DEGREE PROJECT

The mass balance of compounds in source-separated blackwater treatment at RecoLab, Helsingborg

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Abstract

As an alternative to conventional wastewater treatment, the use of source-separating wastewater systems is growing in popularity. This study evaluated processes at RecoLab, a source-separating wastewater treatment and recovery plant, which treats blackwater, greywater and food waste streams from the nearby residential area of Oceanhamnen, in Helsingborg, Sweden. The focus of the study was on the blackwater stream which included the biogas production and the struvite precipitation and ammonia stripping processes which recovered nutrients in the form of phosphorus and nitrogen.

The objectives were to analyse the variation in quantity and quality of incoming blackwater to RecoLab during 2023 and to quantify the mass flows of total solids, chemical oxygen demand, phosphorus, and nitrogen throughout the blackwater treatment process. Furthermore, the end products, including biogas, sludge, and recovered nutrients were analysed in terms of quantity and quality and then compared with products from conventional wastewater treatment plants. Data collected during 2023, including water samples and flow measurements at various points in the blackwater treatment process, were used to calculate mass balances of the investigated parameters.

From the results, it was seen that the anaerobic digestion in RecoLab produced more biogas than most conventional wastewater treatment plants (150 m^3 CH_4/week or approximately 88 kWh/PE/year), with the biogas having a large impact on the total solids and chemical oxygen demand mass balances. The digestion yielded a sludge with low heavy metal concentrations (13 mg Cd/kg P), making it safe for agricultural use. Low metal concentrations were also found in the struvite (0.68 mg Cd/kg P), although decreasing metal concentrations over time point to a diffuse source of leaching in the equipment. The nutrient recovery as sludge and struvite, a recovery of 27% of the total phosphorus and 2.5% of the total nitrogen, was impacted by the struvite precipitation process showing a low collection grade. Substantial total phosphorus and total solids losses were found over the struvite precipitation process in the mass balances. Total nitrogen loss over the struvite precipitation process was mainly deemed to be due to aeration, though a possible overestimated value. Hypothetical calculations of the ammonia stripper showed great potential for nitrogen recovery in the form of ammonium sulphate, where a fully working ammonia stripper would contribute with the largest fraction of nutrient recovery in the RecoLab plant.

Overall, the thesis presents an alternative to conventional wastewater treatment methods by possibilities to enhance nutrient recovery and support circular economy principles. However, the study highlights several areas for improving the blackwater treatment at RecoLab, such as optimizing the hydrocyclone to increase the struvite recovery and conducting direct measurements of the aeration process to better understand nitrogen pathways. The implementation of source-separating systems, as demonstrated by RecoLab, can serve as a model for future urban developments, aiming for resource efficiency and minimal environmental impact.

Sammanfattning

Ett alternativ till konventionell avloppsrening är användandet av källsorterande avloppssystem, som ökar allt mer i popularitet. Denna studie utvärderade processer vid RecoLab, en källsorterande avloppsrenings- och återvinningsanläggning, som behandlar svartvatten, gråvatten och matavfall från det närliggande bostadsområdet Oceanhamnen, i Helsingborg, Sverige. Studiens fokus låg på svartvattenströmmen, inklusive biogasproduktionen från en anaerob nedbrytning, samt struvitfällning och ammoniakstrippning som återvann näringsämnen i form av fosfor och kväve.

Syftet var att analysera variationer i mängd och kvalitet på inkommande svartvatten till RecoLab under 2023 och att kvantifiera massflödena av total torrsubstans, kemisk syreförbrukning, fosfor och kväve genom svartvattenreningsprocessen. Vidare analyserades mängden och kvaliteten av slutprodukterna, inklusive biogas, slam och återvunna näringsämnen, vilket sedan jämfördes med produkter från konventionella avloppsreningsverk. Data insamlat under 2023, inklusive vattenprover och flödesmätningar vid olika punkter i svartvattenreningsprocessen, användes för att beräkna massbalanser för de undersökta parametrarna.

Resultaten visade att den anaeroba nedbrytningen i RecoLab producerade mer biogas än i de flesta konventionella avloppsreningsverk (150 m^3 CH_4/vecka eller cirka 88 kWh/PE/år), där biogasproduktionen dessutom hade en stor inverkan på massbalanserna för total torrsubstans och kemisk syreförbrukning. Nedbrytningen gav ett slam med låga tungmetallkoncentrationer (13 mg Cd/kg P), vilket gör det säkert för jordbruksanvändning. Låga metallkoncentrationer hittades också i struviten (0,68 mg Cd/kg P), med minskande metallkoncentrationer över tid som antyder en diffus urlakning från utrustning. Näringsåtervinningen i form av slam och struvit, en återvinning av 27% av totalfosforn och 2,5% av totalkvävet, påverkades av struvitfällningsprocessen som hade en låg insamlingsgrad. Stora förluster av totalfosfor och totalt torrsubstans syntes över struvitfällningsprocessen i massbalanserna. Totalkväveförlusten över struvitfällningsprocessen ansågs främst bero på luftning, även om det var ett möjligt överskattat värde. Hypotetiska beräkningar av ammoniakstrippningen visade stor potential för kväveåtervinning i form av ammoniumsulfat, där en fullt fungerande ammoniakstrippning skulle bidra med den största andelen näringsåtervinning i anläggningen.

Sammanfattningsvis presenterar studien ett alternativ till konventionella avloppsreningsmetoder, där exempelvis möjligheter till förbättrad näringsåtervinning bidrar till den cirkulära ekonomin. Det finns däremot flera förbättringsområden gällande svartvattenbehandlingen i RecoLab, såsom att optimera hydrocyklonen för att öka struvitåtervinningen och att genomföra direkta mätningar av luftningsprocessen för att bättre förstå var förluster av kväve sker. Implementeringen av källsorterande system, likt RecoLab, kan fungera som modeller för framtida stadsutvecklingsprojekt som strävar efter resurseffektivitet och minimal miljöpåverkan.

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List of Abbreviations

AC system	Accumulation system
AD	Anaerobic digester
BOD	Biological oxygen demand
BOD_7	Biological oxygen demand during 7 days
BW	Blackwater
CH_4	Methane
CO_2	Carbon dioxide
COD	Chemical oxygen demand
CSTR	Continuous Stirred Reactor
FW	Food waste
GW	Greywater
H_2S	Hydrogen sulphide
HRT	Hydraulic retention time
$MgCl_2$	Magnesium chloride
MgO	Magnesium oxide
NH_3	Ammonia
NH_4^+	Ammonium
NL	Normal litre, at STP
NSVA	Nordvästra Skånes Vatten och Avlopp
PE	Population equivalent
SRT	Sludge retention time
SS	Suspended solids
STP	Standard temperature and pressure
TN	Total nitrogen
TOC	Total organic carbon
TP	Total phosphorus
TS	Total solids
UASB	Upflow anaerobic sludge blanket reactor
UASB-ST	Upflow anaerobic sludge blanket reactor septic tank reactor
VFA	Volatile fatty acids
WWTP	Wastewater treatment plant

1. Introduction

To manage wastewater in urban areas, the implementation of sewage systems began in Sweden in the late 19th century (Swedish EPA, 2022). The first sewage system which was built were combined sewers, in which stormwater and wastewater was collected in the same pipe (ibid). Later, duplicate sewers were built, which consist of separate pipes for stormwater and wastewater. At first, wastewater was discharged into water bodies without any treatment, though water pollution led to the development of wastewater treatment plants (WWTP) (ibid).

In conventional WWTPs, mechanical, chemical and biological treatment methods are used to remove particles, organic matter, phosphorus and nitrogen from the wastewater (Swedish EPA, 2022). Despite the removal, discharge from municipal wastewater is the second largest anthropogenic source of nutrients, such as nitrogen and phosphorus, to the waters surrounding Sweden (ibid). As a result of the discharge of excess nutrients, natural waters can suffer from eutrophication.

Sludge is formed during processes in WWTPs, and it has a high organic content (Elalami et al., 2019). Anaerobic digestion (AD), which brings a production of biogas, is a common method to stabilize sludge (ibid). The energy which is needed for anaerobic digestion is usually covered by the WWTPs' own production of biogas, though additional energy is needed to run, for example, the biological wastewater treatment (Kjellén & Andersson, 2002). Sludge can be applied to agricultural land as a fertilizer as it contains nitrogen, phosphorus, and organic matter, amongst other elements, which is crucial for plant growth (Fijalkowski et al., 2017). However, as described by Fijalkowski et al. (2017), pathogens, heavy metals, nanoparticles, pharmaceuticals, personal care products and antibiotics found in sludge make the application to agricultural land problematic.

An alternative way to manage wastewater is by implementing source-separating wastewater systems, as discussed by Otterpohl et al. (2004). In these systems, domestic wastewater flows are separated based on their vastly different characteristics. An example of a separated stream is the low volume, nutrient rich toilet water, known as blackwater (BW). Another stream is the high volume, shower, laundry, and kitchen water, known as greywater (GW).

BW can be collected using vacuum sewer systems, which decreases the amount of water used for toilet flushing and brings a highly concentrated stream, making it possible to apply anaerobic digestion to the blackwater directly (Otterpohl et al., 2004). Anaerobic digestion of blackwater leads to a potential for increased biogas production compared with conventional sludge digestion (Kjerstadius et al., 2015). Furthermore, the nutrient rich blackwater brings possibilities to recover nutrients to a higher degree (Kjerstadius et al., 2015) which is important for the increasing fertilizer demand (Desmidt et al., 2015).

Other advantages with source-separating wastewater systems include lower emissions and possibilities of targeted treatment of pathogens and pollutants (McConville et al., 2023). Two other examples of drivers, stated by Skambraks et al. (2017), for the implementation of source-separation, are sustainability goals and knowledge generation, to combat future challenges caused by climate change.

One urban source-separating system which has been implemented is in the H+ project, in Helsingborg, Sweden (Schelbert et al., 2023). In a triplicate sewer network, food waste (FW), blackwater and greywater are collected separately from households and then transported to the wastewater treatment and recovery plant RecoLab. The treatment process of blackwater results in produced biogas as well as nitrogen and phosphorus fertilizers in the form of struvite and ammonium sulphate, respectively.

However, the lack of knowledge about source-separation within the wastewater sector hinders the implementation of further systems (McConville et al., 2017). Additionally, the technology is deemed as immature which can lead to technical problems and decreased legitimacy of the systems. Though, the inspiration and knowledge that already implemented source-separating sites bring has been seen to be important for the growth of the technology (McConville et al., 2023; Skambraks et al., 2017).

This thesis contributes to the mentioned lack of knowledge about source-separating wastewater systems and the results that these systems can bring. Therefore, the aim was to evaluate the RecoLab source-separation blackwater resource recovery efficiency. Calculations of the blackwater flow and mass balances of solids, nitrogen, phosphorus, and chemical oxygen demand (COD) were conducted. The specific objectives were to:

- Analyse the variation in the quantity and quality of incoming blackwater to RecoLab during 2023.
- Quantify the mass flows of total solids, chemical oxygen demand, phosphorus, and nitrogen throughout the blackwater treatment process at RecoLab.
- Compare the production and quality of end products, including biogas, sludge, and recovered nutrients, at RecoLab with those at conventional wastewater treatment plants.

1.1. Scope of study

This study focuses on source-separating wastewater systems in urban environments, covering the treatment processes of source separated blackwater in the RecoLab facility, in Helsingborg, Sweden. The study does not cover the RecoLab treatment processes of greywater nor of food waste.

The assessment of the treatment performance is limited to the flows of total solids, nitrogen, phosphorus, and the chemical oxygen demand. This is based on the importance of nutrients for fertilizer production.

The mass balances were calculated and analysed using data from 2023. However, for certain parameters data outside of the timeframe was used to complete the dataset. Due to limitations within treatment processes and analysis, some mass balances were based on data from half of the year of 2023.

2. Background

In this chapter, the topic of source-separated wastewater systems is covered, with a focus on H+ and processes related to RecoLab. This includes a description of the streams in source-separated wastewater, how biogas is produced, and nutrients recovered from BW. In Figure 1 an illustration of how source-separated wastewater systems can bring re-use and recovery of materials is shown, thus closing the loop as described by Hoffmann et al. (2020).

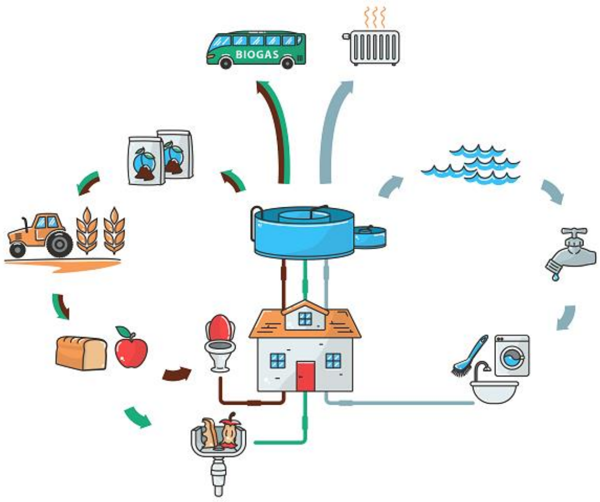

Figure 1: An illustration of source-separated wastewater systems. Figure from Kjerstadius et al (2016).

2.1. H+: Oceanhamnen and RecoLab

H+ is an urban development project in the city of Helsingborg, Sweden (Helsingborgs Stad, 2021). The project covers the development of the housing district of Oceanhamnen, which will have apartments, offices, restaurants, and shops divided into four blocks. The first residents moved into Oceanhamnen in 2020. However, the first block, as shown in Figure 2, was finished in 2022 with approximately 340 apartments and three office buildings (Helsingborgs Stad, 2024).

Within the properties, there are three separate piping systems for BW, GW, and grinded FW. BW collection is done by a vacuum sewer network, consisting of parallel vacuum pipes which are connected to groups of properties (Kvarnström et al., 2022). The sewer pipes are 75 – 90 mm in diameter and have transport pockets every 30 meters (H. Kjerstadius, personal communication, 2024). The vacuum pipes go to a central pumping station in Oceanhamnen, from which a pressurized sewer transport the BW to RecoLab, the wastewater treatment and recovery plant which is managed by the municipal water utility Nordvästra Skånes Vatten och Avlopp (NSVA) (Kvarnström et al., 2022).

Figure 2: Sketch of Oceanhamnen when finished. The red rectangle shows the currently constructed housing block. The red circle shows RecoLab. Figure modified from Helsingborg Stad (2021).

In 2021, the source-separated wastewater from Oceanhamnen started to be treated at RecoLab, which has a capacity for approximately 2 000 population equivalents (PE) (RecoLab, 2021). The amount of PE is conventionally based on an assumption of 60 g of biological oxygen demand (BOD) per person and day (Henze et al., 2008). RecoLab consists of three separate treatment lines, for BW, GW, and FW respectively (RecoLab, 2021). As seen in Figure 3, the BW treatment process consists of a collection tank, an anaerobic digester, a drum filter, and processes for struvite precipitation and ammonia stripping.

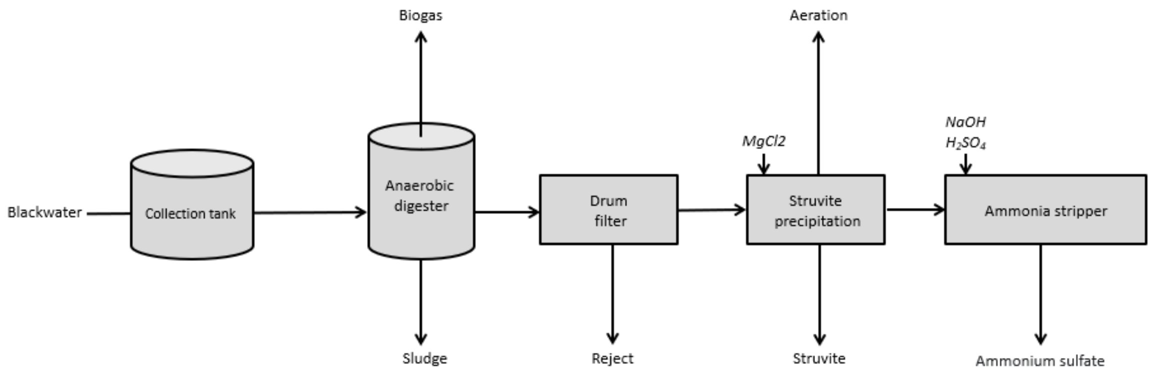

Figure 3: Sketch of the BW treatment process at RecoLab.

