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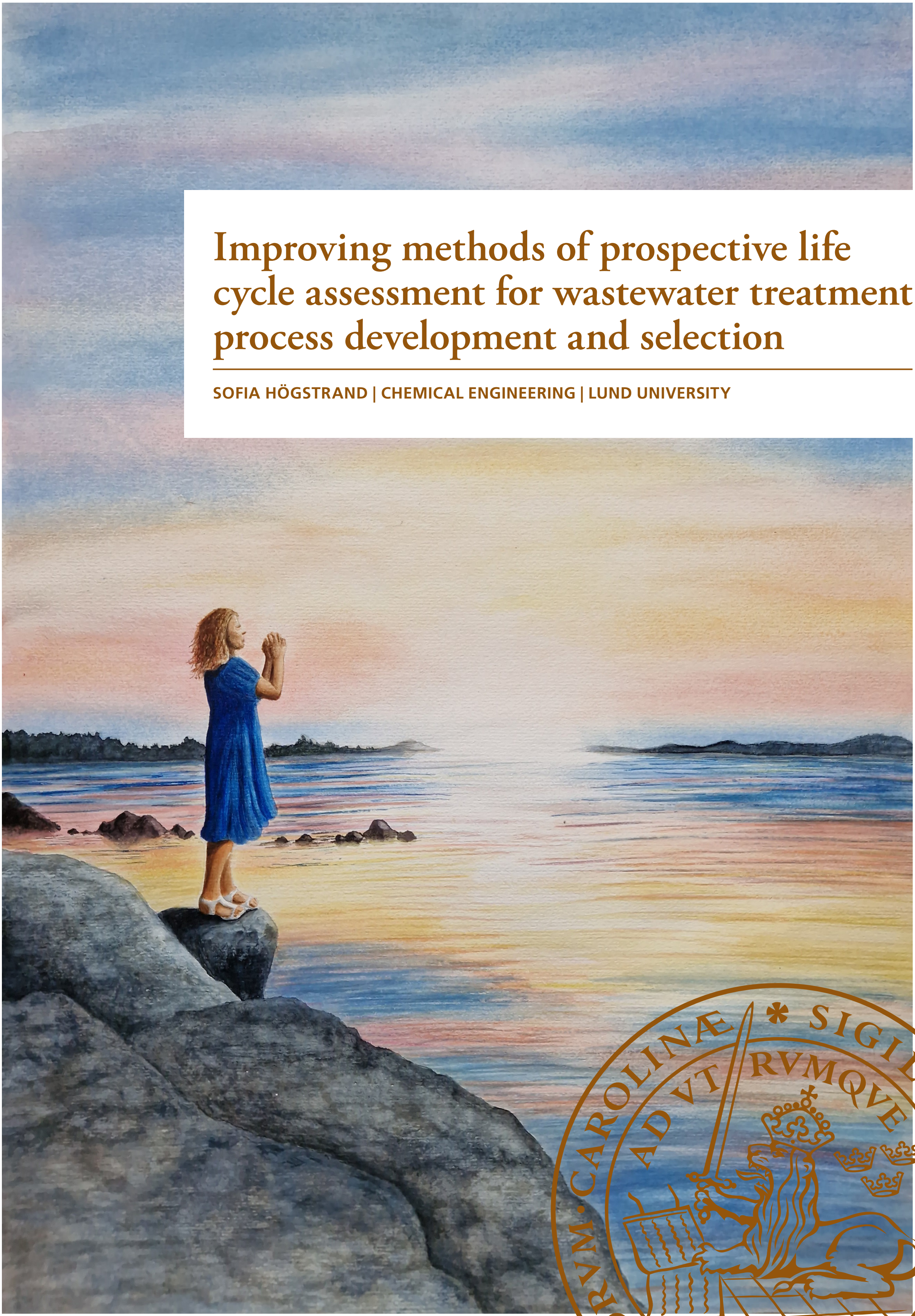

Improving methods of prospective life cycle assessment for wastewater treatment process development and selection

SOFIA HÖGSTRAND | CHEMICAL ENGINEERING | LUND UNIVERSITY

Improving methods of prospective life cycle assessment for wastewater treatment process development and selection

Sofia Högstrand

DOCTORAL DISSERTATION

Doctoral dissertation for the degree of Doctor of Philosophy (PhD) at the Faculty of Engineering at Lund University to be publicly defended on the 5th of December at 09.00 in lecture hall KC:A, Kemicentrum, Department of Process and Life Science Engineering, Naturvetarvägen 14, Lund

Faculty opponent

Associate Professor Anders Damgaard, Department of Environmental and Resource Engineering, Technical University of Denmark (DTU), Denmark

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Title and subtitle: Improving methods of prospective life cycle assessment for wastewater treatment process development and selection

Abstract:

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Regarding LCA methodology, this thesis shows that it is possible to simplify the system by narrowing the system function to reduce the need for data while still maintaining relevance—e.g., by delimiting the technological system while using a simplified model to account for impacts on the larger system. It also shows the extent of which pilot plants and dynamic models are useful for creating data inventories when accounting for differences in scale and model uncertainties, respectively. Furthermore, the thesis demonstrates that alternative impact assessment approaches, such as the application of bioassays rather than conventional chemical analysis, can capture missing aspects of toxicity. However, further research on important parameters—such as choice of reference substances, types of bioassays and biological endpoints—is necessary to aid interpretation and increase the meaningfulness.

As for WWT-specific results, the thesis describes the hotspots and the desirable paths forward for processes for nutrient removal (biological and chemical phosphorus removal) and recovery (NPHarvest, struvite precipitation, biochar production, and sludge stabilisation), as well as micropollutant removal (biological and mechanical post-treatment after ozonation). Moreover, the thesis demonstrates how prospective LCA is an important tool for WWT process development and selection to portray environmental aspects and facilitate improved decision-making.

Keywords: Environmental assessment, enhanced biological phosphorus removal (EBPR), nutrient recovery, quaternary treatment, sludge management, bioassay

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Improving methods of prospective life cycle assessment for wastewater treatment process development and selection

Sofia Högstrand

LUND
UNIVERSITY

Front cover illustration: “The art of playing the trumpet without a trumpet” by Madeleine Flyman
Back cover photo by Hilde Skar Olsen

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Playing the trumpet without a trumpet
is the hardest thing there is.
-Pippi Longstocking

Table of Contents

Preface: Where is the trumpet?

The story of my research begins with a wastewater treatment plant. Back in the old days of 2018, a municipality in the south of Sweden was planning to build a new wastewater treatment plant filled to the brim with all the latest technology and the capability to recover both energy and nutrients. However, Swedish bureaucracy came in the way, and the wastewater treatment plant is still waiting to be built. I was a new PhD candidate at that time, diving head-first into that project and thinking this was the opportunity of a lifetime—for what PhD student really gets the possibility to do research on a full-scale, brand-new facility with the latest technology? Well, not me, as it turned out. My plans had to change drastically. Several times over, in fact, as attempts to do more lab-oriented research also failed due to various reasons.

But there were blessings to be found in the struggles. I discovered the intriguingly interesting yet mind-bogglingly frustrating field that is life cycle assessment—or LCA for short—and made life hard for my supervisor by changing the scope of my research to entail systems thinking. However, the unbuilt treatment plant was still an issue. The words by Pippi Longstocking stood out to me: "*Playing the trumpet without a trumpet is the hardest thing there is*". I felt these words quite fittingly described my dilemmas. How could I measure and analyse wastewater treatment without a treatment plant? How could I perform LCA without data? I had to seek other means of finding, collecting, and using data to assess the impacts of the treatment plant.

As I soon found out, this is actually an ever-growing research field: future-oriented, *prospective* LCA. It offers means to evaluate the impacts of something not yet fully realised, such as a new material or a novel technology, or—as in my case—an unbuilt wastewater treatment plant. With the help of great colleagues and co-writers, I contributed to this field by further developing three methodological approaches on different levels of LCA: simplifying the technological system, creating data through pilot plants and models, and using alternative impact assessment approaches to capture missing aspects of toxicity. I then used these methods to assess the possible impacts of process development and selection of different parts of the wastewater treatment plant. Although not quite making the sounds of a real trumpet played by a proper philharmonic, these methods may still provide sounds reminiscing of music. In this digital age, I believe that the use of models and digital twins—especially with the help of AI, although that is outside the scope of my current work—will continue to evolve so that these digital "sounds" will be more and more useful to guide the development and selection of both current and future trumpets, I mean, treatment plants. Nevertheless, there is, for sure, an art to it.

Acknowledgements

Although mine is the sole name on this thesis, many are those who have contributed and without whose support this work would not be possible. I would therefore like to acknowledge …

… the economic funding and provision of the European Life programme through the LIWE LIFE project (LIFE17 ENV/SE/000384), the VA-teknik Södra research cluster (18-130, 21-116, 25-103), the Swedish Water Association (SVU 20-120); the Richert Foundation (2020-00635) and the Swedish Environmental Agency (NV-06632-23).

… my main supervisor, Karin Jönsson. Thank you for taking me on for all these years and supporting me, even though the research changed from the original plans to outside your comfort zone. I have learned so much working alongside you these past years.

… my co-supervisor, Magdalena Svanström. Thank you for being an inspiration, generously sharing your expertise of all things LCA-related and welcoming me to your department at Chalmers. Without you, I would be completely lost in the world that is LCA.

… my LIWE LIFE project partners at the municipality of Lidköping and Kompetenzzentrum Wasser Berlin. Thank you, Pernilla Bratt, for bringing enthusiasm and inspiration; Gudrun Magnusson, for sharing your life-long experience and answering all my questions; and Amanda Andersson, for your helpful collection of data. Thank you, Ulf Miehe, Michael Stapf, and Fabian Kraus, for interesting discussions and the warm welcome during my visit in Berlin.

… my co-authors Raed A. Al-Juboori (Aalto University), David Gustavsson (Sweden Water Research, VA SYD), Jes la Cour Jansen (Professor Emeritus, Lund University), Juho Uzkurt Kaljunen (Aalto University, NPHarvest), Hamse Kjerstadius (NSVA), Anna Mikola (Aalto University), Greg Peters (Chalmers University of Technology), Maja Rydgård (University of Copenhagen), Christoffer Wärff (RISE, Lund University), and Linda Önnby (IVL). Thank you for sharing your knowledge and so undeniably helping me on my way. It has been an honour working with you!

… my colleagues at the division of Chemical Engineering at Lund University for contributing to a creative environment and all the laughs in the fika room. A special thank you to Michael Cimbritz and Per Falås for guiding initial lab work, reading manuscripts of papers and the thesis frame, generously giving me feedback, and checking in on me at regular intervals—in practice, being some sort of bonus supervisors. Thank you, Gertrud Persson, for guiding me in the lab, listening to the joys and progresses, as well as offering your shoulder to cry on. And, of course, a big shout-out to all my former and current fellow PhD student colleagues for bringing joy and colour to this time and sharing all the struggles of striving for the PhD title. A special thank you to Ellen Edefell for bringing the Pippi quote to my attention during one of our lunches out.

… my colleagues at the Division of Environmental Systems Analysis at Chalmers for creating space for me and making me feel so welcome during my short and irregular visits.

… my family and friends. Thank you for your unwavering support, your encouraging cheers from the sideline, and for carrying me in your prayers. A special thank you to my husband Kristoffer and our children Elias and Livia for putting up with me during this time, allowing me to work and finish in time for “mum’s big party”. You are an inspiration, and I love you so much.

Lastly, I also want to thank God, the source of all wisdom, creativity, and life, for leading me these years. You led me into it, and You got me through it. I am curious to see where You will take me next. And a special thank you to the guardian angel that must have held my computer and protected the data and documents when, three months before the public defence, the battery exploded mid-sentence.

Abstract

Future-oriented environmental assessments—in particular, prospective life cycle assessments (LCAs)—are increasingly applied within the wastewater treatment (WWT) sector. This thesis contributes to this field of research by addressing *how* to estimate the environmental impact (improving LCA methodology) and *what* the impact amounts to (WWT-specific results) for a wastewater treatment plant not yet constructed. To this end, four case-studies were performed, resulting in the four accompanying papers.

Regarding LCA methodology, this thesis shows that it is possible to simplify the system by narrowing the system function to reduce the need for data while still maintaining relevance—e.g., by delimiting the technological system while using a simplified model to account for impacts on the larger system. It also shows the extent of which pilot plants and dynamic models are useful for creating data inventories when accounting for differences in scale and model uncertainties, respectively. Furthermore, the thesis demonstrates that alternative impact assessment approaches, such as the application of bioassays rather than conventional chemical analysis, can capture missing aspects of toxicity. However, further research on important parameters—such as choice of reference substances, types of bioassays and biological endpoints—is necessary to aid interpretation and increase the meaningfulness.

As for WWT-specific results, the thesis describes the hotspots and the desirable paths forward for processes for nutrient removal (biological and chemical phosphorus removal) and recovery (NPHarvest, struvite precipitation, biochar production, and sludge stabilisation), as well as micropollutant removal (biological and mechanical post-treatment after ozonation). Moreover, the thesis demonstrates how prospective LCA is an important tool for WWT process development and selection to portray environmental aspects and facilitate improved decision-making.

Populärvetenskaplig sammanfattning:
Konsten att spela trumpet utan trumpet

Hur gör man för att uppskatta miljöpåverkan av ett reningsverk som inte finns? Och hur ska man veta vilken sorts rening som ger bäst resultat för minst miljömässiga kostnad medan man fortfarande har en chans att påverka designen? Det är minst sagt en konst som jag har ägnat de senaste sju åren åt att försöka bemästra.

Miljöbedömning, närmare bestämt livscykelanalys (LCA) är ett verktyg för att uppskatta miljöpåverkan från en produkt eller process—i mitt fall ett reningsverk. Genom att samla in data över allt som går in (som kemikalier, energi, material) och ut (som utsläpp till luft, vatten och mark) och därefter bearbeta och analysera detta, kan man kartlägga exempelvis hur stor den potentiella påverkan blir på klimatet, miljön eller den mänskliga hälsan. För att kunna göra en ordentlig bedömning behövs således en väldig mängd data. Men hur gör man om data inte finns? Om reningsverket inte är byggt än?

Framåtblickande studier, så kallade *prospektiva* LCAer, är ett växande fält inom forskningen. Denna metodik används exempelvis vid utveckling av nya material eller produkter för att försöka göra sig en bild av vilka åtgärder som är viktigast för att den fortsatta utvecklingen ska bli så hållbar som möjligt. Till detta fält kan också nu mitt arbete sälla sig.

Mitt arbete har haft ett tvådelat forskningsfokus: **Hur** kan man göra en miljöbedömning över, och **vad** är miljöpåverkan av, ett reningsverk som inte är byggt än? Den första delen har inneburit utveckling av LCA-metodik medan jag för den andra delen använt och testat dessa metoder på avloppsrening för processval och processutveckling. För att svara på de två övergripande frågorna har jag genomfört fyra fallstudier (mina fyra artiklar) i vilka jag både utvecklat metodik samt använt den för att undersöka konkreta reningsprocesser.

I korthet har jag vidareutvecklat tre metoder för att hantera bristen på explicit mätdata i LCA. För det första har jag utvecklat en modell för att bedöma påverkan på hela reningsverket när man bara har mätdata för en del av den. Detta ger fördelen att man kan förenkla det undersökta systemet men ändå fånga viktiga faktorer för hur det större systemet berörs. För det andra har jag skapat representativa data genom att använda pilotanläggningar och simuleringsmodeller. Oväntade resultat i form av höga ammoniakutsläpp upptäcktes när pilotdata kombinerades med beräkning av massflöden. Likaledes oväntat indikerade en simuleringsmodell på skillnader i metanutsläpp mellan processkonfigurationer som visar på vikten av fortsatta mätningar i fullskala för att undersöka detta närmare. För det tredje har jag utvecklat själva miljöpåverkansbedömningen genom att använda en metod som förvisso använts inom andra områden men först på senare tid funnit sin väg in i

LCA-världen. För att bedöma den toxiska påverkan av avloppsvattnet på omgivande natur används generellt kemiska analyser för enskilda ämnen. Genom att istället använda så kallade effektbaserade mätmetoder kan vattenprovets hela sammansättning få visa på hur stor påverkan mottagande vatten kan tänkas få. Mitt arbete har visat på styrkor och utmaningar med denna nya metod.

Gällande de mer praktiska resultaten av mina fyra fallstudier kan jag kortfattat sammanfatta dem med följande fyra punkter:

1) Det finns för- och nackdelar med både biologisk och kemisk rening av fosfor, där den förstnämnda har en större driftsäkerhet vid händelse av kemikaliebrist medan den sistnämnda potentiellt har en lägre klimatpåverkan från direkta luftutsläpp.

