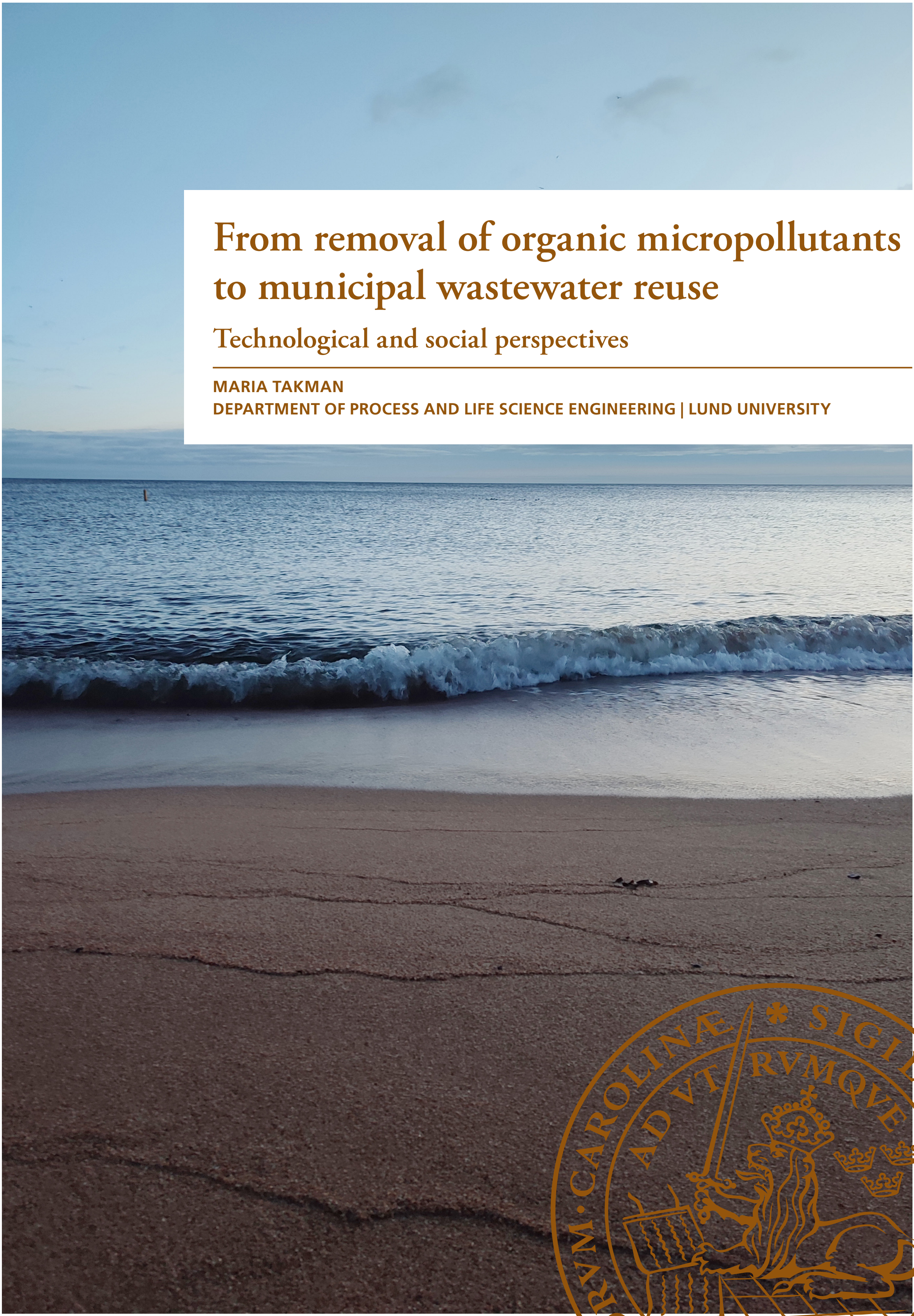

From removal of organic micropollutants to municipal wastewater reuse

Technological and social perspectives

MARIA TAKMAN
DEPARTMENT OF PROCESS AND LIFE SCIENCE ENGINEERING | LUND UNIVERSITY

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Maria Takman

LUND
UNIVERSITY

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 26th of April at 09.00 in lecture hall KC:A, Kemicentrum, Department of Process and Life Science Engineering, Naturvetarvägen 14, Lund

Faculty opponent
Associate Professor Foon Yin Lai, Department of Aquatic Sciences and Assessment, Swedish University of Agricultural Sciences

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Title and subtitle: From removal of organic micropollutants to municipal wastewater reuse
– Technological and social perspectives

Abstract:

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The results showed that the treatment of wastewater with a process combination of a membrane bioreactor, GAC filtration, and disinfection with ultraviolet radiation improved its quality to a level that approached that of drinking water. Limits on some organic micropollutants, such as pharmaceuticals, are generally not included in drinking water legislation or legislation for irrigation with reused wastewater, adding insecurities to the evaluation of water quality. Nevertheless, the water was treated to a high degree and potentially constitutes a beneficial supplementary resource for irrigation or drinking water production during drought or times of water scarcity. Whether the water criteria need to be complemented with additional parameters merits further investigation.

Organic micropollutants are removed by GAC filters primarily through adsorption but also through biological degradation. Measurements from a full-scale GAC filter indicated degradation of certain pharmaceuticals, which was confirmed in laboratory experiments with granules from various GAC filters. The degradation appeared to be affected specifically by the oxygen concentration in the filters and by operation time.

In the second part of the thesis work, the discourses over wastewater reuse and desalination on the Swedish islands of Öland and Gotland were examined and compared with general discourses identified from literature. Wastewater reuse and desalination are ways of producing drinking water when groundwater and surface water resources are not sufficient and are often compared in the literature. The results showed that the local discourses often had similarities with the general ones and that there were differences between the two islands. Desalination on Gotland seemed to be more controversial than wastewater reuse and desalination on Öland, and the perceptions of wastewater reuse and desalination were affected by many factors, such as visions and values with regard to welfare or sustainability and other political topics—for example local industries.

Key words: Wastewater reuse; Water reclamation; Organic micropollutants; Granular activated carbon; UV disinfection; Discourses

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The history of men is reflected in the history of sewers.
Victor Hugo

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Preface

This dissertation is the result of a PhD project that was associated with the Water and Environmental Engineering Group in the Department of Process and Life Science Engineering and the Agenda 2030 Graduate School at Lund University. The Agenda 2030 Graduate School is interdisciplinary, with PhD students from all faculties at Lund University, aspiring to address the multiple challenges in the 2030 Agenda and to contribute to advancing research on sustainable development (Lund University, 2024). One of the studies (Paper I) was also financed by Svenskt Vatten (project number 20-112). The dissertation work was performed in collaboration with Österlen VA and the Division of Applied Microbiology in the Department of Chemistry, Lund University.

The influence of the Agenda 2030 Graduate School on the thesis work is reflected by the interdisciplinary collection of papers, ranging from chemical and microbial analyses of wastewater treatment processes to a discourse analysis of wastewater reuse and desalination.

Acknowledgments

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To the Water and Environmental Engineering Group: thank you for all of the interesting discussions, workouts, and lab cleanings. To Ellen, Maja, and Simon, for your always-positive spirits; to Gertrud, for welcoming me into the group; to Ashley, for all your fun initiatives; to Alexander, for discussions and cleaning in the scintillation lab; and to Sofia, Rubén, and Isabell, for being great officemates! I would also like to thank the Division of Chemical Engineering. I have really enjoyed all of the fikas and lunches with you guys, and I will truly miss this friendly workplace. Also, a thank you to Mahla, my officemate during the last time.

To the Lund University Agenda 2030 Graduate School, for financing this PhD position and to Svenskt Vatten, for project financing. To all of the people who were involved in the Agenda 2030 Graduate School for all of the seminars, retreats, and courses, giving me new insights that without whom I would probably never have gotten. To Linn, for all of the bike rides and other adventures; to Ida, for workouts and movie nights; and to Naja, for the interesting lunch discussions.

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Also, thank you to my friends and family. To my parents, Anders and Christina, for always helping out. To Anna, Einar, and Marion, for discovering Copenhagen, France, and Oregon together. To Louis, Thompa, Metsi, Jonte, Gustav, Linn, Stefan, and Sebbe, for introducing me to Scania. To Hedvig, ille, Rasmus, Emma, Jonas (Pp), Malin, Jennie M.B., and Hugo, for hikes, ski trips, and other adventures. And Jolinn: it was really nice to have you as a fellow Lundensare for a while! To Jennie H.H., for discovering the coastal towns between Lund and Göteborg. To Emelie, for all of the nice walks and fikas. To Felicia, MVP, and everyone we played soccer against—I loved playing soccer with you!

Abstract

In this thesis, wastewater reuse was investigated from technological and social perspectives, based on which the thesis was divided into two parts. In the first part, the removal of chemical and microbial contaminants by full-scale and pilot-scale granular activated carbon (GAC) filters, in different process combinations, was examined. In the second part, the perceptions of reused wastewater were studied through a discourse analysis and compared with those of desalinated seawater, which is another alternative water source.

The results showed that the treatment of wastewater with a process combination of a membrane bioreactor, GAC filtration, and disinfection with ultraviolet radiation improved its quality to a level that approached that of drinking water. Limits on some organic micropollutants, such as pharmaceuticals, are generally not included in drinking water legislation or legislation for irrigation with reused wastewater, adding insecurities to the evaluation of water quality. Nevertheless, the water was treated to a high degree and potentially constitutes a beneficial supplementary resource for irrigation or drinking water production during drought or times of water scarcity. Whether the water criteria need to be complemented with additional parameters merits further investigation.

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Populärvetenskaplig sammanfattning

Vi kan återanvända avloppsvatten – men vilket vatten ska vi använda till vad?

Klimatförändringar, tillsammans med en ökad vattenanvändning, kan i vissa områden leda till vattenbrist. Samtidigt byggs en del avloppsreningsverk om så att de ska kunna rena läkemedel från avloppsvattnet. Om vi renar avloppsvattnet mer och mer, är vattnet till slut så rent att vi kan börja använda det till olika saker? Kan vi vattna med det, eller till och med dricka det?

SVERIGE
Akut vattenbrist i Kivik – tankbilar kör i skytteltrafik på nätterna
Uppdaterad 2021-07-31 Publicerad 2021-07-31

Återanvänt toalettvatten kan bli dricksvatten på Österlen
UPPDATERAD 28 JUNI 2023 PUBLICERAD 28 JUNI 2023
Det krävs nya metoder för att rena avloppsvatten så att det går att använda till bevattning av åkrar, parker, trädgårdar – och på sikt även till dricksvatten. I Kivik på Österlen finns ett av få reningsverk i Europa som har kommit långt i den här processen.
– Vi kommer att få se fler sådana här anläggningar framöver. Kivik ligger 20 år före många andra, säger Michael Cimbritz, docent i

Bevattningsförbud i hela kommunen
- Den långvariga torkan och de många turisterna på Ö gör att Simrishamns kommun ännu en gång inför bev förbud.
- Förbudet gäller från och med i dag torsdag i hela kom

Vattenbrist en risk på svenska semesterorter

SVERIGE
Reningsteknik kan göra avloppsvattnet drickbart
Uppdaterad 2021-08-08 Publicerad 2021-08-08

ONSDAG 24 JUNI 2020
Extremhettan på väg – kan bli 30 grader i Sydsverige
HÄR GÄLLER BEVATTNINGSFÖRBUD
HÄR GÄLLER ELDNINGSFÖRBUD

Akut vattenbrist plågar Österlen
Nyheter. Höga temperaturer, lite nederbörd och högt turisttryck har tärt på grundvattennivåerna i Kivik.
– Detta är extremt, säger Johan Nyholm på Österlen VA, som låter köra in dricksvatten i tankbilar från Simrishamn om nätterna. **Sidan 12**

t. Foto: Pontus Lundahl/TT

Vattenbrist i var fjärde kommun
2:05 min Min sida Dela

SÅ DRABBAS BONDEN PETER AV EXTREMA TORKAN
"Kommer det inte regn inom 14 dagar förlorar jag halva höstvetet"

Vattenbrist kan hanteras på flera sätt. Dels kan vattenförbrukningen minskas genom exempelvis bevattningsförbud. Om det inte räcker kan alternativa vattenresurser användas, såsom havsvatten efter avsaltning, eller avloppsvatten efter rening i flera steg.

Samtidigt pågår andra förändringar inom vattenområdet. Ny lagstiftning förväntas inom EU, med krav på rening från bland annat läkemedel från avloppsvatten. Vissa avloppsreningsverk har redan byggts om för rening från dessa ämnen, exempelvis avloppsreningsverken i Kivik, S:t Olof och Degeberga i östra Skåne. I delar av östra

Skåne har också vissa år varit så torra år att vatten har blivit en bristvara. Att återanvända det renade avloppsvattnet skulle då kunna vara ett smart sätt att hantera de begränsade vattentillgångarna.

Återanvändning av avloppsvatten – inte bara en fråga om teknik

Återanvändning av avloppsvatten kan vara kontroversiellt, men det behöver inte vara det. På vissa platser har det blivit protester när avloppsvatten har använts till produktion av dricksvatten, medan det i andra fall har funnits en stor förståelse och tillit. Människors attityd gentemot återanvänt vatten kan påverkas av flera faktorer, och återanvändning av avloppsvatten uppfattas bland annat ofta som mer hållbart än alternativet avsaltning, som ibland associeras med negativ miljöpåverkan, men som kan uppfattas som säkrare och mer beprövat.

Avloppsvattenrening

För att få bort läkemedel från vattnet och för att kunna återanvända det renade vattnet till bevattning eller dricksvattenproduktion behöver vattnet renas, och det är inte alltid självklart hur mycket det ska renas eller vad det kan användas till efter olika grader av rening.

En reningsteknik som är vanlig för att rena läkemedel från avloppsvatten är filtrering med granulerat aktivt kol (GAK). Fullskaleanläggningar med GAK för detta ändamål är ovanliga, men avloppsreningsverken i Kivik, S:t Olof och Degeberga är utrustade med just GAK-filter. Detta har skapat en unik möjlighet för oss att studera hur sådana filter fungerar i verkligheten, till exempel hur reningsgraden förändras över tid.

Vi såg att mikroorganismer började växa relativt snabbt på kolet, att några av dessa bröt ned olika läkemedel, bland annat diklofenak, vilket är ovanligt i biologiska reningsprocesser. Dessutom minskade koncentrationerna av flera indikatorbakterier, såsom *E. coli*, i vattnet.

Är vattnet tillräckligt rent?

Trots att GAK-filtren minskade koncentrationerna av *E. coli* och läkemedel så var minskningen av *E. coli* inte stor nog för att vattnet skulle bedömas som tillräckligt rent och säkert för återanvändning (det vill säga fritt från sjukdomsspridande bakterier). Därför gjordes också experiment med desinfektion i form av ultraviolett (UV) ljus.

