Comparative Life Cycle Assessment of Sludge Treatment Systems

Is recycling aluminium based coagulant from chemical sludge the way of the future?

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Is recycling aluminium based coagulant from chemical sludge the way of the future?

Jämförande livscykelanalys av slamhanteringssystem
Är återvinning av aluminiumbaserad koagulant från kemslam framtidsvägen?

Degree project in Strategies for sustainable development, Second Cycle
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Preface

This master thesis has been written from January to September 2017 to conclude my graduation at the department of sustainable development at the Royal Institute of Technology (KTH). The study was conducted on behalf of ÅF Industry AB and Purac AB in order to evaluate the environmental impacts of a newly developed sludge treatment method from a life cycle perspective.

I would like to express my utmost gratitude to my tutor at KTH, Anna Björklund, and my supervisors at ÅF and Purac, Therese Johansson and Madeleine Nilsson respectively, for your guidance, cooperation and support. It has been a privilege to analyze and debate all aspects of my thesis with you. Moreover, I would like to thank Mårten Krogerus at ÅF who help me with the energy and mass balance. My thanks go also out to Daniel Ling and Ingemar Heidfors at Purac for all your input and feedback.

To my parents, for always believing in me and encouraging me to keep going despite all the setbacks during this journey.

Patrick Henriksson

Stockholm, October 30, 2017
Abstract

Chemical coagulation is a widely used wastewater treatment method around the world to reduce impurities from the process water in various industries. However, the large amounts of coagulation chemicals that are required for the removal of dissolved particles create a chemical sludge which poses a great environmental problem. Purac AB, a Swedish wastewater treatment company attempts to solve this problem with a new technology called the ReAl process. The ReAl process can recycle the aluminium ions from the commonly used coagulant aluminium sulfate, which reduces the amount of chemical sludge and the amount of aluminium sulfate needed in the coagulation process. In this study, a comparative life cycle assessment was conducted with a cradle-to-grave approach and mostly in accordance with the ISO-14040 series with the only deviation of not including resource-based impact categories. The goal was to evaluate the environmental impact of two sludge treatment systems – a conventional system (system 1) and a system which includes the ReAl process (system 2). Furthermore, the environmental performance of two dewatering equipment’s, a decanter centrifuge and a filter press, were examined in system 1, while in system 2, the exclusion of sludge drying was investigated.

The scope of the study did not include the infrastructure of the sludge treatment systems and the ReAl process since previous studies have shown that, the environmental impact from the infrastructure in the wastewater treatment industry is relatively small compared to other factors, such as the energy and coagulation chemical used in these systems.

The characterization results showed that system 2 had the lowest environmental impact on all the evaluated impact categories. The results also revealed that system 1 would have a slightly lower environmental impact if the chemical sludge was dewatered with a decanter centrifuge instead of a filter press. Similarly, system 2 would have a slightly lower environmental impact if sludge drying was excluded from the system. However, the environmental performance gain from selecting the best dewatering and drying equipment is limited and considered within the margin of error. Thus, this thesis suggests selecting the sludge treatment equipment based on their economic and technical factors before their environmental performance.

The largest environmental impact in system 1 derived from the use of the coagulation chemical aluminium sulfate, while in system 2, sulfuric acid used in the ReAl process contributed the most to its environmental impact. The sensitivity analysis showed that a “clean” electricity mix is essential for system 2 and the ReAl process overall impact on the environment compared to system 1.

Keywords: LCA, chemical coagulation, sludge treatment, ReAl process, Purac, ÅF, wastewater treatment
Sammanfattning


Undersökningen omfattade inte infrastrukturen i slambehandlingssystemen och ReAl-processen eftersom studier har visat att miljöpåverkan från infrastrukturen i avloppsvattenreningsindustrin är relativt liten jämfört med andra faktorer som t.ex. den energi och fällningskemikalier som används i dessa system.


Den största miljöpåverkan i system 1 härrörande från användningen av fällningskemikalien aluminiumsulfatet, medan i system 2 bidrog svavelsyran från ReAl-processen mest till dess miljöpåverkan. Känslighetsanalysen visade b.la. att en "ren" el-mix är nödvändig för att system 2 och ReAl-processen ska ses som mer miljövänlig än system 1.

Nyckelord: LCA, kemfällning, slambhantering, ReAl-processen, Purac, ÅF, avloppsvattenrenning
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## Abbreviations

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<tr>
<th>Abbreviation</th>
<th>Full Form</th>
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<tbody>
<tr>
<td>ADT</td>
<td>Air-dry tonne</td>
</tr>
<tr>
<td>CAB</td>
<td>County administrative board</td>
</tr>
<tr>
<td>CIP</td>
<td>Clean-in-place</td>
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<tr>
<td>CO₂</td>
<td>Carbon dioxide</td>
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<tr>
<td>COD</td>
<td>Chemical oxygen demand</td>
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<tr>
<td>DAF</td>
<td>Dissolved air flotation</td>
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<td>DW</td>
<td>Dry weight</td>
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<tr>
<td>ESP</td>
<td>Electrostatic precipitation</td>
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<td>GLO</td>
<td>Global</td>
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<td>GWP</td>
<td>Global warming potential</td>
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<td>GHG</td>
<td>Greenhouse gas</td>
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<td>ISO</td>
<td>International standard organization</td>
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<td>LCA</td>
<td>Life cycle assessment</td>
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<td>LCCA</td>
<td>Life cycle cost analysis</td>
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<td>LCIA</td>
<td>Life cycle impact assessment</td>
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<td>LCI</td>
<td>Life cycle interpretation</td>
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<tr>
<td>NOₓ</td>
<td>Nitrogen oxides</td>
</tr>
<tr>
<td>RER</td>
<td>Europe</td>
</tr>
<tr>
<td>SCOD</td>
<td>Soluble chemical oxygen demand</td>
</tr>
<tr>
<td>NVV</td>
<td>Swedish Environmental protection Agency (Sv. Naturvårdsverket)</td>
</tr>
<tr>
<td>SOₓ</td>
<td>Sulfuric oxides</td>
</tr>
<tr>
<td>TOC</td>
<td>Total organic carbon</td>
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<tr>
<td>VOC</td>
<td>Volatile organic compound</td>
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1. Introduction

Chemical coagulation is a widely used wastewater treatment method around the world in many industries e.g. pulp and paper and it is also used to treat raw water to process water. However, chemical coagulation requires large amounts of metal salts e.g. based on aluminium or iron to coagulate dissolved particulates. Consequently, when dissolved particulates and water are separated, large amounts of chemical sludge is produced. The amount of chemical sludge for further treatment and the coagulation chemicals are two substantial environmental problems in various industries.

At the Swedish Environmental Protection Agency (NVV) they are discussing whether it is good for the environment to impose stricter regulations on wastewater treatment in the pulp and paper industry. These regulations would most likely require a third (tertiary) treatment stage (chemical coagulation) upon a primary (often mechanical) and a secondary (often biological) treatment stages. The tertiary treatment stage would result in cleaner water and less chemical oxygen demand (COD) to the recipient. However, the drawback with chemical coagulation is the production of residue in the form of chemical sludge, and higher energy consumption and extra input of consumables such as coagulation chemicals. Relying on only a primary and secondary treatment stages would avoid chemical sludge and require fewer inputs such as energy but the water to the recipient would consequently have an increased COD and phosphorus level. (Nilsson, pers.comm., 12 February)

The discussion is also heavily related to Sweden’s problem with eutrophication due to emissions of phosphorus. In fact, both the NVV and the county administrative board (CAB) are very keen to make chemical coagulation a standard wastewater treatment method adjacent to industries emitting substantial amounts of phosphorus (Holmgren, pers.comm., 17 March). Chemical coagulation can reduce the phosphorous content by up to 90 % (Lenntech, 2016), compared to biological treatment methods, which have very limited effect on the phosphorous reduction (20-30 %) (Persson, 2011). The pulp and paper industry is one such industry, where the introduction of a chemical coagulation stage, in a normal sized plant, could lead to significant reductions of phosphorous of the order of several tons per year (Holmgren, pers.comm., 17 March). This could be a more environmental and cost-effective approach to handle the phosphorus issue compared to today’s efforts, where municipalities are enforcing property owners with requirements on costly and often environmentally inefficient measures such as infiltration trenches. However, for chemical coagulation to have a breakthrough in Sweden, it is very important to solve the management of chemical sludge (Ridderstolpe, 2009).

This is where the ReAl process comes into play. It is a newly developed technology which is globally licensed for the Swedish water treatment company Purac AB (Purac, 2016). The ReAl process can recover aluminium ions from chemical sludge which has the potential to reduce the amount of chemical sludge and the need for coagulation chemicals (Johansson, 2016).

1.1. Purpose

The purpose of this master thesis is to determine, from a life cycle perspective, which sludge treatment system, a conventional system or a system which includes the ReAl process, is more environmentally sustainable. This knowledge is important for both Purac and ÅF since it will inform them if the results can be used to market the ReAl process as an environmentally friendly technology. From a broader perspective, the LCA is also meaningful for the wastewater treatment market as a whole. The NVV and the CAB are also very interested in the results because they want
to impose stricter limits on industrial discharge water quality, and the only way to achieve it today, is by using chemical wastewater treatment – and the ReAl process could accelerate the development of chemical coagulation in Sweden.

1.1.1. Objective

1. Analyze the environmental impact of the two sludge treatment systems by carrying out a comparative Life cycle assessment (LCA).
   a. Identify which sludge treatment system is the most environmentally friendly.
   b. Identify which inputs that have the largest contribution to the environmental impact.
   c. Identify which combination of sludge treatment equipment within the systems is the most environmentally friendly.

2. Propose recommended actions which ÅF, Purac and the intended audience should take based on the findings of the LCA.
2. Background

The background establishes an underlying understanding of the most important processes (chemical coagulation and the ReAl process) of the sludge treatment systems. Furthermore, chemical sludge and ReAl sludge are defined and the treatment of the sludge is put into a legal context. Relevant previous research is presented in order to compare those findings with the findings of this thesis.

2.1. Treatment with chemical coagulation and flocculation

Chemical treatment methods are common in municipal wastewater treatment processes but also in industries. Chemical treatment is used as a polishing stage which can achieve very high treatment degrees. Chemical coagulation means that suitable chemicals are added to transform dissolved pollutants into precipitates. The formed particles can then be separated from the water using e.g. sedimentation. It is suited for removal of larger particles and less degradable compounds while letting the smallest compounds pass through. It is well-used for phosphorus reduction in municipal wastewater treatment and the separation of metals from the process water in many industries. (Persson, 2011, pp. 122)

Chemical coagulation and flocculation are used to separate particulate matter and it is also possible to separate and remove compounds in suspended or colloidal form (very small and otherwise non-settable particles). Colloidal particles are normally non-settable because they have a negative surface charge which makes them repel each other so that the Van der Waals forces between these particles are too weak, thus, they cannot form particles heavy enough to settle. Through the addition of coagulants to the wastewater, mostly in the form of a salt with a positive charge, the positive ions neutralize the negative colloidal particles which enable the Van der Waals forces to form larger settable particles. Another way to settle colloidal particles is by adding long-chain polymers as flocculation agents. The polymer and colloidal particles attract each other and form flocks which can be removed in a settling basin. (Persson, 2011, pp. 123-125)

There are some essential drawbacks with chemical treatment methods that should be pointed out. For instance, chemical coagulation and flocculation require the use of expensive coagulants and they generate larger quantities of chemical sludge, around ten times more sludge is produced in relation to the COD reduction. Recovering coagulation chemicals and finding a good use for the chemical sludge will determine the economic sustainability of using a chemical wastewater treatment (Granström et al., 2014). The ReAl process attempts to solve this problem and it could maybe lead to an increased adoption in the future.

2.2. Definition of sludge

No proper definition of sludge was found in, neither the Swedish Code of Statutes nor from the NVV. In fact, Naturvårdsverket (2004) points to the European standard EN 12832:1999 (Characterization of sludges, Utilization, and disposal of sludge – Vocabulary) for a definition. In this standard, sludge is defined as “Mixture of water and solids separated from various types of water as a result of natural or artificial processes”.

In this report, two types of sludges are distinguished; chemical sludge and ReAl sludge. The difference between these sludges is that the chemical sludge is a by-product of the chemical coagulation stage, entailing a substantial amount of aluminium. The ReAl sludge is the sludge after
the recovery of aluminium ions in the ReAl process. Thus, the ReAl sludge entails a much lower amount of aluminium than the chemical sludge.

2.3. ReAl process
Feralco AB, a Swedish chemical producer, and Purac AB have for many years worked on a process to recycle coagulation chemicals in laboratory conditions and by conducting pilot tests at multiple wastewater treatment plants in Sweden. These tests have resulted in what is now known as the ReAl process. The technology is patented by Feralco and Purac is globally licensed to contract the wastewater treatment system. (Purac, 2016a)

Coagulation with metals produces large amounts of sludge. Introducing the ReAl process in connection to a chemical coagulation stage would significantly reduce the amount of purchased coagulants needed, thus lowering the operational costs. The environmental benefits of recovering aluminium are, first and foremost, the significant reduction of coagulation chemicals and sludge, but also, the increased organic content of the sludge. Less inorganic matter makes the sludge more suitable for incineration since more material is combustible and fewer ash residues can be expected. Reduced phosphorus emissions from the pulp and paper industry is another significant environmental benefit (Purac, 2016a). Purac tested the process at a semi-industrial scale in China (Rizhao) in 2009. Pilot tests have also been performed at Iggesunds pulp mill during the year 2010-2011 and the year 2013-2015 (Purac, 2016b). The ReAl process is now a commercial technology ready for the market but it has not yet been implemented as a full-scale technology. It is suited for various scenarios, both for water and wastewater treatment, where lots of coagulants are used and costs for sludge deposit are substantial (Purac, 2016a).

Simply put, the ReAl process recovers the coagulant (aluminium sulfate) by hydrolyzing, acidifying and pre-heating the chemical sludge. The aluminium ions (Al^{3+}) are then separated through continuous cross-flow ultrafiltration (see Figure 1) and approximately 75%-90% of the coagulant is expected to be recovered. The amount of recovered coagulant combined with sludge concentration in several steps results in about 60%-70% sludge reduction (Purac, 2016b). A more detailed look at the ReAl process is provided in subchapter 4.3.2.

2.4. Laws and regulations related to sludge treatment in Sweden
Two of the most common ways to manage chemical sludge is through incineration and landfilling. These two processes were included in the sludge treatment systems that were evaluated in the comparative LCA. Moreover, many of the relevant Swedish regulations surrounding sludge treatment are fairly new with few guiding court cases. It is important to understand how laws and
regulations can impact the development of chemical coagulation and sludge management. A deeper look into the Swedish regulatory aspects of landfilling and incineration is described in the two upcoming subchapters.