2) Införande av kvartär rening för avskiljning av mikroföroreningar kan uppvisa betydligt större nytta då toxiciteten beräknats med effektbaserad analys snarare än med kemisk analys av ett fåtal ämnen och antyder därmed att man faktiskt tjänar miljömässigt på införandet av tekniken.

3) NPHarvest, tekniken för återvinnande av näringsämnen ur koncentrerat avloppsvatten, uppvisade liknande miljöpåverkan som teknologin den jämfördes med (dvs. bästa tillgängliga teknik). Med fördel bör dess fortsatta utveckling inkludera sätt att minska ammoniakavgång, utvärdera andra kemikalier samt öka energieffektiviseringen.

4) När det kommer till slamhantering med näringsåtervinning kan det konstateras att det även för tekniker som rötning, struvitfällning och pyrolys finns både för- och nackdelar. Generellt visade dock konfigurationer med struvitfällning på en lägre miljöpåverkan än de utan. Rötning med slamspridning vore kanske en tekniskt enklare lösning, men skulle även bidra till ökade utsläpp. Pyrolys, å sin sida, skulle kunna minska utsläpp bland annat genom ökad kolinlagring, men den resulterande biokolen har potentiellt sämre gödselegenskaper vilket skulle kunna innebära minskade skördar.

Avslutningsvis påminns jag av ett citat av Pippi Långstrump: ”*Att spela trumpet utan trumpet är det svåraste som finns*”. Och så har det stundtals känts i min forskning. Hur kan man säga något om ett reningsverk som inte finns? Hur kan man göra en miljöbedömning utan data? Men faktum är att mitt arbete, och det växande fält som är prospektiv LCA, har visat att det inte bara är möjligt utan också rent utav nödvändigt för att kunna leda utvecklingen framåt i en hållbar riktning. Med det sagt är det inte lätt. Det är en konst.

List of papers

Paper I

Högstrand, S., Uzkurt Kaljunen, J., Al-Juboori, R.A., Jönsson, K., Kjerstadius, H., Mikola, A., Peters, G., Svanström, M., 2023. Incorporation of main line impact into life cycle assessment of nutrient recovery from reject water using novel membrane contactor technology. *J. Clean. Prod.* 408, 137227. https://doi.org/10.1016/j.jclepro.2023.137227

Paper II

Högstrand, S., Wärff, C., Svanström, M., Jönsson, K., 2024. Dynamic process simulation for life cycle inventory data acquisition - Environmental assessment of biological and chemical phosphorus removal. *J. Clean. Prod.* 479, 144047. https://doi.org/10.1016/j.jclepro.2024.144047

Paper III

Högstrand, S., Peters, G., Svanström, M., Önnby, L. Testing the incorporation of bioassays in life cycle assessment: A case study on advanced wastewater treatment. *Accepted for publication in Int. J. Life Cycle Assess.*

Paper IV

Högstrand, S., Gustavsson, D., Jönsson, K., Rydgård, M., Svanström, M. Struvite or biochar? Environmental assessment of sewage sludge management for improved nutrient recycling. *Manuscript*

Author's contribution to the papers

Paper I

While the primary data for the NPHarvest pilot plant runs were obtained by co-author Juho Uzkurt Kaljunen, I collected additional data for the background system, as well as for the alternative technology. I also created the LCA model, ran the software, and performed the LCA. I reviewed literature to find and develop, with inputs from my co-authors, the mathematical model for main line impact (MLI). I analysed the results and created graphs and figures. I also wrote the original draft with input from the other authors.

Paper II

I developed research questions and the project scope together with my co-authors. I set up and led the project. While co-author Christoffer Wärff created the dynamic simulation model, I performed all simulations, as well as created and ran the LCA model. I gathered additional data, created figures and graphs, wrote the original draft, as well as led the work on review and editing with input from the other authors.

Paper III

I developed research questions and the project scope together with my co-authors. I set up and led the project. I gathered data, and I created and ran the LCA model. I created figures and graphs, wrote the original draft, as well as led the work on review and editing with input from the other authors.

Paper IV

I contacted and invited co-authors. I developed research questions and the project scope together with my co-authors. I set up and led the project. I performed all process model simulations, as well as created and ran the LCA model. I gathered additional data, created figures and graphs, and wrote the original draft with input from the other authors.

Related publications

Högstrand, S., Uzkurt Kaljunen, J., Kjerstadius, H., Al-Juboori, R., Jönsson, K., Mikola, A., Peters, G., Svanström, M., 2022. *Rejektvattenrening med näringsåtervinning - Teknisk och miljömässig bedömning av NPHarvest-teknologin genom piloförsök och livscykelanalys*, SVU-rapport 2022-12.

Jönsson, K., Jansen, J. la C., **Högstrand, S.**, Svanström, M., Wärff, C., 2025. *Möjlighet till ökad tillämpning av biologisk fosforavskiljning (bio-P) på svenska avloppsreningsverk - En kunskapssammanställning med omvärldsbevakning och livscykelanalys*, VA-teknik Södra Rapport nr. 2025-15. Lund, Sweden.

Abbreviations

Wastewater treatment-related terms

AS	Activated sludge
AWT	Advanced wastewater treatment
CHP	Combined heat and power process
CP	Chemical precipitation of phosphorus
EBPR	Enhanced biological phosphorus removal
GAC	Granular activated carbon
MBBR	Moving bed biofilm reactor
O_3-GAC	Ozonation with GAC post-treatment
O_3-MBBR	Ozonation with MBBR post-treatment
PAO	Polyphosphate-accumulating organism
PE	Population equivalent
VFA	Volatile fatty acid
WRRF	Water resource recovery facility
WWT	Wastewater treatment
WWTP	Wastewater treatment plant
UWWTD	Urban Wastewater Treatment Directive

Life cycle assessment-related terms

EF	Environmental footprint
FU	Functional unit
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
MLI	Main line impact

1 Introduction

How do we estimate the impacts and benefits of wastewater treatment before it physically exists? How can we guide decision-making towards sustainable options before implementation of new or developing technologies?

Wastewater treatment (WWT) nowadays is indeed different than just a hundred years ago, when the activated sludge process was invented (Ardern and Lockett, 1914). Nevertheless, the main purpose, or **function**, is the same: treatment of wastewater for man's health and well-being—that is, the **function of preventing disease**. But over the years, new problems have appeared, prompting new solutions and changes in management. In the middle of the 1900s, eutrophication—i.e., over-fertilisation causing overgrowth and algal blooms—became apparent and connected to the high organic and nutrient content of wastewaters. Thus, a new function emerged: the **function of mitigating eutrophication**.

More recently, other pollutants have made it to the headlines: pharmaceuticals, pesticides, personal care products, antibiotics, PFAS, etc. Last year, the European Urban Wastewater Treatment Directive (UWWTD) (EU Directive 2024/3019) was updated to include requirements for removal of these substances, thus prompting the **function of reducing toxicity**.

In parallel to that, the focus has shifted to also include aspects of circularity and sustainability. Efforts are made to recover valuables from wastewater, such as energy, fertilisers, and, more recently, even water for reuse. A paradigm shift is underway. Instead of viewing wastewater as something to be handled and removed in a wastewater treatment plant (WWTP), it is increasingly seen as a stream with potential for recovery of valuables in a water resource recovery facility (WRRF) (Guest et al., 2009). This is the **function of resource recovery**. But circularity through resource recovery is only one aspect of sustainability and perhaps not always the most sustainable solution. During the last decades, the concern and already visible impacts of climate change have been high on the public agenda. The wastewater sector has therefore set the aim of reducing the carbon footprint and use of (in particular, fossil) energy. This is the **function of minimising climate change**.

In the UWWTD, all these functions are enforced and encouraged. There are demands on extensive removal of nutrients and organic micropollutants—of recovering resources while simultaneously striving for climate neutrality without violating the overall function of disease prevention. But there are inherent trade-offs between these functions, as more stringent effluent requirements need more chemicals and energy, whereas to reach climate neutrality, energy and chemicals must be used more sparingly and efficiently.

One way of exploring these trade-offs is through life cycle assessment (LCA). LCA is a tool for investigating the potential environmental impacts of a product or process in terms of both resource use and emissions for a wide range of categories, such as climate change, eutrophication, and toxicity, to name a few. The earliest LCAs were developed in the 1970s in the packaging industry (Bjørn et al., 2018), but the use of the tool has spread to many different fields since—to the field of wastewater management already in the 1990s (Tillman et al., 1998). The use of LCA within wastewater treatment is thus not new. But in recent years, the focus has shifted. From looking at the current state, there has been an increase in studies aiming at the future, targeting questions like "What is the impact of this new technology, not yet fully developed?" or "What will the impact be if we build a facility like this or that?"

This growing field of future-oriented, **prospective** LCAs is where my research is located, specifically targeting the wastewater sector. My work circles around a soon-to-be-built WWTP in Lidköping, in the south of Sweden, with the purpose of evaluating the environmental implications of process design choices before construction begins shortly. I have had a dual research focus with two overarching questions: **how** can we assess the possible environmental impacts, and **what** are the possible environmental impacts of wastewater treatment prior to implementation? More specifically, the following research questions (RQs) define the work as presented in this thesis in the context of planning a Swedish WWTP capable of fulfilling current and foreseeable environmental legislation (i.e., the UWWTD) with current (2025) technological understanding of wastewater treatment processes:

RQ 1) Is it possible to maintain the relevance while simplifying the system to reduce the need for data?

RQ 2) To what extent are pilot plants and models useful for creating data inventories?

RQ 3) Can alternative impact assessment approaches meaningfully capture missing aspects related to toxicity?

RQ 4) What are the hotspots and desirable paths forward according to beyond state-of-the-art LCA for WWT process development and selection?

The first three questions relate to the **how**—i.e., the methodology used for data acquisition, utilisation, and modelling, with more general conclusions for the LCA community. The fourth question relates to the **what**—i.e., the interpretation of the results, with more specific conclusions targeting the wastewater sector. This duality in research objective is also evident in my papers. Each paper covers both research areas and, in so doing, contributes to knowledge on LCA methodology, as well as wastewater treatment.

To answer these questions, I used a case study methodology applying a life cycle perspective—i.e., a holistic view of the WWTP, quantifying impacts from raw material acquisition to disposal of residues, assessing them through different, relevant environmental aspects. Specifically, four LCA case studies (the papers) were conducted, targeting different parts of the WWTP—as visualised in Figure 1. In each study, different LCA methodological challenges were addressed to which improvements were suggested and applied in the specific context of the study. To that end, I have studied and utilised different types of data acquisition methods and I have engaged with experts within various fields to ensure a deep understanding of technologies, theories and models. In all case studies, the WWT problem owners—i.e., the municipality of Lidköping—have constituted a reference group that helped me obtain a correct understanding of current practices as well as the needs and possibilities of the setting where the future plant is to be constructed.

The first case study is detailed in **Paper I**, in which system simplification (RQ 1) is applied on a novel technology for nutrient recovery using primary data from pilot plants (RQ 2) to aid early-stage process development (RQ 4). The second case study is described in **Paper II**, where the use of a dynamic simulation model (RQ 2) for data acquisition is applied on secondary and tertiary treatment in order to compare processes of phosphorus removal (RQ 4) in the event of potential chemical shortage. In **Paper III**, the third study is presented, where a complementary approach of toxicity assessment (RQ 3) is used on pilot plant data (RQ 2) to evaluate ozonation post-treatment alternatives for quaternary treatment (RQ 4). Finally, the fourth case study is detailed in **Paper IV** and combines different mathematical and computational models (RQ 2) for evaluation of sludge management and nutrient recovery practices (RQ 4) to identify in which form—sludge, struvite or biochar—phosphorus is most desirable from an environmental perspective.

The outline of my thesis is deliberately slightly unorthodox with the two main themes—LCA and wastewater treatment—highlighted through two background chapters and two chapters with results and discussion, intertwined with methodological comments along the way. In **Chapter 2**, the history and general context of wastewater treatment are described, accompanied by two more specific sub-sections on wastewater treatment processes used in my studies. In **Chapter 3**, LCA is introduced and described using the four phases of LCA procedure, along with comments and examples of my own methodological choices. In **Chapter 4**, the first three research questions are discussed and answered, with the input of LCA methodology contributions from all four papers. In **Chapter 5**, the fourth question is answered, with the wastewater-treatment-related lessons learned from the four case studies. After that follows a reflection on my research in **Chapter 6**, looking back on my contributions, as well as looking forward to what the future might hold. Finally, with the conclusions in **Chapter 7**, my thesis is completed.

With this thesis, my main objective is to help guide the decision-making towards sustainable options prior to their implementation. I do this by finding the methodological challenges, improving these methods of prospective LCA, and applying them to generate insights for the wastewater treatment context. In other words, I hope to demonstrate the art of playing the trumpet without a trumpet and what lessons we can draw from it.

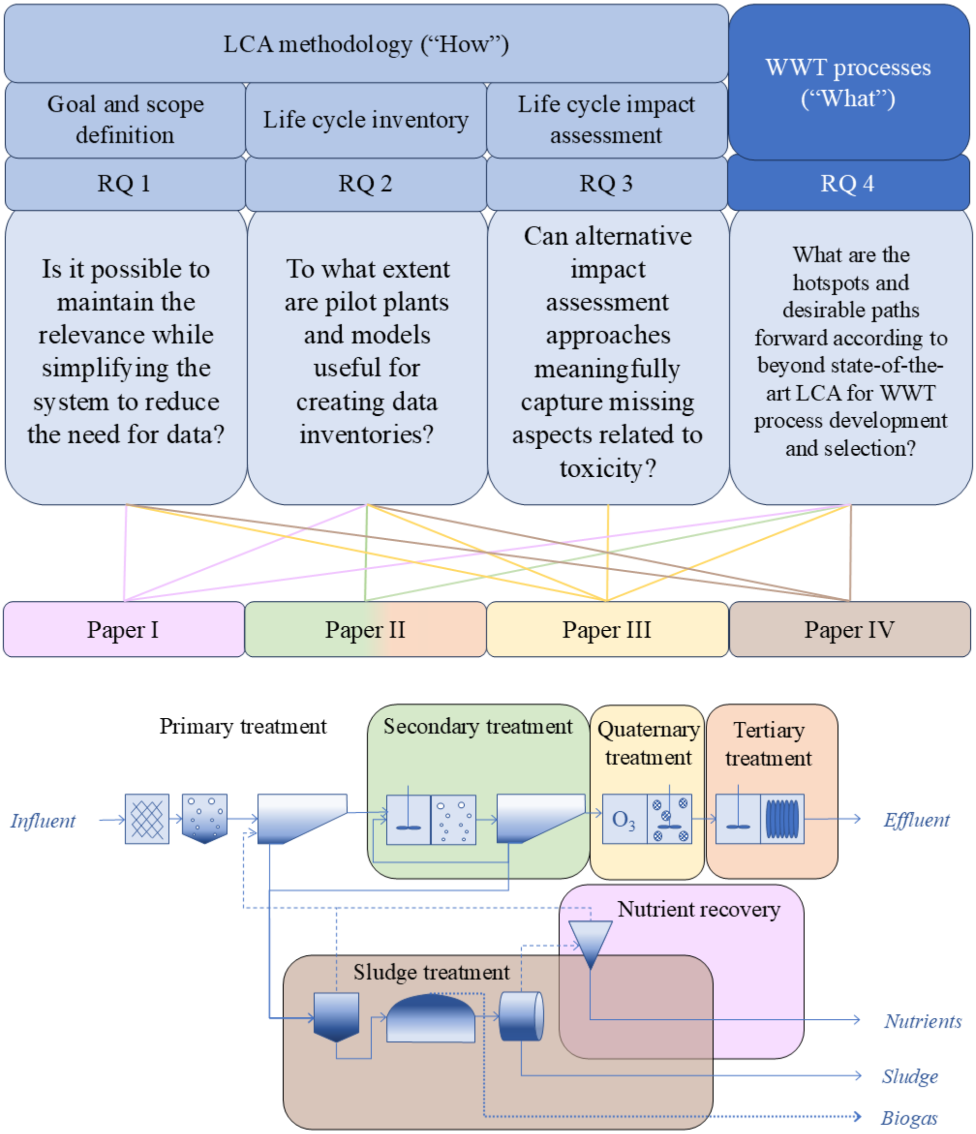

Figure 1
Connections between the research areas, research questions, papers, and wastewater treatment stages.

