Potential for mitigating GHG emissions at a Swedish wastewater treatment plant – a life cycle approach

Nina Aldén
Preface

This Master’s thesis is Nina Aldén’s degree project in Geography at the Department of Physical Geography, Stockholm University. The Master’s thesis comprises 30 credits (one term of full-time studies).

Supervisor has been Salim Belyazid at the Department of Physical Geography, Stockholm University and Frida Österdahl at Roslagsvatten AB. Examiner has been Mattias Winterdahl at the Department of Physical Geography, Stockholm University.

The author is responsible for the contents of this thesis.

Stockholm, 1 September 2020

Björn Gunnarson
Vice Director of studies
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Abstract

To meet the national and international climate goals every potential GHG mitigating effort needs to be addressed. The aim of this thesis is to investigate if the wastewater treatment plant (WWTP), Ekebyhov, can reduce its GHG emissions by making changes in the treatment process. The main GHGs emitted from WWT are N₂O, CH₄ and CO₂. To begin with, Ekebyhov’s current carbon footprint was calculated in a base line scenario, using a calculation tool (ECT). The results showed that the total footprint amounted to 522 tons CO₂eq per year, with the majority of the emissions (83 %) from the activated sludge process. Five GHG-mitigating measures were identified and potential GHG emission reduction (PGER) was calculated from 1) optimized WWT, 2) urea treated sludge, 3) change of chemicals, 4) green transports and 5) added anaerobic digestion (AD) process. The largest PGER came from added AD, followed by optimized WWT. Finally, the PGER for all measures was calculated and resulted in net negative emissions of -95 tons CO₂eq per year. The thesis shows that it is possible to reduce the carbon footprint of Ekebyhov WWTP, even to a net negative result. It is, however important to address other impact categories in a full LCA to be able to make fully informed decisions.

Keywords
GHG emissions, Wastewater treatment, Sewage sludge, Life cycle analysis, Mitigation, Carbon footprint, Global warming potential
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Abbreviations

ASP – Activated sludge process
BOD$_7$ – Biochemical oxygen demand. A measurement of organic material as the amount of oxygen required to biologically break down the material
C – Carbon
CDR – Carbon dioxide removal
CH$_4$ – Methane
CO$_2$ – Carbon dioxide
CO$_2$eq – Carbon dioxide equivalents
COD – Chemical oxygen demand. A measurement of organic material as the amount of oxygen required to chemically break down the material
DS – Dry substance
ECT – Excel calculating tool
EF – Emission factor
FU – Functional unit
GHG – Greenhouse gas
GWP – Global warming potential
LCA – Life cycle assessment
N – Nitrogen
NH$_4^+$ – Ammonium
N$_2$O – Nitrous oxide
NET – Negative emissions technique
P – Phosphorus
PE – Population equivalent, the average amount of BOD$_7$ per person and day
PGER – Potential GHG emission reduction
Ppm – Parts per million
SS – Sewage sludge
TS – Total Solids
VS – Volatile Solids
WW – Wastewater
WWT – wastewater treatment
WWTP – Wastewater treatment plant
Acknowledgements

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Nina Aldén
1. Introduction

The possible effects of global warming are a great threat to our civilization, including rising sea levels, extreme weather, loss of species, food and water shortages, human health impacts etc. (IPCC, 2018; Steffen, et al., 2018). In the Paris agreement from COP21 in Paris 2015, the world’s leaders agreed to a common cause to undertake ambitious mitigating efforts to combat climate change and limit global warming to a temperature rise well below 2°C, aiming for 1,5°C (UNFCCC, 2015). Despite this, it is estimated that human activities have already caused a global temperature rise of 1°C, and if the greenhouse gas (GHG) emissions continue in the current rate, global warming will reach 1,5°C between 2030 and 2052 (IPCC, 2018).

Models from the Intergovernmental Panel for Climate Change (IPCC) estimate that, to stay below 1,5°C, the anthropogenic CO₂ emissions need to decline by 45% from 2010 levels by 2030 and reach net zero by 2050. Simultaneously, other GHGs such as methane and nitrous oxide, should be reduced as much as possible, more than 35%. To reduce the emissions in time, the actions need to be ambitious and include every level of every society (IPCC, 2018).

The treatment of waste and wastewater has been identified as key sectors amongst the anthropogenic sources of GHG emissions since it, in 2010, represented 3% of total global GHG emissions and is correlated to economic and population growth (Blanco, et al., 2014; Fang, et al., 2019). In 2010, wastewater treatment represented 54% of the GHGs emitted from the waste sector (Blanco, et al., 2014). The main GHGs emitted from wastewater treatment (WWT) are methane (CH₄) and nitrous oxide (N₂O) (Jönsson, et al., 2015; Koh & Shaw, 2015), gases that have, respectively, 34 and 298 times stronger radiative forcing effect than carbon dioxide (CO₂) (Myhre, et al., 2013). This entails that reducing the emissions of CH₄ and N₂O is of vital importance and makes WWT plants (WWTP) important actors in the effort to mitigate GHG-emissions and combat climate change.

In 2017 a new climate framework was adopted by the Swedish parliament to meet the targets set by the Paris agreement (Swedish Government, 2017). The framework consists of three parts: 1) A legislative “Climate Act”, ensuring the continued work towards reduced GHG emissions; 2) ambitious climate goals, the leading one being “net zero emissions by 2045”; and 3) A climate policy council, instructed to review decisions and make sure they are leading in the right direction.

To reach the goals set by both the Paris agreement and the Swedish parliament, the per capita carbon footprint needs to be as far below 2 tons as possible by 2050 (Swedish EPA, 2020). The Swedish GHG emissions per capita and year varies today between 5 and 9 tons depending on how it is measured. The last few years the average consumption based GHG emissions has been stable around 9 tons per person and year (Swedish EPA, 2020) and territorial emissions were 5,11 tons per capita 2018 (Swedish EPA, 2019), out of which 21 kg originated from the municipal WWT (SCB, 2020).

In 2018 214,7 kilo tons CO₂eq were emitted from the Swedish WWT sector (SCB, 2020). Swedish WWTP need to do better, but can they do better?

2. Aim

The aim of this thesis is to answer the following question:

Can the wastewater treatment plant, Ekebyhov, reduce its greenhouse gas emissions by making changes in the treatment process?
3. Background

The core function of a WWTP is to remove nutrients (P and N), organic compounds (BOD\(_7\)) and other substances that may harm the recipient ecosystem, e.g. by eutrophication and toxification. In that sense WWT is positive for the environment. However, the various processes involved in the treatment are sources of the GHGs carbon dioxide (CO\(_2\)), methane (CH\(_4\)) and nitrous oxide (N\(_2\)O) that are harmful for the environment and lead to global warming (Jönsson et al., 2015). Depending on which techniques are used throughout the WWT, phosphorus (P), nitrogen (N) and carbon (C) originating from the wastewater and sludge can become either valuable resources or be emitted as environmentally negative substances (Heimersson et al., 2016).

This chapter will explain the fundamentals behind WWT and the connection to global warming.

3.1 Wastewater treatment

3.1.1 Municipal wastewater treatment

Before the 1960s most of the municipal wastewater was totally uncleaned, letting nutrients, organic compounds and toxins out into rivers and lakes (Swedish EPA, 2018). This resulted in eutrophication, death of fish, loss of bathing shores and spread of diseases. Around 1960, water related environmental and public health issues came high up on the agenda, and finally lead to a big governmental investment in municipal WWTP. At first, the WWT focused on removing particles, organic compounds (BOD\(_7\)) and phosphorus (P) from the WW, but since the late 80s techniques have been added to remove nitrogen (N) (Swedish EPA, 2018). The degree of removal of P and BOD\(_7\) has been stable over 95 % for the last decade, whereas the removal of N is lower and more varying, from 37 to 72 %. In 2016 the mean degree of removal of N was 62 %, but the degree increases with increased capacity (Swedish EPA and Statistics Sweden, 2018).

When the cleaned water leaves the WWTP, sewage sludge (SS) is left behind. The sludge contains the nutrients, organic compounds and other leftovers from the WW. The composition of the sludge is dependent on what the incoming WW contains, what have been flushed down up streams (Henriksson, et al., 2012). Pharmaceutical residues, heavy metals and various organic toxins are examples of substances the sludge may contain. Some are transformed and removed during the treatment processes, but some remain in the sludge (Henriksson, et al., 2012). There are several uses for SS. The three most common in Sweden are agricultural soil improvement, construction soil and landfill cover (Swedish EPA and Statistics Sweden, 2018).

3.1.2 Wastewater treatment process

The treatment process is generally divided into two steps: water treatment and sludge treatment.

Water treatment

The water treatment in Sweden is usually a combination of mechanical, biological and chemical treatment as shown in figure 1 (Swedish EPA, 2018). The mechanical step removes solid waste, grit, plastics, sand etc. from the WW by using screens and grit chamber. This is important to prevent problems for the pumps in the rest of the process.
The biological step is called the active sludge (AS) process, where bacteria removes N and organic material (BOD$_7$) from the WW. Aerobic and anaerobic bacteria transform organic nitrogen and ammonia (NH$_4^+$) to nitrogen gas (N$_2$) through nitrification and denitrification (Table 2). The aerobic bacteria require oxygen to transform NH$_4^+$ to nitrate through two steps (nitrification). Following this step, the anaerobic bacteria require carbon input and an anoxic environment where they can transform nitrate to nitrogen gas (denitrification)(Carlsson & Hallin, 2003; Swedish EPA, 2018). Unsuccessful nitrification or denitrification caused by e.g. too high or too low oxygen levels are sources of N$_2$O emissions (Carlsson & Hallin, 2003).

<table>
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<th>NITRIFICATION</th>
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<th>DENITRIFICATION (SIMPLIFIED REACTION)</th>
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<tr>
<td>nitrate $\rightarrow$ nitrite $\rightarrow$ nitric oxide $\rightarrow$ nitrous oxide $\rightarrow$ nitrogen gas</td>
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<td>NO$_3^-$ $\rightarrow$ NO$_2^-$ $\rightarrow$ NO $\rightarrow$ N$_2$O $\rightarrow$ N$_2$</td>
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<th>DENITRIFICATION (TOTAL REACTION)</th>
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<td>Step 4</td>
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In the chemical treatment, a chemical coagulant (e.g. aluminium sulphate or ferric sulphate) is added to flocculate the phosphorus before the water is released to the recipient (Swedish EPA, 2018).

