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Improving life cycle inventories for nitrogen and phosphorus in sewage sludge used in agriculture

Master of Science Thesis
in Industrial Ecology

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Cover:
Sludge management of digested and dewatered sewage sludge at Gryaab, Ryaverket in Gothenburg.

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Abstract

When treating wastewater, large amounts of sewage sludge are generated. Sewage sludge contains substances that may be of use, e.g. nutrients such as nitrogen and phosphorus, but also substances that are harmful such as pathogens and heavy metals. In order to reuse the nutrients in sewage sludge it can be used as a fertilizer on agricultural fields in Sweden but there are regulations that specify how the sludge must be hygienized before use. One way of hygienizing sewage sludge in Sweden is by storing it. However, this method will probably not will be allowed as an hygienization method in the future, since the Swedish Environmental Protection Agency has not listed it as one of the hygienization methods they recommended to be acceptable in future legislation on the use of sludge in agriculture.

A way of assessing the environmental impact when sewage sludge is stored and applied to agricultural fields as a fertilizer is by using life cycle assessment (LCA). Previous reviews of LCAs performed in this field have shown a variety when it comes to which flows that are included in the life cycle inventories, and there is a need to improve the data quality when the flows are quantified. Therefore, this study was made to investigate if the life cycle inventory for nitrogen and phosphorus compounds could be improved and if there are differences in the characterization of nitrogen and phosphorus compounds in different methods, and what LCA practitioners needs to be aware of when assessing this kind of system. The three impact categories chosen were global warming potential, acidification potential and eutrophication potential.

The study was performed by literature searches and by personal contact with experts on nutrients and the practice when using sewage sludge in agriculture in Sweden. Also, an LCA case study was performed to illustrate how choices of inventory data and LCIA methods affect the environmental impact results.

The study concluded that sewage sludge storing and application to agricultural fields have a significant environmental impact regarding the three impact categories and the used characterization methods. The literature review combined with the personal contact and LCA case study indicated that there are many factors that affect the emissions of nitrogen and phosphorus from sewage sludge, e.g. soil type, soil chemistry, pH, how the sewage sludge has been treated, when the sewage sludge is incorporated into the soil and weather conditions. Therefore, it was shown that it is important to do a life cycle inventory for the specific system and not to use generic numbers of emissions since it can vary a great deal.

Key words: *Sewage sludge, Life cycle assessment, Nitrogen, Phosphorus, Mineral fertilizers, Agriculture, Storage.*

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Amanda Karlsson
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Abbreviations

AP – Acidification Potential

CF – Characterization Factor

EP – Eutrophication Potential

GWP – Global Warming Potential

ILCD – International Life Cycle Data system

IPCC – Intergovernmental Panel on Climate Change

LCA – Life Cycle Assessment

LCI – Life Cycle Inventory

LCIA – Life Cycle Impact Assessment

N – Nitrogen

P – Phosphorus

SS – Sewage Sludge

STRB – Sludge Treatment Reed Bed

TN – Total Nitrogen

WWT – Waste Water Treatment

WWTP – Waste Water Treatment Plant

1. Introduction

In the world today, collection of wastewater and wastewater treatment are usually implemented in densely populated areas in order to avoid eutrophication and dissemination of odors and diseases (Svanström, Laera & Heimersson, 2015). In Sweden, most of the collected water is treated in municipal wastewater treatment plants (WWTP). There are also households that have their own individual drains which require the individual property owner to ensure that the water meets the treatment requirements that exist (Svensk Avloppsrening, n.d.). The ambition of today's wastewater treatment is to treat the water, recycle energy and manage nutrients to create closed circuits (Svanström, Heimersson, & Harder, 2016). One way of doing this is to use the biogas that is generated during anaerobic digestion of sewage sludge as fuel by upgrading it to vehicle gas (Palm & Ek, 2010) or by using the sewage sludge as a fertilizer in order to reuse the nutrients contained in the sewage sludge (Svenskt Vatten, 2016a). However, in addition to the nutrients in the sewage sludge, the wastewater and sewage sludge also contain harmful substances. A few examples are heavy metals and organic micropollutants, which can make the reuse of sewage sludge difficult (Svanström et al., 2015).

One way of using sewage sludge, and recycle its nutrient content, is to use it as a fertilizer in agriculture. For the last centuries, Earth's population has grown tremendously and with the growth there has been an increasing demand for food. This has meant that the food industry, in turn, has an increased need for mineral fertilizers in order to obtain high crop yields. Two common nutrients in mineral fertilizers are nitrogen (N) and phosphorus (P) (Cordell, Drangert, & White, 2009). The N is mainly synthesized industrially in the Haber Bosch process, which is when atmospheric N is linked to hydrogen, forming ammonia (NH_3), which is the main form of N in mineral fertilizers (Schröder, 2014). P, on the other hand, is obtained from phosphate rock extraction. The rock reserves today could be exhausted in the closest 50 to 100 years if the extraction rate will continue at the same pace (Cordell et al., 2009). However, authors make different predictions of how long the phosphate reserves will last based on different assumptions. If the extraction rates stay at current levels the reserves are predicted to last until somewhere between 2080 and 2410 (Cordell & White, 2011).

One of the main tasks for the WWTPs of today is to remove both P and N from the wastewater before it is released to the recipient. The wastewater is commonly treated with a biological as well as a chemical treatment processes, resulting in a great deal of N and P ending up in the sewage sludge. The WWTPs in Sweden produce 1 million tons of sewage sludge per year (Svenskt Vatten, 2016b), and sewage sludge has many areas of use. In Sweden it is used as soil for road slopes, sealing layers for landfills or as fertilizer on arable land (Svenskt Vatten, 2016a). But before the sewage sludge can be used it needs to be managed in order to reduce its volume, the smell and to make it more stable (Jönsson et al., 2015). The most common sewage sludge management technique today is that the sewage sludge is thickened by a polymer additive. Thereafter it is stabilized by a digestion process and dewatered (Svenskt Vatten, 2013).

In order to be allowed to apply sewage sludge on arable land in Sweden, the sewage sludge needs to be sanitized (SNFS 1994:2). As an addition to the regulations in Sweden, a certification system called REVAQ has been developed as an industry initiative (Svenskt Vatten, 2019). It has stricter requirements for example regarding acceptable heavy metal contamination levels and demands constant improvements of the WWTP. The members are encouraged only to deliver REVAQ certified sewage sludge to agriculture (Svenskt Vatten, 2013), to ensure that the applied sewage sludge is of good enough quality and has been properly sanitized (Svenskt Vatten, 2018).

During the summer of 2018, the Swedish Government initiated an investigation on how P in the future could be recycled from wastewater and the possibilities for, and consequences of, a ban on spreading sewage sludge on soil in Sweden (Dir 2018:67). The current regulation concerning the use of sewage sludge on agricultural land is from 1998 and states the maximum levels of seven heavy metals, but nothing about hygienization (1998:944). In 2013, the Swedish Environmental Protection Agency requested, in a written report on sustainable recycling of P, stricter requirements of heavy metals and other substances, and that hygienization needs to be regulated (Naturvårdsverket, 2013). If the current investigation determines that sewage sludge may not be used in agriculture, it is necessary to combine this decision with a method and a requirement for how P is to be extracted from the sewage sludge. The investigation is expected to be completed in the autumn of 2019 (Dir 2018:67).

A way of assessing the benefits and potential impacts of applying sewage sludge on arable land is by having a life-cycle-perspective and use life cycle assessment (LCA) methodology (Svanström et al., 2015). A review of previous LCA studies focusing on sewage sludge management has shown shortcomings in how the LCA-methodology have been applied on these kind of systems (Yoshida, Christensen, & Scheutz, 2013). A review of LCA studies done by Heimersson, Svanström, Laera, & Peters (2016) shows that there are differences when it comes to which flows that have been included and the data that have been used for quantifying these flows. It also showed that there is a need for better modeling of sewage sludge applied on arable land. The replacement ratios are factors of how much mineral fertilizers that are being replaced by sewage sludge. The ones that are regularly used today seem arbitrarily chosen and is often the same values and they are poorly substantiated (Heimersson et al., 2016).

This study aimed to facilitate and improve the life cycle inventory (LCI) in future LCAs assessing systems in which sewage sludge is hygienised and applied on agricultural fields. The project focuses on the content of N and P in the sewage sludge, as these are not only a potential resource in agriculture, but also gives rise to important emissions to air and water in the studied systems.

1.1 Aim and objectives

The purpose of the study is to improve life cycle inventories for N and P in sewage sludge in systems where sewage sludge is stored and applied on agricultural fields. This is in order to improve assessment of the impact categories global warming potential (GWP), acidification potential (AP) and eutrophication potential (EP). A second purpose is to examine how the characterization of N and P containing compounds are done in different life cycle impact assessment (LCIA) characterization methods when assessing GWP, AP, and EP, for knowing what is needed for the inventory when applying these methods.

The objectives of the study are to answer:

- In what way is the LCI for the studied system (see Figure 1) in need of improvement and how can it be done?
- How to reason when choosing an LCIA-method for this system and what differences are there between the methods that LCA-analysts need to be aware of?

- How shall LCA-analysts reason when choosing a replacement ratio for how much mineral fertilizers that are replaced when using sewage sludge?

1.2 Delimitations

The study focused on direct emissions of N and P compounds originating from the sewage sludge during storage and application on agricultural fields, see Figure 1. The inventory of emissions from transports were thus not improved.

The results were tested in a case study under Swedish conditions, including also the wastewater treatment (WWT). However, much of the conclusions, as well as the inventory practices, are also relevant for a broader geographical area and application on different sorts of land. This makes the study highly relevant even with a possible future ban on the use of sewage sludge on agricultural fields in Sweden.

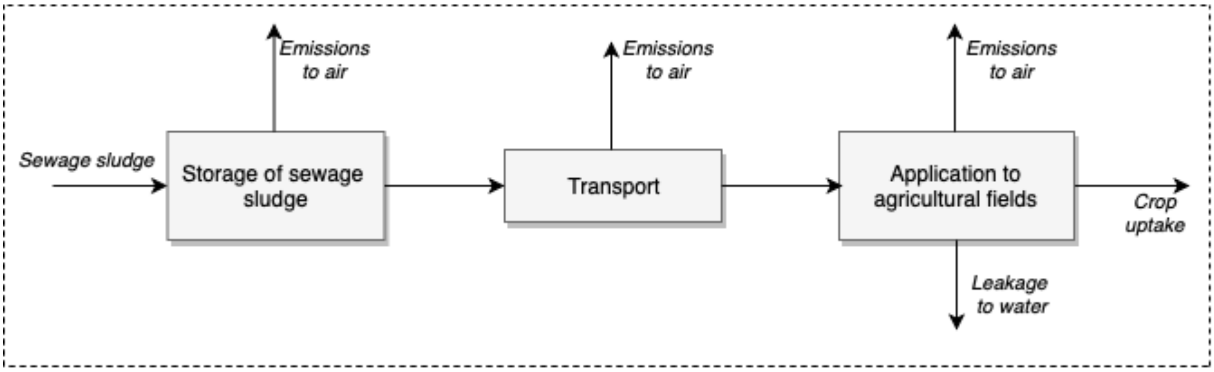

Figure 1: The studied system where sewage sludge is stored for hygienization reasons and thereafter applied as a fertilizer on agricultural fields.

The study focused on flows of substances of N and P that were of importance for the chosen impact categories: AP, EP and GWP. The LCIA methods that were studied was Accumulated Exceedence, ReCiPe, EDIP2003 and IPCC's GWP for climate change, selected from the recommendations in an ILCD-handbook (European Commission, 2010).

The geographical boundaries were set to be Sweden. This was due to the study being focused on sewage sludge produced and used in Sweden, but when necessary the gathering of data was expanded to other countries as well. The study was done in order to make future LCIAs when sewage sludge is applied on land, with focus of application on agricultural fields, more relevant and the data collection attempted to focus on the latest 20-30 years.

2. Background

To describe the different parts of the studied system (Figure 1), underlying background information will be presented. Sections 2.1 and 2.2 will present different treatment methods for sewage sludge and how storing and application to agricultural fields are done. In section 2.3 N and P and its emissions from agricultural soil are described.

2.1 Treatment methods for sewage sludge

Treatment of sewage sludge aims to reduce the number of pathogenic microorganisms but also to stabilize it, i.e. to partially break down the organic matter. Both the hygienization and stabilization are of importance in order to inhibit the spread of infection and preventing reinfection and regrowth of pathogenic bacteria. It can be done in separate processes for stabilization and hygienization or in one single process. Most often, the pH and temperature have the greatest effect on killing off pathogens. There are a number of different hygienization methods, for example anaerobic digestion, liming, storing, composting, pasteurization, urea treatment and thermal drying (Naturvårdsverket, 2011). Here follows a short description of a few treatment methods for a better understanding of the report and its results.

2.1.1 Anaerobic digestion

Anaerobic digestion is one of the most important and common stabilization processes for sewage sludge used today (Gavala, Yenal, Skiadas, Westermann, & Ahring, 2003). The stabilizing effect highly depends on temperature and exposure time (Kjerstadius et al., 2012). Mesophilic digestion is more commonly used today compared to thermophilic digestion, since it is performed during lower temperatures (about 35°C) and therefore it uses less energy. The process is also more stable and easily controlled (Gavala et al., 2003). However, it does not provide a sufficient hygienization effect to live up to the standards proposed in the EU or the proposed regulation from the Swedish Environmental Protection Agency (Kjerstadius et al., 2012). Thermophilic digestion, which is when the digestion process is done with higher temperature (about 50-55°C), has an enhanced hygienization effect and eliminates pathogens, which makes thermophilically digested sewage sludge a good choice when the intention is to apply sewage sludge on land (Gavala et al., 2003).

The ongoing work on recycling P from wastewater in Sweden (Dir 2018:67, n.d.) has led to an investigation into the possibilities of expanding the use of thermophilic digestion which is a less used method today because of the increased energy consumption in relation to the mesophilic digestion. Thermophilic digestion can give a good reduction of selected pathogens (Kjerstadius et al., 2012), however, it is important to investigate what exposure time is required to achieve sufficient hygienization for the thresholds that exist for different pathogens. Kjerstadius et al. (2012) investigated the potential effect of thermophilic digestion on other substances, such as drugs and PAH, but saw no greater effect.

One additional advantage of anaerobic digestion is that it is generating biogas. When the sewage sludge is digested there are bacteria that decompose the sewage sludge and biogas is produced. At Gryaab (Ryaverket) in Gothenburg more than 70 GWh of biogas is produced every year. It is then sold to an energy company in Gothenburg that compresses the gas and upgrades it to a methane (CH_4) content of 95 – 98 % and it is then used as vehicle fuel (Gryaab, n.d.).

2.1.2 Lime treatment

One way to both stabilize and sanitize sewage sludge is to add lime, which raises the pH. Several studies have shown that a pH over 12 leads to a reduction of pathogenic substances (Naturvårdsverket, 2011). Liming sewage sludge, which can be used as a complement to anaerobic digestion, does not demand a lot of energy but the production of quicklime is however energy demanding. One disadvantage is that lime is a finite resource and using lime when treating sewage sludge generates larger sludge amounts, which can be negative from a transport perspective (Svenskt Vatten, 2013). Also, if too much lime is added and the pH gets too high, it can be hard to restore a lower pH in the soil again (Linderholm, 2011).

2.1.3 Urea treatment

One method for hygienization of sewage sludge, that is relatively new and that is mainly developed in Sweden during the last 15 years, is urea treatment. The NH_3 in the urea is not consumed during the hygienization but can easily be taken up by crops in the form of NH_4^+. Therefore, treatment with ammonia has double value in both being sanitizing and giving the sewage sludge an increased fertilizer value. Thus, there are many benefits of adding ammonia to sewage sludge. Vinnerås, Nordin, & Jönsson (2017) claims that it is an uncomplicated method, and from a working environment perspective, a safe method when using urea. The sewage sludge gets a high fertilizer value and it has been proven to be effective against several evaluated organisms. However, the method has although shown to be slow for some viruses and, if the treatment temperature is low the treatment time can be very long (Vinnerås et al., 2017).

Emissions of greenhouse gases N_2O and CH_4 are considerably lower from urea treated sewage sludge. This is since the treatment inhibits growth of bacteria, and if there are less bacteria present in the sewage sludge there will be less formation of N_2O and CH_4 (Vinnerås et al., 2017). Jönsson et al. (2015) showed that urea treated sewage sludge generated smaller emissions of N_2O compared to a mesophilically digested sewage sludge when being spread on agricultural fields. Also, when storing sewage sludge, the emissions of both N_2O and CH_4 was showed to be smaller compared to other sewage sludge types and scenarios. However, the urea treated sludge was also covered up during storage (Jönsson et al., 2015; Willén, Rodhe, Pell, & Jönsson, 2016)

2.1.4 Sludge treatment reed bed systems

Another hygienization method for sewage sludge is sludge treatment reed bed systems (STRB) which is a common method in Denmark. The method is cost-efficient and environmentally friendly because no chemicals and only a minimal use of energy is needed (Orbicon, n.d.), and it takes care of surplus sludge from conventional wastewater treatment. The liquid sludge is supplied into great basins were the sewage sludge is, over a period of many years, dewatered and mineralized and turns into a sludge that can be used as fertilizer in agriculture (Steen Nielsen, Peruzzi, Macci, Doni, & Masciandaro, 2014). Low treatment costs compared to other treatment methods and that the cost of transporting and spreading the sewage sludge is reduced are some examples of what makes the method cost-efficient. The basins though are big and demands a lot of areal space and the method is also slow compared to others which can be seen as disadvantages (Orbicon, n.d.).

2.1.5 Sewage sludge storing

Storing of sewage sludge is one way of sanitizing it from pathogens and other microorganisms that the sewage sludge contains before applying it to soil. However, storing of sewage sludge can lead to large emissions of CH_4 and nitrous oxide (N_2O) (Jönsson et al., 2015). Storing the sewage sludge over a year could fulfill the lowest demand for hygienization according to the Swedish Environmental Protection Agency (Naturvårdsverket, 2002). This kind of storage is often done on an open hard surface. The hygienization effect can be hard to predict even though it is known that the microorganisms decrease over time (Naturvårdsverket, 2011).