2.4.1. Landfilling

Landfills accumulate large amounts of pollutants and toxins on a limited surface area. Over time, the substances leak into the surrounding environment which could affect human health and the environment through pollution of soil and water. Landfilling organic matter poses a risk of emitting methane which is a very potent greenhouse gas (GHG) (Naturvårdsverket, 2016a). Furthermore, emissions of methane from organic waste at landfills still account for the largest share of greenhouse gas emissions from waste management (Naturvårdsverket, 2012). Because of this, it is prohibited since 2005 to landfill organic waste in accordance with Waste landfill (2001:512) ordinance 10 §. Organic matter can also be used as a resource instead of sending it to a landfill e.g. by material and energy recovery through incineration or as fertilizer (Sundberg, 2013). Sorted combustible waste is forbidden to be landfill since 2002 in accordance with (2001:512) 9 §. Sludge containing more than 10 weight percent total organic carbon (TOC) dry weight (DW) is classified as organic waste and thus prohibited to be sent to landfill in accordance with the NVV statute (NFS 2004:4) 12 § unless otherwise stated by the NVV. A study analyzed among other things the composition of chemical sludge from the pulp and paper industry (Ek et al., 1996) and found that ten different samples ranged from 52-88 % TOC DW. Since these numbers are much higher than 10 % TOC, the sludge in this thesis should be classified as organic waste, which means that it cannot be sent to landfill. However, the county administrative board has given dispensation to some landfills, including the landfill at Iggesunds pulp mill, where they can use chemical sludge as a component in construction material which can be used to end-cover landfills. There is thought a limit to how much and for how long chemical sludge can be used to cover landfills. Landfilling is not a long-term sustainable solution, thus all actors who produce sludge of this type must work on better solutions for manage it (Holmgren, pers.comm., 17 March).

One such solution, Instead of receiving a dispensation to landfill the chemical sludge, is to co-incinerate it with e.g. bark and use the ash that arises in construction materials or at landfills. Within factory areas and especially at landfills, any leaching of heavy metals from the ash has a small significance (IVL, 2003).

2.4.2. Incineration

In the pulp and paper industry, it is common to incinerate the waste they produce, mostly bark but also sludge (Naturvårdsverket, 2012). In fact, 96 % of the heat demand in the Swedish forestry industry is generated with their own bioenergy (Skogsindustrierna, 2015). There are two types of incineration plants, co-incineration and waste incineration. A co-incineration plant is according to the waste incineration (2013:253) ordinance 7 § p.1 an incinerator which is mainly intended for the production of energy or material, where the waste is used as normal or additional fuel, or which is treated with heat in order to be disposed of. The incineration of chemical sludge shall

1 Swedish: Förordning (2001:512) om deponering av avfall. The purpose of this ordinance is to prevent and reduce the negative effects of waste disposal on human health and the environment, in particular with regard to surface water, groundwater, soil and air pollution

2 Swedish: Förordning (2013:253) om förbränning av avfall. This ordinance applies to incineration of solid or liquid waste in a combustion plant.
therefore normally follow the regulations associated with co-incineration plants because sludge is mainly incinerated together with other fuel types such as bark for energy production.

The pulp and paper industry can under the right circumstances avoid (2013:253) through an exception described in § 17 p. 4. This paragraph states that (2013:253) shall not be applied to plants where the treated waste is only fibrous vegetable waste arising in the production of virgin pulp or paper production from pulp if the waste co-incinerates at the production site and if the heat generated by incineration is recovered. It is unclear whether chemical sludge and/or ReAl sludge can take advantage of 17 § p.4 and be classified as biofuel instead of waste, under the much less strict large combustion plants (2013:252) ordinance\(^3\) 3 § p.4 because there have not been any legal cases that can settle this matter. Nonetheless, one of the most credible person’s (Holmgren, pers.comm., 17 March)\(^4\) within this particular area of legislation has stated that chemical sludge from the pulp and paper industry containing significant amounts of metal salts based on e.g. aluminium cannot be exempted from (2013:253). However, Holmgren is open to the possibility, to exempt sludge from (2013:253) that has been treated or were coagulation chemicals have been recovered from the sludge with methods such as the ReAl process.

2.5. Literature review
In this chapter, previous research regarding the production of coagulation chemicals and the importance of the impact of infrastructure in the wastewater treatment industry are presented.

2.5.1. Aluminium sulfate (coagulation chemical)
An LCA and carbon footprinting was conducted on behalf of INCOPA (2014) for various coagulants produced by nine INCOPA member companies. Aluminium sulfate which is the coagulant used for chemical coagulation in this thesis was also one of the coagulants included in the INCOPA report. The study encompassed every stage of the production of the coagulant from resource extraction (cradle) to the factory gate (up until the coagulant is shipped to the customer), also known as a cradle-to-gate life cycle. The result showed that the carbon footprint for Aluminium sulfate 8.25% aluminium oxide (liquid) and aluminium sulfate 17% aluminium oxide (solid) was equal to 0.148 kg CO\(_2\)-eq/kg product and 0.295 kg CO\(_2\)-eq/kg product respectively. Aluminium hydroxide had by far the largest impact on the total carbon footprint with a contribution of about (70%) followed by truck (12%), sulfuric acid (9%) and electricity (7%) (see Figure 2).

![Figure 2 The impact contribution from the aluminium sulfate production.](image)

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\(^3\) Swedish: Förordning (2013:252) om stora förbränningsanläggningar. This ordinance regulates large combustion plants with at least 50 megawatts (MW) installed power input.

\(^4\) Torbjörn Holmgrens statement is based on his professional opinion and not on behalf of the county administrative board in Gävleborg where he is employed.
2.5.2. The importance of infrastructure in LCA of sewage treatment

In this subchapter, a prior LCA study will be presented in order to provide a basis for deciding, whether to include or exclude the infrastructure (extracted and processed materials) necessary for the sludge treatment systems.

An LCA study performed by Igos et al. (2013) investigated the contribution of infrastructure and operation at a unit process level for two water treatment plants in France. The two water treatment plants are used for the production of drinking water. The total single score results for the whole plants showed that the contribution of infrastructure varies between 4-12 % depending on the use of impact assessment method. Steel and concrete and the lifetime of these two materials were found to be the most important contributors to the environmental impact of the infrastructure. Along these lines, a sensitivity analysis was conducted with a variation of (±50 %). The baseline for the lifetime was set to 30 years ± 15 years. With a lifetime of 15 years, the environmental impacts from infrastructure could increase by 5-30 %. Similarly, using a lifetime of 45 years could reduce the impacts by up to 10 %. Figure 3 a) shows the percentage of infrastructure and operational environmental impact, as well as, the total single score for each unit process. The contributing fractions of the background processes to the single score are shown in b).

Ultrafiltration which is part of the ReAl process only contributes with 4% of the total environmental impact (see Figure 3). In conclusion, it is evident that the operational phase of a system in the wastewater treatment industry has by far the largest environmental impact. Thus, the author of this thesis has decided to not include the infrastructure of the studied sludge treatment systems.
3. Methodology

This study was conducted in accordance with the four LCA phases of the ISO 14040 series. A goal and scope definition was formulated, with among other things; an appropriate functional unit, system boundaries, assumptions and limitations. Furthermore, a life cycle inventory analysis for data collection and calculation was executed, as well as a life cycle impact assessment to assess the magnitude of the potential environmental impacts and a life cycle interpretation for understanding the LCA results. A large part of LCA is the collection of data which was needed in order to model the sludge treatment systems. Data related to the ReAl process was mostly provided by Purac. However, some of the data was not given due to confidentiality, consequently, assumptions based on literature was necessary to fill the data gap. Literature research was based on normal Google searches and databases such KTHB Primo, Google Scholar and Web of Science.

The next subchapters will describe what a life cycle assessment is and outline how to conduct one in accordance with the ISO 14040 series.

3.1. Life cycle Assessment

Life cycle assessment is a holistic decision-making tool providing help, to analyze environmental aspects and potential impacts of a product or system, throughout its entire life cycle. An entire life cycle is also called a cradle-to-grave approach i.e. from raw material extraction, along with material processing, manufacturing, use phase, and to waste management (Finnveden et. al., 2009).

The International Organization for Standardization (ISO) has put forward a series of documents regarding LCA which is referred to the ISO 14040 series (LCA Student Handbook, p. 12). This series includes international standards, describing the principles and a methodological framework (ISO 14040), as well as requirements and guidelines (ISO 14044), on how to carry out an LCA (ISO, 2006a; ISO, 2006b). Another international standard (ISO 14045) in the series is about guidelines and requirements about Eco-efficiency assessment. Additionally to the international standards, there are some technical reports; (ISO/TR 14047) which give examples on how to apply ISO 14044 to impact assessment situations, (ISO/TS 14048) describes how to format and document data, and (ISO/TR 14049) covers, how to apply ISO 14044 to goal and scope definition and inventory analysis. Another important source of guidelines for conducting an LCA is the International Reference Life Cycle Data System (ILCD) Handbook which is a set of documents that are in line with ISO 14040/44 (European Commission, 2016).

LCA is only one environmental management tool among many (e.g. environmental impact assessment and risk assessment). Consequently, an LCA is not always the best-suited tool in all decision situations. For example, an LCA normally lacks economic and social aspects of a product. However, the life cycle approach described in (ISO 14040), can very well be applied to these aspects. A unique feature of an LCA is its comprehensive and broad perspective on environmental issues. It is very important when one wants to avoid problem shifting i.e. for example, when a problem from one part of the system is transferred to another part or when a problem is solved and a new one occurs somewhere else (Finnveden et al., 2009).

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5 The definition of a “product” includes services in the international standard.
According to (ISO 14040, 2006), an LCA consists of four stages (see Figure 4):

- Goal and scope definition;
- Life cycle Inventory analysis;
- Life cycle Impact assessment (LCIA); and
- Life cycle Interpretation (LCI)

Figure 4 Life cycle assessment framework (ISO, 2006a).

LCA results can vary even if the same product or system has been assessed. This is due to the fact that LCA involves simplifications, assumptions and subjective choices during the process. Hence, transparency regarding all the assumptions and value-based choices made become crucial for understanding the result of an LCA study. (LCA Student Handbook, p. 15)

3.1.1. Goal and scope definition

The goal and scope definition is more than a simple “introduction” of an LCA. It is the foundation, where conditions and constraints are created for the continuation of the whole study. The goal and scope of an LCA must clearly be defined according to (ISO 14044:2006, p. 7). However, the scope of a study can be refined, since LCA is an iterative process. The goal definition shall include:

- The intended application i.e. how will the results of the LCA be used;
- Why the study is carried out;
- The intended audience i.e. who will be interested in the results of the study; and
- Whether the results are intended to be used in comparative assertions intended to be released to the public. (ISO 14044:2006, p. 7)

The scope of an LCA shall be defined in accordance with the goal and include:

- A description of the studied system;
- A functional unit;
- Defined system boundaries (scoping);
- Allocation procedures or allocation methods (Co-product allocation);
- Impact categories and LCIA methodology;
- Assumptions and limitations;
- Data requirements;
- Data quality requirements;
- Type of critical review and;
- Type and format of the report. (ISO 14044:2006, p. 7)

Two of the most important aspects of the goal and scope definition is the functional unit and the system boundary. A functional unit is a quantifiable baseline to which all the inputs and outputs of a system is referenced to. More specifically, a functional unit shall provide information regarding a system or product provided function (what it performs), in what quantity, for how long (duration), and to what quality. The System boundary describes what is included in the system and when resource flows are no longer considered. Generally, it is appropriate and ideal to follow the flows from “cradle-to-grave”. There are five aspects of boundaries that are of interest and shall be presented: temporal (time period), spatial (geographic), (energy, material, and natural system), cut-off-criteria, and allocation. It is helpful to draw a flow diagram, including, all unit processes and flows of material and energy of the studied system. (ISO 14044:2006, pp. 8-9)

3.1.2. Life cycle inventory analysis

The second phase of an LCA is the life cycle inventory analysis. This is the phase where all the necessary, qualitative and quantitative input and output data, to meet the goals by modeling the studied system, is collected. In order to quantify the inputs and outputs, the data has to be collected for each unit process within the system boundary, either via measurements, calculations or estimates. Data collected from public sources must be referenced properly, and whenever data has had a considerable impact on the study results, it is especially important to document: the collection process, the data collection date, and the data quality indicators. Since the data needed to describe a product or system can be found from various sources, it is important to collect data that underpin and support a unified model. Hence, an inventory analysis shall also include (ISO 14044:2006, p. 11):

- A description of each of the unit processes, including all the inputs and outputs;
- A process flow diagram, showing every unit process and their interrelationships;
- An outline on how the data has been collected and calculated, including assumptions; and
- A documentation experimental data. (ISO 14044:2006, p. 11)

The inventory analysis shall check the data validity and confirm the data quality requirements during the data collection phase. Unit process flow data shall relate to the functional unit. The system boundaries set in the scope definition shall be refined based on the results of a sensitivity analysis. (ISO 14044:2006, p. 13)

3.1.3. Life cycle impact assessment

Life cycle impact assessment (LCIA), analyzes in a quantitative way, a product or systems potential environmental impacts of all its inputs and outputs (e.g. material, energy, emissions), collected and modeled in the inventory analysis by using relevant impact category indicators (LCA Student Handbook, p. 137). LCIA simplifies the analysis and the communication of the inventory analysis by converting the inventory data into common units and groups them into a few, often around 15 impact categories (Carlson & Pålsson, 2008). According to ISO 14044, there are three compulsory and four optional elements in a LCIA (see Table 1).
Table 1 Compulsory and optional elements in a LCIA.

<table>
<thead>
<tr>
<th>Compulsory Elements</th>
<th>Impact categories, classification, characterization</th>
</tr>
</thead>
<tbody>
<tr>
<td>Optional Elements</td>
<td>Normalization, grouping, weighting and data quality</td>
</tr>
</tbody>
</table>

The mandatory elements start with choosing impact categories, category indicators and characterization models, followed by classification i.e. linking the inventory results (inputs and outputs) with the impact categories. Lastly, characterization is the quantification of the impact category indicator results (ISO 14044:2006, p. 16).