2 On wastewater treatment

As evidenced in Figure 1, there are numerous terms used in wastewater treatment (such as ‘secondary’, ‘tertiary’, ‘quaternary’, ‘sludge management’, and ‘nutrient recovery’) that need to be properly defined when discussing wastewater treatment—and they are perhaps not as unambiguous as one might think. I will therefore use this chapter to briefly describe the history and general context of wastewater treatment and the origin of these terms (Section 2.1); how they are or, perhaps rather, might be used today; and how I have decided to use them in my research (Section 2.2). Then, I will describe the specific context of my research: Ängen WWTP, the planned WWTP in Lidköping (Section 2.3), and NPHarvest, the nutrient recovery pilot plant used in **Paper I** (Section 2.4).

2.1 Wastewater treatment throughout history

All throughout human history, wastewater has been considered dirty and unclean, but the ways of its management or treatment have varied over time and space. A common way of disposing human waste before greater civilisations were created was through holes in the ground, as commanded already in the Mosaic Laws from 1400 BC:

> “Designate a place outside the camp where you can go to relieve yourself. As part of your equipment have something to dig with, and when you relieve yourself, dig a hole and cover up your excrement.” (Deuteronomy 23:12-13, NIV)

The first signs of wastewater management on a community scale were in the Mesopotamian Empire (3500-2500 BC), where drainage systems, latrines, and cesspits have been uncovered from the ruins of Ur and Babylonia (Lofrano and Brown, 2010). The earliest signs of wastewater treatment come from the Indus civilisation (2500 BC), where wastewater was drained from houses through pipes into a sump, where solids were allowed to settle, while the liquids overflowed to further drainage (Lofrano and Brown, 2010). This type of treatment with solids removal through gravity sedimentation is often referred to as **primary treatment**. It removes the solids without altering their composition.

As time went on, the Egyptians built toilet stools, the Greeks built drainage systems for storm- and wastewater, and the Romans managed the whole water cycle—from collecting clean water through their aqueducts to disposing of the storm- and wastewater, and even recycling water from the spas to flush their latrines (Lofrano and Brown, 2010). After the fall of the Roman empire, the management of wastewater deteriorated, and it was not until the industrial age where wastewater management once again was prioritised. In the 1800s, sewage systems were built in the larger cities in Europe, leading untreated water to rivers, such as the Seine and Thames, using water as a transport medium for pollutants, out of the cities, resulting in severe pollution of these water courses (Hahn, 2014; Harremoës, 1999). The Thames was even referred to as a "monster soup" by the Victorians (Lofrano and Brown, 2010).

It was not until the early 1900s when treatment facilities first were constructed which not only removed but also treated the wastes. The use of micro-organisms for degrading the organic material in the wastewater is referred to as **secondary treatment** (Metcalf & Eddy, 2014). The first trickling filter (micro-organisms attached to filter media) was installed in 1893 in the UK (Lofrano and Brown, 2010), and activated sludge (AS)—i.e., retention of micro-organisms that are suspended in the wastewater through aeration or stirring—was patented in 1913 (Ardern and Lockett, 1914; Daigger, 2014; Lofrano and Brown, 2010). Rapid implementation of the AS processes was seen, so that within twelve years, around 20 full-scale plants were constructed in the UK and USA (Stensel and Makinia, 2014). After removal of organics, the treatment advanced to mitigate eutrophication by targeting nutrients, such as nitrogen and phosphorus. Biological nitrogen removal processes, such as nitrification and denitrification, were developed in the 1960s and 1970s by several different research groups, suggesting different process configurations of aerobic and anoxic reactors (Khunjar et al., 2014). Biological phosphorus removal was first observed in the 1960s, when alternating between aerobic and anaerobic conditions, and the first full-scale plant with enhanced biological phosphorus removal (EBPR) was constructed in South Africa in the 1970s (Barnard and Comeau, 2014). Addition of chemicals, such as iron, aluminium, and calcium salts, was performed already in the 1930s to remove organic matter, but in the 1960s, the idea that these salts also precipitated phosphorus took notion (Jenkins et al., 1971). These nutrient-removing technologies were originally referred to as **advanced treatment** (Lofrano and Brown, 2010) but are today more commonly referred to as **tertiary treatment** (Metcalf & Eddy, 2014). Tertiary treatment may also include disinfection, as well as more advanced filtration steps, such as ultrafiltration, which was developed already in the 1970s (Crawford et al., 2014).

In recent decades, the issue of micropollutants has prompted the development of even more advanced technologies, such as ozonation, activated carbon, and membrane technologies (e.g., reverse osmosis) to further mitigate environmental impacts and allow for water reuse. Today, these treatments are often referred to as

advanced treatment (Metcalf & Eddy, 2014), at least when the purpose is water reuse (Davis, 2010). But, they may also fall under the rather recently defined **quaternary treatment** (EU Directive 2024/3019, 2024). The updated UWWTD from 2024 defines the purpose of quaternary treatment to be:

> "… to ensure that a large spectrum of the remaining micropollutants are removed from urban wastewater." (EU Directive 2024/3019, 2024)

As for the residuals of wastewater treatment—i.e., the solids, or the sludge—they have throughout the civilised ages been used as fertilisers on farmland (Lofrano and Brown, 2010). This way of managing the residuals is still commonplace today in countries such as France and Sweden—although there has been an ongoing debate for several decades in Sweden whether sludge application should be allowed (SOU, 2020:3)—and increased concerns related to its pollutant content have banned this practice in countries such as the Netherlands, Slovakia, and Malta (Hudcová et al., 2019). One way of mitigating some of the risks is through sludge digestion—both aerobic and anaerobic—which reduces the pathogens as well as volatile solids. In Sweden, a voluntary but common certification system for wastewater sludge, called Revaq, exists, which among other things requires outdoor storage of the sludge for at least six months prior to spreading to further minimise risks (Revaq, 2022).

With the discovery of the Haber-Bosch process in the early 1900s and the following large-scale production of mineral fertilisers, the need for nutrient recycling through sludge application diminished (Zilio et al., 2022). However, with the rising concern of climate change and the awareness of how highly energy-demanding and fossil-dependent the Haber-Bosch process is, recovery of nitrogen from nitrogen-containing waste streams, such as wastewater, has once again become interesting (Zilio et al., 2022). Furthermore, as phosphorus for mineral fertilisers is mined from finite phosphate rock, concerns regarding the limited supply (Cordell et al., 2009) have put it on the EU Critical Raw Materials list (European Commission, 2014), which further prompts the importance of phosphorus recycling. As both nitrogen and phosphorus are macronutrients common in wastewater, recovery from these streams has become increasingly interesting in recent years (Rout et al., 2021; Sengupta et al., 2015).

But there are more resources than nutrients to recover from wastewater. Biogas production from anaerobic digestion was practiced already in the 10th century BC in Assyria, although research on the topic intensified in the 1800s, with the first wastewater sludge digester being built in the UK in 1895 and more widespread implementation of biogas production facilities after World War II (Kasinath et al., 2021). Recovery of carbon and organic matter may increase the soil organic carbon content and increase the soil's water-holding capacity and thus increase crop yields (Börjesson et al., 2012; Hedlund, 2012).

2.2 Wastewater treatment in general

Returning to the terminology used for wastewater treatment, it is noted that definitions may vary, depending on temporal and geographical factors. The definitions by Metcalf and Eddy (2014) may possibly be the most utilised within the wastewater sector today, at least in English-speaking countries. However, the definitions as described in the UWWTD (EU Directive 2024/3019, 2024) differ to the ones suggested by Metcalf and Eddy. Moreover, the common convention may also differ between countries. In Sweden, a common way to categorise the different parts of the WWTP is by using the terms **mechanical/physical**, **biological**, and **chemical**. Mechanical and physical treatment often coincides with primary treatment, biological with secondary, and chemical with tertiary. However, chemicals used for phosphorus removal may be added already in the primary treatment, and biological processes may be used in the tertiary treatment, such as in the biofilms on sand filters.

When looking into these three definitions, I noticed differences and tried to make a distinction between them. I would consider the Swedish convention of terms to be more process-specific, the Metcalf and Eddy definition to be more function-specific, and the UWWTD to be more result-specific (see the three textboxes), and they may therefore be more relevant in different settings and serve different purposes. What I mean is, for example, the Swedish physical treatment may refer to processes as diverse as grit removal, sedimentation basins, and flotation, whereas utilising the Metcalf and Eddy definitions would categorise grit removal within preliminary treatment, sedimentation within primary or secondary, and flotation as secondary. The UWWTD definition does not look at the processes per se—rather, what result they have on the wastewater quality.

Process-specific terms

– Swedish convention (Terminologicentrum TNC, 1977)

Mechanical: “Treatment consisting of preliminary treatment and primary sedimentation”

Physical: “Treatment utilising the pollutants’ physical characteristics —e.g., sedimentation, flotation, centrifugation, filtration, sifting”

Biological: “Treatment with micro-organisms to convert organic compounds into more stable forms. Main processes include activated sludge (AS), trickling filters, and ponds”

Chemical: “Treatment with added chemicals—e.g., iron salts, aluminium salts, and lime. Chemical treatment can occur through oxidation, precipitation, coagulation, flocculation, etc.”

Function-specific terms

– Metcalf and Eddy (2014)

Preliminary: “Removal of wastewater constituents such as rags, sticks, floatables, grit, and grease, that may cause maintenance or operational problems with the treatment operations, processes, and ancillary systems.”

Primary: “Removal of a portion of the suspended solids and organic matter from the wastewater.”

Advanced primary “Enhanced removal of suspended solids and organic matter from the wastewater. Typically accomplished by chemical addition or filtration.”

Secondary: “Removal of biodegradable organic matter (in solution or suspension) and suspended solids. Disinfection is also typically included in the definition of conventional secondary treatment.”

Secondary with nutrient removal: “Removal of biodegradable organics, suspended solids, and nutrients (nitrogen, phosphorus, or both nitrogen and phosphorus)”

Tertiary: “Removal of residual suspended solids (after secondary treatment), usually by granular medium filtration or microscreens. Disinfection is also typically a part of tertiary treatment. Nutrient removal is often included in this definition.”

Advanced: “Removal of dissolved and suspended materials remaining after normal biological treatment when required for various water reuse applications”

Perhaps the definitions according to Metcalf and Eddy are the most universally accepted within the wastewater sector globally. However, as time moves on, the use of the term “advanced” may prove to change—as it already has before—and might therefore not be sufficiently unambiguous. I believe that the definitions as put forward in the UWWTD may be more clear-cut and, as such, may prove a better way forward. However, there is always a lag phase of confusion when going from one paradigm to another. Perhaps the biggest differences in definitions between Metcalf and Eddy and UWWTD concerns secondary and tertiary treatment—more specifically, the definition as to where nutrient removal occurs. According to UWWTD, secondary treatment is only removal of organic matter, where nutrient removal is strictly categorised as tertiary treatment. Conversely, in Metcalf and Eddy, nutrient removal may be part of the secondary treatment, although it is also an important part in tertiary treatment.

Result-specific terms

– UWWTD (EU Directive 2024/3019)

Primary: “Treatment of urban wastewater by a physical or chemical process, or both, involving settlement of suspended solids, or other processes in which the BOD5 of the incoming wastewater is reduced by at least 20 % before discharge and the total suspended solids of the incoming wastewater are reduced by at least 50 %”

Secondary: “Treatment of urban wastewater by a process generally involving biological treatment with a secondary settlement or another process which reduces biodegradable organic matter from urban wastewater”

Tertiary: “Treatment of urban wastewater by a process which reduces nitrogen or phosphorus, or both, in urban wastewater”

Quaternary: “Treatment of urban wastewater by a process which reduces a broad spectrum of micropollutants in urban wastewater”

Within **Paper II**, I do not specifically classify the technologies (biological and chemical phosphorus removal) as either secondary or tertiary treatment. Within the thesis frame, however, as seen in Figure 1, I have chosen to utilise the more Metcalf-and-Eddy-esque definition, labelling biological phosphorus removal as a secondary treatment (with nutrient removal) and chemical precipitation as tertiary treatment. This is motivated, since the baseline biological scenario also utilises post-precipitation as a polishing step to reach the low effluent requirements, meaning that this scenario performs nutrient removal both in secondary and tertiary treatment. Perhaps this distinction is more understandable from the Swedish perspective of differentiating between biological and chemical processes.

Regarding the usage of quaternary versus advanced wastewater treatment, I believe that the term “advanced treatment” may be more ambiguous and change meaning over time. I have therefore decided to use the term “quaternary treatment” within the thesis frame when discussing the technologies evaluated in **Paper III**. Within said paper, the UWWTD definition is used when stating the functional unit (quaternary treatment of 1 m^3 secondary effluent). However, I used the current common literary jargon of referring to the evaluated technologies as advanced wastewater treatment (AWT) technologies (see e.g., Gallego-Schmid et al., 2019; Li et al., 2019; Risch et al., 2022). In doing so, I may involuntarily have contributed to continued ambiguity, but I use this platform to call for further update and standardisation of definitions. The latest edition of Metcalf and Eddy was written a decade ago—not to mention the Swedish terminology list that is half a century old; a lot has happened in that time, and both technologies and languages have developed since.

Although, I guess that when it comes down to it, perhaps it is not really that important whether EBPR should be classified as secondary or tertiary treatment. Or if ozonation is advanced or not. The more important question, and what I focussed my research on, is what environmental impacts the implementation of the different technologies would have. And to answer that, we need to know how to perform these environmental assessments. But before we go into that, let me describe the treatment plants that I have been working with, around which my research centres.

2.3 Wastewater treatment in particular: Ängen WWTP

Ängen WWTP is a soon-to-be-built treatment plant in Lidköping municipality, Sweden, that is planned to contain a lot of the latest technologies. At the time of writing, detailed process plans are being developed, and the finished WWTP may thus differ from what is explained in the following.

Figure 2 shows the locations of the current and the planned WWTP in Lidköping. The current WWTP is outdated and operated at a level that exceeds its capacity. The new WWTP will be built on a new location on the other side of the town. The effluent will be emitted to a new recreational area with an artificial stream before it reaches the river Lidan and, a couple of kilometres further downstream, the lake Vänern (the largest lake in Sweden and the third largest in Europe). It will be of (Swedish) medium size with a capacity of 45 000 population equivalents (PE) and ability to treat 14 900 cubic meters of wastewater per day (Dahlberg, 2019).

A simplified process schematic of Ängen WWTP is depicted in Figure 3. The WWTP will implement EBPR by adding an anaerobic reactor in the main line, as well as side-stream hydrolysis for production of volatile fatty acids (VFAs)—i.e., easily degradable substrate needed for the polyphosphate-accumulating organisms (PAOs) responsible for the excess uptake of phosphorus. Nitrogen removal is facilitated through pre-denitrification and nitrification. After the secondary clarifier, the water phase enters quaternary treatment, consisting of ozonation and post-treatment with moving bed biofilm reactors (MBBRs). In these reactors, there will be the option of further nitrogen removal through post-denitrification by adding an external carbon source (ethanol). Subsequently, there will be the possibility of post-precipitation with poly-aluminium chloride and disc filtration to ensure low levels of phosphorus in the effluent. The return sludge from the secondary clarifier is led through aerobic and anoxic reactors before a part of the sludge stream goes to the side-stream hydrolysis and is returned to the biological main line.

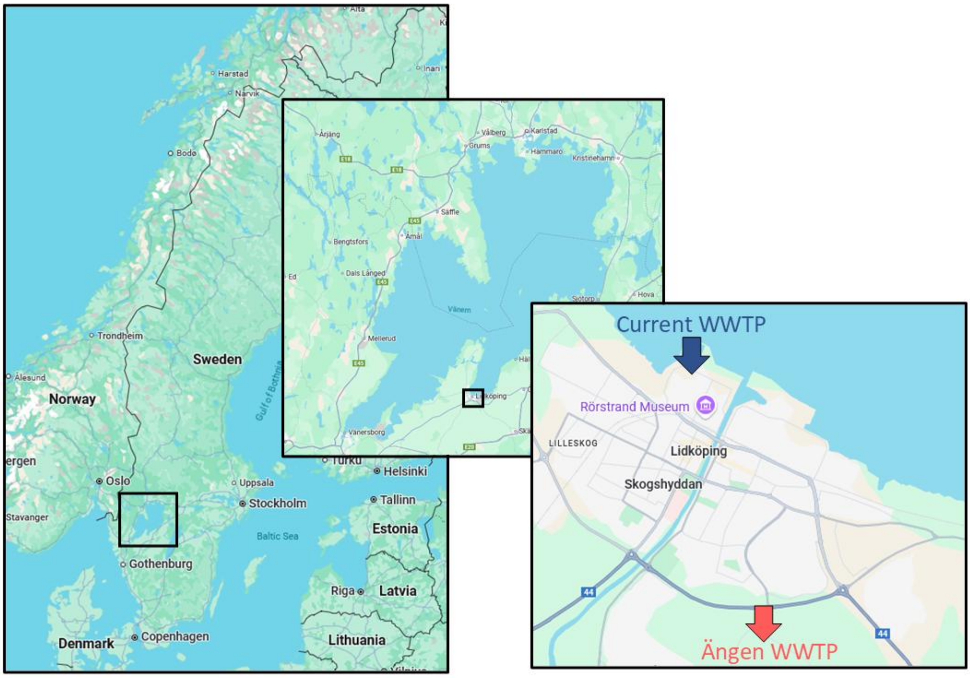

Figure 2
Locations of the current and planned (Ängen) WWTP in Lidköping municipality, Sweden.

