.98 M € to the total benefits of the system, increasing the benefits of DWP by 8%, the internal costs required are significant high. In Table 22 the relevant internal costs and benefits that occur in a year after the system's implementation are shown. Each sub system's costs are presented.

Table 19. Internal costs and benefits of the post-ZB system for 1 year of implementation

Internal Costs/Benefits	DWP	Site I	Site II	Post-ZB system (M €)
Benefits				
Demi water	24.27	-	-	
Magnesium hydroxide ($Mg(OH)_2$)	-	0.07	-	
Calcium hydroxide ($Ca(OH)_2$)	-	0.04	-	
Sodium bicarbonate ($NaHCO_3$)	-	-	1.87	
Total Benefits				**26.25**
Capital Costs	1.03		6	7.03
Operational Costs				
Raw materials				
Sodium Chloride (NaCl)	0.15	-	-	0.15

Sodium Hydroxide (NaOH)	0.20	0.08	5.92	6
Hydrochloric acid (HCl)	0.08	0.004	-	0.084
Iron(III) Chloride (FeCl3)	0.04	-	-	0.04
Polyacrylamide	0.01	-	-	0.01
Antiscalant (Vitec 3000)	-	0.11	-	0.11
Antiscalant (Vitec 4000)	-	-	4.46	4.46
Sulfuric acid (H2SO4)	-	-	4.84	4.84
Energy				
Electricity	0.84	0.03	0.94	1.81
Stream	0.69	-	-	0.69
Maintenance	0.05	0.3		0.35
Labor	1.14	0.65		1.79
Personnel	5.72	3.27		8.99
Total Costs				**36.58**

As we can see, the costs for raw materials of Site II represent 96.4% of the overall cost for raw materials and 41% of the total internal costs. To that end, we can derive to the conclusion that the bad economic performance of the post-ZB system is assigned to the Site II consumables. As already elaborated in the LCA results, the large quantities of sodium hydroxide and sulfuric acid are behind the poor environmental performance of Site II and as indicated here also the economic one. Furthermore, the considerable expenses for the antiscalant (Vitec 4000) are owned to its high price, 14.11 €/kg. Therefore, in order for the company to reduce the costs and the environmental impact of the chemicals at Site II, the reduction of their quantities is required.

6.2.2. eLCC & sLCC

As mentioned above, the poor economic performance of the post-ZB system is mainly attributed to internal costs. However, valuable insights about environmental and social costs could be provided by analyzing the results of eLCC and sLCC. eLCC consists of internal and transfer costs. Concerning transfer costs, as internal costs analyzed above, they depend on electricity consumption and CO_2 emissions. The main reason that the costs for CO_2 tax are significantly higher than the costs for electricity tax lies in the purpose of CO_2 tax. The Netherlands is classified as one of the bigger GHG emitters in Europe as a result of its energy-intensive and fossil-dependent industrial sector (CBS, 2018). Thus has led the Dutch parliament to enact policies to reduce these emissions ("Climate deal makes halving carbon emissions feasible and affordable", 2019).

Regarding sLCC, as shown in Table 20, the highest external costs per 1000 m^3 demi water concern the particulate matter formation that equals 461.58€. The assigned price per kg PM_{10} – eq is 69€ and derived from the high impact of the particulates on human health. Particulate matter (PM) is a mixture of particles (liquid or solid) of varying size and composition and could be distinguished into primary particles that are emitted directly into the atmosphere by a wide range of sources and secondary particles that formed in the atmosphere in chemical reactions involving gaseous compounds like ammonia (NH_3), sulphur dioxide (SO_2), nitrogen oxides (NOx) and organic chemicals. Of all the environmental pollutants to which humans are exposed, it is primary and secondary particulates that cause the greatest health damage, because they transport a wide range of toxic substances directly into the air passages and lungs (De Bruyn et al., 2018). As observed in the contribution analysis, the large quantities in the TOC removal process of sulfuric acid are responsible for the particulate matter formation and hence for the high external costs.

Following, global warming has a weighting factor of 0.057€/kg CO_2-eq. resulting in 106.07€/ 1000 m^3 demi water. Climate change refers to anthropogenic changes to the Earth's climate (temperature, weather). The climate is currently changing as a result of rising atmospheric concentrations of greenhouse gases, which consist of carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O), but there are many others. The single largest source of GHG emissions is fossil fuel combustion (De Bruyn et al., 2018). In the post-ZB system, the amount of electricity required for the processes as well as for the production of raw materials contributes to high CO_2 emissions, translating to high external costs for global warming.

Finally, the acidification impact category contributes 122.10€ to external costs. Acidification refers to the collective impacts of airborne pollutants that are converted to sulfuric and nitric acid and deposited on soils and vegetation by means of wet or dry deposition. The main source of potentially acidifying emissions is anthropogenic activities like agriculture (particularly livestock farming, NH_3) and fossil fuel consumption (SO_2, NO_x) (De Bruyn et al., 2018). As also mentioned above, the large quantities of sulfuric acid in Site II are mainly responsible for the costs for the acidification.

6.2.3. Total economic performance

The main factors that influence the estimation of internal, transfer and external costs are described above. However, it is essential to investigate the contribution of each type of cost to the overall economic performance of the post –ZB system. Figure 34 outlines the contribution of Capex, Opex, environmental and social costs of DWP and Site I & II to the economic performance of the post-ZB system that occur a year after the system's implementation.

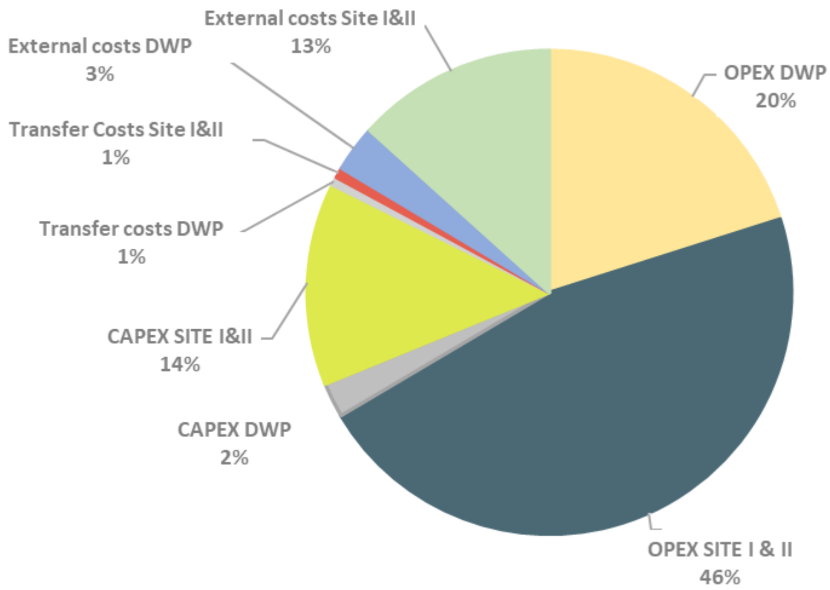

Figure 34. Cost's contribution to the post-ZB system after 1 year of implementation

The operational costs comprise the largest expense for the company. Both DWP and Site I&II operational expenses contribute 66% to the total costs. However, the operational costs of Site I&II are almost twice the DWP's costs. As mentioned earlier, the underlying reasons are the large quantities of Site II consumables and more specifically the amounts of sodium hydroxide and sulfuric acid. Concerning the DWP, the personnel and labor costs contribute the most to its operational expenses. Following, the capital costs along with the external costs of Site I & II contribute by 14% and 13%, respectively. The external costs of Site I & II are due to the environmental consequences that Site I & II induce. Again, the large quantities of consumables required for the operation of Site II are mainly responsible for the high percentage of Site's I & II external costs. Finally, the transfer costs along with the external costs of DWP have a low percentage of 1% and 3%, accordingly.

6.3. Sensitivity analysis

6.3.1. Upscaling of Site I & II

To be able to analyze the environmental and economic impacts of the Site I & II at industrial scale and to compare them with the DWP's impacts, the systems were upscaled. However, the upscaling of processes from demonstration and pilot to industrial scale is a very complex and difficult procedure where many variables need to be taken into account. In this report, linear upscaling was performed. However, during the upscaling procedure, the consumption of raw materials and energy could be improved due to efficient process design or due to the implementation of continuous processing compared to batch processing. For this reason, a sensitivity analysis was performed where the required inputs of raw materials and energy for Site I & II were reduced. The amount of raw materials and energy were reduced by 10-40 % based on the calculated quantities for the linearly upscaled systems. In figure 35, 36, the environmental impacts of the post-ZB system are presented.

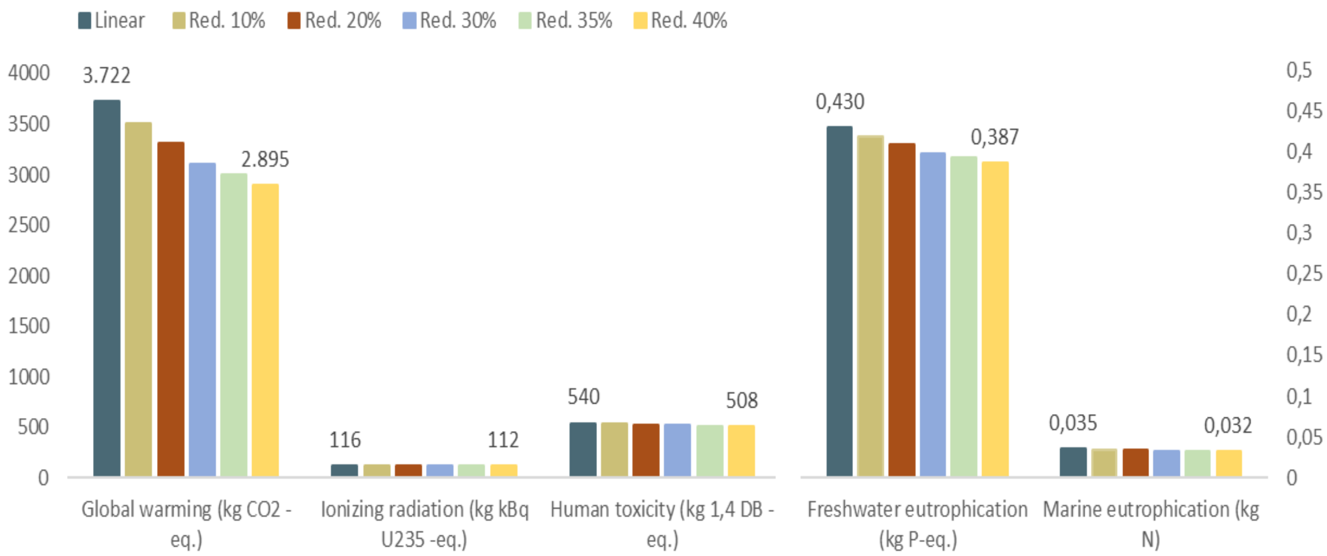

Figure 36. Environmental impacts of post-ZB system per 1000m^{e} demi water – Reduction of material's inputs (Global warming, Ionizing radiation, human toxicity, freshwater & marine eutrophication)

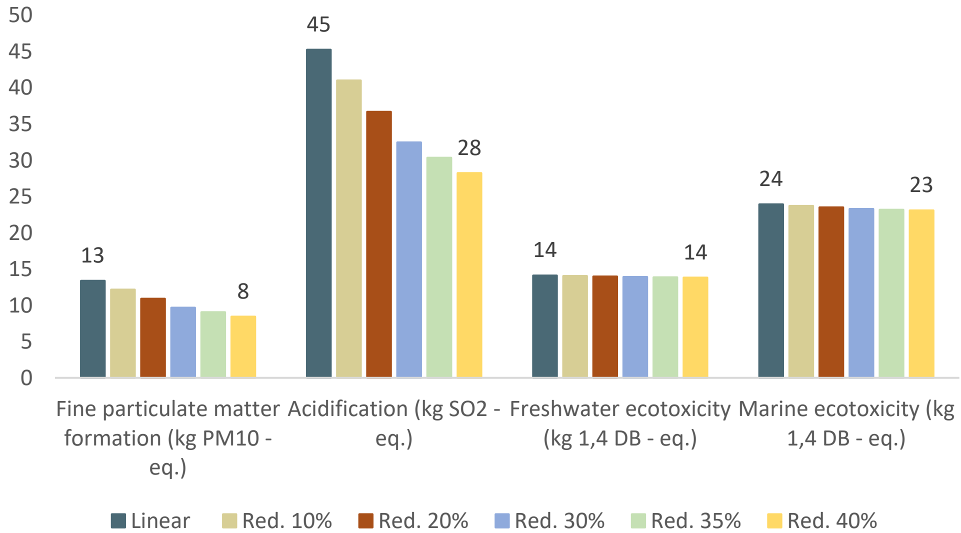

Figure 35. Environmental impacts of post-ZB system per 1000m^{e} demi water – Reduction of material's inputs (Fine particulate matter formation, acidification, freshwater ecotoxicity, marine ecotoxicity)

The environmental burdens of the post –ZB system for the production of 1000m^3 demi water production are decreased when the quantities of the required raw materials are reduced. As it is shown, the environmental burdens are significantly decreased in some impact categories. For example, if the inputs of Site I & II reduced by 40%, global warming, acidification, and particulate matter formation categories are reduced by 22%, 37%, and 38%, respectively. This is reasonable since as it was highlighted in the impact assessment, Site II is mainly responsible for the environmental impacts of these impact categories.