The collection tank had a volume of 10 m^3, after which the BW was led to an upflow anaerobic sludge blanket reactor, UASB-ST, which worked at a constant temperature of 37°C (RecoLab, 2021). During AD, sludge, which was regularly emptied from the reactor, and biogas were produced. At the BW flow rate during the time of the study, the hydraulic retention time (HRT) of the reactor was approximately 5 days, whereas the sludge retention time (SRT) was approximately 75 days (NSVA, n.d.). The decantate then went through a drum filter of 40 µm pore size to reduce the amount of suspended solids (SS) in the water (H. Kjerstadius, personal communication, 2024).

When the BW entered the struvite precipitation, which was done in batches, an aeration tank decreased the amount of carbon dioxide (CO_2) in the water and increased the pH (EkoBalans, n.d.). In a precipitation tank magnesium chloride ($MgCl_2$) was added and struvite ($MgNH_4PO_4 \cdot 6H_2O$) precipitated. The struvite was separated from the water and into a bag by a hydrocyclone.

Further on, the treated water went to an equalization tank in which sodium hydroxide (NaOH) was added to raise the pH. To collect suspended material left after the previous treatment, the BW went through two filters before being heated up in a heat exchanger. Then, in a stripper tower the ammonia (NH_3) transferred from the water phase to gas. From there, the water exited the BW treatment process. Sulfuric acid (H_2SO_4) was added to the gas in a reactor and crystals of ammonium sulphate ($(NH_4)_2SO_4$) formed. The ammonium sulphate was separated in a hydrocyclone. The recovered mineral nutrients could later be mixed with dewatered sludge from the AD to form a pellet fertilizer (RecoLab, 2021).

Figure 4: From left to right, struvite, ammonium sulphate, and pellet fertilizers from RecoLab.

2.1.1. *Other source-separating wastewater systems*

In Hamburg, the municipal water utility has implemented a source-separation concept for the urban development of Jenfelder Au, an old military area which will be connected to more than 2000 people (Kvarnström et al., 2022). By using vacuum sewers for BW, the water use has decreased 30% compared with other areas in Hamburg (HAMBURG WASSER, n.d.). AD of the BW brings biogas, which is used for production of electricity and heating. Furthermore, GW treatment technologies are tested in the project.

Another example is De Nieuwe Dokken, Ghent, an urban development project where approximately 1200 PE will be connected to a source-separation system when it is finished, by 2025 (Kvarnström et al., 2022). Similarly to Oceanhamnen and Jenfelder Au, vacuum sewers are used in the system (DuCoop, n.d.). In this case, FW and BW are anaerobically co-digested, resulting in biogas which covers 1/3 of the heat consumption in De Nieuwe Dokken. By struvite precipitation, a fertilizer is produced. After treatment of the source-separated waste streams, the treated water is sold and used in industrial processes.

2.2. Streams in domestic source separated wastewater

In source separated wastewater systems, wastewater is separated into streams due to their different characteristics, for example into blackwater (BW) from toilets and greywater (GW) from showers, laundry, and kitchen sinks. Food waste (FW) can also be incorporated into the systems.

2.2.1. *Greywater*

GW consists of water produced from bathing/showering, washing hands, laundry and using the kitchen sink (Kujawa-Roeleveld & Zeeman, 2006). It contains a large fraction of the heavy metals which are found in wastewater, for example originating from cleaning products. In a study of an ecological housing district in Germany, Sievers et al. (2016) showed GW with mean total nitrogen and phosphorus concentrations of 13.7 and 6.9 mg/L. In a study where GW from Oceanhamnen was used in an experimental setup, the GW had mean total nitrogen and phosphorus levels of 16 and 2.0 mg/L (Sami et al., 2023). There is a large variability in GW quantity, with the flow pattern being diurnal and depending on the water use in the country (Shaikh & Ahammed, 2020). Approximately 100 L per person and day of GW is produced in high-income countries (Hammes et al., 2000). According to Jönsson et al. (2005), GW approximately has a total solids (TS) level of 26 kg/PE/year whereas the chemical oxygen demand (COD) and BOD are 23 respectively 12 kg/PE/year in Swedish settings.

2.2.2. *Blackwater*

BW contains faeces, urine, toilet paper and flush water and thus have high levels of organic matter, nutrients such as nitrogen, phosphorus and potassium but also pathogens, pharmaceutical residues and hormones (Kujawa-Roeleveld & Zeeman, 2006).

Urine mostly contains water, though inorganic salts and organic compounds are also present (Kujawa-Roeleveld & Zeeman, 2006). The most frequent dissolved compound in urine is urea (CH_4N_2O), which contains approximately 85% of the nitrogen present in urine (Udert et al., 2006). The spontaneous process of urea hydrolysis causes ammonia (NH_3) and bicarbonate (HCO_3^-) to form, together with a strong pH increase. Phosphorus is mostly present as dissolved phosphate in urine, though precipitation reactions are frequent and phosphate minerals, struvite and hydroxyapatite, are found in wastewater systems. Urine is also the main source of pharmaceuticals in wastewater (Kujawa-Roeleveld & Zeeman, 2006).

TS in urine is approximately 7 kg/PE/year, nitrogen around 4 kg/PE/year and phosphorus 0.33 kg/PE/year in Swedish conditions according to Jönsson et al. (2005). Additionally, the COD is deemed to be approximately 3 kg/PE/year and the BOD is 2 kg/PE/year.

Faeces contains high amounts of organic matter, which means high levels of carbon, and is rich in phosphorus and potassium (Harder et al., 2019). There are also contaminants in faeces, including bacteria, viruses, protozoa, and parasitic worm eggs. Moreover, of the heavy metals we ingest through our food, more metals are found in faeces than in urine. The concentrations of elements in faeces vary depending on if toilet paper is included. However, TS of 19 kg/PE/year with toilet paper included is proposed as a typical value in Swedish settings by Jönsson et al. (2005). Furthermore, nitrogen levels are approximately 0.5 kg/PE/year and phosphorus 0.18 kg/PE, year. Considering the total COD and BOD levels of faeces, Jönsson et al. (2005) deemed 23 and 12 kg/PE/year, respectively, to be a reasonable level.

When combining faeces and urine, BW has, amongst other parameters, high levels of suspended matter. Measurements of BW in the ecological housing district of Flintenbreite in Lübeck, Germany, showed an average nitrogen load of 1.4 g/L and approximately 73 mg/L of phosphorus (Sievers et al., 2016). Another study of BW in Flintenbreite showed TS levels of approximately 6.5 g/L, COD of approximately 8.7 g/L, total nitrogen levels of 1.5 g/L and total phosphorus levels of 175 mg/L in the BW stream of 5 L/PE/day (Wendland et al., 2007).

2.3. Vacuum toilets and sewers

Vacuum sewer systems use differential air pressure, generated by vacuum pumps at a vacuum station, to transport the wastewater (Mohr et al., 2016). The pumps draw in air from air inlets which are attached to valve pits that open after wastewater has accumulated. Then, the wastewater is sucked into the vacuum main towards the central vacuum station. A driving mechanism behind the system is the expansion of air in negative pressure, which causes the wastewater to move towards the pump (ibid). In vacuum sewers the pipes are laid with bends, so called transport pockets, in which accumulation of the wastewater leads to build up of the negative pressure (Flydén & Lindblom, 2022). The wastewater is turbulently mixed with air which causes lifting of the wastewater out of the transport pocket and further into the system. An example of a vacuum sewer system is seen in Figure 5.

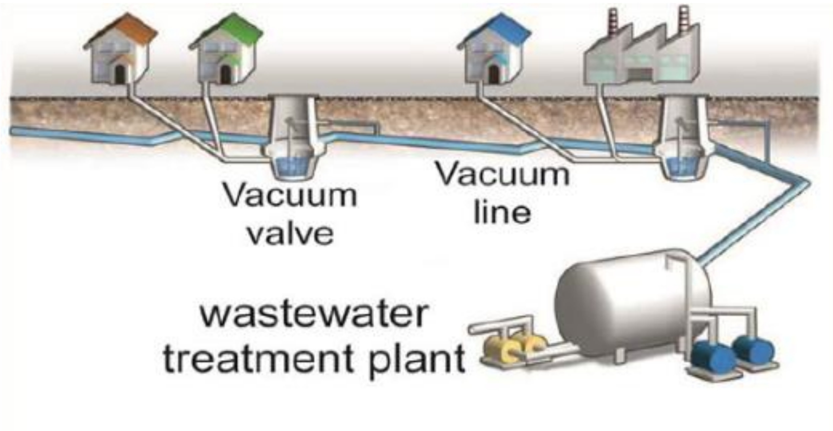

Figure 5: Example of a vacuum sewer system (Coway EnTech, n.d.).

As described by Otterpohl et al. (2004), the use of vacuum toilets and sewers in source separated systems brings a decrease in the volume of wastewater produced and saves fresh water. Furthermore, a highly concentrated wastewater stream with nutrients is produced. This is due to vacuum toilets requiring 0.5 – 1.2 L per flush (Gao et al., 2019) as compared with modern Swedish dual-flush toilets which require 2/4 L per flush (Avloppsguiden, n.d.).

An advantage with the vacuum system is that the pipes can have relatively small dimensions, leading to savings of space, costs and energy (Kjerstadius et al., 2012). Zeeman et al. (2008) reported that vacuum systems resulted in an energy usage of 90 MJ per person and year as compared with conventional sewers which required 108 MJ per person and year. Furthermore, the negative pressure causes homogenisation of the blackwater which is useful for treatment of the wastewater (Kjerstadius et al., 2012). One thing to consider is the noise level of vacuum systems, as relatively high noise levels have been noticed in the past.

2.4. Anaerobic digestion

Anaerobic digestion (AD) brings a conversion of organic matter into methane, in an oxygen-free environment with low amounts of excess sludge produced (de Graaff et al., 2010). Furthermore, nutrients are mostly conserved in the liquid phase of the wastewater and can be recovered downstream in the process. To make AD more efficient, a concentrated influent is of importance, which is why source-separating systems where less dilution of the wastewater occurs is a good pairing for AD (Wendland, 2008). This includes the use of low-flush or vacuum toilets in the collection system. The process of AD depends on several factors, such as the characteristics of the feed, pH, redox potential, temperature and mixing of the substrate (Kariyama et al., 2018).

There are four stages of AD: 1) Hydrolysis, 2) Acidogenesis, 3) Acetogenesis and 4) Methanogenesis (Deena et al., 2022). The stages are all dependent on activity from different microorganisms. During hydrolysis, insoluble organic complexes such as proteins and fats are broken down into soluble compounds such as amino acids and sugars. In the next step, the acidogenesis, acidogenic bacteria convert the compounds into volatile organic compounds such as volatile fatty acids (VFA). During acetogenesis, the acidic compounds are turned into acetate and hydrogen which then is consumed during methanogenesis, where acetate is converted into carbon hydroxide, hydrogen, and methane.

2.4.1. Biogas

As a result of AD biogas is produced, which mainly consists of methane (CH_4, between 50 – 65%), and carbon dioxide (CO_2, between 35 – 50 %), saturated with water vapour (Miltner et al., 2017). Traces of hydrogen sulphide (H_2S), and ammonia (NH_3) are also present in the biogas together with nitrogen and oxygen from air. Depending on which material is fed to the AD, pollutants may be found in the biogas, e.g. siloxanes and volatile hydrocarbons (ibid). To make use of the biogas, trace gases need to be removed and biogas upgrading takes place (Nguyen et al., 2021). Biogas upgrading decreases the volume of CO_2 in the gas and thus increases the calorific value, meaning the amount of heat produced during combustion, which makes it economically feasible to compress and transport the biogas for further use. Furthermore, reducing the amount of H_2S and water vapour decreases the risk of process problems downstream. Several biogas upgrading processes are available, for example water scrubbing and membrane separation. Internationally, the produced biogas is mainly used for electricity and heat (Nguyen et al., 2021). Looking at Sweden as an example, the upgrading of biogas to biomethane has led to the main use of biogas as vehicle fuel.

The combustion of biogas leads to CH_4 conversion into CO_2, and in case it is not a full conversion there will be CH_4 emissions (Paolini et al., 2018). CH_4 has a 28 – 36 times higher global warming potential than CO_2 and CH_4 emissions should therefore be minimized during biogas production. A significant contribution of CH_4 emissions come from biogas storing and sludge management. Nevertheless, the use of biogas and the successive decrease of fossil fuel dependence means that the biogas production mitigates the greenhouse effect (ibid).

Source-separating systems combined with vacuum sewers can produce more CH_4 than conventional systems, due to a higher amount of organic matter being treated with the AD (Kjerstadius et al., 2015 & other studies, Table 1). Based on 1 m^3 of CH_4, approximately 10 kWh of electrical energy can be produced (Suhartini et al. 2019). BW could bring an electrical energy production of 66 kWh/PE/year when excluding the FW, as seen by calculations using data from Kjerstadius et al. (2016). Based on the numbers in

Table 1, the CH_4 production from source-separating systems can yield approximately 50% more energy compared to conventional wastewater sludge AD. The average electrical energy production in the investigated conventional WWTPs is 69 kWh/PE/year whereas the investigated source-separating systems had an average of 103 kWh/PE/year (Table 1). As described by Kjerstadius et al. (2012), the varying production is due to differences in generation, collection, transport, treatment and digestion of the sludge.

Table 1: Electrical energy production in conventional and source-separating WWTPs according to a few studies. The source-separating systems include anaerobic co-digestion of BW and FW.

Conventional WWTPs (sludge AD)	Source-separating systems (BW AD)	Unit	Source
80	128	kWh/PE/year	Kjerstadius et al. (2016)
12.1	44.4	kWh/PE/year	Remy & Jekel (2008)
61	122	kWh/PE/year	STOWA (2014)
121.5	116.5	kWh/PE/year	Thibodeau (2014)

2.4.2. Sludge

The sludge after AD varies in quality depending on the origin of the organic matter which has been decomposed (Makádi et al., 2012). Furthermore, the SRT plays a role. A higher SRT, i.e. the time the sludge stays in the AD reactor, the less organic matter will there be in the sludge after digestion (ibid). A characteristic of sludge is whether it is liquid or solid, based on their TS content. Liquid sludges have a TS < 15% (ibid).

Harder et al. (2019) describes how pathogens accumulate in AD sludge to a certain extent, while highlighting the increased precipitation of heavy metals into the sludge compared with aerobic treatments due to sulphide precipitation. However, when combining source-separating wastewater systems with AD, lower concentrations of heavy metals are found in the BW to begin with due to no industrial effluents and surface run-off affecting the quality of the water (Tervahauta et al., 2014).

In conventional WWTPs, sludge is formed during several treatment steps (Elalami et al., 2019). Using the sludge in agriculture is the main way to recover nutrients, as it can contain 95% of incoming phosphorus and 27% of nitrogen (Kjerstadius et al., 2017). In a report by Malovanyy et al. (2022), it is stated that 13 – 34% of incoming nitrogen can end up in dewatered sludge in conventional WWTPs. The recycling of the nutrients then depends on the amount of sludge which is used for agricultural purposes.

In Sweden, the amount of sewage sludge re-used on agricultural land has reached over 50% in the recent years (Naturvårdsverket, n.d.). During 2022, the re-used sludge had average metal and nutrient levels as seen in Table 2. To reduce the spread of harmful substances by sludge re-use, Swedish WWTPs can become Revaq-certified (RISE, n.d.-a). This certification includes that the sludge does not have more than 22 mg Cd/kg P. Furthermore, products from source-separating wastewater treatment plants can be SPCR178 certified, and in that case the products must not exceed 17 mg Cd/kg P (RISE, n.d.-b). This certification includes maximum metal levels as seen in Table 2.