Över 100 olika mikroorganismer och kemikalier analyserades i det UV-behandlade vattnet från Kiviks avloppsreningsverk. Det visade sig att koncentrationerna klarade gränsvärdena för dricksvattenkvalitet (enligt Svenska Livsmedelsverket), med nitrat som enda undantag. Nitrat är lyckligtvis enkelt att rena från avloppsvatten, och rening av nitrat är vanligt på stora svenska avloppsreningsverk. Detta är goda nyheter, eftersom UV-desinfektion kommer att byggas på Kiviks avloppsreningsverk, med förhoppningen att vattnet ska kunna användas för bevattning.

Trots att vattnet klarade gränsvärdena för dricksvattenkvalitet är vi inte helt säkra på att det är säkert att dricka under lång tid, eftersom gränsvärden saknas för vissa av kemikalierna som finns i avloppsvatten, till exempel läkemedel. Om vi ska producera dricksvatten från eller vattna med renat avloppsvatten bör det utredas om vi behöver gränsvärden på dessa substanser också. Trots de osäkerheter som finns kan vi fortfarande konstatera att vattnet som renats i Kiviks avloppsreningsverk, efter desinfektion med UV-ljus, når i princip dricksvattenkvalitet, och att det renade vattnet bör kunna nyttjas som en välkommen extra vattenresurs om det blir torka och vattenbrist.

List of Papers

Paper I

Takman, M., Svahn, O., Paul, C.J., Cimbritz, M., Blomqvist, S., Struckmann Poulsen, J., Lund Nielsen, J., Davidsson, Å. (2023) Assessing the potential of a membrane bioreactor and granular activated carbon process for wastewater reuse – A full-scale WWTP operated over one year in Scania, Sweden. *Science of the total environment*. 895, 165185. https://doi.org/10.1016/j.scitotenv.2023.165185

Paper II

Takman, M., Paul, C.J., Davidsson, Å., Jinbäck, M., Blomqvist, S., Cimbritz, M. MBR and GAC filtration followed by UV disinfection – implications for wastewater reuse at full scale. Manuscript accepted by the journal *Water Reuse*.

Paper III

Takman, M., Betsholtz, A., Davidsson, Å., Cimbritz, M., Svahn, O., Karlsson, S., Karstenskov Østergaard, S., Lund Nielsen, J., Falås, P. Biological degradation of organic micropollutants in GAC filters – temporal development and spatial variations. Manuscript submitted.

Paper IV

Takman, M., Cimbritz, M, Davidsson, Å. and Fünfschilling, L. (2023) Storylines and imaginaries of wastewater reuse and desalination: the rise of local discourses on the Swedish islands of Öland and Gotland. *Water Alternatives*, 16(1).

Author's contribution to the papers

Paper I

With the other authors, I applied for grants and planned the study. I conducted the sampling, analyzed the wastewater quality parameters as well as concentrations of *E. coli*, total coliforms, total cells, and organic micropollutants, and performed the DNA extraction. I analyzed the data and wrote the manuscript in dialogue with the other authors.

Paper II

I planned the experimental work, performed the experiments, and analyzed the wastewater quality parameters, as well as the *E. coli*, total coliform, and total cell concentrations. I performed the data analysis and wrote the manuscript in dialogue with the other authors.

Paper III

I participated in planning the study and performing the experiments with the Kivik and Degeberga GAC. I analyzed the wastewater quality parameters and data and wrote the manuscript in dialogue with the other authors.

Paper IV

I planned the study, designed the interview guide, and planned the interviews. I performed the interviews and media analysis, analyzed the results, and wrote the manuscript in dialogue with the other authors.

Related publications

Takman, M., Cimbritz, M., Davidsson, Å., Paul, C.J., Svahn, O., Blomqvist, S. (2022) Återanvändning av renat avloppsvatten. Potential efter rening med en membranbioreaktor följt av granulerat aktivt kol. Svenskt Vatten. Report No.: 2022-14.

Gidstedt, S., Betsholtz, A., Cimbritz, M., Davidsson, Å., Hagman, M., Karlsson, S., **Takman, M.**, Svahn, O., Micolucci, F. (2024) Chemically enhanced primary treatment, microsieving, direct membrane filtration and GAC filtration of municipal wastewater: a pilot-scale study. *Environmental Technology*. 45(1). https://doi.org/10.1080/09593330.2022.2099307

Takman, M. (2019) Opportunities for wastewater reuse in Sweden. 75: 4. 2019. *Vatten (Journal of Water Management and Research).*

Abbreviations

BV	Bed volume
CAS	Conventional activated sludge
DO	Dissolved oxygen
DOC	Dissolved organic carbon
EBCT	Empty bed contact time
GAC	Granular activated carbon
LRV	Log removal value
LOQ	Limit of quantification
MDG	Millennium development goal
MF	Microfiltration
MBR	Membrane bioreactor
nd	Not detected
PAC	Powdered activated carbon
PAHs	Polycyclic aromatic hydrocarbons
PFASs	Per- and polyfluoroalkyl substances
PE	Population equivalents
RO	Reverse osmosis
SDG	Sustainable development goal
TCC	Total cell concentration
TCL	Total cell load
THMs	Trihalomethanes
UF	Ultrafiltration
UV	Ultraviolet (for example, ultraviolet radiation)
UVT	UV transmittance
WTP	Water treatment plant
WWTP	Wastewater treatment plant

1 Introduction

Humans are and always have been dependent on their surrounding environment and ecosystems, requiring air for breathing, insects and soil for agriculture, and water for drinking. Human activities can impact these systems, sometimes deteriorating the essentials for human life. Such historic examples include deforestation, for example, on the island Rapa Nui (Easter Island) (Hunt and Lipo, 2009; Mann et al., 2008), soil salinization from irrigation, as in Mesopotamia (Jacobsen and Adams, 1958; Zaman et al., 2018), and contamination of drinking water, contributing to disease outbreaks, including the cholera outbreak in London in 1854 (Walford, 2020). Early water and wastewater treatment aimed to prevent such outbreaks, whereas today, wastewater treatment is directed primarily toward environmental protection in terms of prevention of eutrophication and oxygen deficiency.

The removal of organic micropollutants, such as pharmaceuticals, is also gaining attention, notably in the European Union (EU) proposal for a new Urban Waste Water Treatment Directive (2022/0345/COD). The current suggestion is that WWTPs treating a load equal to or greater than 150 000 population equivalents (PE) will be mandated to remove some of these substances (Council of the European Union, 2024). New treatment processes, such as activated carbon or advanced oxidation processes, will thus be necessary.

In parallel, more regions in Sweden have been experiencing droughts and water shortages, and the trend toward more advanced wastewater treatment, in light of depleting freshwater resources, has increased the interest in wastewater reuse in Sweden, thus guiding the theme of this thesis. The issues of how the aquatic environment should be protected through wastewater treatment and how droughts and water shortages must be managed are also covered by the sustainable development goals (SDGs) in the 2030 Agenda that was adopted by the UN General Assembly in 2015.

1.1 Sustainable development goals

The SDGs combine social, economic, and ecological sustainability into a single agenda. They cover various aspects of ecological sustainability, such as protection of the environment on land and below water, climate change, consumption and production, and drinking water.

The different SDGs on environmental sustainability are entangled. Pollution of the aquatic environment, for example, has necessitated the treatment of our wastewater to protect freshwater resources, whereas climate change impacts precipitation patterns and thus, together with an increased population and water use, our freshwater resources (Ungureanu et al., 2020).

1.2 Wastewater reuse

To protect aquatic environments and freshwater resources from harmful substances, such as organic micropollutants, wastewater need to be treated. Wastewater treatment processes to remove organic micropollutants, including granular activated carbon (GAC) and ozonation, have been implemented at a number of WWTPs in southern Sweden. Together with water scarcity—for example, when tanker trucks had to shuttle drinking water to the town of Kivik during the summer of 2021 (Vodopija Stark, 2021)—these implementations have increased the interest in using the treated wastewater for irrigation, in industries, or as source water for drinking water production.

Using treated wastewater instead of lake water and groundwater is associated with benefits, such as a decreased pressure on freshwater bodies, and positive effects for industries, agriculture, and society in general, which otherwise would be constrained by a lack of freshwater (Silva, 2023). However, domestic wastewater contains chemicals, such as organic micropollutants (for example, pharmaceuticals) (Fatta-Kassinos et al., 2011; Verlicchi et al., 2023) and salts (Muyen et al., 2011), and pathogens, including, for example, bacteria, parasites, and viruses (Jaramillo and Restrepo, 2017), posing a potential risk to human health, the soil, or the irrigated plants (Yalin et al., 2023).

To mitigate the chemical and microbial risks from compounds in the wastewater, treatment processes for wastewater reuse often include microfiltration (MF) or ultrafiltration (UF) membranes, followed by reverse osmosis (RO) and disinfection with, for example, ultraviolet (UV) radiation (Drewes and Horstmeyer, 2016; Jeffrey et al., 2022; Rattier et al., 2012). Since it can be costly to treat water with RO, due to the energy demand and the management of the concentrate (brine) (Giammar et al., 2022; Kehrein et al., 2021), other process combinations, such as

GAC filtration, should be evaluated for reuse purposes (Hogard et al., 2021; Rattier et al., 2012).

GAC primarily removes chemical contaminants through adsorption, but the removal is also affected by the biofilm that develops over time on the filter media (Gibert et al., 2013; Weber et al., 1978). This biofilm can decrease the adsorptive capacity of the GAC (Stewart, 2003) but can also contribute to the removal of some organic micropollutants through biological degradation (Altmann et al., 2016; Betsholtz et al., 2021). Various factors, such as upstream treatment processes (Torresi et al., 2018) and the type of GAC media (Vignola et al., 2018), can influence the composition and function of the biofilm. Further, influent oxygen concentration and nutrient and organic carbon loads can affect the microbial community in biological wastewater treatment processes (Chen et al., 2017) and thus likely the biofilm in GAC filters.

Due to the many factors that affect the function of GAC filters and to the temporal development of the biofilm and the decrease in adsorptive capacity, long-term studies of full-scale GAC filters are needed to generate a comprehensive description and derive a profound understanding of their function.

Apart from the temporal variations in GAC function and other technological aspects, the implementation of wastewater reuse is a matter of attitudes and perceptions. Wastewater reuse is subject to many varying opinions and can cause controversies, but does not necessarily do so. An understanding of the perceptions of wastewater reuse is thus necessary to understand how and where wastewater reuse can be applied.

1.3 Aim

The aim of the work in this thesis was to examine the effects of wastewater treatment by GAC filtration on effluent microbial and chemical water quality and evaluate their implications for wastewater reuse. Wastewater reuse is subject to many perceptions that can affect its implementation and thus the design of water and wastewater infrastructure. The aim was therefore further to analyze the discourses of wastewater reuse generally and in Sweden, compared with desalination, which is also an alternative method that can be applied when groundwater and lake water resources are insufficient to meet societal water demands. The work was structured into two parts, with associated objectives.

Part 1: Wastewater reuse with granular activated carbon

- To study temporal variations in the removal of chemical and microbial contaminants by a full-scale GAC filter.
- To assess the potential to reuse municipal wastewater—treated with a membrane bioreactor (MBR) followed by GAC filtration and UV disinfection or with a conventional activated sludge (CAS) process followed by sand filtration, GAC filtration and UV disinfection—for irrigation and drinking water production.
- To investigate the biological degradation of pharmaceuticals in GAC filters and factors that may affect it, including operation time, oxygen concentration and filter material.
- To study the removal of microbial contaminants by GAC filters with different upstream treatment processes.

Part 2: Discourses of wastewater reuse

- To analyze the discourses of wastewater reuse and desalination in Sweden and identify factors that affected these.

1.4 Outline of dissertation

The thesis is based on an interdisciplinary collection of papers on various aspects of wastewater reuse. Chapter 2 in the thesis gives a background to relevant aspects of wastewater reuse. Chapter 3 gives historical and social perspectives on contemporary wastewater treatment and sustainability initiatives. In Chapter 4, the methods used in the thesis are described. The effects of GAC filtration on microbial and chemical water quality were investigated, and the potential to reuse the treated wastewater was evaluated, the results of which are discussed in Chapter 5 (Paper I, II and III). Paper IV comprises an analysis of the discourse that surrounds wastewater reuse compared with other alternative raw water sources; the results are discussed in Chapter 6. The main findings are summarized in Chapter 7, and ideas for future research are discussed in Chapter 8.

2 Wastewater reuse

Wastewater reuse is the reuse of, usually, treated wastewater for, for example, irrigation, industrial use, and direct or indirect drinking water production (referred to as direct potable reuse, or indirect potable reuse) (Figure 1). Direct potable reuse is the treatment of wastewater in several steps and subsequently its direct delivery to the drinking water distribution system, whereas in indirect potable reuse, after treatment, the wastewater is passed through a natural system that allow for retention of the water, such as an aquifer or a lake, before it is treated again and used as drinking water. Wastewater is often released to rivers or lakes that are also used as source waters for drinking water production, corresponding to unplanned reuse, which can be referred to as *de-facto* reuse.

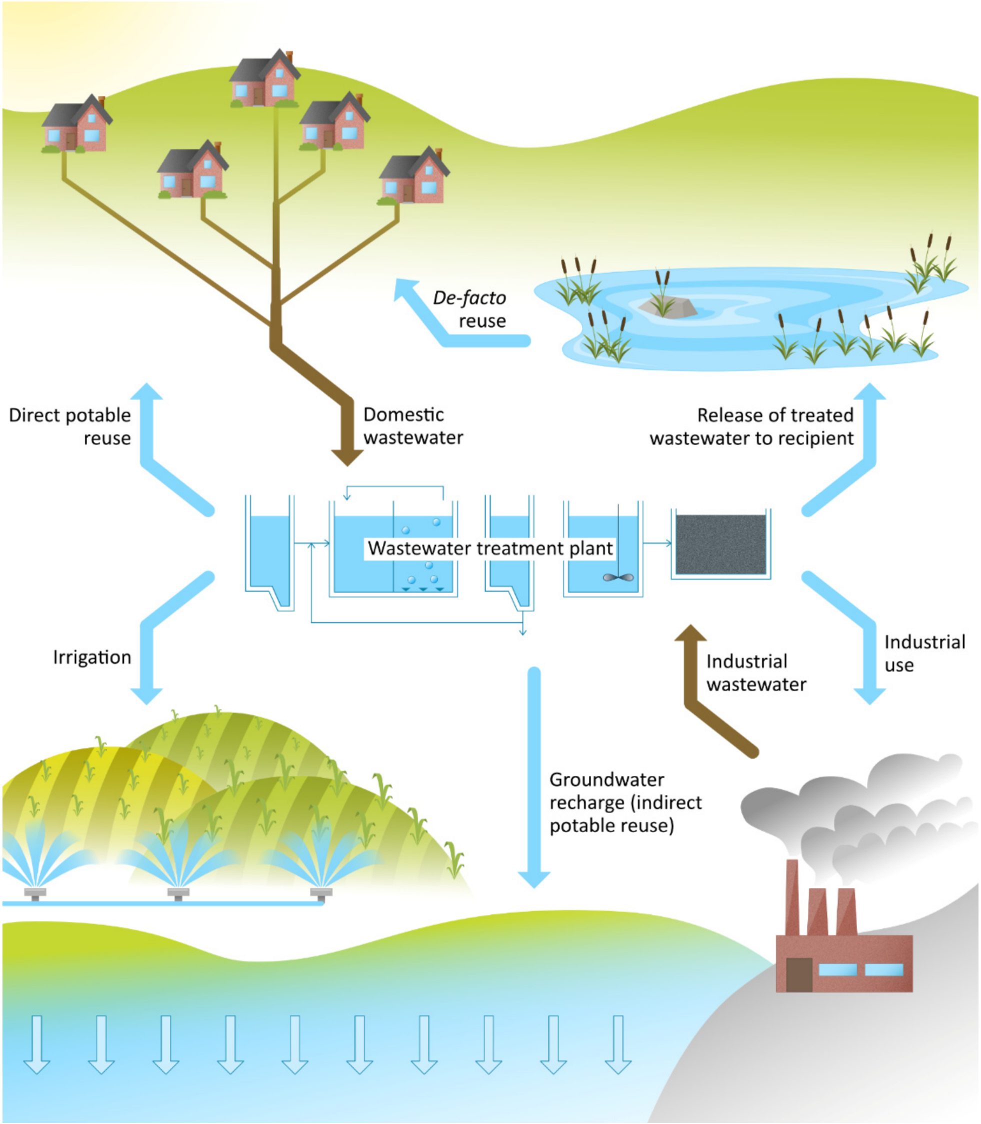

Figure 1. Schematic of various types of wastewater reuse.