When storing sewage sludge, it gets dryer and more aerobic over time. More aerobic sludge can increase the nitrification which increases the emissions of N_2O. Rainfall is also a contributing factor that can increase the emissions if nitrate (NO_3^-) in the top layers as runoff and it can follow the water further down in the sewage sludge pile to anaerobic zones where it denitrifies. One way of reducing the N_2O emissions when storing sewage sludge is to cover the sewage sludge, since it will not dry up as much and get as aerobic (Jönsson et al., 2015).

However, this method will likely not be approved for hygienization of sewage sludge in Sweden in the future since the Swedish Environmental Protection Agency has not listed it as an approved method for this purpose in their recommendations for sustainable recycling of P in 2013 (Naturvårdsverket, 2013).

2.2 Storing and application practice

Storing of sewage sludge can thus be seen as a hygienization method, but the main reason for why sewage sludge is stored is because of supply and demand. There are clear regulations when it comes to application of sewage sludge to agricultural fields regarding how and when it is allowed to be used as fertilizer and how it should be incorporated into the soil (SJVFS 2004:62). Immediate incorporation is known to reduce odors and emissions of NH_3 (Jönsson et al., 2015). Delayed incorporation is however also used, and in Sweden 4 hours is the longest time before the sludge needs to be incorporated according to regulations (SJVFS 2004:62). Application of sewage sludge to agricultural fields usually happens after the harvest in late summer, or in the fall, to avoid packing of the soil (Svenskt Vatten, 2013).

Since the demand of sewage sludge as a plant nutrient can differ from year to year, the storage area should have the capacity to store one annual production. Also, if there are a year with much rainfall, it can also inhibit the application of sewage sludge. This due to the increased leakage from the sludge with the rainfall (Svenskt Vatten, 2013).

The usage of sewage sludge as fertilizer is allowed for cultivation of cereals, oilseeds and energy crops. It is not allowed for the cultivation of organic food. Food companies and operators are spreading doubt about sewage sludge among farmers (Svenskt Vatten, 2013). The farmers are worried that they will not be able to sell the crops that they cultivate because of the worries of the food industry and consumers about the purity of sewage sludge and that it contains e.g. heavy metals. It adversely affects farmers as they will not be as open to use sewage sludge as they might have been if organizations and the food industry would be open to the option.

2.3 Nutrients and its emissions from agricultural soil

In order for crops to grow they need different nutrients from the soil. The nutrients can be divided in macro- and micronutrients. A macronutrient is a substance that the plants need in a larger amount to survive and thrive. The macronutrients that all plants need are N, P, sulfur, potassium and magnesium (Nationalencyklopedin, 2019a). Micronutrients, that plants need to absorb in smaller amount, are iron, boron, chlorine, copper, manganese, molybdenum and zinc (Nationalencyklopedin, 2019b). In the following subsections, N and P in soil and its pathways will be presented.

2.3.1 Nitrogen

Nitrogen (N) is an element that exits naturally in earth's crust and in the atmosphere and is of big importance for plant growing. Therefore, humans apply N to soils as mineral or organic fertilizer to maintain crop yields. But even if N is applied to agricultural fields, plants are unable to assimilate all the N available to them, as it is very reactive and mobile (Schröder, 2014).Therefore, the losses of N are important to map when performing an LCA on when sewage sludge is applied to agricultural soil. Losses of N to the air are in the forms of ammonia-N (NH_3), elementary N (N_2) and nitrous oxide (N_2O) (Schröder, 2014). IPCC claims that of total amount of N applied in sewage sludge, 1 % is emitted to air as direct N_2O emissions (IPCC, 2006). N_2O, which is a strong greenhouse gas, can be formed in two ways in the complex nitrogen cycle, both in an intermediate stage when ammonia/ammonium (NH_4^+) is oxidized to nitrite (NO_2^-) and through denitrification from NO_2^- to N_2 (Jönsson et al., 2015), see Figure 2.

The emissions of N_2O are affected by when and how the sewage sludge is applied to the soil. $NH_4^+$is naturally present in sewage sludge and can also be formed when microorganisms mineralize organic N (Figure 2). Both wind and temperature affect the emissions of NH_3. These emissions imply losses of plant-available N and can contribute both to acidification and eutrophication. There is a proposal that sewage sludge should only be allowed to be applied to agricultural fields if it is directly harrowed into the soil (Jönsson et al., 2015).

The availability of N to plants can fluctuate with several factors, as pH, precipitation, wind, temperature and soil type. Therefore, the form of N that the plants prefer to assimilate depends on the soil conditions. Generally, plants in soils with low pH absorb NH_4^+ or amino acids and plants in more aerobic soils with higher pH prefer nitrate (NO_3^-) (Masclaux-Daubresse et al., 2010). When talking about leakage of nitrogen to water, nitrogen is mainly represented by NO_3^- (see Figure 2). The N supply to agricultural fields consists mainly of nitrate and is then inevitably exposed to leakage because agricultural fields do not have active roots deep within the soil (Schröder, 2014).

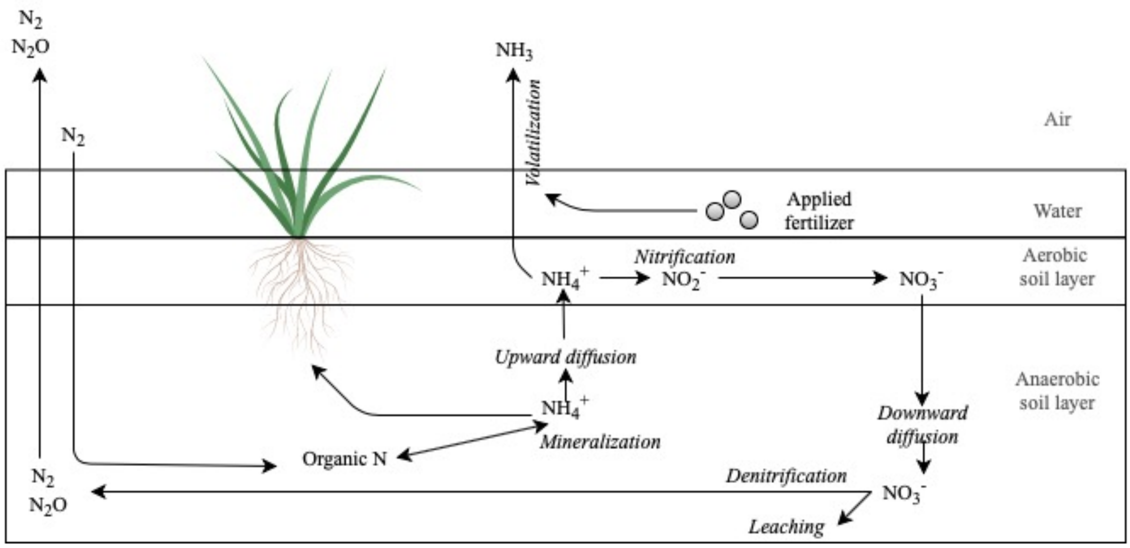

Figure 2: Pathways of nitrogen compounds through the soil and to water and air, based on Brady & Well (2013).

2.3.2 Phosphorus

Phosphorus (P) is of big importance as a nutrient in soil, and in todays industrilized agriculture mineral fertilisers is brought into the soil to increse crop yields (Butusov & Jernelöv, 2013). Removing P from wastewater can be done by either biological treatment or chemical precipitation (Linderholm, 2011). Chemical treatment is more frequently used in Sweden and uses different salts to precipitate the P, for example, ferric chloride or aluminum sulfate (Linderholm, 2011). The benefits of using the biological treatment process is that it makes the nutrients more accessible to plants (Svenskt Vatten, 2016c).

When applying nutrients in the form of mineral fertilizers to soil it is normally available to the plants at the moment it dissolves in soil water (Linderholm, 2011). The largest amount of P in the soil is not available for the plants, since it is too hard chemically bound. One part is also bound in organic form and a smaller part is available as ions stuck on soil particles (Naturvårdsverket, 2005). The crops mainly take up P as the phosphate ions $H_2PO_4^-$ and HPO_4^{2-}. In Swedish soils, $H_2PO_4^-$ dominates due to the pH which usually is around 6 (Linderholm, 2011). s

Krogstad, Sogn, Asdal, & Sæbø (2005) showed that using biological P removal, followed by lime treatment of the sewage sludge, generated the highest amounts of plant available P. It also showed that chemical treatment with Fe or Al chemicals proved to generate lower plant availability of P, but when applying sewage sludge combined with inorganic fertilizer it seemed to increase the plant available P from the sewage sludge. Delin & Nyberg (2014) showed the opposite - that a limed treated sewage sludge decreased the availability of P, which they explain by the fact that it may have been the result of the soil type used in the study.

The sorption of P in the soil can be affected by several parameters such as fertilizing, liming and tillage of the soil, which can change the plant available P (Naturvårdsverket, 2005). Also, pH has shown to be an important factor. In Sweden a pH around 6 is recommended when applying sewage sludge to agricultural fields because $H_2PO_4^-$ is dominating at that level. A common measure to increase the pH in low pH-soils is to add lime. If a soil has a too high pH on the other hand, it is difficult to restore a lower pH, and it can hinder the crop's uptake of micronutrients, such as copper, boron and manganese. Therefore, it is not always beneficial to add lime or lime-treated sewage sludge to soils (Linderholm, 2011).

Beyond erosion and surface water runoff, leaching is known to be an important pathway of P losses from agricultural fields (Börling, 2003). It is mainly a concern in drained areas. When excessively applying fertilizers to the soil, it gets saturated with P that can start leaching (Magdoff & Van Es, 2009). Leaching of P is however often limited due to its low solubility, but it can be higher depending on soil type and if the sewage sludge is not incorporated into the soil (ten Hoeve et al., 2018).

2.4 Replacement ratios

One difficulty when performing an LCA is to decide how much mineral fertilizer that is being replaced by sewage sludge (hereafter referred to as the replacement ratio), which is one of the choices that needs to be made in the LCI (section 3.2.3) if a substitution approach has been selected to handle allocation. The replaced mineral fertilizer production, that can be credited to the system, depends on the quantity of N and P in the sewage sludge that is used and effectively taken up by the plants, compared to if mineral fertilizer have been used. Heimersson, Svanström, Cederberg, & Peters (2017) argued that it is challenging to choose replacement ratios, since the availability of nutrients are depending on many factors (e.g. soil type, the climate, precipitation chemicals etc.). They mean that the true replacement ratio will be site specific due to local conditions, and it is therefore difficult to decide a generic replacement ratio.

3. Method

This chapter describes the methodology of the project and how the work has been done (section 3.1). It also contains a description of the LCA framework (section 3.2) that introduces the reader to the different parts of an LCA which is of importance for understanding the later chapters. In section 3.3 the LCA case study is defined.

3.1 Methodology

This study was done in three steps, corresponding to its three objectives (section 1.1). The first objective was to identify in what way LCIs for N and P compounds originating from the sewage sludge when it is stored and applied to agricultural fields need to be improved, and to suggest improvements. The need for improvements was partly identified based on a previous study (Heimersson et al., 2016). Improvement of the selection and quantifications of flows was mainly done through a literature study, with focus on gathering data from field-, pilot- and lab-scale studies on sewage sludge storing and application to agricultural fields. Collecting data from previous LCA studies was not prioritized, partly because such a review has already been done (Heimersson et al., 2016) and a conclusion from that work was that the data used are often the same.

The studies that were selected for this project were identified through searches using different combinations of the keywords: “sewage sludge”, “biosolids”, “amendments”, “nitrogen”, “phosphorus”, “storing”, “hygienization” and “leakage” using the Scopus database bus also Google Scholar. Depending on how specified the searches were, it generated a varied number of search results. If the searches generated far too many hits, they were narrowed down by combining with one or more other keywords. The search results were scanned by looking at the title and if the title seemed suitable the abstract was read to find studies that contain data relevant for this project. This initial selection was made to find studies that had examined sewage sludge applied to soil or storing the sewage sludge for a more thorough review. Reference tracking was also used to the extent that appropriate references was found in other studies.

Based on the inventory results, a mass balance for N was compiled to understand how much of the flows to and from the soil that is unaccounted for. A best-case and worst-case scenario was made to be able to see differences of the amount of N remaining in the soil depending emissions to air, water and crop uptake of N. The best-case scenario is when the emissions of N compounds were the lowest and the crop uptake the highest, and the worst-case scenario was if the emissions of N compounds were the highest and the crop uptake the lowest.

The second objective (section 1.1) was how to reason when choosing an LCIA-method and what differences there are between the methods that LCA-analysts need to be aware of, specifically for systems where sewage sludge is used in agriculture. A selection of LCIA methods to review was done by using the ILCD-handbook that make recommendations about which LCIA-methods that are suitable for each impact category (European Commission, 2011). In order to analyze each chosen LCIA-method for the three previously chosen impact categories (GWP, AP and EP), literature searches in Scopus database and Google Scholar were then used. Knowledge was gained about what characterization factors that exist for each method and which flows that are needed to be inventoried to get as complete LCIA results as possible. The underlying model for the LCIA methods were also looked into to understand how the characterization factors take emissions from soil into consideration.

The third objective (section 1.1) of the study was to provide a knowledge basis to be used when determining how much mineral fertilizer that sewage sludge on arable land can be assumed to replace, under different conditions. The replacement ratios used today are often similar as Heimersson et al. (2016) shown. The review study was used as a basis to map what replacement ratios that are usually used. To gain knowledge of how this is considered in practice, interviews were done with two persons. The first person was Emma Hjelm, who works at The Swedish Board of Agriculture and is part of the project "Greppa Näringen" as an expert on nutrients and the environment. "Greppa Näringen" aims to reduce the greenhouse gas emissions and eutrophication and works for the safe use of plant protection products. The second person, Kjell Ivarsson, was contacted in his role as responsible for research and development of food production at The Federation of Swedish Farmers. It is an organization for entrepreneurs in agriculture, the forestry industry, horticulture and other entrepreneurs who have their base in rural areas. The interview with Emma Hjelm was conducted only over e-mail, but the interview with Kjell Ivarsson was done by telephone.

In order to illustrate results for all three objectives of this study, an LCA case study was performed in the end. It was done to show how both different choices of inventory data, which LCIA methods and replacement ratios affect the resulting environmental impact. A detailed description of the LCA case study can be seen in section 3.3.

3.2 Life cycle assessment framework

To understand the purpose of this thesis and the case study presented in section 3.3, and because the LCA method is the reason this thesis has been conducted, the LCA framework is here presented. Both the general procedure (section 3.2.1) and the LCI and LCIA phases are presented in more detail (section 3.2.2).

3.2.1 The general procedure

The idea behind the internationally standardized tool Life Cycle Assessment (LCA) is to assess a product or service from its "cradle" to "grave", or from raw material extraction to disposal and waste management (Baumann & Tillman, 2004). An LCA consists of four parts, which is illustrated in Figure 3.

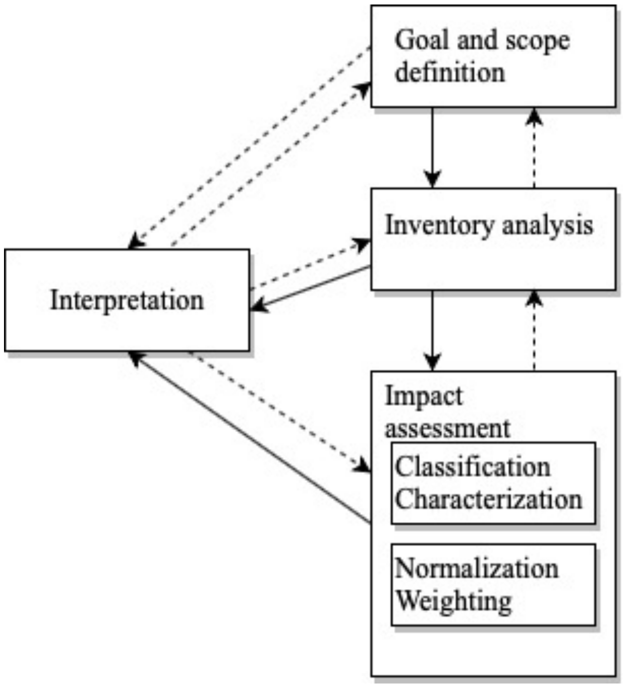

Figure 3: LCA procedure. The boxes signify procedural steps within the methodology and the arrows the order in which these are performed. The broken arrows indicate possible iterations between the steps, based on Baumann & Tillman (2004).

In the first part, the goal and scope definition, the specific product or system that is intended to be studied and the purpose of the study is decided upon (Baumann & Tillman, 2004). The LCI (phase) consists of data collection and calculations to quantify flows in to and out of the product system (Rydh, Lindahl, & Tingström, 2002). The LCI results, see section 3.2.2, state how much emissions one product or system generates and how much resources it consumes. The environmental impacts caused by the flows in to and out of the system are quantified in the third step of the LCA, the life cycle impact assessment (LCIA), see section 3.2.3 (Baumann & Tillman, 2004).

3.2.2 Life cycle inventory

The LCI phase of an LCA is the part where the data collection is done. The data collection is the most time-consuming activity during an LCA. Guidelines for the method often point out that it is important to know what type of data needs to be gathered. Data can be collected in different ways, e.g. using data from an organization or company, data from LCI databases or from previous studies in the same field. But regardless how ambitious the LCA analyst is, there will almost always be data gaps in the LCI, that will require one to make assumptions or do estimations (Baumann & Tillman, 2004). The data can also be of different quality. Primary data, which can be data from own observations from the specific system could be of benefit, but it can be hard to find. Therefore, secondary data can be used, which can be data from literature or similar studies, to complement the primary data of the specific system.