Table 2: The mean level of metals and nutrients in sewage sludge spread on agricultural land in Sweden, 2022 (Naturvårdsverket, n.d.), in the limit values according to the SPCR178 certification (RISE, n.d.-b), and in Öresundsverket's (ÖV) influent wastewater and digested sludge (NSVA, n.d.).

Element	Swedish sewage sludge (mg/kg TS)	SPCR178 limit values (mg/kg TS)	ÖV influent (µg/L)	ÖV sludge (mg/kg TS)
Cd	0.606	1	0.10	0.6
Cu	334	600	44	319
Ni	16.8	50	3.3	19.6
Pb	13.72	100	2.0	16.2
Zn	470	800	82	528
Hg	0.341	1	0.022	0.43
Cr	19.78	100	2.5	26.5
TN	49 510		39 000	61 535
TP	28 672		4 000	29 430

2.4.3. Reactor types

There are several types of reactors which can be used for AD. These include Continuous Stirred Reactor Tank (CSTR), Upflow Anaerobic Sludge Blanket (UASB) and an Accumulation (AC) system to name a few alternatives. The choice between reactor types depends on the incoming wastewater, e.g. the concentration of the BW or FW, and the end-use of the treated wastewater (Kujawa-Roeleveld & Zeeman, 2006).

In a CSTR, the substrate is regularly fed into the reactor while an equal amount leaves the tank (Kariyama et al., 2018). It is assumed that the HRT and SRT is equal. The HRT normally exceeds 15 days at mesophilic (20 – 40 °C) conditions (Wendland, 2008). In turn, this results in the need for a large reactor volume. Having a continuous stirring leads to homogeneity in the mixture and a uniform environment for the anaerobic bacteria (Kariyama et al., 2018). Although, the homogeneity usually leads to a lower digestion grade since some of the feed will exit the reactor without being digested (Kjerstadius et al., 2012). Some disadvantages with the CSTR include a high capital cost, and the operational and maintenance costs of the stirring (Kariyama et al., 2018).

In contrast to the CSTR, the UASB can have a longer SRT than HRT, which makes it possible to digest more of the organic matter since it stays longer in the reactor (de Graaff et al., 2010). In the UASB, the wastewater enters from the bottom and travels upwards through the reactor, which causes a stirring within the reactor (Dutta et al., 2018). The reactor can be described as having different zones, including the sludge bed, the sludge blanket and a separator, as shown in Figure 6. The sludge bed consists of granular biomass which has a high settling velocity, and it is where the main chemical reactions occur. Due to the formation of biogas in the reactor, as well as the upward motion of the wastewater, flocs become suspended and form the sludge blanket. The separator ensures that the produced biogas is collected, and that the liquid wastewater, known as decantate, exits through the overflow. By letting the wastewater pass through the biomass, there is a high efficiency of capturing and digesting the influent substrate which gives the advantage of a high SRT but a low HRT (Wendland, 2008).

According to Kujawa-Roeleveld & Zeeman (2006), UASB treatment of wastewater can result in a reduction of COD of up to 80-90%, depending on e.g. the temperature of the system. This is further shown in a study by de Graaff et al. (2010) where treatment of BW resulted in an average COD removal of 78% with a HRT of 8.7 days.

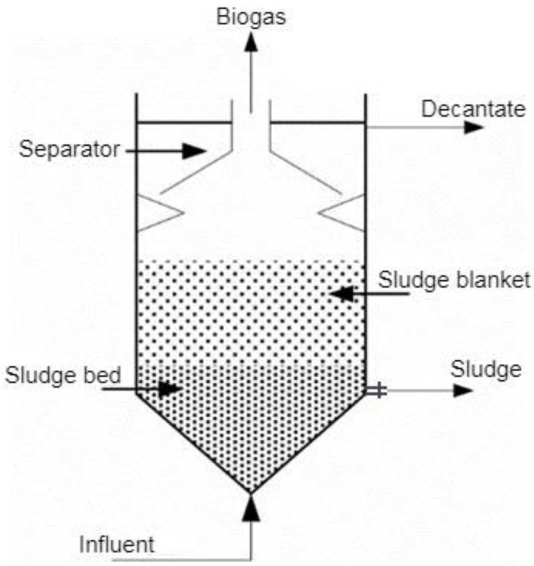

Figure 6: Schematic sketch of an UASB, based on figure from Dutta et al. (2018).

A variant of the UASB is the UASB – Septic Tank (UASB-ST), in which the main difference is an increased accumulation and stabilization of the sludge (Wendland, 2008). The UASB-ST does not have granules, but instead a flocculated sludge (Kjerstadius et al., 2012). The flocculated sludge needs a slower upwards movement of the wastewater, as compared to an UASB with granular sludge, and therefore the need for a larger reactor (Kujawa-Roeleveld & Zeeman, 2006). There is some accumulation of sludge in the UASB-ST, meaning that not all organic matter can be digested.

An AC system is a continuously fed reactor in which the reactor is emptied after maximum storage or desired reaction time has been reached (Kujawa-Roeleveld & Zeeman, 2006). It can be seen as the simplest AD system and can be combined with biogas collection. When emptying the AC system, some digested volume should be left in the reactor to increase the treatment of later incoming wastewater. The system should be used for concentrated waste, such as source separated BW, to keep the required volume relatively small.

2.4.4. Ammonia inhibition

One important factor to AD is the C:N-ratio, meaning the ratio between the mass of carbon and nitrogen in the substrate (Yang et al., 2023). Blackwater has a low C:N-ratio, due to the high levels of nitrogen in urine, which can lead to ammonia inhibition of the digestion process. The problem lies in high concentrations of free ammonia (NH_3) affecting the microorganisms, methanogens, during the methanogenesis stage (Wendland, 2008).

A study by Gao, Zhang et al. (2019) showed that BW from vacuum toilets generated lower amounts of biochemical methane potential than from conventional toilets, due to ammonia inhibition. This indicates the need to observe pH and ammonia in source-separating systems combined with AD. To counteract the low C:N-ratio of BW and the ammonia inhibition, a possibility is co-digestion with substrates of higher C:N-ratio (Yang et al., 2023). The addition of other substrates also increases the amount of organic matter and consequently the biogas yield. Different substrates can be chosen depending on for example biodegradability and availability, with FW being a common choice. An example of co-digestion is presented in Wendland et al. (2007), in which BW and FW were co-digested in a CSTR. The results were higher COD removal and methane production than when only BW is digested.

2.5. Nutrient recovery

Phosphorus is a common element in fertilizers, which traditionally has derived from excavated phosphate rock (Desmidt et al., 2015). It is described by Desmidt et al. that the peak of phosphate mining will be reached in the coming decades, which together with an increased demand of fertilizers lead to the need for alternative phosphorus sources. One option is to recover phosphorus from anaerobically treated wastewater since it brings high enough phosphorus concentrations, as the mineral struvite for example (de-Bashan & Bashan, 2004).

Fertilizers also generally contain nitrogen, which can be industrially produced from nitrogen in the air through the Haber-Bosch process (Smith et al., 2020). However, this process is energy intensive, consuming 1 – 2 % of the total energy usage in the world (Kyriakou et al., 2020), and has conventionally relied on fossil fuels as feedstock (Smith et al., 2020). This makes the Haber-Bosch process a large greenhouse gas emitter. Although there is potential to make Haber-Bosch more environmentally friendly, as described by Smith et al. (2020), an alternative is to recover nitrogen that is already in circulation.

2.5.1. Struvite precipitation

Struvite ($MgNH_4PO_4 \cdot 6H_2O$) has been known to spontaneously form in wastewater systems, causing consequences such as fouling of pipes, although extraction of the mineral is also possible (de-Bashan & Bashan, 2004). The formation of struvite is dependent on concentrations of phosphate, ammonium and magnesium being high enough for the equilibrium system to cause precipitation (Siciliano et al., 2020).

Precipitation of struvite affects both the nitrogen and phosphorus concentrations in wastewater, as seen by the mineral formula. The N:P molar ratio for struvite is 1:1 while according to Zeeman et al. (2008), BW can have a N:P ratio ranging from 35:1 to 42:1. This means that almost all phosphorus, in theory, could be extracted. However, only up to 3% of nitrogen would be incorporated in the precipitate. In wastewater, magnesium will limit the struvite precipitation and it can therefore be added in form of e.g. $MgCl_2$ or MgO (Desmidt et al., 2015). The pH also affects the precipitation; however, the optimum pH level varies depending on the characteristics of the wastewater.

Siciliano et al. (2020) concluded that pH in the ranges of 7.6 – 10.5 have been suitable for struvite precipitation. For BW, de Graaff et al. (2011) showed a phosphorus removal efficiency of 90% in an UASB effluent by having the molar ratio of Mg:P at 1.5:1 or having a pH of 8.0. A study by Gell et al. (2011) showed that BW previously treated in an UASB reactor gave struvite with suitable levels of phosphorus, nitrogen and magnesium for use as a fertilizer. Furthermore, the levels of heavy metals and pathogens were below the regulatory limits, thus pointing to it being a safe fertilizer.

Considering the additives, $MgCl_2$ has proven to be successful in aiding the precipitation process (Siciliano et al., 2020). Though a downside to $MgCl_2$ includes the chloride negatively affecting the struvite quality. If using MgO instead, acid dissolution of the additive is needed since MgO has a low solubility. However, the downside of chloride is counteracted with the use of MgO. MgO can also be used as an alkaline reagent to raise the pH to sufficient levels (ibid). Another method for increasing the pH is wastewater aeration, in which CO_2 is stripped from the solution affecting the carbonate balance of the water and causing the pH to increase.

A simple reactor which is commonly used for struvite precipitation is the stirred tank reactor (Siciliano et al., 2020). The main feature of the tank is a mixing system to make the additives and the wastewater homogeneous, though there are additional features to control the process. These reactors can work continuously or in batch mode and can also be combined with wastewater aeration. The aeration process not only increases the pH but also contributes to the mixing of the waste, with the airflow rate being an important factor to reach high struvite precipitation rates.

2.5.2. Ammonia stripping

A technology for the previously mentioned nitrogen recovery is ammonia stripping, which is suitable for effluents from anaerobic digesters (Kujawa-Roeleveld & Zeeman, 2006). The nitrogen in the effluent will be mainly found as ammonium (NH_4^+); however, when raising the pH it will be converted to ammonia (NH_3). Ammonia stripping is conducted by applying a gas flow to an ammonia rich wastewater, causing NH_3 to transfer to the gas. This is done in stripping towers, which can be of two types: either air- or steam stripping. To access the nitrogen from the stripping towers, treatment of the gas can be done by scrubbing. One common method is acid scrubbing, in which e.g. sulfuric acid (H_2SO_4) adsorbs the NH_3 and forms ammonium sulphate ($(NH_4)_2SO_4$) (Wu & Vaneeckhaute, 2022).

Application of ammonia stripping could give >90% recovery of ammonium in wastewaters with ammonium concentrations larger than 1000 mg/L (Wu & Vaneeckhaute, 2022). Air stripping also requires an elevated temperature of the wastewater. Furthermore, a pH above 9.25 is required, making it possible to combine ammonia stripping with struvite precipitation, which also requires an alkaline environment (Wu & Vaneeckhaute, 2022). If ammonia stripping is conducted after struvite precipitation, the pH must be maintained between the treatment processes due to the acidity formed during struvite precipitation. The use of ammonia stripping could result in a recovery of 69% of nitrogen in BW when vacuum systems are used for collection (Kjerstadius et al., 2016).

2.6. Fertilizers from wastewater

A common conventional practice of nutrient reuse is the spreading of combined sewage sludge on farmlands. However, hazardous chemical substances in the sludge makes this a controversial topic (Hudcová et al., 2019). These substances include for example heavy metals, organic chemicals such as polycyclic aromatic hydrocarbons (PAHs) and polychlorinated biphenyls (PCBs), nanoparticles and pharmaceuticals including antibiotics (Fijalkowski et al., 2017; Hudcová et al., 2019). The hazardous substances as well as the negative perception of human excreta means that the social acceptance of wastewater derived fertilizers is an issue (Barquet et al., 2020). However, the use of source-separating systems leads to increased possibilities of advanced treatment methods to decrease the levels of dangerous substances (Otterpohl et al., 2004).

Furthermore, nutrients can be recovered to a greater extent in source-separated systems, as described in previous sections. Struvite works well as a fertilizer, especially due to its low solubility leading to a slow release of nutrients which is suitable for plant growth (Siciliano et al., 2020). Depending on the additives used, there are several possible products from ammonia stripping, which can be used as fertilizers (Harder et al., 2019). These include ammonium sulphate, ammonium borate, ammonium chloride and ammonium nitrate. The process of ammonia stripping and the subsequent acid sorption means a phase separation between the BW and the product (Bisschops et al., 2019). In turn, pathogens, organic pollutants and heavy metals are generally not found in the ammonium products (Harder et al., 2019). On the other hand, struvite precipitation occurs together with the BW and if other solids are precipitated together with the struvite there is a risk for pollutants (Bisschops et al., 2019).

To reach a circular economy, sustainable waste management is pivotal and this includes the recovery of materials for use as fertilizers (Rizzioli et al., 2023). The authors describe how up until 2019, the EU did not have legislation which covered nutrient recovery and production of biobased fertilizers. This meant that the production of fertilizers, e.g. from wastewater, was discouraged. In the recent years, the European Commission has worked to recognize organic and waste-based fertilizers, to improve the European market for nutrient recovery (European Parliament, 2019).

2.7. Mass Balances

Mass balances are used to quantitively evaluate the materials which enter, leave and accumulate within a system with defined boarders, based on the law of conservation of mass (Von Sperling & Chernicharo, 2005). In other words, mass is not created or destroyed. A mass balance can be expressed according to the following general equation:

$$Input = Output + Accumulation. \quad (1)$$

Steady state can be assumed, and then accumulation is negligible (Von Sperling & Chernicharo, 2005). In that case, flows and concentrations within the system are assumed constant.

When doing mass balance calculations in WWTPs, one parameter which can be used is total solids (TS). TS is the material left after evaporation and drying of a sample (Lipps et al., 2017a). TS is the sum of dissolved solids plus suspended solids in water, where dissolved solids pass through a 2 µm filter while suspended solids (SS) do not (ibid).

Nitrogen and phosphorus are also of interest for mass balance calculations of wastewater, especially for source-separation systems such as RecoLab where nutrient recovery is a driver for the system. Nitrogen can be measured as total nitrogen (TN), which consists of organic nitrogen, ammonia (NH_3), nitrate and nitrite (Lipps et al., 2017b). The sum of the organic nitrogen and ammonia can be analysed as the Total Kjeldahl Nitrogen. Generally, nitrate and nitrite are found in low concentrations in raw wastewater (ibid). Phosphorus concentrations can be measured as total phosphorus (TP) (Lipps et al., 2017c), with most of the phosphorus in wastewater found as phosphates (PO_4^{3-}) which is inorganic (Tchobanoglous et al., 2003). The phosphorus can be either dissolved or suspended.

As described by Li et al. (2018), the chemical oxygen demand (COD) is a measurement of how much oxygen is needed to oxidize organic compounds, by adding strong chemical oxidants such as potassium dichromate. COD is deemed as an important water quality parameter for the level of organic pollution. A theoretical calculation of the COD of an organic compound $C_nH_aO_b$ is based on the complete oxidation of the compound (Henze et al., 2008):

$$C_nH_aO_b + \frac{1}{4}(4n + 1 - 2b)O_2 \rightarrow nCO_2 + \left(\frac{a}{2}\right)H_2O \quad (2)$$

The theoretical oxidation of the organic compound can then be calculated according to (ibid):

$$COD_{tot} = \frac{8(4n+a-2b)}{12n+a+16b}\left[\frac{g\ COD}{g\ C_nH_aO_b}\right] \quad (3)$$

2.8. Öresundsverket

The effluent streams from RecoLab go to the main sewage line of the connecting Öresundsverket, which is the conventional WWTP for the municipality of Helsingborg (NSVA, n.d.). Öresundsverket has around 149 000 PE connected. The treatment process consists of mechanical treatment in sedimentation basins, biological nitrogen and phosphorus treatment with activated sludge and a final sedimentation step with a sand filter (NSVA, n.d.). Sludge from the sedimentation and biological treatment is degraded in an AD, with biogas production. Then, the digested sludge is dewatered. Shown in Table 2 is the average metal and nutrient concentrations in Öresundsverket's influent and sludge. The sludge had an average of approximately 20 mg Cd/kg P.