Furthermore, the reduction of raw materials and energy consumption affects not only the environmental impacts but also the costs required for the production of demi water. Figure 37 highlights how the NPV including the PV of transfers and external costs reacts in the reduction of the inputs of Site I & II. As indicated, even though the NPV increases, it is still negative rendering the implementation of Site I & II financially unattractive.

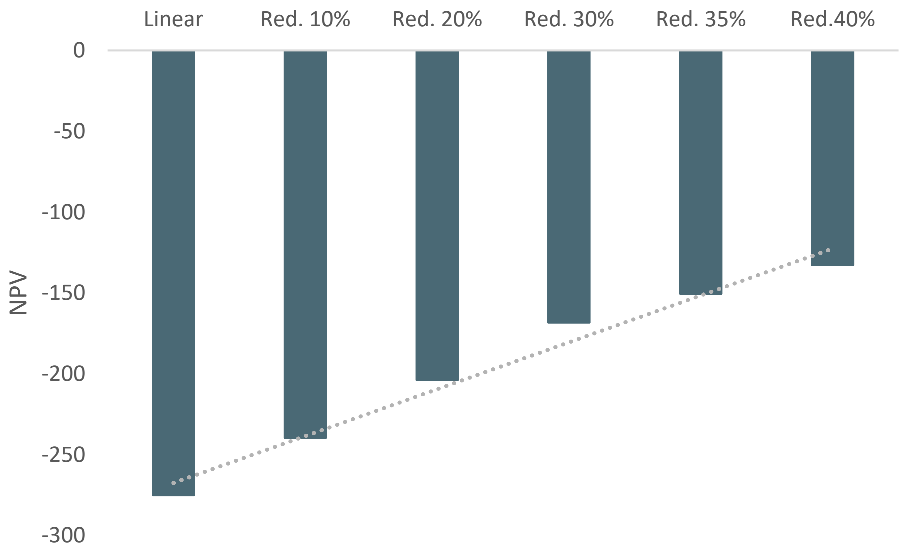

Figure 37. Sensitivity analysis of post-ZB system NPV's – Reduction of raw material's inputs

6.3.2. Direct and indirect emissions

As already discussed, LCA usually involves supply chain emissions related to material and chemical production. On one hand, from an LCA perspective, the identified environmental impacts for the production of demi water are attributed to all the involved supply chain actors. On the other hand, LCC usually takes either the perspective of the manufacturer or the manufacturer together with the supply chain actors or the consumer and user. This report has conducted the LCA and LCC analysis in order to be coherent, from the manufacturer's perspective since the supply chain perspective required intensive data gathering that was not feasible in the context of this master thesis. Nevertheless, the question arising is who has to pay for the environmental as well as the social costs. The cost estimations applied in eLCC and sLCC are based on the environmental impacts of the raw materials and energy consumption on-site, yet produced from other actors in the supply chain. Therefore, it is reasonable to claim that the estimated costs represent the costs of all energy and raw materials manufacturing processes engaged in the Dutch chemical process industry. To that end, all actors involved are jointly but not equally accountable for the identified environmental and social costs. Someone could argue that since there are no emissions attributed directly to Evides, it should not pay environmental and social costs for the production of demi water. In this work, it is assumed that 50% of the identified environmental burdens are attributed to the company. However, this is a disputable subject and lies in the taste and perspective of the analyst. For this reason, sensitivity analysis is performed to calculate the external costs by considering that the company is accountable for 0% to 40% of the burdens since it is not reasonable to attribute more than 50%. As it is shown in Figure 38, the external costs are increasing exponentially with the higher percentage attributed to the manufacturer.

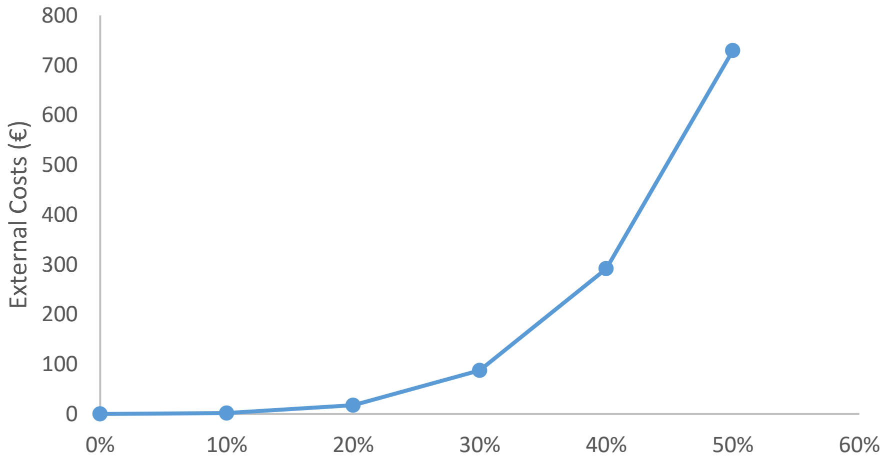

Figure 38. Sensitivity analysis for external costs attributed to the manufacturer per 1000 m^3 demi water

6.3.3. Discount rate

In general, to estimate cash flows that occur at different periods in the life cycle of a system or product, discounting is used. Depending on the system that is analyzed, the goal and the scope of the research, the discount rate could usually vary from 0% to 15%. Generally, the discount rate is slightly higher than the local inflation rate. In order to evaluate the impact of the discount rate's selection, it is appropriate to perform sensitivity analysis. During the cost assessment of a system, it is reasonable to employ a high discount rate for financial estimations and a low one for environmental effects.

In cLCC, a company is interested to be aware of the profit that a technology might yield. It then at least has to deal with the real cost of borrowing. The market rate of interest shows the trustworthiness of the company. Usually, the discount rate for private financing ranges from 5% to 20%. However, in this report, the discount rate set at 4%, following the guidelines of the European Commission. In sLCC, the expected future expenses are underestimated by choosing a high discount rate when an investment decision should be made in the present. For this reason, the costs associated with environmental damages are overlooked and not considered with equal importance as the private costs. The particular study calculated the associated costs with the detected environmental damages by choosing the lowest possible discount rate, which is 0.001%.

As depicted in Figure 39, the NPV is increasing when the level of the discount rate is increasing since the present value of costs is declining. Concerning the external costs, it is of essential importance to be taken into consideration when making a decision because the environmental impacts might cause negative effects. Therefore, the selection of a low discount rate gives incentive to the firm to invest in the present with the aim of preventing higher costs in the future.

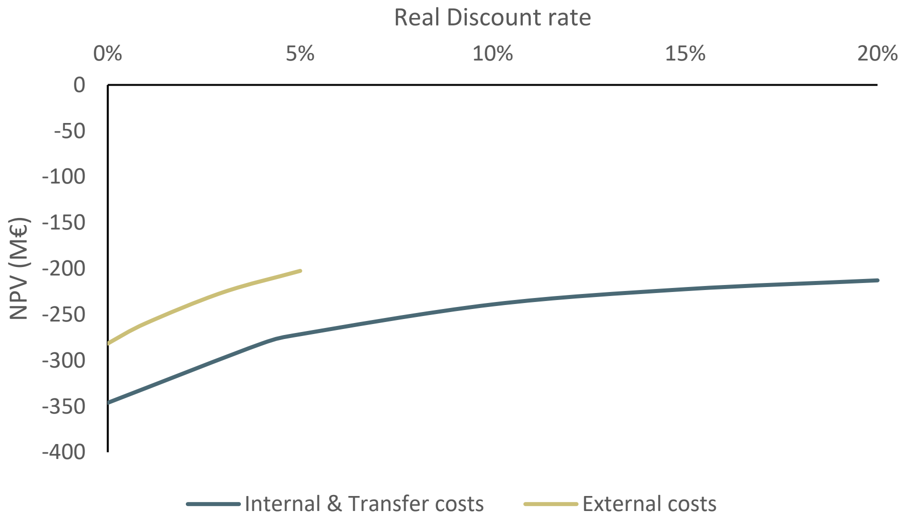

Figure 39. Sensitivity analysis for real discount rate

6.3.4. Economic Data

One main issue regarding the economic inputs used for economic analysis is their reliability. The types of costs used for the estimation post-ZB system's economic performance are divided into three categories; namely internal, transfer and external costs. In this section, sensitivity analysis is performed in regards to internal costs to assess their influence on the economic analysis. Concerning the transfer costs, they comprise only 2% of overall costs and hence they left out of the sensitivity analysis. As far as the external costs are concerned, they highly depend on the weighting factors that are used to translate the environmental impacts into monetary terms. The comparison among different methods for the monetization of the environmental burdens was out of the scope of this research and as a result the selected weighting method used as default.

The internal costs consist of capital and operational costs. The latter is divided into costs for consumables, energy, maintenance, labor, and personnel. In order to identify how the economic values for the different types of costs affect the total economic performance of the post-ZB system, two extreme scenarios were developed and presented in Table 23. These scenarios were developed based on the fluctuations that the economic data could demonstrate in accordance with the baseline scenario, namely the economic inputs used for the analysis in this report. For the prices of consumables and energy, the minimum and maximum prices were selected according to the prices' range found in the literature (see Table 16) for the low and high extreme scenarios, respectively. For the electricity price, the minimum and maximum price selected based on the price's fluctuation for the Dutch non-households consumers for the period of 2010 -2019 (Eurostat, 2019). Regarding the capital costs and personnel costs an increment of ± 50% and ± 30% was chosen based on the baseline scenario. The maintenance and labor costs are altering respectively since they have been calculated as percentages of capital and personnel costs.

Table 20. Different scenarios for the economic analysis of post-ZB system

Costs/Scenarios	Low Extreme	Baseline	High Extreme
Internal Costs			
Capital Costs *(1 year of implementation)*	3.51 M €	7.03 M €	10.55 M €
Operational Costs			
Consumables			
Sodium Chloride	0.07 €/kg	0.135 €/kg	0.2 €/kg
Sodium Hydroxide	0.36 €/kg	0.48 €/kg	0.6 €/kg
Hydrochloric acid	0.15 €/kg	0.235 €/kg	0.32 €/kg
Iron(III) Chloride	0.14 €/kg	0.37 €/kg	0.6 €/kg
Polyacrylamide	4.51 €/kg	4.73 €/kg	4.89 €/kg
Antiscalant (Vitec 3000)	8.13 €/kg	8.345 €/kg	8.56 €/kg
Antiscalant (Vitec 4000)	13.75 €/kg	14.11 €/kg	14.47 €/kg
Sulfuric acid	0.17 €/kg	0.265 €/kg	0.36 €/kg
Energy			
Electricity	0.0603 €/kg	0.0679 €/kWh	0.0769 €/kg
Steam (200^{o}C, 18 bar)	0.02 €/kg	0.025 €/kg	0.03 €/kg
Maintenance	0.11 M €/year	0.21 M €/year	0.32 M €/year
Labor	0.54 M €/year	1.6 M €/year	2.33 M €/year
Personnel	57,190€/year/employee	81,700 €/year/employee	106,210€/year/employee

Figure 40 depicts the contribution of the different types of costs to the internal costs after 1 year of implementation of the post-ZB system for the referred scenarios. In all three scenarios, the costs of consumables comprise the highest expense for the company. These costs depend on the amount of raw materials required and their price. In this report, different sources were employed to obtain reliable estimates regarding the prices. However, chemicals' prices are fluctuating over time and shifting with market supply and demand. Furthermore, each provider offers different prices according to the customer and the required quantities. Even though the chemical's prices are highly influencing the outcomes of the economic analysis, they are quite uncertain and vary among different providers, consumers, geographic location.