2.1 Water quality

Wastewater may contain microbial and chemical contaminants that can be harmful to human health—for example, if they are consumed with drinking water or with crops that have been irrigated with treated wastewater—or have negative impacts on irrigated plants or soil. To ensure safe reuse, they must be removed to safe concentrations.

Microbial contaminants include bacteria, viruses, parasites (Drewes and Horstmeyer, 2016; Kristanti et al., 2022), fungi (Becerra-Castro et al., 2015), and antibiotic resistance genes (Lai et al., 2021) and can negatively affect human health as well as the soil properties (Jaramillo and Restrepo, 2017). Chemical contaminants can be organic, such as pharmaceuticals, or inorganic—for example, heavy metals. Organic contaminants that occur at concentrations on the µg/L or ng/L level can be referred to as organic micropollutants and include, for example, pharmaceuticals, per- and polyfluoroalkyl substances (PFASs), and pesticides.

Concern has been raised over a potential connection between environmental exposure to pharmaceutical residues and adverse health effects (Miarov et al., 2020), and exposure to PFASs has been linked to negative reproductive effects, negative effects on the immune system, and certain types of cancer (US EPA, 2024). Consumption of water that contains antimicrobial residues can have negative health consequences, and irrigation with water that contains these substances can harm irrigated plants (Adeel et al., 2017; Janeczko and Skoczowski, 2005; Treiber and Beranek-Knauer, 2021; Yalin et al., 2023). Plant uptake of PFASs (Ghisi et al., 2019) and some pharmaceuticals (Mordechay et al., 2022, 2021; Wu et al., 2014) has also been reported—to varying extents, depending on the plant, compound, and concentration in the irrigation water. Some pharmaceuticals or their metabolites, such as lamotrigine, ciprofloxacin, and 10,11-epoxycarbamazepine, have been found in edible plants at levels that warrant further investigation (Malchi et al., 2014; Riemenschneider et al., 2016).

Some of the Swedish municipalities that might have an interest in wastewater reuse, due to water shortage, are small and thus will be exempt from implementing advanced treatments in the proposed EU directive (2022/0345/COD), given the size of their WWTPs. Consequently, the removal of organic micropollutants is not ensured at WWTPs where the water might be reused, suggesting a need for further regulations on contaminants in reused wastewater.

2.1.1 Evaluation of water quality

Sweden currently lacks detailed regulations on wastewater reuse, as discussed by Johansson et al. (2022). Since 2020, EU Regulation 2020/741 of the European Parliament and of the Council of 25 May 2020 on minimum requirements for water reuse has governed irrigation with reused wastewater, wherein microbial risks are managed through limits on the indicator bacteria *E. coli*, for example. The water quality criteria are divided into four quality classes—A, B, C, and D—depending on what is irrigated and how the irrigation is performed (Table 1 and Table 2). Management of chemical risks from, for example, organic micropollutants and heavy metals are not detailed, but other countries sometimes have more guidelines on such compounds. US EPA guidelines for water reuse (2012) define (among other parameters) limits on heavy metal concentrations in wastewater that is reused for irrigation, and can be used to complement local regulations (Table 3).

Regulations from the Swedish Food Agency on the quality of drinking water can be used as a reference to evaluate the potential for potable reuse, although future legislation on potable reuse could include additional substances, including certain pharmaceuticals, that can occur at higher concentrations in wastewater compared with lake water and groundwater (Table 2). Organic micropollutants are included in the Australian guidelines for water recycling (2008), and Reungoat et al. (2012) (a technical report that complements the Australian guidelines), both of which concern indirect potable reuse through the augmentation of surface water or groundwater sources (Table 4).

Few countries include limits on pharmaceuticals in their legislation on wastewater reuse or drinking water production, despite research that reports uptake of such compounds by plants. Due to the lack of restrictions on these compounds, measurements described in the literature of concentrations in lakes that are used as source waters for drinking water production and as recipients for treated wastewater were used as complementary references. The comparison provides information on concentrations that we currently accept in our source waters for drinking water, although they are not based on a risk evaluation nor constitutes legally binding limits. Here, a comparison has been performed against concentrations in the lakes Vänern, Vättern, Mälaren (Malnes et al., 2020), and Ringsjön (Svahn and Björklund, 2017).

Table 1. Quality classes for irrigation with reused water per Regulation (EU) 2020/741.

Minimum reclaimed water quality class	Crop category	Irrigation method
A	All food crops consumed raw where the edible part is in direct contact with reclaimed water and root crops consumed raw.	All irrigation methods.
B	Food crops consumed raw where the edible part is produced above ground and is not in direct contact with reclaimed water, processed food crops and non-food crops including crops used to feed milk- or meat-producing animals.	All irrigation methods.
C	Food crops consumed raw where the edible part is produced above ground and is not in direct contact with reclaimed water, processed food crops and non-food crops including crops used to feed milk- or meat-producing animals.	Drip irrigation or other irrigation method that avoids direct contact with the edible part of the crop.
D	Industrial, energy and seeded crops.	All irrigation methods.

Table 2. Summary of a selection of legislation and guidelines on microbial and physical parameters. The summary is based on the parameters that were analyzed in this thesis. More parameters are included in the legislation and guidelines but are not listed in the table.

Microbial and physical parameters							
	Drinking water, limits, and indicator parameters (Sweden, LIVSFS 2022:12)			Irrigation, Regulation (EU) 2020/741 on minimum requirements for water reuse (EU, 2020/741)			
	Finished drinking water	*Drinking water at the user*	*Bottled drinking water*	*Class A*	*Class B*	*Class C*	*Class D*
E. coli	Present in 100 mL	Present in 100 mL	Present in 100 mL	≤10 cfu/100 mL	≤100 cfu/100 mL	≤1000 cfu/100 mL	≤10 000 cfu/100 mL
Total coliforms	Present in 100 mL	Present in 100 mL	Present in 100 mL				
Intestinal enterococci	Present in 100 mL	Present in 100 mL	Present in 100 mL				
Actinomycete		100 cfu/100 mL					
Clostridium perfringens		Present in 100 mL					
Slow-growing bacteria		No unusual changes*					
Culturable microorganisms at 22°C	No unusual changes*	No unusual changes*	No unusual changes*				
Microfungi		100 cfu/100 mL					
TSS				≤10 mg/L			
Turbidity	0.5 NTU	1.5 NTU	1.5 NTU	≤5 NTU			

*Translation from Swedish (“Ingen onormal förändring”)

Table 3. Summary of a selection of legislation and guidelines on metals and other inorganic trace contaminants. The summary is based on the parameters that were analyzed in this thesis. More parameters are included in the legislation and guidelines but are not listed in the table.

Metals and other inorganic trace contaminants			
	Drinking water, limits, and indicator parameters (Sweden, LIVSFS 2022:12)		US EPA (2012)
	Finished drinking water	*Drinking water at the user and bottled drinking water*	*Irrigation*
Aluminum		200 µg/L	5.0 mg/L
Ammonium		0.50 mg NH_4/L	
Antimony		10 µg/L	
Arsenic		5.0 µg/L	0.10 mg/L
Beryllium			0.10 mg/L
Boron		1.5 mg/L	0.75 mg/L
Cadmium		0.50 µg/L	0.01 mg/L
Chromium		25 µg/L	0.1 mg/L
Cobalt			0.05 mg/L
Copper		2.0 mg/L	0.2 mg/L
Iron	100 µg/L	200 µg/L	5.0 mg/L
Lead		5.0 µg/L	5.0 mg/L
Lithium			2.5 mg/L
Magnesium		30 mg/L	
Manganese		50 µg/L	0.2 mg/L
Molybdenum			0.01 mg/L
Nickel		20 µg/L	0.2 mg/L
Nitrate		50 mg NO_3^-/L	
Nitrite	0.10 mg NO_2 /L	0.50 mg NO_2 /L	
Quicksilver		1.0 µg/L	
Selenium		20 µg/L	0.02 mg/L
Sodium		200 mg/L	
Vanadium			0.1 mg/L
Zinc			2.0 mg/L

Table 4. A summary of limits and guideline values from the Australian guidelines for water recycling, 2008 (recommended drinking water guideline values); Reungoat et al., 2012 (recommended drinking water guideline values); and Swedish drinking water legislation (limit values), with average concentrations that have been measured in the Swedish lakes of Vänern, Vättern, Mälaren (Malnes et al., 2020) [1], and Ringsjön (Svahn and Björklund, 2017) [2]. This summary is based on the parameters that were analyzed in this thesis. More parameters are included in these documents than in the table. nd, not detected.

Organic micropollutants				
	Augmentation of drinking water supplies (µg/L)		Finished drinking water (µg/L)	Lake water (ng/L)
	Australian guidelines for water recycling (2008)	*Reungoat et al. (2012)*	*Swedish drinking water legislation (LIVSFS 2022:12)*	*Average (max) concentrations in Swedish lakes*
17α-ethinylestradiol	0.0015			<8.6 (<8.6)^{1}
17α-estradiol	0.175			
17β-estradiol	0.175			2.8 (3.9)^{1}
Atenolol		25		1.6 (4)^{1}
Azithromycin	3.9			2.18 (3.6)^{1}
Bisphenol A			2.5	
Carbamazepine	100			5.4 (22)^{1}, 12.4^{2}
Ciprofloxacin	250			<10 (<10)^{1}
Citalopram		10		0.65 (4.2)^{1}
Clarithromycin	250			0.68 (1.2)^{1}, 0.25^{2}
Diclofenac	1.8			6.3 (23)^{1}, 1.5^{2}
Erythromycin	17.5			3.5 (12)^{1}, 0.5^{2}
Estrone	0.03			0.6^{2}
Fluconazole				2.9 (15)^{1}, 1.2^{1}
Furosemide		10		<17 (<17)^{1}
Hydrochlorothiazide		12.5		17 (61)^{1}
Ibuprofen	400			<8.25 (<8.25)^{1}, nd^{2}
Irbesartan				0.85 (2.1)^{1}
Ketoconazole				nd^{2}
Losartan				6.9 (29)^{1}, 1.7^{2}
Metoprolol	25			4.1 (32)^{1}, 3.9^{2}
Naproxen	220			3.1^{2}

Oxazepam		7.5		2.7 (10)[1], 4.7[2]
Paracetamol	175			<5.4 (<5.4)[1]
PFAS 4 (sum of PFOS, PFOA, PFNA, PFHxS)			4.0 ng/L	
PFAS 21 (sum of 21 PFAS substances)			100 ng/L	
Propranolol	40			0.79 (1.6)[1]
Sertraline		25		<1.1 (<1.1)[1], nd[2]
Sulfamethoxazole	35			3.0 (12)[1], 3.4[2]
Tramadol		50		6.1 (59)[1], 1.2[2]
Trimethoprim	70			0.55 (2.8)[1], 0.6[2]
Venlafaxine		37.5		16 (43)[1]
Zolpidem				nd[2]

Studies have reported an uptake, to levels that warrant further investigation, of some pharmaceuticals or their metabolites in plants that have been irrigated with wastewater (Malchi et al., 2014; Riemenschneider et al., 2016). The concentrations of these compounds in the wastewater that was used for the irrigation was, for example, 300 ng/L of ciprofloxacin (Riemenschneider et al., 2016), and 20–1700 ng/L of carbamazepine (Malchi et al., 2014; Riemenschneider et al., 2016). The limits in the Australian guidelines for water recycling (2008) by far exceed these levels (100 000 ng/L of carbamazepine, 250 000 ng/L of ciprofloxacin), raising questions on what constitutes safe levels in drinking water and irrigation water.

2.2 Treatment processes

To remove unwanted compounds from the water, treatment processes for water reuse in many cases include RO with MF or UF as pre-treatment, commonly followed by disinfection with, for example, UV radiation (Drewes and Horstmeyer, 2016). Due to their energy demands and the management of the brine residue, treatment processes that include RO can be costly, prompting an interest in advanced wastewater treatment without RO, such as combinations of MBR, advanced oxidation processes, UV disinfection, and GAC filtration (Giammar et al., 2022; Hogard et al., 2021; Kehrein et al., 2021; Rizzo et al., 2019).

This thesis focuses on the treatment process GAC filtration, implemented for the removal of organic micropollutants from wastewater. Three full-scale WWTPs with GAC filtration and one pilot-scale GAC filter were studied. UV disinfection was examined in laboratory experiments. An overview of GAC filtration and UV disinfection is provided below.

2.2.1 Granular activated carbon filtration

Activated carbon has a high surface area and high porosity and removes organic matter and contaminants through adsorption. It can be in the shape of powder (powdered activated carbon, PAC) or granules (granular activated carbon, GAC). PAC has a smaller particle size compared with GAC and can be dosed continuously to the water at various locations in the treatment process, whereas GAC is operated as a filter that consists of a bed of granules, through which the water flows. If the PAC is removed with the sludge, it pollutes the sludge with the contaminants that it has adsorbed. This property renders PAC a viable option if the sludge is incinerated but not if it is to be used as fertilizer. Consequently, GAC—not PAC—is currently more common in Sweden.

The costs for GAC filtration are generally somewhat higher than for ozonation and PAC (Pistocchi et al., 2022) but depend on many factors, such as the size of the

WWTP, influent dissolved organic carbon (DOC) concentrations, and regeneration intervals (Pistocchi et al., 2022; Rizzo et al., 2019; Tarpani and Azapagic, 2018).

Activated carbon can be produced from renewable sources, such as coconut, and non-renewable sources, including coal, and can be activated physically or chemically (Heidarinejad et al., 2020). The quality and properties of the activated carbon can vary, depending on carbon source and activation method.