3. Method

To make the mass balances, several assumptions were made which may affect the results. These assumptions are summarized in Table 3.

Table 3: Assumptions made regarding sampling, sample analysis and data analysis.

Affected	Assumption
Water vapour in biogas	Has a negligible effect on the mass balances for liquid mass.
Drum filter reject	Has a negligible effect on all studied mass balances.
Methane levels in biogas	Average level of methane content observed before this study is representative for all measurements of biogas during 2023.
TS and TSS	The measurements are equal; thus, dissolved solids have no impact.
TN and Kjeldahl-N	The measurements are equal; thus, concentrations of nitrate and nitrite are assumed to be negligible.
Concentrations measured monthly	Representative for all weeks of the month. Variations over the month are neglected.
Decantate volume	Is equal to the influent volume subtracted with the sludge volume, due to no decantate flow samples taken.
Struvite effluent volume	Is equal to the decantate volume, due to no flow measurements available.
Missing or excluded values	The average of the rest of the dataset is a good replacement to complete the dataset.
Density of inflow, decantate and sludge	Is equal to the density of water.
TS lost by biogas	Is the remaining TS mass from the other flow paths.
TN lost by aeration	Is the remaining TN mass from the other flow paths.

3.1. Mass balance boundaries

The system boundaries were the limits considered in the calculations done in this study, shown in Figure 7 below. Two different boundaries were considered, one data-based and one which included a hypothetical calculation. The data-based boundary included the influent BW until the struvite effluent, thus not including the ammonia stripper. The reason for that being the ammonia stripper at RecoLab was not yet working properly, with no available data to make mass balances with. Instead, hypothetical mass balances concerning the potential to recover nitrogen in the ammonia stripper was done.

The inflow to the data-based mass balances was the incoming BW. Outflows include the biogas, sludge, drum filter reject, solid struvite, aeration during the struvite precipitation and the struvite effluent. As seen in Figure 7, certain flow paths affected certain mass balances, shown by the elements within parenthesis. Practical experience with the system has shown that the water vapour which leaves with biogas as well as all losses from drum filter reject is assumed to have a negligible effect on the mass balances and was therefore excluded.

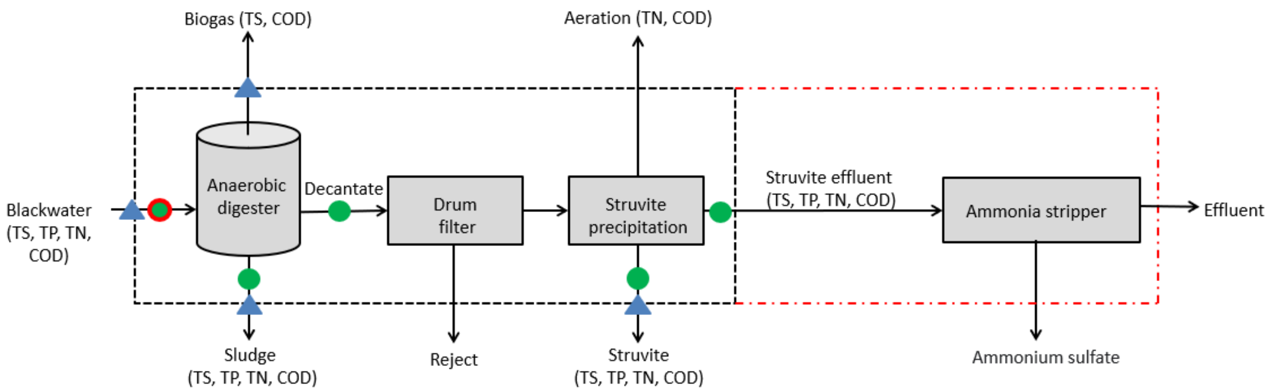

Figure 7: The system boundary of the data-based mass balance calculations is within the black dotted line. Grab sample locations are shown by the green dots while flow proportional sampling is shown by the green dot with red contour. The blue triangles show where flow or weight was measured. In parentheses are the parameters which are affected by that flow path. The red line shows the additional limit of the hypothetical mass balance calculation.

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3.2. Sampling locations and occurrences

All the sampling was conducted by personnel at NSVA. Locations of flow and weight measurements are shown in Figure 7.

Measurements of the incoming BW, using a flow proportional sampler, were taken daily. Sludge was emptied from the UASB-ST twice a week. Larger than normal volumes were withdrawn at the beginning of 2023, with the aim to decrease the height of the sludge blanket in the UASB-ST reactor. These increased sludge withdrawals were done from a height of 3 m. After half the year passed, the increased sludge withdrawals stopped and instead smaller volumes were emptied from the bottom of the reactor, which helped achieve a more stable SRT and less suspended solids in the digester decantate.

The biogas volume was measured daily, in room temperature of 20°C and an overpressure of 50 mbar. At the same time the CH_4 content of the biogas was measured. However, due to faulty equipment, the CH_4 levels were inaccurately measured between January – December 2023.

The weight of solid struvite was measured when the bag, see Figure 8, had been exchanged and dried in room temperature, thus in a semi-dry state. The changing of the bags was random, in total occurring four times in 2023. See Appendix B for when the struvite bags were exchanged.

Figure 8: A bag in which struvite is collected.

Flow proportional samples of the incoming BW, and grab samples of the decantate, the sludge, the solid struvite and the struvite effluent were conducted in different locations (Figure 7) and time schedules (Table 4). For the concentrations in the sludge, the monthly measured levels were assumed to be representative for the whole month.

Table 4: The location and frequency of parameters which were sampled.

Location	Parameter	Frequency
Influent	TS, TP, COD	Weekly
Influent	TN	Twice a week
Decantate	TS, TP, TN, COD	Weekly
Sludge	TS, TP, TN, COD	Monthly
Struvite	TS, TP, TN	When semi-dry
Struvite effluent	TS, TP, TN, COD	Weekly

3.3. Sample analyses

The samples from the influent, sludge and solid struvite were analysed by SGS Analytics Sweden AB laboratory. The influent was also analysed by the in-house NSVA laboratory, together with analyses of the decantate and the struvite effluent. Thus, two datasets of concentrations in the influent BW were available. Overall, the influent analyses conducted by the SGS laboratory showed lower standard deviations as well as higher numbers of analysed samples compared with the locally analysed NSVA samples (Appendix A). If excluding data points based on concentrations outside of two standard deviations from the mean, fewer samples had to be excluded from the influent SGS data set than the NSVA data as illustrated in Appendix A. Based on these differences between the laboratories, the calculations were conducted using the SGS data set.

The samples were analysed with the methods shown in Table 5. The NSVA laboratory performed Hach Lange cuvette tests for the TP, TN, and COD, with the product numbers shown within parenthesis in the table below.

Table 5: Analysis methods of TS, TSS, TP, TN, Kjeldahl N, and COD in the different mediums conducted by the in-house NSVA laboratory (shown in blue) and the SGS laboratory (shown in black).

Parameter	Method	Medium
TS	SS-EN 12880:2000	Sludge, struvite
TSS	SS-EN 872:2005	Influent, decantate, struvite effluent
TP	Hach (LCK350)	Influent, decantate, struvite effluent
TP	Hach (LCK349)	Decantate, struvite effluent
TP	Hach (LCK348)	Decantate
TP	SS-EN ISO 15681-2:2018	Influent
TP	SS-EN ISO 54321:2021	Sludge
TP	SS-EN 16171:2016	Sludge
TP	SS-EN ISO 11885:2009	Struvite
TN	Hach (LCK338)	Influent, decantate, struvite effluent
TN	Hach (LCK138)	Influent, struvite effluent
TN	Hach (LCK238)	Decantate
TN	SS-EN ISO 20236:2021	Influent
Kjeldahl N	SS-EN 16169:2012	Sludge
Kjeldahl N	SS-EN 13342	Struvite
COD	Hach (LCK514)	Influent, decantate, struvite effluent
COD	ISO 15705:2002	Influent, sludge

For the sake of this study, an assumption was made that the analysed concentrations of total suspended solids (TSS) were equal to the total solids (TS), due to the lack of consistent analyses. This means that dissolved solids are neglected, which brings a limitation in the accuracy of the study. Furthermore, analyses of TN and Total Kjeldahl Nitrogen are used interchangeably, neglecting any concentration of nitrate or nitrite which were assumed to be negligible and not influence the results.

3.4. Data analysis

Some data was excluded from the dataset (Table 6) for various reasons, as described in the following paragraphs.

The daily measured flows were summed to form accumulative weekly flows. Weekly flows which were outside of two standard deviations from the average were excluded from the dataset (two weekly flows in total). The monthly sludge concentrations were, as mentioned previously, considered representative for the whole month. Therefore, the sludge withdrawal volumes per month were divided by the number of weeks in the month, to get an average weekly sludge volume to compare with the concentration. Due to the increased sludge withdrawal volumes, from 3 m height, which is not representative of the analysed sludge concentrations, the initial sludge volumes up until week 26 were excluded from the dataset. Furthermore, sludge removal from 3 m of the digester was not deemed representative for normal operation by NSVA personnel (H. Kjerstadius, personal communication, 2024).

Due to a non-functional methane sensor, an average CH_4 content in the biogas was deemed a representative replacement to use in further calculations. The average was calculated using measurements of the CH_4 content from June 2021 to January 2023, before the equipment was faulty.

No flow measurements, besides measurements of the struvite weight, were taken after the AD as shown in Figure 7. Therefore, the mass balance calculations after the AD are based on calculations of the volume of BW. The decantate volume was assumed to be the volume of incoming BW subtracted with the sludge volume. Furthermore, this calculated volume was assumed to stay constant throughout the struvite precipitation process and is thus used as the struvite effluent volume.

The initially analysed TS concentrations in the sludge were deemed to be obviously false as they reached above 9%, which is too high considering that no sludge thickening process was used. Therefore, similarly to the sludge volumes, the sludge TS values up until week 26 were also excluded from further calculations. Due to variations of the data quality, certain concentrations of TP, TN and COD in the influent BW, the decantate and the struvite effluent were excluded from the dataset. The excluded values were outside of 2 standard deviations from the average value for the sample in question. Furthermore, one additional value of TN in the struvite effluent was excluded due to it being obviously false, as it was close to 0 mg TN/L (compared to the average value of 906 mg TN/L).

Table 6: Number of samples (n) used in analysis together with the number of excluded values. Samples analysed by the NSVA laboratory in blue and by SGS in black.

Parameter	Influent		Decantate		Sludge		Struvite		Struvite effluent	
	n	**Nr of excluded**	**n**	**Nr of excluded**	**n**	**Nr of excluded**	**n**	**Nr of excluded**	**n**	**Nr of excluded**
Flow/weight	50	2	-	-	7	5	5		-	-
TS	37	3	42	1	6	5	10	0	28	0
TP	47	1	33	3	10	0	10	0	26	3
TN	90	3	32	1	10	0	10	0	26	2
COD	47	1	36	2	10	0	-	-	28	2

The weight of the solid struvite was divided into weekly values of produced struvite. Due to the analysis of struvite was done in a semi-dry state, a conversion to the original wet weight of the struvite was needed for the wet flow mass balance. The conversion was made based on one weight difference of fresh, wet struvite compared with the dry struvite weight. A TS calculation according to,

$$TS\ (\%) = \frac{dry\ weight\ (kg)}{wet\ weight\ (kg)} * 100, \tag{4}$$

resulted in a TS of 34% in the wet sample. The TS value was assumed to be the same for all wet struvite, and thus all semi-dry weights could be converted to their original wet weight. A picture of the difference between the semi-dry and wet struvite is found in Appendix B.

Mass balances were calculated for all weeks of 2023. To get a full dataset, weeks where data was missing or excluded were replaced with the average values of weeks with existing data.

To make the wet matter mass balance, the measured and calculated volumes of influent, decantate and sludge were multiplied with the density of water, assuming all flows had the same density. These masses, together with the wet weight of struvite completed the dry matter balance. The struvite precipitation process started working properly in mid-2023. To make representative mass balances, mass flows of the struvite precipitation and the aeration were only calculated from July 2023. Due to only four bags of struvite being sampled for chemical analysis in the last half of 2023, average struvite concentrations of metals and nutrients were based on all struvite bags, thus from 2021 and forward.

After calculating the mass balances in the unit of kg per week, the averages were transformed to g per person and day using an estimated PE value based on default values from the URWARE model (Jönsson et al., 2005).

3.4.1. Total solids calculations

To calculate the weekly TS mass flows of the inflow, the AD decantate and the struvite effluent, equation (5) was used.

$$TS_{inflow,\ decant.,\ str.effl.} \left[\frac{kg}{week}\right] = flow \left[\frac{m3}{week}\right] * 1000 * \frac{TS\ concentration\ [\frac{mg}{L}]}{10^6}. \tag{5}$$

The TS which leaves through biogas is assumed to be the mass which does not go to the decantate or the sludge, i.e.:

$$TS_{biogas} \left[\frac{kg}{week}\right] = TS_{inflow} - TS_{decant.} - TS_{sludge}. \tag{6}$$

The sludge mass flows were calculated using the density of water, $\rho = 0.997\ kg/L$, according to equation (7).

$$TS_{sludge} \left[\frac{kg}{week}\right] = Accumulated\ sludge \left[\frac{L}{week}\right] * \rho * \frac{TS\ [\%]}{100}. \tag{7}$$

Then, using the semi-dry weight of the struvite:

$$TS_{struvite} \left[\frac{kg}{week}\right] = struvite\ mass \left[\frac{kg}{week}\right] * \frac{TS\ [\%]}{100}. \tag{8}$$

According to the mass balance boundary in Figure 7 and equation (1), the mass outflows were added together and compared with the inflow. These calculations were done for all weeks of 2023. Then, average values over the year for all mass flows were calculated.

3.4.2. Total phosphorus calculations

Similarly to the TS calculations, TP mass flows in the inflow, the decantate and the struvite effluent were determined by using equation (5), by exchanging TS with TP.

The sludge and solid struvite mass flows were calculated using the results from (7) and (8) respectively:

$$TP_{sludge,\ struvite}\left[\frac{kg}{week}\right] = TS_{sludge,\ struvite}\left[\frac{kg}{week}\right] * \frac{TP\ concentration\left[\frac{g}{kg\ TS}\right]}{1000}. \tag{9}$$

Then, the mass outflows were added and average mass flows over the year calculated.

3.4.3. Total nitrogen calculations

The TN mass balance was calculated in the same way as the TP balance, except for the aeration during the struvite precipitation process affecting TN. Due to the aeration not being sampled, the TN lost to aeration was assumed to be equal to the difference between the influent and effluent mass of nitrogen, i.e. equal to the TN which does not go to the solid struvite or the struvite effluent, as summarized in equation 10,

$$TN_{aeration}\left[\frac{kg}{week}\right] = TN_{decant.} - TN_{str.\ effl.} - TN_{struvite}. \tag{10}$$

To calculate the hypothetical nitrogen recovery in the ammonia stripper, an assumption of an ammonium nitrogen (NH_4-N) recovery rate of 78.4% was used based on a study by Sagberg & Grundnes Berg (2000) who studied the ammonia stripper of VEAS in Oslo. For that reason, every measured NH_4-N concentration in the struvite effluent was multiplied by 0.784. Then, to convert the NH_4-N concentrations into TN concentrations which was needed for the mass balances, the ratio between the measured TN and NH_4-N concentrations was used. The mass of recovered TN in the form of ammonium sulphate was calculated similar to equation (5).