Following, also the capital costs constitute a great expenditure for the company. Concerning the implementation of ZB design, the amount of capex utilized for the economic analysis was based on preliminary estimations made on ZB Grant Agreement (Zero Brine, 2017). Although these appraisals are not completely accurate, the Capex of each technology employed in ZB was difficult to be gathered since some of these do not exist in the market. Furthermore, even though each technology provider has given a cost estimate, it concerns the pilot scale and not the full-scale implementation. Regarding the DWP, the data provided concerning the plant's operation concern the overall consumption of consumables and energy and they are not process-specific. Hence, no clear conclusion could be drawn in regards to each equipment's capacity, efficient and energy consumption, resulting in treating the DWP as a "black box". For this reason, it was assumed that the Capex of DWP are 10% of the company's total capital costs for the industrial water production since the water produced from the DWP accounts for 10% of the total industrial water production of Evides (Evides, 2018). Overall, the estimation of the Capex considerably affects the results of this study, making this category of costs a dominant factor for the applied model. However, the limitations mentioned above for this study results in uncertainty regarding the economic values used.

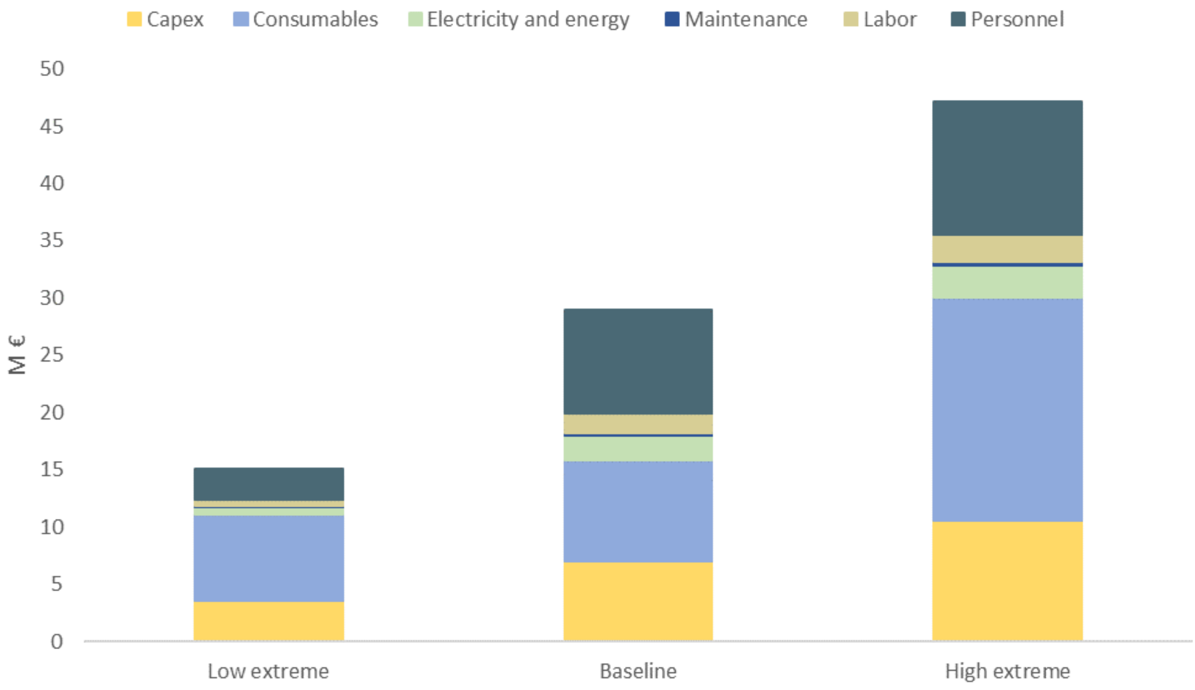

Figure 40. Internal costs of post – ZB for different scenarios after one year of implementation

Personnel costs contribute as well to the overall internal costs with a high percentage, namely 18% to 25% in the different scenarios. These costs are a function of the number of employees and the average personnel cost. Since the average personnel cost was based on the annual report of Evides for 2018 (Evides Waterbedrijf, 2018) and the number of employees was provided by Evides, it can be reasonable to argue that this category of cost comprises a reliable factor. Furthermore, labor costs which contribute 4% to 6% to the overall costs across the employed scenarios, have been calculated as 20% of personnel costs.

Finally, electricity and steam, as well as maintenance, have a low contribution to the overall costs. Regarding electricity and steam prices, they were obtained from sources that concern the production of the Netherlands. Electricity price was based on Eurostat statistics for 2019 for the Netherlands and steam price was based on a report that concerns the analyzed company. Hence, even though they do not significantly influence the results of the economic analysis, the uncertainty regarding those data is low. Concerning maintenance costs, they usually comprise a great expense for the companies. However, in this report, it was found that contributes 1% to the overall costs. The maintenance expenses were estimated as a percentage of capital costs. This choice was based on scientific papers that estimate the economic performance of technologies used in water treatment plants. More specifically, Trieb, F., Moser, M., & Fichter, T. (2012) examined the costs of MED, RO and NF plants and found that maintenance costs comprise almost 3% of the capital expenses in each case. Furthermore, Micari et al. (2019) used the same percentage for the estimation of these costs for the techno-economic assessment of MED for the treatment and recycling of ion exchange resin spent brine. Since these technologies are employed in the post-ZB system, this percentage was selected. However, it is apparent that maintenance costs are dependent on capital costs.

Overall, the most dominant factors that influence the results of the economic analysis are the consumables, the capital, and personnel costs. Consumables and capital costs comprise the most problematic factors in the sense that are quite uncertain due to the choices made and limitations addressed above. In fact, developing a consistent data set for a study can be challenging. To tackle this issue, suggestions will be given in the next section regarding the type of data that should be given from stakeholders and the strategic choices that should be made to increase the preciseness of this kind of economic evaluation.

6.4. Conclusions of Chapter 6

In this chapter, the interpretation of results obtained from LCA and LCC analysis in Chapter 5 was presented. Regarding the LCA, contribution analysis of the post-ZB sub system's and the processes of DWP and Site II was provided to determine the "hot-spots" of the system. Firstly, the post-ZB system was divided into three major sub-systems; the DWP (after the implementation of ZB design), the Site I and Site II. The percentage contribution of each sub-system to the total environmental impacts engaged in the production of 1000 m^3 demi water was obtained. It was concluded that both the DWP and Site II have the greatest impact on the environment with 50.6% and 48.5%, respectively. Site I had nearly zero environmental burdens compared to DWP and Site II.

Following, since Site II and DWP had the highest environmental impacts, process-specific contribution analysis was used to reveal which are the "hot-spots" in their processes. Regarding the DWP, the data provided by the company concerned the overall consumption of consumables and energy in the plant and were not process-specific. However, a conclusion is drawn regarding the raw materials. It was noticed that electricity and stream are mainly responsible for the environmental burdens across all impacts categories comprising more than 50% of the overall burdens across all impacts categories. It is concluded that even though the environmental performance of DWP was enhanced after the application of ZB design, still improvements could be made in terms of energy efficiency. Furthermore, it was acknowledged that the absence of data regarding each process's energy consumption was proved to be a limitation in regards to the identification of the most energy-intensive process of DWP. As far as Site II is concerned, it was observed that the TOC removal process had the greatest contribution to all the impact categories, accounting for 94% to 98% of the overall impact. In particular, the underlying reason behind the bad environmental performance of the TOC removal process found to be the large quantities of sodium hydroxide and sulphuric acid employed. The environmental impact of sodium hydroxide was the largest contributor in all categories except acidification and particulate matter formation, ranging from 66% to 87%. In acidification and particulate matter formation, sulphuric acid comprised 67% and 79% of the overall environmental damages.

Concerning the LCC, the results obtained from the cLCC, eLCC and sLCC were analyzed. For the cLCC, the internal costs and benefits that occur in a year after the system's implementation were shown with each sub system's costs and benefits to be presented. It was observed that the costs for raw materials of Site II represent 96.4% of the overall cost for raw materials and 41% of the total internal costs. To that end, it is concluded that the bad economic performance of the post-ZB system is assigned to the Site II consumables. The large quantities of sodium hydroxide and sulfuric acid were again behind the poor economic performance of Site II. Therefore, in order for the company to reduce the costs and the environmental impact of the chemicals at Site II, the reduction of their quantities is required.

Even though the poor economic performance of the post-ZB was proved to mainly attributed to the internal costs, valuable insights were gained by analyzing the results of eLCC and sLCC. The eLCC analysis consisted of costs related to energy consumption and CO_2 emissions. It was found the costs for the CO_2 tax were significantly higher. Furthermore, the highest external costs in sLCC concerned the particulate matter formation, global warming, and acidification categories. It was found that the large quantities of sulfuric acid utilized in Site II are responsible for the high external costs of particulate matter and acidification impact categories. Regarding the global warming potential, the amount of electricity required for the processes as well as for the production of raw materials contributes to high CO_2 emissions.

For the total economic performance of the post-ZB system, it was observed that it is the implementation of Site I & II rendered the project economical unsustainable. The contribution of Capex, Opex, environmental and social costs of DWP and Site I & II to the economic performance of the post-ZB system were presented. It was observed that operational costs comprise the largest expense for the company. Both DWP and Site I&II operational expenses contribute 66% to the total costs. However, the operational costs of Site I&II are almost twice the DWP's costs. The capital costs along with the external costs of Site I & II contribute by 14% and 13%, respectively. Furthermore, the transfer costs along with the external costs of DWP have a low percentage of 1% and 3%, accordingly.

Finally, sensitivity analysis was carried out for the assumptions that were made regarding the upscaling of Site I & II, the emissions attributed to the company, the selected discount rates, and the economic data used. Regarding the upscaling of Site I & II, the amount of raw materials and energy were reduced by 10-40 % based on the calculated quantities for the linearly upscaled systems. It was found that environmental burdens for the production of 1000m^3 demi water are significantly decreased in some impact categories. However, by estimating the NPV with the reducted quantities of consumables, it was found that the NPV is still negative rendering the implementation of Site I & II financially unattractive.

With respect to the economic inputs used for the economic analysis, two extreme scenarios were developed to identify how the economic values for the different types of costs affect the total economic performance of the post-ZB system and their reliability. The choices for the presented economic data were explicitly defined. It was concluded that the most dominant factors that influence the results of the economic analysis are the consumables, the capital, and personnel costs. Consumables and capital costs comprise the most problematic factors in the sense that are quite uncertain due to the choices made and the limitations of this study.

Chapter 7

Conclusions and Recommendations

In this final chapter, the results of this research and the general conclusions are discussed. In Section 7.1, all sub-research questions are answered individually, providing an answer to the main research question in Section 7.2. Additionally, the research contribution of this study is presented in Section 7.3. In Section 7.4, how the results of this study can be generalized is discussed. Next, the limitations of the study are provided in Section 7.5. This is followed by propositions for future research possibilities in Section 7.6.

7.1. Answers to Sub-Questions

SQ1. How to perform an environmental and economic assessment in terms of LCA and LCC techniques?