The waste sludge is thickened and let into a phosphorus release reactor and another thickener, from which the phosphorus-rich water phase is led to the nutrient recovery stage, while the sludge is led, together with the primary sludge, to the digester. The produced biogas is led to a combined heat and power (CHP) unit for electricity and heat production which will be used on site, and any excess will be sold to the grid. After digestion, the sludge is dewatered with the ammonium-rich reject water sent to nutrient recovery before joining the return sludge treatment. The management of the dewatered sludge is not fully decided at the time of writing this thesis but may entail direct use as fertiliser or possibly pyrolysis (i.e., combustion in an oxygen-free environment to produce biochar).

Nutrient recovery is performed by adding magnesium to the phosphorus- and ammonium-rich streams to create magnesium ammonium phosphate ($MgNH_4PO_4 \cdot 6H_2O$)—i.e., struvite. This struvite is dried and then used directly as a fertiliser or as a raw material in the fertiliser industry.

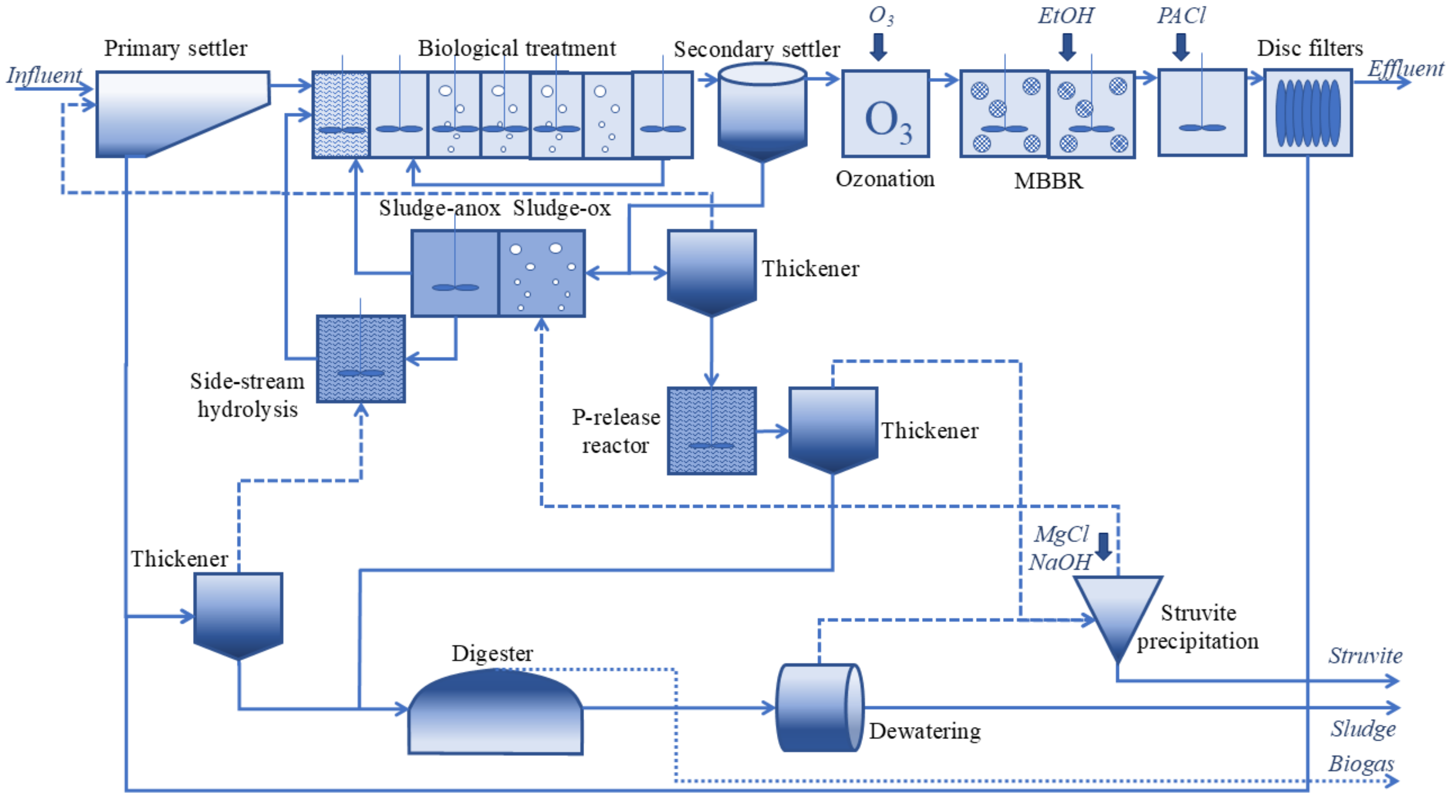

Figure 3
Simplified process schematic of the planned Ängen WWTP plant, Lidköping, Sweden.

However, as the Ängen WWTP is not yet constructed, a dynamic simulation model created in the project as a rendition of the planned plant, became an important tool for data acquisition in this thesis. Based on the technical description in the planning documents (Dahlberg, 2019) a model was created (Wärff, 2021) which was further refined in **Paper II**, and even more so in **Paper IV**. The model was developed in the Sumo22 simulation software (Dynamita, France) in the Sumo4N setting allowing for estimations of nitrous oxide emissions (Hiatt and Grady, 2008; Pocquet et al., 2016) as well as EBPR (Varga et al., 2018) and chemical precipitation (Hauduc et al., 2015).

2.4 Wastewater treatment in particular: NPHarvest

The NPHarvest technology for recovering nutrients from concentrated wastewater streams was developed at Aalto University, Finland, in the beginning of the 2020s (Righetto et al., 2021a, 2021b; Uzkurt Kaljunen et al., 2021). Through the use of solid-tolerant, hydrophobic, gas-permeable membranes, nitrogen in the shape of ammonia is recovered in a pure form. The membranes are operated at atmospheric pressure, potentially resulting in a low energy need. The NPHarvest process works well on highly concentrated waste streams, such as reject water from dewatering of digested sludge (Uzkurt Kaljunen et al., 2021). As displayed in Figure 4, prior to the membranes, there is a chemical pre-treatment with lime addition to precipitate phosphorus and organic material. The solids are removed in a settling tank with a subsequent bag filter in the form of a phosphorus- and calcium-rich sludge-like product. The lime addition also increases the pH so that the nitrogen in the solution is in the form of ammonia that permeates the membranes into the acid tank, where it reacts with sulphuric acid to form ammonium sulphate. The treated wastewater is then returned to the WWTP main line.

Apart from the function of nutrient recovery, this technology also has the function of wastewater treatment. When used on a reject water stream, the load on the WWTP main line decreases, meaning that, for example, less chemicals for phosphorus removal and less energy for nitrogen removal are required in the main line.

In 2021, the NPHarvest pilot plant (2.4 m^3/d) was moved to Sweden and placed in the RecoLab testbed at Öresund WWTP in Helsingborg. The project, which was a collaboration between the local wastewater utility NSVA, Aalto University, Lund University, and Chalmers University of Technology in Gothenburg, resulted in my **Paper I** and a report in the Swedish Water Association's development series (Högstrand et al., 2022). My main contribution to this project was the environmental assessment to aid further development of the technology.

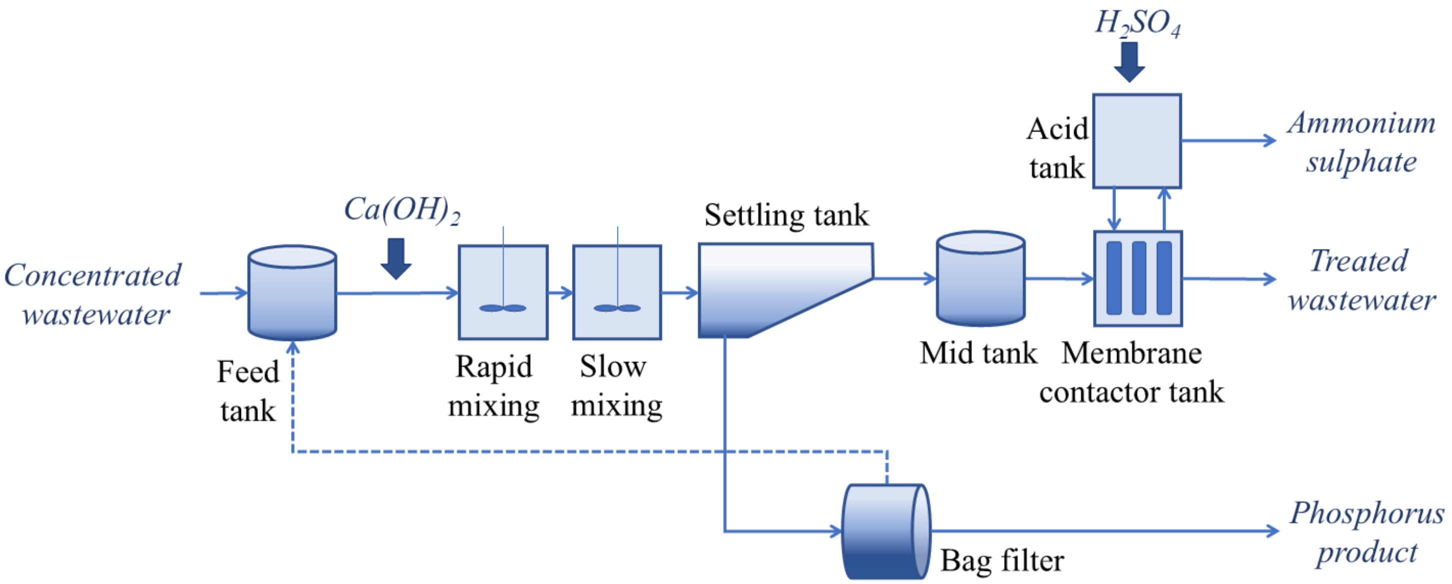

Figure 4
Process schematic of the NPHarvest pilot plant as used at the time of **Paper I**.

3 Life cycle assessment explained

One of the first LCAs to see the light of day was a study conducted by a famous drink producer in 1969 to evaluate what type of packaging to use for their beverages (Bjørn et al., 2018). As time went on, the use of LCA penetrated other sectors; the first wastewater treatment LCAs (WWT-LCAs) were published in the 1990s (Tillman et al., 1998). Today, it has become an important tool also within governance to aid policymaking (Jegen, 2024). The early LCAs lacked consistency, prompting the need for international standardisation—which has been a continuous hot topic since (Bjørn et al., 2018). In 1993, SETAC[1] published a Code of Practice, and in 2006, a general framework and methodological guidelines were standardised in the ISO 14040 and ISO 14044. Other important milestones for the LCA community were the releases of the ILCD[2] handbook in 2010 and the PEF and OEF[3] guidelines in 2012 (Bjørn et al., 2018).

The LCA perspective is a holistic perspective, systematically accounting for—and quantifying—environmental impacts connected to the life cycle of a specific product or function. By clearly defining the scope of the studied system, thoroughly collecting data, carefully assessing the impacts, and cautiously interpreting the results, a (for the goal) relevant assessment of the environmental impacts of a function may be obtained. The purpose of an LCA may be to aid learning by identifying hotspots and unintended burden-shifting between alternatives. It can also be used in decision-making or as part of external communication (Baumann and Tillman, 2004). In ISO 14044:2006, LCA is described like this:

> "LCA addresses the environmental aspects and potential environmental impacts (e.g. use of resources and environmental consequences of releases) throughout a product's life cycle from raw material acquisition through production, use, end-of-life treatment, recycling and final disposal (i.e. cradle-to-grave)."

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[1] The Society of Environmental Toxicology and Chemistry is a global, non-profit organisation founded in the US in the 1970s for communication between environmental scientists, managers, engineers, and others (https://www.setac.org/learn-about-setac/our-story.html).

[2] The International Reference Life Cycle Data System (ILCD) was developed by the Joint Research Centre (JRC) of the European Commission.

[3] Product Environmental Footprint and Organisational Environmental Footprint are LCA methods developed by the European Commission for companies to assess their products or organisations.

LCA is also a methodological tool, an iterative process consisting of four phases, as displayed in Figure 5: goal and scope definition, life cycle inventory, life cycle impact assessment, and interpretation. After completion of one phase, results are evaluated and interpreted to decide whether the scope needs to be altered, new data gathered, the impacts redefined or if the results are sufficiently robust. In this chapter, each phase is described in general, with a specific focus on WWT-LCA practices, together with how it connects to my research.

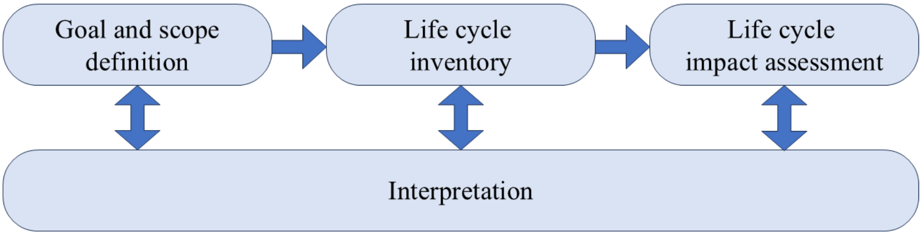

Figure 5
The four phases of LCA according to ISO 14044:2006.

3.1 Goal and scope definition

In the first phase, the goal of the study is defined, pinpointing what question or questions need to be answered. It describes the why, the how, and for whom the study is. Then, the scope is outlined, describing methodology, choices, and assumptions made to answer these questions.

Here, the **system** is defined, and the different options, or scenarios, that will be modelled are explained. An important parameter is the **system function**—what is the purpose of the system? For wastewater treatment, there are several different system functions, as described in the introduction. Oftentimes, the system function "wastewater treatment" is used with a common **functional unit** (FU), being "1 m^3 of treated wastewater" (Corominas et al., 2020). In recent years, studies focussing on nutrient recovery increasingly aim at the nutrient recovery function and may have a FU along the lines of "recovery of 1 kg phosphorus" (Sena and Hicks, 2018). For LCA studies targeting sludge management, the system may be delimited to only account for the sludge treatment processes of the WWTP and commonly adopt an FU of "treatment of 1 tonne total solids (TS)" (Ding et al., 2021). Also, the **system boundaries** need to be defined in terms of technology, geography, and time. Examples of technology-related system boundaries of wastewater treatment systems are shown in Figure 6. For some studies, inclusion of the full WWTP might not be necessary to still catch the major differences between alternatives. The choice of system boundaries depends on the goal of the study and what question is to be answered. For example, it is quite common to delimit sludge treatment studies to only the sludge treatment processes (Ding et al., 2021), quaternary treatment studies to only quaternary treatment processes (Pesqueira et al., 2020), or nutrient recovery studies to nutrient recovery processes—often within the sludge treatment (Lam et al., 2020). This approach of simplifying the technical system is directly related to my RQ 1 and elaborated further on in the next chapter (Section 4.1).

Furthermore, the type of LCA is defined in this phase. This is, for example, whether it is an attributional or consequential LCA. This can, in turn, determine whether impacts from different products in a multifunctional system should be allocated between functions or accounted for by system expansion, as well as whether average or marginal data are to be used. For WWT-LCAs, a hybridisation of methodologies is rather common, where multifunctionality is handled through system expansion by substitution while using average data (Heimersson et al., 2019). This hybridisation is also seen in my papers.

Subsequently, the type of impact assessment methodology used is explained and motivated, together with a description of the intended type of interpretation. Finally, important assumptions, limitations, data requirements, etc., are stated.