**Sludge treatment**

Throughout the water treatment, the sludge is sedimented and collected. In Sweden, the most common treatment of sludge is anaerobic digestion to stabilize the sludge and take care of the emitted biogas for electricity, heat or vehicle fuel. After the anaerobic digestion the sludge is usually spread on agricultural soils or used as construction soil (Swedish EPA, 2018).
3.1.3 Regulations


These regulations specify allowed contents of nutrients and heavy metals, obligations when it comes to permits and reports, and required samples and inspections (Swedish EPA, 2018). The regulations may affect which measures for reduced carbon footprint that are available.

The sludge can become certified by Revaq, with the aim to provide high quality, nutrient rich sludge to agricultural soils and reduce harmful substances that the sludge may contain (Svenskt Vatten, 2020). The certification requires the organisation to be transparent and have a systematic process to improve the quality of the sludge. This includes improvements in techniques as well as working with upstream users to reduce harmful substances in the sludge. The sludge has to be tested for harmful substances with results below the allowed limits, and become properly hygienised to remove pathogens before it is spread on agricultural soils. There are several ways to hygienise the sludge, e.g. thermal treatment or storage over 6 months (Swedish EPA, 2013).

3.1.4 Ekebyhov

Ekebyhov is a small scale WWT plant in eastern Stockholm, with a capacity to treat 25000 pe. It was built in 1975 and added to 1989 following new rules for cleaning nitrogen (N). The recipient is lake Mälaren, the fresh water source for all of Stockholm. The plant treats both water from sewage lines and from private cisterns. The sewage sludge is certified by Revaq.

The treatment process goes through the following steps (figure 2):

Wastewater treatment

1. Wastewater (WW) is transported to the plant by sewer lines or trucks.
2. Sand and solid waste is removed from the WW. The screenings that are removed are pressed to reduce water and then transported with the domestic waste to a waste incineration facility. Sand is supposed to be cleaned and returned to nature, but this process is lacking in Ekebyhov, releasing some sand to the following steps. The sand that is collected is sucked into a truck and transported to a landfill.
3. Pre-denitrification by Anaerobic/Anoxic treatment
4. Nitrification and oxidation of BOD by aeration (DO level around 2 mg/L)
5. Deox chamber to remove oxygen, enabling denitrification
6. Recirculation of nitrate rich water to be returned to the anoxic treatment for denitrification (197 % of influent water)
7. Mid sedimentation, most of the sludge is removed to the thickener (and some added to step 3 to preserve the bacteria)
8. Chemical coagulant (aluminium sulphate, ALG) is added to flocculate P
9. Final sedimentation, the rest of the sludge is removed to the thickener (and some added to step 3 to preserve the bacteria)
10. Clean water is released to the recipient Mälaren, some 450 meters from the shoreline. Heat from the clean water is reused to warm the facility, before being released.

Sludge treatment

11. The thickener uses gravity to compress the sludge. Excess water is added to step 5
12. Underground sludge storage holds the sludge before centrifugation.
13. Chemicals (Polymer, FLOPAM™ EM 440 HIB) is added.
15. Transportation of dewatered sludge by truck to over ground sludge storage. Stored for 6 months for hygenisation
16. Hygenised sludge is spread on agricultural soils to fertilize.

Samples are analysed on incoming and outgoing water (P, N, BOD, Metals) and manually on outgoing sludge (Metals and dry substance (DS)). The reduction rate of Tot-N at Ekebyhov is 80.1 % of influent N.

Photos 1-4 shows parts of the process at Ekebyhov, photographed by the author during a study visit to the WWTP.
Figure 2. The treatment process at Ekebyhov (Roslagsvatten, 2019) (used with permission, translated and numbers added by the author).
3.2 Global Warming

Global warming is a phenomenon with great consequences for ecosystems, societies and economies (Steffen, et al., 2015; Steffen, et al., 2018). Since the influence of climate change affects the entire earth system, it has been identified as a top priority amongst the planetary boundaries that makes life on earth possible (Steffen, et al., 2015). Possible effects of climate change are e.g. rising sea levels, increasing frequency of extreme weather, loss of species, food and water shortages, spread of diseases etc. (IPCC, 2018; Steffen, et al., 2018; Al-Ghussain, 2019). Climate scientists have a 97-100% consensus that climate change is already happening, and that it is caused by GHG emissions (mainly CO₂) from human activities (Cook, et al., 2013; Powell, 2017; Oreskes, 2004). The level of CO₂ in the atmosphere has been increasing since the industrial revolution and is the result of overuse of fossil fuels as energy source (Al-Ghussain, 2019; Hawkins, et al., 2017). The pre-industrial CO₂-level was approximately 227 parts per million (ppm), but has since then increased and arose above 400 ppm in 2015 (Le Quéré, et al., 2015). The emissions of GHG:s have led to a global temperature rise of over 1°C above pre-industrial levels (Hawkins, et al., 2017; IPCC, 2018), and have already started affecting the human and ecological systems, e.g. through melting glaciers, droughts, flooding and heatwaves (Lwasa, et al., 2018; Steffen, et al., 2018).

To deal with the effects of global warming, a global agreement has been signed in which both mitigation and adaptation strategies are combined (UNFCCC, 2015). The mitigating strategies aim to prevent climate related risks by reducing GHG- emissions to keep the temperature rise well below 2°C above pre-industrial temperature. Complementary to the mitigation, adaptive strategies must be put into place to minimize the effects that are too late to prevent (IPCC, 2018).

Keeping the temperature well below 2°C requires global net emissions to stay within a specified carbon budget. The carbon budget has been defined as the cumulative amount of net CO₂ emissions that can be released while still limiting warming with a specific minimum probability to below a given temperature threshold (IPCC cited in Fuss et al., 2018).

The remaining carbon budget stated by IPCC (2018) was about 420 Gt CO₂ for a 60 % chance of limiting warming to 1.5°C, and about 580 Gt CO₂ for an even chance. These figures have been highly debated, and are suggested to rather be around 0-200 Gt CO₂ (Minx, et al., 2018). The current net emissions rate is around 40 Gt CO₂/year, which (either way) leaves little time to reduce the emissions to net zero. The goal of reaching net zero is heavily reliant on carbon dioxide removal (CDR) with negative emission techniques (NETs) (Minx, et al., 2018). However, estimations show that the total potential of various NETs to remove CO₂ ranges from 4 to 24.6 Gt/year (Fuss, et al., 2018). This implies that even in the best case (24.6 Gt/year), we still have to make ambitious GHG emissions reductions to meet the goal by 2050.

Although global warming is in its essence a global environmental problem, the impacts of climate change will, most likely, fall disproportionately on developing countries and poor people (Sachs, 2008; Sealey-Huggins, 2017). At the same time, people in poor countries have way smaller carbon footprints than wealthier counties, with the latest data from 2014 showing 0.3 and 10.9 tons CO₂ per capita respectively (World Bank, 2014). Global warming is therefore a question of justice, and it becomes even more important that we in the wealthy part of the world do everything in our power to reduce our footprint.

3.2.1 Greenhouse gases, GWP and CO₂eq

Greenhouse gases (GHG) are substances that, when emitted to the atmosphere, affect the global climate by reflecting heat back to Earth’s surface (NASA, 2020). IPCC has classified GHGs with significant global warming potential (GWP) to be carbon dioxide CO₂, methane CH₄, nitrous oxide N₂O, chlorofluorocarbons and water vapor amongst others (Barber, 2009; Al-Ghussain, 2019).

The gases have different GWP, and to make them comparable to one another, they have been put in relation to the GHG most commonly emitted by humans, CO₂, and converted to CO₂ equivalents...
(CO₂eq) (IPCC, 2018; Barber, 2009). CO₂eq is an internationally accepted measurement, which explains the amount of CO₂ that would result in the same radiative forcing as the gas in question (Table 1) (Barber, 2009). CO₂ stays in the atmosphere for 100 years, which is why most of the CO₂eq are also based on their impact over 100 years in the atmosphere, and expressed as GWP₁₀₀ (Klöpffer & Grahl, 2014). One unit of N₂O or CH₄ has the GWP₁₀₀ of 298 or 34 units of CO₂, respectively (Table 1) (Myhre, et al., 2013).

*Table 2. CO₂eq for GHGs in this study, in relation to different time periods (Myhre, et al., 2013)*

<table>
<thead>
<tr>
<th>GAS</th>
<th>DURABILITY IN THE ATMOSPHERE (YEARS)</th>
<th>GWP 20 YEARS (CO₂EQ)</th>
<th>GWP 100 YEARS (CO₂EQ)</th>
</tr>
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<tbody>
<tr>
<td>CO₂</td>
<td>100</td>
<td>1</td>
<td>1</td>
</tr>
<tr>
<td>CH₄</td>
<td>12</td>
<td>86</td>
<td>34</td>
</tr>
<tr>
<td>N₂O</td>
<td>144</td>
<td>268</td>
<td>298</td>
</tr>
</tbody>
</table>
4. Method

4.1 Literature review

To provide an understanding of the research field, methods and technologies, previous research regarding wastewater treatment and GHG-emissions was broadly reviewed by using the website Web of science. Keywords used during the search were: “Water”, “Sewage”, “GHG-emissions”, “Sludge”, “LCA” and to keep the review up to date, the publication period was set to 2008-2020.

The resulting articles were briefly examined, looking at titles and abstracts and included if considered relevant to the thesis. Relevant cited papers in the collected articles were also included.

Previous research was also used to identify measures for reducing the carbon footprint.

To provide a background to WWT in Sweden, reports on wastewater treatment were collected from the websites Svenskt Vatten (Svenskt Vatten, 2019) and Swedish EPA (Swedish EPA, 2020b), and reports on the most recent knowledge about climate change were collected from IPCC (IPCC, 2020).

4.2 Study visit

In the beginning of the work with the thesis, a visit to the WWTP Ekebyhov was made. It provided a good background understanding of the treatment processes, logistics, chemical use, energy demanding processes and the internal routines for measurements and documentation. It has also established a connection with the staff, making it easier to receive further information.

Additional visits were planned, e.g. to the sludge storage, but were not possible to realize because of the corona pandemic. This made the initial contact with staff even more important, enabling digital communication.

4.3 Life cycle assessment

This study is based on a life cycle approach, but is more simplified than a complete life cycle assessment (LCA). While LCA considers several aspects, I will in this study only focus on global warming potential, which, according to ISO standard, makes it an incomplete LCA since it does not take other environmental aspects into account. The LCA framework, however, is still suitable for the case study.