To provide an answer in this sub-question, the identification of the LCA's and LCC's fundamental theories and principles took place. The approach to integrate the LCA and LCC techniques was found in the literature and concerns their parallel implementation based on the ISO standards given for the LCA structure. Furthermore, the three types of LCC; cLCC, eLCC and sLCC, were also included in the analysis by identifying their key characteristics from the literature and then including them in the integrated LCA and LCC approach. All these concepts were incorporated by employing the environmental impact assessment results to eLCC and sLCC. On the one hand, the costs that are anticipated to be internalized in the decision-relevant future were included in the eLCC while on the other hand, the monetized environmental impacts were incorporated in sLCC. For the monetization of the external costs, LCA-based economic weighting factors were used. In light of this approach, all three types of LCC are utilized to perform the economic analysis of the system, including the assessment of internal, transfer and external costs. Concerning the application of LCA, the environmental impacts of the system were quantified.

Therefore, the four phases of the environmental and economic analysis are the following. First of all, the goal and the scope of this research are explicitly defined in accordance with the objective. Subsequently, the life cycle inventory was established, including all the inflows and outflows in terms of raw material, energy and money. Then, the quantification of the environmental burdens of the analyzed system was performed by employing a suitable LCA software and selecting the impact assessment method. Therefore, the acquired results are translated into monetary terms. For that purpose, internal, transfer and external costs are gathered and estimated for a certain time period. The estimation of the anticipated future cash flows was performed by choosing a suitable discount rate. Lastly, the interpretation along with the discussion of the results is conducted.

For the monetization of the environmental damages, two fundamental principles are generally applied, namely WTP and WTA. Based on these principles, four approaches that further divided into various valuation methods are found with the purpose of monetizing the environmental impacts. These valuation methods are used for the creation of economic weighting factors suitable for implementation in the LCA. Hence, several LCA models have developed their one economic weighting set, such as ReCiPe, LIME, Stepwise, Ecovalue08, and Ecotax02.

However, after thorough literature research, it was found that the identified weighting sets are incomplete due to the following reasons. Firstly, the identified weighting sets do not incorporate factors for the monetization of all LCA midpoint and endpoint indicators, signifying the inconsistency of these factors in terms of applying them in all impact categories. In addition, cultural, social and economic differences among the studies cause the deviation of the results. Therefore, the generalizability and representativeness of the sets are heavily influenced due to their different socio-cultural orientation. Finally, the aforementioned weighting sets concern data that are relatively old.

Therefore, in order to overcome this problem, research was conducted to find LCA-based economic weighting factors that are not outdated and are found on Dutch statistics, since this report deals with a case study regarding the Dutch chemical process industry. As such, the weighting set used in this report concern average values for emissions in 2015 for the Netherlands and is provided by CE Delft.

SQ2. Which are the key differences among the conventional, environmental and societal LCC?

To resolve this question the key characteristics of three LCC types were investigated. cLCC is strictly built upon economic evaluation and includes all the expenses related to a product's or system's life cycle. These costs are covered by the manufacturer or the consumer. Furthermore, the evaluation incorporates only real and internal costs and there is no need for parallel LCA analysis. The perspective of this analysis is that of one actor, namely the producer or the consumer or the user.

Concerning eLCC, it follows the structure of the LCA. The system boundaries along with the functional unit correspond to those of the LCA's as eLCC also deals with the entire life cycle of a system or product. eLCC enhances cLCC by including, on the one hand, all life cycle phases and anticipated costs, and, on the other hand, separate not-monetized LCA results. eLCC concerns the assessment of all costs related to the life cycle of a product or system which are covered by one or several stakeholders throughout the life cycle. Finally, it also may include subsidies and taxes, if they are relevant for the analysis.

Furthermore, sLCC contains a wider range of costs, incorporating all the costs that are significant in the long term for all the actors that are directly and indirectly involved. One major distinction between sLCC and both cLCC and eLCC is the inclusion of governments and public agencies in the group of stakeholders, even though they are not directly involved in the analyzing product or system. To that end, transfer costs like taxes and subsidies are not incorporated into the system since they are internal to it. Additionally, sLCC involves the environmental impacts of the analyzed product or system by monetizing them and it may be complemented by an LCA.

Overall, in eLCC non-monetary impacts are not translated to monetary units since they are included in the LCA impact assessment as environmental terms. sLCC includes the monetization of external costs both in terms of environmental and social impacts. Such an approach contradicts the nature of cLCC and eLCC, which include only real costs covered by one or several stakeholders in the product's life cycle, despite the anticipated costs involved in eLCC. Moreover, all the eLCC and additional external costs that are based on WTP methods are included in the sLCC. Finally, in contrast to eLCC, transfer costs have no net cost effect and as a result, are not incorporated in sLCC.

SQ3. Which LCC type is the most appropriate for the Dutch Zero Brine case study?

Three categories of costs for the implementation of the three types of LCC are distinguished in this report; internal, transfer and external costs. Internal costs are included in all three types of LCC, transfer costs in eLCC and external costs in the sLCC. To that end, it can be concluded that since the costs captured in cLCC are also included in the other two types, the inclusion of cLCC gives nothing more to the analysis and thus it could be excluded. Taking this into account, eLCC along with sLCC are the remaining options.

Concerning eLCC, as it is mentioned it includes transfer costs. These costs incorporate the expenses that the company has to pay for taxes and fees. In this specific case study, the taxes for electricity and CO_2 emissions are calculated and are amount to 0.55 M € per year. On the other hand, sLCC does not include transfer costs as they are internal to the system but incorporates the external costs by monetizing the environmental emissions resulted from the LCA analysis of the post-ZB system. These costs add up to 7.29 M € per year. It is obvious that transfer costs are significantly lower than external costs and hence someone could argue that they could be neglected from the analysis of the post-ZB system. However, this is not the case since they are giving different kinds of information and result in valuable insights about the company's economic performance. Consequently, it is argued that both eLCC and sLCC are appropriate for the Dutch Zero Brine case study and should be included in the analysis.

SQ4. Could the Zero Brine applications enhance the environmental and economic performance of the analyzed Demineralized Water Plant's production scheme?

As far as environmental performance is concerned, it is concluded that the implementation of ZB systems has ambiguous results. On the one hand, the majority of the environmental impacts were decreased by 15% to 22%. Specifically, ionizing radiation, human toxicity, marine eutrophication, freshwater, and marine ecotoxicity categories show better environmental performance after the implementation of ZB design. On the other hand, global warming, acidification as well as particulate matter formation categories were sharply increased in the post – ZB system. In particular, the global warming impact category showed an increase of 114% and ionizing radiation and acidification were risen by thirteen and fifteen times, respectively. Regarding DWP itself, its performance shows an increase of 77% due to the fact that ZB applications reduce its requirements of sodium chloride, water, and electricity. However, this increase does not counterbalance the environmental damages of the ZB system in global warming, particulate matter formation, and acidification categories.

ZB applications incorporate two different systems; Site I and Site II. Through contribution analysis, it is found that Site's I contribution is nearly zero while Site II is responsible for almost 50% of the environmental performance of the post-ZB system. The main reason behind this is the large requirements of electricity and raw materials in order to treat the brine stream that derives from the DWP. Specifically, one particular process in Site II; TOC removal, contributes to its environmental damages by more than 94%. Hence, it is concluded that the bad environmental performance of Site II is attributed to the TOC removal process.

From the economic assessment results, we can conclude that the post-ZB system is not financially viable. By estimating the NPV after the implementation of ZB applications, it is observed that it is negative thus rendering the project unsustainable. Even though, Site I & II increase the benefits of DWP by 7%, the costs for their operation are significant high. By examining the costs of the post-ZB system, it is observed that Site II is also accountable for the poor economic performance. Again, the large quantities of consumables required for the TOC removal process are the main cost contributor.

Overall, with the implementation of ZB systems; namely Site I and Site II, the environmental performance slightly increases in the majority of the impacts categories, however, it also sharply decreases in the remaining ones. Furthermore, the economic performance of the DWP's production scheme is not enhanced, rather it gets worse. The main reason behind this is the operation of Site II and more specifically the large quantities of raw materials for the TOC removal process.

7.2. Answer to Main Research Question

The current research focused on the investigation of the environmental and economic performance of the Dutch ZB case study by following a life cycle thinking. To achieve this, the LCA and LCC tools were applied by incorporating all three types of LCC; cLCC, eLCC and sLCC.

The main research question of the master thesis was:

"Which is the most efficient approach to identify the environmental and economic performance of the Dutch Zero Brine case study, in terms of environmental and economic assessment techniques?"

This research question was answered by merging the answers from the stated sub-questions. In summary, this research has shown how the parallel implementation of LCA and LCC techniques as well as the incorporation of the three different types of the LCC in one study was applied to the Dutch ZB case study. Specifically, since both eLCC and sLCC concepts are relatively new fields of research, with many different approaches present in the literature and room for further development, a lot was left up to the interpretation of the researcher on how to conduct such a study. By applying all the above-mentioned concepts, the main goal of the research – to determine the environmental and economic performance of the ZB project – was achieved. As a result of answering this main research question, valuable insights were noted and reflected upon which may prove useful not only to ZB stakeholders but also to LCA and LCC practitioners and business decision-makers.

7.3. Research Contribution

7.3.1. Scientific and Social Contribution

First of all, the relevance of this report lies in the fact that the knowledge gaps identified in Section 1.2 were filled. Scientific research into sLCC is currently at an initial and developing phase due to the immature and intangible nature of the social dimension of sustainability. Moreover, even though eLCC is more developed than sLCC and guidelines have been published for its implementation, there is a misunderstanding in the scientific community what each LCC type includes. Hence, academia calls for more case studies and methodologies to strengthen the knowledge and reliability in terms of their application in practice. This presented research contributes by thorough examining the differences of the LCC types and proposing an approach to incorporate then in a study. Furthermore, their application to the Chemical process industry is new, which is something that increases the contribution of this work as well. Furthermore, the incorporation and application of the LCA and LCC tools in order to evaluate the environmental and economic efficiency of the Dutch ZB case study contributes to the literature. In particular, the input of this project is enhanced by applying the aforementioned life cycle thinking tools in a country and industry that are missing from the scientific publications.

Following, critical aspects found in the literature regarding the LCA and LCC methodology were also encountered during the implementation of LCA and LCC tools in this study. For example, data gathering comprised a challenge regarding the large amount of necessary data, taking considerable time and resulting in difficulties to incorporate all the required data in high quality and accurate manner. Furthermore, concerning the interpretation of the environmental impacts, it highly depends on the researcher and thus making room for inconsistencies throughout the study and limitations for future applications. Besides these challenges, new issues have arisen during this research such as the inclusion of social impacts in the sLCC or the reliability of the inputs in economic analysis. Consequently, this report foster scholars to transform these issues into research problems.

Finally, this research contributes by bridging the gap between academia and practice, taking the perspective of the company and applying an approach for the incorporation of LCA and LCC to be used as a tool for business practitioners. Through the application of the proposed LCA and LCC approach companies will be able to identify, measure and manage their environmental and economic sustainability and integrate this approach into operational and strategic decision-making. Therefore, this research aimed at including managerial needs in academic research.

7.3.2. Managerial Relevance

As mentioned above, the applied LCA and LCC methodology could be utilized as a tool for decision-making in companies that are keen on incorporating environmental and economic sustainability in their processes. In recent years, the concept of sustainability is gaining ground and comprise an essential goal for various organizations by allowing them to gather, assess and convey information about their sustainability. In this way, they get an overview of their current state and support the management decisions regarding their advancement over the years. The methodology implemented in this report allows the assessment of the environmental and economic performance through a number of well-defined steps that could be used again according to the context of the analysis. The proposed methodology has a two-fold purpose for decision-makers; firstly to convey the essential information about the consequences of their activities and secondly to apply this information as an entry point to decide which actions should be taken to be environmental and economic sustainable.

Furthermore, as addressed above, the current research gives insights on what challenges practitioners may face during the implementation of LCA and LCC methodology. However, it is claimed that this also stands for the decision-makers from organizations that intend to use this methodology. By anticipating possible challenges that the implementation of such a methodology may incur, companies could premediate these issues resulting in the elimination or minimization of such challenges.

7.4. Generalization of results

The LCA and LCC methodology applied in this thesis was developed and implemented specifically in the ZB project, with the analyzed company (Evides) being a part of the Dutch chemical process industry. Regarding the data gathering, all the factors utilized in the applied model were explicitly defined and were well-documented by providing the collected data material and references. The required information for the environmental analysis was obtained by the partners of the ZB project. Concerning the economic analysis, the data were derived from different sources and based on various assumptions since the required inputs for the model were not given the company itself as they might be business sensitive. To strengthen the reliability and validity of the economic inputs, the triangulation of data was achieved by employing more than one source, namely scientific papers, websites, market reports, databases, annual reports, and newspapers. Considerable effort was put to find various sources of evidence so as to acquire accurate data and information.