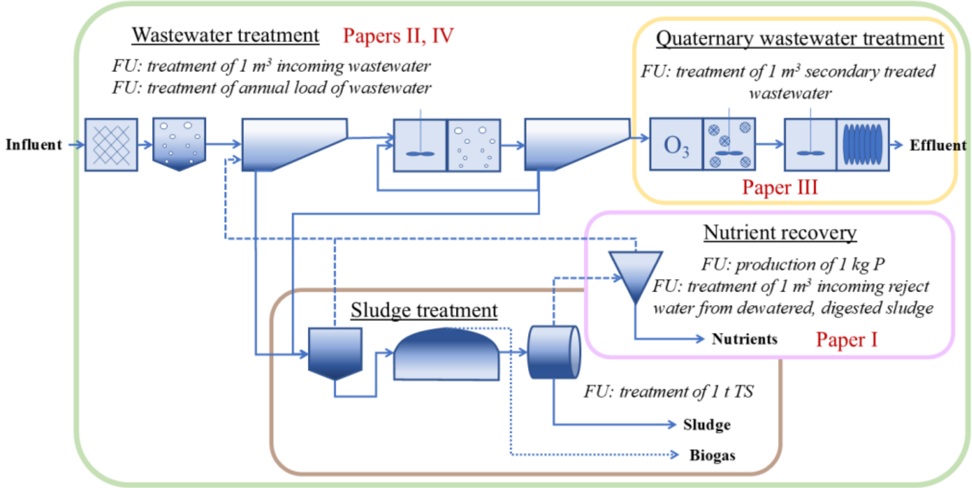

Figure 6

Examples of technical system boundaries drawn around different system functions with suggested functional units (FUs). The systems used in the four papers are also shown.

3.2 Life cycle inventory

In the life cycle inventory (LCI) phase, data for resource use and emissions in the system are collected. All **unit processes** of the system (i.e., small, definable parts of the system, such as “transport of chemicals” or “energy production”) are commonly presented in a flowchart, followed by a detailed description of their inputs (resource requirements) and outputs (emissions). An example of a wastewater treatment flowchart is given in Figure 7.

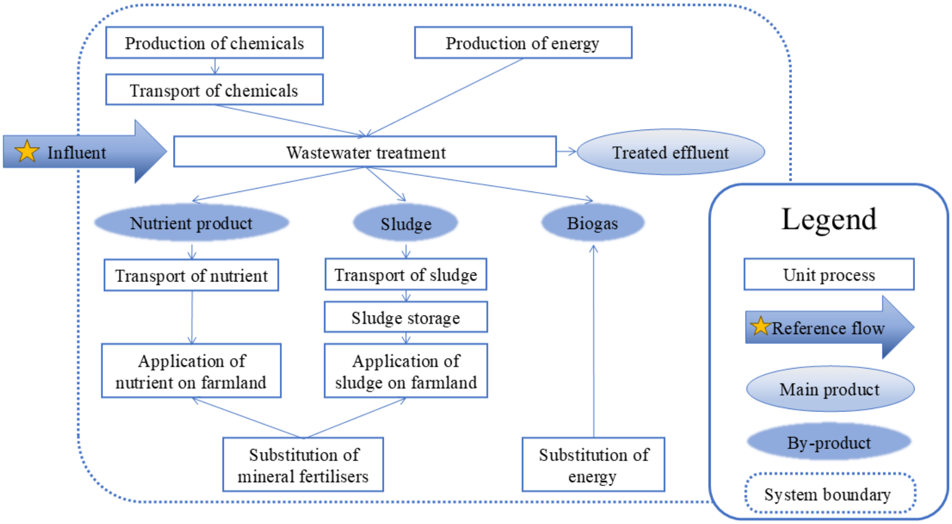

Figure 7
Example flowchart of a wastewater treatment system. The star denotes the reference flow—i.e., the flow connected to the FU—in this case, treatment of a specific amount of incoming wastewater.

Here, data are commonly differentiated as **foreground** or **background data**, with the former referring to the data that are supposedly managed by the owner of the study (e.g., if the owner is a WWTP operator, foreground data are the amount of chemicals used or direct emissions from the wastewater treatment processes). Background data, in contrast, are connected to data that are not so easily controlled by the owner (e.g., emissions from chemical production or emissions from energy production) (Klöpffer and Grahl, 2014). Common foreground data sources consist of measurements at full-scale plants, pilot plants or lab-scale setups as well as models. In contrast, specific background data are often harder to come by—a problem solved using databases such as ecoinvent or literature sources (Corominas et al., 2020). In my work, this collection of foreground data has been an important issue (RQ 2), prompting various solutions, as elaborated further on in Section 4.2.

3.3 Life cycle impact assessment

Life cycle impact assessment (LCIA) consists mainly of six steps: three mandatory (selection, classification, and characterisation) and three optional (normalisation, grouping, and weighting) (ISO 14040:2006, ISO 14044:2006). In the **selection** step, impact categories, such as climate change, eutrophication, etc., that are relevant for the study, are identified and selected. In the **classification** step, the resource use and emissions compiled in the LCI phase are classified with regard to the selected impact categories so that e.g., greenhouse gas emissions are classified into the category 'climate change'. In **characterisation**, all impacts within the same category are translated to the same reference unit by using characterisation factors. Within the 'climate change' category, for example, greenhouse gas emissions are expressed in kg CO_2-equivalents using characterisation factors to convert other gases, such as methane (1 kg CH_4 = 25 kg CO_2-eq.) and nitrous oxide (1 kg N_2O = 298 kg CO_2-eq.). The characterised results may look something like Figure 8. These mandatory steps help quantify environmental impact at midpoint level and are commonly preferred for WWT-LCAs. However, to clarify the results for decision-makers, endpoint level indicators consisting of the three optional steps may be relevant (Corominas et al., 2020).

In the **normalisation** step, the results in all studied impact categories are scaled to a reference value—e.g., global emissions from a certain reference year. Then, the results may be **grouped** and ranked according to impact. Lastly, the results may also be **weighted** by applying a set of weighting factors based on subjectively chosen values—e.g., if climate change is considered a more urgent issue than, say, acidification, it would have a larger weighting factor and thus contribute more to the overall results than if only normalisation was applied. As mentioned, these optional steps may aid interpretation and can be useful for decision-makers. However, they inevitably also increase the subjectivity and decrease the transparency of the results (Corominas et al., 2020).

There are several different LCIA methodologies, each with a specific set of impact categories and characterisation factors. For WWT-LCAs, Corominas et al. (2020) recommend the use of TRACI or ReCiPe (Huijbregts et al., 2017), whereas the European Commission (2021) recommends the use of the Environmental Footprint (EF) 3.1. Several LCIA methodologies are incorporated within various LCA software, such as SimaPro, LCA for Experts (LCA FE), OpenLCA, etc. In my work, the EF 3.1 was utilised in **Papers II**, **III**, and **IV**, whereas the selection of categories in **Paper I** was largely based on the ILCD handbook. All assessment modelling was performed with the LCA FE software.

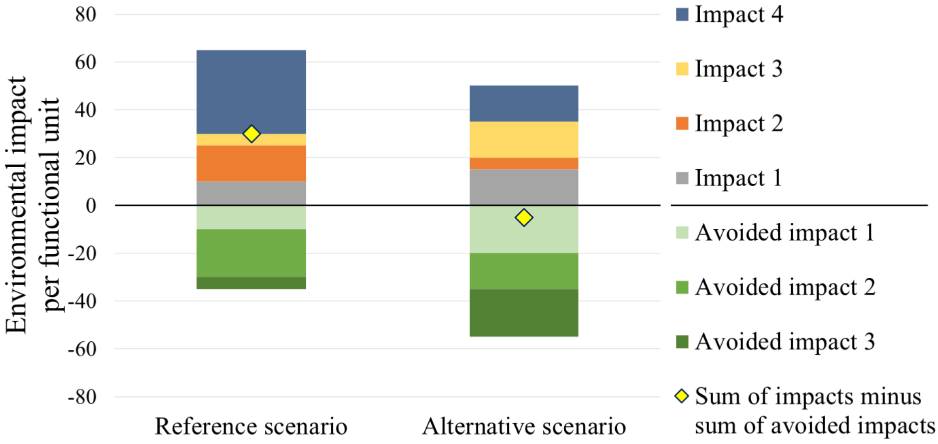

Figure 8
Example of an LCA results graph for a study with two alternatives and with a system expansion to handle differences in functionality. Although the sum of the four impacts is rather similar for both reference and alternative scenarios, the avoided impacts are greater in the alternative scenario, resulting in a net negative impact—i.e., an environmental benefit. Hotspots in impacts are Impacts 4 and 2 for the reference scenario, whereas Impacts 1 and 3 are more important for the alternative scenario.

The most common impact categories within WWT-LCAs include global warming potential (or climate change), acidification, eutrophication, photochemical oxidation, and toxicity (Corominas et al., 2013). However, toxicity is still considered a rather uncertain parameter, lacking in robustness, and its exclusion is (temporarily) recommended in official LCA studies aimed towards the public (European Commission, 2018). Nonetheless, Corominas et al. (2020) list toxicity as a key indicator, recommending its inclusion in WWT-LCAs together with climate change and eutrophication. In my work, I have attempted to approach the issue of toxicity impact assessment (RQ 3), which is further discussed in Section 4.3.

3.4 Interpretation

According to ISO 14044:2006, the results of the LCIA are subject to scrutiny in three steps in the interpretation phase: 1) Identifying significant issues—e.g., assessing the relative contributions of the unit processes to identify hotspots; 2) Evaluating the completeness, sensitivity, and consistency; and 3) Stating the conclusions and recommendations as per identified limitations.

For the evaluation step, a completeness check is made to determine if all relevant data are gathered. If not, the LCI and/or LCIA steps may need to be repeated or the goal and scope adjusted (Klöpffer and Grahl, 2014). The sensitivity check is perhaps the most common way of evaluating the results (Klöpffer and Grahl, 2014) and may entail uncertainty analysis (e.g., Monte Carlo simulations), sensitivity analysis (e.g., adjusting parameters one at a time), and scenario analysis (e.g., testing different energy mixes to account for geographical differences) (Corominas et al., 2020). Lastly, the consistency check assesses whether the methods used and results obtained are in line with what is stated in the goal and scope definition (Klöpffer and Grahl, 2014). In all my papers, creating scenarios for sensitivity assessment has been an important part of the interpretation.

3.5 Future-oriented LCAs

When studying the future, data may many times be missing, and a prospective view is necessary where development of current systems is anticipated. For example, the energy sector of today may be markedly different than that in ten years' time. A future-oriented approach to LCA has been increasingly common in recent years, and many terms related to this field, such as 'prospective', 'ex ante' (meaning 'before the event'), 'anticipatory', and 'lab-scale' exist in literature (Arvidsson et al., 2024). Arvidsson et al. (2024) point out that there are several definitions found in literature of these terms that sometimes coincide and sometimes diverge from each other. Instead, they recommend converging the terms and to use a definition of prospective LCA, stated as:

> "LCA that models the product system at a future point in time relative to the time at which the study is conducted." (Arvidsson et al., 2024)

Furthermore, they also recommend that the temporal position is specified—e.g., a specific year or time span—and that the technological maturity is defined—e.g., using technology readiness levels (TRLs) or scale (such as 'pilot scale'). Looking at my work, the need for a prospective approach for the foreground system is clear (assessing a WWTP that is not yet built), however, to varying degrees for different parts. A compilation of temporal positions and maturity of studied technologies is presented in Table 1.

There are various approaches for applying the prospective perspective, as exemplified by Erakca et al. (2024) in their review on prospective methods for emerging technologies. They identified a list of scale-up methods, including approximation, process engineering, simple extrapolation, simulation, advanced empirical scaling, modular influence estimation, and molecular-structure-based model.

Table 1
Description of temporal position and technological maturity in the four papers.

Paper	Temporal position	Technological maturity
Paper I	Full-scale implementation deemed possible within a few years	**NPHarvest**: TRL 5-6 **Benchmark technology**: TRL 8
Paper II	Immediate present (including seasonal variations over one year)	**EBPR** and **chemical precipitation** are both fully mature and existing technologies (albeit not at the site of interest)
Paper III	Construction of the WWTP is assumed within a few years	**Ozonation**, **MBBR**, and **GAC** are mature technologies, although the combinations (O_3-MBBR and O_3-GAC) are only implemented sparingly
Paper IV	Construction of the WWTP is assumed within a few years	**Anaerobic digestion** is a fully mature technology widely implemented **Struvite precipitation** is a mature technology, with around 120 facilities worldwide (Kabbe, 2023) **Pyrolysis** is not yet fully established in practice (Salva et al., 2025)

In my work, there are examples of several of these. In **Paper I**, the process engineering method was used, as pilot plant data are combined with mass balance calculations. In **Paper II** and **Paper IV**, simulation was the main method for obtaining data at the WWTP-scale. In **Paper III**, simple extrapolation was used on the toxicity reduction rates of the pilot processes, assuming the same rates in the full-scale quaternary treatment in the study. In the same paper, advanced empirical scaling was also used when utilising a power law function to calculate the construction material requirements.

As for the background system, the same assumption—or simplification—was made in all four papers, that any major changes to the energy or transport systems were not expected within the timeframe and that the current systems were assumed relevant. However, by contrasting scenarios with different energy sources (as in **Paper I**, **Paper II**, and **Paper IV**), a preliminary understanding of how an energy transition from fossil-based to fossil-free energy sources might affect the environmental impacts of the system could be generated.

4 Improving methods of prospective life cycle assessment

In this chapter, I focus on the LCA methodological research questions (RQ 1-3). Each of them is associated with a specific phase of the LCA: scope (Section 4.1), inventory (Section 4.2), and impact assessment (Section 4.3).

4.1 Scope: System simplification

RQ 1) Is it possible to maintain the relevance while simplifying the system to reduce the need for data?

One way of approaching the issues with limited or missing data is to consider the possibility of simplifying the system itself through narrowing the functionality as reflected in the FU. By doing so, we could avoid having too many unknowns that may anyway not be that important for the specific study. Let me exemplify by using three instances in my own work: 1) quaternary treatment (**Paper III**), 2) nutrient removal and recovery (**Paper I**), and 3) sludge management (**Paper IV**).

Quaternary treatment
When assessing the impact of quaternary treatment, the technologies of interest are commonly placed towards the end of the WWTP process train where the wastewater has already undergone extensive pretreatment. When the goal of the study is to compare processes rather than to assess the impact of the full WWTP, a delimitation of the technical system may be relevant to make. This is common for LCA studies on advanced or quaternary treatment in literature (Pesqueira et al., 2020) and is also what I did in **Paper III**. This decision was reflected in the FU of "quaternary treatment of secondary effluent" (compared to the FU used in **Paper II** on the full WWTP stating "treatment of incoming municipal wastewater for 45 000 PE for one year"). This allowed me to focus on the latter part of the WWTP process train and to limit the functionality to exclude the nutrient removal impact (while stating and justifying that any differences would be of minor importance anyway). Furthermore, we excluded the sludge impact, as the differences between configurations were deemed minor, as the granular activated carbon (GAC) filter would be placed after

the disc filters from which the sludge would be collected. Any other reject water flows, such as GAC filter backwash water or disc filter cleaning water, were also disregarded. By doing so, the search for data was substantially streamlined to what was deemed most important.

Nutrient removal and recovery

In contrast, when looking at nutrient recovery technologies, these are more commonly placed where nutrient concentrations are higher, such as on the reject water stream from dewatering of sludge digestate. In **Paper I**, we looked at the NPHarvest technology that had two functions: nutrient *recovery* in a desirable form but also wastewater treatment by nutrient *removal* to lessen the load from the reject water on the WWTP. As the data were based on a pilot plant for which no full-scale data were available, limiting the studied system only to the NPHarvest technology was considered the best way forward. However, it was clear that the wastewater treatment function would have impacts on the full WWTP that needed to be accounted for. This gave us the idea of putting the smaller, detailed system within a larger, yet simplified, system and account for those impacts by modelling.

When looking at literature, system delimitation is common in nutrient recovery LCA studies (Lam et al., 2020), but authors have handled the possible impact on the full WWTP in different ways. As elaborated on more in **Paper I**, some authors of LCA studies disregarded these effects altogether (Rodriguez-Garcia et al., 2014), while some included single aspects, such as decreased need for aeration energy or precipitation chemicals (Amann et al., 2018; Remy and Jossa, 2015) or nitrous oxide emissions (Longo et al., 2017; Schaubroeck et al., 2015). Some authors did not specify what impacts were considered nor how (Bradford-Hartke et al., 2015). We decided to develop a model that incorporated several parameters (energy, precipitation chemicals, and nitrous oxide emissions), referring to it as a model of main line impact, or MLI for short. For each of these three parameters, the sensitivity was tested through scenario analysis by varying select assumptions (type and amount of precipitation chemical, theoretical vs practical energy demand, as well as a range of nitrous oxide emission factors). The results showed that the MLI contributed substantially to the benefit of the technology, although the model was quite sensitive to assumptions, such as nitrous oxide emission factor and type of precipitation chemical. We concluded that the MLI method could be applicable for comparisons of reject water technologies and that consideration of the impacts on the main line is important when the process under study is believed to affect the load on the WWTP.