LCA is a tool that enables a quantified analysis of the environmental impacts of the entire production and use of a product or process (Klöpffer & Grahl, 2014; Corominas, et al., 2013). It has been defined in the ISO 14040 standard as:

“LCA studies the environmental aspects and potential impacts throughout a product’s life (i.e. cradle-to-grave) from raw material acquisition through production, use and disposal. The general categories of environmental impacts needing consideration include resource use, human health, and ecological consequences.” (Klöpffer & Grahl, 2014)

LCA does not include economic or social impacts, but focus on environmental aspects only (Klöpffer & Grahl, 2014). When using LCA as a tool to make decisions it is important to not forget about the other aspects of sustainability. The strength of an LCA lies in visualizing and describing the environmental impacts, enabling decisionmakers to make environmentally sound changes in the product line, without laying the environmental burden somewhere else in the lifecycle (Klöpffer & Grahl, 2014; Zang, et al., 2015). Transparency is central to all LCA. It should be possible for anyone else to follow the data sources and make the same analysis. Without transparency it is difficult to make comparisons and draw conclusions from the LCA (Klöpffer & Grahl, 2014; Corominas, et al., 2013).
LCA has become a popular tool to investigate and evaluate the performance and techniques for WWT (Abusoglu, et al., 2017; Alyaseri & Zhou, 2017; Amann, et al., 2018; Chen, et al., 2019; Corominas, et al., 2013; Gallego-Schmid & Tarpani, 2019; Hao, et al., 2019; Lorenzo-Toja, et al., 2016; Niero, et al., 2014; Polruang, et al., 2018). Using LCA enables evaluation of eventual trade-offs, e.g. improved nutrient removal vs. increased chemical and energy use, or reduced GHG emissions vs. increased energy use (Yoshida, et al., 2014). However, it can also show synergies, e.g. improved nutrient removal leads to reduced GHG emissions.

A LCA can analyse the environmental impacts from two main perspectives: midpoint and endpoint (Alyaseri & Zhou, 2017; Lorenzo-Toja, et al., 2015; Zang, et al., 2015). A midpoint perspective analyses the more immediate impacts from the lifecycle and can describe e.g. global warming or eutrophication. The endpoint perspective goes further and calculates the final damages to ecosystems, e.g. looking at changes in biodiversity or climatic systems (Alyaseri & Zhou, 2017; Klöpffer & Grahl, 2014). This study has adopted a midpoint perspective, due to limitations in time and resources.

Key factors and terminologies for a LCA are to specify impact categories and functional unit, select and define system boundaries and have a transparent system inventory (Corominas, et al., 2013; Heimersson, et al., 2016; Klöpffer & Grahl, 2014; Zang, et al., 2015). This will be further elaborated upon in the following sub-sections.

### 4.3.1 Impact categories

Impact categories specify which environmental aspects will be investigated. It can be toxicity, eutrophication, global warming etc. (Klöpffer & Grahl, 2014; Zang, et al., 2015). A complete LCA should include all impact categories affected by the process, or otherwise motivated why they were excluded. Analysing several impact categories provide a better and more thorough understanding of the environmental impacts of a process or product, or changes of the same (Klöpffer & Grahl, 2014).

Impact categories relevant to WWT are: eutrophication potential, global warming potential, toxicity related impact categories, acidification potential, photochemical oxidation potential, ozone layer depletion, energy use, water use and land use (Zang, et al., 2015). However, due to limitations in time and resources, this study will only investigate global warming potential by the measurement of CO$_2$eq. A study that only investigates GWP does not qualify as a complete LCA according to the ISO standard, and can also be named “carbon footprint (CF) study” (Klöpffer & Grahl, 2014).

### 4.3.2 Functional unit

The selection of the functional unit (FU) determines the comparability of the LCA (Klöpffer & Grahl, 2014). Functional units relevant to WWT can be e.g. WW volume, amount of nutrients removed, nutrient content on influent WW, population equivalents (pe) or people connected to the WWTP (Corominas, et al., 2013; Tumlin, et al., 2014). A volume unit does not consider the quality of the WW or the nutrient content, a nutrient based unit does not account for the capacity of the WWTP and a population-based unit does not take external organic waste into account (Tumlin, et al., 2014). All of these factors may differ between WWTPs, which poses a problem. For instance, comparing two systems with different influent loads or with different removal efficiencies might result in misleading conclusions if using volume unit only as the functional unit. This shows the importance of selecting a representative FU, but also that there are pros and cons with all FUs (Corominas, et al., 2013; Tumlin, et al., 2014).

The most frequent used FU in LCA investigating WWTPs is a volume unit of treated WW, e.g. m$^3$ or ML (Corominas, et al., 2013). This thesis will present the results with several FUs, which will increase the comparability with other studies. The tool used for the study (presented below in section 4.4) will calculate the total annual emissions of CO$_2$eq, as well as CO$_2$eq per pe, m$^3$ influent, influent N, P and COD, and removed N, P and COD (Tumlin, et al., 2014).
4.3.3 System description and boundaries

The system boundaries provide a description of the system, explains what have been included and excluded in the analysis and determines the extent of the analysis (Klöpffer & Grahl, 2014). Clearly defined system boundaries are imperative, because two studies on the same system can have completely different results due to different extents of the boundaries (Klöpffer & Grahl, 2014; Yoshida, et al., 2014). A study with extended system boundaries that include all environmental impacts becomes a powerful tool for decision making (Zang, et al., 2015).

The system boundaries for WWT usually contain upstream processes, core processes and downstream processes, to various extents (Corominas, et al., 2013; Zang, et al., 2015). Upstream processes are e.g. collection and transport of WW to the plant. Core processes include construction, use and deconstruction of the plant, although few studies include the construction and deconstruction of the plant in the LCA (Corominas, et al., 2013; Tumlin, et al., 2014). The usage of the plant consists of water treatment process, energy and chemical input, sludge treatment, maintenance etc. (Zang, et al., 2015). Downstream processes are energy recovery, final disposal and sometimes fertilizer substitution (Corominas, et al., 2013; Zang, et al., 2015).

Figure 3. System boundaries for the analysis. Bold arrows indicate transport is included. Dashed lines indicate the processes are excluded from the analysis. Grey boxes represent substituted processes.

This thesis uses the boundaries set by the creators of the calculation tool (presented below in Section 4.4), but is extended in the upstream part to also include transport of WW to the plant. The boundaries of the system start with the collection of WWs from private tanks, through the water and sludge treatment processes and ends with the final use of the sludge, as shown in Figure 3. The boundaries include manufacturing and transport of chemicals, energy use, transports of sludge, sand and waste, impacts on the recipient and substitution of energy and mineral fertilizers. The boundaries do not include maintenance, sewer lines, transport of external organic material or the construction and deconstruction of materials, techniques or facilities.

4.3.4 System inventory

The system inventory makes it clear which material and energy inputs, outputs and various processes affect the defined system. It is a refined version of the system description and can contain transports,
manufacturing of techniques, waste generation etc. (Klöpffer & Grahl, 2014). The WWT system give rise to both direct (N$_2$O and CH$_4$ from water- and sludge treatment) and indirect (transport, manufacturing of chemicals etc.) GHG emissions. In accordance with guidelines from IPCC, it is assumed that 100 % of the CO$_2$ emitted directly from WW or SS is of biogenic origin and is therefore not included in the calculations (Tumlin, et al., 2014).

4.4 Calculation tool

The tool used for this study is an excel calculation tool (ECT), created with Swedish WWTPs in mind but grounded in international scientific knowledge (Baresel, et al., 2016; Tumlin, et al., 2014). It is based on the LCA framework and calculates the GHG emissions from the entire WWT process, including up- and downstream processes and both direct and indirect emissions. It consists of the excel document with accompanying manual and report (Tumlin, et al., 2014).

To enable calculations where plant-specific data is difficult to obtain, the ECT provides data from relevant literature and databases. It is possible to adjust the tool and use individual data, to adapt it to local circumstances (Tumlin, et al., 2014; Baresel, et al., 2016). Since CH$_4$ and N$_2$O are the dominant GHGs in WWT, conversion factors have been used to transform the emissions to CO$_2$eq. The conversion factors are 34 for CH$_4$ and 298 for N$_2$O, in accordance with guidelines from IPCC. Biogenic CO$_2$ emissions are not included (Tumlin, et al., 2014).

The ECT have been used by other researchers to calculate the carbon footprint of WWTPs (Adriansson & Turesson, 2016; Baresel, et al., 2016) with satisfying results. This study uses the ECT similar to previous authors.

First a base scenario will describe the current situation, including which parts of the WWT process that are the main emitters of GHG. After that, the GHG emissions-reducing measures identified from previous research will be tested with the ECT by altering the data. This will create several future pathways towards reduced emissions and visualise the most effective measures. The measures that will be analysed are:

- MBR and optimized processes
- Urea treated sludge
- Change of chemicals
- Optimized transports
- Anaerobic digestion of the SS

4.4.1 Assumptions

The calculations made by the ECT are based on a number of basic assumptions (Tumlin, et al., 2014), listed below.

The treatment used at Ekebyhov is a conventional activated sludge process, in which the aerobic and anaerobic/anoxic conditions generate the GHG emissions. The ECT assumes that the activated sludge process emits 0,0157 kg N$_2$O for every kg removed N, and 0,0025 kg CH$_4$ per kg influent COD.

The calculation for sludge storage assumes that the stored sludge emits 1,1 % of N-tot in sludge as N$_2$O over one year, and 0,0007 m$^3$ CH$_4$/ton VS every hour.

For the calculation of the transports’ footprint, it is assumed that the transports are 40 tons-trucks which are fuelled by diesel. An emission factor of 8,5 kg CO$_2$eq/10 km is assumed.

The ECT assumes that 20 % of the screenings are of plastic material of fossil origin. Therefore, parts of the CO$_2$ emissions from the incineration are not considered biogenic, but are included in the result.

The calculated emissions from the recipient are based on the amount of nitrogen that remains in the effluent water, and assumes a lower emission factor for freshwater than seawater, 0,0005 kg N$_2$O/kg N. It also assumes the lake emits a negligible amount of CH$_4$. 

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The footprint for the sludge use is based on a number of assumptions and modelled values, e.g. the ECT assumes that the soil emits a negligible amount of CH\textsubscript{4}, that 1% of available N is emitted as direct N\textsubscript{2}O emissions and 0,0157 kg N\textsubscript{2}O is emitted per kg emitted NH\textsubscript{3} as indirect emissions, and that 10 % of the C will be stored in the soil for at least 100 years.