During its service life, the capacity and rate of adsorption to GAC decrease, due to blockage of pores and adsorption sites from previously adsorbed organic carbon and particles (Corwin and Summers, 2010; Meinel et al., 2015; Miguel et al., 2010). The biofilm that develops over time on the GAC granules adds to the decline in adsorption (Stewart, 2003), but can possibly also contribute to the removal through the biological degradation of some organic compounds and micropollutants (Altmann et al., 2016; Betsholtz et al., 2021; Edefell et al., 2022) or through biological regeneration of the GAC (El Gamal et al., 2018).

In addition, the total number of bacteria in the water can be increased by GAC filtration (Miller et al., 2020; Whitton et al., 2018) and the bacterial community can be shifted (Kantor et al., 2019; Miller et al., 2020; Piras et al., 2022; Stewart et al., 1990; Vignola et al., 2018). Concerns have been raised over the potential release of opportunistic bacterial pathogens (Kantor et al., 2019; Miller et al., 2020; Vignola et al., 2018; Wullings et al., 2011) and the potential accumulation of antibiotic resistance genes in GAC filters (Wan et al., 2021). However, a decrease in the concentrations of indicator bacteria, such as fecal coliform bacteria (El-Zanfaly et al., 1998), *E. coli* (Hijnen et al., 2010; Spit et al., 2022), and enterococci (Spit et al., 2022), has been observed after GAC filtration.

2.2.2 Disinfection by ultraviolet radiation

UV radiation is an efficient method to disinfect drinking water or wastewater, by damaging the microbial DNA. UV radiation refers to electromagnetic radiation with a wavelength between 100 and 400 nm and is not visible to the human eye. The part of the spectrum that is efficient for disinfection is called the germicidal range (approximately 200–300 nm)—wavelengths under 200 nm are absorbed by the water, and those greater than 300 nm are not absorbed by the DNA and thus do not damage it (Crittenden et al., 2012). The disinfection efficiency depends on the UV transmittance (UVT) in the water, wherein a low transmittance will decrease the effect of the UV radiation (Crittenden et al., 2012).

UV and other light-based disinfection methods do not generate harmful disinfection by-products and do not result in a residual disinfection that, for example, prevents growth in the distribution system (as chlorination does) (Wang et al., 2021).

3 Wastewater and sustainability

Irrigation with reused wastewater traces back to the Bronze Age and was performed by, for example, the Minoan and Indus civilizations, and later in the Roman Empire (Angelakis et al., 2018).

In addition to managing wastewater for irrigation, human settlements had to handle excreta to mitigate the risk of disease spreading (Lofrano and Brown, 2010; Vuorinen et al., 2007). The first civilization with evidence of management of sanitation problems was the Mesopotamian Empire, and the culture in Indus Valley developed systems to manage wastewater, as did the Greek and Roman Empires (Angelakis et al., 2023; Lofrano and Brown, 2010; Vuorinen et al., 2007). Yet, the open and combined wastewater and stormwater networks in ancient Greece caused waterborne diseases like cholera to spread (Prochaska and Zouboulis, 2020), and waterborne disease was a common cause of death in ancient Rome (Vuorinen et al., 2007).

In medieval times, after the fall of the Roman Empire, the water supply systems in Europe became more primitive—for example with the use of cesspits and the practice to throw excreta from the window onto the street—increasing the risk of disease spreading and epidemics (Angelakis et al., 2023; Vuorinen et al., 2007).

3.1 Industrialization and contemporary wastewater treatment

During industrialization, scientific progress was made in terms of understanding diseases, such as cholera, and how they spread with water (Bleakley et al., 2018; Lofrano and Brown, 2010). Filtration and chlorination of drinking water were introduced during the 19th and 20th centuries (Bleakley et al., 2018; Vuorinen et al., 2007). The water closet was established, increasing water use and the need for urban wastewater management. The new wastewater systems that developed in European, US, and Australian cities during the 19th and early 20th centuries often consisted of the collection of wastewater in sewer systems and its release to a water body, such as a river or lake, causing environmental pollution (Bleakley et al., 2018; Katko et al., 2022; Lofrano and Brown, 2010; Radcliffe and Page, 2020). Water pollution was largely understood in terms of pathogens and low dissolved oxygen (DO) concentrations, resulting from emissions of organic carbon, and methods to measure them were developed in the 19th century (Shifrin, 2005).

3.1.1 Wastewater treatment plants

One of the first WWTPs of the modern era in Europe was constructed in Bubeneč, Poland (built in 1900–1906), consisting of sedimentation and removing approximately 40% of the organic matter (Angelakis et al., 2023). In 1913, another milestone for modern wastewater treatment was attained: the results from experiments with a suspended bacterial culture in an aerated system for wastewater treatment were published by Fowler and Mumford, and the discovery of the benefits of recirculation of the suspended solids in the system was reported by Ardern and Lockett in 1914 (Alleman and Prakasam, 1983). These findings formed the basis for the activated sludge process, commonly used today in centralized wastewater treatment. Soon after, WWTPs that used this technology were constructed in European and US cities (Alleman and Prakasam, 1983), and standards for design and water quality developed, especially during the second half of the 20th century (Shifrin, 2005).

During the 20th century, the development in analytical methods, such as gas chromatography and atomic adsorption spectrophotometry, advanced our understanding of environmental pollution (Lofrano and Brown, 2010; Shifrin, 2005). One such type of pollution stems from organic micropollutants, a group consisting of organic compounds, including pharmaceuticals and PFASs. Full-scale treatment to remove these from wastewater has been implemented in a few countries, such as Switzerland (Bourgin et al., 2018), Germany (Neef et al., 2022) and Sweden (Svahn and Borg, 2024).

The technological developments in water treatment and analysis has made it possible to safely reuse wastewater for purposes other than irrigation, such as drinking water, referred to as potable reuse. The first potable reuse plant was commissioned in 1968 in Windhoek, Namibia, and direct or indirect potable reuse is now conducted in the US and in Singapore, among other areas (Angelakis et al., 2018; Du Pisani and Menge, 2013).

Despite the options for advanced wastewater treatment and water quality analysis, 3.4 billion people lacked safe sanitation in 2022, and 2.2 billion people had no access to safe drinking water services (United Nations, 2024). Providing everyone with safe drinking water and sanitation is connected to economic and social development as well as environmental protection, and is addressed in the United Nations 2030 Agenda SDGs. Wastewater treatment and access to drinking water relate to, for example, SDG 6 (clean water and sanitation), SDG 13 (climate action), and SDG 14 (life below water), and wastewater reuse is listed among the targets in SDG 6.

3.2 Sustainable development

The 17 SDGs in the 2030 Agenda have been influenced by many years of development and sustainability initiatives. After the Second World War, a consensus emerged in the western world to promote peace and international cooperation and to aid social progress and increased living standards in industrially less developed countries (Birnie, 1995; Purvis et al., 2019). This discussion initially focused on economic development, whereas environmental issues were incorporated in the development agenda during the 1960s and 1970s, aided by books such as Silent Spring by Rachel Carson (1962), which increased general awareness of the effects of human activities on the environment (Purvis et al., 2019).

The United Nations Conference on the Human Environment in Stockholm in 1972 was the first global meeting to focus on global environmental issues (Birnie, 1995; Paglia, 2021). At this conference, the Stockholm Declaration (United Nations Environment Programme, 1972) was adopted, stating that mankind must protect the environment, while building peace and economic and social development. These endeavors are similar to sustainable development, although they were not yet referred to as such. The term “sustainable development” was described eight years later, in the World Conservation Strategy – Living resource conservation for sustainable development (1980), as the conservation of living resources, and later in the Report of the World Commission on Environment and Development: Our Common Future (1987) (also known as the Brundtland report), as *“a development that guarantees meeting the needs of the current generation without reducing the ability of future generations to meet their own needs.”* Five years later, in 1992, the first Earth Summit, the United Nations Conference on Environment and Development, was held in Rio de Janeiro, resulting in Agenda 21, a document that describes a path for work toward sustainable development in the 21st century (Clémençon, 2012; Francioni, 2016).

3.2.1 A new millennium

At the start of the new millennia, the Millennium Summit was held (2000) in New York, resulting in the United Nations Millennium Report and the United Nations Millennium Declaration, the latter from which eight Millennium Development Goals (MDGs) were formed. These MDGs were aimed at decreasing poverty, promoting health and equality, and ensuring environmental sustainability and global partnership, with a target date in 2015. Two years after the adoption of the MDGs, the next global meeting on sustainable development took place: the World Summit on Sustainable Development (WSSD) in Johannesburg in 2002 (La Viña et al., 2003). The outcome of this conference was the Plan of Implementation and the Johannesburg Declaration on Sustainable Development, the goal of which was to improve implementation of the previous commitments in Agenda 21 and the MDGs (La Viña et al., 2003).

3.2.2 The 2030 Agenda

As the target date for the MDGs approached, the SDGs in the 2030 Agenda, after lengthy negotiations, emerged on the Rio +20 conference in Rio de Janeiro in 2012 and were passed by the UN General Assembly in 2015 (Palmer, 2015). The 2030 Agenda updated the MDGs and combined social and environmental sustainability into one agenda, and the negotiations for it were longer and more transparent and involved more actors than previous negotiations (Stevens and Kanie, 2016). Whereas the MDGs were directed primarily toward poverty, maternal health, and primary school—goals that were already largely reached by many countries in the global north—the SDGs covered matters that concern the entire world (Hajian and Kashani, 2021). In the MDGs, environmental issues were condensed into a single goal (Goal 8, Ensure environmental sustainability), whereas several SDGs cover various aspects of environmental sustainability, including energy (SDG 7), consumption and production (SDG 12), climate change (SDG 13), aquatic ecosystems (SDG 14), ecosystems on land (SDG 15), and drinking water (SDG 6). In the 2030 Agenda, it is also acknowledged that these areas are related. Systems for drinking water production and distribution (SDG 6) are affected by, for example, climate change and pollution, in turn potentially affecting industry, businesses, and welfare in general.

3.3 Water discourses

Societal systems for, for example, water production and distribution, wastewater management, energy, and transportation can be referred to as socio-technical systems, the changes in which can be referred to as socio-technical transitions (Markard et al., 2012). Since such systems are related to, and affect, several aspects of sustainability and society, they are subject to many opinions and perceptions. The different viewpoints among politicians, public officials, the public, and other actors can be understood in terms of discourses, and can impact the choice and design of the infrastructure, and thus society and the environment. According to Hajer (2006: 67), a discourse is defined as *"an ensemble of ideas, concepts and categories through which meaning is given to social and physical phenomena, and which is produced and reproduces through an identifiable set of practices."* It can comprise ideas for solving a problem, such as water scarcity, and opinions on priorities and problem definitions.

Discourses can be analyzed in terms of their components' storylines (Rosenbloom, 2018; Rosenbloom et al., 2016) or imaginaries (Jasanoff and Kim, 2009; Tidwell and Smith, 2015). Storylines are simplified descriptions of the truth, often drawing on common sense, and are described as *"a condensed statement summarizing complex narratives, used by people as 'short hand' in discussions"* (Hajer, 2006: 69). Imaginaries are visions of a desirable future and are considered *"collectively imagined forms of social life and social order reflected in the design and fulfillment of nation-specific scientific and/or technological projects"* (Jasanoff & Kim, 2009: 120). The effects of discourses on socio-technical transitions constitute a research field in social sciences. Water infrastructure has been subject to this type of research and it has been argued that discourses have affected the design of water infrastructure around the world. Therefore, a discourse analysis was performed on water infrastructure in general and specifically on reused wastewater and desalinated seawater, both of which are alternative source waters when groundwater and surface water resources are insufficient to meet societal water demands (Paper IV).

4 Methods

In this chapter, the methods that were used in the thesis are discussed and described. Four full-scale WWTPs, all located in Scania in southern Sweden and comprising full-scale or pilot-scale GAC filters, were examined: Kivik WWTP (Papers I–III), S:t Olof WWTP (Paper II), Degeberga WWTP (Paper III), and Svedala WWTP (Paper III) (Figure 2). Long-term, full-scale studies were performed to capture phenomena that evolve slowly in GAC filters and that are not possible to fully simulate on a laboratory or pilot scale. Laboratory experiments were conducted when full-scale studies were not possible—to analyze biological degradation in detail using ^{14}C-labeled organic micropollutants, the use of which is restricted due to their radioactivity, and to evaluate UV disinfection, which was not implemented at the studied WWTPs.

Further, an analysis of discourses on water treatment plants (WTPs) that incorporate wastewater reuse and desalination for drinking water production, on Öland and Gotland, was performed through interviews and a media analysis (Paper IV) (Figure 2).

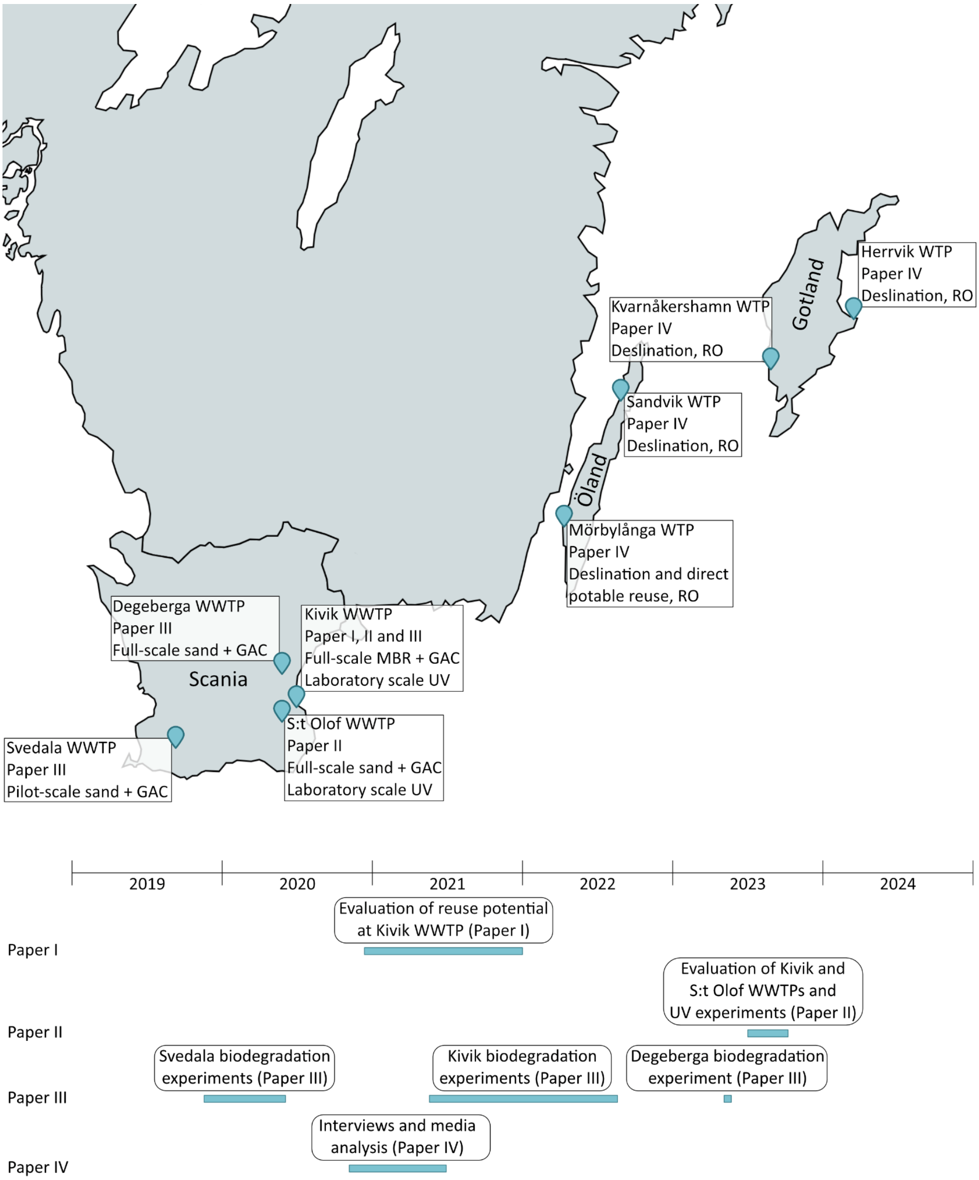

Figure 2. Map with approximate locations of wastewater and water treatment plants in this thesis, with an approximate timeline of the data collection.