Sludge management

The third example is delimitation of the scope of the sludge management systems. These are also common in literature (Ding et al., 2021), but the way any impacts on the WWTP main line are included differs greatly between studies—from not accounting for it at all, even when comparing scenarios with and without digestion

(Hosseinian et al., 2024), to accounting for the reject water to the WWTP as a separate, external process using data from databases (Rydgård et al., 2024a; Thomsen, 2018). Even more elaborately, Gievers et al. (2025) modelled the aerobic treatment of return process water from hydrothermal carbonisation and compared with anaerobic treatment and wet oxidation process in a sensitivity analysis. For the baseline, aerobic treatment, they adapted a database process so that "the loads of N, P, and C in the process water were introduced into the original process as wastewater pollutants and removed at predefined abatement levels". In some studies, the reject water flows are included in their process descriptions, but neither the water flows' constituents nor impacts are detailed in the inventory (Havukainen et al., 2022; Huang et al., 2022; Luo et al., 2021; Mayer et al., 2021). In a similar manner did Morales et al. (2024) include flows of water from dewatering and drying without specifying constituents; however, instead of sending them back to the WWTP, they modelled them as "discharged into water basins (continental freshwater compartment) as common practice in Norway". Then, there are a few examples of similar approaches as our MLI model in **Paper I**: Faragò et al. (2022) modelled the drying condensate as containing 9% of total nitrogen, which was assumed to correlate to nitrous oxide emissions using a range of emission factors. Yoshida et al. (2018) included more parameters, as they modelled the impact from the digester reject water on the reject water treatment through electricity requirement, ferric chloride dose, and nitrous oxide emissions. However, due to limited data, they did not differentiate between reject water from dewatered mixed sludge and anaerobically treated sludge.

In **Paper IV**, the focus was on sludge management and nutrient recovery and thus a limited part of the WWTP. However, the full WWTP was modelled, essentially because we had the means to do it—the model existed at large already from **Paper II**, though alterations were needed—but also because we believed that the different scenarios (with and without digestion, struvite precipitation, and pyrolysis in different combinations) would render such diverse loads back to the main line and that they would need to be accounted for. And the results were clear: accounting for MLI is highly important when comparing scenarios that so greatly affect the return loads (as with or without digestion does). Our results confirmed the importance of nitrous oxide emissions and precipitation chemical, as included by Yoshida et al. (2018), but also highlighted other parameters, such as methane emissions from the digester and CHP, chemicals for struvite precipitation, and external carbon source for enhanced post-denitrification. Due to the largely fossil-free energy mix in Sweden, differences in energy requirements between the scenarios only showed minor impact; however, when applying a more fossil-dependent energy source, energy optimisation would be recommended.

Returning to the research question, it seems that simplification of the technical system may be a relevant approach of reducing the need for data if aligned with the goal and purpose of the study. However, I would argue that impacts on the larger system often need to be accounted for, especially if the return loads from different alternatives differ considerably (such as when comparing scenarios with and without digestion). The praxis in literature varies greatly, and guidance on this matter would be beneficial. I would further argue that in general, application of a thought-through MLI model with a simplified system is a relevant path forward when data on the larger system are limited, although when applicable, modelling of the full WWTP may uncover unforeseen environmental hotspots.

4.2 Data inventory: Pilot plants and models

RQ 2) To what extent are pilot plants and models useful for creating data inventories?

The next approach to manage the issues of limited full-scale, locally measured data is to try to create it. For this, there are various ways, such as lab experiments, pilot plant measurements, model simulations, or even literature data extrapolation. In my work, and what I will focus on in this sub-chapter, are the pilot plant and model options.

Pilot plants

I may not have had my hands down the slippery, slobbery processes of a pilot plant myself, but I have watched from a front-row seat how incredibly much work it entails and how often things go wrong. So, let me just take a moment to express my gratitude and admiration to those of my colleagues who have spent so many working hours (and time outside working hours) on wrestling with their equipment. I have the impression that running a pilot plant is adhering strictly to Murphy’s law: anything that can go wrong will go wrong. As one of my colleagues suddenly expressed in a phone call: “Det glöder på fel ställe” (“It’s burning in the wrong place”)—and he quickly hung up.

Nevertheless, there are numerous benefits of running pilot plants, as well, and there are generally plenty of useful lessons to learn. Pilot plants provide an amazing opportunity to try things out, to test new technologies in a setting more advanced and technologically mature than lab scale, while still having time to further develop things before going full scale. The NPHarvest pilot plant used in **Paper I**, was the size of a shipping container and could therefore be shipped between locations and tried on different feedstocks from both biogas plants and WWTPs (Uzkurt Kaljunen et al., 2021).

But, the scale of the pilot plant is also one of the drawbacks, at least when performing LCA and wanting results for full scale. As exemplified in the review by Erakca et al. (2024), there are a number of different methods for scaling up lab- and pilot-scale results for assessment of emerging technologies. In **Paper I**, we avoided scale-up completely by stating that the comparison was for this smaller scale and that no scale-up to full scale was necessary to adhere to the purpose of the study. In **Paper III**, construction data were scaled from larger full-scale facilities to our medium-sized WWTP by using an equation to account for non-linearity in economy of scale (Gallego-Schmid et al., 2019; Remer and Chai, 1993; Risch et al., 2022). For treatment efficiency, pilot plant data were used but were not corrected for scale.

However, one of our main findings in **Paper I** was discovered through the combination of pilot plant measurements and mass balance calculations. This combination indicated a substantial loss of nitrogen through ammonia evaporation—not seen in the measurements as the gas phase was not monitored—which in turn proved to be one of the greatest environmental hotspots shown in the LCA (more on these results in Section 5.4).

Thus, to answer the research question, I would argue that pilot plants are decidedly useful but need proper consideration—or at least transparent discussion—of scale. Furthermore, the combination with other methods, such as mass balance calculations, could render valuable insights and should be considered when applicable.

Models

A scientific model is an approximation, a representation of an object or phenomenon one wishes to use to better understand the real world. Typically, there are three types of models: physical models (e.g., globe representing Earth), conceptual models (e.g., a diagram of the water cycle), and mathematical models (e.g., a set of equations). Connected to the mathematical model—but used for considerably more complex systems and larger datasets—is the computational model (e.g., as used for climate simulations). In my work, when I talk about models, I generally mean the computational model in the shape of a dynamic simulation model of a WWTP or in the shape of the LCA model of environmental impacts at system-level. To some extent, I also refer to the mathematical ones. But before going deeper, let me preface this section with a quote by the British statistician George E.P. Box:

> "All models are wrong, but some are useful." (Box, 1976)

That said, there may very well be useful lessons to learn when employing models, while bearing in mind that humility and transparency in interpretation are highly encouraged. And there are great benefits in using computational, simulation models for data acquisition in WWT-LCAs, as the ever-growing body of literature has shown (see e.g., Besson et al., 2021; Bisinella de Faria et al., 2015; Foley et al., 2010; Igos et al., 2017; Monje et al., 2022; Ontiveros and Campanella, 2013). Assessments on WWTP-levels can be performed at a fraction of the cost and time than if trials would have been carried out in real, full scale. For a WWTP that already exists, using a dynamic simulation model as a digital representation, or a digital twin, could allow for running different scenarios on the same treatment plant, changing operational or external parameters without affecting actual effluents and recipients (Molin et al., 2024).

Dynamic simulation models could also be used, as in my case in **Paper II** and **Paper IV**, before the WWTP is commissioned to aid decision-making prior to implementation. The model allowed us to compare different configurations that were tested on the same influent wastewater—something that would not be possible in real, full scale. The biggest drawback, however, and something that severely impacted the robustness of the results, was the obvious lack of a WWTP to calibrate the model against.

Nonetheless, we were able to learn from our model used in **Paper II**. It indicated an notable difference in methane emissions between biological and chemical phosphorus removal configurations not reported elsewhere. When digging deeper, it was noted that the model was highly sensitive to the apparent fermentation rate and that the parameters guiding that rate varied between common simulation software in use (Downing et al., 2023). These differences on the process-model level translated into the LCA results, with great impact on the overall results.

Thus, to answer the research question, I would argue that employing dynamic simulation models for data acquisition is highly useful, but I want to stress the difference in selected parameters between different modelling software and the importance of further measurement to refine this so that the impact of methane in anaerobic wastewater treatment processes would be more correctly established.

Regarding **Paper IV**, a combination of computational and mathematical models was used. As there were no ready-made units for drying or pyrolysis within the WWTP simulation software, I created a simplified process unit by using the mathematical model of transfer functions (Thomsen, 2018) for determination of the partitioning of selected wastewater constituents and adding these to point separators within the simulation programme. Mathematical modelling was also used when assessing the impact of increased soil carbon content from agricultural application of sludge and char as a corresponding increase in crop yield. In addition, the computational PLCI 2.0 model (Rydgård et al., 2024b) was applied to estimate the impact of agricultural application of the different products in terms of mineral fertiliser substitution, phosphorus loss and crop uptake of phosphorus.

Hence, there were several different models in use to produce the inventory data. There are examples in literature of constructed interfaces between models, such as the WWTP dynamic simulation model and the LCA model (e.g., Bisinella de Faria et al., 2015), but in my case, I was the interface. A seeming patchwork of models was required to reach the goal of the study, and selected uncertainties in the models were evaluated in a sensitivity analysis. This analysis showed that for most scenarios, the preference order of the configurations (from lowest to highest impact) was not extensively altered when changing specific parameters. The largest deviation from the baseline scenario was seen in the scenario utilising fossil energy sources—i.e., altering background data on the LCA-model-level—which resulted in substantially larger impacts (but also increased benefits for configurations with energy recovery) as well as affecting the impact order. This model assumption thus had a larger impact on the results than the choice of transfer functions in the pyrolysis model, the selected region in the PLCI 2.0 model, the inclusion of biochar stability in the soil organic carbon model as well as accounting for bioavailability of heavy metals in the LCA model. This highlights two things: 1) background datasets may have larger impact on the results than foreground considerations, and 2) a combination of different models may still be robust. I would therefore argue—in response to the research question—that this patchwork-integration of models may prove useful when illuminating major uncertainties, and carefully—and transparently—selecting impactful parameters.

4.3 Impact assessment: Capturing toxicity

RQ 3) Can alternative impact assessment approaches meaningfully capture missing aspects related to toxicity?

When attempting to perform toxicity estimations based on chemical analysis results, the lack of data is striking. According to a recent study, there may be as many as 350 000 chemical substances on the market (Wang et al., 2020), while "only" about 20 000 commercial chemicals have been analysed in environmental media, with the 100 most studied substances accounting for 34% of the dataset (Muir et al., 2023). Many of these chemicals used in society end up in the wastewater (Venkatesan and Halden, 2014). Regular measurements at the WWTPs mostly just entail organic matter, nutrients, and heavy metals, and even in specific measurement campaigns, less than 200 substances may be targeted (e.g., Loos et al., 2012). But even if we had more measured results on a large number of substances, characterisation factors within LCIA are also lacking. In the USEtox 2.1 model, there are currently some 4 000 substances listed—not even close to the 350 000 substances reported by Wang et al. (2020). Besides, chemical analysis of specific substances does not target unknowns or indicate potential mixture effects.

One potential approach to manage this issue is to use effect-based methods, such as bioassays. This is common within the field of risk assessment, and these types of measurements have been done for a long time (Escher et al., 2021). However, within the field of LCA, bioassays have only recently been introduced (Pedrazzani et al., 2018). This was the main topic of **Paper III**, in which we tested the approach as introduced by Pedrazzani et al. (2018) in a Swedish context. In this method, results from bioassays are translated through dose-response curves into the bioanalytical effect concentration (BEQ) of a reference substance. This reference substance may then, in turn, be used in the LCA with the corresponding LCIA characterisation factor. Thus, this method still requires the availability of USEtox characterisation factors but not as many as for chemical analysis.

Alas, the availability of characterisation factors is a bottleneck. In our study, of the ~100 substances detected at Lidköping's current WWTP, removal rates of the quaternary treatment could be calculated for ~30 compounds, of which only 14 had corresponding characterisation factors in USEtox 2.1. Still, performing an LCA based on 14 substances is more than what is done in many studies on advanced wastewater treatment technologies (Pesqueira et al., 2020).

In **Paper III**, we used the biological endpoints estrogenicity, aryl hydrocarbon receptor activity, and oxidative stress, as recommended for wastewater monitoring by Escher et al. (2021). Data for the *in vitro* bioassays and chemical analysis were found in literature and included measurements at the current WWTP in Lidköping, as well as two quaternary treatment pilot plants (Baresel et al., 2024; Holm and Önnby, 2022). The results of our study indicated a much larger benefit of implementing quaternary treatment when assessed with bioassays than if based on chemical analysis. In fact, quaternary treatment could now be portrayed as having a net negative toxicological impact—i.e., a toxicological benefit—also when accounting for the increased use of chemicals and energy. This means that trade-offs can potentially be more realistically captured.

That said, we also noted methodological issues that need to be attended to, going forward. For example, it is possible to translate the bioassay results into other reference substances (Enault et al., 2023), and the resulting impact on the LCA level may vary considerably, as we showed in a sensitivity analysis. To answer the research question, I would therefore argue that for this method to be more meaningful and reliable, guidance on choice of reference substances, in parallel with further development of the USEtox database, is required. I would further argue that guidance on choice of bioassays, biological endpoints, as well as impact categories is encouraged, although we did not explicitly assess the impact of different selections in our study.

Lastly, using bioassays only for parts of the system (such as on the wastewater effluent in the foreground as compared to background datasets of chemical production) might render a bias in interpretation that needs to be acknowledged. Presumably, if background datasets were also to employ bioassays, the relative importance of the wastewater effluent would likely diminish, and our results should therefore be taken as preliminary. However, not all contexts exhibit the issues of low concentrations of a very large number of substances potentially with mixture toxicity, as is the case for wastewater treatment. Nevertheless, I would argue that incorporation of bioassays into LCA is a relevant, yet complementary, approach to conventional LCA, with promising prospects, should guidance on methodological choices be developed.

5 Wastewater treatment process development and selection

RQ 4) What are the hotspots and desirable paths forward according to beyond state-of-the-art LCA for WWT process development and selection?

The purpose of this chapter is to showcase the lessons learned from the four case studies as a reply to RQ 4. The chapter is divided into sections according to treatment stage, starting off with secondary/tertiary treatment (Section 5.1), followed by quaternary treatment (Section 5.2), reject water treatment (Section 5.3), and, lastly, sludge management (5.4).

5.1 Phosphorus removal in secondary/tertiary treatment

In **Paper II**, I evaluated the environmental life cycle differences between biological and chemical phosphorus removal. The aim was to assess hotspots and trade-offs, especially through the perspective of potential chemical scarcity. Although comparisons between these two technologies had been made previously (e.g., Rahmberg et al., 2020), the frame of chemical scarcity was new. This had its roots in the global supply chain instabilities highlighted since the war in Ukraine began in 2022. Preparations and evaluations of the state of readiness for emergencies were done on several levels of society—so, also in the Swedish water sector (Kemikalieinspektionen, 2022; Nilsson, 2021).

My research group was given the assignment from the Swedish EPA to evaluate the possibilities of increasing the implementation of biological phosphorus removal (EBPR) in Sweden, which resulted in a report (Jönsson et al., 2025). Within the frame of that project, my task was to perform an environmental assessment and comparison of EBPR and conventional chemical precipitation (CP), and the result of this study is found in the report and in more depth in **Paper II**.

In short, the results showed that there are indeed environmental trade-offs when comparing EBPR and CP. As mentioned in Section 4.2, the model indicated increased air emissions of methane from the EBPR configuration. Though the values are uncertain and prompt further evaluation of the hydrolysis rate, the results highlighted a lack of measurements from existing EBPR plants in published literature. Nevertheless, the main differences between the configurations are the anaerobic reactors in the main line, in the side stream, as well as before digestion of the EBPR configuration, suggesting an increased risk of unwanted methane production at unfavourable conditions.

Among the benefits of the EBPR process, on the other hand, is that since the precipitation chemicals inherently contain pollutants, such as heavy metals, decreased use of these leads to fewer pollutants in effluent and sludge. This means increased favourable conditions, should the sludge be used for agricultural application. However, the arguably largest benefit of running an EBPR plant as compared to a CP plant is the stability in process, should there be a scarcity of precipitation chemicals. Looking at the normalised, weighted results in Figure 9, the increased impact on the effluent in the CP shortage scenario compared with the CP baseline, when chemicals are available, is substantial. This increase in impact from the effluent is even greater than the difference in air emissions between the EBPR and CP scenarios. Notably, the difference in impact from production of precipitation chemicals is also larger than the difference in air emissions. The results show that the net results are rather similar for both configurations for the baseline, but in case of a long-term, complete chemical shortage, EBPR is a more stable and recommended process.