### 4.5 Data collection

Klöpffer & Grahl (2014) stated that

“It is rarely ever possible to procure all data as primary data, that means to gather specific data at specific plants for specific processes. Therefore, a real LCI always consists of primary data, generic data and, where the one or other is not available, of estimations.” (Klöpffer & Grahl, 2014)

This thesis has used primary data as far as possible, and otherwise relied on standard data from literature or provided by the ECT.

The majority of the data have been collected from internal reporting systems at the WWTP. One extra analysis on SS has been made to receive the C-content. Data have also been provided by entrepreneurs via phone or e-mail. Below is a summary of the data used in the ECT. As previously stated, transparency is essential in any part of a system analysis, especially the collection of data (Klöpffer & Grahl, 2014). Therefore, a full dataset including sources can be found in Appendix 1.

#### Wastewater treatment

Data on nutrients and organic content on influent (COD 913 tons/year, BOD\textsubscript{7} 347 tons/year, N 137 tons/year and P 12 tons/year) and effluent (COD 131,7 tons/year, BOD\textsubscript{7} 5,9 tons/year, N 26,1 tons/year and P 0,4 tons/year) water are measured on routine at the WWTP. Data for 2019 have been provided from environmental reports for the WWTP (Roslagsvatten, 2020a).

No measurements have been made on the direct emissions of N\textsubscript{2}O and CH\textsubscript{4} from the WWT. Instead the standard values and calculations from the ECT have been used (Tumlin, et al., 2014).

#### Chemicals

The chemicals used at the plant are a coagulant (aluminium sulphate 222,9 tons/year) (Kemira, 2020a) for flocculating P, and polymer (8,4 tons/year) (SNF Nordic, 2020) for the drying of the sludge. The carbon footprint for aluminium sulphate (293 kg CO\textsubscript{2}eq/ton) was provided by the vendor (Kemira, 2020b) and the footprint for the polymer (805 CO\textsubscript{2}eq/ton) was provided by the ECT (Tumlin, et al., 2014).

#### Energy

Electricity (1871,8 MWh/year 2019) is provided by Energi Sverige and is 100 % renewable (Roslagsvatten, 2020a; Energi Sverige, 2020) and therefore generate no GHG emissions.

The plant does not generate any electricity, but reuses the heat from the water to generate enough heat to cover the needs of the plant (Khadhouri, 2020). There are no measurements of how much heat is generated and used over the year.

The plant has no anaerobic digestion of the sludge, and therefore no energy is conserved from the sludge treatment (Khadhouri, 2020).

#### Transports

Emissions from transports of chemicals, WW, SS and waste are accounted for in the analysis. The distances were calculated using google maps (Google, 2020) and the number of transports were provided by the entrepreneurs (Kemira, 2020a; SNF Nordic, 2020; Ragn Sells, 2020). For transportation of external sludge, it was estimated that the average tour was 10 km, and that each truck carried 10 m\textsuperscript{3}. 

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**Sludge and waste treatment**

Data on the amount of produced dried sludge (2442 tons 2019) and the contents of nutrients and heavy metals (68.33 kg N/ton TS, 24.83 kg P/ton TS, 3.78 kg K/ton TS) in dried sludge are measured on routine at the WWTP. Data for 2019 have been provided from environmental reports (Roslagsvatten, 2020a). An extra analysis was ordered to establish the carbon content (369 kg C/ton TS) (Eurofins, 2020).

No measurements have been made on the direct emissions of N₂O and CH₄ from the sludge. Instead the standard values and calculations from the ECT have been used (Tumlin, et al., 2014).

The amount of generated screenings and sand is not measured at Ekebyhov, so for the purposes of this analysis the measurements from a sister plant have been used and adjusted to the number of people connected to Ekebyhov (22.6 and 2.2 tons respectively) (Roslagsvatten, 2020b).

**Data for proposed measures**

The data collected for the proposed measures (Section 5.3) are presented below each measure.
5. Results and discussion

This chapter contains the carbon footprint of Ekebyhov, a sensitivity analysis of the data and suggested measures to mitigate GHG emissions. A full account of the GHG emissions by source and measure can be found in Appendix 2.

5.1 Baseline

The carbon footprint from the WW and SS treatment at Ekebyhov’s WWTP, based on the data from 2019 and emissions factors from the ECT, was calculated to be 522 tons CO$_2$eq per year (Table 3).

Table 3. The carbon footprint from Ekebyhov’s WWTP with different functional units

<table>
<thead>
<tr>
<th>TOTAL CO$_2$eq PER YEAR</th>
<th>522</th>
<th>TON</th>
</tr>
</thead>
<tbody>
<tr>
<td>CO$_2$eq per pe (70 g BOD$_7$ per day)</td>
<td>38,5 kg</td>
<td></td>
</tr>
<tr>
<td>CO$_2$eq per m$^3$ treated wastewater</td>
<td>0,3 kg</td>
<td></td>
</tr>
<tr>
<td>CO$_2$eq per removed N-tot</td>
<td>4,7 ton</td>
<td></td>
</tr>
<tr>
<td>CO$_2$eq per removed P-tot</td>
<td>44 ton</td>
<td></td>
</tr>
<tr>
<td>CO$_2$eq per removed COD-tot</td>
<td>0,7 ton</td>
<td></td>
</tr>
<tr>
<td>CO$_2$eq per influent N-tot</td>
<td>3,8 ton</td>
<td></td>
</tr>
<tr>
<td>CO$_2$eq per influent COD-tot</td>
<td>0,6 ton</td>
<td></td>
</tr>
<tr>
<td>CO$_2$eq per influent P-tot</td>
<td>42,8 ton</td>
<td></td>
</tr>
</tbody>
</table>

522 tons CO$_2$eq may seem large compared to 338 tons from Ängstorp (Adriansson & Turesson, 2016), or small compared to e.g. 8719 tons from Himmerfjärdsverket (Baresel, et al., 2016) or 3730 from Västra stranden (Adriansson & Turesson, 2016), but this functional unit does not account for the different capacities of the plants. For the purpose of comparison between plants, the FU kg CO$_2$eq per pe and year is more suitable. Ängstorp has a carbon footprint of 32,2 kg CO$_2$eq per pe, Västra stranden 36,9 kg CO$_2$eq per pe and Himmerfjärdsverket 37,4 kg CO$_2$eq per pe (Adriansson & Turesson, 2016; Baresel, et al., 2016). Ekebyhov falls just above these three with 38,5 kg CO$_2$eq per pe (Table 3), but below the average of 46 kg CO$_2$eq per pe identified in Gustavsson and Tumlin (2013) based on 16 Scandinavian WWTPs (Gustavsson & Tumlin, 2013).

Figure 4 shows the fractions of the emissions by percent of total emissions, and Figure 5 shows the positive and negative CO$_2$eq emissions by source of emission.

Figure 4. The percent of total emissions for every source
The major part (83%, 432 tons CO₂eq) of the GHG emissions are direct emissions originating from the activated sludge process (ASP) (Figure 4), out of which 356 tons CO₂eq comes from N₂O emissions, and 77 tons CO₂eq from CH₄ emissions. This result agrees with a substantial body of literature that attributes the major source of direct GHG to the WWT process (Gustavsson & Tumlin, 2013; Mannina, et al., 2020; Baresel, et al., 2016; Adriansson & Turesson, 2016; Jönsson, et al., 2015; Vasilaki, et al., 2019; Parravicini, et al., 2016). Tumlin et al. (2014), however, found the electricity to be a greater GHG source at 3 out of 4 plants due to the choice of electricity source (see section about electricity below), but WWT followed closely behind and was the greatest source at the fourth plant.

N₂O is created when the dissolved oxygen level in the aeration tank is too low and inhibits complete denitrification or when the nitrification is incomplete due to high levels of DO or nitrite (Carlsson & Hallin, 2003). Although there seems to be high agreement that WWT give rise to the majority of GHGs at WWTPs, Arnell (2013) and Vasilaki et al. (2019) show that the measured levels of N₂O can vary greatly between plants and even over time within the same plant. Model values for N₂O emissions are therefore inappropriate to use and it is advised to make local measurements over time to enable identification of proper measures (Arnell, 2013; Vasilaki, et al., 2019).

Sludge storage

During storage the sewage sludge emits direct GHG emissions of 80 tons CO₂eq through both CH₄ and N₂O emissions (0.8 and 0.2 tons, respectively, which corresponds to 52 and 28 tons CO₂eq, respectively). This represent 15% of total GHG emissions, and is the second largest source of GHG emissions at Ekebyhov. The sludge has to be stored for six months to become properly hygienised and available for agricultural use. Storage of sludge provide a more or less anaerobic environment, which favours production of both CH₄ and N₂O.

The results agree with Baresel et al. (2016), who also appointed sludge storage the second largest GHG source at Himmerfjärdsverket (not counting the use of methanol, which leaves a substantial footprint at Himmerfjärdsverket but is not used at Ekebyhov). Parravicini et al. (2016), also found sludge storage to be the second largest GHG source. Adriansson and Turesson (2016), however, does not include the storage time in their calculations, because the plants use other ways to hygenise the sludge. Possible hygenisation techniques are pasteurisation, thermophilic anaerobic digestion or hydro thermal
techniques. These are however, expensive and may yet leave some storage time of the end product awaiting transportation or agricultural spreading (Adriansson & Turesson, 2016; Swedish EPA, 2013). The calculations are based on data on emissions from digested sludge, since that is the standard in Sweden and therefore best available data. It is unclear if undigested sludge, as in the case of Ekebyhov, emits more or less CH₄ than digested sludge. The measurements made by Parravicini et al. (2016) shows that undigested sludge emits more CH₄ during storage than digested sludge. Contradicting results were received by Willén (2016) when comparing GHG emissions from digested and undigested cattle slurry. Therefore, it is essential to make local measurements to make better decisions. It is, however clear that sludge storage is a meaningful source of CH₄.

**Chemicals**

The production of the chemicals Aluminium sulphate (ALG) and Polymer (FLOPAM™ EM 440 HIB) provide a carbon footprint of 65 and 7 tons CO₂eq per year, respectively. This represents 14 % of total GHG emissions, and is the third largest source of GHG emissions at Ekebyhov. The emissions are indirect since they arise from upstream processes. Baresel et al. (2016), Adriansson and Turesson (2016) and Tumlin et al. (2014) uses more chemicals, and especially an external carbon source which provide a substantial carbon footprint. Since Ekebyhov has no pre-sedimentation tank, the COD from the influent is enough for the denitrification, and therefore the footprint of an external carbon source is saved.

Jones et al. (2016) also find that aluminium sulphate provides a large carbon footprint and states that a switch from aluminium sulphate to ferric sulphate is a viable option for reducing the carbon footprint (Jones, et al., 2016) (see measure 3).