4.1 Full-scale wastewater treatment plants

In full-scale WWTPs, many factors simultaneously impact the treatment processes and effluent water quality. Such factors include temperature, and influent DOC and nutrient concentrations, the latter of which are influenced by households and industries that are connected to the WWTP and by infiltration and inflow to the sewer system. At a full-scale WWTP, all of these factors will affect the treatment results but will typically not be accounted for in laboratory experiments. Conversely, a drawback of full-scale studies is that simultaneous variations in several factors prevent conclusions from being drawn on causal relationships. Further, full-scale WWTPs must fulfill legal responsibilities regarding effluent water quality. Thus, the water can not be spiked with contaminants, and the possibilities for altering the process to allow various comparisons to be made are limited.

All of the GAC filters in the thesis have been commissioned without existing demands on organic micropollutant removal, but with an expectation of future demands. All of the studied WWTPs are located in Scania, in southern Sweden, and are surrounded by agricultural land and forest.

The towns of Kivik and S:t Olof lie in Simrishamn municipality, which acquired funds in 2018 from the Swedish Environmental Protection Agency (EPA) to implement wastewater treatment for the removal of organic micropollutants, helping finance reconstruction of the WWTPs in the two towns (Svenskt Vatten, 2024). The area in which they lie has a many visitors and a high density of vacation homes, increasing the population especially in June, July, and August. The larger population during these months increases the load to the WWTPs, potentially affecting the wastewater treatment. Thus, the Kivik WWTP was studied for one full year.

4.1.1 Kivik wastewater treatment plant

Water and GAC media from Kivik WWTP were collected and analyzed throughout one year from the date of commissioning (Paper I). GAC media, collected on four occasions, was also studied in incubation experiments (Paper III), three of which overlapped with the sampling described in Paper I. Water was further collected on three occasions for analysis and UV disinfection experiments (Paper II) (Figure 2).

The reconstructed Kivik WWTP—and, consequently, the Kivik GAC filters—became operational in December 2020 and is dimensioned for a maximum of 7500 PE. The WWTP receives water from the towns of Kivik (~900 inhabitants), Vitaby (~300), Vitemölla (~100), and Södra Mellby (~100). The high dimensioning is due to the rising population during the summer months.

The treatment process comprises mechanical treatment with screens, grease trap, sand trap and disc filters, which is followed by an MBR that consists of an anoxic zone, an aerated zone, and two parallel UF membranes with a pore size of 38 nm (Figure 3 and Figure 4). Downstream of the MBR follows two parallel GAC filters (Jacobi Aquasorb 6100, filter volume 18 m^3 per filter), after which the water is released to the Baltic Sea. Until March 2022, chemical precipitation with an aluminum-based chemical took place upstream of the disc filters, and was then moved to the anoxic and aerated zones in the MBR, i.e. pre-precipitation was exchanged for simultaneous precipitation. The empty bed contact time (EBCT) of the GAC filters during the study period was approximately 60 minutes. The GAC filters were backwashed approximately once per month, starting after approximately one year of operation. The treatment process is detailed in Papers I, II, and III.

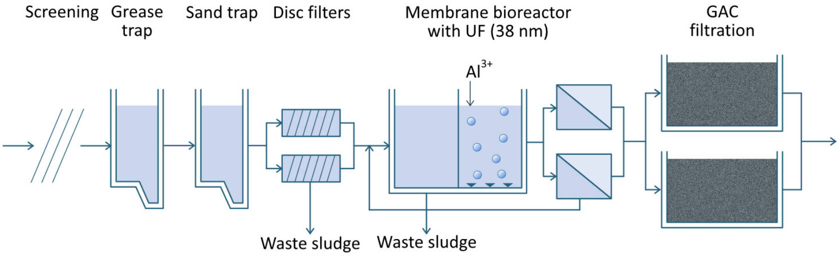

Figure 3. Process scheme for Kivik WWTP.

Figure 4. Photograph of parts of Kivik WWTP, with the GAC filters to the left.

4.1.2 S:t Olof wastewater treatment plant

Water from S:t Olof WWTP was collected on three occasions for analysis and UV disinfection experiments (Paper II) (Figure 2).

The reconstructed S:t Olof WWTP was commissioned in 2021 and is dimensioned for 1000 PE. The treatment comprises mechanical treatment with sand trap and screens, followed by a conventional activated sludge (CAS) process that consists of an aerated zone and sedimentation, followed by chemical post-precipitation with an aluminum-based chemical (Figure 5). The water then passes through two parallel rapid sand filters (filter volume 6.4 m^3 per filter) followed by two parallel GAC filters (Jacobi Aquasorb 6100, filter volume 6.4 m^3 per filter) (Figure 5 and Figure 6), which are backwashed approximately twice per year. The EBCT in the sand and GAC filters was approximately 170 min during the study period. The treatment process is detailed in Paper II. UV experiments were performed also on the effluent from Kivik WWTP, with MBR instead of CAS and sand filtration prior to the GAC filters, allowing for the comparison between GAC filters with different upstream processes.

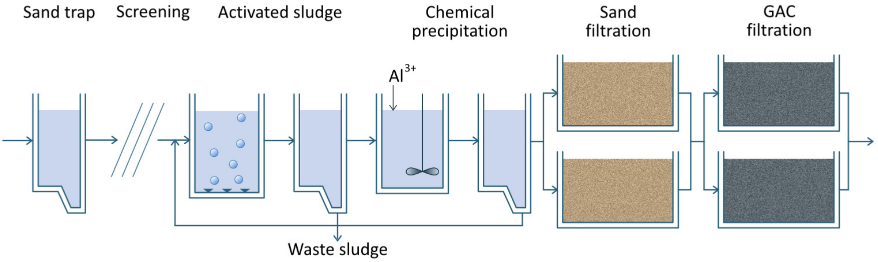

Figure 5. Process scheme for S:t Olof WWTP.

Figure 6. Photograph of GAC filter at S:t Olof WWTP. Photograph by Moa Jinbäck.

4.1.3 Degeberga wastewater treatment plant

GAC and sand media from Degeberga WWTP were collected on one occasion for incubation experiments (Paper III) (Figure 2).

The Degeberga GAC filters were commissioned in April 2020 and were the first full-scale GAC filters to be installed in Sweden for domestic wastewater treatment (Svahn and Borg, 2024). At the time of writing, the GAC filters had never been backwashed.

The treatment process consists of mechanical treatment with screens and sand trap, followed by an activated sludge process with an anoxic zone, an aerated zone, and sedimentation, which is then followed by chemical precipitation, rapid sand filtration, and two parallel GAC filters (Jacobi Aquasorb 5000, bitumen-based GAC, and Jacobi Aquasorb CS, coconut-based GAC; filter volume 5.5 m^3 per filter) (Figure 7 and Figure 8). The EBCT was approximately 50 minutes in the bitumen-based filter and 80 minutes in the coconut-based filter. The treatment process is detailed in Paper III and by Svahn and Borg (2024).

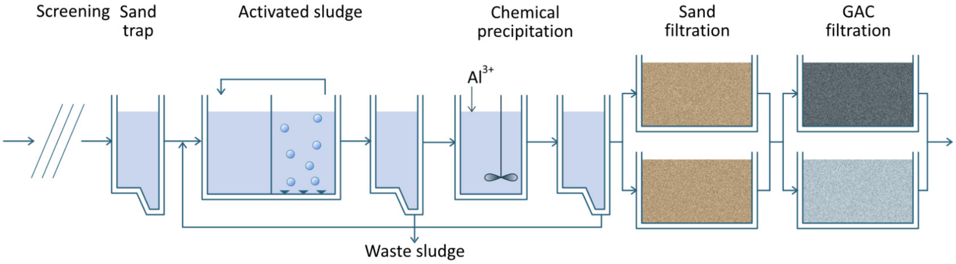

Figure 7. Process scheme for Degeberga WWTP.

Figure 8. Photograph of GAC filters at Degeberga WWTP. Photograph by Ola Svahn.

4.1.4 Svedala wastewater treatment plant

From 2019 to 2020, a pilot plant with sand and GAC filtration was operated at Svedala WWTP. GAC media from the pilot-scale GAC filter was collected on four occasions for incubation experiments (Paper III) (Figure 2). These experiments were performed before I began this study and were compared with the results from the Kivik and Degeberga incubation experiments described in the same paper (Paper III).

Svedala WWTP has a capacity to treat wastewater from 18 500 PE. The treatment process consists of mechanical treatment with screens and sand trap, followed by an activated sludge process that includes an anoxic zone, an aerated zone, and sedimentation, followed by chemical precipitation (Figure 9). A small fraction of the water was passed through a pilot-scale rapid sand filter and GAC filter (Figure 10). The pilot GAC filter (Jacobi Aquasorb 5000, filter volume 0.019 m^3) was operated at low influent oxygen concentrations (DO=0.5–1 mg/L). It was backwashed twice per month, and the EBCT was approximately 10 minutes. The treatment process is detailed in Paper III and by Gidstedt et al. (2022).

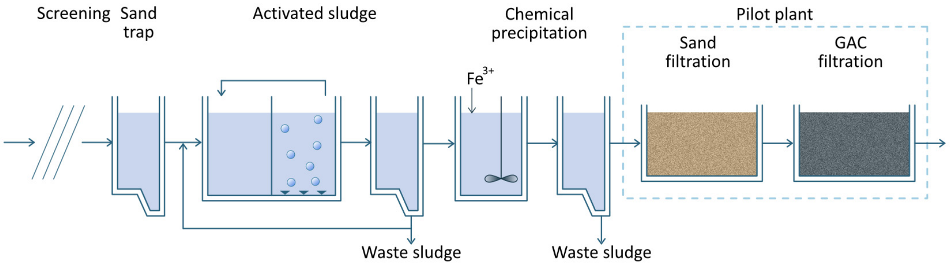

Figure 9. Process scheme for Svedala WWTP, including the pilot plant that was operated during 2019–2020.

Figure 10. Photograph of pilot-scale GAC (left) and sand (right) filter at Svedala WWTP. Photograph by Alexander Betsholtz and Simon Gidstedt.

4.2 Laboratory experiments

UV disinfection was examined in laboratory experiments, as was biological degradation of organic micropollutants. Laboratory experiments are particularly suitable for the detailed study of specific phenomena, such as biological degradation, that require a controlled environment.

Three UV experiments with two wastewater effluents and three UV fluences (Paper II) and in total nine degradation experiments comprising 81 incubations (i.e., bottles with GAC or sand media and spiked wastewater) (Paper III) were performed.

4.2.1 UV disinfection experiments

UV disinfection was performed in laboratory-scale experiments on the effluents from two full-scale WWTPs that were equipped with GAC filtration (Kivik and S:t Olof), with different types of upstream treatment processes (MBR at Kivik WWTP, and CAS and sand filtration at S:t Olof WWTP) (Paper II). The two processes were compared regarding removal of microbial contaminants by the GAC filters and by the UV disinfection. UF removes particles and bacteria to a larger extent than sand filtration, possibly affecting the GAC filtration and the UV disinfection. Microbial parameters in the Swedish drinking water criteria were analyzed, and one of the UV-treated samples from Kivik WWTP was selected for extensive chemical analysis, comprising approximately 100 parameters.

The experiments were conducted with a monochromatic, low-pressure mercury lamp. The UV reactor was placed vertically on the wall with upward flow (Figure 11). Wastewater in a 20-L plastic bucket was forced through the UV reactor with a garden pump (MEEC Tools, 800 W, 53.3 L/min), and the UV fluence was adjusted by altering the flow with a valve. UV irradiance was monitored with a UV sensor that was factory-calibrated to issue an alert if the irradiance was lower than 70% of the maximum, which did not occur during the experiments.

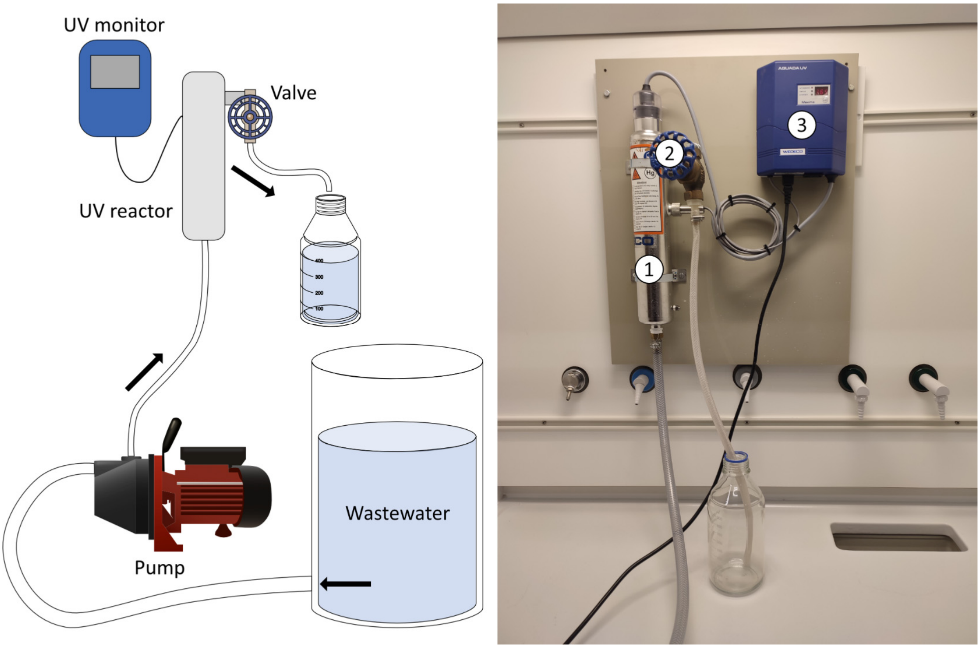

Figure 11. Left: Schematic of the UV setup; right: photograph of the UV equipment, including the UV reactor (1), valve (2), and UV monitor (3).

Three UV fluences (200 J/m^2, 400 J/m^2, and 700 J/m^2) were tested on three occasions (Table 5), of which 400 J/m^2 is a normal fluence for drinking water disinfection in Sweden (Saguti et al., 2022).

Table 5. Experimental parameters for the UV disinfection experiments.