Overall, it is argued that the applied methodology could be utilized from other organizations in the water industry as well as the chemical industry not only in the Netherlands but also in other countries. It may be utilized by managers and decisions makers for assessing the environmental and economic sustainability in a manner that is purposeful to them. Nonetheless, it is evident that alterations will possibly be needed with respect to the analyzed processes, technologies and systems configurations. An important issue that needs to be addressed is the type of data required for economic analysis. As highlighted in section 6.3.4, the prices of chemicals, as well as the Capex expenditures, comprise the most dominant factors that affect the results of the performed economic analysis. Companies are often reluctant to give this kind of information. On the one hand, when companies performing an economic analysis for themselves, they have access to these types of data and with the application of the proposed model they could arrive at reliable and valuable results. On the other hand, LCC practitioners or decision-makers that are outside of the company's boundaries will confront the challenges that were mentioned regarding the availability and reliability of the data. However, it is argued that through triangulation of data, transparent methodological choices and sensitivity and uncertainty analysis the uncertainty could be reduced and reliable results to be obtained.

7.5. Limitations

This project has its constraints. First of all, the environmental assessment of the DWP before the implementation of ZB systems did not include the effect of the brine streams that are discharged into the seawater. In LCIA models in Simapro software, the environmental impacts of saline effluent are ignored and not included in the marine and aquatic impact categories. Therefore, the environmental damages of these categories are underestimated and we cannot claim that results regarding the environmental performance of the pre-ZB system are completely representative. Furthermore, the data obtained for the operation of Site I & II are based on bench-scale tests and computer simulations and thus may not completely portray the actual operating information of the pilot-scale plants. In addition, the upscaling performed for Site I & II was based on various assumptions. To that end, the quantities of energy and raw materials that will be required for the actual upscaled Site I & II may vary, with fewer materials probably to be employed.

Concerning the assumption of a steady-state production system, it is not representative of the actual performance of the system. In a dynamic environment, firms strive to address the customer's requirements by constantly adapting to the state of the market. Therefore, considering the price's variation along with the feedstock's availability that are critical for producing a product or a system, the steady-state assumption does not reflect the real market circumstances. Furthermore, there are constraints regarding the scope of the analysis. The exclusion of use and disposal phases from the system boundaries prohibits us to assess the overall efficiency of the ZB project. In addition, for the analysis of the environmental impacts, ReCiPe was the only method employed. Even though this method has global coverage, the credibility of results is influenced by not incorporating other impact assessment methods.

Concerning the economic modeling, the choice of analyzing the system from the manufacturer's perspective has neglected the impact that each decision has in regards to the relevant actors in the supply chain. To that end, conclusions cannot be made for the beneficial or adverse impacts that the analyzed system may bring to the other stakeholders. Consequently, it is not sure that our recommendations will be compatible with everyone's interests. Overall, as noted in the previous sections the validity of the results is highly based on the type of dataset provided by third parties. Apart from that, since the sample of the data will cover a specific period of time, the generalization of the results might not be accurate after some years.

7.6. Recommendations for future research

Taking into consideration the limitations of the study, it is considered appropriate to give suggestions for future research. Regarding the ZB project, considerable research has to be made in order to make the applications more sustainable with regard to environmental and economic efficiency. In particular, the “hot-spots” of the ZB system and DWP were identified in this report. Hence, improvement needs to be made regarding the overall design of ZB applications before its implementation. Moreover, decision-makers in Evides could take into account the processes that need improvement in the DWP with regards to its environmental performance. Consequently, it would be useful to examine different practices to be applied or the modification of the proposed ones. Moreover, a wider scope of LCA and LCC is important, whereupon the actions of all stakeholders involved in the production of demi water could elucidate in what way alterations in one industry might influence the other. As it was stated in limitations different stakeholders have different perspectives of what is environmentally and financially sustainable for them.

REFERENCES

A, R., Pati, R., & Padhi, S. (2019). Sustainable supply chain management in the chemical industry: Evolution, opportunities, and challenges. *Resources, Conservation And Recycling*, *149*, 275-291. doi: 10.1016/j.resconrec.2019.05.020

Ahlroth, S., Nilsson, M., Finnveden, G., Hjelm, O., & Hochschorner, E. (2011). Weighting and valuation in selected environmental systems analysis tools–suggestions for further developments. Journal of Cleaner Production, 19(2-3), 145-156.

Ahmad, N., & Baddour, R. E. (2014). A review of sources, effects, disposal methods, and regulations of brine into marine environments. *Ocean & Coastal Management*, *87*, 1-7.

Ahmed, M., Arakel, A., Hoey, D., Thumarukudy, M., Goosen, M., Al-Haddabi, M., & Al-Belushi, A. (2003). Feasibility of salt production from inland RO desalination plant reject brine: A case study. *Desalination*, *158*(1-3), 109-117. doi: 10.1016/s0011-9164(03)00441-7

Ahmed, M., Shayya, W. H., Hoey, D., Mahendran, A., Morris, R., & Al-Handaly, J. (2000). Use of evaporation ponds for brine disposal in desalination plants. *Desalination*, *130*(2), 155-168.

Angelakoglou, K., & Gaidajis, G. (2015). A review of methods contributing to the assessment of the environmental sustainability of industrial systems. *Journal Of Cleaner Production*, *108*, 725-747. doi: 10.1016/j.jclepro.2015.06.094

Ariono, D., Purwasasmita, M., & Wenten, I. (2016). Brine Effluents: Characteristics, Environmental Impacts, and Their Handling. Journal Of Engineering And Technological Sciences, 48(4), 367-387. doi: 10.5614/j.eng.technol.sci.2016.48.4.1

Aryanti, P. T. P., Yustiana, R., Purnama, R. E. D., & Wenten, I. G. (2015). Performance and characterization of PEG400 modified PVC ultrafiltration membrane.

Attia, N., Jawad, M., & Al-Saffar, A. (2015). The integration of desalination plants and mineral production. *Desalination And Water Treatment*, *57*(45), 21201-21210. doi: 10.1080/19443994.2015.1115376

Auer, J., Bey, N., & Schäfer, J. M. (2017). Combined Life Cycle Assessment and Life Cycle Costing in the Eco-Care-Matrix: A case study on the performance of a modernized manufacturing system for glass containers. *Journal of Cleaner production*, *141*, 99-109.

Badruzzaman, M., Oppenheimer, J., Adham, S., & Kumar, M. (2009). Innovative beneficial reuse of reverse osmosis concentrate using bipolar membrane electrodialysis and electrochlorination processes. *Journal Of Membrane Science*, *326*(2), 392-399. doi: 10.1016/j.memsci.2008.10.018

Bakshi, B. (2011). The path to a sustainable chemical industry: progress and problems. *Current Opinion In Chemical Engineering*, *1*(1), 64-68. doi: 10.1016/j.coche.2011.07.004

Barranco, C. R., Balbuena, M. B., García, P. G., & Fernández, A. G. (2001). Management of spent brines or osmotic solutions. *Journal of Food Engineering*, *49*(2-3), 237-246.

Beery, M., & Repke, J. U. (2010). Sustainability analysis of different SWRO pre-treatment alternatives. Desalination and water treatment, 16(1-3), 218-228.

Bierer, A., Götze, U., Meynerts, L., & Sygulla, R. (2015). Integrating life cycle costing and life cycle assessment using extended material flow cost accounting. *Journal of Cleaner Production*, *108*, 1289-1301.

Boardman, A., Greenberg, D., Vining, A., Weimer, D.: Cost-Benefit Analysis, 4th edn. Prentice

Bonton, A., Bouchard, C., Barbeau, B., & Jedrzejak, S. (2012). Comparative life cycle assessment of water treatment plants. Desalination, 284, 42-54.

Brandhuber, P., Cerone, J., Kwan, P., Moore, E.L. and Vieira, A. (2007) A Look at Conventional and Emerging Brine Disposal and Waste Minimization Technologies. HDR Waterscapes, 19, 7-10.

Brundtland, G. (1987). Report of the World Commission on Environment and Development: Our Common Future. United Nations General Assembly document A/42/427.

Business.gov.nl. (2019). Retrieved 28 October 2019, from https://business.gov.nl/

Cartier, S., Theoleyre, M. A., & Decloux, M. (1997). Treatment of sugar decolorizing resin regeneration waste using nanofiltration. *Desalination*, *113*(1), 7-17.

Choe, J. K., Bergquist, A. M., Jeong, S., Guest, J. S., Werth, C. J., & Strathmann, T. J. (2015). Performance and life cycle environmental benefits of recycling spent ion exchange brines by catalytic treatment of nitrate. *Water research*, *80*, 267-280.

Climate deal makes halving carbon emissions feasible and affordable. (2019). Retrieved 28 October 2019, from https://www.government.nl/latest/news/2019/06/28/climate-deal-makes-halving-carbon-emissions-feasible-and-affordable

Damania, R., Desbureaux, S., Hyland, M., Islam, A., Moore, S., & Rodella-Boitreaud, A. et al. (2017). Uncharted Waters: The New Economics of Water Scarcity and Variability. Washington, DC: World Bank.

De Schryver, A. M., Brakkee, K. W., Goedkoop, M. J., & Huijbregts, M. A. J. (2009). Characterization Factors for Global Warming in Life Cycle Assessment Based on Damages to Humans and Ecosystems. *Environmental Science & Technology*, *43*(6), 1689–1695. https://doi.org/10.1021/es800456m

De Bruyn, S., Ahdour, S., Bijleveld, M., de Graaff, L., Schep, E., Schroten, A., & Vergeer, R. (2018). Environmental Prices Handbook 2017: Methods and numbers for valuation of environmental impacts. *Delft: CE Delft*, 05-2018.

Diafonidis, O. (2019). Comparative enviro-economic impact assessment of Dutch chlorine production systems: Investigating different methods for monetizing environmental externalities (Master Thesis). TU Delft.

Długołęcki, P., Gambier, A., Nijmeijer, K., & Wessling, M. (2009). Practical potential of reverse electrodialysis as process for sustainable energy generation. *Environmental science & technology*, *43*(17), 6888-6894.

Edwards, J., Othman, M., Crossin, E., & Burn, S. (2018). Life cycle assessment to compare the environmental impact of seven contemporary food waste management systems. Bioresource technology, 248, 156-173.

Eldh, P., & Johansson, J. (2006). Weighting in LCA based on ecotaxes-development of a mid-point method and experiences from case studies. The International Journal of Life Cycle Assessment, 11(1), 81-88.

El-Nashar, A. M. (2001). Cogeneration for power and desalination—state of the art review. Desalination, 134(1-3), 7-28.

European Commission. (2014). *Guide to Cost-benefit Analysis of Investment Projects: Economic appraisal tool for Cohesion Policy 2014-2020*. *Publications Office of the European Union*. https://doi.org/10.2776/97516

Eurostat. (2019). Electricity prices for non-household consumers - bi-annual data (from 2007 onwards). Retrieved June 23, 2019, from https://appsso.eurostat.ec.europa.eu/nui/submitViewTableAction.do

Evides Waterbedrijf, 2017. Evides Waterbedrijf – Annual report 2018 – Summary.

FAO. (2017). The future of food and agriculture – Trends and challenges. Rome.

Fauzi, R. T., Lavoie, P., Sorelli, L., Heidari, M. D., & Amor, B. (2019). Exploring the Current Challenges and Opportunities of Life Cycle Sustainability Assessment. *Sustainability*, *11*(3), 636.

Fauzi, R. T., Lavoie, P., Sorelli, L., Heidari, M. D., & Amor, B. (2019). Exploring the Current Challenges and Opportunities of Life Cycle Sustainability Assessment. *Sustainability*, *11*(3), 636.

Fernández-Torres, M. J., Randall, D. G., Melamu, R., & Von Blottnitz, H. (2012). A comparative life cycle assessment of eutectic freeze crystallisation and evaporative crystallisation for the treatment of saline wastewater. *Desalination*, *306*, 17-23.