**Impact categories, category indicators and characterization models**

Selecting impact categories, category indicators, and characterization models are becoming more and more difficult since there are more and more LCIA methods to choose from. The practitioner must select these methods in the starting phase of an LCA, during the goal and scope definition, since it is a fundamental decision, which can affect the outcome of a study. To make things easier for the practitioner, guidelines and recommendations to choose an appropriate LCIA method have been established in the ISO 14044 standard. (Curran, 2017)

The first step is to decide which impact categories reflect the studied product or system the most by choosing a pre-made impact assessment method which includes predefined impact categories, category indicators and characterization models (Pålsson, 2011). Impact categories represent different environmental impacts e.g. global warming, eutrophication, acidification etc. (Baumann & Tillman 2004). The selected Impact categories should represent and capture the fundamental and overall environmental burden of the studied product or system (ISO 14044:2006, p. 17).

Inputs and outputs collected in the inventory analysis are converted into impact category indicators (or characterization results) with the help of characterization factors which describe the relative contribution of inputs or outputs to an impact category (EC, 2010). Characterization factors represent the impact categories in a quantitative way e.g. methane contributes to climate change with 25 kg CO₂-eq, while CO₂ contributes with 1 kg CO₂-eq to climate change (ISO 14044:2006, p. 6).

**Optional elements (normalization, grouping and weighting)**

Normalization is a value based on the category indicator results in relation to a reference value, or in other words, it is the ratio of the characterization result relative to a reference value. Grouping is the ranking (e.g. a scale from 1-10) and/or sorting (e.g. by attribute) of impact categories. Weighting is an approach where numerical factors based on subjective values are used to transform indicator results assigned to various impact categories (ISO 14044:2006, pp. 21-22).

**3.1.4. Life cycle interpretation**

Life cycle interpretation is the final phase of an LCA procedure in which the results of an inventory analysis or a LCIA, or both, are summarized and discussed as a basis for conclusions, recommendations and decision-making in accordance with the goal and scope definition. The interpretation shall include identification of the significant issues of the inventory and impact assessment phases. Moreover, a data completeness, data consistency and sensitivity check must be provided. In the last phase of an LCA, it is also important to evaluate the potential impact the chosen functional unit and system boundaries have had on the results. (ISO, 2006a)
4. Comparative LCA of sludge treatment systems

In this chapter, the comparative LCA of the two sludge treatment systems is presented for the most part in accordance with the ISO 14040 series and in line with the four LCA phases: goal and scope definition, life cycle inventory analysis, life cycle impact assessment, and life cycle interpretation.

4.1. Goal of the study

4.1.1. Purpose

The purpose of this LCA is to investigate the environmental impacts of two sludge treatment systems that treat chemical sludge deriving from the pulp and paper industry. More specifically, the environmental impacts are evaluated based on a specific set of impact categories determined in subchapter 4.4. The research questions asked are:

1. Is a sludge treatment system including the ReAl process a more environmentally friendly alternative in regards to the selected impact categories, from a life cycle perspective, compared to a conventional system which does not recover any coagulation chemicals from the sludge?

2. Which inputs in the systems are contributing the most to the environmental impacts?

Lastly, since the ReAl sludge and the chemical sludge can be treated with a range of different treatment equipment, the LCA shall also provide an answer to:

3. Which dewatering and sludge drying equipment can handle these two sludge types in the most environmentally friendly way with respect to the chosen impact categories?

4.1.2. Intended audience and application of the LCA results

This study is intended, first and foremost to Purac and ÅF since they have requested the study. They want to know, whether the ReAl process is a sustainable technology to treat chemical sludge with from an environmental life cycle perspective and if they can recommend it to potential future customers in this regard. Furthermore, the Swedish Environmental Protection Agency and the county administrative board are also very interested in the LCA results because one of their agendas is to combat eutrophication, and the ReAl process has the potential to solve the sludge problem of chemical coagulation. Solving this problem is necessary for chemical coagulation to have its breakthrough in Sweden. Thus, this comparative LCA could provide these authorities with a decision basis regarding how to regulate the management and classification of the chemical sludge treated in a sludge treatment system which includes the ReAl process. Actors within the water treatment industry, as well as, the pulp and paper industry will also have an interest in this LCA.

4.2. Scope of the study

This LCA attempts to assess the entire life cycle of both sludge treatment systems, which includes the environmental impacts from material extraction, manufacturing, transport, usage and end-of-life management.

4.2.1. A description of the studied system

The scope of this study includes two sludge treatment systems with the function to treat the sludge deriving from wastewater treatment of pulp mill sewage. The major difference between the two systems is the ReAl process and the variation of dewatering and drying equipment. A
visual representation of the two systems is presented in Figure 5. The first stage in both systems is chemical coagulation where coagulants are added in the form of aluminium sulfate which results in the sludge production. The chemical sludge in system 1 (the conventional system without the ReAl process) enters a dewatering stage and a sludge dryer with the purpose to increase the sludge dry content before incineration. In system 1 two dewatering options, a decanter centrifuge and a filter press are evaluated with different dewatering efficiencies (22% DM and 30 % DM respectively). The chemical sludge dewatered with a decanter centrifuge is called system 1a which is the conventional dewatering equipment at Iggesunds pulp mill. The chemical sludge dewatered with a filter press is called system 1b which was included in order to ensure that the results are not affected by the use of a different dewatering equipment in system 2.

When the sludge in system 2 enters the ReAl process the coagulants are recycled back to the chemical coagulation stage, thus, creating a ReAl sludge containing far less aluminium than the chemical sludge in system 1. In System 2, two options are evaluated, system 2a (excludes drying) and system 2b (includes drying). System 2b was considered and evaluated with a dryer in order to ensure that the comparative results between system 1 and system 2 would not be affected by the exclusion of drying in system 2a. The last stage in both system 1 and system 2 is co-incineration of bark and sludge which creates ash as a final waste flow. The ash can be used for example in construction materials on site to cover landfills or for road construction (IVL, 2003).

**Incineration**

Incineration was chosen as the last process to manage the sludge. One reason being that pulp mills have incinerators on site and even if some parts of the incinerator might have to be upgrade by e.g. installing a new electrostatic precipitator (ESP) and/or increase the capacity of the ash handling (Gyllenhammar, pers.comm., 7 March), it is still the best option out there. Directly landfilling the sludge is as stated in subchapter 2.4.1 prohibited since 2005 in Sweden and processes such as digestion and composting is not suited for chemical sludge handling. Chemical sludge from the pulp and paper industry which entails high amounts of sulfur cannot be digested because the methane-forming organisms are inhibited (Granström et al., 2014). Even if it would work there are still many obstacles which prevents digestion from being a suitable option. Digestion would for example require large investments in either a digestion plant or the cost of transporting the sludge to someplace else where one is located (Granström, pers.comm., 23 February). Spreading compost from chemical sludge from the pulp and paper industry on fields or woods as soil improvers is not allowed in accordance with SPCR 120 certification. It is not mandatory to follow SPCR 120 but many relevant actors are using this certificate because it ensures a certain quality of the compost and thus it is not a good option to compost chemical sludge in Sweden (Persson, 2015).

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6 Notice, landfilling of ash is part of the sludge treatment systems but it is prohibited to landfill chemical sludge except if the county administrative board has given out a dispensation.
Figure 5 Initial flowchart showing sludge treatment system 1 to the left and sludge treatment system 2 to the right. In each system, there are two options of equipment which are categorized, from left to right, as system 1a, system 1b, system 2a, and system 2b.

4.2.2. Type of LCA
This study is conducted as a comparative LCA which means that the different properties of the two systems have been taken into account. Hence, processes with similar inputs and outputs in both systems have been omitted.

There are generally speaking two life cycle inventory modeling approaches, an attributional and a consequential approach. An attributional approach refers to a way of accounting the
environmental impact over a product or systems life cycle with factual and measurable data, while a consequential modeling approach is based on changes and effects a decision has on the studied product or system. Since the purpose of the study is to compare and assess the current environmental impact of the two sludge treatment systems an attributional modeling approach was chosen. In other words, this LCA does not assess any change oriented effects which would be more suited with a consequential life cycle inventory approach. (EC, 2010)

4.2.3. Functional unit
The functional unit is especially important in a comparative LCA in order to ensure a fair comparison. The first function of the studied systems was the treatment of the sludge deriving from the chemical coagulation of the pulp mill sewage. The functional unit was therefore based on the annual amount of sewage which enters the chemical coagulation (50 057 195 m$^3$). The second function was to treat the sewage to a particular quality. More specifically, to reduce the soluble COD (SCOD) to the recipient by 60% which was estimated to 4 500 tonnes/year (see subchapter 4.3.2).

4.2.4. System boundaries
The main unit processes included in the system boundaries of the sludge treatment systems are shown in Figure 5 and they are chemical coagulation, the ReAl process, dewatering, drying, incineration, and landfill. The configuration of the unit processes in each system is described in subchapter 4.2.1 and are as well displayed in Figure 5.

Time horizon
The data required for the modeling of the sludge treatment systems are based on the year of 2017. The data generated by the energy and mass balance in subchapter 4.3.2 is based on the production of bleached pulp per year, thus the results from the model created in the LCA software will represent the annual environmental impact of the sludge treatment systems. Furthermore, so-called long-term emissions (emissions that occur over 100 years) (Pré, 2016c) are excluded due to the uncertainty involved in such modeling.

Geographical boundaries
The sludge treatment systems that are being assessed are based on and dimensioned after a hypothetical but typical modern pulp mill in Sweden. Thus, some inputs such as electricity are modeled in accordance with Swedish circumstances. Unfortunate, there was a lack of Swedish data and thus most inputs were based on European and global markets. However, since ÅF and Purac are interested to export the ReAl process beyond the Swedish border, still makes these two markets relevant. Furthermore, the distances of transportation involved in the study are modeled as market averages$^7$ from the Ecoinvent 3 database since no particular place in Sweden has been selected for the pulp mill and the sludge treatment systems.

Energy, material, and technical and natural system
The cradle-to-grave approach, which this LCA is based on includes: the extraction and processing of raw materials used to produce the consumables (such as the coagulant), the operation phase (electricity, heat, water etc.), and the waste management (recovery of coagulant, incineration of sludge and disposal of ash). All the unit processes in the sludge treatment systems and a number of inputs and the associated emissions on site are considered to be in the foreground. However,

$^7$ The average distance of transportation from a production site to a user.
the production of the inputs from a third party and their associated emissions are considered to be in the background. The boundary between the technical systems (technosphere) and the natural system (environment), as well as the associated inputs and emissions, are displayed in Figure 7.

Cut-off criteria

The infrastructure of the unit processes was not included in this study because the operation phase has a far larger environmental impact as concluded in subchapter 2.5.2. Processes that are the same in both systems, such as the biological wastewater treatment, have not been considered because it will not affect the conclusion in a comparative sense. Lastly, the access to data for the ReAl process was limited due to classified information. The ReAl process was therefore viewed as a “semi-black box” and some data could not be appointed to a specific sub process (acidification, heat exchanger, and ultrafilter). Hence, the discussion and interpretation of the environmental impacts deriving from the ReAl process had some limitations.

4.2.5. Allocation procedures

The chosen allocation approach in the integrated database (Ecoinvent 3) in SimaPro was “allocation recycle content”, also known as a “cut-off system model”. This approach is easy to use and less convoluted than other models. This approach allocates the full burden of waste and by-products to the primary user of a product (the first waste producer). When a product enters the second life cycle through recycling or reuse, it will be “burden-free” i.e. there will be no environmental impact, except the recycling process, from the forthcoming life cycles. (Ecoinvent, u.d.)

An allocation problem arises when the energy from co-incineration with sludge and bark is recovered and allocated to the sludge. This problem is solved by system expansion – which means that the recovered energy can be subtracted from the energy demand in previous unit processes such as the sludge dryer.

4.2.6. Assumptions and limitations

The list below summarizes the overall assumptions and limitations of the study:

- The infrastructure was not included in the assessment because the effort to model this would probably not be proportional to its impact on the conclusion.
- The inputs and outputs of the sludge treatment systems are based on a modern pulp mill in Sweden which is assumed to produce 800 000 tonnes of bleached pulp per year.
- The study is not based on a particular geographical area, thus mostly European, global and Swedish data sets were used from the Ecoinvent 3 database.
- Transportation was modeled from average market data in Ecoinvent 3.
- There was limited access to data concerning the sub processes of the ReAl process (acidification, heat exchanger, and ultrafiltration).
- Long-term emissions were excluded.
- Only the amount of ash was monitored. In other words, the environmental impact of the ash used in construction materials such as roads and to cover landfills was not considered.
- Limited knowledge about the elemental content of chemical sludge and ReAl sludge. NO\textsubscript{X} emissions from sludge and bark incineration were therefore modeled based on emission requirements in Sweden and SO\textsubscript{X} emissions were based on literature data.
4.2.7. Impact assessment method

There are multiple impact assessment methods with pros and cons which must be considered so that the most appropriate one is selected. These methods can, for example, include different impact categories, their selection of indicators and geographical focus. The chosen impact assessment method for the life cycle impact assessment was ReCiPe. ReCiPe is a widely used, state-of-the-art, impact assessment method which transforms, the many and hard to understand results from the Life cycle inventory analysis (inputs and outputs), to a few simpler indicator scores (SimaPro, 2017). These indicator scores represent the effect on the impact categories.

The diagram shown in Figure 6 displays the structure of the ReCiPe method and the relationship between the inventory result, environmental mechanisms, midpoint and endpoint indicators, as well as the single score. The environmental mechanisms part 1 are based on certain scientific data which are then converted into the midpoint indicators. Midpoint indicator can be hard to interpret, hence, it is possible to convert them with the environmental mechanism part 2, based on relatively uncertain data, to easier to understand but more uncertain endpoint indicators. The estimated indicators are then aggregated into single scores by characterization factors related to a specific impact indicator (ReCiPe, 2015). One example of such an impact indicator is the impact of methane emissions on climate change which is calculated with the scientific factor of 25 kg CO2 eq./kg of CH4 (Lehtinen et al., 2011). Similar, assumptions (uncertainties) and choices linked to the aggregation steps are grouped together in three different perspectives (time horizons). These perspectives are:

- Individualist (I) - short time frame based on an optimistic view that technology can solve most problems in the future;
- Hierarchist (H) - based on scientific consensus and policies and;
- Egalitarian (E) - long time frame based on precautionary thinking (PRé, 2012).

ReCiPe has eighteen midpoint indicators and three endpoint indicators (see Figure 6) and each method have been created for three different perspectives (explained in the previous paragraph). Hence, the practitioner of an LCA must decide which type of indicator and perspective is best suited in their particular case. Guidance and recommendation regarding the choice of LCIA methodology are provided by the European Commission’s ILCD Handbook (EC, 2017). In this study, the European ReCiPe midpoint (H) version 1.12 April 2015 method was chosen. Moreover, the
default time perspective (Hierarchist) was selected since it is a balanced approach based on scientific consensus with less uncertainty.