**Transports**

5 % of Ekebyhov’s GHG emissions arise from indirect emissions from transports. The transports included in this analysis are chemicals, external sludge, sludge to storage, sludge to farmland, screenings and sand (Figure 6). The transports amount to a total of 27,41 tons CO₂eq per year, of which the majority is attributed to the transport of external sludge and chemicals to the WWTP (16,09 and 6,46 tons respectively), followed by sludge to farmland (4,62 tons). The transport of sludge to storage is relatively small, but could be reduced further had they not had to move the stored sludge to another facility because of lack of space.

Similar to Baresel et al. (2016), Adriansson and Turesson (2016), Tumlin et al. (2014) and Parravicini et al. (2016), is that the transport makes up a relatively small portion of the total GHG emissions, but Ekebyhov gives a slightly larger portion than the others. Unlike Baresel et al. (2016), Adriansson and Turesson (2016) and Tumlin et al. (2014), transport of external sludge was included in Ekebyhov’s footprint. It is appropriate since Ekebyhov accepts sludge both from smaller WWTPs and from private household tanks. The calculations on transport of external sludge is, however, based on estimations, which makes this result somewhat uncertain.

![EMISSIONS FROM TRANSPORTS, TON CO₂EQ/YEAR](image)

*Figure 6 shows the CO₂eq emissions from the transports included in the processes at Ekebyhov*
**Screening, sand and recipient**

The indirect emissions from the handling of screening and sand amount to 5 tons CO$_2$eq per year or 1% of the total GHG emissions. Out of the 5 tons almost all emissions come from the incineration of waste, and the landfilling of the sand generates 0.04 tons CO$_2$eq. The heat generated from the incineration is used for central heating and substitutes alternative heat sources, which leaves a small negative footprint of -0.002982 tons CO$_2$eq.

The direct N$_2$O emissions from the recipient of Ekebyhov are estimated to be equivalent to 4 tons CO$_2$eq per year or 1% of the total GHG emissions.

These results are in line with the results of Baresel et al. (2016), Adriansson and Turesson (2016) and Tumlin et al. (2014), who all concluded that GHG emissions from sand, waste and recipient were the smallest part of the total GHG emissions, together with transports.

**Electricity**

Roslagsvatten has chosen to use 100% renewable energy for the operation of Ekebyhov, which has resulted in zero emissions from energy use. Tumlin et al. (2014), on the other hand, found electricity to be the largest source of GHG emissions in three out of four plants, depending on electricity use and local generation of electricity from biogas. The source of energy in Tumlin et al. (2014) was set as EU future mix, which emits 350 tons CO$_2$eq/GWh, whereas Adriansson & Turesson (2016) chose Swedish mix (10 tons CO$_2$eq/GWh) and Baresel et al. (2016) used 100% renewable energy.

The electricity demanding processes such as aeration and pumps show the importance of the decision to select renewable energy as source of electricity. This is also clearly demonstrated in figure 7 which shows the CO$_2$eq that would be emitted, had Roslagsvatten chosen another source of electricity.

![Emissions from electricity use, Ton CO$_2$eq/year](image)

*Figure 7. Ton CO$_2$eq emissions from electricity use, based on source.*

**Sludge use**

The footprint for Ekebyhov’s sludge use is negative by -99 tons CO$_2$eq per year (Figure 8). The sludge is used on agricultural soil, where the soil becomes a carbon sink and the nutrients (N, P and K) act as fertilizers. The sludge is therefore a substitute to synthetic fertilizers. The production of synthetic fertilizers leaves a carbon footprint (40.3 tons CO$_2$eq for N, 4.5 tons CO$_2$eq for P and 0.7 tons CO$_2$eq for K), which can be subtracted from the sludge use as a negative footprint. The carbon sequestration also leaves a negative footprint (-67.8 tons CO$_2$eq per year). However, the N in the sludge is a source of
N₂O and leaves a positive footprint (9.7 tons CO₂eq per year), as does the CO₂ emissions from the loading and spreading of the sludge (1.2 and 3.2 tons CO₂eq per year, respectively).

Other studies have also received negative emissions from the use of sludge on agricultural soils, but contrary to this study (which have 80 tons CO₂eq emissions from storage, see above), the positive emissions from the storage of sludge were larger than the negative emissions from the use of sludge (Baresel, et al., 2016; Tumlin, et al., 2014). Adriasson & Turesson (2016) did receive net negative emissions from sludge use from both Västra stranden and Ångstorp. However, they did not account for any GHG emissions from storage of the sludge.

### 5.2 Sensitivity analysis

This study is based on a number of assumptions and emissions factors (EF), see section 4.4.1. To evaluate the impact and importance of the chosen EFs on the total carbon footprint (CF), a sensitivity analysis was made based on best- and worst-case scenarios from the literature (Table 4). The result can indicate which data are uncertain and which data require local measurements to draw clear conclusions (Tumlin, et al., 2014). See full results of the sensitivity analysis in Appendix 3.

The analysis shows that the main uncertainties are surrounding EFs on N₂O emissions from the recipient (H) and WWT (A) (Figure 9). The carbon footprint may be up to 48 % smaller or 136 % larger than the baseline result, if other EFs for N₂O emissions from WWT are more suitable for the situation at Ekebyhov (A). Yet, even in the best-case scenario, the N₂O emissions from WWT represent the majority of the emissions (107 out of 273 tons CO₂eq per year). The EF for CH₄ emissions from WWT (B) is somewhat more stable, but there is still a 42 % difference between best- and worst-case scenarios.

The N₂O emissions from the recipient can result in a much larger CF (238 %) than calculated in the base result, which shows there are great uncertainties surrounding N₂O emissions from recipient (H), whereas a change of EF for CH₄ emissions from the recipient (G) gives a small increase to the total CF. The change of EFs for level of substitution for N and P (C and D) and level of carbon sequestration (I) make little or no difference to the CF.

The EFs for CH₄ and N₂O emissions from storage of sludge (E and F) can impact the CF by 18 % and 16 %, respectively. However, even the EFs for the best- and worst-case scenarios are based on data for digested sludge whereas the sludge at Ekebyhov is undigested. It is unclear how this affects the GHG emissions from the sludge. Since there is a higher C-content (because of no extraction of biogas) an argument can be made that the CH₄ emissions can be even higher than the results and sensitivity analysis.
show. The emissions can also be lower, due to the fact that no digestion process has started and there is a lower temperature in the sludge.

Given the result of the sensitivity analysis, local measurements on CH$_4$ and N$_2$O emissions from WWT and sludge storage are highly motivated.

**Table 4. Parameters with corresponding best- and worst-case data for sensitivity analysis.**

<table>
<thead>
<tr>
<th>SCENARIO</th>
<th>PARAMETER</th>
<th>USED VALUE</th>
<th>UNIT</th>
<th>BEST CASE</th>
<th>REFERENCE</th>
<th>WORST CASE</th>
<th>REFERENCE</th>
</tr>
</thead>
<tbody>
<tr>
<td>A</td>
<td>Nitrous oxide emissions WWT</td>
<td>1</td>
<td>% of denitrified N</td>
<td>0,03</td>
<td>Foley et al. (2010)</td>
<td>3</td>
<td>Foley et al. (2010)</td>
</tr>
<tr>
<td>B</td>
<td>Methane emissions WWT</td>
<td>0,0025</td>
<td>Kg CH$<em>4$/kg COD$</em>{in}$</td>
<td>0</td>
<td>Gunnarson et al. (2005)</td>
<td>0,007</td>
<td>Göte (2013)</td>
</tr>
<tr>
<td>C</td>
<td>Substitution synthetic N</td>
<td>32,5</td>
<td>% of N in sludge</td>
<td>75</td>
<td>Foley et al. (2010)</td>
<td>15</td>
<td>Peters &amp; Rowley (2009)</td>
</tr>
<tr>
<td>D</td>
<td>Substitution synthetic P</td>
<td>70</td>
<td>% of P in sludge</td>
<td>75</td>
<td>Foley et al. (2010)</td>
<td>25</td>
<td>Foley et al. (2010)</td>
</tr>
<tr>
<td>E</td>
<td>Methane emissions sludge storage</td>
<td>0,0007</td>
<td>Nm$^3$ CH$_4$/ton VS</td>
<td>0,0025</td>
<td>Gabriel et al. (2003)</td>
<td>0,0025</td>
<td>Gabriel et al. (2003)</td>
</tr>
<tr>
<td>F</td>
<td>Nitrous oxide emissions sludge storage</td>
<td>1,1</td>
<td>% of N-tot in sludge</td>
<td>0,5</td>
<td>Kirkeby et al. (2005)</td>
<td>1,1</td>
<td>Willén et al. (2011)</td>
</tr>
<tr>
<td>G</td>
<td>Methane emissions recipient</td>
<td>0</td>
<td>kg CH$<em>4$/kg COD$</em>{in}$</td>
<td>0</td>
<td>IPCC (2006a)</td>
<td>0,025</td>
<td>IPCC (2006a)</td>
</tr>
<tr>
<td>H</td>
<td>Nitrous oxide emissions recipient</td>
<td>0,0005</td>
<td>kg N$<em>2$O/kg N$</em>{en}$</td>
<td>0,0003</td>
<td>IPCC (2006a)</td>
<td>0,16</td>
<td>IPCC (2006a)</td>
</tr>
<tr>
<td>I</td>
<td>Carbon sequestration</td>
<td>0,1</td>
<td>kg C/kg C-applicated</td>
<td>0,2</td>
<td>Foley et al. (2010)</td>
<td>0</td>
<td>Foley et al. (2010)</td>
</tr>
</tbody>
</table>

![Figure 9. The percentual deviation from the baseline carbon footprint for each parameter in the sensitivity analysis (Table 4.).](image-url)
5.3 Possible future measures

The identified measures are presented below in a descending order, starting with the largest source of GHG emissions and ending with possible sinks. For every measure the potential GHG emission reduction (PGER) is calculated. The results below are presented with the functional unit ton CO$_2$eq per year, but a full presentation with the results of the measures with different functional units can be found in Appendix 4.

5.3.1 Measure 1, regarding wastewater treatment

This section proposes an optimization of the WWT to reduce GHG emissions.