Friedrich, E., Pillay, S., & Buckley, C. (2010). The use of LCA in the water industry and the case for an environmental performance indicator. *Water SA*, *33*(4). doi: 10.4314/wsa.v33i4.52938

Gallego, A., Hospido, A., Moreira, M. T., & Feijoo, G. (2008). Environmental performance of wastewater treatment plants for small populations. *Resources, Conservation and Recycling*, *52*(6), 931-940.

Garcia-Muiña, F., González-Sánchez, R., Ferrari, A., & Settembre-Blundo, D. (2018). The Paradigms of Industry 4.0 and Circular Economy as Enabling Drivers for the Competitiveness of Businesses and Territories: The Case of an Italian Ceramic Tiles Manufacturing Company. Social Sciences, 7(12), 255.

Ghasemipanah, K. (2013). Treatment of ion-exchange resins regeneration wastewater using reverse osmosis method for reuse. *Desalination and Water Treatment*, *51*(25-27), 5179-5183.

Gilron, J., Folkman, Y., Savliev, R., Waisman, M., & Kedem, O. (2003). WAIV—wind aided intensified evaporation for reduction of desalination brine volume. *Desalination*, *158*(1-3), 205-214.

Global CCS Institute. (2018). Rotterdam Opslag en Afvang Demonstratieproject. Retrieved from https://www.globalccsinstitute.com/wp-content/uploads/2019/09/ROAD-Close-Out-Report-on-Capture-and-Compression-final.pdf

Goedkoop, M., Heijungs, R., Huijbregts, M., De Schryver, A., Struijs, J., & Van Zelm, R. (2013). *ReCiPE 2008: A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level (version 1.08), Report I: Characterisation, first ed*.

Gryta, M. (2008). Fouling in direct contact membrane distillation process. *Journal of membrane science*, *325*(1), 383-394.

Gryta, M., Karakulski, K., Tomaszewska, M., & Morawski, A. (2005). Treatment of effluents from the regeneration of ion exchangers using the MD process. *Desalination*, *180*(1-3), 173-180. doi: 10.1016/j.desal.2005.01.004

Guinée, J. B. (2002). Handbook on life cycle assessment operational guide to the ISO standards. *The international journal of life cycle assessment*, *7*(5), 311-313.

Heijungs R. & Guinée J.B. (2015), Some fundamentals on ALCA and CLCA. In: Blanc I. (Ed.) *EcoSD Annual Workshop. Consequential LCA.* Paris: Presses des Mines. 41-48.

Heijungs, R., Settanni, E., & Guinée, J. (2013). Toward a computational structure for life cycle sustainability analysis: unifying LCA and LCC. *The International Journal of Life Cycle Assessment*, *18*(9), 1722-1733.

Heimersson, S., Svanström, M., & Ekvall, T. (2019). Opportunities of consequential and attributional modelling in life cycle assessment of wastewater and sludge management. *Journal of Cleaner Production*, *222*, 242-251.

Himma, N. F., Anisah, S., Prasetya, N., & Wenten, I. G. (2016). Advances in preparation, modification, and application of polypropylene membrane. *Journal of Polymer Engineering*, *36*(4), 329-362.

Hiremath, T., Roberts, D. J., Lin, X., Clifford, D. A., Gillogly, T. E., & Lehman, S. G. (2006). Biological treatment of perchlorate in spent ISEP ion-exchange brine. *Environmental engineering science*, *23*(6), 1009-1016.

Hoepner, T., & Lattemann, S. (2003). Chemical impacts from seawater desalination plants—a case study of the northern Red Sea. Desalination, 152(1-3), 133-140.

Hunkeler, D., & Rebitzer, G. (2003). Life Cycle costing—paving the road to sustainable development?. *The international journal of life cycle assessment*, *8*(2), 109-110.

Hunkeler, D., Lichtenvort, K., & Rebitzer, G. (2008). Environmental life cycle costing. Crc press.

Huppes, G., Kleijn, R., Huele, R., Ekins, P., Shaw, B., Esders, M., & Schaltegger, S. (2008). Measuring eco-innovation: framework and typology of indicators based on causal chains: final report of the ECODRIVE Project.

Huysegoms, L., Rousseau, S., & Cappuyns, V. (2018). Friends or foes? Monetized Life Cycle Assessment and Cost-Benefit Analysis of the site remediation of a former gas plant. Science of the Total Environment, 619, 258-271.

Hydrochloric Acid Europe & US Prices, Markets & Analysis | ICIS. (2019). Retrieved 10 November 2019, from https://www.icis.com/explore/commodities/chemicals/hydrochloric-acid/europe-us/

Ihr Onlineshop für Wasseraufbereitungsanlagen und Zubehör. (2019). Retrieved 12 November 2019, from https://www.wasseraufbereitung-shop24.de

IHS Markit. (2018). *Global Chlor-alkali Market Outlook*. Mexico.

ISO. (2006). ISO 14044, Environmental Management - Life Cycle Assessment - Requirements and Guidelines. Switzerland, Geneva.

Itsubo, N., Sakagami, M., Washida, T., Kokubu, K., & Inaba, A. (2004). Weighting across safeguard subjects for LCIA through the application of conjoint analysis. *The International Journal of Life Cycle Assessment*, *9*(3), 196–205. https://doi.org/10.1007/BF02994194

Ji, X., Curcio, E., Al Obaidani, S., Di Profio, G., Fontananova, E., & Drioli, E. (2010). Membrane distillation-crystallization of seawater reverse osmosis brines. *Separation And Purification Technology*, *71*(1), 76-82. doi: 10.1016/j.seppur.2009.11.004

Jones, E., Qadir, M., van Vliet, M., Smakhtin, V., & Kang, S. (2019). The state of desalination and brine production: A global outlook. Science Of The Total Environment, 657, 1343-1356. doi: 10.1016/j.scitotenv.2018.12.076

Kabsch-Korbutowicz, M., Wisniewski, J., Łakomska, S., & Urbanowska, A. (2011). Application of UF, NF and ED in natural organic matter removal from ion-exchange spent regenerant brine. Desalination, 280(1-3), 428-431.

Kalbar, P., Karmakar, S., & Asolekar, S. (2012). Selection of an appropriate wastewater treatment technology: A scenario-based multiple-attribute decision-making approach. *Journal Of Environmental Management*, *113*, 158-169. doi: 10.1016/j.jenvman.2012.08.025

Kalbar, P., Karmakar, S., & Asolekar, S. (2016). Life cycle-based decision support tool for selection of wastewater treatment alternatives. *Journal Of Cleaner Production*, *117*, 64-72. doi: 10.1016/j.jclepro.2016.01.036

KEMCORE. (2019). Hydrochloric Acid 35% - Market Price. Retrieved June 27, 2019, from https://www.kemcore.com/hydrochloric-acid-35.html

Kim, D. H. (2011). A review of desalting process techniques and economic analysis of the recovery of salts from retentates. *Desalination*, *270*(1-3), 1-8.

Kim, J., Park, M., Snyder, S., & Kim, J. (2013). Reverse osmosis (RO) and pressure retarded osmosis (PRO) hybrid processes: Model-based scenario study. *Desalination*, *322*, 121-130. doi: 10.1016/j.desal.2013.05.010

Kloepffer, W. (2008). Life cycle sustainability assessment of products. *The International Journal of Life Cycle Assessment*, *13*(2), 89.

Klöpffer, W. (2003). Life-cycle based methods for sustainable product development.

Korngold, E., Aronov, L., & Daltrophe, N. (2009). Electrodialysis of brine solutions discharged from an RO plant. *Desalination*, *242*(1-3), 215-227. doi: 10.1016/j.desal.2008.04.008

Kress, N. (2019). *Marine Impacts of Seawater Desalination: Science, Management, and Policy*. Elsevier.

Le Dirach, J., Nisan, S., & Poletiko, C. (2005). Extraction of strategic materials from the concentrated brine rejected by integrated nuclear desalination systems. *Desalination*, *182*(1-3), 449-460. doi: 10.1016/j.desal.2005.02.037

Lee, K. L., Baker, R. W., & Lonsdale, H. K. (1981). Membranes for power generation by pressure-retarded osmosis. *Journal of Membrane Science*, *8*(2), 141-171.

Lennetech. (2017). Retrieved 7 November 2019, from https://www.lenntech.com/Data-sheets/Ion-Exchange-for-Dummies-RH.pdf

Levasseur, A., Lesage, P., Margni, M., Deschenes, L., & Samson, R. (2010). Considering time in LCA: dynamic LCA and its application to global warming impact assessments. *Environmental science & technology*, *44*(8), 3169-3174.

Lim, S., Park, D., & Park, J. (2008). Environmental and economic feasibility study of a total wastewater treatment network system. *Journal Of Environmental Management*, *88*(3), 564-575. doi: 10.1016/j.jenvman.2007.03.022

Liu, D. (2015). Water Treatment by Adsorption and Electrochemical Regeneration Development of a Liquid-Lift Reactor. Manchester, United Kingdom.

Loganathan, P., Naidu, G., & Vigneswaran, S. (2017). Mining valuable minerals from seawater: a critical review. *Environmental Science: Water Research & Technology*, *3*(1), 37-53. doi: 10.1039/c6ew00268d

Lorenzo-Toja, Y., Alfonsín, C., Amores, M. J., Aldea, X., Marin, D., Moreira, M. T., & Feijoo, G. (2016). Beyond the conventional life cycle inventory in wastewater treatment plants. Science of the Total Environment, 553, 71-82.

Martinetti, C., Childress, A., & Cath, T. (2009). High recovery of concentrated RO brines using forward osmosis and membrane distillation. *Journal Of Membrane Science*, *331*(1-2), 31-39. doi: 10.1016/j.memsci.2009.01.003

Martinez-Sanchez, V., Kromann, M. A., & Astrup, T. F. (2015). Life cycle costing of waste management systems: Overview, calculation principles and case studies. Waste management, 36, 343-355.

Martinez-Sanchez, V., Tonini, D., Møller, F., & Astrup, T. F. (2016). Life-cycle costing of food waste management in Denmark: importance of indirect effects. Environmental science & technology, 50(8), 4513-4523.

Mastali, M., Abdollahnejad, Z., & Pacheco-Torgal, F. (2018). Carbon dioxide sequestration of fly ash alkaline based mortars with recycled aggregates and different sodium hydroxide concentrations. Carbon Dioxide Sequestration In Cementitious Construction Materials, 349-371. doi: 10.1016/b978-0-08-102444-7.00014-9

McAdam, E. J., & Judd, S. J. (2008). Biological treatment of ion-exchange brine regenerant for re-use: A review. *Separation and Purification Technology*, *62*(2), 264-272.

McManus, M. C., & Taylor, C. M. (2015). The changing nature of life cycle assessment. *Biomass and bioenergy*, *82*, 13-26.

Meneses, M., Pasqualino, J. C., Céspedes-Sánchez, R., & Castells, F. (2010). Alternatives for reducing the environmental impact of the main residue from a desalination plant. *Journal of Industrial Ecology*, *14*(3), 512-527.

Mericq, J., Laborie, S., & Cabassud, C. (2010). Vacuum membrane distillation of seawater reverse osmosis brines. *Water Research*, *44*(18), 5260-5273. doi: 10.1016/j.watres.2010.06.052

Micari, M., Moser, M., Cipollina, A., Fuchs, B., Ortega-Delgado, B., Tamburini, A., & Micale, G. (2019). Techno-economic assessment of multi-effect distillation process for the treatment and recycling of ion exchange resin spent brines. *Desalination*, *456*, 38-52.

Michaud, C.F.C., 'Zero D' *A Method of Reducing Industrial Softener Discharge*, Water Conditioning & Purification, 52, pp. 26-30, 2010

Michaud, C. F. (1994). Zero Discharge Softener Regeneration. *Water Conditioning & Purification*, *39*, 40-45.