Figure 6 Structure of the ReCiPe methodology (ReCiPe, 2015).

4.2.8. Normalization and Weighting

Normalization and weighting are two of the optional steps in the ISO 14040 series because of the involvement of subjectivity. Weighting is not allowed in comparative assertions intended to be disclosed to the public (ISO 14044, 2006), thus weighting was excluded. Normalization was also excluded because the practitioner of this thesis does not think that relating the characterization results to a reference value based on the average environmental impact from a European citizen per year (ReCiPe, 2011; Ecoinvent, 2014) does not add any value to the life cycle interpretation. In contrary, there is a risk of misinterpretation of the normalization results.

4.2.9. Data quality and data requirement’s

As concluded in chapter 2.5.2, the operation phase of similar sludge treatment systems to the once in this study, contribute the most to the total environmental impact. With this in mind, the focus was to use as much “site-specific data” as possible. However, since the ReAl process has only been pilot tested by Purac and it does not exist as a full-fledged system, thus the comparison between the two systems had to be dimensioned based on a modern pulp mill in Sweden. Moreover, the operational data has been estimated by carrying out an energy and mass balance

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8 The ReCiPe Midpoint (H) method includes normalization based on European impacts per year and person
based on results from the pilot tests and the assumed pulp mill size. Average and generic data, as well as the database Ecoinvent 3, have been used to fill data gaps.

It was particularly important to have accurate data for the coagulant (aluminium sulfate) since it is a key input in the sludge treatment systems. The intent was therefore to find data appropriate for the geographical boundary of Sweden and Europe. Furthermore, the electricity required in all processes was produced in Sweden, while most of the remaining datasets used in Ecoinvent 3 were based European markets and a few were based on global data.

4.2.10. Modeling software
There is a wide variety of LCA software’s on the market today where some of them are fully compliant with the ISO 14040/1444 standards e.g. SimaPro and GaBi, and others are more limited but simpler to use e.g. COMPASS and Package Modeling. The four Software’s already mentioned are all commercially available but there are also some that are not. However, the choice of software can have a significant effect on the outcome of a study (Speck et al., 2015), hence, it is important to choose the most suited software in each particular case.

The LCA software used for modeling the two sludge treatment systems was SimaPro 8.2.3 developed by Pré Consultants. SimaPro was chosen since it is by far the most widely used LCA software (Speck et al., 2015) which is an advantage when one wants to compare LCA studies. SimaPro also supports many different impact assessment methods and multiple databases. The practitioner of this study has selected, in SimaPro, the impact assessment method “ReCipe (version 1.12, April 2015)” and the database “Ecoinvent 3” (allocation, recycled content – system and unit).

4.2.11. Critical review considerations
A critical review can be performed by an independent internal or external expert and it shall be included in a LCA according to ISO 14044 (2006) If the study is intended to be used as a comparative assertion intended to be disclosed to the public. However, it important to clarify that a master thesis such as this study is conducted for educational purposes and cannot be used in a public comparative assertion as a marketing tool from neither ÅF, Purac nor anybody else.

Within the scope and process of a master thesis, each student has a supervisor (an expert within the study area which in this case is LCA). Thus, the supervisor of this master thesis has provided insightful remarks on how to perform an LCA. The supervisor has no interest in the study result and it is safe to say that the guidance and advice have been impartial. However, the supervisor has not reviewed all the data in the life cycle inventory analysis in greater detail. Thus, the supervisor cannot be equated with an external reviewer.

4.3. Life cycle inventory
This chapter begins with a detailed process flowchart of the two sludge treatment systems. Furthermore, all the necessary, quantitative data that was collected in accordance with the goal and scope definition are described here – including datasets from the database Ecoinvent 3, inputs and output data from the energy and mass balance, as well as, calculations based on estimates and assumptions.

4.3.1. Process flowchart
In Figure 7, a process flowchart shows all the inputs and outputs from the two main sludge treatment systems (system 1 on the left-hand side and system 2 on the right-hand side). An
asterisk in system 1 describes that there are two dewatering equipment’s – a decanter centrifuge (22% DM) in system 1a and a filter press (30% DM) in system 1b. Similarly, in system 2, an asterisk describe that two drying options have been evaluated — one option excluding drying (system 2a) and one option including drying (system 2b). To summarize, there were four sludge treatment systems that were modeled and evaluated.

4.3.2. Energy and mass balance

An energy and mass balance was created to determine the inputs and outputs of the studied sludge treatment systems shown in Figure 7. The energy and mass balance was put together in collaboration with representatives from ÅF and Purac. The quantitative data determined by the
balance is based on assumptions, estimates and experience data from ÅF and Purac. Much of the experience data regarding the ReAl process was obtained from pilot tests conducted by Purac at Iggesunds pulp mill. All the data presented in the following subchapters are based on annual values in accordance with the functional unit. Also, the sludge treatment systems were assumed to run every day during the year.

Assumptions
The sludge treatment systems studied in this thesis are only a small part of a bigger system which also includes a pulp mill and treatment stages before the first unit process (chemical coagulation). It is necessary to assume the size of the pulp mill and the SCOD reduction in prior treatment processes in order to determine what the SCOD content is in the sewage when it enters the chemical coagulation. These assumptions are important because the sludge production from the chemical wastewater treatment is very much dependent on the SCOD content in the sewage.

The first assumption that was made in the life cycle inventory analysis and the energy and mass balance was that the sludge treatment systems were part of, and connected to, a pulp mill producing 800 000 tonnes of bleached Kraft pulp per year. This production capacity was chosen since it represents a modern sized pulp mill in Sweden.

The sewage from the pulp mill, before it enters the chemical coagulation, was assumed to be treated in a biological activated sludge process. The amount of SCOD in the sewage leaving the pulp mill and entering a biological wastewater treatment stage (not included in the studied systems), was estimated with 30 kg per air-dry tonne (ADT) pulp. Furthermore, the biological wastewater treatment was assumed to reduce the SCOD by 70% (ÅF, 2017). With these assumptions in mind, the SCOD in the sewage which enters the chemical coagulation was calculated as 7 500 tonnes/year (see Figure 8).

The following subchapters will present all the important assumptions and the data determined by the energy and mass balance for each unit process (chemical coagulation, ReAl process, dewatering, drying, incineration, and landfill). Every assumption made in the life cycle inventory analysis is summarized in Table 16 (see appendix) and the sludge flow in each unit process for each system is shown in Table 2.

*Table 2 The Sludge flow into each unit process for each sludge treatment system expressed as m3/year.*

<table>
<thead>
<tr>
<th>Unit Process</th>
<th>System 1a</th>
<th>System 1b</th>
<th>System 2a</th>
<th>System 2b</th>
<th>Unit</th>
</tr>
</thead>
<tbody>
<tr>
<td>Chemical coagulation</td>
<td>50 057 195</td>
<td>50 057 195</td>
<td>50 057 195</td>
<td>50 057 195</td>
<td>m³/year</td>
</tr>
<tr>
<td>ReAl process</td>
<td>-</td>
<td>-</td>
<td>365 365</td>
<td>365 365⁹</td>
<td>m³/year</td>
</tr>
<tr>
<td>Dewatering</td>
<td>262 070</td>
<td>262 070</td>
<td>45 990</td>
<td>45 990</td>
<td>m³/year</td>
</tr>
<tr>
<td>Drying</td>
<td>35 770</td>
<td>26 280</td>
<td>-</td>
<td>10 694.5</td>
<td>m³/year</td>
</tr>
<tr>
<td>Incineration</td>
<td>22 155.5</td>
<td>18 308.4</td>
<td>10 694.5</td>
<td>7 533.6</td>
<td>m³/year</td>
</tr>
</tbody>
</table>

⁹ Notice, the sludge flow entering the ReAl-process is higher than the sludge flow entering the dewatering. This is due to the fact that the recovery of aluminium sulfate in the ReAl-process also leads to recycling of COD. System 2a and system 2b will therefore have to treat more COD in the chemical coagulation which leads to the production of more sludge.
**Chemical coagulation (system 1 and system 2)**

In Figure 8, the sewage enters the chemical coagulation with 7,500 tonnes of SCOD per year. The addition of the coagulant (aluminium sulfate) was assumed to reduce the SCOD by 60% (4,500 tonnes/year) (ÅF, 2017) which means that the remaining 40% of SCOD (3,000 tonnes/year) goes to the recipient.

![Diagram of chemical coagulation process](image)

*Figure 8 The amount of SCOD entering and leaving the chemical coagulation. Notice, the sludge continues to the dewatering stage in system 1 and to the ReAl process in system 2.*

The aluminium sulfate with 9.1% concentration was dosed in accordance with the amount of SCOD in the sewage. Consequently, the total amount of aluminium sulfate (9.1%) needed to coagulate the SCOD in system 1 was estimated to 11,190 tonnes/year, while in system 2, the ReAl process recovers about 70% of the Al\(^{3+}\), thus only approx. 3,180 tonnes/year of “fresh” aluminium sulfate was needed (ÅF, 2017). A dataset for aluminium sulfate was created in SimaPro which is shown in Table 4.

The sludge particles formed during the chemical coagulation are separated by dissolved air flotation (DAF)\(^\text{10}\). A polymer is also added to the DAF flocculation. The total annual consumption was estimated as 100 tonnes of polymer and the polymer usage in both system 1 and system 2 was considered to be equal (ÅF, 2017). The polymer was modeled in SimaPro as a polyacrylamide since most polymers are based on those (Kemira, 2003; Flocculants, 2015).

DAF takes advantage of the density difference between the particles and the surrounding liquid. During flotation, microbubbles attach to particles and carry them to the surface where the sludge is removed with a scraper. The microbubbles are produced when the air is dissolved under high pressure and then released at atmospheric pressure, which makes DAF a very energy intensive process (Féris et al., 2000; Persson, 2011). Thus, the electricity for the DAF has been estimated by

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\(^{10}\) Dissolved air flotation is considered to be part of the chemical coagulation stage. Thus, it is not visualized in any of the figures.
assuming that the dispersion flow is 30% of the sewage flow which resulted in 3 GWh/year (ÅF, 2017). The electricity demand from DAF was considered to be the same in both system 1 and system 2. The inputs in the chemical coagulation is shown in Figure 3.

Table 3 Inputs in the chemical coagulation.

<table>
<thead>
<tr>
<th>Input</th>
<th>System 1</th>
<th>System 2</th>
<th>Unit</th>
<th>Dataset in Ecoinvent 3</th>
</tr>
</thead>
<tbody>
<tr>
<td>Aluminium sulfate</td>
<td>11 190</td>
<td>3 180</td>
<td>tonne/year</td>
<td>Aluminium sulfate</td>
</tr>
<tr>
<td>(9.1% Al³⁺)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Polymer</td>
<td>100</td>
<td>100</td>
<td>tonne/year</td>
<td>Polyacrylamide {GLO}</td>
</tr>
<tr>
<td>Electricity</td>
<td>3</td>
<td>3</td>
<td>GWh/year</td>
<td>Electricity, medium voltage {SE}</td>
</tr>
</tbody>
</table>

The amount of sludge leaving the dissolved air floatation and continuing to the next sludge treatment stage is dependent on the leakage of sludge from the dissolved air floatation. It is assumed that the sludge retention is 95%, thus 5% of the sludge is leaking to the recipient (ÅF, 2017).

Aluminium sulfate dataset
An aluminium sulfate dataset was created in SimaPro based on the production of 1 kg of solid aluminium sulfate (9.1% Al³⁺ and 17.2% Al₂O₃). Solid aluminium sulfate is still produced in Sweden but in the rest of the world, it is usually produced as a liquid (4.33% Al³⁺ and 8.25 Al₂O₃) because it requires less advanced technology which makes the process cheaper (Stendahl, pers.comm., June 28). The chemical reaction in Equation 1 describes the relationship between the reactants (aluminium hydroxide, sulfuric acid and water) and the products (aluminium sulfate and water of crystallization).

\[
2\text{Al(OH)}_3 + 3\text{H}_2\text{SO}_4 + 8\text{H}_2\text{O} \rightarrow \text{Al}_2(\text{SO}_4)_3 \cdot 14\text{H}_2\text{O} \quad \text{Eq. 1}
\]

The mass of each reactant was determined by the fact that the Al³⁺ concentration is 9.1% (which means that there is 91g per kg aluminium sulfate), the sulfuric acid has a concentration of 96% and by applying molar mass relationships. The dataset of aluminium sulfate with all inputs and quantities are shown in Table 4.

Table 4 Inputs to aluminium sulfate production (9.1% Al³⁺) (Stendahl, pers.comm., June 28).

<table>
<thead>
<tr>
<th>Input</th>
<th>Quantity</th>
<th>Unit</th>
<th>Dataset in Ecoinvent 3</th>
</tr>
</thead>
<tbody>
<tr>
<td>Aluminium hydroxide</td>
<td>263</td>
<td>g</td>
<td>Aluminium hydroxide {GLO}</td>
</tr>
<tr>
<td>Sulfuric acid (96%)</td>
<td>517</td>
<td>g</td>
<td>Sulfuric acid</td>
</tr>
<tr>
<td>Water</td>
<td>220</td>
<td>g</td>
<td>Tap water {RER}</td>
</tr>
</tbody>
</table>

A dataset was created in SimaPro (see Table 4).

This data set was created as a modification of “Sulfuric acid {GLO} | production | Alloc Rec, U”. The modification entailed changes to resemble European conditions. All data inputs but sulfur had a European dataset and were therefore changed accordingly.
All the inputs in Table 4 include transport, however, the transport of aluminium sulfate must also be considered. The aluminium sulfate is assumed to be produced in Sweden which was modeled based on an aluminium sulfate data set\(^{13}\) from Ecoinvent 3. The transport data from the data set was modified by excluding transoceanic ship as a mean of transport, changing from global to European train, and changing from global to EURO\(^{3}\) lorry. The distance was assumed to be applicable in Sweden and thus it was kept as it is. The transport data of aluminium sulfate used in this study is shown in Table 5.