WWTP	**Date**		**200 J/m^2**		**400 J/m^2**		**700 J/m^2**	
		UVT (%)	Flow rate (L/s)	UV fluence (J/m^2)	Flow rate (L/s)	UV fluence (J/m^2)	Flow rate (L/s)	UV fluence (J/m^2)
Kivik	2023-06-12	84	0.346	202	0.156	450	0.100	702
Kivik	2023-06-26	82	0.349	197	0.161	428	0.096	719
Kivik	2023-10-02	85	0.315	225	0.159	447	0.102	699
S:t Olof	2023-06-12	83	0.349	198	0.158	437	0.100	691
S:t Olof	2023-06-26	81	0.348	192	0.154	432	0.094	707
S:t Olof	2023-10-02	81	0.326	204	0.159	419	0.099	675

4.2.2 Biodegradation of organic micropollutants

Since biofilm in GAC filters develops over time (Gibert et al., 2013), biological degradation in the filters is likely affected by operation time. The degradation of organic micropollutants in biological wastewater treatment processes is in general also influenced by oxygen availability (Edefell et al., 2021; Suarez et al., 2010), which thus is likely to affect the degradation also in GAC filters. Therefore, the effects of operation time and oxygen concentration on biological degradation of organic micropollutants were studied by sampling GAC filters that were operated with different oxygen concentrations, after various numbers of bed volumes (BVs). Filter depth and material can also be expected to affect the biofilm and thus the degradation (Benstoem et al., 2017; Moreno-Castilla, 2004; Sauter et al., 2023; Velten et al., 2011; Vignola et al., 2018), prompting the analysis of variations in degradation over depth and between filters comprising different materials. To study differences in biofilm composition between the different time points and filters, 16S rRNA gene sequencing was performed on the GAC and sand samples.

GAC and sand media was collected from three WWTPs: Kivik WWTP (one full-scale GAC filter, referred to as GAC1 in section 5.1, operated with high influent oxygen concentration, bitumen-based), Degeberga WWTP (two full-scale GAC filters, bitumen and coconut based, and one full-scale sand filter, all operated with high influent oxygen concentrations), and Svedala WWTP (one pilot-scale GAC filter, operated with low influent oxygen concentration, bitumen-based). Biological degradation was studied in incubation experiments using ^{14}C-labeled micropollutants and liquid scintillation counting (Figure 12). The treatment processes are described in Section 4.1, and the methods are detailed in Paper III.

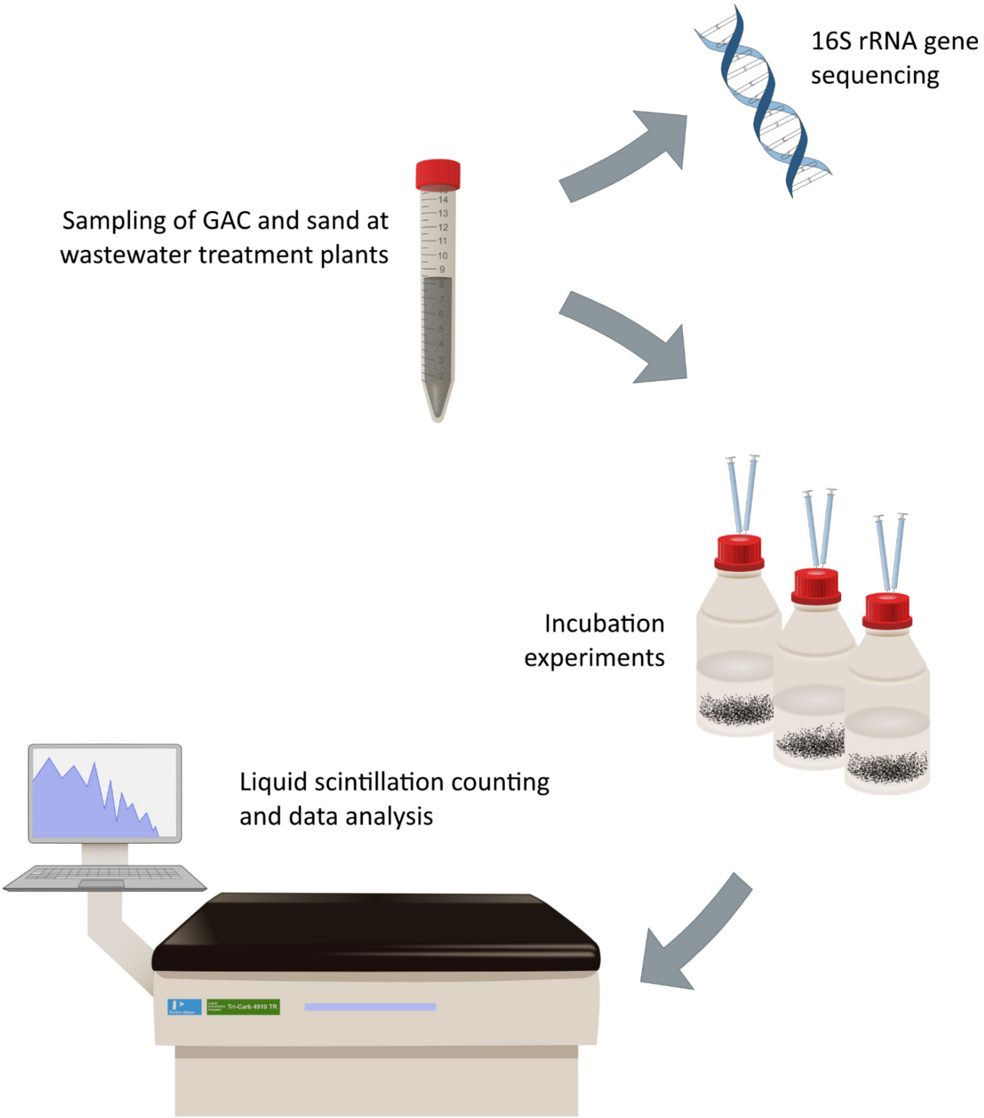

Figure 12. Schematic of the method for examining biological degradation in GAC and sand filters.

Sampling

To study the temporal development of biological degradation, samples of GAC media were collected from the surface layer of the Kivik filter and the Svedala pilot-scale filter after various BVs (Kivik: 6000 BVs, 9000 BVs, 15 000 BVs, 23 000 BVs; Svedala: 6000 BVs, 12 000 BVs, 18 000 BVs, 27 000 BVs). From the two GAC filters at Degeberga WWTP, samples were collected from different depths (surface: 0–10 cm, middle: 40–50 cm, bottom: 90–100 cm) on one occasion—after

29 000 BV (bitumen-based GAC) respective 19 000 BV (coconut-based GAC). Sampling from different depths is challenging in full-scale filters that are not designed with this option, and a sampler was constructed from a metal pipe with a plug to trap the GAC from different depths. Media from the sand filter (in operation since 1975) at Degeberga WWTP was collected from the surface layer on the same occasion as the GAC media.

Experimental setup

The experiments were conducted in glass bottles that contained treated wastewater, GAC or sand media, and ^{14}C-labeled organic micropollutants (Figure 13). The compounds were studied in separate bottles. A glass vial, with 20 mL 1 M NaOH, was placed in each bottle to capture the $^{14}CO_2$ that formed from degradation. Control incubations, with GAC media that was sterilized by heat treatment, were included.

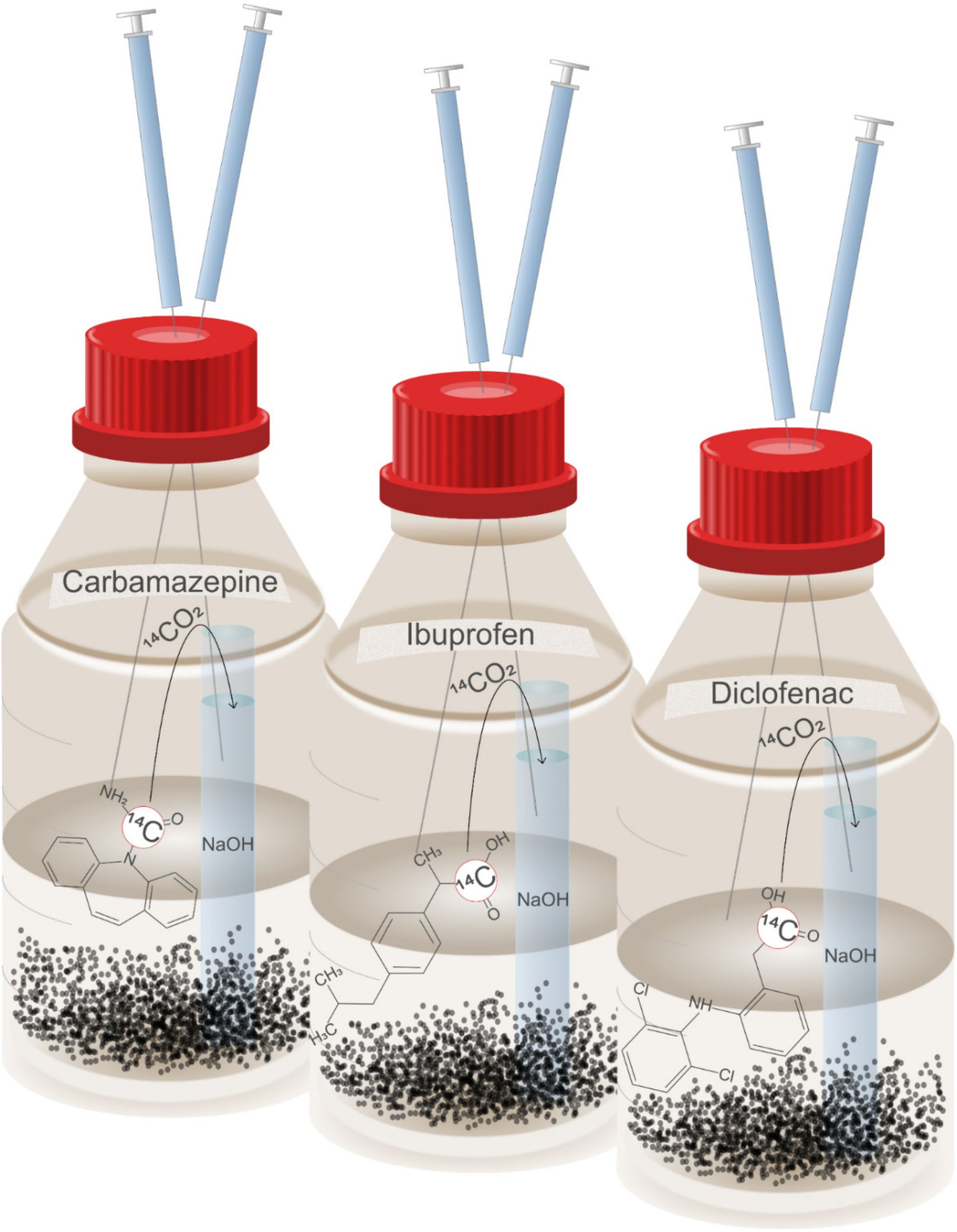

Figure 13. Schematic of incubations with ^{14}C-labeled organic micropollutants. The positions of the ^{14}C are indicated with red circles.

Liquid scintillation counting

Liquid scintillation counting is a method to quantify radioactivity from, for example, β-emitting isotopes such as ^{14}C. A sample that contains radioactive isotopes—for example, the spiked wastewater or NaOH from the $^{14}CO_2$ trap—is mixed with a scintillation cocktail that usually consists of an aromatic, organic solvent, the scintillator, and surfactants (Perkin Elmer, 2024). The energy from the β-radiation from the ^{14}C decay is transferred to the molecules in the scintillator, which results in an electron excitation. On the de-excitation of the electrons, photons are emitted and detected by the liquid scintillation counter (Figure 14).

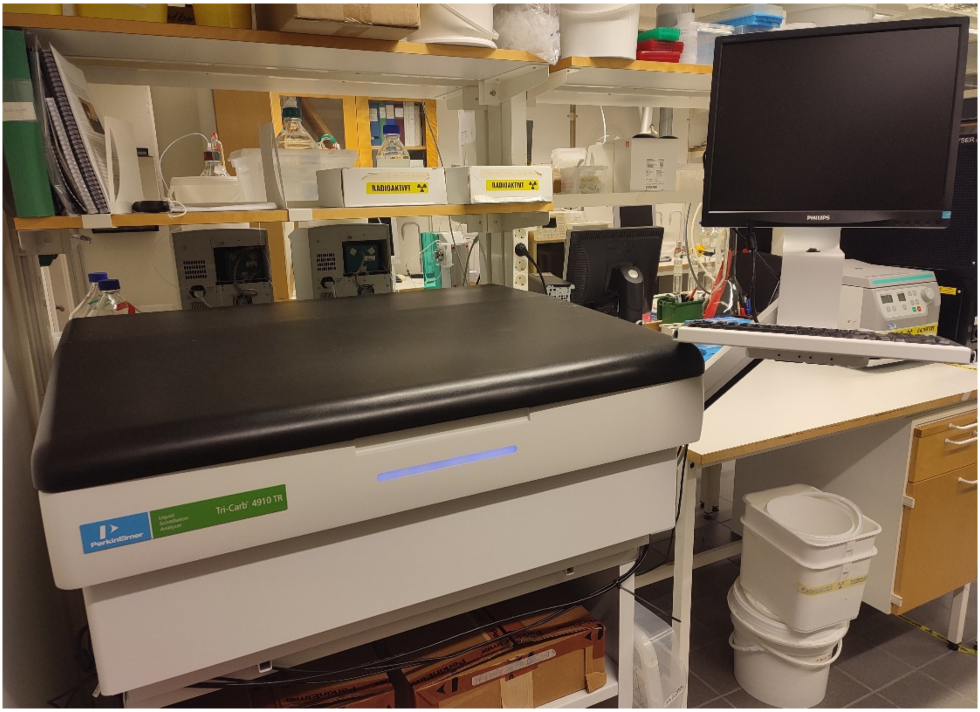

Figure 14. Photograph of the liquid scintillation counter.

The method provides a direct measure of biological degradation, by monitoring the $^{14}CO_2$ formation from ^{14}C-labeled organic compounds. However, only degradation of the ^{14}C-labeled moiety will be measured, which means that other transformations are not detected.

Selection of ^{14}C-labeled organic micropollutants

The organic micropollutants that were studied were ibuprofen, diclofenac and carbamazepine (Figure 15), chosen based on their varying susceptibilities to biodegradation. Ibuprofen is known to be easily degradable in biological processes, such as the activated sludge process (Joss et al., 2006), diclofenac can be degraded in some biofilm processes (Betsholtz et al., 2021; Falås et al., 2012), and carbamazepine is considered to be inert to biological wastewater treatment (Joss et al., 2006) and was included as a reference.