3.4.4. Chemical oxygen demand calculations

Similarly to previous descriptions, the COD in the inflow, decantate, and the struvite effluent were calculated according to equation (5), by exchanging TS with COD. The COD in the sludge was calculated by equation (9).

Based on equation (2) and (3), the theoretical COD of the oxidation of CH_4 was calculated considering the general formula for organic compounds $C_nH_aO_b$ according to:

$$COD = \frac{8(4n+a-2b)}{12n+a+16b} = \frac{8(4+4)}{12+4} = 4\ \frac{g\ COD}{g\ CH_4}. \tag{11}$$

At the standard temperature and pressure (STP), 0°C and 1 atm pressure, the density of CH_4 is $0.72\ g/m^3$. Thus, the COD can also be described as 1 g COD per 0.35 L CH_4.

To calculate how much COD theoretically leaving with the biogas, the volume of the measured biogas was converted to standard litre, at the conditions of STP, by using the ideal gas law:

$$p * V = n * R * T, \tag{12}$$

where p (atm) is the pressure, V (L) the volume, n (mol) the number of moles of gas, R the ideal gas constant and T (K) the temperature. Due to the number of moles being constant, the ideal gas law is transformed to the following equation used to convert the volume of biogas to STP:

$$V_{STP} = \frac{p*V*T_{STP}}{T*p_{STP}}. \tag{13}$$

The weekly CH_4 production at STP was then calculated using the biogas volume at STP from (13) and multiplying with the average RecoLab CH_4 content in the biogas. Then, to calculate the COD in [kg/week]:

$$COD_{biogas} \left[\frac{kg}{week}\right] = CH_{4_{STP}} \left[\frac{m^3}{week}\right] * \frac{1}{0.35} \tag{14}$$

No analysis of the COD reduction, occurring during the struvite precipitation process, was done by the laboratories. Therefore, based on a 30 – 35% reduction of COD in struvite precipitation according to Diamadopoulos et al. (2007) the mass of COD in the struvite was calculated by:

$$COD_{struvite} \left[\frac{kg}{week}\right] = COD_{decant.} * 0.35 \tag{15}$$

4. Results

4.1. Quantity and quality of the blackwater

Considering the incoming flow from Oceanhamnen, including all three streams of wastewater, small fluctuations in the volume were seen, as graphically illustrated in Figure 9. These fluctuations occurred regularly over the course of 2023. Of the approximately 70 000 m^3 of source-separating wastewater incoming to RecoLab, BW contributed with an average of 64 ± 8.6 m^3/week (see Table 7) to the total flow. In contrast, the nearby Öresundsverket both had a larger influent volume and spikes in the volume on a few occasions.

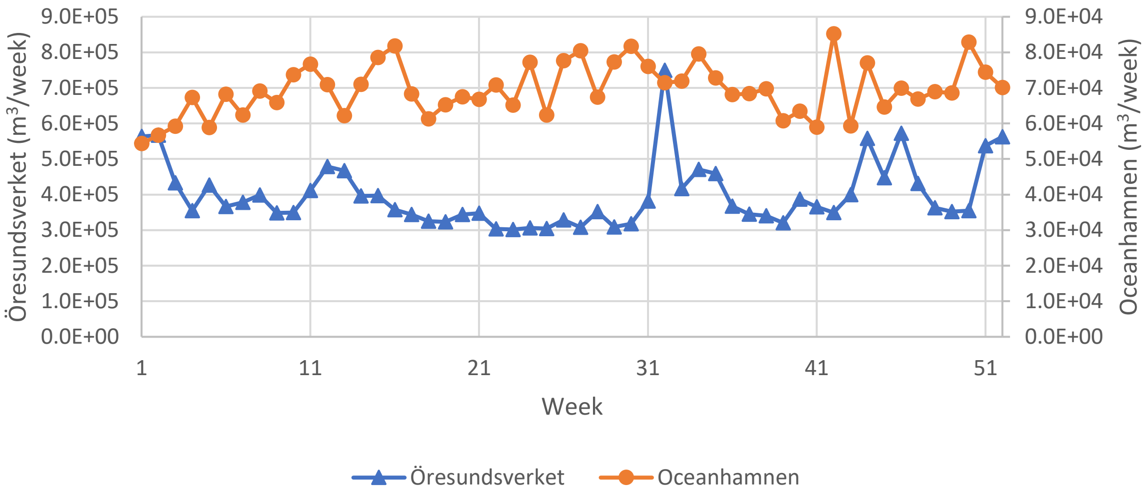

Figure 9: The accumulated flow per week of 2023 for Öresundsverket and RecoLab, including all streams of wastewater.

Seen in Table 7 are the average BW incoming flow and concentrations of TS, TP, TN, and COD, which all had a standard deviation < 25%. The COD:N ratio was 6.6:1. By using a linear relationship between COD and TOC as described by Dubber & Gray (2010), the TOC:N in the influent was 2.2:1. The metals with the highest concentration levels in the BW influent were Zn, Cu, Ni, and Cr, which had average concentrations of 1175, 174, 38 and 31 µg/L respectively. Cd, Pb and Hg had average concentrations beneath 2.3 µg/L. The influent had an average of 8.1 mg Cd/kg P.

Table 7: Flow and mean concentrations of TS, TP, TN, COD, and metals in the influent BW together with the standard deviation (SD) and the number of samples (n) included in the calculations for the year 2023.

Parameter	Mean	SD	Unit	n
Flow	64	8.6	m^3/week	50
TS	4 108	880	mg/L	37
TP	123	19	mg/L	47
TN	1311	112	mg/L	90
COD	8757	1526	mg/L	47
Cd	1.0	0.11	µg/L	11
Cu	174	28	µg/L	11
Ni	38	9.5	µg/L	11
Pb	2.3	0.32	µg/L	11
Zn	1175	130	µg/L	11
Hg	0.14	0.12	µg/L	11
Cr	31	12	µg/L	11

While conventional WWTPs base the estimation of PE on the levels of BOD found in the wastewater, source-separating wastewater systems bring a challenge in which the BW does not have the same levels of BOD as the GW does (Jönsson et al., 2005). Therefore, an estimation of RecoLab having a load of 885 PE was based on an average of the default values from the URWARE model of COD, BOD_7, TP and TN in BW (Table 8). The PE based on the flow and the TS were excluded due to them varying from the rest. The estimation of PE was used to illustrate the mass balances.

Table 8: Default values of COD, BOD_7, TS, TP, and TN in BW (faeces + toilet paper + urine) according to the URWARE model described in Jönsson et al. (2005). The default BW flow of vacuum-based systems is based on Wendland et al. (2007). The average weights of the parameters in RecoLab's influent are seen together with an estimated amount of pe.

Parameter	Default values	RecoLab	Pe
COD	71 (g/pe/day)	429 (kg/week)	860
BOD_7	38 (g/pe/day)	247 (kg/week)	919
TS	71 (g/pe/day)	260 (kg/week)	522
TP	1.4 (g/pe/day)	7.8 (kg/week)	802
TN	12 (g/pe/day)	83 (kg/week)	957
Flow	5 (L/pe/day)	63.52 (m^3/week)	1815
Estimated			885

4.2. Mass balances

For the wet mass balance, the average outflows were 100% of the average inflow (see Table 9). Thus, the wet mass is considered as a “closed mass balance” (i.e. where inflows equals outflows). Regarding the TS and COD mass balances, the average outflows were 91 and 104% of the inflow respectively. Furthermore, the TP and TN mass balances had average outflows which were 55 and 92% of the inflow respectively. Hence, TS, TP, TN, and COD were all considered to have had “open mass balances” (i.e. where inflows and outflows are not equal) since the outflows/inflow ratio was not equal to 100%. However, only the TP balance showed major inconsistencies when comparing the inflow and outflows.

Table 9: Mean wet mass and mass of TS, TP, TN, and COD in the outflows from the mass balances together with standard deviations (SD) and the number of samples used in analysis (n).

	Wet mass			TS			TP			TN			COD			
	Mean	**SD**	**n**	**Mean**	**SD**	**n**	**Mean**	**SD**	**n**	**Mean**	**SD**	**n**	**Mean**	**SD**	**n**	**Unit**
INFLOW	63334	8315	50	260	55	37	7.8	1.6	47	83	12	90	553	97	47	kg/week
OUTFLOWS																
Biogas	-	-	-	181*	60	-	-	-	-	-	-	-	429	110	51	kg/week
Sludge	1023	151	26	30	4.3	5	1.5	0.5	10	1.7	0.6	10	20	8.3	10	kg/week
Struvite	16	7.3	5	3.8	1.8	10	0.68	0.30	10	0.30	0.13	10	57*	34	-	kg/week
Aeration	-	-	-	-	-	-	-	-	-	37*	18	-	-	-	-	kg/week
Struvite effluent	62311*	8331	-	25	15	28	2.2	0.6	26	56	16	27	85	47	28	kg/week
SUM OUTFLOWS	63342	8315		237	62		4.1	0.8		77	26		562	142		kg/week
Outflows/Inflow	100	0.016		91	14		54	15		92	25		104	29		%

**Only based on calculations*

As shown in Figure 10, there were variations of the ratio between the outflow and the inflow over the course of 2023. The outflow and inflow ratio for the TS and COD balances had average standard deviations of 14 and 29% respectively (Table 9) which can be explained by the TS mass balance consistently varying just below or at 100%, whereas the COD had an increasing trend over the year. By the end of the year, more COD was found in the outflow than in the inflow. A linear regression analysis showed that there was a statistically significant correlation between the COD outflow/inflow ratio and the week number. The point at which the struvite precipitation process was added to the calculations is shown by the red vertical line in Figure 10. No difference between inflow and outflows of TS and COD due to the struvite precipitation is seen.

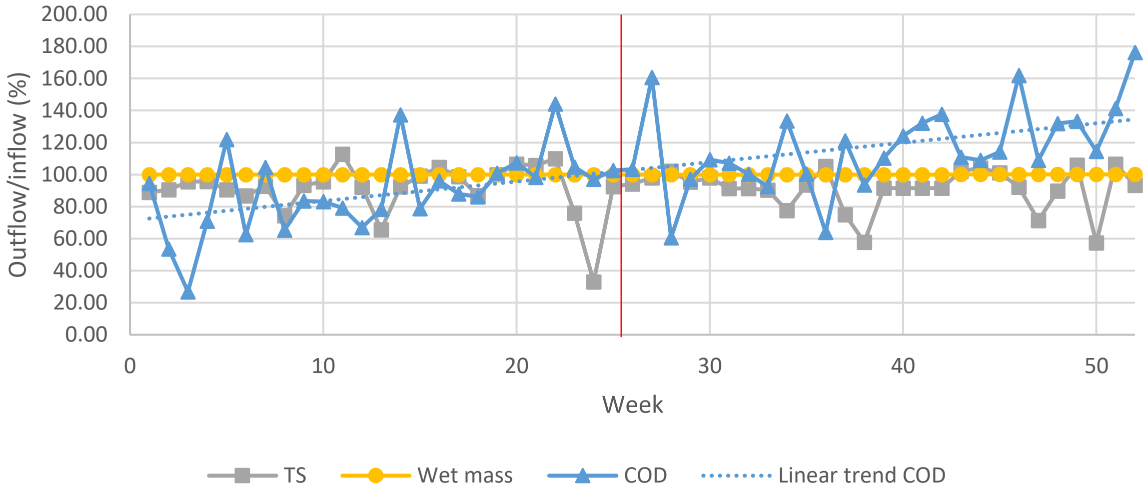

Figure 10: The variation of the ratio of outgoing and incoming wet mass, TS, and COD over the course of 2023. The dotted blue line is the linear trend line of the COD outflow/inflow ratio. The red line shows when the struvite precipitation process was added to the calculations.

The variation of TP and TN over 2023 is seen in Figure 11. Before the struvite precipitation process, less TN was found in the outflow than the inflow at a rather consistent difference of approximately 70%. When the struvite precipitation was considered, the difference between outflows and inflow for TN varied around approximately 110%. Looking at TP on the other hand, less TP was consistently found in the outflows than in the inflow.

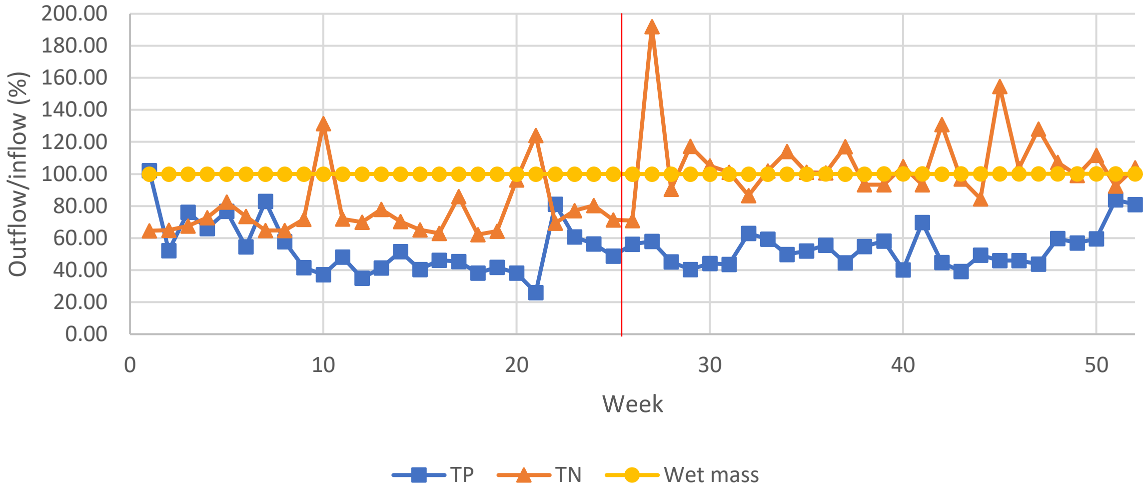

Figure 11: The variation of the ratio of outgoing and incoming wet mass, TP, and TN over the course of 2023. The red line shows when the struvite precipitation process was added to the calculations.

4.2.1. Wet mass balance

The wet mass was approximately 10 kg/pe/day into RecoLab during 2023, shown in Figure 12. While some wet mass went to the sludge and the struvite, 170 and 2.6 g/pe/day respectively, nearly all the wet mass continued to the decantate and struvite effluent. Both the decantate and effluent masses were based on calculations, and therefore the mass balance can be considered as closed.

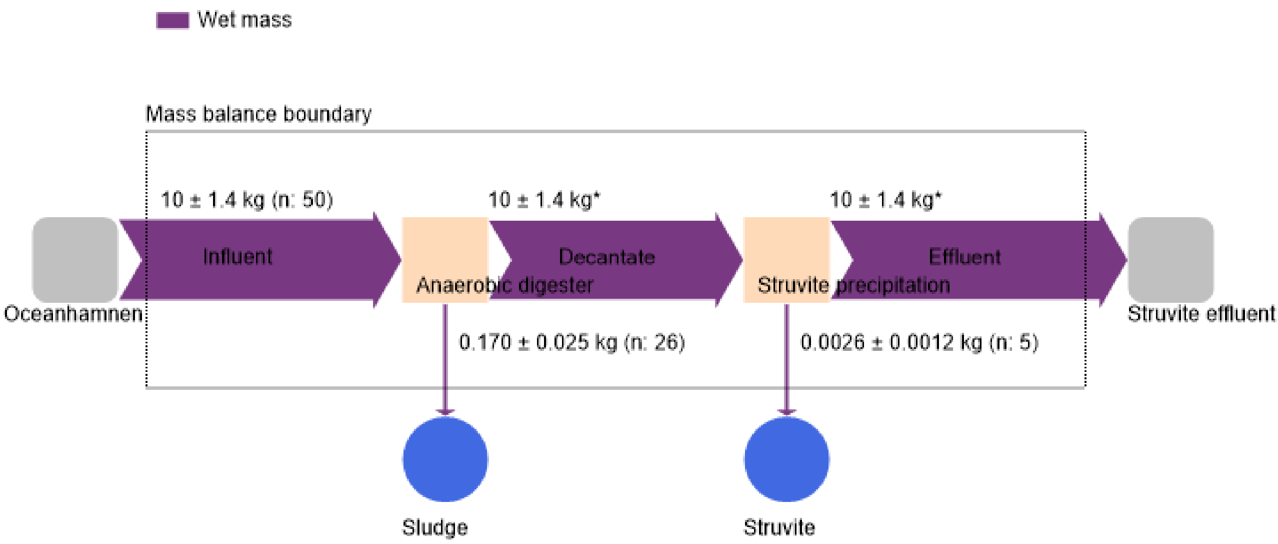

*Figure 12: The wet matter mass balance per PE and day, including standard deviation and the number of data points used for the calculations. * Indicates that no samples were taken of the stream.*

4.2.2. *Total solids and COD mass balance*

In the influent BW, approximately 89 and 42 g/pe/day respectively of COD and TS was found. A large fraction of COD and TS, 78 and 69% respectively, was converted to biogas in the AD. In other words, 831 NL CH_4/kg TS was formed. When considering the influent COD subtracted with the sludge and decantate, an average difference of 367 kg COD/week was seen. This corresponds with a possible production of 409 NL CH_4/kg COD, which is higher than the theoretical maximum of 350 NL CH_4/kg COD. Since the COD reduction to biogas was based on the CH_4 volume, which in turn was based on the biogas volume, the increasing COD trend (Figure 10) can be due to the increasing biogas production over 2023. This is further discussed in section 4.3.2.