\textit{Table 5 Transport of aluminium sulfate.}

<table>
<thead>
<tr>
<th>Means of transport</th>
<th>Distance</th>
<th>unit</th>
<th>Dataset in Ecoinvent 3</th>
</tr>
</thead>
<tbody>
<tr>
<td>\textit{Train}</td>
<td>0.31</td>
<td>tkm</td>
<td>Transport, freight train {Europe without Switzerland}| market for | Alloc Rec, U</td>
</tr>
<tr>
<td>\textit{Lorry}</td>
<td>0.21</td>
<td>tkm</td>
<td>Transport, freight, lorry 16-32 metric ton, EURO3 {GLO}| market for | Alloc Rec, U</td>
</tr>
<tr>
<td>\textit{Barge}</td>
<td>0.025</td>
<td>tkm</td>
<td>Transport, freight, inland waterways, barge {GLO}| market for | Alloc Rec, U</td>
</tr>
</tbody>
</table>

\textit{ReAI process (system 2)}

In Figure 9, the first stage of the ReAI process is acidification. In this stage, sulfuric acid (96\%) is added to the sludge in order to dissolve \(\text{Al}^{3+}\) from it. The addition of steam, supplied via the heat exchanger, increases the movement speed of the molecules in the sludge which dissolve \(\text{Al}^{3+}\) faster. The steam generated for the ReAI process is the same as in the drying stage. More details regarding the steam generation are provided in the sludge drying section. Before the ultrafiltration, the pressure is increased and process water is added to the sludge in order to dissolve even more \(\text{Al}^{3+}\). When the sludge reaches the ultrafiltration the \(\text{Al}^{3+}\) are separated by a semipermeable membrane while water and sludge pass through it. The separated \(\text{Al}^{3+}\) is then recycled back to the chemical coagulation stage. A so-called clean-in-place (CIP) system is connected to the ultrafilter which cleans the process equipment without the need for disassembly (Sani-Matic, 2016). The consumables used for the cleaning are \(\text{NaOH}\) in water (50\%) and sulfuric acid (96\%) and the ReAI process reduces the amount of sludge by 60-70\% (Purac, 2016b). Lastly, the sludge leaving the ReAI process is from this point on called ReAI sludge.

\begin{figure}[h]
\centering
\includegraphics[width=\textwidth]{fig9.png}
\caption{Detailed flowchart of the ReAI process. Notice, transparent inputs are not considered in this study.}
\end{figure}

\(^{13}\) Aluminium sulfate, without water, in 4.33\% aluminium solution state \{GLO\}\| market for \| Alloc Rec, U. the means of transport and the distance in this dataset was modeled based on surveys and statistics from the USA.

\(^{14}\) European emission standard
Data related to the ReAl process is confidential and can therefore not be published in this thesis. However, the data was entered into the SimaPro model. The datasets that were chosen to represent the inputs are shown in Table 6.

Table 6 Inputs to the ReAl process. The quantity of each input is not published in this thesis since it is confidential.

<table>
<thead>
<tr>
<th>Input</th>
<th>Dataset in Ecoinvent 3</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sulfuric acid (96%) Steam</td>
<td>Sulfuric acid¹⁵</td>
</tr>
<tr>
<td>CIP, Sulfuric acid wash (96%)</td>
<td>Steam¹⁶</td>
</tr>
<tr>
<td>CIP, Sodium hydroxide base wash (50%)</td>
<td>Sodium hydroxide, without water, in 50% solution state (RER)</td>
</tr>
<tr>
<td>Process water</td>
<td>Process water, ion exchange, production mix, at plant, from groundwater RER U</td>
</tr>
<tr>
<td>Electricity</td>
<td>Electricity, medium voltage (SE)</td>
</tr>
</tbody>
</table>

Dewatering (system 1 and system 2)

In system 1, the chemical sludge goes directly to the dewatering process after the chemical coagulation, while in system 2, the sludge goes through the ReAl process and then into the dewatering process. The purpose of dewatering sludge is to decrease the water content which facilitates the incineration process in many ways, not to say the least, it increases the energy efficiency. Dewatering also reduces the sludge volume which results in less hauling costs (Tramfloc, 2016). However, in this particular study the sludge is incinerated at the pulp mill, thus hauling costs are not important.

The dewatering of chemical sludge in system 1 is evaluated for both a decanter centrifuge and a filter press, achieving 22% DM and 30% DM respectively. While the dewatering of the ReAl sludge is only assessed for a filter press which also achieves 30% DM (see Figure 5). In both systems, a polymer is added to agglomerate sludge particles into larger flocs which increase the sludge thickness and ultimately improves the dewatering process (SNF, 2015). The polymer consumption to dewater the chemical sludge is assumed to be 78.6 tonnes/year and 38.6 tonnes/year of the polymer is needed for the ReAl sludge. The electricity demand from the decanter centrifuge is assumed to be 0.45 GWh/year and for the filter press 1.18 GWh/year of electricity is required (ÅF, 2017). The ReAl sludge is dewatered to 30% DM using the filter press and requires only 0.27 GWh/year. Dewatering sludge with a decanter centrifuge or a filter press omits according to Monteith and Bell (1998) very low amounts of volatile organic compounds (VOCs), thus these emissions have been neglected. All the inputs in the dewatering process are shown in Table 7.

¹⁵ See footnote 12
¹⁶ A dataset was created in SimaPro which represents the incineration of bark to generate steam for the ReAl-process and the sludge drying. All inputs and outputs in the dataset is described in the drying stage.
¹⁷ See footnote 12
Table 7 Inputs included in the dewatering process for the following systems: system 1a (decanter centrifuged chemical sludge), system 1b (filter pressed chemical sludge), and system 2 (filter pressed ReAl sludge). Notice, there is only one dewatering option for system 2.

<table>
<thead>
<tr>
<th>Input</th>
<th>System 1a</th>
<th>System 1b</th>
<th>System 2</th>
<th>Unit</th>
<th>Dataset in Ecoinvent 3</th>
</tr>
</thead>
<tbody>
<tr>
<td>Polymer</td>
<td>78.6</td>
<td>78.6</td>
<td>38.6</td>
<td>tonne/year</td>
<td>Polycrylamide [GLO]</td>
</tr>
<tr>
<td>Electricity</td>
<td>0.45</td>
<td>1.18</td>
<td>0.27</td>
<td>GWh/year</td>
<td>Electricity, medium voltage [SE]</td>
</tr>
</tbody>
</table>

Drying (system 1 and system 2b)

In the drying stage, the sludge is dried with steam generated from a bark boiler located on site. The energy required to dry the sludge was obtained by multiplying the difference in sludge flow rate, before and after the drying, with the enthalpy of vaporization of water, which was assumed to be 2.52 MJ/kg at 30 °C (Engineeringtoolbox, 2016). The steam needed to dry the decanter centrifuged chemical sludge (system 1a) from 22% DM to 60% DM was estimated to 9.51 GWh/year while drying the filter pressed chemical sludge (system 1b) from 30% DM to 60% DM would require 5.52 GWh/year (see Table 8). As can be seen in Figure 5, the ReAl sludge has one option without a dryer and another option where it is dried from 30% DM to 60% DM. The amount of energy needed to vaporize the water in the latter option equals to 2.23 GWh/year. The electricity demand during sludge drying was neglected in all systems.

Table 8 The amount of steam required to dry the sludge in system 1a (decanter centrifuged chemical sludge), system 1b (filter pressed chemical sludge), and system 2b (filter pressed ReAl sludge). Notice, system 2a does not include a drying stage.

<table>
<thead>
<tr>
<th>Input</th>
<th>System 1a</th>
<th>System 1b</th>
<th>System 2b</th>
<th>Unit</th>
<th>Dataset in Ecoinvent 3</th>
</tr>
</thead>
<tbody>
<tr>
<td>Steam</td>
<td>9.51</td>
<td>5.52</td>
<td>2.23</td>
<td>GWh/year</td>
<td>Steam&lt;sup&gt;18&lt;/sup&gt;</td>
</tr>
</tbody>
</table>

During the sludge drying process, VOCs are omitted which have to be considered. VOC emissions from drying sewage sludge were analyzed by Gomez-Rico et al. (2008) and their findings showed that 545 to 591 mg VOCs per kg dry weight were omitted. The mean value of these results (568 mg) was used for this inventory analysis and the percentage of VOCs per kg dry weight of sludge is 0.0568%. The estimated VOC emissions for each sludge treatment system are shown in Table 9.

Table 9 Emissions from sludge drying from system 1a (decanter centrifuged chemical sludge), system 1b (filter pressed chemical sludge), and system 2b (filter pressed ReAl sludge). Notice, system 2a does not include a drying stage.

<table>
<thead>
<tr>
<th>Emission</th>
<th>System 1a</th>
<th>System 1b</th>
<th>System 2b</th>
<th>Unit</th>
<th>Dataset in Ecoinvent 3</th>
</tr>
</thead>
<tbody>
<tr>
<td>VOC</td>
<td>12.24</td>
<td>12.24</td>
<td>5</td>
<td>Kg/day</td>
<td>VOC, volatile organic compounds</td>
</tr>
</tbody>
</table>

Dataset for steam generation

A dataset was created in SimaPro in order to model the incineration of bark to generate steam for the ReAl process and the sludge drying. The dataset is a hypothetical representation of bark incineration because the data is based on literature and not on data from a specific bark boiler.

<sup>18</sup> A dataset was created in SimaPro which represents the incineration of bark to generate 1 kWh of steam. By multiplying the amount of steam required in each system with the steam dataset for 1 kWh will result in the environmental impact of generating steam.
The inputs and emissions surrounding the incineration of bark to generate 1 kWh of steam are described in this section. The data required to estimate the amount of bark needed to generate the steam is the heat value of bark, the moisture content of bark, and the efficiency of a bark boiler. According to Bioenergihandboken (1996), the effective heat value of bark is 7.3 MJ/kg on a wet basis (as received) with a moisture content of 55% and 3% ash content. The bark boiler is assumed to generate steam with an efficiency of 90% (Naturvårdsverket, 2005) and the amount of bark required for each drying process is displayed in Table 10.

Table 10 Bark needed to generate 1 kWh of steam.

<table>
<thead>
<tr>
<th>Input</th>
<th>Quantity</th>
<th>Unit</th>
<th>Dataset in Ecoinvent 3</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bark</td>
<td>0.548</td>
<td>Kg/kWh</td>
<td>Bark chips, wet, measured as dry mass (RER)</td>
</tr>
</tbody>
</table>

Residual products are formed during incineration which mainly consists of inert material from the incinerated fuel. The residual product is mostly divided into two categories, bottom ash and fly ash. The total amount of residual is largely dependent on the ash content of the fuel and the ash content, in turn, varies depending on the fuel type, type of incinerator (e.g. fluidized bed and moving grate incinerator), and to some extent also the operation phase. Dust removal equipment such as electron filter also affects the residual products. The typical ash distribution for a fluidized bed is approx. 10% bottom ash and 90% fly ash. (Hjalmarsson et al., 1999)

The calculation of ash distribution was made based on a fluidized bed because it is a more modern technology compared to for example a moving grate. Moreover, it is assumed that an electrostatic precipitator is installed which removes 99% of the particles (Nazarooft & Alverz-Cohen, 2000). The distribution of ash as bottom ash and fly ash was then estimated with the distribution fraction and an ESP. The bottom ash and 99% of the fly ash were assumed to be used in construction materials such as roads at landfills or as landfill cover material. Unfortunately, the practitioner of this study could not find a data set for bark ash in SimaPro. The bark ash was therefore instead modeled as a “final waste flow” with wood ash (see Table 11). It is important to point out that the final waste flow in SimaPro does only monitor the amount of ash and not its environmental impact (Pré, 2016a).

Table 11 Ash, NOX and SOX emissions from incinerating bark to generate 1 kWh of steam.

<table>
<thead>
<tr>
<th>Emission</th>
<th>Quantity</th>
<th>Unit</th>
<th>Dataset in Ecoinvent 3</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bottom ash to landfill</td>
<td>0.74</td>
<td>g/kWh</td>
<td>Final waste flow - Wood ash</td>
</tr>
<tr>
<td>Fly ash to landfill</td>
<td>6.6</td>
<td>g/kWh</td>
<td>Final waste flow - Wood ash</td>
</tr>
<tr>
<td>Fly ash to air</td>
<td>0.067</td>
<td>g/kWh</td>
<td>Final waste flow - Fly ash</td>
</tr>
<tr>
<td>NOX</td>
<td>24.7</td>
<td>mg/kWh</td>
<td>Emissions to air – Nitrogen oxides</td>
</tr>
<tr>
<td>SOX</td>
<td>8.3</td>
<td>mg/kWh</td>
<td>Emissions to air – Sulfur oxides</td>
</tr>
</tbody>
</table>

Incinerating bark also releases emissions such as nitrogen oxides (NOX) and sulfuric oxides (SOX). The practitioner of this report chose to estimate the NOX emissions from sludge incineration with the Swedish Environmental Protection Agency’s provisional emission condition for large incineration boilers (100-300MW). The NVVs guideline value for large combustion plants (LCP) is
80 mg/MJ\textsubscript{fuel} which is based on EUs BREF-LCP reference document for best available technology (Naturvårdsverket, 2016b). The NO\textsubscript{X} emissions per kWh of steam generated from incinerating bark (see Table 11) was calculated with 80 mg NO\textsubscript{X}/MJ\textsubscript{fuel} and the energy in the bark. The energy in bark was in turn obtained by taking the boiler efficiency (90\%) into account. The SO\textsubscript{X} emissions were estimated with the same approach. Boersman et al. (2008) found that the SO\textsubscript{X} emissions from incinerating biomass (particle board, wood chips, MDF, and bark) ranged from 10 to 44 mg/MJ\textsubscript{fuel}. Thus, the mean value (27 mg/MJ\textsubscript{fuel}) was used (see Table 11).

**Incineration (system 1 and system 2)**

The best way to handle the chemical sludge and the ReAl sludge is through co-incineration with bark. Co-incineration makes sense in many ways due to financial, legal and technical reasons. The legal reason is simply that of the less strict regulations described in subchapter 2.4.2. Incinerating only sludge would be economically unsustainable since it has a relatively low heat value and a high ash content which would result in a low or negative net energy generation. In technical terms, it is very hard to incinerate wet fuel such as sludge because it is sticky (Marklund, pers.comm., 6 June).

According to Christer Andersson (2017, pers.comm., 9 March), the rule of thumb, when co-incinerating sludge with a biofuel such as bark, is to have a sludge fraction of no more than 15 dry-weight\%. The lower the sludge fraction is, the easier it will be to mix the fuel properly and the heat value will be higher. With this in mind, the author of this report assumed a chemical sludge fraction of 10 dry-weight\% and the remaining 90 dry-weight\% is bark. The ReAl sludge will have the same fuel mix fraction. The only difference is that a small amount of sawdust, which most pulp mills have on site, will be mixed into the ReAl sludge before it is co-incinerated with the bark. The reason for this is to reduce sludge cakes and the stickiness of the otherwise wet fuel (Liu et al., 2017). However, the sawdust is not considered in the SimaPro model because of the small amount that is needed. The bark in the co-incineration is assumed to have the same properties as the bark for steam generation (see “dataset for steam generation”).