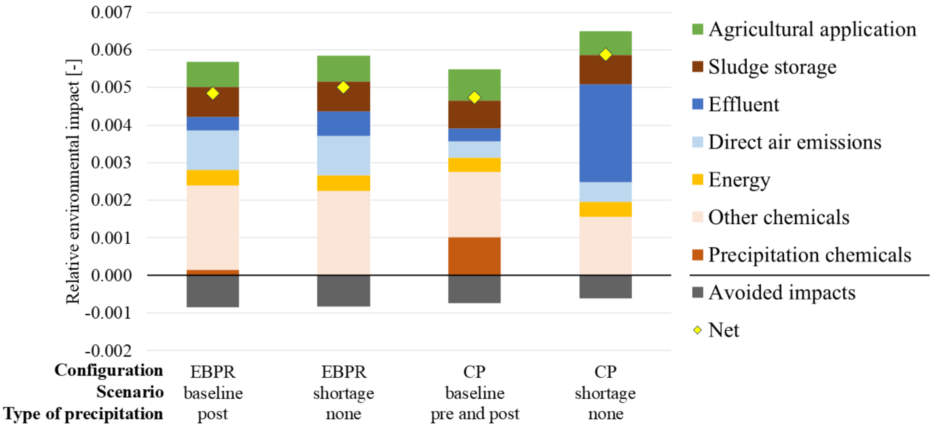

Figure 9
Normalised and weighted results comparing EBPR to CP with pre-precipitation, showing the dimensionless, relative impact of each unit process and scenario. Baseline scenario means that post-precipitation is used to reach the effluent requirements of 0.2 mg phosphorus per litre. Shortage scenario means no precipitation chemicals are added without further change of configuration. “Avoided impacts” include avoided crop production; avoided heat and electricity production; and avoided production, transport, spreading and application of mineral fertilisers.

In Sweden today, there are 430 WWTPs with a capacity larger than 2 000 PE, of which around 40 WWTPs implement EBPR to some degree. Just short of 400 WWTPs (or 6.5 million PE) have, as recipients, either inland water courses or the Baltic Sea (Villner and Myhr, 2022). These fresh- and brackish waters are generally more susceptible to phosphorus inputs, causing eutrophication and algae blooms, etc. This indicates that a long-term, nationwide shortage of precipitation chemicals may affect many of these sensitive waters.

It was therefore in our interest to evaluate the possible impacts of increased implementation of EBPR in Sweden, and preliminary calculations based on the inventory data were performed to initiate a discussion (Jönsson et al., 2025). We assumed that the inventory data were linearly scalable from the 45 000 PE at the WWTP level to 6.5 million PE at the nationwide level. The results should therefore only be considered as possible indications, rather than more elaborate predictions. Among the drawbacks of increased implementation were increased electricity use (corresponding to about 0.01% of the total electricity use in Sweden), decreased biogas recovery potential (corresponding to about a sixth of the amount of biogas that is currently produced at WWTPs, or slightly more than what is currently flared), and potentially increased emissions of greenhouse gases (as discussed earlier, there is uncertainty regarding this claim but, in the worst case, this increase would correspond to 0.16% of Swedish total consumption-based emissions in 2021 and a larger share if only territorial emissions were considered). The benefits, on the other hand, entail decreased dependency on precipitation chemicals (corresponding to the amount of chemicals required for production of drinking water for 6.8 million PE annually, allowing for strict prioritisation in times of shortage), decreased emissions of heavy metals (corresponding to a reduction in flows of heavy metals to sludge of 40% chromium, 20% nickel, and 10% copper compared with current levels), increased potential of struvite recovery (potentially corresponding to 7% of Swedish annual use of mineral phosphorus fertilisers), and, of course, decreased risk of eutrophication in case of chemical shortage (a marginal increase of phosphorus compared to the five-times-higher phosphorus release, should the bulk of WWTPs continue with chemical phosphorus removal).

So, returning to the research question, what are the hotspots and desirable paths forward when it comes to WWT process selection of secondary/tertiary treatment? In short, I would argue that:

- Direct emissions from biological reactors need to be mitigated. Further studies on surveying and optimising operational parameters are recommended (as are already ongoing);
- The hotspots of conventional chemical treatment are the production and use of chemicals. The pollutant content in chemicals should not be overlooked;

- EBPR could facilitate increased nutrient recovery in desirable—i.e., less polluted—forms (e.g., struvite), which should be considered; and
- Although chemical processes may render an easier and more stable daily operation, in the event of chemical shortage (a perhaps not-as-unlikely event as one would like to think, given the present state of the world), a biological process would ensure substantially lower effluent levels of phosphorus and, thus, a lower risk if crisis appears. The possibilities of implementing biological phosphorus removal at new or existing WWTPs should be considered, perhaps by using the guidelines suggested by Jönsson et al. (2025).

5.2 Micropollutant removal in quaternary treatment

For quaternary treatment, I will use the case study in **Paper III** as an example. At the Ängen WWTP, quaternary treatment through ozonation is to be implemented. When using ozonation, a post-treatment step of some sort is highly recommended to reduce the potentially toxic impacts of transformation- and by-products (Stalter et al., 2010b, 2010a). Commonly utilised types of post-treatment include sand filters and GAC filters (Lim et al., 2022), where the latter is suggested as the most efficient (Bourgin et al., 2018). But, other types of biological treatments, such as MBBRs, have also been implemented in a few facilities (e.g., Stapf et al., 2020). At Ängen, the O_3-MBBR combination has been suggested in the initial plans, but due to recent increased concerns regarding PFAS, the O_3-GAC combination has gained more interest. Furthermore, what was lacking in available literature was an environmental assessment on different ozonation post-treatments. We sought therefore to fill that gap and to aid the decision-making process for the new WWTP by performing an LCA, comparing O_3-MBBR with O_3-GAC.

As mentioned in Section 4.3, the benefits of using the treatment technologies in terms of reduced toxicity in the effluent were assessed by bioassays and conventional chemical analysis. And, as already mentioned, the bioassays portrayed a vastly better picture of quaternary treatment than conventional chemical analysis. As for the comparison between O_3-MBBR and O_3-GAC, Figure 10 shows the overall, normalised, and weighted impacts of the two technologies. The difference between the technologies when it comes to toxicity reduction efficiency was perhaps not so clear, and more data are needed to be able to see those differences more clearly. Instead, the largest hotpots are GAC and ozonation operation, showing that the O_3-GAC had a larger environmental impact than O_3-MBBR, although the net impact, when including the benefits based on bioassays, portrayed them as equally beneficial. The results also showed that technological variations, such as source of ozone (from air or liquid oxygen) and GAC filter lifespan (longer = 40 000 bed volumes, or standard = 20 000 bed volumes), had a profound impact on the LCA results, meaning that wise selection of these may have as much impact on the results as choice of configuration (O_3-MBBR or O_3-GAC).

Revisiting the research question: What are the hotspots and desirable paths forward when it comes to WWT process selection of quaternary treatment? I would argue that:

- Production and transport of activated carbon is an important hotspot. Maximising GAC filter lifespan, as well as producing activated carbon within the country to reduce transport distances (also, making sure the carbon is sufficiently dry before sending it to reactivation), would reduce impact substantially. Furthermore, as suggested by others, using biogenic rather than fossil carbon would also reduce the impacts (Vilén et al., 2022), although the resulting treatment efficiency has been debated (Takman et al., 2024); and
- Comparing the two configurations, O_3-MBBR may have less environmental impact than O_3-GAC, but the treatment efficiency needs to be further established; especially, more bioassay measurements need to be conducted.

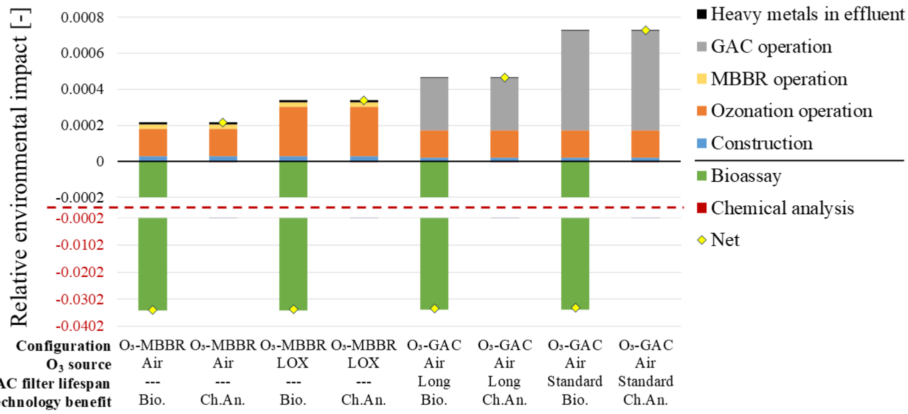

Figure 10
Normalised and weighted results showing the dimensionless, relative impact of each unit process and scenario. A comparison of O_3-MBBR to O_3-GAC whilst varying source of ozone (air or LOX = liquid oxygen), GAC filter lifespan (long = 40 000 bed volumes, or standard = 20 000 bed volumes), and benefit assessment method (Bio. = bioassays, or Ch.An. = chemical analysis). Note the broken y-axis and the different scales.

5.3 Nutrient recovery in reject water treatment

In the NPHarvest project, mentioned in Section 2.4 and detailed in **Paper I**, the reject water treatment and nutrient recovery technology NPHarvest was assessed from a life cycle environmental point of view and compared with a benchmark technology. This benchmark technology consisted of the current best available technology—i.e., struvite precipitation and ammonia stripping. The location at RecoLab testbed at Öresund WWTP allowed for direct comparison between the two technologies on the same influent wastewater (although campaigns were run at different times; so, absolute identicality, such as when performing the dynamic simulations in **Paper II** and **Paper IV,** was not possible to achieve).

The results of the hotspot evaluation are displayed in Figure 11. For climate change, the main impact is from chemical production for both technologies, but the effect is basically outweighed by the benefits from substituted fertilisers and decreased nitrous oxide emissions. Both technologies are essentially climate-neutral in this setup and within the system boundaries, assuming Swedish electricity mix and that the benefits may be realised. Looking at the primary energy demand, both technologies require quite a bit of energy, but due to the Swedish electricity mix being basically fossil-free, this is not reflected in the climate change category. However, should another energy source be used, such as European average, the environmental impact would increase substantially.

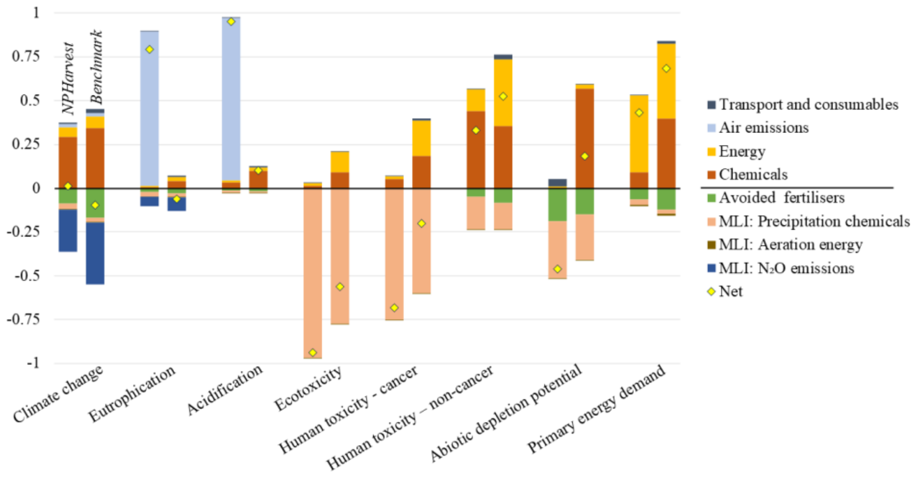

Figure 11
Hotspot identification for NPHarvest (left bar in each pair) and benchmark technology (right bar in each pair) for the eight evaluated midpoint impact categories. The results are normalised, so that the length of the largest bar in each impact category equals one (1). MLI = Main line impact.

As for the eutrophication and acidification categories, the mass balance calculations of the NPHarvest pilot trial revealed potentially large emissions of ammonia, which completely dominated these two categories. These emissions need to be reduced before going full scale, which should be achievable when ensuring air-tight reactors. Figure 11 also shows the large benefits of considering the main line impact (MLI)—in particular, with regard to avoided nitrous oxide emissions and use of precipitation chemicals.

Thus, when circling back to the research question: What are the hotspots and desirable paths forward when it comes to WWT process development of nutrient recovery? I would argue for the specific case of NPHarvest that:

- The main hotpots consisted of ammonia emissions, chemical use, and energy requirements; and
- Before implementation in a larger setting, ammonia emissions need to be adequately mitigated, use of other chemicals should be properly evaluated, and further energy optimisation should be considered.

5.4 Nutrient recovery in sludge management

The purpose of the fourth case study, as described in **Paper IV**, was to evaluate the life cycle environmental impacts of sludge management from a resource recovery perspective. In other words, we wanted to evaluate three different processes (anaerobic digestion, struvite precipitation and pyrolysis) and assess in which form the recovered phosphorus would be most favourable from a life cycle perspective—as sludge, struvite or biochar?

To that end, we created five different configurations: anaerobic digestion only (A), anaerobic digestion and struvite precipitation (AS), anaerobic digestion, struvite precipitation and pyrolysis (ASP), anaerobic digestion and pyrolysis (AP), and pyrolysis only (P). Utilising the dynamic simulation model created in **Paper II**, and refining it further to include the pyrolysis process, rendered—together with a patchwork of other models as expanded upon in Section 4.2—the foreground data used as input to the LCA model.

In short, the largest differences were seen between configurations with and without pyrolysis (A and AS compared with ASP, AP and P). In six of the 16 assessed midpoint impact categories of the baseline scenario, the configuration with lowest net impact contained pyrolysis, although the configuration with only pyrolysis had the lowest impact in only one. For eleven categories, configurations with struvite precipitation exerted the lowest net impact. The configuration with anaerobic digestion only (A) showed the lowest impact in two categories.

Looking at the normalised and weighted results of the baseline scenario, Figure 12a shows that the net relative environmental impact (denoted with the yellow diamond) for different configurations was rather similar, but with lower impact from configurations with pyrolysis (configurations ASP and P). The main contributors to this difference were emissions from sludge storage and application on arable land, as also noted as important contributors in earlier works on sludge management (Svanström et al., 2017). The benefit of carbon sequestration has been deemed the most important reason for implementing pyrolysis (Morales et al., 2024). And for the climate change category, the benefit of carbon sequestration was indeed notable. However, after weighting, the negative effect on crop yield when using biochar instead of mineral fertilisers, outweighed the benefit of carbon sequestration in the present study.

The main hotspot of the weighted results relates to direct air emissions. Adding a digester and CHP increased direct air emissions by 50% (AP vs P), but adding a pyrolysis unit also increased air emissions by 14% (AS vs ASP). However, as pointed out in Sections 4.2 and 5.1 as well as in **Paper II**, the methane emissions from the dynamic simulation model were uncertain. Therefore, a sensitivity analysis of the impact of an altered anaerobic fermentation rate was made, similar to what was done in **Paper II**—depicted in Figure 12b. It should be noted, however, that a major difference between **Paper II** and **Paper IV** was that in the latter, emissions from the sludge-line—i.e., methane slips from the digester and the CHP unit—were included along with the assumption that 15% of the influent carbon was of fossil origin (Reppas-Chrysovitsinos et al., 2024), and thus contributed to fossil carbon dioxide emissions. In Figure 12, it can be seen that although there was a considerable difference between the baseline and sensitivity analysis results with regard to direct emissions, the sludge-line and pyrolysis emissions also contribute to a similar extent as the waterline emissions in the baseline scenario. Furthermore, this change in model parameter also affected the phosphorus uptake. More precipitation chemicals were required, especially for configurations A and AP which needed a lower feedback setpoint to reach the phosphorus effluent requirement. This also meant less struvite recovered (configurations AS and ASP) which can be seen in configuration ASP as more phosphorus remained in the biochar and resulted in a slightly larger loss of crop production than in the baseline scenario. Nevertheless, as the main difference pertained to air emissions, and the waterline emissions of methane were rather similar for all configurations, the net order of the configurations (from lowest to highest impact) did not change drastically between the two scenarios, except for configurations AS and AP switching 3rd and 4th places.