Approximately 3.2 and 4.8 g/pe/day of COD and TS, respectively, was collected as sludge while the decantate contained approximately 27 and 8.1 g/pe/day of COD and TS, respectively. The struvite precipitation affected the COD concentrations to a higher extent than the TS, with approximately 9.2 g/pe/day of COD in the struvite, based on a calculated value. In one analysis of the solid struvite, a total organic carbon (TOC) level of 1.6% of the TS was found. The TOC level corresponds with a COD reduction of 0.083 g/pe/day.

Considering the average measured concentration of suspended solids over the height of the UASB-ST reactor, see Appendix C, and the unlikely possibility of all the solids accumulating at the bottom in a volume representing only 10% of the reactor, the maximum theoretical TS of 6.8% in the sludge would be possible, based on calculations. In comparison, the highest analysed TS sludge levels were 17.4% and 14.5% respectively, much higher than the theoretical maximum of 6.8%.

When looking at the difference between inflow and outflows of the process steps separately, it was seen that the average TS outflows from the struvite precipitation does not add up to the average concentration in the decantate. Approximately 43% of the decantate TS was lost during the struvite precipitation process according to the mass balance.

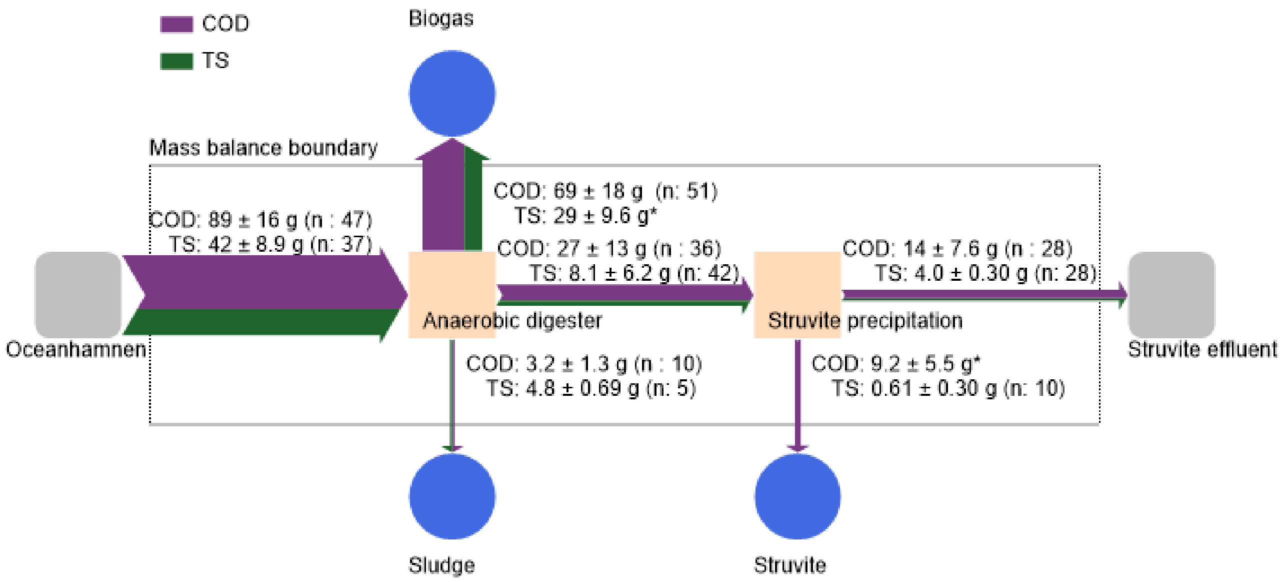

*Figure 13: The TS and COD mass balances per PE and day, including standard deviation and the number of data points used for the calculations. * Indicates that no samples were taken of the stream.*

4.2.3. Phosphorus and nitrogen mass balance

Around 10 times as much TN than TP was found in the influent BW, with 13 g/pe/day of TN and 1.3 g/pe/day of TP, as shown in Figure 14. The average TN value in the decantate was 14 g/pe/day which is slightly larger than the inflow. However, when considering the standard deviaton of 3.2 g/pe/day the influent and decantate can be considered equal.

The amounts of TN and TP found in the sludge were approximately 0.28 and 0.24 g/pe/day, respectively. Thus, TP is bound to to the sludge to a higher degree than TN. The same is true for the struvite precipitation where just 0.048 g/pe/day of TN precipitated whereas 0.11 g/pe/day of TP was found. Concerning the struvite precipitation, the mass balance showed that TP was lost somewhere in the struvite precipitation process, since the solid struvite and the struvite effluent masses do not add up to the decantate mass. Approximately 57% of the TP was lost during the struvite precipitation. There was some loss of TN to aeration, which was approximately 5.9 g/pe/day, calculated based on the difference between the decantate, the struvite and the effluent.

Looking at the hypothetical TN recovery in the ammonia stripper, 7.5 g/pe/day or 82% would be possible to recover. In turn, 1.6 g/pe/day of TN would be found in RecoLab's overall effluent. The effluent TP concentration was assumed not to be affected by the ammonia stripper. The hypothetical calculations were based on the NH_4-N:TN concentration ratio which on average was 1:1.3.

Since the sludge, struvite, and ammonium sulphate are the possible sources for nutrient re-use, these results show that approximately 27% of the incoming TP was available for re-use during 2023. Without the ammonia stripper, only 2.5% of the incoming TN is found in the reusable products. When considering the hypothetical production of ammonium sulphate, 60% of the incoming TN would be possible to reuse.

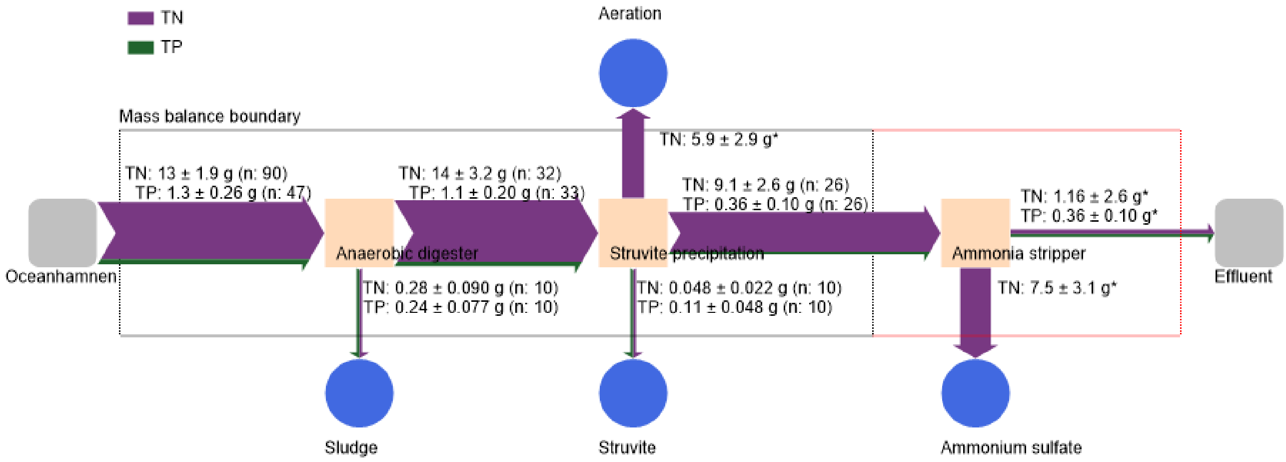

*Figure 14: The TN and TP mass balances per PE and day, including standard deviation and the number of data points used for the calculations. * Indicates that no samples were taken of the stream.*

4.3. Products of blackwater treatment

4.3.1. Sludge

An average of 1023 kg/week of sludge was withdrawn from the AD reactor during 2023, when the non-representative sludge volumes taken from the height of 3m were excluded. The levels of TS in the sludge varied between 2.1 – 3.7 %. Concerning the metal levels in the sludge (Table 10), Zn and Cu had the highest concentrations of 614 respectively 98 mg/kg TS. The concentrations showed no clear trend over time (Appendix C), and the standard deviations were rather low. An average of 58 and 49 g/kg TS of TN and TP respectively were found in the sludge. The sludge had an average of approximately 13 mg Cd/kg P.

Table 10: Mean concentrations and the standard deviations of metals, TN, and TP during 2023 in the sludge, after removal from the AD reactor, and the struvite bags nr 1 - 8.

Element	Sludge mean (mg/kg TS)	Struvite mean (mg/kg TS)
Cd	0.57 ± 0.047	0.14 ± 0.012
Cu	98 ± 6.0	8.1 ± 3.1
Ni	24 ± 3.4	3.55 ± 2.2
Pb	< 2	2.3 ± 0.22
Zn	614 ± 59	61 ± 48
Hg	0.24 ± 0.057	0.060 ± 0.0075
Cr	20 ± 4.2	5.0 ± 3.1
TN	58 000 ± 21 000	72 000 ± 9 000
TP	49 000 ± 14 000	210 000 ± 23 000

4.3.2. Biogas

The average biogas production at RecoLab during 2023 was 194 m^3/week, with an average methane production of 150 m^3/week. As shown in Figure 15, the methane production had an increasing trend over 2023. In a linear regression analysis, a statistically significant correlation between the methane production and the week number was seen. Though, some weeks the production was low and did not reach 100 m^3/week. The trend was not due to increasing influent volumes, as the ratio between the produced biogas and the influent volume increases over time. Overall, RecoLab produced 88 kWh/PE during 2023 based on 10 kWh of electrical energy per m^3 CH_4 (Suhartini et al. 2019).

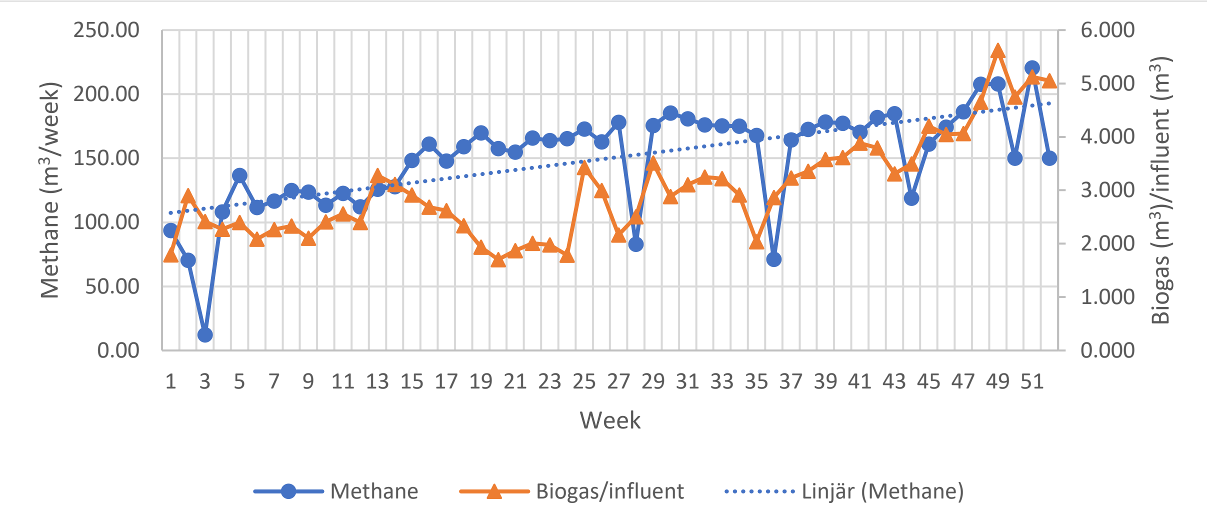

Figure 15: The methane production at RecoLab over the course of 2023 shown by the blue line, together with the ratio between the produced biogas and the influent BW volume. The dotted line shows the linear trend of the methane production.

4.3.3. Struvite

Due to the exchange of struvite bags, only two full bags were weighed during the second half of 2023, which was the period when the struvite precipitation process worked properly. These bags, nr 9 and 10, had a semi-dry weight of 55 and 74 kg respectively as seen in Table 11.

When looking at the difference in the mass of TP in the decantate and in the effluent, an average of 4.59 kg TP/week during the second half of 2023 was separated in the struvite precipitation process. Considering the molecule formula of struvite ($MgNH_4PO_4 \cdot 6H_2O$) and the amount of separated TP, a theoretical struvite production of 515 respectively 324 kg was calculated for bag 9 and 10. Therefore, the measured struvite mass of the bags make up approximately 11 respectively 23% of the theoretical amount, when neglecting the impact of drying of the struvite.

On further investigation of the struvite quality, based on the molecule formula, the TP mass content should theoretically be 12.62% of the entire weight. However, the average TP mass of all struvite bags from RecoLab were 20%.

Table 11: The weight of produced struvite in RecoLab together with the theoretical struvite weight based on the amount of separated TP.

Bag (nr)	Weeks (n)	Semi-dry weight (kg)	Theoretical dry weight (kg)	Produced (%)
9	14	55	515	11
10	9	74	324	23

Shown in Table 10 are the average metal, TP, and TN levels of struvite bags 1 - 8 produced at RecoLab. Whilst Cd, Pb, and Hg concentrations have remained rather constant over the course of struvite production, Cu, Cr, Ni and Zn concentrations have varied in the different bags. This is illustrated in Figure 16, where Cu, Cr, Ni and Zn show a decreasing trend from the earlier bags to the more recent bags which were produced. For information regarding when the bags were produced, see Appendix B. The struvite had an average Cd content of 0.68 mg Cd/kg P.

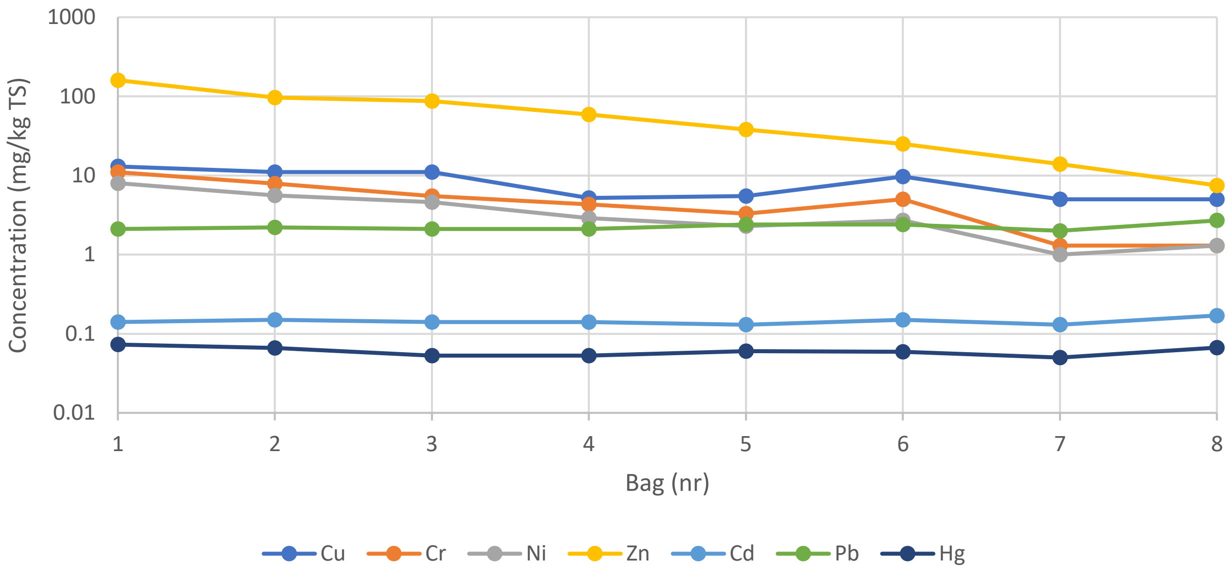

Figure 16: Concentrations of Cu, Cr, Ni, and Zn in the bags of produced struvite.