At a pulp mill, electricity is normally generated through incineration of bark. However, waste or a by-product such as the sludge can be used as a substitute to the bark in the co-incineration. This is known as expanding the system boundaries (system expansion). The emissions and resources associated with the substituted bark will be subtracted from another process in the system where bark is used (Pré, 2016a). The energy generated in the different co-incineration configurations, with the same boiler efficiency (90\%) as in the bark boiler are shown in Table 12.

**Table 12 Annual energy generation from sludge.**

<table>
<thead>
<tr>
<th>Input</th>
<th>System 1a</th>
<th>System 1b</th>
<th>System 2a</th>
<th>System 2b</th>
<th>Unit</th>
<th>Avoided product</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sludge</td>
<td>30.14</td>
<td>25</td>
<td>9.33</td>
<td>6.80</td>
<td>GWh/year</td>
<td>Bark</td>
</tr>
</tbody>
</table>

The total amount of ash from incinerating the chemical sludge (system 1a and system 1b) and the ReAl sludge (system 2a and system 2b) is shown in Table 13. The ash distribution of bottom ash and fly ash was based on a fluidized bed and an electrostatic precipitator with the same assumptions as for the steam generation with bark (described in “Dataset for steam generation”). The NO\textsubscript{X} and SO\textsubscript{X} emissions from incinerating sludge are estimated, again, with the assumptions described in “Dataset for steam generation”. The NO\textsubscript{X} and SO\textsubscript{X} emissions are also listed in Table 13.
Table 13 Ash, NO\textsubscript{X} and SO\textsubscript{X} emissions from sludge incineration.

<table>
<thead>
<tr>
<th>Output</th>
<th>System 1a</th>
<th>System 1b</th>
<th>System 2a</th>
<th>System 2b</th>
<th>Unit</th>
<th>Dataset in Ecoinvent 3</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Energy</strong></td>
<td>82576.8</td>
<td>25552.8</td>
<td>68493.6</td>
<td>18621.5</td>
<td>kWh/day</td>
<td>Steam (Avoided product)</td>
</tr>
<tr>
<td><strong>Bottom ash</strong></td>
<td>1.2384</td>
<td>1.0253</td>
<td>0.1321</td>
<td>0.1849</td>
<td>tonne/day</td>
<td>Sludge ash</td>
</tr>
<tr>
<td><strong>Fly ash to air</strong></td>
<td>0.111456</td>
<td>0.092277</td>
<td>0.011889</td>
<td>0.016641</td>
<td>tonne/day</td>
<td>Fly ash</td>
</tr>
<tr>
<td><strong>Fly ash to landfill</strong></td>
<td>11.034144</td>
<td>9.135423</td>
<td>1.177011</td>
<td>1.647459</td>
<td>tonne/day</td>
<td>Sludge ash</td>
</tr>
<tr>
<td><strong>NO\textsubscript{X}</strong></td>
<td>744.21</td>
<td>617.29</td>
<td>230.29</td>
<td>167.82</td>
<td>kg/year</td>
<td>Emissions to air – Nitrogen oxides</td>
</tr>
<tr>
<td><strong>SO\textsubscript{X}</strong></td>
<td>251.17</td>
<td>208.33</td>
<td>77.72</td>
<td>56.64</td>
<td>kg/year</td>
<td>Emissions to air – Sulfur oxides</td>
</tr>
</tbody>
</table>

4.4. Life cycle impact assessment

In the life cycle impact assessment, all the inputs and outputs described in the life cycle inventory are organized and assigned by the impact assessment method, ReCiPe (H) Midpoint, to the appropriate impact categories in SimaPro. Due to the many impact categories included in the chosen impact assessment method, only the most significant and relevant impact categories to this thesis were selected for further evaluation in consultation with ÅF and Purac (see Table 14). In the next subchapter the LCIA results for the chosen impact categories are presented.

Table 14 Chosen impact categories and their unit of measurement - with a short description and motivation.

<table>
<thead>
<tr>
<th>Impact category</th>
<th>Unit</th>
<th>Description and motivation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Climate Change</td>
<td>Kg CO\textsubscript{2} eq.</td>
<td>Our planet’s climate has the perfect temperature to inhabit life due to the existent of greenhouse gases such as CO\textsubscript{2}, water vapor and methane, in the atmosphere. However, human’s enormous use of fossil fuels has increased the GHG emissions so much that the planet’s temperature is increasing in a way that can have severe consequences to humankind and ecosystems around the world (IDB, 2015). The environmental mechanisms used by ReCiPe to model climate change are radiative forcing, temperature effects, damage to human health, and damage to ecosystem diversity (ReCiPe, 2008). Many of the raw materials and process involved in the studied systems are emitting greenhouse gases. Climate change was also chosen because of its global scale and the fact that it is one of the most urgent threats facing humanity.</td>
</tr>
<tr>
<td>Category</td>
<td>Unit</td>
<td>Description</td>
</tr>
<tr>
<td>-------------------------------</td>
<td>---------------</td>
<td>------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------</td>
</tr>
<tr>
<td>Terrestrial Acidification</td>
<td>Kg SO₂ eq.</td>
<td>The release of sulfates, nitrates, and phosphates to the atmosphere causes acidification to the soil when it rains. Most plant species will be harmed if the soil acidity changes enough – this is called acidification. ReCiPe models acidification with “fate factors” which takes the environmental persistence (fate) of an acidifying substance into account, and “effect factors” which considers an acidifying substance damage to ecosystems (ReCiPe, 2008). Terrestrial Acidification was selected due to the use of aluminium sulfate and sulfuric acid and also because the sewage from the pulp mill contains considerable amounts of sulfates, nitrates, and phosphates.</td>
</tr>
<tr>
<td>Freshwater Eutrophication</td>
<td>Kg P</td>
<td>Aquatic eutrophication is the enrichment of phosphorus and nitrogen nutrients to a water body. The limiting factor for freshwater eutrophication is typically P. The growth of phytoplankton’s are dependent on the availability of N and P. Eutrophication can lead to severe local ecological effects e.g. oxygen depletion in water bodies. ReCiPe models freshwater eutrophication with the characterization factor for the environmental persistence (fate) in regards to P emissions to freshwater (ReCiPe, 2008). Freshwater eutrophication was selected due to the use of aluminium sulfate and sulfuric acid and also because the sewage from the pulp mill contains considerable amounts of sulfates, nitrates, and phosphates.</td>
</tr>
<tr>
<td>Marine Eutrophication</td>
<td>Kg N</td>
<td>Aquatic eutrophication is the enrichment of phosphorus and nitrogen nutrients to a water body. The limiting factor for marine eutrophication is typically N. The growth of phytoplankton’s are dependent on the availability of N and P. Eutrophication can lead to severe local ecological effects e.g. oxygen depletion in water bodies. ReCiPe models marine eutrophication with the characterization factor for the environmental persistence (fate) in regards to N emissions to marine water (ReCiPe, 2008). Marine eutrophication was selected due to the use of aluminium sulfate and sulfuric acid and also because the sewage from the pulp mill contains considerable amounts of sulfates, nitrates, and phosphates.</td>
</tr>
<tr>
<td>Particulate Matter Formation</td>
<td>Kg PM₁₀ eq.</td>
<td>Particulate matter (PM) is a mixture containing very small particles and liquid droplets which mostly form in the atmosphere due to chemical reactions between pollutants.</td>
</tr>
</tbody>
</table>


PM pollution can have serious negative human health effects on heart and lungs when they are inhaled (EPA, 2017). ReCiPe models particulate matter formation with the characterization factor of PM$_{10}$ intake fraction (ReCiPe, 2008).

Particulate matter formation was chosen because the studied systems include incineration processes and the extraction of raw material such as sulfur.

<p>| | |</p>
<table>
<thead>
<tr>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Freshwater Ecotoxicity</strong></td>
<td>Kg 14-DCB$^{19}$</td>
</tr>
<tr>
<td></td>
<td>Freshwater ecotoxicity describes the impacts from toxic substances released on freshwater ecosystems (Pré, 2016b). ReCiPe calculates ecotoxicity from environmental persistence (fate), exposure and effects of toxic substances (ReCiPe, 2008). Freshwater ecotoxicity was selected due to the use of aluminium sulfate and the sulfuric acid in the sludge treatment systems.</td>
</tr>
</tbody>
</table>

<p>| | |</p>
<table>
<thead>
<tr>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Marine ecotoxicity</strong></td>
<td>Kg 14-DCB$^{20}$</td>
</tr>
<tr>
<td></td>
<td>Marine ecotoxicity describes the impacts from toxic substances on marine ecosystems (Pré, 2016b). ReCiPe calculates ecotoxicity from environmental persistence (fate), exposure and effects of toxic substances (ReCiPe, 2008). Freshwater ecotoxicity was selected due to the use of aluminium sulfate and the sulfuric acid in the sludge treatment systems.</td>
</tr>
</tbody>
</table>

### 4.4.1. LCIA Results

This subchapter presents the LCIA results which describe the relative contribution of inputs and outputs to an impact category.

The LCIA results are first presented as a comparison between the four sludge treatment systems in which a holistic view of the results is established. The results of the four systems are displayed in the figures, while in written form, the results refer to system 1 and system 2 as the average of (system 1a and system 1b) and (system 2a and system 2b) respectively. In this way it is easier to communicate the results in written form and still maintain great accuracy – the error which occurs when taking the average instead of the specific aggregated results for each system is within 0.2% - 1.5%. The specific aggregated results for each system and impact category can be found in Table 17 Specific aggregated results for the sludge treatment system comparison. Table 17 (see Appendix).

---

$^{19}$ 14-DCB: 1,4 dichlorobenzene to freshwater  
$^{20}$ 14-DCB: 1,4 dichlorobenzene to marine water
The second part of the LCIA results displays the contribution from various inputs to each impact category in system 1 and system 2. All LCIA results presented exclude long-term emissions in accordance with the system boundaries.

**Sludge treatment system comparison**

The characterized results shown in Figure 10 describe the relative difference among the compared systems for each chosen impact category. System 2 which includes the ReAl process performs better than system 1 in every impact category. Moreover, system 1a (dewatering with a decanter centrifuge) and system 2a (excluding drying) have a lower impact on all impact categories in relation to their system counterpart, system 1b (dewatering with a filter press) and system 2b (includes drying) respectively. In system 1a, the decanter centrifuge requires less than half of the electricity demand of the filter press (see Table 7) and the net energy output (steam generated from co-incineration minus steam required in the drying stage) from system 1a is about 6% higher than in system 1b. Similarly, in system 2a the net energy output is about 53% greater than in system 2b (see Table 8 and Table 13). However, the environmental performance gain from selecting the best dewatering and drying equipment is limited and considered within the margin of error.

![Figure 10 Characterization results of the sludge treatment systems. Long-term emissions have been excluded.](image)

**Sludge treatment system 1**

The characterization results of system 1a and system 1b are almost identical and thus system 1 is represented by system 1a (see Figure 11). Figure 11 shows that the aluminium sulfate represents approx. 87% - 95% of the total potential impact across all impact categories except for marine eutrophication. The impact on marine eutrophication is divided in the following order: polyacrylamide i.e. polymer (38%), dewatering (28%), aluminium sulfate (33%), and electricity (1%).
The SO\textsubscript{X} and NO\textsubscript{X} emissions that arise from incinerating the sludge and the bark contributed only with 0.11-0.96% across three impact categories (terrestrial acidification, marine eutrophication and particulate matter formation). The specific emission contributions for each sludge treatment system can be found in the Appendix (see Table 18).

**Sludge treatment system 2**

The characterized results of system 2 are represented by system 2a (see Figure 12) for the same reasons as for why system 1a is representing system 1. Figure 12 shows that the ReAl process is the single largest contributing factor on all impact categories except for marine eutrophication. The ReAl process contributes between 47% - 79% to the total potential environmental impact distributed among the categories. The second largest contributing factor to all impact categories, except marine eutrophication, is aluminium sulfate. The two largest impacts on marine eutrophication are the polymer (49%) and the ReAl process (39%).

The characterization results of the ReAl process are shown in Figure 13 since it is of particular interest in this study to know what inputs of the ReAl process contribute the most to each impact category. The sulfuric acid which is added to the sludge to dissolve and separate the Al\textsuperscript{3+} from the sludge is by far the largest contributing input on all impact categories, ranging from approx. 70% -
96%. The second largest contributing input into the ReAl process is electricity. Sulfuric acid and sodium hydroxide which is used for cleaning the ultrafilter has almost a neglectable contribution to all impact categories. Sodium hydroxide contributes with a few percent to climate change and freshwater eutrophication.

Figure 13 Characterization results of the ReAl process. Long-term emissions have been excluded.

4.5. Life cycle interpretation
In the last phase of the LCA, the inventory data and the LCIA results are interpreted. However, the interpretation shall not only relay on the LCIA results but also on additional information that will complement the fundamental flaws of LCIA. Thus, significant issues were identified and a sensitivity analysis was performed. The data quality was considered in terms of consistency. Lastly, assumptions and limitations associated with the study were assessed. All these steps were taken to evaluate the reliability of the LCIA results.

4.5.1. Identification of significant life cycle inventory and LCIA contributions
The LCIA results of system 1 showed that aluminium sulfate has the largest impact on every impact category except for marine eutrophication in system 1, while in system 2, both aluminium sulfate and sulfuric acid have the largest impacts. However, since approx. 50 weight-% of aluminium sulfate is produced with sulfuric acid (see Table 4), which can be interpreted that sulfuric acid is actually the most important input in this study. The environmental impact from the inputs which are required in the aluminium sulfate production is shown in Figure 14. The impact distribution of the inputs on climate change is somewhat comparable to the results in INCOPA’s LCA of aluminium sulfate (see subchapter 2.5.1). The results from INCOPA state that aluminium hydroxide contributes with 70% of the impact on climate change, while truck, sulfuric acid and electricity contribute with 12%, 9% and 7% respectively.
The results of the seven impact categories which were chosen to be included in the LCIA are now in the following subchapters interpreted one by one.

**Climate change**

The reason why system 2 showed a lower potential impact on climate change compared to system 1 is that it requires far less aluminium sulfate which has a large carbon footprint during its raw material extraction and production (see subchapter 2.5.1). The ReAl process ability to recover aluminium sulfate results in approx. 72% less use of the coagulant (see Table 3). However, the ReAl process has a high electricity demand with a considerable impact on climate change (see Figure 13) due to the burning of fossil fuels. Nevertheless, system 2 was estimated to potentially have 40% less effect on climate change than system 1.