Mining Chemicals and Services From Kemcore. (2019). Retrieved 1 December 2019, from https://www.kemcore.com/

Mohamed, A., Maraqa, M., & Al Handhaly, J. (2005). Impact of land disposal of reject brine from desalination plants on soil and groundwater. *Desalination*, *182*(1-3), 411-433. doi: 10.1016/j.desal.2005.02.035

Mohapatra, P. K., Siebel, M. A., Gijzen, H. J., Van der Hoek, J. P., & Groot, C. A. (2002). Improving eco-efficiency of Amsterdam water supply: a LCA approach. *Journal of water supply: Research and technology-AQUA*, *51*(4), 217-227.

Møller, F., Slentø, E., & Frederiksen, P. (2014). Integrated well-to-wheel assessment of biofuels combining energy and emission LCA and welfare economic Cost Benefit Analysis. Biomass and Bioenergy, 60, 41-49.

Morillo, J., Usero, J., Rosado, D., El Bakouri, H., Riaza, A., & Bernaola, F. J. (2014). Comparative study of brine management technologies for desalination plants. *Desalination*, *336*, 32-49.

Muñoz, I., Gómez, M. J., Molina-Díaz, A., Huijbregts, M. A., Fernández-Alba, A. R., & García-Calvo, E. (2008). Ranking potential impacts of priority and emerging pollutants in urban wastewater through life cycle impact assessment. Chemosphere, 74(1), 37-44.

Neugebauer, S., Forin, S., & Finkbeiner, M. (2016). From life cycle costing to economic life cycle assessment-introducing an economic impact pathway. Sustainability (Switzerland), 8(5), 1–23. https://doi.org/10.3390/su8050428

Ng, H., Lee, L., Ong, S., Tao, G., Viawanath, B., & Kekre, K. et al. (2008). Treatment of RO brine–towards sustainable water reclamation practice. *Water Science And Technology*, *58*(4), 931-936. doi: 10.2166/wst.2008.713

Nicolini, J. V., Borges, C. P., & Ferraz, H. C. (2016). Selective rejection of ions and correlation with surface properties of nanofiltration membranes. Separation and Purification Technology, 171, 238-247.

Norris, G. (2001). Integrating Economic Analysis into LCA. *Environmental Quality Management*, *10*(3), 59-64. doi: 10.1002/tqem.1006

Oren, Y. (2011). Corrigendum to "Pilot studies on high recovery BWRO-EDR for near zero liquid discharge approach" [Desalination 261 (2010) 321–330]. *Desalination*, *266*(1-3), 291. doi: 10.1016/j.desal.2010.09.019

Panagopoulos, A., Haralambous, K. J., & Loizidou, M. (2019). Desalination brine disposal methods and treatment technologies-a review. *Science of The Total Environment*.

Peters, T., & Pintó, D. (2008). Seawater intake and pre-treatment/brine discharge—environmental issues. *Desalination*, *221*(1-3), 576-584

Peterskova, M., Valderrama, C., Gibert, O., & Cortina, J. (2012). Extraction of valuable metal ions (Cs, Rb, Li, U) from reverse osmosis concentrate using selective sorbents. *Desalination*, *286*, 316-323. doi: 10.1016/j.desal.2011.11.042

Petit-Boix, A., Llorach-Massana, P., Sanjuan-Delmas, D., Sierra-Perez, J., Vinyes, E., Gabarrell, X., Rieradevall, J. & Sanye-Mengual, E. (2017). Application of life cycle thinking towards sustainable cities: A review. *Journal of Cleaner Production*, *166*, 939-951.

Pizzol, M., Weidema, B., Brandão, M., & Osset, P. (2015). Monetary valuation in life cycle assessment: a review. Journal of Cleaner Production, 86, 170-179.

Raluy, G., Serra, L., & Uche, J. (2006). Life cycle assessment of MSF, MED and RO desalination technologies. *Energy*, *31*(13), 2361-2372.

Raluy, R. G., Serra, L., Uche, J., & Valero, A. (2005). Life cycle assessment of water production technologies-Part 2: reverse osmosis desalination versus the Ebro river water transfer (9 pp). *The International Journal of Life Cycle Assessment*, *10*(5), 346-354.

Randall, D. G., & Nathoo, J. (2015). A succinct review of the treatment of Reverse Osmosis brines using Freeze Crystallization. *Journal of water process engineering*, *8*, 186-194.

Randall, D. G., Nathoo, J., & Lewis, A. E. (2011). A case study for treating a reverse osmosis brine using Eutectic Freeze Crystallization—Approaching a zero waste process. *Desalination*, *266*(1-3), 256-262.

Rebitzer, G., Ekvall, T., Frischknecht, R., Hunkeler, D., Norris, G., Rydberg, T., & Pennington, D. W. (2004). Life cycle assessment: Part 1: Framework, goal and scope definition, inventory analysis, and applications. *Environment international*, *30*(5), 701-720.

Reich, M. C. (2005). Economic assessment of municipal waste management systems—case studies using a combination of life cycle assessment (LCA) and life cycle costing (LCC). Journal of Cleaner Production, 13(3), 253-263.

Resende, J., Nolasco, M., & Pacca, S. (2019). Life cycle assessment and costing of wastewater treatment systems coupled to constructed wetlands. *Resources, Conservation And Recycling*, *148*, 170-177. doi: 10.1016/j.resconrec.2019.04.034

RIVM. (2018). LCIA: the ReCiPe model. Retrieved May 21, 2018, from https://www.rivm.nl/en/life-cycle-assessment-lca/recipe

Roberts, D., Johnston, E., & Knott, N. (2010). Impacts of desalination plant discharges on the marine environment: A critical review of published studies. Water Research, 44(18), 5117-5128. doi: 10.1016/j.watres.2010.04.036

Rödger, J. M., Kjær, L. L., & Pagoropoulos, A. (2018). Life cycle costing: an introduction. In *Life Cycle Assessment* (pp. 373-399). Springer, Cham.

Salih, H. H., Li, J., Kaplan, R., & Dastgheib, S. A. (2017). Life cycle assessment of treatment and handling options for a highly saline brine extracted from a potential CO2 storage site. *Water research*, *122*, 419-430.

Scarcity | UN-Water. (2018). Retrieved from http://www.unwater.org/water-facts/scarcity/

Schewe, J., Heinke, J., Gerten, D., Haddeland, I., Arnell, N., & Clark, D. et al. (2013). Multimodel assessment of water scarcity under climate change. Proceedings Of The National Academy Of Sciences, 111(9), 3245-3250. doi: 10.1073/pnas.1222460110

Seigworth, A., Ludlum, R., & Reahl, E. (1995). Case study: Integrating membrane processes with evaporation to achieve economical zero liquid discharge at the Doswell Combined Cycle Facility. Desalination, 102(1-3), 81-86. doi: 10.1016/0011-9164(95)00044-3

Sharaai, A. H., Mahmood, N. Z., & Sulaiman, A. H. (2009). Life Cycle Impact Assessment (LCIA) of potable water treatment process in Malaysia: comparison between dissolved air flotation (DAF) and ultrafiltration (UF) technology. *Australian Journal of Basic and Applied Sciences*, *3*(4), 3625-3632.

SimaPro. (2019). Retrieved 8 November 2019, from https://simapro.com/

Singh, R., Murty, H., Gupta, S., & Dikshit, A. (2009). An overview of sustainability assessment methodologies. *Ecological Indicators*, *9*(2), 189-212. doi: 10.1016/j.ecolind.2008.05.011

Solbrine. (2017). Retrieved 7 December 2019, from http://solbrine.uest.gr/index.php/el/

Sombekke, H. D. M., Voorhoeve, D. K., & Hiemstra, P. (1997). Environmental impact assessment of groundwater treatment with nanofiltration. *Desalination*, *113*(2-3), 293-296.

Steen, B. (1999). A systematic approach to environmental priority strategies in product development (EPS): version 2000-Models and data of the default method (p. 67). Chalmers tekniska högsk..

Subramani, A., DeCarolis, J., Pearce, W., & Jacangelo, J. (2012). Vibratory shear enhanced process (VSEP) for treating brackish water reverse osmosis concentrate with high silica content. *Desalination*, *291*, 15-22. doi: 10.1016/j.desal.2012.01.020

Swarr, T. E., Hunkeler, D., Klöpffer, W., Pesonen, H. L., Ciroth, A., Brent, A. C., & Pagan, R. (2011). Environmental life-cycle costing: a code of practice.

Tabellen tarieven milieubelastingen. (2019). Retrieved 8 November 2019, from https://www.belastingdienst.nl/wps/wcm/connect/bldcontentnl/belastingdienst/zakelijk/overige_belastingen/belastingen_op_milieugrondslag/tarieven_milieubelastingen/tabellen_tarieven_milieubelastingen

Taelman, S., Tonini, D., Wandl, A., & Dewulf, J. (2018). A holistic sustainability framework for waste management in European cities: Concept development. Sustainability, 10(7), 2184.

Tanaka, Y., Ehara, R., Itoi, S., & Goto, T. (2003). Ion-exchange membrane electrodialytic salt production using brine discharged from a reverse osmosis seawater desalination plant. *Journal Of Membrane Science*, *222*(1-2), 71-86. doi: 10.1016/s0376-7388(03)00217-5

The Ministry of Infrastructure and the Environment and the Ministry of Economic Affairs. (2016). *A Circular Economy in the Netherlands by 2050*. The Ministry of Infrastructure and the Environment and the Ministry of Economic Affairs.

Trieb, F., Moser, M., & Fichter, T. (2011). MENA Regional Water Outlook Part II: Desalination using Renewable Energy. *FICHTNER (Germany)*.

Tufa, R. A., Curcio, E., van Baak, W., Veerman, J., Grasman, S., Fontananova, E., & Di Profio, G. (2014). Potential of brackish water and brine for energy generation by salinity gradient power-reverse electrodialysis (SGP-RE). *RSC Advances*, *4*(80), 42617-42623.

Turek, M., Was, J., & Dydo, P. (2009). Brackish water desalination in RO–single pass EDR system. *Desalination And Water Treatment*, *7*(1-3), 263-266. doi: 10.5004/dwt.2009.710

UNESCO. (2019). *The United Nations World Water Development Report l Leaving no one behind*. France: UNESCO.

United Nations. (2018). *The Sustainable Development Goals*. New York: United Nations.

Von Medeazza, G. M. (2005). “Direct” and socially-induced environmental impacts of desalination. *Desalination*, *185*(1-3), 57-70.

Wadley, S., Brouckaert, C. J., Baddock, L. A. D., & Buckley, C. A. (1995). Modelling of nanofiltration applied to the recovery of salt from waste brine at a sugar decolourisation plant. *Journal of Membrane Science*, *102*, 163-175.

Walmsley, T. G., Ong, B. H., Klemeš, J. J., Tan, R. R., & Varbanov, P. S. (2019). Circular Integration of processes, industries, and economies. Renewable and Sustainable Energy Reviews, 107, 507-515.

Water use in industry. (2016). Retrieved from https://ec.europa.eu/eurostat/statistics-explained/index.php/Archive:Water_use_in_industry

Weidema, B. P. (2009). Using the budget constraint to monetarise impact assessment results. Ecological economics, 68(6), 1591-1598.

Weldu, Y. W. (2018). A Societal Life Cycle Costing of Energy Production: The Implications of Environmental Externalities. Low Carbon Transition: Technical, Economic and Policy Assessment, 109.

Weldu, Y. W., & Assefa, G. (2017). The search for most cost-effective way of achieving environmental sustainability status in electricity generation: Environmental life cycle cost analysis of energy scenarios. Journal of cleaner production, 142, 2296-2304.

Wenten, I. G. (2016). Reverse osmosis applications: prospect and challenges. *Desalination*, *391*, 112-125.

Wenten, I. G., Ariono, D., Purwasasmita, M., & Khoirudin. (2017, March). Integrated processes for desalination and salt production: A mini-review. In *AIP Conference Proceedings* (Vol. 1818, No. 1, p. 020065). AIP Publishing.

Wood, R., & Hertwich, E. G. (2013). Economic modelling and indicators in life cycle sustainability assessment. The International Journal of Life Cycle Assessment, 18(9), 1710-1721.

World Economic Forum. (2019). *The Global Risks Report*. Geneva: World Economic Forum. Retrieved from https://www.weforum.org/reports/the-global-risks-report-2019

Xevgenos, D., Moustakas, K., Malamis, D., & Loizidou, M. (2016). An overview on desalination & sustainability: renewable energy-driven desalination and brine management. *Desalination and Water Treatment*, *57*(5), 2304-2314.