Central when making an inventory of data is to identify which inventory data that needs to be collected, for the studied system. This depends on the purpose of the study which affects the choice of impact categories chosen to be assessed in the LCA, as well as the specific method chosen for the assessment of each of the impact categories. It is also important to know how the specific methods characterize environmental impacts and what characterization factors that exist within different LCIA-methods, e.g. which flows the

LCIA method have CFs for. The advantage of using these LCIA-methods is that LCA-analysts can use them and not have to go in-depth into the LCIA procedure, since in these LCIA methods the impact assessment is packaged inside. If, for some reason, the characterization would be done without using an LCIA method it would be very time-consuming (Baumann & Tillman, 2004). One disadvantage, however, is that if the LCA-analysts do not know how the method's characterization works and which characterization factors that exist for each LCIA method it may result in inventory data missing which can lead to misleading results.

3.2.3 Life cycle impact assessment

In order to find out the total environmental impact of a product or a system, the environmental impacts need to be summed up. The calculations are often done in two steps. Firstly, it is determined what impact a consumed resource or produced pollution has on various environmental problems which is called "midpoint categories" (i.e. climate change, acidification, eutrophication, ozone depletion etc.). Secondly, the impact of the midpoint categories can be summed up to the endpoint categories of human health, ecosystem quality and finite resources.

The characterization (see Figure 3) includes that the inventory data is multiplied with pre-calculated characterization factors (CFs) which are specific for each impact category and emitted substance. LCIA is often done using ready-made LCIA methods, a few examples of methods are ReCiPe, IPCCs indicator of GWP, EDIP2003 and Ecoindicator'99. The characterization factors are therefore individual for each LCIA method. The results of the LCIA are usually presented in bar charts or other types of graphic presentations (Baumann & Tillman, 2004).

3.3 LCA case study

A life cycle case study was made to illustrate how the choice of inventory data and of LCIA methods can affect the outcome of an LCA, and how the choice of replacement ratio affects the results of an LCA. The case study was conducted in collaboration with Sara Heimersson as part of a project called "Improved life cycle assessment – the case of sewage sludge" funded by the Swedish Research Council for Environment, Agriculture Sciences and Spatial Planning.

3.3.1 Goal and Scope definition

The goal of the LCA case study was to assess the environmental impacts from a WWTP in Sweden and its transport, storage and application of sewage sludge to agriculture fields. The results are supposed to give LCA-practitioners a basis and a better understanding of how different choices within the LCA-framework affects the environmental impact when assessing a similar system to the one in this case study.

The system that was chosen as a reference system was the Gryaab WWTP (Ryaverket) in Gothenburg, Sweden (Gryaab, 2019a). Gryaab is located at the island Hisingen, and after going through the WWT the water is released into the sea. The transport, storage of sewage sludge and application of it to soil have been modeled according to how Gryaab treats and takes care of its generated sewage sludge. In reality, the sewage sludge produced at Gryaab is not only used in agriculture, but that is an assumption we do in this study. The purpose is to evaluate three scenarios when different levels of emissions are used to model the situation of when sewage sludge is stored and being applied to agricultural fields and see how the environmental impact is affected by these different choices. Also, a second purpose are to see the difference in environmental impact of different LCIA-methods when they are used to assess these systems.

For the three modeled scenarios, only the processes of storage and agricultural use of sewage sludge and replacement ratios of mineral fertilizers were changed. The rest of the processes in the system were modeled in the same way in the scenarios. The three scenarios were:

- *Low scenario:* Emissions of N and P compounds from storage and application of sewage sludge to agricultural fields were the lowest found in the inventory review (see Tables 2 to 7). The replacement ratios of mineral fertilizers were also modeled at their lowest.
- *Sludge specific scenario:* Here, inventory data were chosen to suit the sludge produced at Ryaverket, e.g. mesophilically digested sludge, and emissions of storage of mesophilically digested sludge with no cover. The replacement ratios for mineral fertilizers were chosen to be commonly used numbers from previous studies (see review by Heimersson et al., 2016).
- *High scenario:* Emissions of N and P compounds from storage and application of sewage sludge to agricultural fields were the highest found in the inventory review (see Tables 2 to 7). The replacement ratios of mineral fertilizers were also modeled at their highest.

The studied system (Figure 4) encompassed the WWTP, biogas production, the storage and use of sewage sludge and transport of sewage sludge between the WWTP, the storage site and the agricultural field. The production and construction of facilities, equipment and vehicles used in the system was not included in the study. The study considered the environmental impacts from direct emissions and production of chemicals and energy used in the system and also the benefits of resource recovery (biogas that replaces the production of natural gas and sewage sludge that replaces the use and production of mineral fertilizers). These benefits were handled through substitution, which is the most common way used in previous similar studies (Heimersson et al., 2017). What happens is that the system was credited for alternative ways of generating an equal function. In order to be able to compare the use of resources and emissions it was decided upon to use an annual reference flow, e.g. all flows were calculated per year and the functional unit was the yearly inflow to Ryaverket.

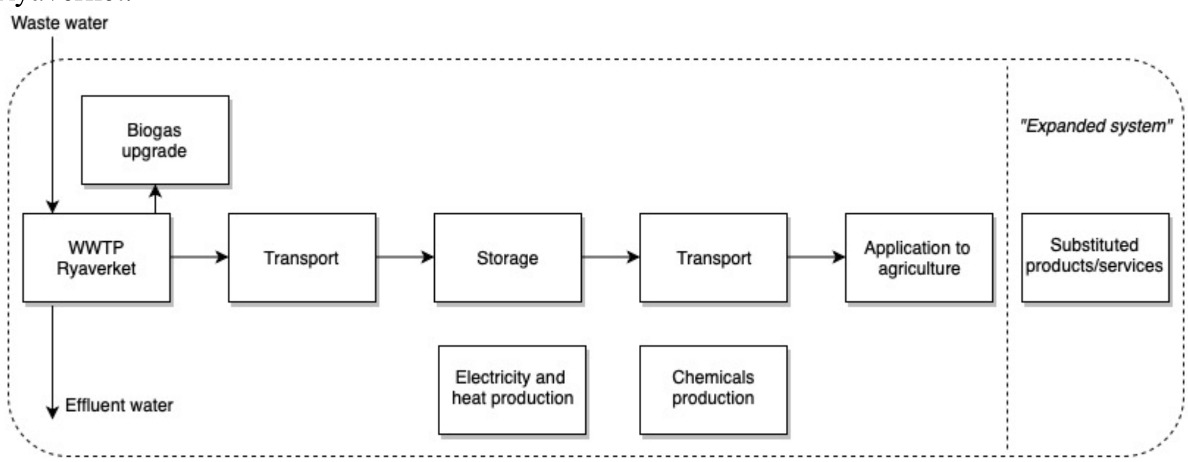

Figure 4: Flowchart of the studied system.

The environmental impact was assessed for the following impact categories: climate change, acidification and eutrophication. In order to characterize the inventory data for each impact category, a selection of LCIA-methods have been made using the ILCD-handbook. Which LCIA methods chosen for each impact category can be seen in Table 1. For handling the inventory and impact assessment, the GaBi software tool was applied (www.gabi-software.com).

Table 1: The three chosen impact categories and the corresponding LCIA methods selected.

	LCIA methods
Climate change	IPCC indicator Global Warming Potential (IPCC, 2006)
Acidification	Accumulated Exceedence (Posch et al., 2008) ReCiPe (Netherlands National Institute for Public Health and the Environment, 2018)
Eutrophication terrestrial	Accumulated Exceedence (Posch et al., 2008) EDIP2003 (Potting & Hauschild, 2005)
Eutrophication aquatic (freshwater)	ReCiPe (Netherlands National Institute for Public Health and the Environment, 2018)
Eutrophication aquatic (marine)	EDIP2003 (Potting & Hauschild, 2005)

3.3.2 Data quality requirements and life cycle inventory

In this section, the data requirements for the system and the three scenarios will be described. As mentioned above, the WWTP Gryaab (Ryaverket) was chosen as a reference system and data have been selected based on it. In Appendix I, the inventory of data is presented.

The following processes presented are the same from an inventory point of view for the three scenarios:

- *WWTP:* The WWTP Gryaab (Ryaverket) in Gothenburg was chosen as a reference system and specific data from the WWTP was gathered from their annual environmental report (Gryaab, 2019b). There was information about emissions to both air and water, energy requirements, district heating and quantities of used chemicals. The electricity production was modeled as Swedish energy mix, with data from Gabi Professional database. The district heating required and used was also modeled using Gabi Professional database and was chosen to be of EU conditions and not of Swedish district heating. The chemicals used, and their production have been modeled using Gabi Professional database. The production of polymer has been omitted due to time constraints in the LCI.
- *Biogas upgrading:* The biogas generated at the WWTP is upgraded, since vehicle gas must contain at least 97% CH_4 (i.e CO_2 and other substances need to be removed) by Göteborgs Energi. Data about energy consumption during biogas upgrading has been collected from Palm & Ek (2010), and electricity production was modeled as Swedish energy mix, with data from Gabi Professional database.
- *Transport:* For modeling the transportation of sewage sludge for all three scenarios it was illustrated as if the sewage sludge was transported from Ryaverket to Kuskatorpet's storage facility near Halmstad in the south of Sweden. The distance was measured to be 150 km using Google maps and the transport was done by truck and trailer. The distance from the storage to when the sewage sludge is applied to agricultural fields has been

assumed to be 50 km, and this transport was also done by truck and trailer. The truck and trailer transport and diesel production have been modeled using Gabi Professional database. Other transport options within the background systems (e.g. chemicals production, biogas upgrading and transport of mineral fertilizers) have not been included in the study.

The following processes were modeled by three scenarios and data was collected from the inventory review described in section 4.1. The processes were:

- *Sludge storage:* Data of direct emissions from the sewage sludge during storage was gathered from Jönsson et al. (2015). In the high and low scenarios, no account was taken of how sludge was treated at the WWTP or how it was stored (i.e., with or without cover) but only the highest and lowest value of emissions was chosen. For the sludge specific scenario, a mesophilically digested sludge was selected that was stored without no cover.
- *Agriculture:* Emission data was collected from studies made of sewage sludge applied to soil. For the high and low scenarios no consideration was taken into account of how the sludge had been treated; the choice was made only on the basis of which sludge had the highest and lowest emissions. For the sludge specific scenario, data was selected to fit the sludge generated at Ryaverket.
- *Mineral fertilizer replacement:* replacement ratios of how much of the mineral fertilizer that is being replaced by sewage sludge have been chosen from an earlier review (Heimersson et al., 2016). For the low scenario, the lowest number was used; for the high scenario, the highest number was used (100 % for both N and P) and for the sludge specific scenario, commonly used numbers were chosen. The mineral fertilizer production was modeled as calcium ammonium nitrate (27% N) fertilizers production and triple superphosphate (46% P_2O_5) fertilizers production using Gabi Professional database.

4. Results

The literature study for improving the inventory from sewage sludge storage and application on agricultural field resulted in inventory tables of relevant N and P flows to air and water (section 4.1) and a mass balance of N flows (section 4.2). The effort to improve the inventory for accounting for the use of byproducts (nutrients) from the studied system provides a knowledge basis useful for future LCA analysts in guiding their selection of replacement ratios (section 4.3). The comparison of LCIA-methods for characterizing N and P flows highlights differences between available methods (section 4.4). At last the LCA case study illustrates the importance of making an inventory that is specific for the particular system under study and the choice of LCIA method (section 4.5).

4.1 Inventory of nitrogen and phosphorus flows

The inventory results are described and presented in section 4.1.1 to 4.1.3 for the compartments air, leakage to water and crop uptake.

4.1.1 Emissions to air

The results from the inventory of N emissions to air is presented in Tables 2 and 3. The most important compounds, for the assessed impact categories (GWP, AP and EP), originating from sewage sludge on agricultural fields is N_2O and NH_3. In Table 2, data from four different earlier studies are presented, and it shows for both N_2O and NH_3 that different parameters such as soil type, type of sewage sludge and how the study has been performed affects the magnitude of emissions.

Jönsson et al. (2015) made a field study of both N_2O emissions and NH_3 emissions. It showed that mesophilically digested dewatered sewage sludge that is directly incorporated in the soil generate higher N_2O emissions than sludge incorporated into soil four hours later. On the other hand, immediate incorporation of sludge that had been urea treated and stored for five months generated lower emissions of NH_3 and the emissions were higher if the same sludge was not incorporated directly. Yoshida et al. (2015) made a pilot study of differently treated sewage sludges applied on one type of soil. They showed that dewatered digested sewage sludge generated the highest level of N_2O emissions and reed bed sludge the lowest. A third study also applied reed bed sludge on soil to measure N_2O emissions (Nielsen et al., 2017). Compared to the other studies, this study showed that reed bed sludge generated relatively low N_2O emissions. Heimersson et al. (2016) reviewed how major flows of N and P from wastewater and sludge management has been quantified in earlier LCAs. Two common references, IPCC (2006) and Hobson (1999), stated that N_2O emissions when applying sewage sludge on agricultural soil generated emissions of 1 % of added N. The inventory review of N_2O can be seen in Table 2. Table 2 also lists the inventory results for NH_3 emissions from sludge on agricultural soil.

In Table 3, the inventory result of N_2O emissions from sewage sludge storing are presented. The study is a pilot study done in Sweden, that measured N_2O emissions of mesophilically and thermophilically anaerobically digested sewage sludge with and without cover. It shows that the N_2O emissions can vary depending on how the sludge has been treated and if it has been stored with or without cover, since in this study the soil properties were the same for all four scenarios (Jönsson, Rodhe, Junestedt, Willén, & Pell, 2016; Jönsson et al., 2015).

Table 2: Review of studies done of N_2O and NH_3 emissions to air when sludge has been applied to agricultural soil. TN stands for total nitrogen.

Origin of emission	Flow	Reference		Type of sludge	Soil type	Other comment
Emissions to air						
Sludge applied on agricultural soil	**N_2O**	Jönsson et al. (2015)	0.34 % of TN added.	Mesophilically digested SS dewatered and stored for 1 year.	Silty clay (44 % clay, 46 % silt, 20 % sand).	Measurements of N_2O emissions in an arable field experiment in Uppsala, Sweden. The emissions were measured for 45 days in the autumn. With delayed incorporation of SS.
			0.71 % of TN added.	Mesophilically digested SS dewatered and treated with urea.		Measurements of N_2O emissions in an arable field experiment in Uppsala, Sweden. The emissions were measured for 45 days in the autumn. With immediate incorporation of SS.
		Yoshida et al. (2015)	0.18 % of TN added.	Primary sludge.	Sandy clay loam (14 % clay, 20 % silt, 66% sand)	A pilot study, in Denmark, with a 190-days long incubation with different sludge types collected from different WWTPs that were using different treatment methods.
			0.66 % of TN added.	Dewatered digested sludge.		
			0.37 % of TN added.	Dried digested sludge.		
			0.28 % of TN added.	Reed bed sludge (20 cm core).		
			0.10 % of TN added.	Reed bed sludge (80 cm core).		
			0.50 % of TN added.	Limed sludge.		
		Nielsen et al. (2017)	0.22 % of TN added.	Reed bed sludge (core at 100 cm depth).	Sandy loam.	Pilot study of reed bed stabilized sludge in Denmark with residue sludge from three different STRBs in Denmark. Incubation for 160 days.
			0.17 % of TN added.	Reed bed sludge (core at 100 cm depth).		
			0.13 % of TN added.	Reed bed sludge (40 cm core).		

NH_3	Jönsson et al. (2015)	12.2 % of TN added. 10.5 % of TN added.	Mesophilically digested and dewatered sludge, urea treated and stored five months.	Muddy clay (44 % clay, 46 % silt, 20 % sand).	Field study during the spring for 67 days. Delayed incorporation of SS into the soil. Immediate incorporation of SS into the soil.
	Mendoza, Assadian, & Lindemann (2006)	171 mg N (no urea) 1916 mg N (with urea). 349 mg N (no urea) 2350 mg N (with urea).	Anaerobically digested. Limed sludge.	Bluepoint loamy sand.	Pilot study of a closed soil column system of both anaerobically digested SS and lime-stabilized SS irrigated with and without urea.

Table 3: Review of studies done of emissions of nitrous oxide to air from sewage sludge during storage.

Origin of emission	Flow	Reference		Type of sludge	Other comment
Emissions to air					
Storage	**N_2O**	Willén, Rodhe, Pell, & Jönsson (2016)	0.34 % of TN added.	Mesophilically digested SS, no cover.	In a pilot study, measurements of N_2O emissions when storing digested SS were done in four different sets.
			0.19 % of TN added.	Mesophilically digested SS, with cover.	
			0 % of TN added.	Mesophilically digested and treated with ammonia SS, with cover.	
			1.30 % of TN added.	Thermophilically digested SS, with cover.	

4.1.2 Leakage to water

Studies that have explored emissions of N and P compounds that leak into the surrounding water is presented in Tables 4 and 5. Leakage of N, represented by NO_3^-, differs in the two presented studies. Shepherd (1996) and Esteller, Martínez-Valdés, Garrido, & Uribe (2009) used a loamy medium sand respectively a clay loam, and the results differed quite much. The leakage of NO_3 in the loamy medium sand were almost four times higher than in the clay loam. Leakage of P however, were only found measured by Esteller et al. (2009) and were much lower than the N leakage.

4.1.3 Crop uptake

Fertilizers are applied to soil to nourish the crops in order to obtain higher crop yields. It is interesting to measure how much of the N and P from the sludge that the crops take up to know how to dose application. Review results can be seen in Tables 6 and 7. Esteller et al. (2009) used aerobically digested sludge on a clay loam with high porosity and it generated a crop uptake of 1.73 % of total applied N. The uptake of P in the same study was 0.35 % of total added P. A study done over a time period of 30 years has been conducted in the southern of Sweden, to investigate soil fertility effects when repeatedly applying sewage sludge during several years. It showed a crop uptake of N that could differ between 3-8% and for P 20 % (Börjesson & Kätterer, 2018, 2019). None of the studies discussed the time perspective of the crop uptake.

According to Maguire, Sims, Dentel, Coale, & Mah (2010), P in sewage sludge is said to have different availability to the crops depending on the precipitation chemical when using chemical treatment of wastewater. A study made in Sweden points out that it is also important how the chemical treatment is done and how much of the chemical that is used. The study showed that when using less of the precipitation chemical, the fertilizing effect became higher than in previous studies when larger amounts of chemical had likely been used (Delin & Nyberg, 2014).