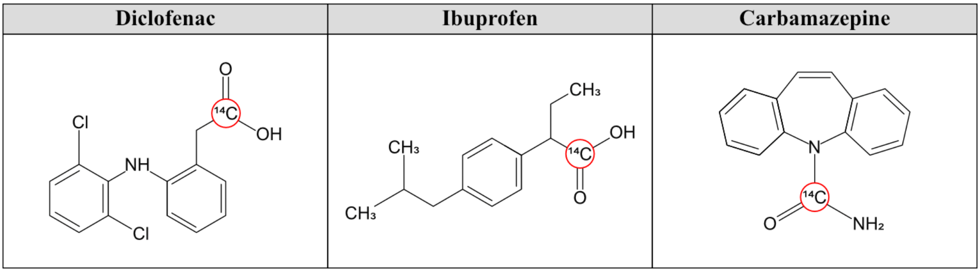

Figure 15. ^{14}C-labeled organic micropollutants studied in the incubation experiments. ^{14}C positions are indicated by red circles.

4.3 Analytical methods

Water quality was assessed based on the analysis of a panel of microbial and chemical parameters. The analytical methods are detailed in Papers I, II, and III.

4.3.1 Microbial analysis

E. coli and total coliforms were measured using Colilert and Quantitray 2000, and total cell concentration (TCC) was analyzed using flow cytometry (Papers I and II). 16S rRNA gene amplicon sequencing was performed of the full-length (Paper I) and of the V1V8 region (Paper III) of the 16S rRNA gene. Water samples were also sent to the commercial laboratory Eurofins for additional microbial analysis (Paper II).

16S rRNA gene sequencing was performed on GAC influent, effluent, and media (Paper I) and on GAC and sand media (Paper III). The water samples were filtered onto 0.22 µm Isopore filters to collect the bacteria for DNA extraction.

Flow cytometric results were analyzed in FlowJo (v 10.6.2). Sequencing data were visualized in R, versions 4.1.2 (Paper I) and 4.2.1 (Paper III), using Rstudio, versions 1.4.1717 (Paper I) and 2022.07.1 (Paper III), with the ampvis 2 package v.2.7.27 (Paper I) and v.2.8.3 (Paper III).

Bacteria that likely originated from contamination were identified and removed using the R package Decontam (https://github.com/benjjneb/decontam, method="frequency") (Paper I). This step was deemed necessary for the analysis in

Paper I, due to the low DNA concentrations after DNA extraction of the water samples.

4.3.2 Chemical analysis

The chemical analysis comprised metals (Paper I), organic micropollutants (Paper I), DOC, PO_4^{3-}-P, NO_3^--N, NO_2^--N, and NH_4^+-N (Papers I, II, and III).

The selection of the analyzed organic micropollutants was based on the EU watch list[1] and the Swiss list for micropollutant removal from wastewater (VSA, 2024). 9 of the 12 substances in Annex I to Proposal 2022/0345 (COD)[2] were analyzed (Paper I). Organic micropollutants were concentrated through solid-phase extraction and analyzed using ultra-performance liquid chromatography with tandem mass spectrometry, as described in Paper I.

The metals and other inorganic trace compounds were selected based on Swedish drinking water legislation (LIVSFS 2022:12) and the US EPA (2012) and analyzed using inductively coupled plasma optical emission spectrometry or, when necessary due to low concentrations, with inductively coupled plasma mass spectrometry, as described in Paper I.

PO_4^{3-}-P, NO_3^--N, NO_2^--N, and NH_4^+-N were analyzed using ion chromatography. DOC was analyzed using HACH Lange LCK 385 cuvettes and a HACH DR2800 spectrophotometer (Papers I and III) or Shimadzu (TOC-L) (Paper II).

4.4 Qualitative methods

The discourses surrounding wastewater reuse and desalination were examined through interviews and a media analysis. The discourses over four water treatment plants (with desalination and wastewater reuse) on two islands (Öland and Gotland) were compared. To identify general and regional traits in these discourses, they were contrasted to general ones that were summarized from the literature.

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[1] Latest updated version: Commission implementing decision (EU) 2022/1307 of 22 July 2022 establishing a watch list of substances for Union-wide monitoring in the field of water policy pursuant to Directive 2008/105/EC of the European Parliament and of the Council.

[2] 2022/0345 (COD) Proposal for a Directive of the European Parliament and of the Council concerning urban wastewater treatment (recast).

4.4.1 Media analysis

In the media analysis, 649 newspaper and debate articles in local media were examined (Paper IV). Articles that dated from five years before inauguration until the year of inauguration of the respective plant were included. Local newspapers were included to capture the local discourse. The search strings were "dricksvatten Gotland" (drinking water Gotland) during 2011–2019, and "dricksvatten Öland" (drinking water Öland) during 2012–2019. The methods are detailed in Paper IV.

4.4.2 Interviews

A total of 23 semi-structured interviews were held with water utility employees, politicians, consultants, and local representatives who were involved in discussions and decisions regarding water supply (Paper IV). The selection of interviewees was based on a media search on the terms "Återanvändning av avloppsvatten" (Wastewater reuse) and "Avsaltning" (Desalination) and the name of respective location. Additional interviewees were selected through snowball sampling, in which the interviewees suggested other relevant participants for the study. Interviews were performed until data saturation occurred, i.e. until additional interviews did not contribute with new information that was assessed as relevant for the study.

The interviews followed an interview guide, based on relevant themes identified in the literature: 1) background on the choice of wastewater reuse and desalination, 2) legitimacy of wastewater reuse and desalination, 3) knowledge on wastewater reuse and desalination, and 4) local and regional factors. The media analysis was performed after completion of approximately half of the interviews, and based on this analysis, the interviews were then shifted toward perceptions and discussions of wastewater reuse and desalination. The interviews were transcribed and coded.

4.4.3 Coding and analysis

The interviews and debate and newspaper articles were coded using inductive coding (Paper IV). From the inductive coding, the discussions around water resources, desalination, and wastewater reuse emerged as a salient theme, and the coding was thus shifted towards opinions, perceptions, arguments and visions of these. Frequently brought up arguments, perceptions or opinions were summarized into storylines and imaginaries.

5 Wastewater reuse with granular activated carbon

Several aspects of GAC filtration were studied, including temporal development of and annual variations in treatment capacity (Paper I), UV disinfection of GAC effluents (Paper II), and biological degradation of organic micropollutants in the filters (Paper III). The results are described in the following sections.

5.1 Temporal development and annual variations in treatment capacity

The GAC filter at Kivik WWTP was studied over one full year after commissioning, allowing the temporal development and annual variations in treatment capacity to be examined. Large variations were observed throughout the year—for example, notably higher inflow to the WWTP during winter than during summer (Figure 16).

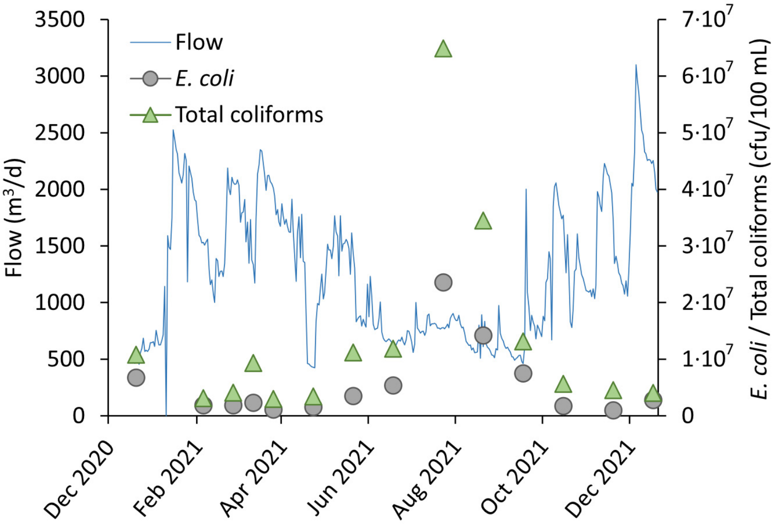


Figure 16. Flow and concentrations of *E. coli* and total coliforms in the influent to Kivik WWTP.

This pattern was likely attributed to the lower infiltration and inflow to the sewer system during summer. Together with an increased population in June–August, this contributed to a more concentrated influent, as evidenced by the concentrations of *E. coli* and total coliforms (Figure 16).

5.1.1 Microbial water quality

The average log removal value (LRV) over the entire treatment plant, calculated from the MBR influent to the GAC effluents, was 5.8 for *E. coli* and 5.5 for total coliforms, and the highest removal was achieved in the MBR (Figure 17).

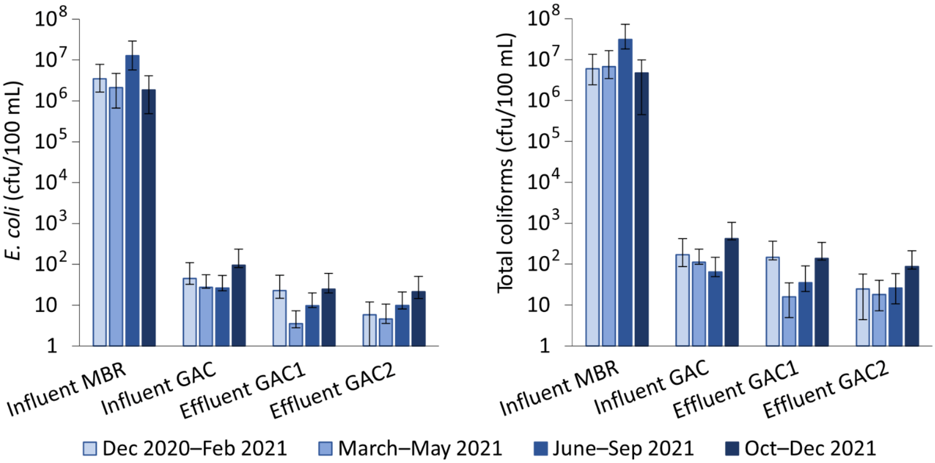


Figure 17. Average concentrations and 25th and 75th percentiles of *E. coli* and total coliforms through Kivik WWTP.

An increase in TCC was observed downstream of the GAC filters and during the summer months (Figure 18, left), likely explained in part by the development of a biofilm in the GAC filter, causing a release of cells to the effluent. The annual flow variations also affected this pattern, and by normalizing the TCC, *E. coli*, and total coliform concentrations to the flow, we calculated their total number, or total load, in the GAC influent and effluents (Figure 18, right). The total cell load (TCL) increased from commissioning of the filter, with highest values in the GAC effluents in October and November of 2021, after which they declined, potentially due to changing environmental conditions, such as lower temperatures from October onward. Whereas the TCC increased in the GAC effluent compared with the influent, the concentrations of *E. coli* and total coliforms decreased (Figure 18), although only slightly, with an LRV of 0.37 and 0.29, respectively. These data confirmed previous observations of the removal of these indicator bacteria in GAC filters, albeit a low removal (El-Zanfaly et al., 1998; Hijnen et al., 2010; Spit et al., 2022).

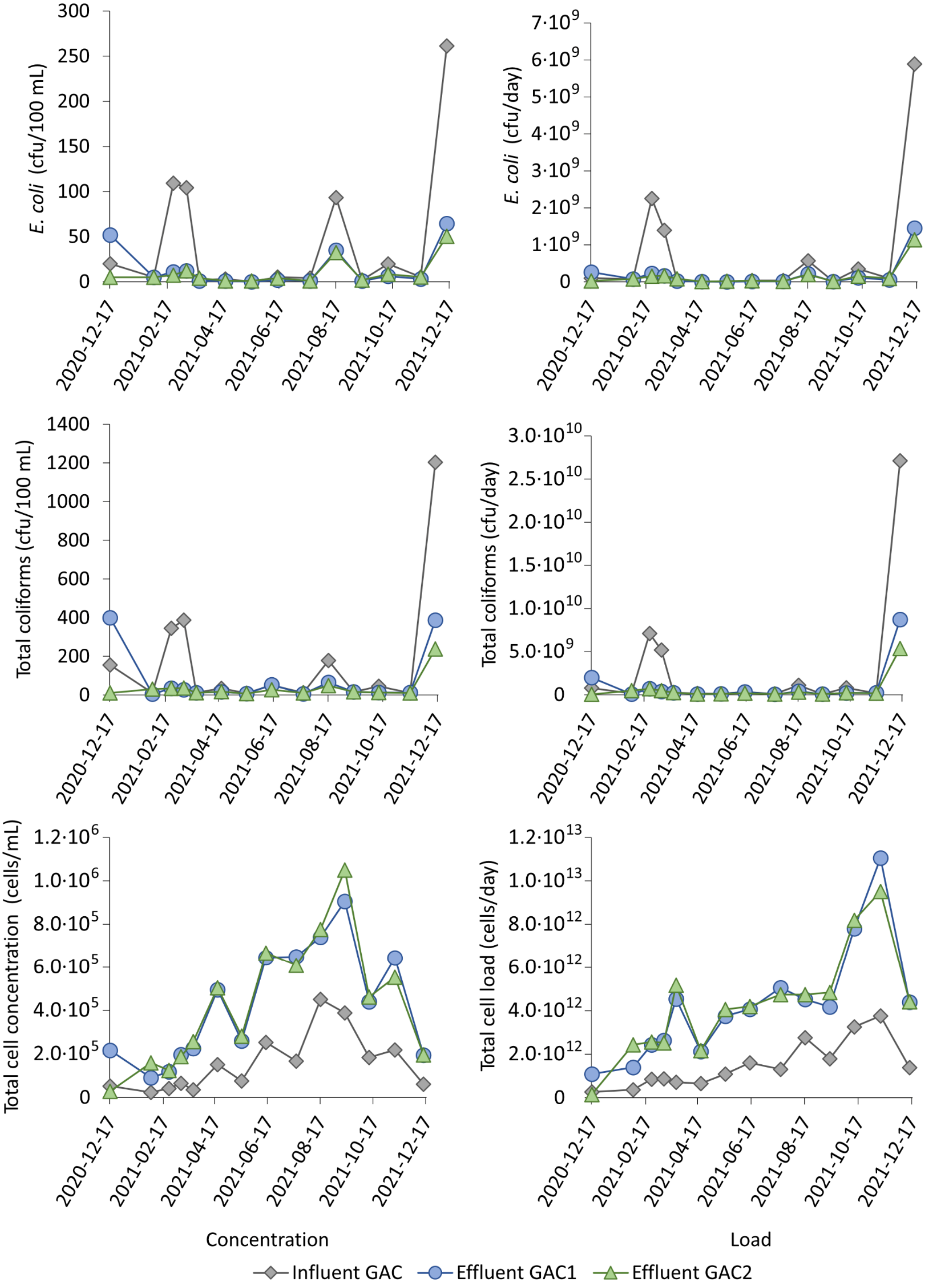


Figure 18. Concentrations (left) and loads (right) of *E. coli*, total coliforms, and total cells in the GAC influent and effluents.

Bacterial communities

Together, the increased TCC and decreased concentrations of *E. coli* and total coliforms indicate the selection of certain bacteria in the GAC-bound biofilm and its influence on the bacterial composition in the GAC effluent. This mechanism is supported by the temporal changes in the bacterial communities on the GAC1 granules and in the GAC1 effluent (Figure 19).