5. Discussion

5.1. Quantity and quality of the blackwater

One positive aspect of source-separating wastewater systems is the separation of streams based on their different quantity (Otterpohl et al., 2004). While the total incoming flow to RecoLab shows some weekly variations, the variations are not as large as with Öresundsverket's wastewater flow (Figure 9). The rather consistent incoming BW volumes make the mass balance calculations more reliable since the water flow impacts the calculations in all steps. Furthermore, the quality of the black water was rather consistent over the year of 2023, with standard deviations of the investigated parameters <25%.

The average BW concentrations of TP and TN were approximately 120 mg/L and 1.3 g/L, respectively. In comparison with another source-separating systems (Table 12), the TP levels were higher than shown by Sievers et al. (2016) though lower than showed by Wendland et al. (2007) . The TN levels in RecoLab were slightly lower than both mentioned studies. The N:P ratio is similar to what was found by Wendland et al. (2007). Sievers et al. (2016) discussed the seasonal effect on the TP concentrations, with higher levels in the summer season possibly due to increased intake of nutrient-rich vegetables. Such a trend is not seen for RecoLab where the levels are consistent over the year, which could be due to differences in demographics and people's diets between the investigated populations. Concerning the levels of TS and COD in the influent, the average TS of 4.1 g/L is slightly lower than 6.5 g/L seen in Wendland et al. (2007) while the average COD levels are the same in this study as well as in Wendland et al.

Table 12: Influent concentrations of TS, TP, TN, and COD to RecoLab together with results from two other studies of BW. Unit g/L.

Parameter	RecoLab influent	Sievers et al. (2016)	Wendland et al. (2007)
TS	4.1	-	6.5
TP	0.12	0.073	0.18
TN	1.3	1.4	1.5
COD	8.8	-	8.7

When combining low C:N ratios, for example ratios <10:1, with high total ammonia concentrations, the AD process could be disturbed (Yang et al., 2023). In this case, based on a relationship between COD and TOC (Dubber & Gray, 2010), the influent TOC:N ratio was calculated to 2.2:1. This indicates that ammonia inhibition could become a problem in the AD reactor, causing less biogas to be produced that what is possible. The conversion from COD to TOC is based on a relationship from conventional WWTPs, which could mean that the TOC:N ratio is wrongly estimated due to the differences between conventional wastewater and BW. However, other studies show that a COD:N ratio of 40:1 is optimal for AD processes (Zappi et al., 2019) and therefore it seems that the ratio between carbon and nitrogen in the RecoLab influent are lower than the optimal for AD.

While conventional WWTPs base the estimation of PE on the levels of BOD found in the wastewater, source-separating wastewater systems bring a challenge in which the BW does not have the same levels of BOD as the GW does (Jönsson et al., 2005). For this reason, to be able to view and compare the results of the study per PE, the default values of several parameters (ibid) and the resulting estimated PE based on RecoLab was used (Table 8). Both TS and the flow resulted in amounts of PE which deviated from the other parameters. A possible explanation for the differences is extra flushes of the vacuum toilets in Oceanhamnen being done without any faeces, urine or toilet paper being flushed down. This would result in a larger volume of flow and in turn making the flow PE assumptions too big. Due to the flow and TS PE values deviating from the other parameters, these were excluded and an average amount of 885 PE was estimated to be representative for RecoLab during 2023 (Table 8).

5.1.1. Comparison with Öresundsverket

In comparison with average metal concentrations in Öresundsverket's influent (Table 2) (NSVA, n.d.), RecoLab has higher concentrations of all metals (Table 7). When considering that all metal levels in the influent are higher than Öresundsverket's influent, it is important to remember the role of dilution. While the RecoLab BW had a weekly accumulated average of 64 m^3/week or approximately 72 L/PE/week, during 2023, Öresundsverket had an accumulated average of 400 000 m^3/week or 2.7 m^3/PE/week. Thus, a difference between Öresundsverket's conventional wastewater treatment system and RecoLab is more dilution occurring in the conventional WWTP. Therefore, to investigate whether the BW or the conventional wastewater contains less metals, the metal levels should be considered per PE. To accurately compare the wastewaters, influent GW concentrations to RecoLab should also be considered. This is deemed to be outside the scope of the study.

5.2. Mass balances

5.2.1. Wet mass

As a result of the assumption that the decantate and effluent volume is the same as the influent volume subtracted with the sludge volume, the wet matter balance becomes a closed mass balance. The struvite mass is so small in comparison that it does not affect the result of the wet mass balance, with the outflows being 100% of the inflow. In the first half of 2023, sludge was withdrawn in larger quantities and from a height of 3 m in the AD, with the aim of decreasing the height of the sludge bed in the reactor. However, since the analysis of concentrations in the sludge were conducted as previously on samples from the bottom of the reactor, sludge taken from a different height should not affect the measured concentrations. For that reason, the initial sludge volumes were excluded and then replaced with an average sludge withdrawal volume from the second half of the year.

5.2.2. Total Solids

A shortcoming in the mass balance calculations is the assumption that total solids and total suspended solids are equal measurements of the solids content. The wet samples, the influent, decantate, and the effluent, were analysed for total suspended solids whereas the sludge and struvite were analysed for total solids. This means that the dissolved solids of the liquid fractions were neglected from the mass balances.

Some of the excluded TS values were high enough for the sludge to be classed as a solid sludge according to Makádi et al. (2012), whereas practical experience of the system showed that the sludge was always liquid. If all solids in the AD reactor accumulate in the bottom, which does not happen in reality, the maximum TS of 6.8% is lower than some of the analysed values. Considering these facts, the exclusion of certain TS sludge values was reasonable, since some kind of analytical error must have occurred.

The mass balances show that TS nearly reached a closed mass balance, though with less mass in the outflows than in the inflow, with no clear trend over the year (Figure 10). It was also shown that the TS mass balance over the struvite precipitation process does not add up (Figure 13). Approximately half of the TS does not leave the struvite precipitation. A possible explanation is that the drum filter, which was assumed to have a negligible effect on all the mass balances, in reality does have some outflow of TS. However, it seems unlikely that a 43% loss of TS in the struvite precipitation process was entirely caused by the drum filter. Furthermore, the precipitation of struvite means that previously aqueous ions form crystalline structures, in turn causing the TS concentrations to increase. This increase of TS was not seen in the mass balance which indicates that some outflow was underestimated.

In contrast with the struvite precipitation, the mass balance over the AD reactor is closed due to the assumption that all TS which does not go to the decantate or the sludge, instead goes to the biogas (Figure 13). The assumption resulted in a large fraction of TS going to the biogas. When considering that RecoLab had a high SRT of 75 days combined with the fact that the longer the SRT the more organic matter is digested (de Graaff et al., 2010), it seems plausible that a lot of TS is reduced and leaves with the biogas. However, when comparing the production of 831 L CH_4/kg TS with data of BW from a LCA of the source-separation system done by Kjerstadius et al. (2015, 2016), only 520 L CH_4/kg TS was produced. That leads to question whether the assumption of a closed TS mass balance over the AD reactor led to an overestimated of the amount of TS which was reduced to biogas.

5.2.3. COD

The mass balance of COD was nearly closed, with the average being slightly more COD in the outflows than inflow (Table 9). An increasing trend of more COD in the outflows than the inflow was seen over the course of 2023 (Figure 10). Since the COD mass balance calculations are based on the biogas production and the methane content of the biogas, the increasing trend is likely connected to the increasing biogas production over the year, see section 5.3.2. for further details.

The methane levels in the biogas were based on an average from earlier than 2023, due to faulty equipment. However, this is a reasonable replacement since the methane levels in the biogas production in the dataset used had a standard deviation of 0.8%. Important to consider is that the COD reduction to biogas was based on the theoretical maximum of 350 L CH_4/kg COD. A theoretical COD reduction in the AD reactor of on average 78% (Figure 13) is in the same range as in studies conducted by Kujawa-Roeleveld & Zeeman (2006) and de Graaff et al. (2010).

However, when looking at the difference between the influent and the sludge and decantate, it was seen that 409 NL CH_4/kg COD could have been produced. Since this is a larger value than the theoretical maximum, some kind of error must have occurred in the measurement. Possibly, the COD measurements could have been overestimated by the laboratory. It could have also been due to a wrongful measurement of the biogas flow rate, which affects the CH_4 levels. The biogas was measured with a drum type flow meter, which after discussion with NSVA has been noticed to contain too much water. This decreases the volume of gas in the gauge and causes an over estimation of measured flow rate. Thus, the main hypothesis is that the gauge was the reason for the biogas flow rate to be overestimated in the used dataset, causing errors in the estimation of produced methane.

The calculation of the COD reduction to struvite was based on a study by Diamadopoulos et al. (2007), which showed a reduction between 30 – 35%. That study was based on struvite precipitation of an UASB effluent, though with the influent being conventional municipal wastewater. Another possible way to estimate the COD reduction to struvite would be to look at the amount of organic matter (TOC) found in the struvite and convert that into a value of the COD. The results from such a calculation showed that the COD reduction would be 100 times smaller than what was calculated using data from Diamadopoulos et al. Though, only one TOC analysis of the struvite was available for RecoLab, which leads to an uncertainty of how true that TOC analysis is for all the struvite bags. Interestingly, the mass balance showed that more COD goes into the struvite than the sludge (Figure 13) which is surprising considering that the sludge has a high level of organic matter. This fact together with the low amount of organic matter found in the struvite points to an overestimation of the COD reduction to struvite. Overall, the reduction of COD during the struvite precipitation should be further investigated at RecoLab.

5.2.4. Phosphorus

The mass balance calculations showed that TP is bound to sludge to a higher extent than TN (Figure 14), which could be due to nitrogen being ionic and thus staying in the aqueous phase as compared to phosphorus which precipitates more (Jönsson, 2019).

Considering the mass balance as a whole, TP had the worst results with the outflows only reaching an average 53% of the inflows (Table 9, Figure 11). 27% of the influent TP ended up in the re-usable products of sludge and struvite (Figure 14), which means a low possible re-use of TP compared to conventional WWTPs where up to 95% of the influent TP can be found in the sludge (Kjerstadius et al., 2017). Furthermore, the mass balances showed that it was over the struvite precipitation process in which a lot of TP, approximately 57%, were lost (Figure

14). Thus, while studies, e.g. Kjerstadius et al., show the potential nutrient recovery in a source-separating system, the RecoLab facility has yet to achieve a good collection of produced struvite. With further improvements, it is possibly that the struvite production could give higher TP recovery than at a conventional WWTP.

A possible reason for the low struvite collection is the fact that there are two filters, one mechanical and one bag filter, after the struvite precipitation and before the ammonia stripper to collect leftover suspended material. Practical experience of the system has shown struvite clogging these filters, struvite which the hydrocyclone has not been able to separate into the struvite bags. Furthermore, coatings in the equalizer tank and the heat exchanger have also been noticed. The clogging and coating could be the explanation for the low TP mass balance. Interestingly, the TS levels also did not add up over the struvite precipitation process, which is possibly connected to the precipitation of solids such as struvite occurring in the system. Overall, the clogging and coating points to the hydrocyclone not being well enough optimized to separate the struvite.

While the struvite crystal sizes in RecoLab has not been analysed, a study by Park et al. (2020) showed that crystals similar to struvite which were larger than 150 µm had a 77% separation efficiency in a hydrocyclone. Possibly, the size of RecoLab's struvite crystals were smaller than 150 µm, resulting in a low separation grade. Park et al. also discussed that struvite crystals of a larger size resulted in a higher purity grade, which can be of importance for further use of the fertilizer.

As described by Wu & Vaneeckhaute (2022), increasing the chemical dosage of magnesium could increase the struvite precipitation and increase the purity of the struvite. This means that the precipitation of other minerals such as calcium phosphates, which is the most common impurity, is decreased. In RecoLab during 2023, $MgCl_2$ was added so that the PO_4-P:Mg ratio was 1.72:1. This is a lower magnesium dosage than what for example De Graaff et al. (2011), a ratio of 1:1.5, showed was most effective. The rather low level of struvite precipitation which occurred at RecoLab could in part be due to the low magnesium rate, however it is important that not too much $MgCl_2$ is added since it can affect the struvite quality negatively (Siciliano et al., 2020).

5.2.5. Nitrogen

The mass balance showed that while some TN was found in the struvite, it was less than the TP found in the struvite (Figure 14). This is as expected since the struvite has the N:P ratio of 1:1 (Zeeman et al., 2008) but nitrogen has a lower molar mass which results in a lower nitrogen mass.

When adding the struvite precipitation process to the mass balance calculations, the nitrogen balance got closer to being a closed balance (Figure 11). The change in the mass balances when adding the struvite precipitation process is likely due to the loss of TN to aeration, with an aeration loss of 42%. The loss by aeration was calculated based on the difference between the inflow and the outflows of the struvite precipitation process, which causes the mass balance to be closed over the struvite precipitation (Figure 14). The loss to aeration is due to the wastewater stream being aerated to reduce the CO_2 and increase the pH. This causes TN removal to occur due to ammonium ions (NH_4^+) being transformed into ammonia (NH_3) molecules and transported by the air (Karri et al., 2018). The smell of ammonia has also been noticed in proximity to the struvite bags, which points to loss of TN in several places along the treatment line. However, calculating the loss to aeration based on the differences between the inflow and outflows mean that the loss to aeration in the mass balances was large. In reality, some of the TN which was assumed to be lost to aeration could instead be precipitated to struvite but not caught by the hydrocyclone, as is the case with TP and TS.

The hypothetical nitrogen recovery calculations showed that 7.5 g/PE/day could be possible to recover in the ammonia stripper (Figure 14). This means that the TN recovery would increase from approximately 2.5% of the incoming TN found in sludge and struvite to approximately 60% being recovered when including ammonium sulphate, with further recovery possible if more struvite is captured. The possible recovery of 60% is slightly lower than the recovery stated in a LCA study conducted by Kjerstadius et al. (2016), where 69% of the nitrogen ended up as ammonium sulphate. Considering the hypothetical calculation, when the nitrogen recovery process works it will bring a large fraction of the nutrient recovery of the RecoLab treatment plant. It should give higher TN recovery rates than by the re-use of sludge as conducted in conventional WWTPs (Kjerstadius et al., 2017; Malovanyy et al., 2022).

The hypothetical calculations were based on a recovery rate during ammonia stripping of reject water, which is the water stemming from the dewatering process of sludge. It has similar characteristics to BW, including the TN and TP concentrations being in a similar range (Solon, 2015). For that reason, the recovery rate from Sagberg & Grundnes Berg (2000) seemed plausible to apply for this study. Furthermore, as stated by Wu & Vaneeckhaute (2022), over 90% of ammonium could be recovered if concentrations were larger than 100 mg NH_4^+/L. In RecoLab's case, the average ammonium concentration incoming to the ammonia stripper was 783 mg/L. Therefore, it seems reasonable that assumed ammonium recovery rate was 78.4% (Sagberg & Grundnes Berg, 2000).