Table 4: Review of studies done on leakage of N when sewage sludge has been applied to agricultural soil.

Origin of emission	Flow	Reference		Type of sludge	Soil type	Other comment
Emissions to water						
Sludge applied to soil	**NO_3-N**	Shepherd, (1996)	39 % of applied N	Digested liquid sludge, surface spread.	Loamy medium sand. Topsoil containing 80 % sand, 15 % silt, 5 % clay and 1.6 % organic matter.	Annual field experiment in central England measuring the effects of nitrate leaching depending on sludge type, application time, application method and two different crops. Surface spread digested liquid sludge.
		Esteller et al. (2009)	11.43 % (year 1) of N applied. 11.94 % of N applied.	Aerobically digested SS.	Clay loam, high porosity.	A two-year field experiment in a corn field in Mexico to establish possible N leaching under real field conditions. Calculated [kg N ton^{-1} applied sewage sludge] is a mean value of the two-year experiment.

Table 5: Review of studies on leakage of P when sludge is applied to soil.

Origin of emission	Flow	Reference		Type of sludge	Soil type	Other comment
Emissions to water						
Sludge applied to soil	**Tot-P**	Esteller et al. (2009)	0.23 % (year 1) of P applied. 0.18 % of P applied.	Aerobically digested SS.	Clay loam, high porosity.	A two-year field experiment in a corn field in Mexico to establish possible P leaching under real field conditions. Calculated [kg P ton^{-1} applied sewage sludge] is a mean value of the two-year experiment.

Table 6: Review of studies on crop uptake of N when sewage sludge is applied to soil.

Origin of emission	Flow	Reference		Type of sludge	Soil type	Other comment
Crop uptake						
Sludge applied to soil	**Tot-N**	Esteller et al. (2009)	1.73 % of applied	Aerobically digested SS.	Clay loam, high porosity.	A two-year field experiment in a corn field in Mexico to establish possible N leakage under real field conditions.
		Börjesson & Kätterer (2018)	3-8 % of TN added.	Anaerobically digested and dewatered.	Sandy loam (14 % clay, 56 % sand) and loam (26 % clay, 44 % sand).	A 30-year field experiment in Sweden at two locations where effects and soil fertility are investigated when applying sewage sludge on agricultural fields.

Table 7: Review of studies on crop uptake of P when sewage sludge is applied to soil.

Origin of emission	Flow	Reference		Type of sludge	Soil type	Other comment
Crop uptake						
Sludge applied to soil	**Tot-P**	Esteller et al. (2009)	0.35 % of applied	Aerobically digested SS.	Clay loam, high porosity.	A two-year field experiment in a corn field in Mexico to establish possible P leaching under real field conditions.
		Börjesson & Kätterer (2019)	20 %	Anaerobically digested and dewatered SS.	Sandy loam (14 % clay, 56 % sand) and loam (26 % clay, 44 % sand).	A 30-year-old field experiment in Sweden at two locations where effects and soil fertility were investigated when applying sewage sludge on agricultural fields.

4.2 Mass balance of nitrogen flows

A mass balance of the flows of N were done based on the inventory results in Tables 2-7, for both a best-case scenario i.e. when emissions were the smallest and a worst case when emissions were at their highest. The system of the mass balance can be seen in Figure 5.

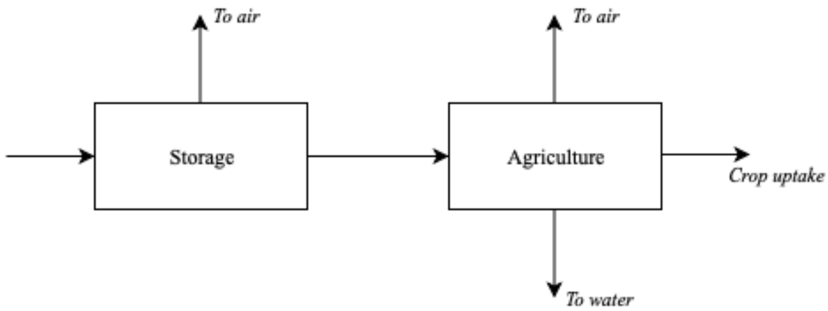

Figure 5: The system of the modeled mass balance of N.

The best-case scenario is where the crop uptake is the highest and the emissions to air and water are at their lowest. The worst-case scenario is the opposite, were the crop uptake is at its lowest and emissions to air and water are highest. Here, no account has been taken of different soil and sludge types, nor if the studies were in a pilot lab scale or as an actual large-scale field study. The results of the mass balance can be seen in Table 8 and which studies that were used in the two scenarios can be seen in Appendix II. The numbers of the best- and worst-case scenarios are not the same as the high and low scenarios in the LCA case study (see section 3.3).

Table 8: Results of a mass balance of N from the sludge generated from the WWTP, to storage and sewage sludge to final application to agricultural fields.

	Total Nitrogen					
	From WWTP	**Emissions from storage**	**Emissions from agricultural fields**			
	Input [%]	To air [% of added TN]	To air [% of added TN]	Leakage [% of added TN]	Crop uptake [% of added TN]	Unaccounted [% of added TN]
Best case	100	0	9.3	11.7	8	71
Worst case	100	1.3	39.6	39	1.7	18.4

4.3 Replacement ratios for mineral fertilizers

In order to map how practitioners in this field and farmers in real life judge the fertilizing effect of sewage sludge, compared to mineral fertilizers, interviews were done. The first person that was interviewed was Emma Hjelm, nutrient and environmental expert at The Swedish Board of Agriculture. She is also a member of the project "Greppa Näringen" (see section 3.1). The second person was Kjell Ivarsson, responsible for research and development of food production at The Federation of Swedish Farmers. The interviews were based on the questions in Appendix III.

In many respects, sewage sludge is similar of manure, but does not contain as much NH_4^+ (Börling, Kvarmo, Listh, Malgeryd, & Stenberg, 2018). According to Emma Hjelm

(Personal contact, 9 May 2019), the N effect is usually judged to be equal to the NH_4^+ content in the sludge, which is an assumption also made for manure. And, as for manure and other organic amendments, the availability of nutrients in sludge and how much the crop can benefit depends on the type of crop, the time of the year the sludge is spread on land, the practice for incorporation of the sludge into the soil and the dry matter of the sludge. For P, Swedish law limits the supply to 22 kg P/ha on average at the spreading area during a period of five years (Jordbruksverket, n.d.). Emma Hjelm believes that in practice it is expected that, in the long run, 100 % of the P in sludge will be plant available. For sewage sludge, it is not really relevant for the farmer to know the exact P effect in the short term when one expects that enough of the sludge input will benefit the crop during the first year anyway. On the contrary, Kjell Ivarsson (Personal Contact, 24 May 2019), believed that there is no big difference in the quality of N and P in sewage sludge and in other fertilizer products (e.g. mineral fertilizers or manure). Kjell Ivarsson argued that it is more important to focus on the properties of the soil to determine how accessible the nutrients will be. He also pointed out that precipitation chemicals used in the WWTP affect the P for a short while, but when harrowing it into the soil, the soil properties dominate the fate of P. It is the chemistry and pH in the soil that decides how much of the nutrients that the plants can benefit from.

Emma Hjelm believed that the view on using sewage sludge as a fertilizer in agriculture in Sweden varies greatly between farmers. Users of sludge value the P and N supplements, but also the organic material. Emma Hjelm thinks that the big challenge for the farmers who use sludge are to find purchasers who want to receive the cultivated goods (Personal contact, 9 May 2019). Kjell Ivarsson claimed that the farmer does not have a free choice of using sewage sludge or not. It is more controlled by what crops that are being cultivated and that it is not allowed to use sewage sludge for direct food production to humans or for organic food production in Sweden. Farmers are also cautious to not become limited in the future to the possibility of varying what is cultivated in the arable land. He also claimed that there is not one special type of sludge that is preferred when using it on arable land. It is more affected by the distance from where the sludge is produced and stored, to final use in agriculture. It is therefore more important that the sludge is nearby and there seems not to be any consideration of what sludge type that is used (Personal Contact, 24 May 2019).

The regional differences in usage of sewage sludge in agriculture in Sweden is likely, according to Emma Hjelm, explained by the varying soil type. Sewage sludge is often wanted in high yielding crop farms with soil with low soil organic content, often with a deficit of P in the crop rotation. In soils with a low humus content, the sludge has a positive effect as it, in addition to nutrients, adds organic matter which increases or maintains the organic content and obtain the harvest (Personal contact, 9 May 2019). Kjell Ivarsson explained the regional differences of using sewage sludge as fertilizer in Sweden as a consequence of the different crops grown in different regions and the accessibility of manure. In western Sweden, farmers are skeptical. Kjell Ivarsson believes that this is due to that there is more cultivation of food there, and relatively many animal farms which generate manure, compared to in other regions. In eastern Sweden, the Stockholm region in particular, there is a more positive attitude to using sewage sludge as a fertilizer, according to Kjell Ivarsson. Kjell Ivarsson points out that there is a lot of sludge available in that region, but few animal farms which means low availability of manure. In the region, the cultivation is more focused on cereal cultivation for transport fuels (e.g. cereals for the ethanol factory in Norrköping), which makes sewage sludge a good fertilizing option (Personal Contact, 24 May 2019).

4.4 Life cycle impact assessment methods

The LCIA, which is the third part of the LCA-methodology as can be seen in Figure 3, focuses on describing the environmental consequences and thus the impacts that can be calculated from the inventory results (Baumann & Tillman, 2004).

4.4.1 Climate change

For assessing climate change, there has become a broad consensus within the LCA community to use the IPCC's GWP's for characterization at midpoint level, and due to that, only this method was selected as a representative method. The method has three different versions with different timeframes; 20, 100 and 500 years. The recommendation is that the method with the 100-year timeframe shall be used as a standard, and the shorter (20 years) and longer (500 years) timeframes can be used as a sensitivity analysis (European Commission, 2011). They also point out that this check is of particular importance when assessing systems within the agricultural sector, since N_2O emissions have a long lifetime (European Commission, 2011). The characterization factors according to IPCC from 2007 and 2014 is presented in Table 9.

Table 9: Characterization factors for global warming potential (GWP) for nitrous oxide, according to IPCC, both Assessment Report 4 (AR4) (Forster et al., 2007) *and Assessment Report 5 (AR5)* (IPCC, 2014).

		20 years [kg $CO_{2\ eq}$/kg]	**100 years [kg $CO_{2\ eq}$/kg]**	**500 years [kg $CO_{2\ eq}$/kg]**
AR4	**N_2O**	289	298	153
AR5	**N_2O**	264	265	NA[1]

4.4.2 Acidification

For LCA applications, acidification is known as one of the most commonly used impact categories. However, emissions with acidifying potential can cause different effects depending on the emission point's location, which makes development of general characterization factors difficult (Seppälä, Posch, Johansson, & Hettelingh, 2018). The ILCD handbook have, however, recommended the method of Accumulated Exceedence for evaluating acidification at midpoint level (European Commission, 2011), and that the updated characterization factors of Posch et al. (2008) should be used. They have developed country-specific characterization factors (eq/kg) by using an atmospheric model but also soil fate factors. They state however, that Accumulated Exceedence should not be used as a midpoint indicator outside of Europe because there is a lack of a fitting atmospheric dispersion model (Posch et al., 2008). The characterization factors for Sweden for acidification according to Accumulated Exceedence is presented in Table 10.

Table 10: Characterization factors for acidification for Sweden according to Accumulated Exceedence (Posch et al., 2008).

	CF [H^+]
NO_2	1.3
NH_3	7.2

[1] Not available.

One other midpoint indicator that the ILCD handbook has evaluated (but not recommend) is ReCiPe. It is chosen here as an alternative to AE since it is a commonly used LCIA method. It focuses on terrestrial acidification, and the ILCD handbook points out that in order to be a suitable LCIA method for different continents, the characterization factors need to be extended to other ecosystems than forests (European Commission, 2011). This could possibly make this method more limited, than if a method takes more ecosystem into account. The terrestrial acidification CFs for Europe is presented in Table 11.

Table 11: Characterization factors for acidification according to ReCiPe (Goedkoop et al., 2008).

	20 years [kg $SO_2^-{}_{eq}$/kg]	**50 years [kg $SO_2^-{}_{eq}$/kg]**	**100 years [kg $SO_2^-{}_{eq}$/kg]**	**500 years [kg $SO_2^-{}_{eq}$/kg]**
NO_X (to air)	0.49	0.52	0.56	0.71
NH_3 (to air)	1.99	2.23	2.45	2.89

4.4.3 Eutrophication

The phenomenon of eutrophication can influence both aquatic and terrestrial ecosystems (Baumann & Tillman, 2004). Therefore, ILCD have evaluated midpoint indicators for both aquatic and terrestrial eutrophication (European Commission, 2011). However, ecosystems in different locations are limited by nutrients in their own way, and characterization factors are therefore hard to develop that account for eutrophication in different geographic areas (Baumann & Tillman, 2004).

For terrestrial eutrophication the ILCD handbook recommends Accumulated Exceedence as midpoint indicator. It contains, as for acidification, both atmospheric and soil fate factors, which makes this method a good choice in order to contain consistency between both terrestrial acidification and terrestrial eutrophication, see Table 12 (European Commission, 2011).

Table 12: Characterization factors for terrestrial eutrophication in Sweden according to Accumulated Exceedence (Posch et al., 2008).

	CF [H^+]
NO_2	3.5
NH_3	7.3

EDIP2003 also tackles terrestrial eutrophication for emissions to air. The model itself is based on European conditions, and its characterization factors can be seen in Table 13.

Table 13: Characterization factors of EDIP2003 for terrestrial eutrophication (Potting & Hauschild, 2005).

	CF [0.01 m^2 unprotected ecosystem/g]
NO_2	2.54
NO	3.88
HNO_3	1.85
NH_3	10.10

The majority of the characterization models for the aquatic eutrophication have in common that the modelling of the fate is rather week and it also ignores some of the most vital removal pathways for N and P (European Commission, 2011). One of the models that were considered the best by the ILCD handbook are ReCiPe that uses a more up-to-date model for the atmospheric fate (European Commission, 2011). The characterization factors for marine eutrophication and freshwater eutrophication is presented in Tables 14 and 15. ReCiPe is the only evaluated method that have special CFs for emissions to agricultural soils. The CFs are calculated based on the assumption that 10 % of all P is moved from the soil to surface waters (Huijbregts et al., 2016).

Table 14: Characterization factors of ReCiPe for marine eutrophication (Netherlands National Institute for Public Health and the Environment, 2018). *Emissions are expressed as according to the reference.*

	Emission compartment		
	Freshwater	**Agricultural soil**	**Seawater**
N	0.30	0.13	1.00
NH_4^+	0.23	0.10	0.78
NH_3	0.24	0.10	0.82
NO	0.14	0.06	0.47
NO_2	0.09	0.04	0.30
NO_3	0.07	0.03	0.23
NO_x	0.09	0.04	0.30

Table 15: Characterization factors of ReCiPe for freshwater eutrophication (Netherlands National Institute for Public Health and the Environment, 2018). *Emissions are expressed as according to the reference.*

	Emission compartment		
	Freshwater	**Agricultural soil**	**Seawater**
Phosphorus (P)	1.00	0.100	0
Phosphate (PO_4^{3-})	0.33	0.033	0
Phosphoric acid	0.32	0.032	0
Phosphorus pentoxide	0.22	0.022	0

The strength of EDIP2003 is that it accounts for both terrestrial and aquatic eutrophication. However, the central scientific criteria suffer from a weaker performance. EDIP2003 is based on the CARMEN model, which is used to express "*the fraction of a nutrient emission from agricultural soil or wastewater treatment plant that will reach and expose inland waters or marine waters*". It is common in LCA practice to consider the topsoil compartment in agriculture to be part of the technosphere. The LCI data for N and P supply to agriculture soil therefore often insinuate to the amount of N and P that leaves the topsoil after the crop uptake and binding into the soil (Potting & Hauschild, 2005). If that number is not known Potting & Hauschild (2005) provide factors that can be used to estimate the runoff of N and P from agricultural soil if the fertilizer amount is known. The characterization factors for EDIP2003 is presented in Table 16.

Table 16: Characterization factors for EDIP2003 both for emissions to inland and marine waters (Potting & Hauschild, 2005).

		Factor
Emission to inland waters	**P-Agriculture** (*)	0.06
	P-Waste Water (**)	0.88
	N-Agriculture (*)	0.53
	N-Waste Water (**)	0.59
Emission to marine waters	**P-Agriculture** (*)	0.06
	P-Waste Water (**)	1.00
	N-Agriculture (*)	0.54
	N-Waste Water (**)	0.70

(*) – The factors relate to emissions of nutrients after plant uptake.
(**) – The emission factors for waste water express the share of emissions that is released directly to the marine water or indirectly through rivers (according to a European average situation).

4.5 LCA case study

In Figures 7 to 14, the results of the LCA case study is shown in bar charts for the three impact categories using the different LCIA-methods chosen. Note that it is only the sludge storing, application on agricultural fields and the credited mineral fertilizer production that changes between the scenario results. The other modeled parts of the systems are the same in all scenarios. The system is presented in Figure 4, and with details in Figure 6. The boxes in Figure 6 is color-coded to facilitate the understanding of the legends in the graphs in Figures 7 to 14.