**Terrestrial acidification**

The characterization results in Figure 10 show that the impact on terrestrial acidification from system 2 is approx. 10% lower than from system 1. There is a distinct difference between the systems which is that the impact from system 1 is almost solely (97%) due to aluminium sulfate, whereas in system 2 approx. 65% of the impact constitutes from sulfuric acid which is added in the ReAl process and almost the rest of the impact is caused by aluminium sulfate. The impact of the two systems is comparable because the extra amount of sulfuric acid required in the ReAl process is similar to the larger use of aluminium sulfate in system 1.

**Freshwater and marine eutrophication**

The characterization results in Figure 10 of freshwater eutrophication are very similar to the results of terrestrial acidification i.e. that system 2 potentially could perform 10% better than system 1. Furthermore, the impact of each input in both system 1 and system 2 are very similar. Hence, the freshwater eutrophication results are interpreted in the same way as for terrestrial acidification.

The characterization results in Figure 10 show that system 2 impacts marine eutrophication approx. 30% less than system 1. In system 1, the inputs and their impact contribution are shown in Figure 11 as follows: polymer (38%), aluminium sulfate (32%), dewatering (29%), and electricity (1%). Since the dewatering stage has an important role in the impact on marine eutrophication, the characterization results of dewatering are displayed in Figure 15 – which shows that the
polymer use carries almost the entire impact (99%). Thus, the conclusion is that about two-thirds of the nitrogen emissions to the marine water body from system 1 comes from the polymer use, while the remaining one-third comes from the aluminium sulfate.

In system 2, the contributing inputs are the polymer (49%), the ReAl process (37%), and aluminium sulfate (13%), and electricity (1%). To fully understand why system 1 has a higher impact on marine eutrophication than system 2, it is helpful to investigate the data from the inventory analysis. The polymer used in the chemical coagulation is equal in both system 1 and system 2 (see Table 3), while the polymer used in the dewatering is about twice as large in system 1 than in system 2 (see Table 7). Moreover, the greater use of the polymer in system 1 is connected to the larger flow of sludge that needs to be dewatered. Thus, one can conclude that the amount of sulfuric acid required in the ReAl process, to recover the aluminium sulfate which consequently reduces the amount of sludge, has a lower impact on marine eutrophication than the extra use of the polymer in system 1.

![Figure 15](image)

**Figure 15** Characterization results of the dewatering. Long-term emissions have been excluded.

**Particulate matter formation**

The characterization results in Figure 10 show that system 2 potentially has 17% less particulate matter formation than system 1. By analyzing the inputs that affect particulate matter formation in system 1 and system 2, see Figure 11 and Figure 12 respectively, it is evident that the reduced use of aluminium sulfate in system 2 means less formation of particulate matter. Furthermore, Figure 13 shows that 95% of the ReAl process impact on particulate matter formation comes from the sulfuric acid.

**Freshwater and marine ecotoxicity**

The characterization results in Figure 10 show that system 1 contributes with approx. 38% more too freshwater ecotoxicity in relation to system 2. A comparison of the characterized results of system 1 (see Figure 11) and the characterized results of system 2 (see Figure 12) show that the total increase of sulfuric acid in system 2, due to the ReAl process, releases less toxic substances than the reduced emissions of toxic substances from recovering and lowering the use of aluminium sulfate. This same logic also explains the characterization results of marine ecotoxicity. However, the relative difference between the two systems in regards to marine ecotoxicity is approx. 31% instead of 38%.
4.5.2. Sensitivity analysis

A sensitivity analysis was performed to check the robustness of the LCA model in relation to key inputs and outputs that involve uncertainties. The LCIA results are in this subchapter called baseline results, while LCIA results which include the chosen sensitivity parameters are called sensitivity results.

**Aluminium sulfate**

The most significant input in system 1 was the coagulant (aluminium sulfate) according to the LCIA results. One uncertainty regarding aluminium sulfate is in which concentration it is delivered. In this study, aluminium sulfate was modeled as a solid which was assumed to contain 9.1% Al\(^{3+}\). However, the most common practice around the world is to produce and transport aluminium sulfate as a liquid based on e.g. 4.33% Al\(^{3+}\). It would therefore be interesting to see how much of an impact the choice of coagulant concentration has on the LCIA results.

Aluminium sulfate 4.33% Al\(^{3+}\) (liquid) contains twice as much water than aluminium sulfate 9.1% Al\(^{3+}\) (solid) which means that twice as much coagulant is needed to achieve the same coagulation rate. Moreover, the only difference between the two concentrations in terms of environmental impact is that aluminium sulfate shipped as a liquid would require twice as much transport compared to shipping solid aluminium sulfate.

The characterization results which include the sensitivity parameter are shown in Figure 16. As expected, the most noticeable difference between the baseline results and the sensitivity results are observed in system 1 since it requires much more coagulant than system 2. Furthermore, the most affected impact category was climate change which is related to the increased CO\(_2\)-emissions due to the extra transport needed of the liquid aluminium sulfate. The effect of the sensitivity parameter ranged from approx. 4% - 15% across all impact categories on system 1, while the effect on system 2 was only a few percent.

![Figure 16](image16.png)

*Figure 16 Characterization results of the baseline and the sensitivity analysis. In this sensitivity analysis aluminium sulfate 9.1% Al\(^{3+}\) is modeled instead of aluminium sulfate 4.33% Al\(^{3+}\). Long term-emissions are excluded.*

**Electricity mix**

Purac’s business is not only directed towards Sweden but also towards the Chinese market. With this in mind and the fact that China relies much more heavily on fossil fuels for electricity generation than in Sweden makes it relevant to perform a sensitivity analysis on both countries.
electricity mixes. In Figure 17, the baseline characterization results of system 1 and system 2 included Sweden’s electricity mix which was then compared to the sensitivity analysis with China’s electricity mix. As expected, China’s electricity mix affected all impact categories for the worse. Climate change and particulate matter formation were particularly affected by this change. This was not surprising since the burning of fossil fuels e.g. coal and oil releases emissions such as CO₂, SO₂, and PM₁₀.

The interesting part with Figure 17 is the fact that the sensitivity results do not follow the same pattern which the baseline results have established regarding the relationship between system 1 and system 2. In the sensitivity results, system 2 has a greater effect on four impact categories (climate change, terrestrial acidification, freshwater eutrophication, and particulate matter formation) compared to system 1, while the opposite is true for the baseline results. By looking back on the inventory data, one can see that system 1 has electricity as an input in two unit processes, while system 2 has three unit processes which require electricity. This third unit process is the ReAl process and because it has a very high electricity demand, it changes the dynamic between system 1 and system 2 when China’s electricity mix is considered instead of Sweden’s.

![Figure 17](image)

*Figure 17 Characterization results showing the difference between Sweden’s and China’s electricity mixes. Long-term emissions have been excluded.*

The environmental impacts from both sensitivity parameters, aluminium sulfate with 4.33% Al³⁺ (liquid) and Chinese electricity mix, are also modeled together (see Figure 18). The overall conclusion is that the effect of changing electricity mix is much greater than the choice of coagulant concentration.
Figure 18 Characterization results showing baseline results and sensitivity results. The sensitivity results include Chinese electricity mix and the original aluminium sulfate dataset (4.33%) from a global market. Long-term emissions have been excluded.

The last sensitivity parameter was to include long-term emissions. The sensitivity results of including long-term emissions are displayed in Figure 19. Comparing Figure 18 and Figure 19, the most notable effect of long-term emissions is on freshwater eutrophication and freshwater ecotoxicity, while smaller effects can be seen on marine eutrophication and almost no effect can be identified on the remaining impact categories.

Figure 19 Characterization results for sensitivity analysis including Chinese electricity mix and aluminium sulfate (4.33%) from a global market. Including long-term emissions.

4.5.3. Qualitative assessment
Conducting an LCA involves many decisions regarding for example the type of data, system boundaries, methodological choice and other factors that can have a significant impact on the results. In this section, these kind of factors are discussed and assessed from a qualitative perspective. The underlying purpose of this subchapter is to increase the transparency and understanding of the inventory data and LCIA results.
Consistency check

The consistency of an LCA regarding its methodology, assumptions and data is a qualitative indicator of how comparable two or more studied options are in relation to one another. The consistency between sludge treatment system 1 and system 2 is maintained well because the assumptions that were made and the type of data that were used were based on similar terms for all the shared unit processes. However, the consistency between unit processes is lower. For example, the inputs related to the ReAl process are based on data from specific pilot tests, while e.g. emission data for bark and sludge incineration were based on literature. Moreover, some datasets were directly selected from Ecoinvent 3 while others were created from various sources.

The most influential decision on the geographical boundary was the fact that the sludge treatment systems were located in Sweden but no site-specific location was chosen. This means that all the transport data was based on market averages. Furthermore, it was also difficult to establish geographical data consistency because of the limitations of Ecoinvent 3. For the baseline analysis, Swedish electricity mix and a global polymer were used, while the remaining datasets were based on European markets. The inconsistency of using the Swedish electricity mix is accommodated because it is in line with the scope of the study. However, to increase data consistency, one should consider finding a European dataset for the polymer. As European datasets usually have a lower environmental impact than global datasets, it is reasonable to believe that the replacement of a global with a European produced polymer would decrease both systems impact, especially on marine eutrophication (see Figure 11 and Figure 12). The geographical area for each input in the baseline and sensitivity analysis are displayed in Table 15.

Table 15 Input data and their geographical area for the baseline analysis (see subchapter 4.3.2) and the sensitivity analysis (see subchapter 4.5.2).

<table>
<thead>
<tr>
<th>Input</th>
<th>Baseline analysis</th>
<th>Sensitivity analysis</th>
</tr>
</thead>
<tbody>
<tr>
<td>Electricity</td>
<td>Sweden</td>
<td>China</td>
</tr>
<tr>
<td>Aluminium sulfate</td>
<td>Europe</td>
<td>Europe</td>
</tr>
<tr>
<td>Sulfuric acid</td>
<td>Europe</td>
<td>Europe</td>
</tr>
<tr>
<td>Polymer</td>
<td>Global</td>
<td>Global</td>
</tr>
<tr>
<td>Process water</td>
<td>Europe</td>
<td>Europe</td>
</tr>
<tr>
<td>Sodium hydroxide</td>
<td>Europe</td>
<td>Europe</td>
</tr>
<tr>
<td>Bark</td>
<td>Europe</td>
<td>Europe</td>
</tr>
</tbody>
</table>

Methodological choice and uncertainties

One important cut-off criteria that must be stressed is the fact that the infrastructure of the sludge treatment systems was excluded (see subchapter 4.2.4). This decision could possibly have contributed from a few percent to a considerable part of the total environmental impact (see subchapter 2.5.2.). Thus, the environmental impact from the infrastructure should be included in a more extensive LCA to enable the most accurate results.

The scope of the study was to investigate the current environmental impacts of the two sludge treatment systems. With other words, both systems were assumed to be static which means that nothing changes over time. Although it is much easier to model static systems, it would have been more accurate to model them as dynamic systems since they have a lifetime of several decades. The possible error of assuming that nothing changes over time is of course dependent on the time frame, and the longer the timeframe is, the more uncertain the assumption becomes. A good
example of this would be how the production of electricity will undoubtedly become more environmentally friendly in the future. One could argue that system 2 which demands more electricity than system 1 would perform even better in the future when the electricity mix is produced by more renewable energy sources.

There are many ways to go about choosing the impact categories e.g. by using an environmental paradigm or based on policies. The impact categories from the ReCiPe impact assessment method (see Table 14) were selected in consultation with ÅF and Purac on the basis that they should be relevant to the studied systems. Thus, the seven impact categories that were chosen are considered to be directly linked to the inventory data. However, none of the selected impact categories covered any natural resource such as water, metal or fossil depletion which according the ILCD handbook is required in the ISO 14040 series (EC, 2010). While it is generally a good idea to focus on a few set of impact categories in order to make the LCIA results easier to interpret, it may also increase the risk of losing valuable results related to e.g. energy consumption.

The decision to exclude long-term emissions is also another uncertainty which will influence impact categories that are related to e.g. leachate from landfills. However, it is not obvious that one should include long-term emissions since the time frame is very long (defined as more than 100 years in ReCiPe). High uncertainties are involved in long-term emissions and it is very much a question about how these uncertainties are viewed.

Limitations and completeness check
The most inherent limitation with the life cycle interpretation was that the input data of the ReAl process was confidential and therefore not published in this thesis. The process data was incorporated into the model and the LCIA results but the interpretation and analysis of the ReAl process contribution to the overall impact was limited.

A noticeable limitation with the modeling in SimaPro was how the ash was handled from the co-incineration of bark and sludge. Both the fly ash and the bottom ash was modeled as a “final waste flow” which does not include any environmental impacts – it is only a way to monitor the amount of ash which is landfilled. The practitioner did not find a suitable dataset for using the ash in construction materials or in landfill cover materials. If the ash’s environmental impacts would have been accounted for it might have influenced the baseline results but especially the sensitivity analysis because it included long-term emissions. Many of the substances in the ash could leak from construction materials to the environment over a longer period of time.

A data gap related to the process flow chart in Figure 7 in the life cycle inventory analysis was the absence of heavy metal emissions from the incineration process. There was a lack of available data concerning the heavy metal content in the sludges. However, these emissions could have been accounted for similarly to the SO$_X$ and NO$_X$ emissions (see subchapter “dataset for steam generation”). Despite this, the inclusion of the heavy metal emissions would not affect the LCIA results nor the conclusion of this thesis. This statement can be explained as follows. The overwhelming majority of heavy metals end up in the fly ash during incineration (Rhodin, pers.comm., 16 March). Furthermore, it was assumed that an electrostatic precipitator would remove 99% of the fly ash which means that only 1% of the fly ash would be released to the air and the remaining 99% could be landfill if the heavy metals are stabilized. Thus, since this thesis was not able to account for any environmental impacts from landfiling, it means that the exclusion of heavy metal emissions did not influence the overall results.
5. Discussion

Life cycle assessment is a great tool to estimate a product or systems environmental impact. However, LCA has like any other scientific approach limitations which an LCA practitioner should be aware of during the evaluation and interpretation of the LCIA results. It is important to remember that LCA is only a tool which tries to model the actual environmental impact based on many different assumptions, impact assessment methods, system boundaries, data etc. (see subchapter 4.5.3). Thus, the results of an LCA model is heavily dependent on the accuracy of the data gathered and how well the tool is utilized. It is especially important to communicate the precision and reliability of the LCA results to the intended audience in order for them to make well-thought-out decisions. With that said, the accuracy of the LCIA results in this thesis should be viewed as a rough estimate of the environmental impact of the sludge treatment systems.