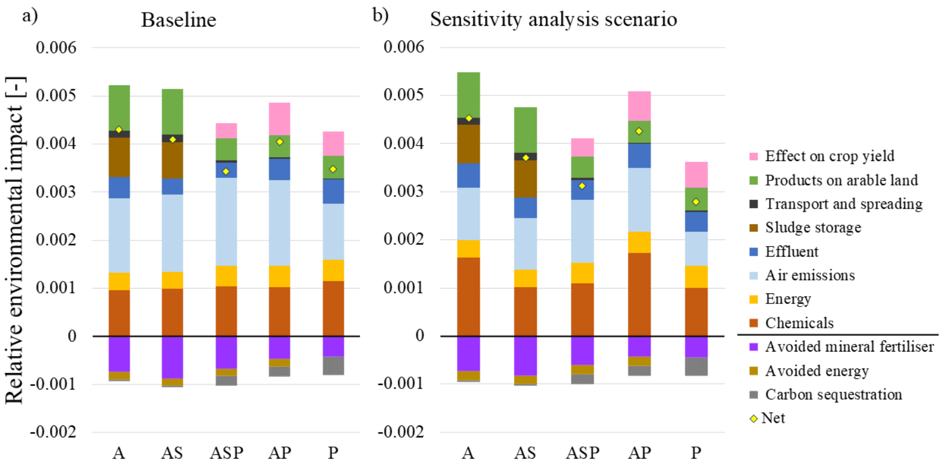

Figure 12
Normalised and weighted results showing the dimensionless, relative impact of each unit process for the different configurations of the baseline scenario (a) and a scenario with altered anaerobic fermentation rate (b).

So, coming back to the research question, what are the hotspots and desirable paths forward when it comes to WWT process selection of sludge management for nutrient recovery? In short, I would argue that:

- Regardless of selected configuration, mitigating direct emissions—from the WWTP itself but also from sludge storage and spreading on arable land—should be top priority, especially in energy systems with low fossil share such as in Sweden;
- Struvite precipitation in the context of an EBPR system seems to be a good idea, at least from a resource recovery perspective and if struvite and mineral fertilisers in fact exhibit similar characteristics in terms of phosphorus loss and crop uptake; and
- Anaerobic digestion without pyrolysis is straightforward albeit with higher risk of increased direct emissions. In contrast, pyrolysis without anaerobic digestion may reduce direct emissions and the produced biochar may increase carbon sequestration capabilities. However, the fertiliser effect of biochar should be further evaluated as the characteristics of biochar could result in a lower crop production, compared with using mineral fertilisers.

6 Looking back, moving forward

As with all research, for each question I answered, new ones were raised. The scope of a thesis is only so large, and things inevitably needed to be left for the ones who come after me. I will therefore use this chapter to reflect on my contributions as well as some important limitations. Furthermore, I will take the opportunity to look at the future, reflect on missing aspects, and ponder on what might come next.

What are my contributions?

As a researcher, my hope is that the results of my work will contribute to increased knowledge in the scientific field, as well as benefit society. Regarding the former, my contribution of knowledge is quite clear, with methodology improvements advancing the LCA field, as well as revealing insights on specific WWT processes. As for the latter, topics such as circular economy, resource recovery, and sustainability are increasingly important in the society of today. My research contributes to this by identifying hotspots of environmental impact for several treatment technologies. Specifically, I believe that my research could lead to cleaner water at decreased environmental cost.

Even more specifically, our research has undoubtedly contributed to the development of the NPHarvest technology in the years following our project[4]. After Juho Uzkurt Kaljunen defended his thesis at Aalto University in March 2023, he continued to work towards commercialisation of the technology and, with the help of business developers, created the company NPHarvest in September of 2023. Six months later, they had managed to secure funding through investors and a government grant and could finally begin to develop the technology. The pilot plant (2.4 m^3/d) used at RecoLab for **Paper I** was reshaped in 2024 so that the liming part was the only thing remaining of the pre-treatment, and the membranes were reinvented without the stirrer to change flow profile and contact time. As the company grew to five people in 2025, efforts were made to scale up the process. Thus, an industrial-scale, mobile demonstration plant (20 m^3/d) was designed in May, constructed in June, transported to Turkey in July, and operated for three months at a biogas plant in Ankara with the start in early August.

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[4] The following two paragraphs are based on personal communication with Juho Uzkurt Kaljunen in late August 2025.

As for how our project helped this development, the finding of the magnitude of ammonia emissions was important, although easily amended. The reactors and tanks of the demo unit are now sealed and airtight, and the installed air space gas analyser has not registered any ammonia emissions at all. Furthermore, and arguably the biggest aid in the process of commercialisation, the conclusion of climate neutrality has been a helpful argument in receiving funding. Next in line for NPHarvest is to secure new funding, optimise the production cost of the demo unit, and then go for full-scale installation. Perhaps it might even be time for a follow-up LCA.

What are important limitations?

One of the major limitations in my work, I believe, is that the simulation model in **Paper II** and **Paper IV** has not been calibrated to a real plant. The obvious reason for this is because the WWTP is not yet built. But that was also the reason as to why we decided to create and use the model in the first place. Nevertheless, uncalibrated models contribute with uncertainty that needs to be highlighted.

I should also comment on the limitations that environmental LCA inherently has and what aspects are not covered. As usual, when there are large projects in the making (such as constructing a new WWTP), the costs need to be considered. The financial aspect has been out of scope for my research but is obviously an important part of the decision-making basis. Furthermore, social aspects, such as work environment, etc., also would need to be included in a more thorough evaluation in a decision-making setting.

What are the prospects of LCA in wastewater treatment?

When it comes to the application of LCA within wastewater treatment, I believe that it is here to stay—but perhaps for other uses than thirty years ago. Now, with LCAs being increasingly applied within policymaking (Jegen, 2024), I believe that it is only a matter of time before this way of quantifying environmental impacts for regulatory purposes is implemented also in the wastewater sector.

I also believe that the prospective perspective is increasingly important in evaluating alternatives, making decisions prior to construction and when developing new technologies. To this end, the use of digital tools will likely increase even further with dynamic simulation models, digital twins, as well as AI for data processing. The combination of LCA, AI, and wastewater treatment modelling would undoubtedly be interesting to explore further.

Should there be more EBPR in Sweden?

In our project on the possibilities of increased implementation of EBPR in Sweden, we found that when looking internationally, there are numerous incentives to apply EBPR. Among them are the high costs for precipitation chemicals, taxes on phosphorus emissions, the overhanging risk for chemical shortage, as well as increased potential for nutrient recovery (Jönsson et al., 2025). However, as touched upon in Section 5.1, there are also challenges that need to be addressed before advocating for increased implementation of EBPR, including the possibility of increased methane emissions, a reduction in biogas production potential, and increased energy demand. Of these three, I believe that clarifying the potential difference in direct emissions between configurations is of utmost importance. By increasing the measurements at full scale, operating WWTPs (especially EBPR plants but also conventional CP plants), a better understanding of these important and highly variable emissions can be reached. Furthermore, a thorough assessment of the costs of performing a large-scale transition on a national level from mainly CP to mainly EBPR needs to be done, as well as establishing who or what will carry those costs. As discussed in our report, there may be many WWTPs that could be converted to EBPR, for at least parts of the year, with only limited reconstruction. I therefore suggest performing an investigation of the (Swedish) WWTP stock to evaluate more precisely how many and which WWTPs could be easily converted.

Do bioassays have a place in WWT-LCAs?

As mentioned in Section 4.3, there are challenges that need to be addressed before bioassays could become a standardised method in LCA. These include further evaluation of how to properly select reference substances, biological endpoints, as well as type of bioassays. Nevertheless, I believe that effect-based monitoring may bring a meaningful contribution to LCA, especially in the light of the new UWWTD. It would be very interesting to see how this assessment approach in the long run could aid the decision-making process to evaluate what WWTPs would benefit from implementation of quaternary treatment. But for this to be possible, I would argue that further research, more guidelines, and updated databases are necessary.

What is the future of nutrient recovery from wastewater?

Recovering valuables from waste streams without risking human health or the environment is important and complex. In my research, I have looked at several different nutrient products of wastewater treatment including sludge, biochar, struvite, and ammonia sulphate.

Applying sludge on farmland is a very straightforward means of disposal, where both macronutrients and organic matter are returned to the fields. However, with that comes the risk of both inorganic and organic pollutants, as well as other direct emissions from sludge storage and spreading. Also, the notion of a possible ban of spreading the sludge on arable land may be a cause to find suitable alternatives even before the ban is in place.

Converting the sludge to biochar might reduce the impacts of storage, transport, and spreading, as well as increase the carbon sequestration potential. However, inorganic pollutants are mostly intact and thus accumulated—although possibly less bioavailable—and the fertiliser potential needs to be further proven. Moreover, legislation of biochar from sewage sludge as a fertiliser is lacking, which poses a major hindrance to large-scale utilisation of the product within agriculture. However, perhaps the biggest hindrance to widespread implementation of pyrolysis at WWTPs, in my opinion, is that although some (and an increasing number of) facilities exist, the general experience of construction and operation is still inadequate. As a more concrete suggestion, I would like to see LCAs on pyrolysis based on “actual” measurements rather than on literature, models, or information from technology providers.

Instead, if creating specific products such as struvite or ammonium sulphate, the pollutant level in the nutrient products ending up on soil is even further decreased while providing a more stable composition of constituents. Furthermore, these two products are also regulated in the EU fertiliser regulation (EU, 2019/1009), which paves the way for more widespread implementation. However, the market is not quite there yet, and I would therefore like to see increased market incentives for these types of products.

What do I then see of the future? I believe that what we need now is clear guidance on a national level, market incentives, and increased technological readiness. These three interdependent factors are necessary for a sustainable, circular management of wastewater valuables.

7 Conclusions

In my thesis, there is a dual research focus on both LCA methodology and WWT processes, which renders a two-fold result. My research contributes with novelty and insight with regards to several phases of the LCA methodology: scope, data inventory, and impact assessment. It also touches upon most parts of the WWTP: secondary, tertiary, and quaternary treatment, as well as sludge management and nutrient recovery. Returning to the questions posted in the introduction—the **how** and the **what** of life cycle environmental assessment of wastewater treatment prior to implementation—I have shown in my thesis some of the possible answers for them.

In short, the **how** can be successfully handled by 1) simplifying the system and thus decreasing the need for data, 2) creating data through pilot plants and models, and 3) using alternative impact assessment approaches to capture missing aspects. More specifically, I found that…

… delimiting the technical system can be one way of reducing the need for data, if accounting for the impacts it might have on the larger system in some other simplified, yet relevant, way. Our "MLI" model showed that these impacts may be as important as the ones from the technology itself.

… the use of pilot plants is decidedly useful, but careful consideration of scale differences needs to be made. Furthermore, the combination of pilot plants with other approaches generating relevant data, such as mass balance calculations, may render valuable insights otherwise overlooked.

… dynamic simulation modelling is a helpful tool for data acquisition, which ensures that scenarios are comparable by allowing the use of the same influent and similar process configurations, etc. However, it is important to consider model uncertainties and the impacts they might have on the results – in our case, the model was highly sensitive to the anaerobic hydrolysis rate, which significantly affected the overall LCA results.

… incorporation of bioassays in LCA to capture missing aspects of toxicity is straightforward, although with challenges. The actual incorporation was rather simple, but questions arose regarding choice of reference substances, bioassays, and biological endpoints. Furthermore, it is important to be aware of inherent bias in the system if bioassays are only implemented in part and not over the entire system (both fore- and background) and how this may affect the interpretation.

The **what** aspect of environmental assessment rendered more specific conclusions from the four case studies on wastewater treatment process development and selection. To elaborate, I found that…

… biological and chemical phosphorus removal in secondary/tertiary treatment both have benefits and drawbacks, showing that trade-offs in selecting one or the other are unavoidable. While chemical phosphorus removal in general has a higher process stability, in times of chemical shortage, this process—for obvious reasons—does not work. EBPR, on the other hand, may allow for easier resource recovery through struvite precipitation while minimising use of heavy-metal-containing precipitation chemicals. Most surprising was the difference in methane emissions as indicated by the model, which highlighted the apparent lack of greenhouse gas measurements at actual, full-scale EBPR plants.

… the benefits of implementing quaternary treatment for removal of micropollutants greatly exceeded the environmental impacts of the increased energy and chemical use when the evaluation was based on bioassays, as opposed to when it was based on conventional chemical analysis. This was true for characterised, as well as normalised and weighted, results. The comparison of the two ozonation post-treatment methods (O_3-MBBR and O_3-GAC) showed that technical considerations for each configuration (such as source of ozone and GAC filter lifespan) could be as important as the choice of technology itself.

… the nutrient recovery technology NPHarvest performed similarly to the benchmark technology. To address the most important hotspots, it is suggested to implement airtight reactors to prevent ammonia emissions, evaluate alternative chemicals, and optimise the energy efficiency in the further development of the technology.

… the sludge management and nutrient recovery technologies anaerobic digestion, struvite precipitation and pyrolysis all had benefits and drawbacks. But configurations with struvite precipitation did, in general, exert lower impact than those without. Anaerobic digestion was straightforward but had notably higher air emissions. In contrast, pyrolysis reduced direct emissions, but the fertiliser effect of the resulting biochar was worse, potentially leading to decreased crop production.

To conclude, my thesis joins the increasing body of literature on future-oriented LCAs and improves methods of life cycle environmental assessment. Although the results show potential impacts and not definite predictions, the usefulness of early indications when designing a WWTP that will last a generation is undeniable. In other words, although their true worth remains for the future to show, the "sounds" made by these limited, digital, or pilot "trumpets" are not only available and practical but indeed valuable if interpreted thoughtfully.

Postface: Who will play the trumpet?

I once attended a seminar with a renowned presenter on the topic of the environmental movement. He talked about many things, but one thought that remained with me afterwards was the process of data turning into impact, and the hierarchy of data, information, knowledge, wisdom and impact. Raw data can be categorised into information. Pieces of information may then be connected into knowledge. From knowledge, new insights can be made. Connecting insights turns into wisdom. And wisdom utilised leads to impact.

The story of my research began as a quest for data and how to utilise them. I do not know how far up the pyramid of impact it will come. Perhaps the biggest impact from my years as a researcher is the impact it has had on me. Being a doctoral student is being on a journey, perhaps not so much in space—especially not during the pandemic years—but in maturity and insight. I have learned so much along the way, and I can really see how I have grown. Both as a researcher and as a human.

As a researcher, my learning curve of LCA methodology is quite visible when, for example, comparing the models of phosphorus recovery impact used in **Paper I**, **Paper II**, and **Paper IV**, respectively. The more I learned and felt I could manage the LCA models, the more complex models I attempted to implement.

Another example of how these years have shaped me is how I've grown as a leader in the projects I've been part of. Although being first-name author in all of them, my roles in the co-author groups varied between the different papers. In **Paper I**, I was basically thrown (a very nice throw, with a soft landing, I might add) into the group, whereas in **Paper IV**, I created the group and contacted relevant members—with **Paper II** and **Paper III** being somewhere in the middle.

As a human, however, perhaps the most important factor for growth during these years was becoming a mother. It has brought me perspective on life, as well as inspiration to do tedious and hard work for the sake of learning. I remember specifically when my son was about one year old, just learning to walk and wanting to go out on the balcony in our apartment. It was a bit tricky, stepping over the threshold, and he almost fell over. But once out, what surprised me (being a lazier adult looking for the most convenient path—I'm an engineer after all), he went straight in again, over the threshold. And then out once more. And then he spent a good deal of possibly ten minutes just going back and forth over the threshold. This moment has ever since spurred me on in my research, inspiring me not to give up when something is hard but to prevail, to practice, and be patient.

And here I am, attempting to play a trumpet that after seven years still does not exist but that hopefully makes some audible sounds. The impact is yet to be heard, but at least we now have one more trumpeter.

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WASTEWATER MANAGEMENT TODAY faces the dire challenge of complying with the diverse goals of protecting human health, ecosystems and the climate simultaneously. This is done through enforcing stricter effluent requirements, mitigating climate impact and recovering valuable resources. However, there are inherent trade-offs between these goals, and a system-perspective is vital to avoid unwanted burden-shifting. To this end, applying the systems thinking approach that is life cycle assessment (LCA) is helpful.

This thesis addresses the issues of wastewater treatment of today and tomorrow from a life cycle perspective by improving and applying LCA methodology in the wastewater treatment context.

SOFIA HÖGSTRAND holds a Master of Science in Environmental Engineering. In 2018, she began to pursue her doctorate in Water and Environmental Engineering at Lund University. Water and sustainability have been her interests for a long time, and her research has focused on the two fields of LCA and wastewater treatment.

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