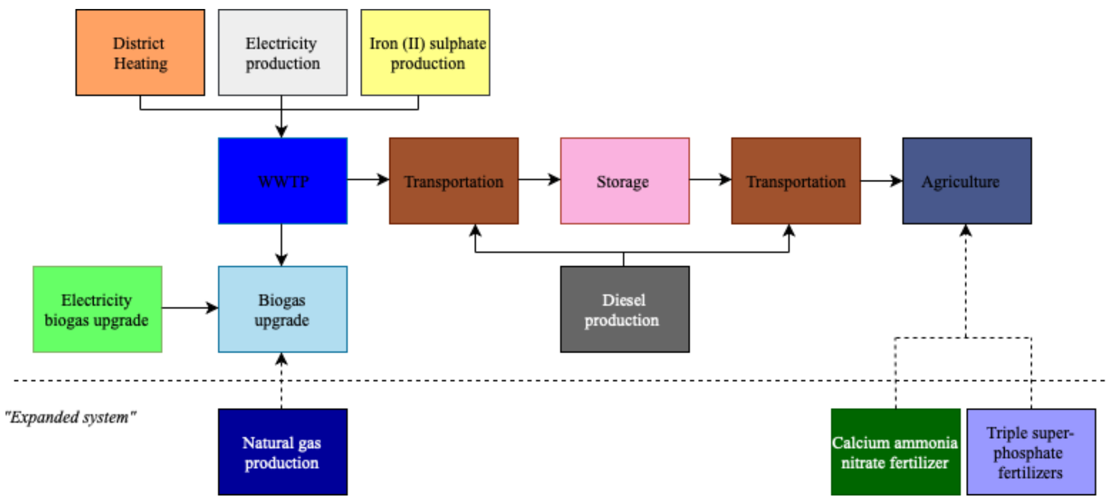

Figure 6: System of LCA case study. Choices of LCI can be seen in section 3.3.2 and in Appendix I.

In Figure 7 the results of GWP is presented. The WWTP, district heating and sewage sludge storing is shown to have a great environmental impact. The natural gas production and the mineral fertilizer production, which is credited to the system due to biogas production and use of sewage sludge as fertilizer, have a significant impact on the results. In Figures 8 and 9 the AP results are presented, using different LCIA methods. Both sewage sludge storage and application to agricultural fields were shown to have a great impact, regardless of the LCIA method applied. The case study results regarding EP is presented in Figures 10 to 14. For terrestrial EP, the impacts of sewage sludge storage and application to agricultural fields was shown to dominate the environmental impact, especially in the sludge specific and high scenarios. The credit of the mineral fertilizer production was not substantial (Figures 10 and

11). For aquatic EP, on the other hand, the application to agricultural fields showed a significant impact for freshwaters according to ReCiPe and to aquatic EP using EDIP2003. For the marine EP according to ReCiPe, the application to agricultural fields also had an important impact but there it was the WWTP that had the dominating environmental impact (Figures 12 to 14).

The sewage sludge storing can be seen to have a significant impact for both GWP, AP and terrestrial EP which contradicts the fact that previous LCAs have not accounted for emissions from sewage sludge storage at all (Heimersson et al., 2016). The sludge application, which is usually accounted for in previous LCA studies (Heimersson et al., 2016), showed to have a substantial impact for AP, and both aquatic and terrestrial EP. The three scenarios showed, for all LCIA methods, that there are clear differences when using different levels of emissions.

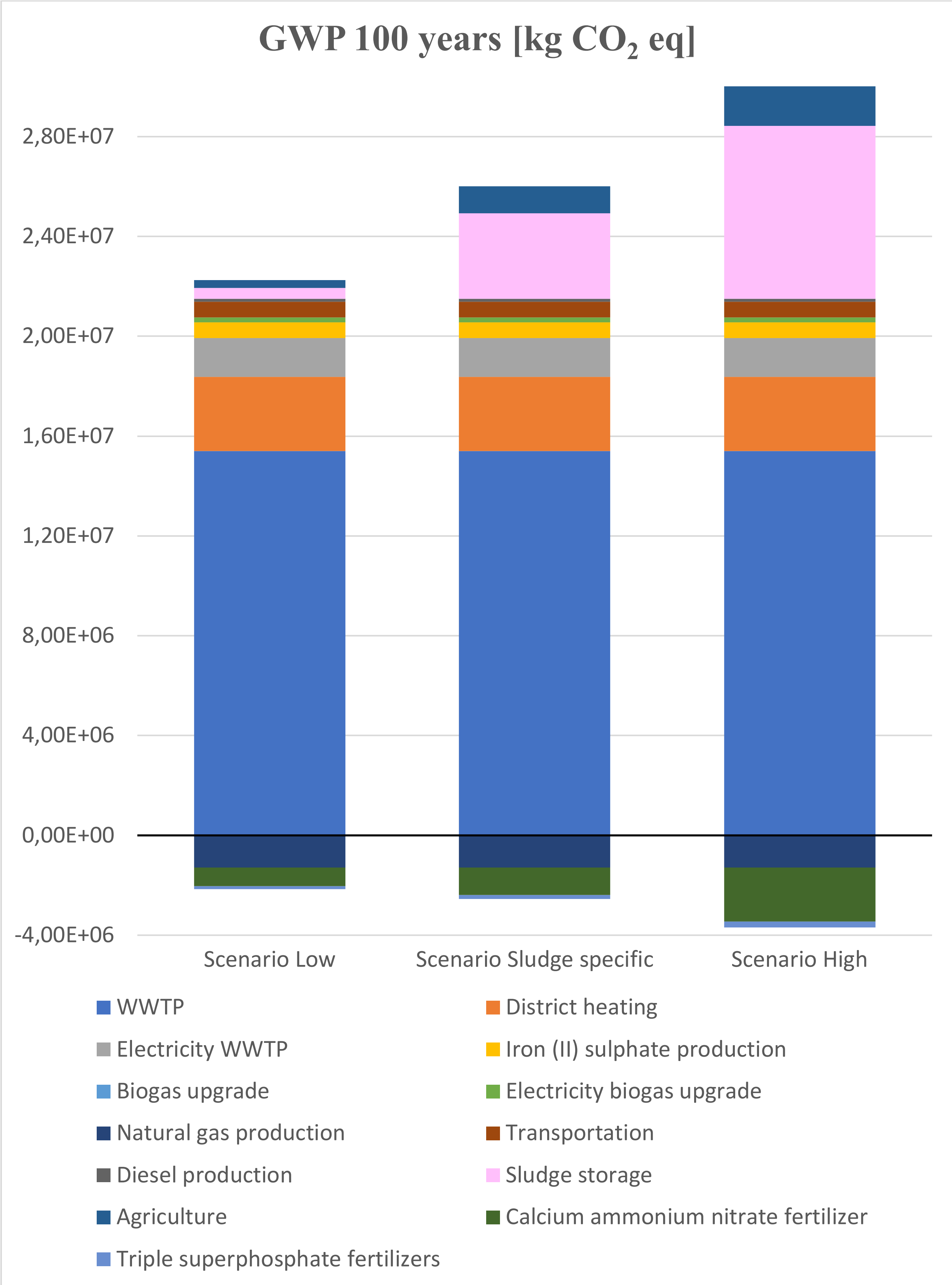

Figure 7: The LCIA result of the global warming potential according to IPCC, in a 100-year perspective, per yearly inflow to Ryaverket.

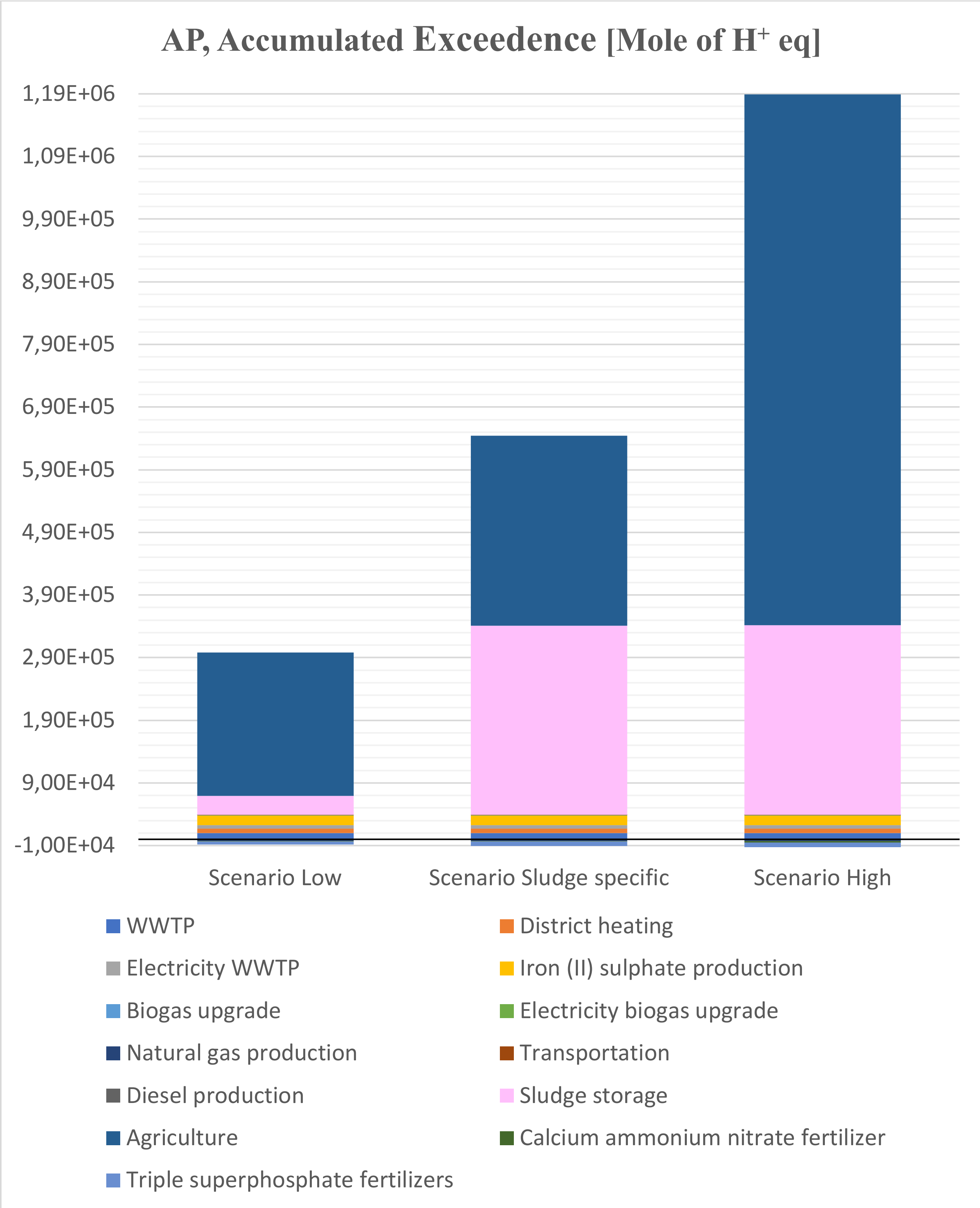

Figure 8: The LCIA results of acidification potential according to accumulated exceedence, per yearly inflow to Ryaverket.

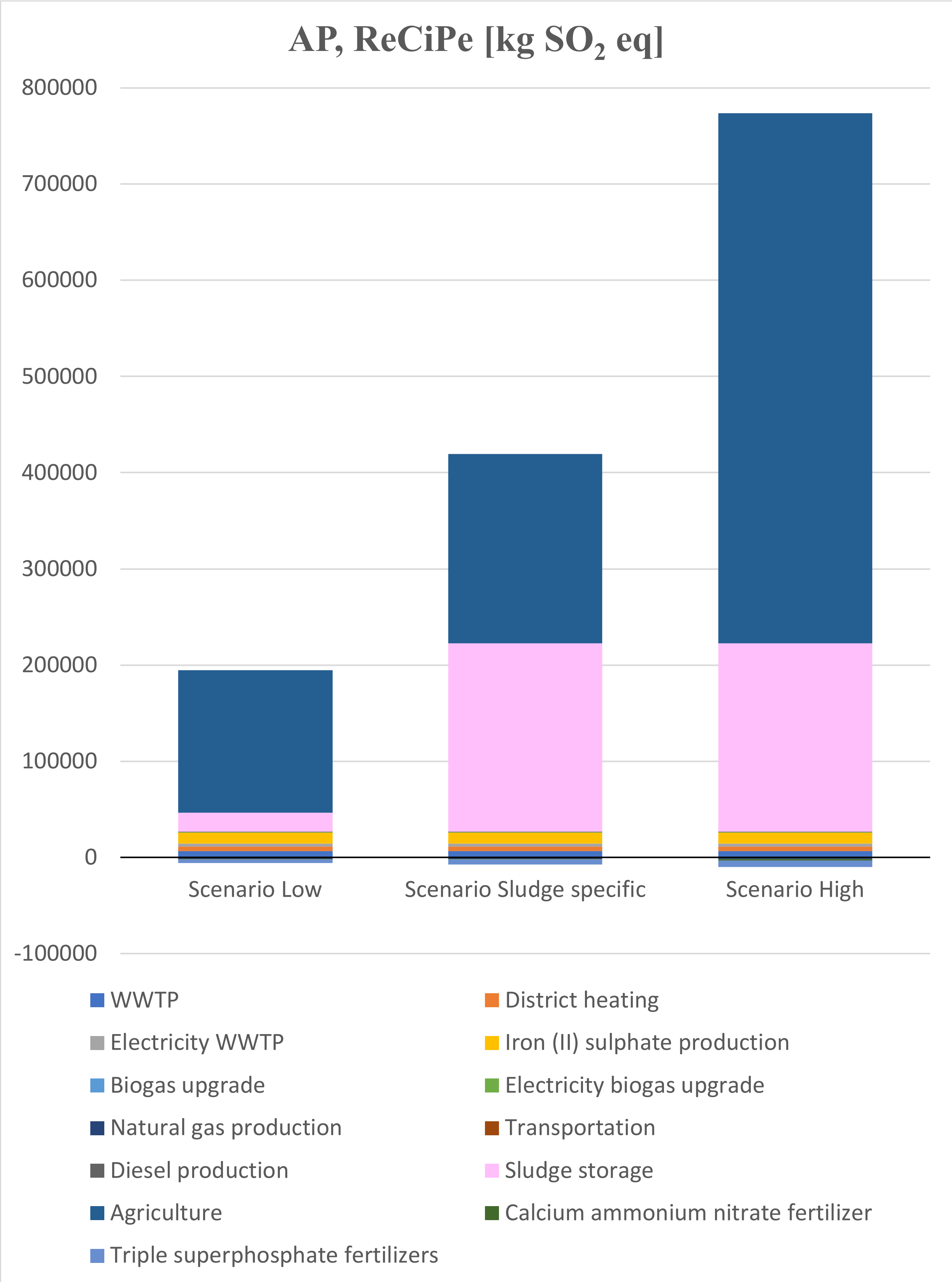

Figure 9: The LCIA results of acidification potential according to ReCiPe, per yearly inflow to Ryaverket.

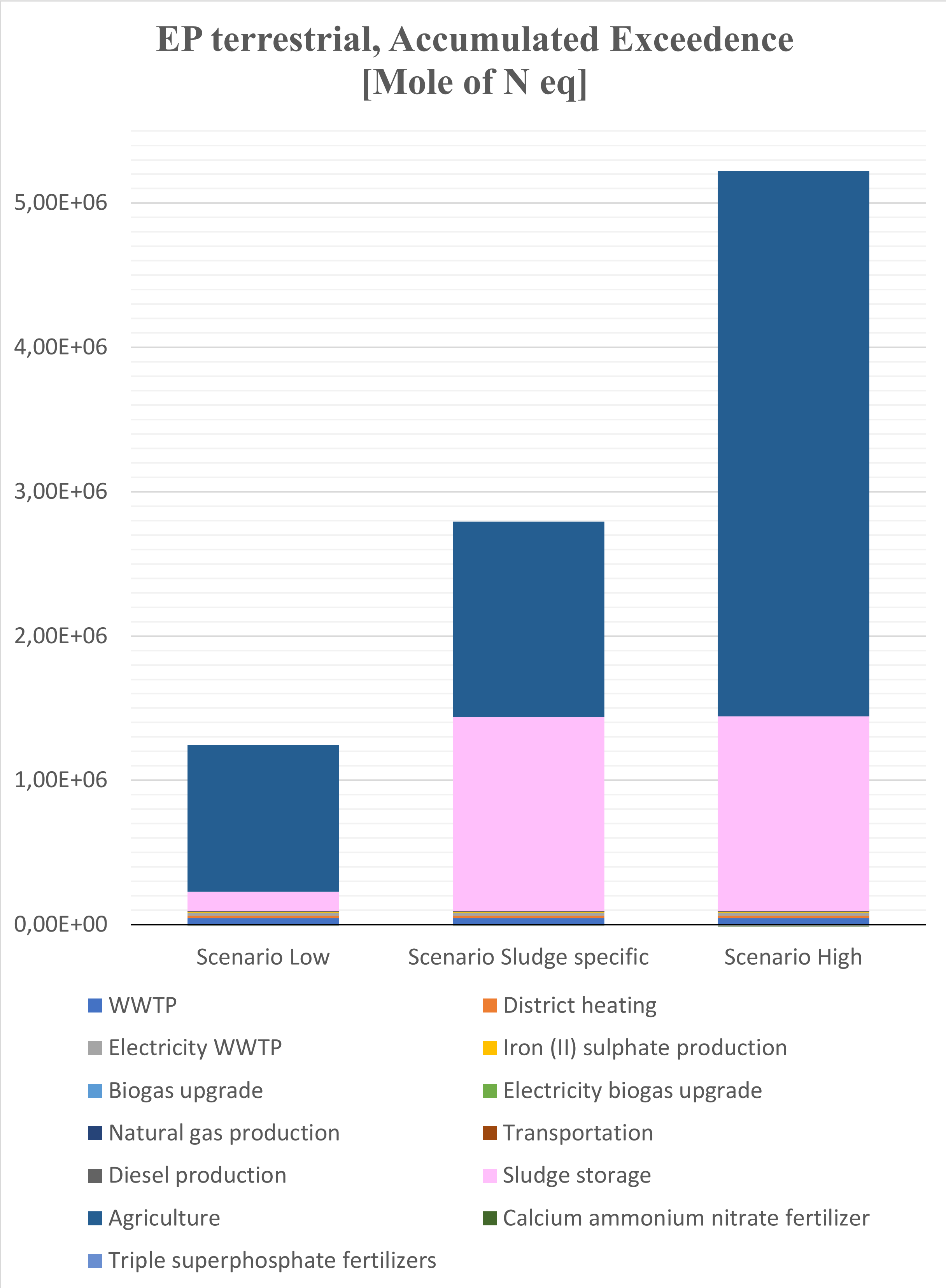

Figure 10: The LCIA results of eutrophication potential (terrestrial) according to accumulated exceedence, per yearly inflow to Ryaverket.