The result of similar comparative life cycle assessment studies can vary a lot due to the variability of access to data and the subjectivity of conducting an LCA. Even though there are comprehensive guidelines such as the ISO 14040 series on how to minimize the risk of conflicting results between studies, there is still enough freedom for individual LCA practitioners to make decisions that affect the outcome of a study.

As an LCA practitioner it is challenging to keep track of whether the extensive guidelines of the ISO 14040 series actually has been followed through. This raises the question whether this LCA is truly in line with the series? The most important aspects of a LCA has been conducted to the best of the practitioner’s knowledge and ability. The most noticeable deviation from the guidelines is probably the fact that none of the selected impact categories where directly related to resources such as energy. However, from the sensitivity analysis it is evident that the electricity mix and the ReAl process large electricity demand plays an important role in the overall environmental impact from sludge treatment system 2. That being said, it is important to consider resource depletion in a more comprehensive LCA.

Part of the purpose of this thesis was to clarify whether a sludge treatment system which includes the ReAl process would be more environmentally friendly than a conventional system. However, even if the LCIA results conclude this, there are still many uncertainties regarding the technical feasibility and regulatory compliance around chemical sludge treatment and its economic implications.

If ReAl sludge, where coagulation chemicals have been recovered, can be exempted from the waste incineration (2013:253) ordinance 7 § p.1 (see subchapter 2.4.2) and be classified as biofuel instead of waste, under the much less strict large combustion plants (2013:252) ordinance 3 § p.4 – than there will be an incentive for Swedish plant owners to invest in technologies such as the ReAl process. Furthermore, the plant owners will also have a better sludge which will reduce the damage done on the incinerator and at the same time generate more energy. With other words, there is an intrinsic value to include the ReAl sludge under less strict regulations, since it can pave the way for an increase of chemical coagulation in various industries which suits the agenda of the CAB and the NVV very well. From the author’s point of view, the hope is that this thesis could give these state authorities a decision basis on how to go forward with the legislations and regulations surrounding chemical sludge treatment.
6. Conclusions
The first research question of this study was aimed to determine, from a life cycle perspective, which of the two studied sludge treatment systems, a conventional system (system 1) and a system which include the ReAl process (system 2), is more environmentally friendly. Moreover, the second research question was designed to investigate which inputs in the systems are contributing the most to the environmental impact. The third and final research question was designed to explore which dewatering equipment (a decanter centrifuge or a filter press) would be more environmentally favorable in system 1, as well as, whether excluding or including drying of the ReAl sludge before incineration in system 2 would lower its environmental impact.

6.1. Research question 1
The main conclusion of this study is that system 2 with the ReAl process has a lower environmental impact on all the evaluated impact categories when the Swedish electricity mix is used or a similar mix with low CO₂-emissions. Thus, the ReAl process can be viewed as an environmentally sound and sustainable technology within the right circumstances. The characterized life cycle impact assessment results showed that system 2 could potentially reduce the impact on climate change and freshwater ecotoxicity by 40%. Furthermore, system 2 showed approx. 20% lower particulate matter formation, 25% lower freshwater eutrophication, and 10% lower terrestrial acidification and freshwater eutrophication than the conventional system.

6.2. Research question 2
Aluminium sulfate was the input that had by far the largest impact on system 1 across all impact categories (87% - 95%) aside from marine eutrophication. Similarly, the ReAl process was the dominating unit process in system 2 which contributed with 47% - 79% across all impact categories apart from marine eutrophication. The highest contributing input on marine eutrophication was the polymer with 49% of the impact. Moreover, sulfuric acid was the most important input in the ReAl process which varied between 70 - 96% across all impact categories. The remaining impact from the ReAl process was due to its electricity demand. Other inputs in the ReAl process had a neglectable contribution.

6.3. Research question 3
The Life cycle impact assessment results showed that the most environmentally friendly dewatering equipment for system 1 is to use a decanter centrifuge and the most environmentally friendly drying option for system 2 is to not include drying at all. However, the environmental performance gain from choosing the best dewatering and drying option in each sludge treatment system is very limited and considered within the margin of error. Hence, it is therefore recommended to consider their economic and technical factors before their environmental performance.

6.4. Sensitivity analysis
A sensitivity analysis was performed on some of the most important inputs of the sludge treatment systems to check the robustness of the LCA model. The coagulant, aluminium sulfate, was in the sensitivity analysis modeled as a liquid (4.33% Al³⁺) instead of a solid (9.1% Al³⁺) which doubled the need for transportation. This further strengthened the baseline result which was that system 2 is more environmentally friendly than system 1 by increasing the relative difference between the two systems environmental impact.
The second sensitivity analysis evaluated the impact from changing the Swedish electricity mix used in the unit processes to a Chinese electricity mix. China's electricity mix had a substantial impact on the baseline results. In fact, four out of the seven impact categories, including climate change, terrestrial acidification, freshwater eutrophication and particulate matter, completely changed in favor of system 1. It is therefore not possible to conclude which sludge treatment system is more environmentally friendly because some of the impact categories are more favorable for system 1 while others are more favorable for system 2.

The evaluation of both sensitivity analyses simultaneously showed that the change in electricity mix had a much larger impact than the choice of aluminium sulfate concentration. The bottom line is that a “clean” electricity mix which is being used in Sweden is essential for system 2 and the ReAl process overall impact on the environment.
7. Recommendations

- The sensitivity analysis showed that the electricity mix played a key role in system 2’s (ReAl process) overall environmental performance. Thus, it is important to produce as much clean energy as possible on site or to buy renewable energy from the grid.

- Purchase, if possible, locally produced aluminium sulfate, sulfuric acid and polymers that are produced in the most environmentally friendly way. Work also on reducing the use of these inputs.

- Even though, the goal of the study was to evaluate the environmental impacts of the sludge treatment systems, it is always important to consider the economic aspects of any project. It is especially relevant to assess the economics of the ReAl process because it is a long-term investment were large capital is needed. Furthermore, in the sensitivity analysis the change from a Swedish electricity mix to a chines electricity mix resulted in that it was not possible to declare which sludge treatment system that is more environmentally friendly. Hence, conducting a life cycle cost analysis (LCCA) of the studied systems could revel at least if the ReAl process is a good economic investment in countries with an electricity mix heavily dependent on fossil fuels.

- Conducting a LCCA and a technical analysis specifically on the dewatering and drying equipment is important in order to determine which equipment is most suited for the sludge treatment systems.
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Appendix

This appendix lists four tables related to the life cycle inventory and life cycle impact assessment.

*Table 16 Assumption made in the life cycle inventory analysis.*

<table>
<thead>
<tr>
<th>Unit Process</th>
<th>Description</th>
<th>Assumption</th>
<th>Unit</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Pulp Mill</strong></td>
<td>Pulp production</td>
<td>800 000</td>
<td>tonnes/years</td>
<td>(ÅF, 2017)</td>
</tr>
<tr>
<td></td>
<td>SCOD in the sewage from the pulp mill</td>
<td>30</td>
<td>kg/ADT pulp</td>
<td>(ÅF, 2017)</td>
</tr>
<tr>
<td><strong>Biological Wastewater</strong></td>
<td>SCOD reduction in the activated sludge process</td>
<td>70</td>
<td>%</td>
<td>(ÅF, 2017)</td>
</tr>
<tr>
<td><strong>Treatment</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Chemical Coagulation</strong></td>
<td>SCOD reduction</td>
<td>60</td>
<td>%</td>
<td>(ÅF, 2017)</td>
</tr>
<tr>
<td></td>
<td>Sludge retention in DAF</td>
<td>95</td>
<td>%</td>
<td>(ÅF, 2017)</td>
</tr>
<tr>
<td><strong>Dissolved Air Flotation</strong></td>
<td>Dispersion flow</td>
<td>30</td>
<td>%</td>
<td>(ÅF, 2017)</td>
</tr>
<tr>
<td><strong>ReAl process</strong></td>
<td>Sludge reduction</td>
<td>70</td>
<td>%</td>
<td>(ÅF, 2017)</td>
</tr>
<tr>
<td><strong>Dewatering</strong></td>
<td>Sludge DM after the decanter centrifuge</td>
<td>22</td>
<td>%</td>
<td>(ÅF, 2017)</td>
</tr>
<tr>
<td></td>
<td>Sludge DM after the filter press</td>
<td>30</td>
<td>%</td>
<td>(ÅF, 2017)</td>
</tr>
<tr>
<td><strong>Drying</strong></td>
<td>Enthalpy of vaporization of water at 30 °C</td>
<td>2.52</td>
<td>MJ/kg</td>
<td>(Engineeringtoolbox, 2016)</td>
</tr>
<tr>
<td></td>
<td>Electricity demand</td>
<td>Neglected</td>
<td>kWh/year</td>
<td>(ÅF, 2017)</td>
</tr>
<tr>
<td></td>
<td>Effective heat value of bark with 55% moisture</td>
<td>7.3</td>
<td>MJ/kg on wet</td>
<td>(Bioenergihandboken, 1996)</td>
</tr>
<tr>
<td></td>
<td>content</td>
<td></td>
<td>basis</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Bark ash content of DM</td>
<td>3</td>
<td>%</td>
<td>(Bioenergihandboken, 1996)</td>
</tr>
<tr>
<td></td>
<td>Boiler efficiency</td>
<td>90</td>
<td>%</td>
<td>(Naturvårdsverket, 2005)</td>
</tr>
<tr>
<td></td>
<td>Sludge DM after drying</td>
<td>60</td>
<td>%</td>
<td>(ÅF, 2017)</td>
</tr>
<tr>
<td></td>
<td>VOC emission</td>
<td>0.0568</td>
<td>% per kg dry</td>
<td>(Gomez-Rico et al., 2008)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>weight of sludge</td>
<td></td>
</tr>
<tr>
<td><strong>Incineration</strong></td>
<td>NO\textsubscript{X} emission</td>
<td>80</td>
<td>mg/MJ\textsubscript{fuel}</td>
<td>(Naturvårdsverket, 2016b)</td>
</tr>
<tr>
<td></td>
<td>SO\textsubscript{X} emission</td>
<td>27</td>
<td>mg/MJ\textsubscript{fuel}</td>
<td>(Boersman et al., 2008)</td>
</tr>
<tr>
<td></td>
<td>Boiler efficiency</td>
<td>90</td>
<td>%</td>
<td>(Naturvårdsverket, 2005)</td>
</tr>
<tr>
<td></td>
<td>Bark co-incineration fraction</td>
<td>90</td>
<td>%</td>
<td>(Andersson, 2017)</td>
</tr>
<tr>
<td></td>
<td>Sludge co-incineration fraction</td>
<td>10</td>
<td>%</td>
<td>(Andersson, 2017)</td>
</tr>
<tr>
<td></td>
<td>Fluidized bed bottom ash</td>
<td>10</td>
<td>%</td>
<td>(Hjalmarsson et al., 1999)</td>
</tr>
<tr>
<td></td>
<td>Fluidized bed fly ash</td>
<td>90</td>
<td>%</td>
<td>(Hjalmarsson et al., 1999)</td>
</tr>
</tbody>
</table>
Electrostatic precipitation particle removal efficiency 99 % (Nazaroofig & Alverz-Cohen, 2000)

<table>
<thead>
<tr>
<th>Unit Process</th>
<th>Description</th>
<th>Assumption</th>
<th>Unit</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Electrostatic precipitation particle removal efficiency</td>
<td>99%</td>
<td>%</td>
<td>(Nazaroofig &amp; Alverz-Cohen, 2000)</td>
<td></td>
</tr>
</tbody>
</table>

Table 17 Specific aggregated results for the sludge treatment system comparison.

<table>
<thead>
<tr>
<th>Impact category</th>
<th>Unit</th>
<th>System 1a</th>
<th>System 1b</th>
<th>System 2a</th>
<th>System 2b</th>
</tr>
</thead>
<tbody>
<tr>
<td>Climate change</td>
<td>kg CO2 eq</td>
<td>4517401</td>
<td>4645170</td>
<td>2677484</td>
<td>2805807</td>
</tr>
<tr>
<td>Terrestrial acidification</td>
<td>kg SO2 eq</td>
<td>65564.75</td>
<td>66241.16</td>
<td>59232.05</td>
<td>60070.85</td>
</tr>
<tr>
<td>Freshwater eutrophication</td>
<td>kg P eq</td>
<td>263.233</td>
<td>267.6907</td>
<td>236.9827</td>
<td>239.2666</td>
</tr>
<tr>
<td>Marine eutrophication</td>
<td>kg N eq</td>
<td>2221.245</td>
<td>2269.063</td>
<td>1669.833</td>
<td>1724.446</td>
</tr>
<tr>
<td>Particulate matter formation</td>
<td>kg PM10 eq</td>
<td>19055.26</td>
<td>19386.04</td>
<td>15493.73</td>
<td>15939.49</td>
</tr>
<tr>
<td>Freshwater ecotoxicity</td>
<td>kg 1,4-DB eq</td>
<td>4606.395</td>
<td>4681.803</td>
<td>2783.132</td>
<td>2875.429</td>
</tr>
<tr>
<td>Marine ecotoxicity</td>
<td>kg 1,4-DB eq</td>
<td>8762.222</td>
<td>8994.611</td>
<td>5955.733</td>
<td>6204.458</td>
</tr>
</tbody>
</table>

Table 18 The SOX and NOX emission contribution of incinerating sludge and bark on three relevant impact categories.

<table>
<thead>
<tr>
<th>Impact category</th>
<th>Unit</th>
<th>System 1a</th>
<th>System 1b</th>
<th>System 2a</th>
<th>System 2b</th>
</tr>
</thead>
<tbody>
<tr>
<td>Terrestrial acidification</td>
<td>kg SO2 eq</td>
<td>0.96%</td>
<td>1.24%</td>
<td>0.11%</td>
<td>0.17%</td>
</tr>
<tr>
<td>Marine eutrophication</td>
<td>kg N eq</td>
<td>0.44%</td>
<td>0.96%</td>
<td>0.11%</td>
<td>0.18%</td>
</tr>
<tr>
<td>Particulate matter formation</td>
<td>kg PM10 eq</td>
<td>0.78%</td>
<td>1.16%</td>
<td>0.14%</td>
<td>0.18%</td>
</tr>
</tbody>
</table>