Since the ASP is the largest emitter in the process, it is essential to address this source. Arnell (2013) suggests that well working plants with minimum disturbances and low levels of N in the effluent water run a low risk for N$_2$O emissions. This requires optimized processes. Vasilaki et al. (2019) made a review of various measures taken to reduce N$_2$O emissions, some of which are adopted below. Baresel et al. (2016) used the assumption that the direct emissions from water treatment would be reduced by 30 % with the use of membrane technique (MBR) and optimized processes. This is supported by Kim et al. (2015), who proposes that optimized processes such as dissolved oxygen levels and internal recycle rate may result in reduced GHG emissions by 38 % (Kim, et al., 2015).

Accounting for local variations, an assumption of 30 % reduction is adopted here. The proposed optimizations are:

- Introduce MBR (Baresel, et al., 2016)
- High levels of dissolved oxygen in aeration tanks (4-6 mg/l) (He, et al., 2017; Kim, et al., 2015)
- Optimal aeration intensity to minimize N$_2$O stripping yet keep the DO level high (Vasilaki, et al., 2019)
- Low levels of DO in the Anoxic zone (Arnell, 2013)
- High internal recycle rate (400-500%) (Kim, et al., 2015; Arnell, 2013)
- Avoid accumulation of NO$_2^-$ (concentration below 0,3-0,5 mg NO$_2^-$-N/l) (Vasilaki, et al., 2019; Arnell, 2013)
- Prevent NH$_4^+$ concentration peaks with equalisation tanks (Vasilaki, et al., 2019; Arnell, 2013)

The optimized WWT results in a PGER of 173 tons CO$_2$eq per year, and gives total emissions of 349 tons CO$_2$eq per year (Figure 10).

This is in line with the results from Baresel et al. (2016), but somewhat contradicts comparative studies between membrane (MBR) and conventional activated sludge (CAS) techniques (Lazarova, et al., 2012; Mannina, et al., 2020). Lazarova, et al., (2012) have compared CAS and MBR, and concludes that the latter gives rise to increased GHG-emissions. Mannina, et al., (2020) arrives at the same conclusion, and attributes the increased emissions to the higher aeration required for MBR to reduce fouling. This results in increased direct CO$_2$eq emissions by stripping of soluble N$_2$O, as well as increased indirect emissions due to the additional energy needed for the aeration. However, both the WWTP studied in Baresel et al. (2016) and Ekebyhov uses 100 % renewable energy, which would eliminate the indirect emissions arising from increased energy consumption. The contradicting results can be due to variations in the operation processes of the plants, or that Baresel et al. (2016) and Kim et al. (2015) may not have accounted for the increased stripping of soluble N$_2$O which affects this study since the assumptions for the calculation are based on their data. Baresel et al. (2016) was not transparent about what kind of process optimizations were made, which makes it difficult to analyse.

Parravicini et al. (2016) states that increased Tot N removal correlates with decreased N$_2$O emissions from the activated sludge. Between their two plants, one had a Tot N removal efficiency of 77 % and N$_2$O EF of 0,75 % /Tot N$_{influent}$, whereas the other had a N$_2$O EF of 0,05 % /Tot N$_{influent}$ with a removal
efficiency of 92% (Parravicini, et al., 2016). Ekebyhov’s removal efficiency was 80.9% in 2019 (Roslagsvatten, 2020a), which means that there is room for improvement.

Figure 10. Positive and negative emissions by source, including the reduced emissions from the measures for WWT. The measures reduce the total emissions by 173 CO₂eq per year.

The fact that there is no conflict, but rather synergies between nutrient removal and reduced N₂O emissions is positive for the integration of optimization measures. It is, however, important to keep in mind that emissions from WWT can vary greatly between and within WWTPs, from 0.0001 to 0.112 kg N₂O-N/kg tot N (Arnell, 2013). These measures are therefore uncertain.

During my research I have not come across any WWTP that uses a closed WWT system with a N₂O destructor to reduce the carbon footprint. Considering the potentially huge impact emissions of N₂O have on global warming, this technique should be further investigated.

5.3.2 Measure 2, regarding sludge treatment

This section proposes urea treated and covered sludge to reduce GHG emissions.

As the second largest emission source, the sludge storage has to be addressed. The pathogen-reducing properties of the ammonia in urea also inhibits the activity of both nitrite- and ammonia-oxidizing bacteria, as well as the anaerobic microorganisms that create methane, and may therefore prevent the production of both N₂O and CH₄ from sewage sludge (Fidjeland, et al., 2013; Willén, et al., 2016a). Willén et al. (2016b) performed measurements of GHGs from sludge treated with various treatments. They received a negligible N₂O EF of 0,00 % of initial tot-N and a low EF for CH₄ of 0,4 % of initial tot C from the storage of sludge.
treated with urea. Mesophilic digested sludge with no treatment, produced EFs of 0,34 % and 1,1 %, respectively (Willén, et al., 2016b).

The reduction of emissions from agricultural application are based on, and correspond with measurements from Willén et al. (2016a). However, two LCAs conducted on various sludge treatments by Willén et al. (2017) and Svanstrom et al. (2017) show that urea treated sludge increases N₂O emissions when spread on agricultural soil, due to the increased concentration of ammonia. This contradicts the studies showing that the high levels of NH₄⁺ or NH₃ inhibits the activity of nitrifying and denitrifying bacteria (Arnell, 2013; Willén, et al., 2016b). They also found that the production of urea is energy intensive and leaves a quite large footprint, which have not been included in the present study due to incomplete data. Still, the urea treated sludge in their analyses emitted only slightly more GHGs than the best option, since the reduction of emissions from sludge storage is of such magnitude.

Besides reducing GHG emissions from sludge, the urea treatment also has the advantages of 1) being cost effective and suitable even for smaller WWTPs, and 2) hygenising the sludge at the same time, removing the need for long time storage (Fidjeland, et al., 2013; Svanstrom, et al., 2017).

Another, even less invasive approach would be to cover the storage. Willén et al. (2016b) show that the N₂O EF from sludge stored under cover is 0,19 % of initial tot-N, which is somewhat higher than urea treated sludge, but lower than untreated sludge. The CH₄ EF, however, is greatly increased to 1,3 % of initial tot C due to the anaerobic conditions when stored under cover. This could be solved by flaring the CH₄ emissions, which would turn the methane into CO₂ that both provide a 34 times smaller GWP than CH₄, and is considered biogenic. The methane could also be used for heating or destructed through thermic or catalytic oxidation techniques (Avfall Sverige; Svenskt Vatten, 2019).

### 5.3.3 Measure 3, regarding chemicals

This section proposes a change of coagulation chemical to reduce the carbon footprint.

The production of the coagulant aluminium sulphate leaves a footprint of 65 tons CO₂eq per year, whereas the coagulant ferric sulphate is a by-product from other production and therefore leaves no footprint. A change to ferric sulphate would therefore create a PGER of 65 tons CO₂eq per year and reduce the total carbon footprint to 456 tons CO₂eq per year (Figure 12).
The same measure is also proposed by Jones et al. (2016) in a study about drinking water treatment. In the study three out of four sites uses aluminium sulphate as coagulant with a footprint of 145 kg CO$_2$eq/ton, and the fourth uses ferric sulphate with the same good result but with a much lower footprint (Jones et al., 2016). Tumlin et al. (2014) has identified a ferric sulphate that is produced as a by-product and therefore leaves no footprint.

### 5.3.4 Measure 4, regarding transports

This section proposes to change the transports from fossil to renewable fuel.

The transports to and from the WWTP do not leave a substantial footprint compared to the other sources. None the less, should all transport of chemicals, sludge and waste be made by renewable fuel (such as biodiesel or electricity), the fossil footprint from transports would be removed since the emissions would be considered biogenic. This would create a PGER of 27 tons CO$_2$eq per year and leave a total carbon footprint of 494 tons CO$_2$eq per year (Figure 13).

A change to green transports was also suggested by Baresel et al (2016) to reduce the carbon footprint at Himmerfjärdsverket. They press the fact that infrastructure needs to be expanded. Some things have happened in this aspect since then, but there is still a lot of room for improvement. Since the largest part of transport emissions at Ekebyhov arises from the transport of external sludge to the plant, this part of the transport should be prioritised in a fuel change.

Besides changes in fuel source, the distribution of GHG emissions from transports to and from Ekebyhov motivates an extension of pipe lines to private households. The logistics of transports could also improve in several ways, e.g. make sure that the sludge only needs to move once.
**5.3.5 Measure 5, regarding biogas use**

This section proposes the use of an anaerobic digestion chamber, to enable extraction of the energy in the sludge by collecting the biogas and use as heat, electricity or upgrade to vehicle fuel and substitute fossil energy sources.

Since there is no data on the amount of biogas produced or the amount of methane slip, modelled values from the ECT together with data from a smaller sister plant (Blynäs, 4563 pe) have been used and adapted to the number of pe at Ekebyhov. Blynäs has a mesophilic anaerobic digestion chamber (32-33°C) and uses 85% of the gas for heating and 15% is torched (Roslagsvatten, 2020c). The measure presented here proposes instead to upgrade 85% of the gas to vehicle fuel since the substitution of fossil fuel can make a substantial impact and 15% for heating of the sludge. The methane content used in the calculation (65%) is adopted from Barresel et al. (2016).

The addition of an AD chamber and upgrading facility results in a total carbon footprint of 295 tons CO$_2$eq per year. This is a PGER of 227 tons CO$_2$eq per year. The reduction is mainly due to the substitution of fossil fuels (~327 tons CO$_2$eq per year) (Figure 14). The operation of an AD chamber often gives rise to CH$_4$ slip. In this calculation, model values from the ECT have been used (2.8% from the AD chamber and 0.2% from the upgrading facility), resulting in a carbon footprint of 100 tons CO$_2$eq per year.

The results are supported by a substantial number of reports, studying the potential to reduce the carbon footprint through AD (Barber, 2009; Fang, et al., 2019; Kiselev, et al., 2019; Teoh & Li, 2020). As can be seen in Figure 14, the substitution of vehicle fuel is an important factor to reduce the carbon footprint, and exceeds the emissions that arise from the production of biogas. The emissions from the production can however be reduced close to zero by the use of e.g. water locks and routine leak searches (Adriansson & Turesson, 2016; Avfall Sverige; Svenskt Vatten, 2019). This is addressed in subsection 5.3.6.

Kiselev et al. (2019) studies the extraction of energy thorough mesophilic or thermophilic AD in Ekaterinburg. The study supports the results of this theses in that the production of biogas through anaerobic digestion gives rise to some GHG-emissions but the benefits of substituting fossil fuels far outweigh the extra emissions (Kiselev, et al., 2019). Contrary to this thesis, Kiselev et al. (2019) concludes that using the biogas for electricity and heat supersedes the use as vehicle fuel. This is
probably because of the high use of fossil energy sources in Russia, whereas Ekebyhov uses 100% renewable energy. Teoh & Li (2020) also conclude that energy or fuel substitution plays an important role in reducing the carbon footprint of WWT.