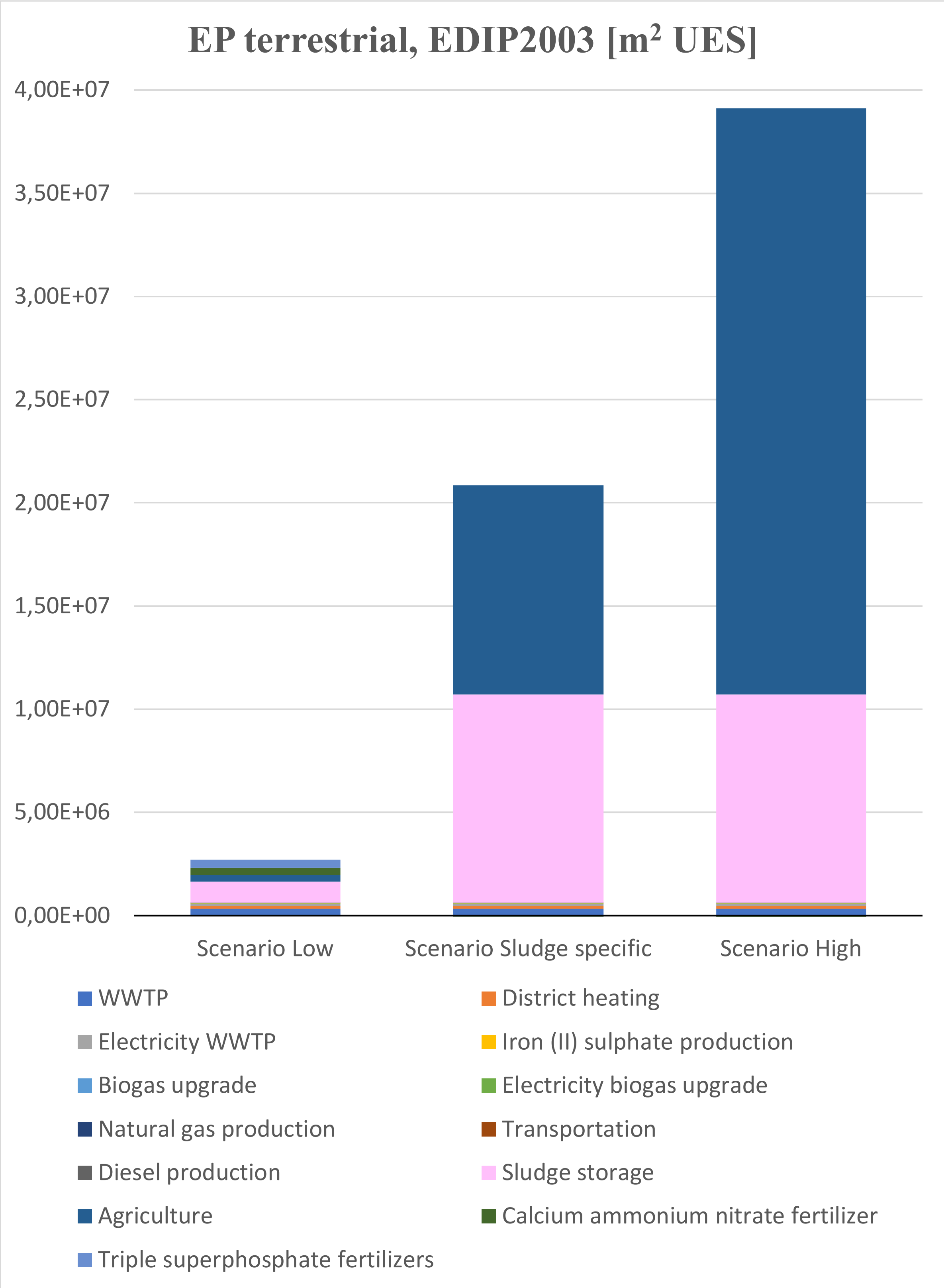

Figure 11: The LCIA results of eutrophication potential (terrestrial) according to EDIP2003, per yearly inflow to Ryaverket.

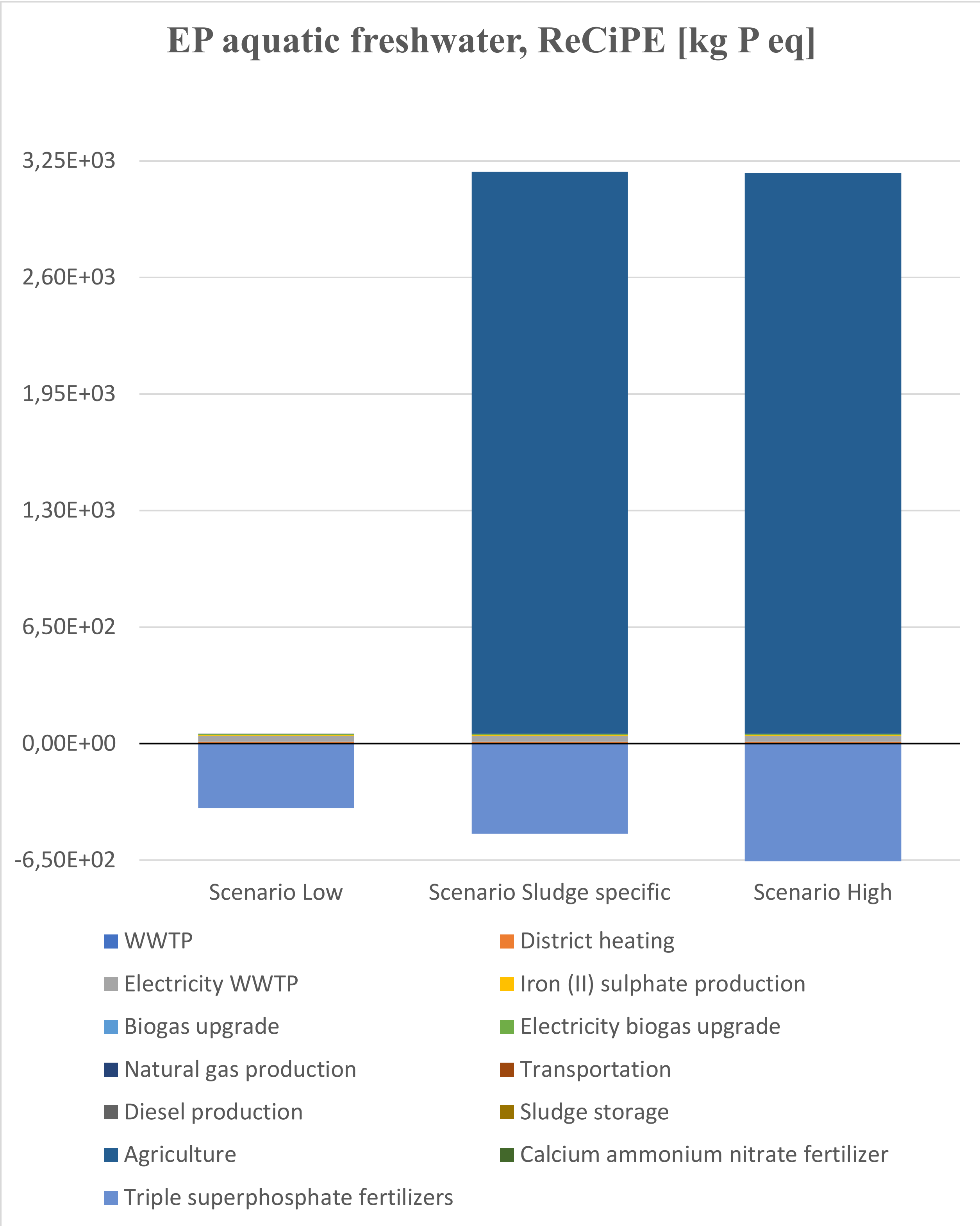

Figure 12: The LCIA results of eutrophication potential (aquatic, freshwater) according to ReCiPe, per yearly inflow to Ryaverket.

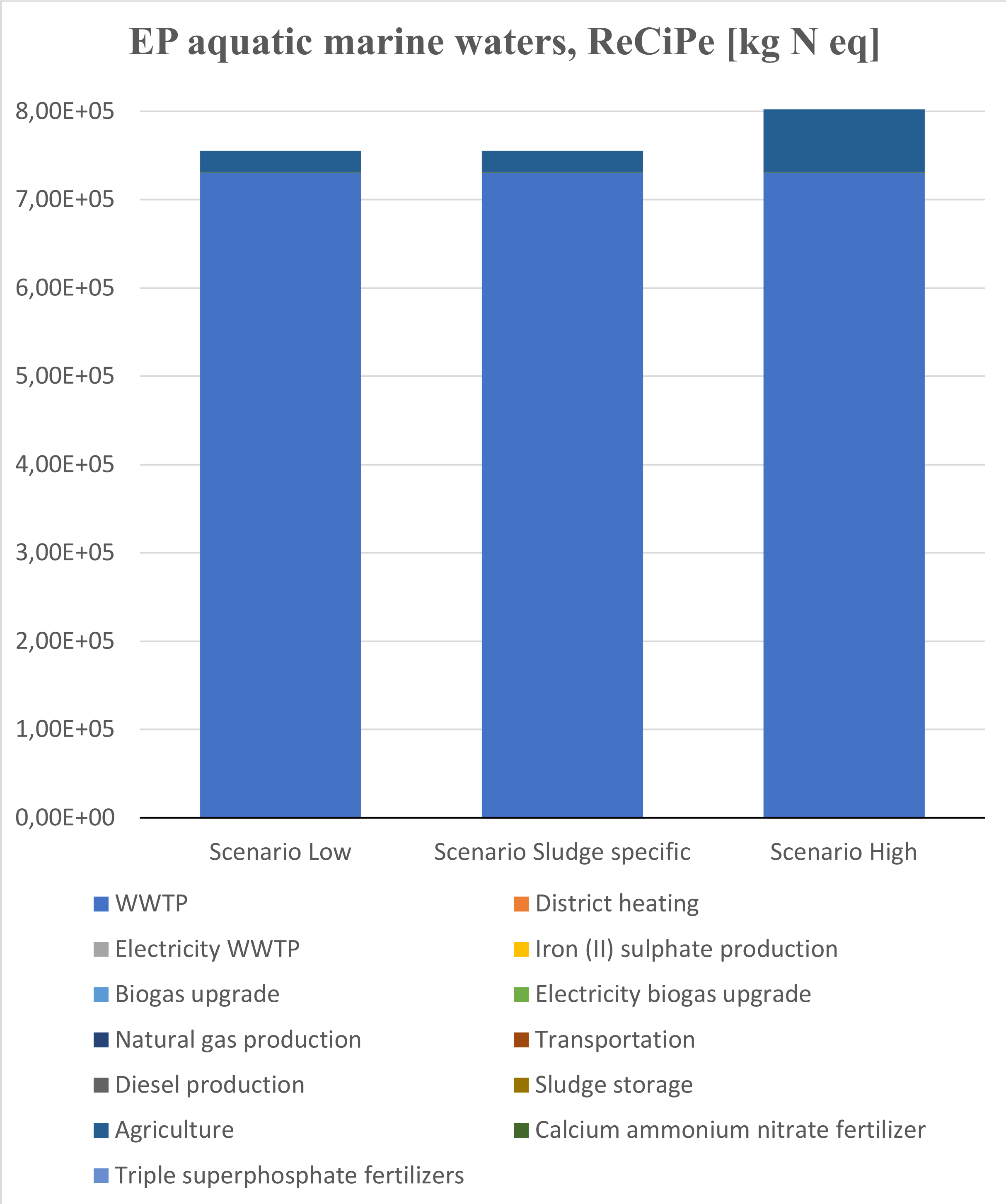

Figure 13: The LCIA results of eutrophication potential (aquatic, marine waters) according to ReCiPe, per yearly inflow to Ryaverket.

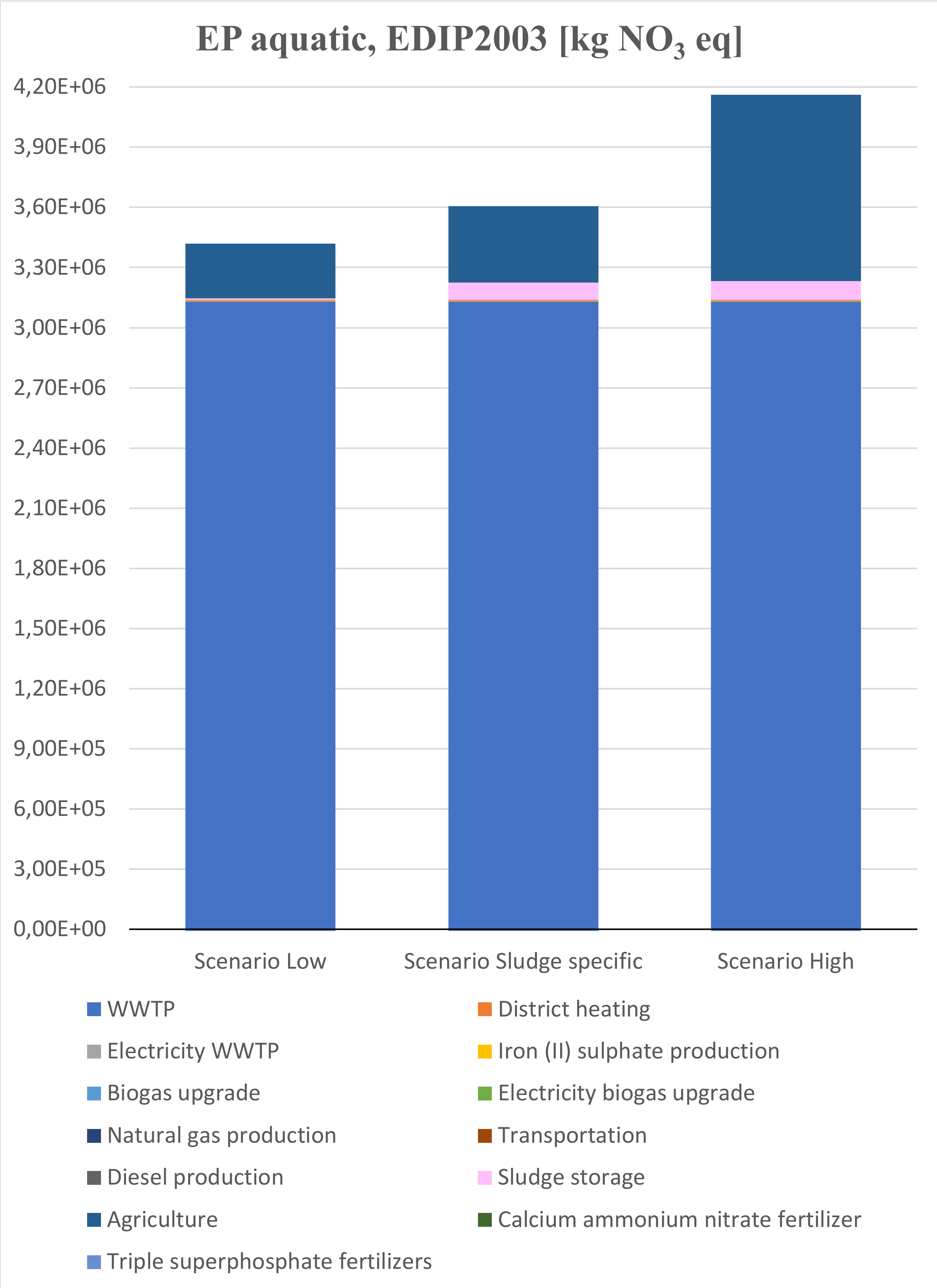

Figure 14: The LCIA results of eutrophication potential (aquatic according to EDIP2003, per yearly inflow to Ryaverket.

5. Discussion

This section contains a discussion of some relevant parts of the study. Firstly, it focuses on reflection of LCI of N and P compounds when sewage sludge is stored and applied to agricultural fields in relation to the case study results (section 5.1), and secondly are the methodological choices of an LCA reflected upon (section 5.2). It ends with some suggestions of recommendations for future studies in this field (section 5.3).

5.1 Life cycle inventory

The literature review, made in order to improve the LCI, revealed that relatively few studies on N and P emissions from sewage sludge storage and application on soil have been published in scientific literature. Despite this, the studies found in the review still gave a relatively good representation of how the emissions can differ. This was despite the fact that the studies were done in different ways (large scale field vs. pilot study) and used different types of sludges and soils, but generated similar results, although there was a certain difference.

The LCA case study results show a clear difference between the three tested LCIA-methods and the three scenarios. The WWTP is shown to have a great impact for the LCIA-methods GWP and EP for marine waters using ReCiPe and EDIP2003. This was expected, as the WWTP contributes with large portions of greenhouse gas emissions to air and large amounts of N to seawater every year. Also, the district heating showed to have a significant impact on GWP which could be surprising. The reason for this was believed to be since it was modeled in Gabi as European produced district heating and not as if it was produced in Sweden. That would be the case in reality since Ryaverket is located in Gothenburg, in which district heating is largely based on recycled waste heat from industries, household waste incineration and renewable energy sources (Göteborgs Energi AB, 2018), and therefore the environmental impact of the district heating should be smaller if modeled as Swedish produced district heat.

The sludge specific scenario, modeled with a mesophilically digested sewage sludge that was not covered during storage, and then spread on agricultural field, was modelled to represent the sewage sludge generated at Ryaverket in Gothenburg. The data selected was from the inventory review in section 4.1, i.e., primary data for Ryaverket's sludge has not been used. For the low and high modeled scenarios, no account was taken of how the sewage sludge has been treated, instead the lowest and the highest values from the literature review was used for the respective scenario. This led to, for example, very high values of NH_3 emissions to air in the high scenario, which was because the sludge was both treated with lime and NH_3. The emissions of N_2O during storage was zero in the case of mesophilically digested sludge since it had been stored with cover. The results for the sludge specific scenario was between the result of the low and high scenario for all LCIA methods. Thus, it can be concluded that modelling the sludge storage and use in agriculture to as far as possible be representative for the specific situation is important. If the handling of the sludge from Ryaverket had been modeled with inventory data for any other type of sludge that was treated in a different way than what is done (as in the high and low scenarios) this could lead to misleading results for several environmental impacts. Here, a discussion about the different types of emissions to different compartments in relation to the LCA case study will follow.