5.3. Products of blackwater treatment

5.3.1. Sludge

The levels of TN and TP in the BW sludge were 17 and 71% higher, respectively, than the average sewage sludge from conventional wastewater treatment in Sweden (Table 13) (Naturvårdsverket, n.d.). When comparing these concentrations, it should be considered that the levels from conventional WWTPs are based on dewatered sludge. In RecoLab, no such process is conducted and thus elements which are contained in the water phase are included. Since BW is the source of RecoLab's sludge, the concentrated nutrient rich wastewater stream could be the reason for the higher nutrient levels in the sludge. According to NSVA (n.d.), Öresundsverket had higher TN concentrations in the sludge whereas RecoLab had higher TP concentrations (Table 13).

Table 13: The mean concentrations of metals and nutrients in RecoLab's sludge, RecoLab's struvite, sewage sludge spread on agricultural land in Sweden (2022) (Naturvårdsverket, n.d.), in the limit values according to the SPCR178 certification (RISE, n.d.-b), and in Öresundsverket's (ÖV) digested sludge (NSVA, n.d.). Unit mg/kg TS.

Element	RecoLab sludge	RecoLab struvite	Sewage sludge	SPCR178 limit values	ÖV sludge
Cd	0.57	0.14	0.606	1	0.6
Cu	98	8.1	334	600	319
Ni	24	3.55	16.8	50	19.6
Pb	< 2	2.3	13.72	100	16.2
Zn	614	61	470	800	528
Hg	0.24	0.060	0.341	1	0.43
Cr	20	5.0	19.78	100	26.5
TN	58 000	72 000	49 510		61 535
TP	49 000	210 000	28 672		29 430

Overall, the concentrations of metals in the sludge were beneath the limit values set by the SPCR178 certification (RISE, n.d.-b). Thus, considering metal levels the sludge is in good condition to use for agricultural purposes. Furthermore, 13 mg Cd/kg P is well below both the limit for the Revaq and SPCR178 certifications (RISE, n.d.-a, n.d.-b). The sludge had a higher Cd/P ratio than the influent BW.

Table 14: The Cd/P ratio in RecoLab's blackwater influent, sludge, and struvite together with certification values from RISE and Revaq (n.d.-b, n.d.-a).

Object	mg Cd/kg P
RecoLab influent	8.1
RecoLab sludge	13
RecoLab struvite	0.68
SPCR178 certification	17
Revaq certification	22

In comparison with average metal levels in sludge produced at conventional WWTPs (Naturvårdsverket, n.d.) as well as Öresundsverket specifically (NSVA, n.d.), higher concentrations of Ni and Zn were found in the RecoLab sludge (Table 13). The sludge from conventional WWTPs arouse from processes such as sedimentation, where especially Ni has been seen to be equally divided between sedimentation and the effluent (Yoshida et al., 2015). For this reason, a smaller fraction of the incoming Ni might have ended up in the sludge than in the case of RecoLab where AD is the first treatment step. Zn concentrations on the other hand is more consistently bound to the particulate phase according to Yoshida et al., and therefore the differences between how the sludges are produced should not have impacted the Zn content. Instead, a possible reason could be that Zn leached from the relatively new metal equipment in Oceanhamnen and RecoLab, which is further discussed in section 5.3.3. However, no clear trend of decreasing Zn concentrations in the sludge was seen over time (Appendix C). Besides Ni and Zn, metal levels were lower in RecoLab's sludge than the sludge from conventional WWTPs which could be due to the BW not containing industrial effluent and surface runoffs which in conventional plants can give increased metal concentrations (Tervahauta et al., 2014). Since industries and stormwater does not affect the BW quality, the relatively high concentrations of for example Hg and Cr in the sludge could be due to leaching equipment, similarly to the Zn concentrations, or due to people's diets.

5.3.2. Biogas

The biogas production increased over the course of 2023, reaching a production of nearly 300 m^3/week of biogas, which corresponds to approximately 200 Nm^3 CH_4/week (Figure 15) and 32 L CH_4/PE/day. Since an error in the biogas measuring gauge has been noticed, the biogas production during late 2023 might have been overestimated. Nonetheless, an increasing trend over the entire year was seen.

The average biogas production in RecoLab equals an electrical energy of 88 kWh/PE/year, which was more than what three out of four investigated conventional WWTPs produced (

Table 1). The production in RecoLab was lower than the electrical production in other source-separating systems, where the investigated systems had an average of 103 kWh/PE/year. This could be due to the fact that the other systems included FW in the AD, which leads to a larger biogas production (Yang et al., 2023).

5.3.3. Struvite

Only 11 and 23% of the theoretical maximum struvite yield was produced in two of the struvite bags produced during 2023 (Table 11). The theoretical separation is based on the amount of TP which was separated in the struvite precipitation process, which varied from approximately 2.5 to 8 kg TP/week. Combining the low percentage of produced struvite with the loss of TS, TP, and possibly TN in the mass balances over the struvite precipitation process, the explanation that struvite forms in the treatment process but is not collected in the hydrocyclone seems further plausible.

Interestingly, more TP was found in the produced struvite than what should be theoretically possible based on the molecular formula of struvite. This trend was seen in all struvite bags with no big change over time, which points to other precipitates than struvite being separated into the bags. A reason for this could be the rather low Mg:P ratio which could decrease the purity of the struvite (Wu & Vaneeckhaute, 2022). Alternatively, when the struvite samples are heated to 105°C during analysis, the molecular structure could have been impacted causing e.g. evaporation of the water molecules which are normally bound to the struvite or breaking of the molecule. This could have impacted the amount of TP found in the struvite during analysis.

Overall, the metal levels and the Cd:P ratio in the struvite are within limits for the SPCR178 certification (Table 13 & Table 14) (RISE, n.d.-b). Some metals had a decreasing trend over time, especially noticeable by the Zn concentrations, which had the highest concentrations (Figure 16). This could correlate with the metal levels in the sludge, where Zn also was the most present metal though no decrease over time was seen in the sludge. Possibly, coatings containing Zn and other metals could have been used in construction of the Oceanhamnen residential area as well as in the RecoLab facility. The leaching of these metals could then have decreased with increased use of the facilities.

6. Conclusions

Source-separating wastewater systems and the treatment of blackwater (BW) provide opportunities for nutrient recovery as well as energy production through anaerobic digestion (AD). To evaluate the RecoLab treatment plant, in terms of BW treatment and nutrient recovery, mass balances of the wet mass, total solids (TS), chemical oxygen demand (COD), total phosphorus (TP), and total nitrogen (TN) were calculated. The quantity of incoming BW to RecoLab was constant over the investigated time, providing a reliable basis for the balance calculations. The concentrations of TS, COD, TP, and TN were generally consistent in the analysed dataset and in line with previous studies conducted on BW.

The study found that the largest impact on the TS and COD mass balances were the biogas production, with 69% and 78% of COD and TS converted to biogas respectively, which was similar to previous studies. However, a faulty gauge connected to the anaerobic digester introduced some uncertainty in the biogas-to-COD calculations, possibly overestimating the biogas production. Furthermore, TS reductions to biogas might have been overestimated due to assumptions made for the mass balance calculations. Nonetheless, RecoLab showed an increasing trend in biogas production over time, with an average methane production of 150 m^3/week and an electrical energy output of 88 kWh/PE/year, 27.5% higher than the conventional wastewater treatment plants (WWTPs) considered in this study. Source-separating wastewater plants co-digesting BW and food waste (FW), had a higher biogas output than RecoLab, suggesting that co-digestion could be a possible avenue for RecoLab to increase their biogas production.

The sludge from the AD reactor had metal concentrations which were within the limit values for SPCR178 certification, including 13 mg Cd/kg P which qualifies the sludge for certification. In comparison with sludge from conventional WWTPs, Zn and Ni concentrations were elevated. These metal concentrations could be connected to the metal levels in the struvite, where Zn was the most prominent. The struvite metal concentrations had a decreasing trend over time, pointing to the impact of leaching from new equipment. Yet, the struvite metal concentrations in the AD sludge were below limit values for SPCR178 certification and it had 0.68 mg Cd/kg P.

The sludge and struvite precipitation contributed to a total recovery share of only 27% of the TP and 2.5% of the TN. The TP and TS mass balances were not closed, with significant loss (57% of TP and 43% of TS) during the struvite precipitation process. The TN mass balance over the struvite precipitation was assumed to be largely affected by aeration, causing a 42% loss of TN. Out of the TP mass separated from the BW during the struvite precipitation, only up to 23% was recovered in the struvite bags and the rest continued in the process which caused clogging of filters downstream. Furthermore, based on elevated TP concentrations in the struvite, other elements than struvite could have precipitated into the bags. Thus, the struvite precipitation process was shown to be less efficient than anticipated, and the process should be improved to increase the nutrient recovery to a satisfactory level.

Overall, this study shows that the anaerobic digestion of BW at RecoLab works as it should, producing a sludge with mostly low metal concentrations and a biogas production higher than at conventional WWTPs. The mass balances indicate that the formation of struvite is efficient, though the separation of the crystals from the BW was inefficient and needs further optimization. Considering the theoretical addition of the ammonia stripper, total recovery rates of TN could be increased to 60%. Furthermore, if the ammonia stripper is used together with a better optimized struvite precipitation process, an even higher TN and TP recovery would be possible.

7. Recommendations of further research

This study identified several uncertainties and potential challenges in the treatment of blackwater (BW) and nutrient recovery. While the results of the mass balances were shown in the unit of g/PE/day, the calculation of population equivalents (PE) is an approximation due to the lack of a definition of PE in BW. For that reason, BW flows in vacuum sewer systems should be analysed to improve the accuracy of PE calculations and to develop a clearer understanding of BW quantity and quality.

Aeration prior to struvite precipitation was assumed to impact TN, but it was a calculated value. Further studies should directly measure the aeration process to determine how much TN is lost during aeration, at other stages in the treatment process, or bound to uncollected struvite. This would clarify the pathways of nitrogen loss during nutrient recovery.

The limited amount of collected struvite, combined with the discrepancies in the mass balances of TS and TP points to the hydrocyclone being inefficient. Investigating and optimizing the hydrocyclone could improve nutrient recovery.

Additionally, the unexpectedly high TP levels in struvite could be due to changes in the crystalline structure during sample heating, leading to inaccurate mass measurements. Further research should explore how struvite crystals react to heat and other conditions. The leaching of Zn from equipment should also be investigated since elevated concentrations were found in the sludge and struvite.

Addressing these issues will help improve the efficiency of BW treatment and nutrient recovery at RecoLab and similar facilities. Further comparisons of source-separating wastewater systems like RecoLab to conventional wastewater treatment plants (WWTPs) could highlight the potential advantages of BW treatment and nutrient recovery, leading to increased knowledge and legitimacy of source-separation.

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Appendix A

Table A 1: Mean concentrations, standard deviation (SD), number of samples and number of excluded samples of TP, TN, and COD analysed by the NSVA laboratory (blue text) and the SGS laboratory (black text).

Parameter	Mean	SD	Unit	n	Nr of excluded
TP	117, 123	26, 19	mg/L	35, 47	2, 1
TN	1286, 1311	163, 112	mg/L	32, 90	3, 3
COD	8399, 8757	2273, 1526	mg/L	37, 47	1, 1

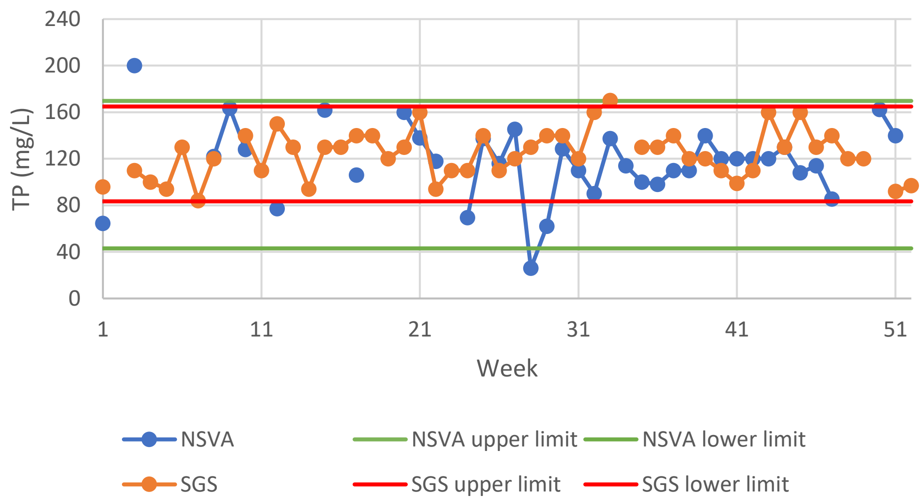

*Figure A 1: Concentrations of TP in the influent BW, analysed by the NSVA laboratory (blue) and the SGS laboratory (orange). The horizontal lines show 2*standard deviation for the average weekly concentration of the NSVA analyses (green lines) and the SGS analyses (red lines). Concentrations outside their respective horizontal lines were excluded.*

Appendix B

Table B 1: Dates of when struvite bags were exchanged.

Bag (nr)	From	To
1	2021	?
2	2021	?
3	2022	?
4	2022	?
5	?	2022-08-02
6	2022-08-02	2022-12-14
7	2022-12-15	2023-02-20
8	2023-02-24	2023-07-14
9	2023-07-14	2023-10-23
10	2023-10-23	2023-12-20
11	2023-12-20	2024-02-12

Figure B 1: Wet struvite to the left and semi-dry struvite which has air-dried to the right.

Appendix C

Table C 1: Mean concentrations of suspended solids (SS) in the AD reactor during 2023.

Week	SS [mg/L]
1	2273
2	2273
3	2273
4	2273
5	2273
6	2273
7	2273
8	2273
9	2273
10	2273
11	7366
12	7276
13	6900
14	7827
15	8089
16	10779
17	7455
18	8508
19	8004
20	9011
21	10742
22	10312
23	10274
24	10370
25	11215
26	12398

Week	SS [mg/L]
27	10553
28	10475
29	10435
30	12142
31	11236
32	10167
33	13796
34	13231
35	13567
36	9531
37	14443
38	13626
39	19211
40	17437
41	14107
42	15605
43	17307
44	13196
45	12347
46	14670
47	13695
48	
49	11475
50	16628
51	11085
52	

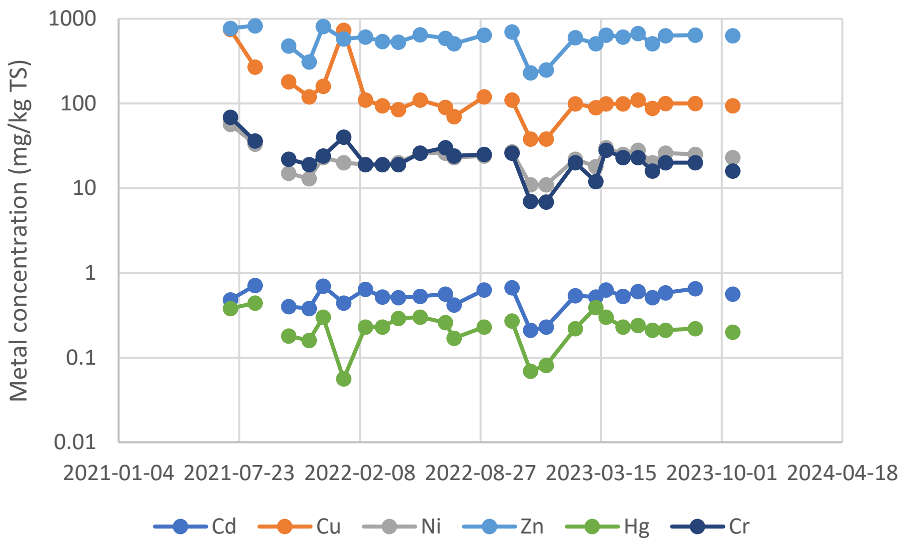

Figure C 1: Concentrations of Cd, Cu, Ni, Zn, Hg, and Cr in the sludge from the AD reactor over time.