Figure 14. Positive and negative emissions by source, including the reduced emissions from the addition of AD. The measure reduces the total emissions by totally 227 tons CO₂eq per year with the negative emissions from the biogas use (-327 tons CO₂eq per year) and the positive emissions from biogas production (100 tons CO₂eq per year).

Fang et al. (2019) studied the potential greenhouse gas emissions reduction (PGER) of anaerobically digesting the SS, with a set of different end uses: agriculture, building material, incineration or land fill in China. They receive a total PGER of -10.77 Mt CO₂eq and conclude that there is huge potential for mitigating GHG emissions if using AD as treatment for SS in China, regardless of the end use. The highest PGER was received from building material as end use (similar to the results by (Chen & Kuo, 2016)), but they do, however consider agricultural application the optimal end use and favour above agricultural application. Barber (2009) also concludes that raw sludge has a higher carbon footprint than AD options regardless of the end use, though (unlike Fang et al. (2019) and this thesis) energy generation through cofiring is favoured. Barber states that “Essentially, in terms of carbon footprint alone […] one could argue that any treatment of sludge without anaerobic digestion is unsustainable.” (Barber, 2009), which is a clear motivation for the measure to add AD to the process.

Another advantage of AD is the reduction of the sludge volume due to destruction of VS and removal of TS, which reduces the indirect emissions from transports (Teoh & Li, 2020). This advantage has however not been included in the calculation of the carbon footprint in this thesis.

To maximize the negative emissions various techniques can be used to improve the extraction of methane (Barber, 2009; Fang, et al., 2019; Teoh & Li, 2020). In fact, Barber (2009) suggest that “all new digestion plants should be designed with pre-treatment technology as a standard” (Barber, 2009). Thermophilic digestion, co-digestion with municipal food waste (Teoh & Li, 2020; Fang, et al., 2019), added pre-sedimentation (though extra carbon source would be needed) and hydrolysis (Barber, 2009) are examples of techniques to increase the methane yield. This is, however, not further elaborated in this thesis.

The calculation has not included the possible extra emissions that can arise from sludge liquor treatment or increased transport of fuel. Also, the construction of the chamber and upgrading facility has been excluded. Cahyani et al. (2019) made an LCA of an AD chamber at a small-scale tapioca industry. They found that the GHG emissions from installation and maintenance of the AD chamber contribute to 18.5
Kt CO₂eq over the lifetime (with fossil electricity sources) (Cahyani, et al., 2019). However, they calculated that the substituted fossil fuel would create a negative footprint of 296 Kt CO₂eq during the same time, creating net negative footprint. Another study, comparing two sludge-to-energy systems, found that the construction left a footprint of 0,12 and 0,05 tons CO₂eq per 500 m² treated sludge (Cao & Pawlowski, 2013). This is not directly applicable to Ekebyhov, but an important aspect to keep in mind.

An interesting alternative to AD is the use of pyrolysis that turn the sludge into bio-oil, syngas and bio-char. Studies have shown that this is a more favourable option in terms of carbon footprint than AD (Barry, et al., 2019; Von Bahr, 2016). The gas can generate electricity or heat, the bio-oil can be upgraded to vehicle fuel and there are several uses for the bio-char, e.g. application on agricultural soil with high level of nutrients, carbon sequestration and stabilizing properties, use in cement kilns as substitute to fossil carbon, use in fuel cells etc. (Barry, et al., 2019; Callegari & Capodaglio, 2018; Khan, et al., 2013; Marazza, et al., 2019; Von Bahr, 2016). This is, however, not further analysed in this thesis.

The relatively small footprint of transports and the ever-growing possibility for green transports raises another possible solution, to transport the sludge to an AD chamber at a larger WWTP or municipal facility for solid waste. The footprint from the construction of an AD-chamber or pyrolysis technique should therefore be analysed by LCA and compared with that of transport. It would remove the negative emissions from substitution of fossil fuel from Ekebyhov’s footprint, but in the bigger picture it may have a larger impact on GHG mitigation.

### 5.3.6 All measures

This section summarizes all previously proposed measures: optimized WWT, urea treated sludge, change of chemicals, green transports and added anaerobic digestion process including reduced methane slip.

The emissions from the production of biogas can in theory be reduced close to zero by the use of e.g. water locks, thermic or catalytic oxidation techniques and routine leak searches (Adriansson & Turesson, 2016; Avfall Sverige; Svenskt Vatten, 2019). In practice, it may be difficult to find and address all leaks, which is why a 50 % reduction is adopted here. Reducing the biogas slip by 50 % would further reduce the total carbon footprint by 50 tons CO₂eq per year.

![Figure 15](image)

*Figure 15. Positive and negative emissions by source before and after measures. The measures reduce the total emissions by 617 tons CO₂eq per year, creating net negative emissions.*

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Altogether, the PGER from the measures add up to 617 tons CO$_2$eq per year, which leaves a net negative footprint of -95 tons CO$_2$eq per year (Figure 14) or -7 kg CO$_2$eq per pe, year.

The largest PGER comes from the addition of an AD chamber and the use of biogas as vehicle fuel (Figure 15). This shows how effective the substitution of fossil fuel is, in terms of mitigating GHG emissions. Gustavsson & Tumlin (2013) concludes that maximizing the biogas production and use of the biogas as vehicle fuel are key components in reducing the carbon footprint of WWT.

Figure 16. Pie chart of the potential GHG emissions reductions, showing that biogas production and use shows the greatest potential with a reduction of 277 tons CO$_2$eq per year, followed by optimised WWT (173 tons CO$_2$eq per year).

The net negative result agrees with the statement by Gustavsson & Tumlin (2013), that “The energy content in wastewater [...] is far beyond the energy needed for wastewater treatment and carbon neutrality, or rather a negative carbon footprint of a WWTP, should be possible.”. Baresel et al. (2016) also received net negative emissions (-1 600 tons CO$_2$eq per year, or –6,9 kg CO$_2$eq per pe, year) after suggesting a number of measures at Himmerfjärdsverket. Measured per pe, the results from Himmerfjärdsverket and Ekebyhov are strikingly similar with -6,9 and -7 CO$_2$eq per year, respectively. Since Himmerfjärdsverket already has an AD chamber, the PGER came from optimising the AD and WWT (Baresel, et al., 2016).

### 5.4 Feasibility of proposed measures

The measures presented above show that there is huge potential to mitigate the GHG emissions from WWT. The practical feasibility and probability of implementing the measures is addressed here.

Optimizing the ASP requires careful measurements and adjustments to the different parameters over time, to monitor the development. This demands time, expertise and resources of the staff and perhaps new techniques for measurements, which may result in extra expenses for the company. The introduction of MBR technique is costly and energy demanding, but it is effective both in space and removal of toxic substances. However, as discussed above it is not clear if the technique would in fact decrease or increase the GHG emissions. Increasing the recycle rate of nitrate-rich water may not be possible with the current pumps, due to their limitations.

The urea treatment of the sludge would increase costs due to the purchase of urea, but decrease the need for long-time storage. It is a cost effective hygenising option compared to thermal treatments (Fidjeland, et al., 2013), and long-time storage may not be enough for future demands of hygenisation (Svanstrom,
et al., 2017). It is important to keep the concentration of N in mind, due to the limitations for applying N on agricultural soils.

A change of flocculating chemicals may already take place at Ekebyhov in the near future. Therefore, a change to ferric sulphate is very probable if it is as effective as aluminium sulphate to flocculate P. The fact that ferric sulphate is a by-product should be considered when deciding on a new coagulant. It may even be a cheaper option.

Changing the transports to non-fossil fuel may increase the expenses. Some entrepreneurs may not have the possibility to change, but this could be demanded in the public procurement when employing an entrepreneur. A change from diesel to HVO does not require a change of motor or vehicle, but it may be somewhat more expensive. If the time and resources are put in to create a system for more effective transport logistics, it should save money in the long run.

Introducing anaerobic digestion at the plant could be costly at first, but there are available governmental funds where you can apply for economic support and the biogas would eventually generate an income from the produced vehicle fuel. There may also be difficulties in gaining permission for the construction, due to time consuming processes and complaints of the smell from neighbours (Swedish EPA, 2011). The political direction towards a fossil free vehicle fleet is, however a major driving force in the expansion of biogas production, as is encouraging and driven staff at municipal and regional levels (Swedish EPA, 2011).

The fact that there is no regulation for GHG emissions from WWTPs, and consequently no economic sanctions, creates low incentives to actually implement the costly measures. There may also be practical difficulties in changing the processes, such as a possible need to close the WWTP for reconstruction or expansion of sewage pipes. Roslagsvatten do, however, have a vision to guide them; “We enable sustainable societies”, in which “sustainable” refers to “…minimise the environmental impact on society...” (Roslagsvatten, 2020d). This vision should include mitigating GHG emissions as far as possible, and therefore include these proposed measures.

It does, however, need to be emphasized that GWP should not be the sole environmental parameter for major changes. The results of this study should be complemented by studies on other impact categories such as eutrophication potential, toxicity potential, acidification potential, land use etc. to test the feasibility of the proposed measures in other aspects. E.g. increased level of DO in the aeration tank may result in much higher energy use, or ferric sulphate may not be as effective as aluminium sulphate to flocculate P which gives a higher eutrophication potential. All aspects need to be considered to make fully informed decisions.

5.5 Implications of methodical limitations

Literature review

There was no strict systematic approach to the literature review, which may result in deviations from what another researcher would have read. Articles that were considered relevant were included, and since relevance is very subjective, someone else may have found other articles relevant.

Study visit

The information from study visits may have been somewhat implicated by the corona virus that made further visits to the plant prohibited.

Life Cycle Analysis

The fact that only one impact category (GWP) is included in the LCA is a clear limitation. The identified measures may lead to improvements in GHG emissions but result in worse impacts for e.g. ecotoxicity, eutrophication or human health.
Calculation tool

The use of the ECT has some limitations. The system boundaries are relevant and necessary, but still clearly leaves some aspects out, e.g. methane emissions from pipe lines and construction of facilities, materials and techniques. According to Tangsubkul et al. (2005) the construction phase makes up 25-38 % of the total carbon footprint.

The sensitivity analysis in section 5.2 address the reliability of the data assumptions made in the ECT. Especially N₂O emissions from WWT can vary between plants, which motivates local measurements of the gas.