5.1.1 Emissions to air

The emissions of N to air that are most problematic, with regards to the chosen impact categories, are N_2O and NH_3. The N_2O that was considered, both during the storage of sewage sludge and when used in agriculture, proved to vary between studies but remained at relatively similar levels. The LCA case study showed that the storage of sewage sludge and application to agricultural fields have a big impact for several of the LCIA-methods. Also, the difference between the three scenarios indicates how important it is to do the inventory of these processes properly and as correctly as possible because there can be a big difference in the environmental impact if using an extremely high or low value of emissions, or if trying to model a scenario to fit the reference system (in this case study Ryaverket as the "Sludge specific scenario").

N_2O generated when sewage sludge is applied to agricultural fields showed to vary between 0.10 and 0.71 % of TN added, as can be seen in Table 8. The IPCC value of 1 % of TN (IPCC, 2006) in sewage sludge applied to agricultural fields was not exceeded by any source in the review. Rather, it was the opposite that the values were well below IPCC's value. Some might argue that it may be better to assume a higher value of nitrous oxide emissions than is actually generated in real life, and some will argue that it will not be beneficial to overestimate the emissions. However, during the final parts of this study, new revised IPCC reports have been approved for publication, however not yet publicly available. With that, the value of N_2O emissions from sewage sludge when applied to agricultural fields may have been updated, which can be good to have in mind.

The studies made by Nielsen et al. (2017) and Yoshida et al. (2015) were both pilot-studies in lab scale and showed emissions of N_2O between 0.10 and 0.66 % of TN added. Sludge treatment reed bed systems were shown in both of these studies to generate relatively low emissions of N_2O. The study made by Yoshida et al. (2015) shows that the emissions of nitrous oxide to air can differ depending on sludge type (applied on the same type of soil). N_2O emissions from agricultural fields is shown in the LCA case study not to have that big an influence on GWP results compared to other parts of the system. CH_4 emissions was in all three scenarios set to be zero, based on the literature review, which can be seen in Appendix I, but should not affect the environmental impact as much as the N_2O due to its much lower CF.

The study of Jönsson et al. (2015) investigated nitrous oxide emissions when sewage sludge is stored for hygienization. Also, regardless of hygienization method, sewage sludge is often stored until it is used in fields due to supply and demand, and it is unfortunate that storage has not been more investigated since it is shown to contribute a great deal in the LCA case study. The inventory result showed clearly that is it a big difference if the sludge is stored with or without a cover and whether the sludge has undergone a mesophilic or thermophilic digestion process and if it has been treated with urea. Treatment with urea gave negligible emissions of nitrous oxide since it slows down the nitrification and denitrification. This due to if the nitrification is affected in the "wrong" direction less NO_3^- is generated and therefore less denitrification happens and then less N_2O is generated. This may be a further reason why ammonia treatment can be a good alternative for hygienization of sewage sludge, besides from increasing the fertilizer value. The environmental impact of GWP is shown to contribute a great deal, both for the sludge specific scenario and even more in the high scenario. Important to point out is that CH_4 is also accounted for in the LCA case study for the storage of sewage sludge.

The emissions of NH_3 represented in the inventory was also shown to differ between the two studies found. The urea treated sludge showed higher emissions of NH_3 which likely was due to that it can easily evaporate to air when have been applied to sewage sludge for hygienization purposes. Also, when applying both urea and lime to sludge it showed even higher emissions of NH_3. The study by Jönsson et al. (2015), that was an actual field study made in Sweden, showed something that also is of importance. For both NH_3 and N_2O, the incorporation of the sewage sludge affected the emissions depending on if it was incorporated directly or delayed. Immediate incorporation of sludge resulted in higher N_2O emissions but lower NH_3 emissions, the sludge however had not undergone the exact same treatment which also can have had an impact on the results. The incorporation of sludge into the soil is regulated in Swedish law (SJVFS 2004:62), which farmers must follow. Knowing how the emissions are affected by incorporation should be important to know both when doing an LCI to know if it affects leakage to water of N and P and not only the emissions to air. It is also of importance for the farmer in practice to know how the incorporation affects the plant uptake. The ammonia emissions to air were shown to contribute a great deal to AP, for both Accumulated Exceedence and ReCiPe. The environmental impact varied among the three scenarios, which indicates that the inventory of ammonia to air is important to consider and also can vary a lot depending on how the sewage sludge has been treated.

5.1.2 Leakage to water

The leakage of N showed to differ quite much between the two reviewed studies (Table 4), which was also reflected in the results of the LCA case study, especially for marine eutrophication which has a CF for NO_3. The leakage of P showed in the inventory review to be less than the N leakage but shown to contribute to marine and freshwater EP using ReCiPe.

5.1.3 Crop uptake and replacement ratios

The crop uptake differed in the two reviewed studies (Börjesson & Kätterer, 2018, 2019; Esteller et al., 2009), see section 4.1. The studies reported for N emissions differed between 1.73 - 8 %, and for P between 0.35 – 20 %. However, the sludge used had not gone through the same treatment processes and the soil types were different. Both studies were field studies, but one conducted over a two-year period and the other over a 30-year period.

It is interesting to connect this to replacement ratios used for how much mineral fertilizers that are being replaced by sewage sludge. What has been shown is that the theoretical and commonly used replacement ratios often refer back to similar studies and have not been frequently updated. According to the personal contact with Emma Hjelm and Kjell Ivarsson, it was indicated that the N and P in sewage sludge does not differ in quality compared to other fertilizing options. Also, according to the results of the interviews, it is the soil and soil chemistry that will decide how much emissions and crop uptake the sewage sludge will generate and therefore how much mineral fertilizers that are being replaced.

In the LCA case study, the replacement of N and P mineral fertilizers gave the greatest credit to the system of environmental impact for GWP. The N mineral fertilizer gave larger credit to the system, and it can be assumed to be of the Haber Bosch process since it is energy demanding. Even though the P fertilizer production did not generate that big a negative impact for this impact category, it is important to have the phosphate rock depletion in mind. It can therefore be of interest to include other impact categories when assessing this system, e.g. resource depletion and toxicity.

Heimersson et al. (2016) listed replacement ratios for both nitrogen and phosphorus that had been used in earlier LCAs on wastewater and sludge treatment systems. Some of the reviewed articles used a replacement ratio of 100 % and assumed an equal availability of N and P in both sludge and mineral fertilizers. Other studies used different ratios but several referred to Lundin, Olofsson, Pettersson, & Zetterlund (2004) (who in return referred to Bengtsson, Lundin, & Molander (1997) and Dalemo, Sonesson, Jönsson, & Björklund (1998)). Lundin et al. (2004) used the replacement ratios 0.4 and 0.7 for nitrogen and phosphorus. Although Bengtsson et al. (1997) stresses that there are a large uncertainty to the values, they are often used in LCAs.

5.2 Methodological choices in LCA

The choices an LCA practitioner needs to make are many. An important part of the LCA is the LCIA and what methods to use to characterize the emissions ad resource flows in the LCI. Section 4.4 presents the LCIA-methods assessed in this study and their CFs. The methods were chosen mainly since they were recommended by the ILCD-handbook (European Commission, 2011), but there are several other methods available as well and it can be difficult to know what method that is suitable for a specific system. Section 4.4 shows that it differs how different LCIA methods characterize the inventory results for the same impact category. It is therefore important to know, for a chosen LCIA method, what needs to be inventoried in order not to get a misleading result.

For climate change it is the IPCC's GWP that is commonly used, and it was the only method in the ILCD handbook that was recommended for assessing climate change. For AP and EP, it is more difficult to state which LCIA method that is the best suited for this type of system. What is important when using any of the LCIA methods is to know what CFs that exist and what emissions that need to be inventoried since it has been shown to vary among the LCIA methods (section 4.4). One thing of interest is that for AP, accumulated exceedence have a CF for NO_2 and ReCiPE have one for NO_X. The atmospheric transformation of N compounds needs to be considered, and therefore it can be of great importance to know what LCIA method that are used for this system. It is important to have this in mind if using an LCA software tool as well. For example, GaBi has the same CFs for both NO_x and NO_2 for both Accumulated Exceedance and ReCiPe, which means that they already have taken this into account. Of the reviewed LCIA methods, the most specific in its CFs are ReCiPe. For freshwater and marine EP there are also different CFs according to three compartments - freshwater, agricultural soil and seawater - which makes it possible to inventory even more flows of emissions than has been done in this report, if possible.

5.3 Suggestions for future studies

After completing this study, the following suggestions for future studies have emerged:

- It would be interesting to conduct a similar study as Yoshida et al. (2015), that did a pilot study on several sludge types under the same conditions (e.g. soil type and incubation time). It would also be interesting to, instead of a pilot study in a lab scale, conduct a field study to measure and analyze emissions of N and P to air, but also to water and the crop uptake. What would be even more valuable are to also do this study with several soil types since the soil and soil chemistry have been pointed out as important factors for emissions and crop uptake of N and P.
- It would be interesting to provide a standardized value for N_2O emissions from sewage sludge storing, similar to the IPCC value of 1 % N_2O emissions when sewage sludge is

applied to agricultural fields, since it has shown to have a significant impact in the LCA case study.

- In the far future, and when this field hopefully has been more investigated, it would be interesting to develop a handbook to LCA practitioners, to be able to guide them through LCA in this field with recommendations of what level of emissions to consider of their sewage sludge depending on the most common factors that affects the emissions. Also, this handbook could present which LCIA methods that can be of benefit to use for this type of system.

6. Conclusions

This study had three main objectives (section 1.1). This section highlights the conclusions and main findings regarding each of them.

First and second objective

The results have shown that storing of sewage sludge and application to agricultural fields can have a significant impact on LCA results. It has proved important not to neglect emissions from sewage sludge storage as they add a significant environmental impact to the selected LCIA methods.

The study has also shown that LCA results can differ when varying the values of emissions. Therefore, it is important to gather data for the specific system, and not use generic numbers that can generate a misleading result. It is therefore important to know how the emissions are affected by different factors (e.g. the soil type, the sewage sludge quality, incorporation into the soil) in order to gain a result of environmental impact that reflects the reality as correctly as possible. There is also a need for continued research to be able to present more clearly how the emissions vary with the different factors.

Third objective

Both the reviewed literature and interviews made during this thesis work have pointed out that the crop uptake of N and P are case specific. It depends on soil type, soil chemistry, soil pH, how the sludge has been treated, the soil incorporation of sludge, weather and what type of crop that is being cultivated. This connects to the first and second objective as well, as there is a need of doing an LCI for the specific system you are assessing. The replacement ratios therefore also need to be chosen with respect to the specific assessed system.

References

Baumann, H., & Tillman, A.-M. (2004). *The Hitch Hiker's Guide to LCA*.

Bengtsson, M., Lundin, M., & Molander, S. (1997). Life Cycle Assessment of wastewater systems. *Technical Environemental Planning Report 1997*, *9*.

Biogas - Gryaab. (n.d.). Retrieved May 30, 2019, from https://www.gryaab.se/vad-vi-gor/biogas/

Börjesson, G., & Kätterer, T. (2018). Soil fertility effects of repeated application of sewage sludge in two 30-year-old field experiments. *Nutrient Cycling in Agroecosystems*, *112*(3), 369–385. https://doi.org/10.1007/s10705-018-9952-4

Börjesson, G., & Kätterer, T. (2019). Correction to: Soil fertility effects of repeated application of sewage sludge in two 30-year-old field experiments. *Nutrient Cycling in Agroecosystems*, *112*(3), 369–385. https://doi.org/10.1007/s10705-018-9952-4

Börling, K. (2003). *Phosphorus Sorption , Accumulation and Leaching*. Swedish University of Agricultural Sciences.

Börling, K., Kvarmo, P., Listh, U., Malgeryd, J., & Stenberg, M. (2018). Recommendations for fertilizing and liming *[In Swedish, Rekommendationer för gödsling och kalkning]*. 2018, 103.

Brady, N. C., & Well, R. R. (2013). *The Nature and Properties of Soils* (Fourteenth). PEARSON EDUCATION LIMITED.

Butusov, M., & Jernelöv, A. (2013). *Phosphorus: An Element that could have been Lucifer* (Vol. 9). https://doi.org/10.1007/978-1-4614-6803-5

Cordell, D., Drangert, J. O., & White, S. (2009). The story of phosphorus: Global food security and food for thought. *Global Environmental Change*, *19*(2), 292–305. https://doi.org/10.1016/j.gloenvcha.2008.10.009

Cordell, D., & White, S. (2011). Peak phosphorus: Clarifying the key issues of a vigorous debate about long-term phosphorus security. *Sustainability*, *3*(10), 2027–2049. https://doi.org/10.3390/su3102027

Dalemo, M., Sonesson, U., Jönsson, H., & Björklund, A. (1998). Effects of including nitrogen emissions from soil in environmental systems analysis of waste management strategies. *Resources, Conservation and Recycling*, *24*(3–4), 363–381. https://doi.org/10.1016/S0921-3449(98)00064-0

Delin, S., & Nyberg, A. (2014). Phosphorus fertilization effect of various residual products *[In Swedish: Fosforgödlingseffekt av olika restprodukter]*.

Dir 2018:67. (n.d.). Committee terms: Non-toxic and cirkular reversal of phosphorus from sewage sludge *[In Swedish: Kommittédirektiv: Giftfri och cirkulär återföring av fosfor från avloppsslam]* . [Elektronisk]. Stockholm: Miljö- och energidepartementet. Retrieved February 14, 2019, from https://www.regeringen.se/4a0f71/contentassets/3eb7bdf7312d426db311ac8d9cf72dc0/giftfri-och-cirkular-aterforing-av-fosfor-fran-avloppsslam-dir.-201867

Esteller, M. V., Martínez-Valdés, H., Garrido, S., & Uribe, Q. (2009). Nitrate and phosphate leaching in a Phaeozem soil treated with biosolids, composted biosolids and inorganic fertilizers. *Waste Management*, *29*(6), 1936–1944. https://doi.org/10.1016/j.wasman.2008.12.025

European Commission. (2010). Analysing of existing Environmental Impact Assessment methodologies for use in Life Cycle Assessment. *European Commission*, 115. https://doi.org/10.2788/38479

European Commission. (2011). *ILCD handbook. International Reference Life Cycle Data System (ILCD) Handbook, Recommendations for life cycle impact assessment in the European context*. https://doi.org/10.2788/33030

Forster, P., Ramaswamy, V., Artaxo, P., Berntsen, T., Betts, R., Fahley, D. ., … Van Dorland, R. (2007). *Changes in Atmospheric Constituents and in Radiative Forcing*. *Climate Change 2007* (Vol. 30). https://doi.org/10.1103/PhysRevB.77.220407

Gavala, H. N., Yenal, U., Skiadas, I. V., Westermann, P., & Ahring, B. K. (2003). Mesophilic and thermophilic anaerobic digestion of primary and secondary sludge. Effect of pre-treatment at elevated temperature. *Water Research*, *37*(19), 4561–4572. https://doi.org/10.1016/S0043-1354(03)00401-9

Goedkoop, M. J., Heijungs, R., Huijbregts, M., De Schryver, A., Struijs, J., & van Zeim, R. (2008). *ReCiPE 2008: Category indicators at the midpoint and the endpoint level*. https://doi.org/10.2307/40184439

Gryaab. (2019a). Retrieved May 25, 2019, from https://www.gryaab.se/

Gryaab. (2019b). Environmental report Ryaverket 2018 *[In Swedish: Miljörapport Ryverket 2018]*.

Göteborgs Energi AB. (2018). Annual and sustainability report *[In Swedish: Års- och hållbarhetsredovisning 2018]*.

Heimersson, S., Svanström, M., Cederberg, C., & Peters, G. (2017). Improved life cycle modelling of benefits from sewage sludge anaerobic digestion and land application. *Resources, Conservation and Recycling*, *122*, 126–134. https://doi.org/10.1016/j.resconrec.2017.01.016

Heimersson, S., Svanström, M., Laera, G., & Peters, G. (2016). Life cycle inventory practices for major nitrogen, phosphorus and carbon flows in wastewater and sludge management systems. *International Journal of Life Cycle Assessment*, *21*(8), 1197–1212. https://doi.org/10.1007/s11367-016-1095-8

Hobson J. (1999). CH4 and N2O Emissions From Waste Water Handling. *Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories*, 441–

454.

Huijbregts, M., Steinmann, Z. J. N., Elshout, P. M. F. M., Stam, G., Verones, F., Vieira, M. D. M., … van Zelm, R. (2016). ReCiPe 2016. *National Institute for Public Health and the Environment*. https://doi.org/10.1007/s11367-016-1246-y

IPCC. (2006). N_2O emissions from managed soils, and CO_2 emissions from lime and urea application, 11.1-11.54. Retrieved from http://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/4_Volume4/V4_11_Ch11_N2O&CO2.pdf

IPCC. (2014). Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. *Histopathology*. Geneva, Switzerland. https://doi.org/10.1046/j.1365-2559.2002.1340a.x

Jönsson, H, Rodhe, L., Junestedt, C., Willén, A., & Pell, M. (2016). Sewage sludge as fertiliser – environmental assessment of storage and land application options. *Water Science and Technology*, *75*(5), 1034–1050. https://doi.org/10.2166/wst.2016.584

Jönsson, Håkan, Junestedt, C., Willén, A., Yang, J., Tjus, K., Baresel, C., … Andersson, S. (2015). Reduce greenhouse gas emissions from sewage treatment and sewage sludge handling *[In Swedish: Minska utsläpp av växthusgaser från rening av avlopp och hantering av avloppsslam]*. Retrieved from www.svensktvatten.se

Jordbruksverket. (n.d.). Spread manure outside nitrate sensitive areas, where can I spread manure? *[In Swedish: Sprida gödsel utanför nitratkänsliga områden Var får jag sprida gödsel?].* Retrieved from www.jordbruksverket.se

Kjerstadius, H., la Cour Jansen, J., Stålhandske, L., Eriksson, E., Olsson, M., & Davidsson, Å. (2012). Digestion of sewage sludge at 35, 55 and 60 °C *[In Swedish: Rötning av avloppsslam vid 35, 55 och 60 °C].* Swedish Water and Wastewater Association.