Xu, P., Cath, T. Y., Robertson, A. P., Reinhard, M., Leckie, J. O., & Drewes, J. E. (2013). Critical review of desalination concentrate management, treatment and beneficial use. *Environmental Engineering Science*, *30*(8), 502-514.

Younos, T. (2005). Environmental issues of desalination. *Journal of contemporary water research and education*, *132*(1), 3.

Yuan, C., Wang, E., Zhai, Q., & Yang, F. (2015). Temporal discounting in life cycle assessment: a critical review and theoretical framework. *Environmental Impact Assessment Review*, *51*, 23-31.

Zero Brine (2017). *Grant Agreement* (No. 730390). European Commission

Zero Brine. (2018). *Bench scale test using equipment from BEC - Deliverable 2.3*.

Zhai, Q., Crowley, B., & Yuan, C. (2011, May). Temporal discounting for life cycle assessment: Differences between environmental discounting and economic discounting. In Proceedings of the 2011 IEEE International Symposium on Sustainable Systems and Technology (pp. 1-1). IEEE.

Zhou, J., Chang, V. W. C., & Fane, A. G. (2011). Environmental life cycle assessment of reverse osmosis desalination: the influence of different life cycle impact assessment methods on the characterization results. *Desalination*, *283*, 227-236.

Zhou, J., Chang, V. W. C., & Fane, A. G. (2013). An improved life cycle impact assessment (LCIA) approach for assessing aquatic eco-toxic impact of brine disposal from seawater desalination plants. *Desalination*, *308*, 233-2

APPENDIX A

Midpoint and endpoint monetary weighting factors of LCIA methods.

LCIA Model € ref. year	Human Health	Ecosystems €/Species.yr	Abiotic resources €/MJ	Acidification €/kg SO_2eq	Global Warming €/kg CO_2eq	Eutrophication €/kg PO_4 eq	Ozone Depletion €/kg CFC11eq
EPS $€_{1999}$	85000€/YOLL	$110E10^9$					
ReCiPe $€_{2008}$	60000€/DALY	$175E10^9$					
Ecovalue08 $€_{2010}$			0.0037	2.792	0.0093	20.289	
Stepwise 2006 $€_{2003}$	74000€/DALY	$30.8E10^9$	0.004	0.146	0.083	1.2	100
LIME $€_{2010}$	119805€/DALY	$115.73E10^9$					
Ecotax02 $€_{2002}$			0.014	1.675	0.059	2.659	111.684

APPENDIX B

Economic and social impacts relevant for sLCC (Hunkeler et al., 2008)

Economic impacts	Relevance for sLCC	
Economic prosperity and resilience	Mainly relevant for LCC studies with major investment decisions only	Could be captured by GNP changes
Income		% change in average income of the affected regions
Employment		% change in average employment rate of the affected regions
Property values	Products or projects related to dispossession, and infrastructure projects (e.g., changing house values)	Value (change) of the affected property
Replacement costs of environmental and social functions (that were formerly provided by the environment, but now have to be paid for)		Avoid double counting with LCA
Economic dependency or freedom	Energy sector projects	Diversity of energy carriers
Burden of national debt	Public investment projects	Change in national debts
Workload or time saving or wasted time	Many electronic products (e.g., dishwasher)	Change in workload or free time, congestion data, % of canceled or delayed trains and planes, and so on

Social impact	Relevance for LCC	Comments
	Health and social well-being	
Death	Products with a direct fatal impact (weapons), accidents due to products, or the like	Could be related to statistical number of fatalities
Reduced number of fatalities in society	Safety product features (e.g., airbags and pedestrian protection)	Could be related to statistical number of reduced fatalities
Nutrition	Products improving nutrition (e.g., fertilizer, food packaging, and refrigerants) or poisoning impacts during the life cycle	Could be related to statistical numbers of changed yield per acre
Actual physical or mental	Pharmaceutical products or negative	Could be related to statistical

health and fertility (reduced or improved by product impact)	impacts during the life cycle	numbers of illness impacts
Perceived health	Placebos (e.g., from electromagnetic pollution)	Percentage of population suffering from diffuse health impacts
Aspirations and image	Luxury products	Market analysis
Autonomy	Products enabling individual mobility, communication, and so on	
Stigmatization or deviance labeling	Energy-efficient appliances	
Feelings in relation to the project	Big infrastructural projects	Survey
	Quality of the living environment (livability)	
Quality of the living environment (actual and perceived)	Similar issues that are treated in environmental impact assessments	Avoid double counting with LCA
Leisure and recreational opportunities and facilities	Landscape-changing and land consuming products	Avoid double counting with LCA
Environmental amenity value and/or aesthetic quality	Landscape-changing and land consuming products	Avoid double counting with LCA
Availability of housing facilities, physical quality of housing (actual and perceived), and social quality of housing (homeliness)	Housing products	Affordability and quality aspects
Adequacy of physical infrastructure	Communication and mobility products and services	Distance to target or average relation between population and infrastructure
Adequacy of and access to social infrastructure	Health care products	Health costs
Personal safety and hazard exposure (actual and perceived)	Hazardous chemicals or waste in the life cycle	Could be related to statistical number of accidents
Crime and violence (actual and perceived)	Security products and indirect impacts along the life cycle	Could be related to statistical numbers of crime and violence
	Cultural impacts	
Change in cultural values (moral rules, beliefs, etc.), or cultural affront	Products in conflict with cultural values in different regions	
Cultural integrity	Media products	
Experience of being culturally marginalized	Roads in areas with indigenous populations	
Profanation of culture	Media products	
Loss of language or dialect	Products standardizing a certain language (software)	Qualitatively
Natural and cultural heritage (violation, damage, or destruction)	Infrastructural projects	Avoid double counting with LCA
	Family and community impacts	
Alteration of family structure	Linked to life cycle impacts of	

	projects or products (e.g., by job losses)	
Family violence Social networks Community identification and connection Community cohesion (actual and perceived) Social tension and violence	Unlikely to be monetized and more reasonably expressed as a separate set of midpoint indicators	To be included in a complementary societal assessment
Institutional, legal, political, and equity impacts		
Functioning of government agencies	Government projects	Could be related to changes in time needed for bureaucratic activities
Access to legal procedures and legal advice Integrity of government and government agencies	Unlikely to be monetized and more reasonably expressed as a separate set of midpoint indicators	
Participation in decision making	Government projects	Could be related to % of participation
Tenure or legal rights	Products and projects related to data safety	To be captured qualitatively
Subsidiary (the principle that decisions should be made as close to the people as possible)	Government projects	
Human rights	Often captured by other social impacts	
Relations between people with different genders, ethnicities, races, ages, sexual orientations, religions, opinions, education levels, income levels, presence of disabilities, and so on		
Physical integrity	Products with encouraging or discouraging features or information	Specific ways for measurement (e.g., psychological analysis)
Personal autonomy	Unlikely to be monetized and more reasonably expressed as a separate set of midpoint indicators	
Fair division of production-oriented labor	Products or projects enabling work for different groups (part-time, or kindergarten) or impacts along the product life cycle	Could be related to changes in % of labor
Fair division of household labor	Unlikely to be monetized and more reasonably expressed as a separate set of midpoint indicators	
Fair division of reproductive labor	Impacts along the product life cycle	Percentage of participation for each group
Fair control over and access to resources	Fair trade products	
Equal access to services (mobility, communication, health care, etc.)	Product features enabling use of, for example, mobility carriers by disabled people	Specific measures (e.g., wheelchair versus vehicle dimension)

APPENDIX C

Allocation factors for LCA analysis

Process	Allocation	Quantity	Price (€)	Proceeds (€)	Allocation Factor
Site I					
NF	**Mass**				
NF permeate (kg)		61,6	-	-	0,215
NF concentrate (kg)		225,6	-	-	0,785
Total		287,2			
MC I	**Mass**				
MC1 effluent (kg)		235,2	-	-	0,948
Mg(OH)2 (kg)		12,8	-	-	0,052
Total					
MC II	**Mass**				
MC2 effluent (kg)		237,6	-	-	0,221
Ca(OH)2 (kg)		67,3	-	-	0,779
Total					
Evaporation	**Economic**				
NaCl (kg)		302,6	0,063	19,07	0,01
Distilled water (l)		3344	0,575	1922,8	0,99
Total				1941,87	
Site II					
Anionic IEX	**Mass**				
An. IEX effluent (kg)		22,8	-	-	0,365
An. IEX brine (kg)		39,6	-	-	0,635
Total		62,4			
RO	**Economic**				
RO concentrate (kg)		22,8	0	0	0
Distilled water (l)		6840	0,575	3933	1
Total				3933	
Evaporation	**Economic**				
Distilled water (l)		649,4	0,575	373,4	0,777
NaCl (kg)		22,6	0,063	1,42	0,004
NaHCO3 (kg)		132,8	0,793	105,31	0,219
Total				480,13	
NF	**Mass**				
NF concentrate		10,9	-	-	0,276
NF permeate		28,5	-	-	0,724
Total		39,4			

APPENDIX D

Environmental impacts for the production of 1000 m^3 for pre- and post-ZB system

Impact Category	Unit	Pre-ZB system	Post-ZB system
Global Warming	kg CO_2 – eq.	$1.68 \cdot 10^3$	$2.94 \cdot 10^3$
Ionizing radiation	kg kBq U_{235}-eq.	$0.82 \cdot 10^1$	$1.35 \cdot 10^1$
Particulate matter formation	kg PM_{10} – eq.	$9.82 \cdot 10^{-1}$	$0.89 \cdot 10^1$
Acidification	kg SO_2-eq.	$0.26 \cdot 10^1$	$3.01 \cdot 10^1$
Freshwater eutrophication	kg P- eq.	$3.98 \cdot 10^{-2}$	$1.02 \cdot 10^{-1}$
Marine eutrophication	kg N	$1.59 \cdot 10^{-2}$	$1.26 \cdot 10^{-2}$
Freshwater ecotoxicity	kg 1,4 DB – eq.	$1.73 \cdot 10^{-1}$	$6.14 \cdot 10^{-1}$
Marine ecotoxicity	kg 1,4 DB – eq.	$0.13 \cdot 10^1$	$0.25 \cdot 10^1$
Human toxicity	kg 1,4 DB – eq.	$2.73 \cdot 10^1$	$7.33 \cdot 10^1$

Environmental impacts for the production of 1000 m3 for the sub-systems of post-ZB system

Impact Category	Unit	DWP	Site I	Site II
Global Warming	kg CO_2 – eq.	$1.60 \cdot 10^3$	$2.32 \cdot 10^1$	$1.32 \cdot 10^3$
Ionizing radiation	kg kBq U_{235}-eq.	$0.74 \cdot 10^1$	$1.75 \cdot 10^{-1}$	$0.58 \cdot 10^1$
Particulate matter formation	kg PM_{10} – eq.	$9.23 \cdot 10^{-1}$	$2.86 \cdot 10^{-2}$	$0.80 \cdot 10^1$
Acidification	kg SO_2-eq.	$0.24 \cdot 10^1$	$5.79 \cdot 10^{-2}$	$2.76 \cdot 10^1$
Freshwater eutrophication	kg P- eq.	$3.16 \cdot 10^{-2}$	$1.01 \cdot 10^{-3}$	$6.93 \cdot 10^{-2}$
Marine eutrophication	kg N	$7,38 \cdot 10^{-3}$	$2.04 \cdot 10^{-4}$	$5.06 \cdot 10^{-3}$
Freshwater ecotoxicity	kg 1,4 DB – eq.	$1.58 \cdot 10^{-1}$	$4.31 \cdot 10^{-3}$	$4.52 \cdot 10^{-1}$
Marine ecotoxicity	kg 1,4 DB – eq.	$0.12 \cdot 10^1$	$1.43 \cdot 10^{-2}$	$0.13 \cdot 10^1$
Human toxicity	kg 1,4 DB – eq.	$2.27 \cdot 10^1$	$6.06 \cdot 10^{-1}$	$5.00 \cdot 10^1$