The ECT does not include biogenic CO₂ emissions because they are assessed not to be of fossil origin (in accordance with recommendations from IPCC). However, Kang et al. show that the content of sewage sludge is not 100% biomass and some of the CO₂ emissions originate from fossil carbon (Kang, et al., 2017; Kang, et al., 2018; Kang, et al., 2019). Therefore, it is possible the GHG emissions from CO₂ are underestimated in this calculation.

Data collection

The primary data, collected from internal documents, are of high reliability. Where primary data have been missing, the secondary data may be flawed. The sensitivity analysis (Section 5.2) address the reliability of some of the parameters.

For Measure 5, the addition of an AD chamber, data from a sister plant was used and adapted to the number of pe at Ekebyhov. It is, however, unclear if this is suitable or if there are other processes involved that would increase or decrease the biogas yield.
6. Conclusion

The results show that Ekebyhov’s wastewater treatment plant can, indeed, reduce its GHG emissions, and the finding that they can even become net negative is encouraging. The general findings are that 1) it is possible to mitigate the GHG emissions from wastewater and sewage sludge treatment, 2) the activated sludge process generate the major part of GHG emissions, 3) the choice of energy source makes a big impact on the carbon footprint, and 4) there is a large mitigation potential in using biogas from anaerobic digestion as vehicle fuel. These findings are likely to apply to most Swedish WWTPs, because, even though the emission factors can vary between WWTPs, there is a clear pattern in the distribution of GHG emissions from this thesis and other studies.

However, the huge potential for mitigation also raises the question of why these measures have not already been implemented. One cause may be that there is no clear incentive. The lack of regulations and economic sanctions regarding GHG emissions, together with the initial cost for implementing the measures probably impact the decisions (or lack thereof) to take mitigating measures. The licencing authority makes demands on the quality of the effluent water from the WWTPs, and should be able to make similar demands to limit GHG emissions. At the present, however, GHG mitigation is not demanded or even mentioned in the licence. There is also no carbon tax for WWTPs. Therefore, there is little incentive to measure the GHG emissions, and even less to mitigate them. Demanding GHG mitigation or putting a tax on GHG emissions would increase the incentives for implementing the proposed measures.

Internal GHG mitigation goals, motivated by the national and international climate goals could also be a means for Roslagsvatten to create tools for GHG mitigation. Following their own vision, to “…minimize the environmental impact on society…” it should result in ambitious goals. The choice to use 100 % renewable energy demonstrates that the organisation is motivated to contribute to the global mission to mitigate GHG emissions.

Another cause for not implementing the measures may be a lack of information and knowledge. The results of this thesis can contribute to filling this gap with knowledge about the main emission sources and mitigation options at Ekebyhov. There is, however, still a need for local measurements, especially of N₂O emissions from the activated sludge process and recipient, and CH₄ emissions from sludge storage. Another important platform of information is the Swedish wastewater association (SWWA). SWWA has a program that addresses CH₄ emissions from anaerobic digestion chambers, based on voluntary commitments to search for and rectify leaks (Avfall Sverige; Svenskt Vatten, 2019). The organisation provides practical information, collects data and give support to the plants who commit to the program. Even though it is voluntary, it has been successful in gaining commitments and mitigating CH₄ emissions. I suggest a similar program should be developed to address the big uncertainties surrounding N₂O emissions from wastewater treatment and support the WWTPs in mitigation efforts. Since this is potentially a huge contributor to global warming, local knowledge is essential.

The results of this thesis attribute the majority of the GHG emissions from the WWTP to the activated sludge process, followed by sludge storage. Addressing these sources is therefore essential to make a substantial reduction of the carbon footprint. The results also identified AD as a major potential sink of GHG emissions with the substitution of fossil fuels. Further elaboration on maximizing the biogas yield is, however, needed to enable the use of the full potential of the sludge. The relatively new field of pyrolysis should also be investigated as a potential alternative to AD. I recommend the use of the calculation tool when considering changes to the processes in the future.

The potential net negative result from the proposed measures in this study leaves room for compensating other GHG emissions. However, the consumption based per capita footprint in Sweden is around 9 tons CO₂eq per year, and the potential sink from wastewater and sewage sludge treatment is -7 kg CO₂eq per pe, year. This means there is still a long way to go to reach the limit of 2 tons CO₂eq per capita and year that is required to meet the national and international climate goals.
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Roslagsvatten, 2020d. Roslagsvattens vision. Åkersberga: Roslagsvatten AB.


## Appendix 1. Data set

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<th>UNIT</th>
<th>DATA</th>
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<td>8,4</td>
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<td>Used chemicals (ALG from Kemira)</td>
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<td>Number of deliveries per year (Polymer)</td>
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<td>(SNF Nordic, 2020)</td>
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<td>Number of deliveries per year (ALG)</td>
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<td>(Kemira, 2020a)</td>
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<td>Share of purchased electricity that is renewable</td>
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<td>Exported electricity from internal production</td>
<td>MWh</td>
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<td>Internally produced electricity from other sources than wastewater/biogas</td>
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<td>-</td>
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<td>Heat Use</td>
<td>Unit</td>
<td>Value</td>
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<td>Local emission factor for heat</td>
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<td>Methane content in biogas</td>
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<td>Biogas emissions (upgrading excluded)</td>
<td>Nm³</td>
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<tr>
<td>Torched biogas</td>
<td>Nm³</td>
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<tr>
<td>Direct emission of biogas to atmosphere</td>
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<td>Used biogas in gas engine and gas boiler</td>
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<td>Upgraded amount of biogas that has been compressed and transported on truck</td>
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<td>Upgraded amount of biogas for grid injection</td>
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<tr>
<td>Transport distance for compressed biogas on truck</td>
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<td>Added propane gas</td>
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<td>Energy use during upgrading</td>
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<th>Transports (sheet 5)</th>
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<tr>
<td>Transport of sludge to storage</td>
<td>km</td>
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<td>Transport of sludge to WWTP</td>
<td>Km</td>
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<td>Transport of sludge to farmland</td>
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<tr>
<td>Transport of screenings/sand/ashes</td>
<td>km</td>
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<td>Amount of dewatered digested sludge</td>
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<td>TS-content in sludge before storage</td>
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<td>VS-content in sludge before storage</td>
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<td>Ntot before storage</td>
<td>kg/ton TS</td>
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<td>K before storage</td>
<td>kg/ton TS</td>
<td>3.78</td>
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<tr>
<td>Ctot before storage</td>
<td>kg/ton TS</td>
<td>369</td>
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<tr>
<td>Ntot after storage</td>
<td>kg/ton TS</td>
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<td></td>
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<tr>
<td>-------------------------</td>
<td>--</td>
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<tr>
<td>K after storage</td>
<td>kg/ton TS</td>
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<tr>
<td>$C_{tot}$ after storage</td>
<td>kg/ton TS</td>
<td>367.4</td>
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<td>Share of sludge to farmland/incineration/land fill/reed beds</td>
<td>%</td>
<td>100 % farmland</td>
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<tr>
<td>Volume reduction after reed bed treatment</td>
<td>%</td>
<td>-</td>
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</table>

| Amount of screenings   | ton/year | 22.6 | (Roslagsvatten, 2020b) |
| TS-content in screenings | % | 40 | ECT |
| TS-content in screenings for incineration | % | 50 | ECT |
| Amount of sand         | ton/year | 2.2 | (Roslagsvatten, 2020b) |
| Share of screenings for incineration/land fill/digestion | % | 100 % Incineration | (Khadhouri, 2020) |
| Share of sand to recycling/land fill | % | 100 % land fill | (Khadhouri, 2020) |
Appendix 2. Results by source

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<th>Source</th>
<th>Base Line</th>
<th>Measure 1</th>
<th>Measure 2</th>
<th>Measure 3</th>
<th>Measure 4</th>
<th>Measure 5</th>
<th>All Measures</th>
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<td>0</td>
<td>0</td>
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<tr>
<td>% of Tot CF</td>
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<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
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<td>5</td>
<td>5</td>
<td>5</td>
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<tr>
<td>% of Tot CF</td>
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<td>2</td>
<td>1</td>
<td>1</td>
<td>2</td>
<td>2</td>
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<tr>
<td><strong>Recipient</strong></td>
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<td>4</td>
<td>4</td>
<td>4</td>
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<tr>
<td>% of Tot CF</td>
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<td>1</td>
<td>1</td>
<td>1</td>
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<td>27</td>
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<td>8</td>
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<td>432</td>
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<td>259</td>
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<td>% of Tot CF</td>
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<td>74</td>
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<td>-106</td>
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<td>-24</td>
<td>-22</td>
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<td>-34</td>
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<td>447</td>
<td>456</td>
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<td>100</td>
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### Appendix 3. Sensitivity analysis

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<th>UNIT</th>
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<th>COMMENT/REFERENCE</th>
<th>WORST CASE</th>
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<th>WORST CASE TOT CO₂EQ PER YEAR</th>
<th>BASE TOT CO₂EQ PER YEAR</th>
<th>BEST CASE DIFFERENCE CO₂EQ PER YEAR</th>
<th>WORST CASE DIFFERENCE CO₂EQ PER YEAR</th>
<th>BEST CASE DIFFERENCE %</th>
<th>WORST CASE DIFFERENCE %</th>
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<td>Foley et al. (2010)</td>
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<td>Foley et al. (2010)</td>
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<td>Peters &amp; Rowley (2009)</td>
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<td>Kirkeby et al. (2005)</td>
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Appendix 4. Results with different functional units

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<th>Measure</th>
<th>Total CO₂Eq per year</th>
<th>CO₂Eq per PE (70 g BOD₅ per day)</th>
<th>CO₂Eq per m³ treated wastewater</th>
<th>CO₂Eq per removed N-tot</th>
<th>CO₂Eq per removed P-tot</th>
<th>CO₂Eq per removed COD-tot</th>
<th>CO₂Eq per influent N-tot</th>
<th>CO₂Eq per influent COD-tot</th>
<th>CO₂Eq per influent P-tot</th>
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<td>0,3 kg</td>
<td>4,7 ton</td>
<td>44 ton</td>
<td>0,7 ton</td>
<td>3,8 ton</td>
<td>0,6 ton</td>
<td>42,8 ton</td>
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<td>30 ton</td>
<td>0,4 ton</td>
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<td>0,4 ton</td>
<td>28,6 ton</td>
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<td>0,6 ton</td>
<td>3,3 ton</td>
<td>0,5 ton</td>
<td>36,7 ton</td>
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<td>39 ton</td>
<td>0,6 ton</td>
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