Krogstad, T., Sogn, T. A., Asdal, Å., & Sæbø, A. (2005). Influence of chemically and biologically stabilized sewage sludge on plant-available phosphorous in soil. *Ecological Engineering*, *25*(1), 51–60. https://doi.org/10.1016/j.ecoleng.2005.02.009

Linderholm, K. (2011). Phosphorus and its plant availability in sludge - a literature study *[In Swedish: Fosfor och dess växttillgänglighet i slam – en litteraturstudie]*. Svensk Vatten Utveckling, 16.

Lundin, M., Olofsson, M., Pettersson, G. J., & Zetterlund, H. (2004). Environmental and economic assessment of sewage sludge handling options. *Resources, Conservation and Recycling*, *41*(4), 255–278. https://doi.org/10.1016/j.resconrec.2003.10.006

Magdoff, F., & Van Es, H. (2009). *Building Soils for Better Crops, 3rd Edition.*

Maguire, R. O., Sims, J. T., Dentel, S. K., Coale, F. J., & Mah, J. T. (2010). Relationships between Biosolids Treatment Process and Soil Phosphorus Availability. *Journal of*

Environment Quality, *30*(3), 1023. https://doi.org/10.2134/jeq2001.3031023x

Masclaux-Daubresse, C., Daniel-Vedele, F., Dechorgnat, J., Chardon, F., Gaufichon, L., & Suzuki, A. (2010). Nitrogen uptake, assimilation and remobilization in plants: Challenges for sustainable and productive agriculture. *Annals of Botany*, *105*(7), 1141–1157. https://doi.org/10.1093/aob/mcq028

Mendoza, C., Assadian, N. W., & Lindemann, W. (2006). The fate of nitrogen in a moderately alkaline and calcareous soil amended with biosolids and urea. *Chemosphere*, *63*(11), 1933–1941. https://doi.org/10.1016/j.chemosphere.2005.10.007

Nationalencyklopedin. (2019a). Macro nutrients *[In Swedish: Makronäringsämne]*. Retrieved April 22, 2019, from https://www-ne-se.proxy.lib.chalmers.se/uppslagsverk/encyklopedi/lång/makronäringsämne

Nationalencyklopedin. (2019b). Micro nutrients *[In Swedish: Mikronäringsämne].* Retrieved April 22, 2019, from https://www-ne-se.proxy.lib.chalmers.se/uppslagsverk/encyklopedi/lång/mikronäringsämne

Naturvårdsverket. (2002). Action plan for reversing phosphorus from sewage sludge *[In Swedish: Aktionsplan för återföring av fosfor ur avlopp]*. Retrieved from www.naturvardsverket.se

Naturvårdsverket. (2005). Losses of phosphorus from soil to water *[In Swedish: Fosforförluster från mark till vatten]*. Retrieved from https://www.naturvardsverket.se/Documents/publikationer/620-5507-0.pdf

Naturvårdsverket. (2011). Risks of spreading infetion via sewage sludge - reporting of treatment methods and regulations *[In Swedish: Risker för smittspridning via avloppsslam - redovisning av behandlingsmetoder och föreskrifter]*.

Naturvårdsverket. (2013). Sustainable recycling of phosphorus *[In Swedish: Hållbar återföring av fosfor].* Report 6580.

Netherlands National Institute for Public Health and the Environment. (2018). ReCiPe2016_CFs_v1.1_20180117. Retrieved April 23, 2019, from https://www.rivm.nl/documenten/recipe2016cfsv1120180117

Nielsen, S., Bruun, S., Bekiaris, G., Gómez-Muñoz, B., Larsen, J. D., Jensen, L. S., & Scheutz, C. (2017). Nitrogen mineralisation and greenhouse gas emission from the soil application of sludge from reed bed mineralisation systems. *Journal of Environmental Management*, *203*, 59–67. https://doi.org/10.1016/j.jenvman.2017.07.042

Nielsen, Steen, Peruzzi, E., Macci, C., Doni, S., & Masciandaro, G. (2014). Stabilisation and mineralisation of sludge in reed bed systems after 10-20 years of operation. *Water Science and Technology*, *69*(3), 539–545. https://doi.org/10.2166/wst.2013.723

Orbicon. (n.d.). Treatment of sludge in Sludge Treatment Reed Beds. Retrieved May 2, 2019, from https://www.orbicon.com/expertise/treatment-of-sludge-in-sludge-treatment-reed-beds

Palm, D., & Ek, M. (2010). Life Cycle Assessment of biogas from sewage sludge *[In Swedish: Livscykelanalys av biogas från avloppsreningsverksslam]*. Sweden.

Posch, M., Seppälä, J., Hettelingh, J. P., Johansson, M., Margni, M., & Jolliet, O. (2008). The role of atmospheric dispersion models and ecosystem sensitivity in the determination of characterisation factors for acidifying and eutrophying emissions in LCIA. *International Journal of Life Cycle Assessment*, *13*(6), 477–486. https://doi.org/10.1007/s11367-008-0025-9

Potting, J. J., & Hauschild, M. Z. (2005). Spatial differentiation in Life Cycle impact assessment - The EDIP2003 methodology-Guidelines from the Danish Environmental Protection Agency. *Danish Environmental Protection Agency*, (80).

Rydh, C. J., Lindahl, M., & Tingström, J. (2002). Life cycle assessment - a method for environmental assessment of products and services *[In Swedish: Livscykelanalys - en metod för miljöbedömning av produkter och tjänster]*.

Schröder, J. J. (2014). The Position of Mineral Nitrogen Fertilizer in Efficient Use of Nitrogen and Land: A Review. *Natural Resources*, *05*(15), 936–948. https://doi.org/10.4236/nr.2014.515080

Seppälä, J., Posch, M., Johansson, M., & Hettelingh, J. (2018). Country-dependent Characterisation Factors for Acidification and Terrestrial Eutrophication Based on Accumulated Exceedance as an Impact Category Indicator (14 pp) | SpringerLink, *11*(6), 403–416. Retrieved from https://link.springer.com/article/10.1065%2Flca2005.06.215%0Ahttp://files/549/10.1065 lca2005.06.html

Shepherd, M. A. (1996). Factors affecting nitrate leaching from sewage sludges applied to a sandy soil in arable agriculture. *Agriculture, Ecosystems and Environment*, *58*(2–3), 171–185. https://doi.org/10.1016/0167-8809(96)01018-3

Svanström, M., Heimersson, S., & Harder, R. (2016). Life cycle assessment of sewage sludge management with phosphorus recycling [In Swedish: Livscykelanalys av slamhantering med fosforåterföring]. *2016–13*.

Svanström, M., Laera, G., & Heimersson, S. (2015). Problem or Resource - Why It Is Important For the Environment to Keep Track of Nitrogen, Phosphorus and Carbon in Wastewater and Sludge Management. *Journal of Civil & Environmental Engineering*, *05*(06). https://doi.org/10.4172/2165-784X.1000200

Svensk Avloppsrening. (n.d.). Eutrpohication *[In Swedish: Övergödning]*. Retrieved February 20, 2019, from https://svenskavloppsrening.se/guides/kretsloppsanpassat-avlopp/

Svenskt Vatten. (2013). Sludge use and strategies for sludge use *[In Swedish: Slamanvändning och strategier för slamanvändning]*. Svenskt Vatten Meddelande M137. Retrieved from http://www.svensktvatten.se/globalassets/avlopp-och-miljo/uppstromsarbete-och-kretslopp/m137-slamanvandning-och-strategier-for-slamanvandning-mars-2013.pdf

Svenskt Vatten. (2016a). Uses for sludge [In Swedish: Användningsområden för slam]. Retrieved February 5, 2019, from http://www.svensktvatten.se/vattentjanster/avlopp-och-miljo/kretslopp-och-uppstromsarbete/slamanvandningsmetoden/

Svenskt Vatten. (2016b). Biogas. Retrieved February 5, 2019, from http://www.svensktvatten.se/vattentjanster/avlopp-och-miljo/slam-och-biogas/

Svenskt Vatten. (2016c). Biological phosphorus treatment *[In Swedish: Biologisk fosforrening]*. Retrieved April 7, 2019, from http://www.svensktvatten.se/vattentjanster/avlopp-och-miljo/reningsverk-och-reningsprocesser/biologisk-fosforrening/

Svenskt Vatten. (2018). Active upstream work Revaq-certification *[In Swedish: Aktivt uppströmsarbete med Revaq-certifiering].* Retrieved February 3, 2019, from http://www.svensktvatten.se/vattentjanster/avlopp-och-miljo/kretslopp-och-uppstromsarbete/revaq-certifiering/?_t_id=1B2M2Y8AsgTpgAmY7PhCfg%3D%3D&_t_q=revaq&_t_tags=language%3Asv%2Csiteid%3Ab47c99cb-a914-4f1b-8fba-9e4836a984f6&_t_ip=78.71.35.36&_t_hit.id=S

Svenskt Vatten. (2019).Rules and templates *[In Swedish: Regler och mallar].* Retrieved April 17, 2019, from http://www.svensktvatten.se/vattentjanster/avlopp-och-miljo/kretslopp-och-uppstromsarbete/revaq-certifiering/revaq-regler-och-mallar/

ten Hoeve, M., Bruun, S., Naroznova, I., Lemming, C., Magid, J., Jensen, L. S., & Scheutz, C. (2018). Life cycle inventory modeling of phosphorus substitution, losses and crop uptake after land application of organic waste products. *International Journal of Life Cycle Assessment*, *23*(10), 1950–1965. https://doi.org/10.1007/s11367-017-1421-9

Vinnerås, B., Nordin, A., & Jönsson, H. (2017). Ammonia hygienization for the safe use of sludge in cultivation *[In Swedish: Ammoniakhygienisering för säker användning av slam i odling]*. *Swedish Water and Wastewater Association*.

Willén, A., Rodhe, L., Pell, M., & Jönsson, H. (2016). Nitrous oxide and methane emissions during storage of dewatered digested sewage sludge. *Journal of Environmental Management*, *184*, 560–568. https://doi.org/10.1016/j.jenvman.2016.10.025

Yoshida, H., Christensen, T. H., & Scheutz, C. (2013). Life cycle assessment of sewage sludge management: A review. *Waste Management and Research*. https://doi.org/10.1177/0734242X13504446

Yoshida, H., Nielsen, M. P., Scheutz, C., Jensen, L. S., Christensen, T. H., Nielsen, S., & Bruun, S. (2015). Effects of sewage sludge stabilization on fertilizer value and greenhouse gas emissions after soil application. *Acta Agriculturae Scandinavica Section B: Soil and Plant Science*. https://doi.org/10.1080/09064710.2015.1027730

Yoshida, H., Ten Hoeve, M., Christensen, T. H., Bruun, S., Jensen, L. S., & Scheutz, C. (2018). Life cycle assessment of sewage sludge management options including long-term impacts after land application. *Journal of Cleaner Production*, *174*, 538–547. https://doi.org/10.1016/j.jclepro.2017.10.175

Appendix I – Inventory for LCA case study

Table A1: Presents the inventory data for the LCA case study for the WWTP, biogas upgrading and transportation.

WWTP Ryaverket		**Quantity**	**Unit**	**Reference**
In				
Energy	Electricity	41059	MWh/year	Gryaab (2019).
	District heat	12165	MWh/year	Gryaab (2019).
Chemicals	Aluminum chlorohydrate	242	ton/year	Gryaab (2019).
	Iron sulphate	3052	ton/year	Gryaab (2019).
Organic material	Methanol	2339	ton/year	Gryaab (2019).
Polymer	Amin/aminoakrylat	200.9	ton/year	Gryaab (2019).
Out				
Biogas production	Biogas	11930988	m^3/year	Gryaab (2019).
To air	CO_2 (fossil)			
	CH_4	187000	kg/year	Gryaab (2019).
	N_2O	28650	kg/year	Gryaab (2019).
	NH_3	3241	kg/year	Gryaab (2019).
	NMVOC	53	kg/year	Gryaab (2019).
To water	N_{tot}	730	ton/year	Gryaab (2019).
	P_{tot}	26.4	ton/year	Gryaab (2019).
Biogas upgrading				
In				
Energy	Electricity	0.72	kWh/Nm^3 CH_4	Palm & Ek (2010).
Out				
Product	Upgraded biogas	7456867,5	m^3/year	Calculated with info from Gryaab (2019).
To air	CH_4			Palm & Ek (2010).
Replacement ratio	CH_4 biogas / CH_4 natural gas	1		Assumption.
Transportation to storage				
Vehicle	Truck and trailer			
Fuel	Diesel			
Load weight	Sewage sludge	55542	ton/year	Gryaab (2019).
Distance	Transport to Kuskatorpet	150	km	Gryaab (2019).
Transport to agricultural fields				
Vehicle	Truck and trailer			
Fuel	Diesel			
Load weight	Sewage sludge	55542	ton/year	
Distance	Transport from Kuskatorpet to use in agriculture.	50	Km	Assumption.

Table A2: Presents the inventory data of the LCA case study for emissions from storage of sewage sludge and application to agricultural fields.

Storage of sewage sludge						
In		Quantity	Unit		Reference	
Amount DM sewage sludge	DM	14074	ton/year		Gryaab (2019).	
Total nitrogen content	N_{tot}	686000	kg/year		Gryaab (2019).	
Total phosphorus content	P_{tot}	477000	kg/year		Gryaab (2019).	
Total content of carbon	C_{tot}	306	kg/ ton DM		Assumption, based on Jönsson et al. (2015)	
Out	**Scenario Low**		**Scenario Sludge specific**		**Scenario High**	
To air		**Reference**		**Reference**		**Reference**
N_2O	0	Jönsson et al. (2015)	0.34 % of TN added	Calculated from Jönsson et al. (2015)	1.3 % of TN added	Jönsson et al. (2015)
CH_4	0.2 % of C_{tot} added	Jönsson et al. (2015)	1.1 % of C_{tot} added	(Willén et al., 2016)	1.3 of C_{tot} added	Jönsson et al. (2015)
NH_3	1.2 % of TN added	Jönsson et al. (2015)	12 % of TN added	Jönsson et al. (2015)	12 % of TN added	Jönsson et al. (2015)
Agriculture						
Out	**Scenario Low**		**Scenario Sludge specific**		**Scenario High**	
To air		**Reference**		**Reference**		**Reference**
N_2O	0.10 % of TN added	Yoshida et al. (2015)	0.34 % of TN added	Jönsson et al. (2015)	1 % of TN added	IPCC (2006)
CH_4	0	Jönsson et al. (2015)	0	Jönsson et al. (2015)	0	Jönsson et al. (2015)
NH_3	9.2 % of TN added	Mendoza, Assadian, & Lindemann (2006)	12.2 % of TN added	Jönsson et al. (2015)	38.9 % of TN added	Mendoza, Assadian, & Lindemann (2006)
To water						
NO_3	11.7 % of TN added	Esteller, Martínez-Valdés, Garrido, & Uribe (2009)	11.7 % of TN added	Esteller, Martínez-Valdés, Garrido, & Uribe (2009)	39 % of TN added	Shepherd (1996)
$H_2PO_4^-$	0	Assumption	0.21 % of added P	Esteller, Martínez-Valdés, Garrido, & Uribe (2009)	0.21 % of added P	Esteller, Martínez-Valdés, Garrido, & Uribe (2009)

Appendix II - Mass balance of nitrogen when sewage sludge is stored and applied to agricultural fields

Table A3 and A4 presents the mass balance of nitrogen when sludge is stored and applied to agricultural fields in more detail. Presents which flows that are accounted for and which references that were used.

Table A3: Presents a mass balance of a best-case scenario for nitrogen when sludge is stored and after applied to agricultural fields.

Best-case scenario			
		[%]	**Reference**
Input	TN	100	
To air (storage)	-	-	
To air	N_2O	0.10	Yoshida et al. (2015)
	NH_3	9.2	Mendoza et al. (2006)
To water	NO_3	11.7	Esteller et al. (2009)
Crop uptake	N	8	Börjesson & Kätterer (2018)
N unaccounted for (calculated)	N	71	

Table A4: Presents a mass balance of a worst-case scenario for nitrogen when sludge is stored and after applied to agricultural fields.

Worst case-scenario			
		[%]	**Reference**
Input	TN	100	
To air (storage)	N_2O	1.3	Willén, Rodhe, Pell, & Jönsson (2016)
To air	N_2O	0.7	Jönsson et al. (2015)
	NH_3	38.9	Mendoza et al. (2006)
To water	NO_3	39	Shepherd (1996)
Crop uptake	N	1.7	Esteller et al. (2009)
N unaccounted for	N	18.4	

Appendix III – Interview questions

The personal contact that was made with Emma Hjelm from Jordbruksverket and Kjell Ivarsson from Lantbrukarnas Riksförbund was in order to get the answer regarding the following questions:

- How is it considered in agriculture that sewage sludge replaces commercial fertilizers? Thus, how much mineral fertilizer does the farmer expect one kg of N and P in the sludge replaces? Is it a difference between the potential replacement ratio (based on studies) and what the farmer actually replaces? What kind of commercial fertilizer is being replaced?

- What I have understood is that the phosphorus being bound differently in the sludge depending on which precipitation chemical that has been used in the treatment plant, or alternatively if P is separated used biological treatment. Does the farmer consider the type of sludge that is spread on arable land? Thus, it is expected that different types of sludge have different P availability when calculating how much commercial fertilizer that can be replaced? Is a particular type of sludge preferred for use in agriculture?

- What is the view of farmers today about using sewage sludge as a replacement for commercial fertilizer?

- Are there any regional differences in Sweden on how sludge is used in agriculture?