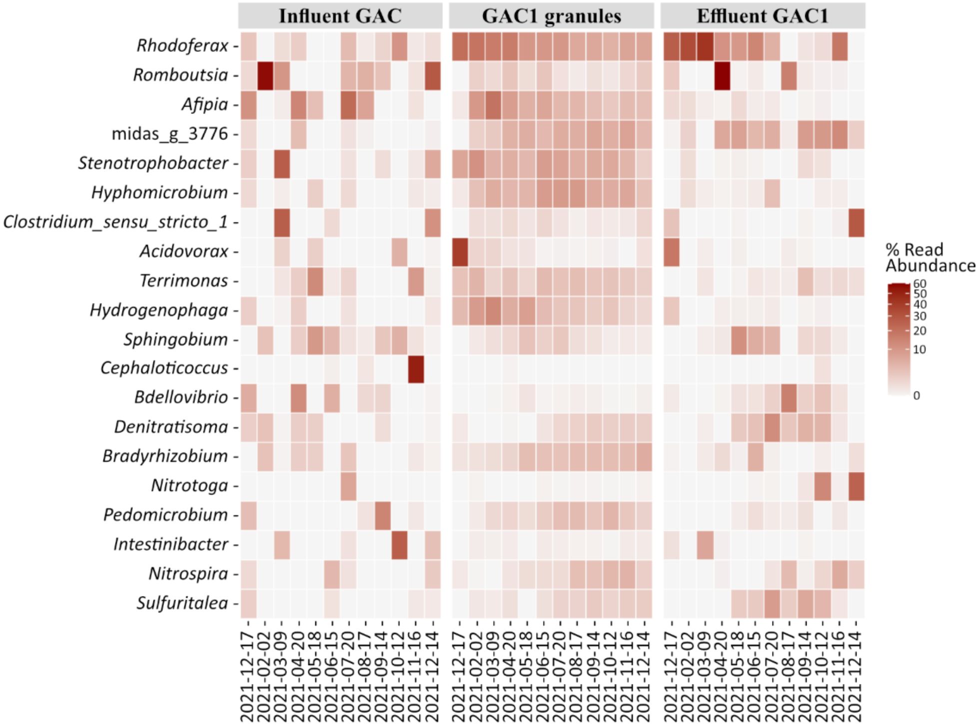


Figure 19. Heatmap of the 20 most abundant genera present in the Kivik GAC influent and effluent and on the granules.

Several of the genera on the GAC granules and in the GAC influent and effluent, such as *Rhodoferax*, *Romboutsia*, *Afipia*, *Hyphomicrobium*, *Terrimonas*, *Bradyrhizobium*, *Nitrospira*, and *Sulfuritalea* (Figure 19), have been observed in activated sludge processes (Dueholm et al., 2022; Freeman et al., 2023; Nierychlo et al., 2020). Further, *Hyphomicrobium*, *Terrimonas*, *Intestinibacter*, and *Nitrospira* have been observed in biofilm in wastewater moving bed biofilm reactors (Cimbritz et al., 2019). Other genera, including *Stenotrophobacter* and *Pedomicrobium*, have been observed in soil samples (Li et al., 2022), and their presence in the samples from Kivik WWTP could have been influenced by the infiltration and inflow to the sewer system.

On the GAC granules, the abundance of some bacterial genera increased with increased operation time, such as *Denitratisoma*, *Bradyrhizobium*, *Pedomicrobium*, *Nitrospira*, and *Sulfuritalea*, whereas the abundance of others decreased, including *Rhodoferax* and *Hydrogenophaga*; these patterns were sometimes also reflected in the microbial community in the effluent (Figure 19). These data indicate an effect from the microbial community on the granules on that of the effluent, potentially contributing to the increased TCC but decreased concentrations of *E. coli* and total coliforms.

Low amounts of DNA were possible to extract from the influent and effluent, perhaps due to the UF in the upstream MBR. A high cell removal was attributed to the UF with a pore size of 38 nm, compared with that of the filters that were used for the up-concentration of bacteria prior to extraction (220 nm).

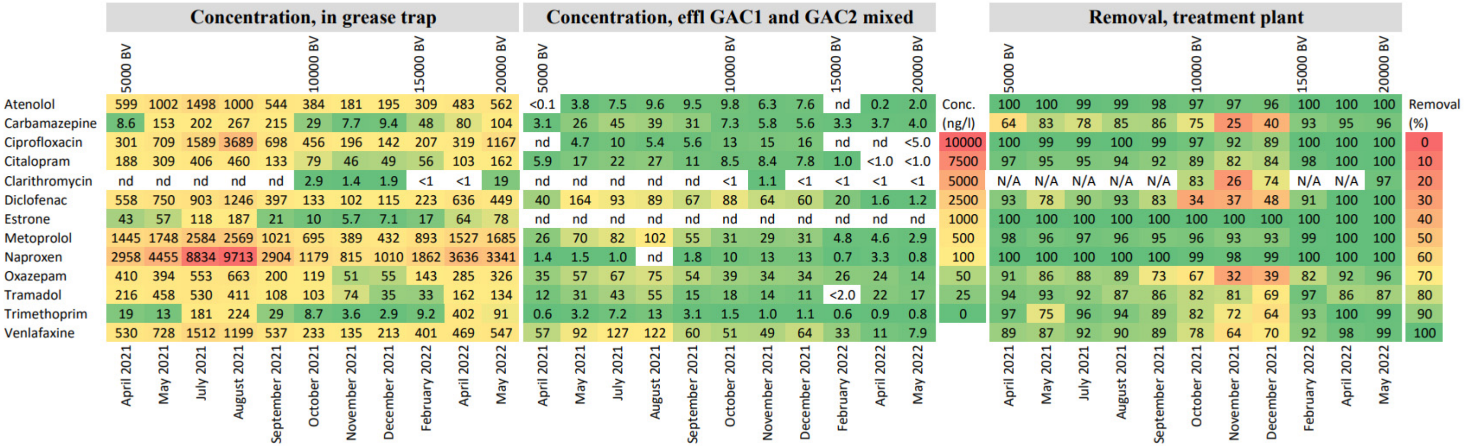
Implications for wastewater reuse

Regulation (EU) 2020/741 on minimum requirements for water reuse for irrigation defines four quality classes: A, B, C, and D. These include limits on *E. coli*, biochemical oxygen demand (BOD_5), total suspended solids (TSS), and turbidity (Table 2). TSS and turbidity were measured on three later occasions (< 1 mg/L and < 1 NTU respectively) (Paper II) and met the criteria for quality class A (≤ 10 mg/L respective ≤ 5 NTU). The average *E. coli* concentration in the GAC1 and GAC2 effluents was 14 ± 21 cfu/100 mL and 10 ± 14 cfu/100 mL, respectively, and the limit for quality class A was exceeded in 5 of 14 samples from the GAC1 effluent and in 3 of 14 samples from the GAC2 effluent. These results imply that *E. coli* is the most critical parameter and that to fulfill the quality class A criteria, measures must be taken to ensure lower concentrations of *E. coli*, such as disinfection with UV.

The average concentration of total coliforms in the GAC1 and GAC2 effluents was 75 ± 136 cfu/100 mL and 37 ± 59 cfu/100 mL, respectively, and the concentrations of *E. coli* and total coliforms thus exceeded the limits in the Swedish drinking water criteria (LIVSFS 2022:12) (Table 2).

5.1.2 Chemical water quality

The removal of organic micropollutants throughout the treatment plant, calculated from the influent concentration ("in grease trap") to the mixed GAC1 and GAC2 effluent concentration, was studied from April 2021 to May 2022 (Figure 20), and that by the GAC1 filter was studied from commissioning of the plant in December 2020 until December 2021 (Figure 21). When the effluent concentration was below the limit of quantification (LOQ), a concentration of 0.5 x LOQ was assumed.



Concentration, in grease trap

	5000 BV					10000 BV			15000 BV		20000 BV
	April 2021	May 2021	July 2021	August 2021	September 2021	October 2021	November 2021	December 2021	February 2022	April 2022	May 2022
Atenolol	599	1002	1498	1000	544	384	181	195	309	483	562
Carbamazepine	8.6	153	202	267	215	29	7.7	9.4	48	80	104
Ciprofloxacin	301	709	1589	3689	698	456	196	142	207	319	1167
Citalopram	188	309	406	460	133	79	46	49	56	103	162
Clarithromycin	nd	nd	nd	nd	nd	2.9	1.4	1.9	<1	<1	19
Diclofenac	558	750	903	1246	397	133	102	115	223	636	449
Estrone	43	57	118	187	21	10	5.7	7.1	17	64	78
Metoprolol	1445	1748	2584	2569	1021	695	389	432	893	1527	1685
Naproxen	2958	4455	8834	9713	2904	1179	815	1010	1862	3636	3341
Oxazepam	410	394	553	663	200	119	51	55	143	285	326
Tramadol	216	458	530	411	108	103	74	35	33	162	134
Trimethoprim	19	13	181	224	29	8.7	3.6	2.9	9.2	402	91
Venlafaxine	530	728	1512	1199	537	233	135	213	401	469	547

Concentration, effl GAC1 and GAC2 mixed

	5000 BV					10000 BV			15000 BV		20000 BV
	April 2021	May 2021	July 2021	August 2021	September 2021	October 2021	November 2021	December 2021	February 2022	April 2022	May 2022
Atenolol	<0.1	3.8	7.5	9.6	9.5	9.8	6.3	7.6	nd	0.2	2.0
Carbamazepine	3.1	26	45	39	31	7.3	5.8	5.6	3.3	3.7	4.0
Ciprofloxacin	nd	4.7	10	5.4	5.6	13	15	16	nd	nd	<5.0
Citalopram	5.9	17	22	27	11	8.5	8.4	7.8	1.0	<1.0	<1.0
Clarithromycin	nd	nd	nd	nd	nd	<1	1.1	<1	<1	<1	<1
Diclofenac	40	164	93	89	67	88	64	60	20	1.6	1.2
Estrone	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Metoprolol	26	70	82	102	55	31	29	31	4.8	4.6	2.9
Naproxen	1.4	1.5	1.0	nd	1.8	10	13	13	0.7	3.3	0.8
Oxazepam	35	57	67	75	54	39	34	34	26	24	14
Tramadol	12	31	43	55	15	18	14	11	<2.0	22	17
Trimethoprim	0.6	3.2	7.2	13	3.1	1.5	1.0	1.1	0.6	0.9	0.8
Venlafaxine	57	92	127	122	60	51	49	64	33	11	7.9

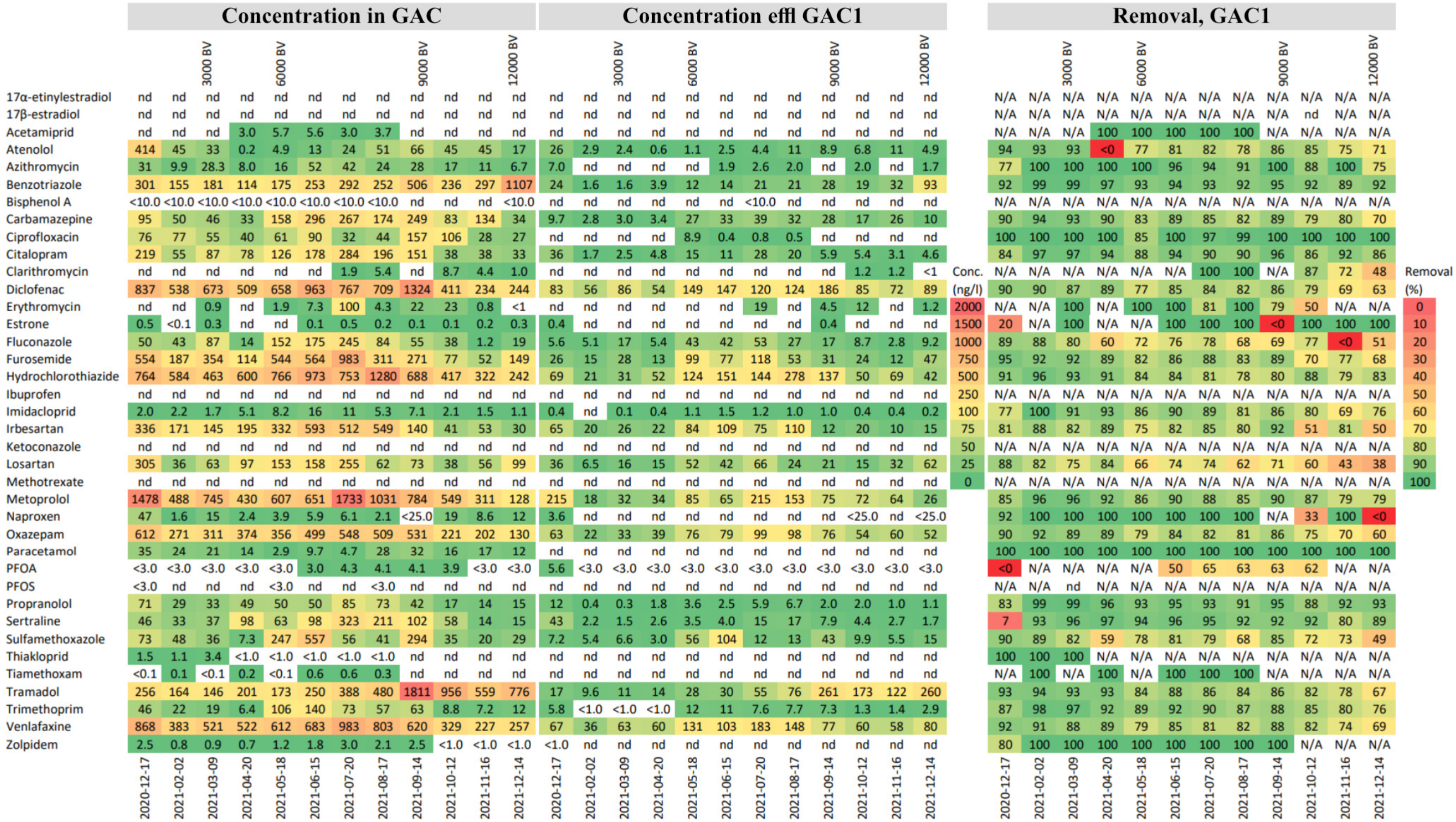
Conc. (ng/l)
10000
7500
5000
2500
1000
500
100
50
25
0

Removal, treatment plant

	5000 BV					10000 BV			15000 BV		20000 BV
	April 2021	May 2021	July 2021	August 2021	September 2021	October 2021	November 2021	December 2021	February 2022	April 2022	May 2022
Atenolol	100	100	99	99	98	97	97	96	100	100	100
Carbamazepine	64	83	78	85	86	75	25	40	93	95	96
Ciprofloxacin	100	99	99	100	99	97	92	89	100	100	100
Citalopram	97	95	95	94	92	89	82	84	98	100	100
Clarithromycin	N/A	N/A	N/A	N/A	N/A	83	26	74	N/A	N/A	97
Diclofenac	93	78	90	93	83	34	37	48	91	100	100
Estrone	100	100	100	100	100	100	100	100	100	100	100
Metoprolol	98	96	97	96	95	96	93	93	99	100	100
Naproxen	100	100	100	100	100	99	98	99	100	100	100
Oxazepam	91	86	88	89	73	67	32	39	82	92	96
Tramadol	94	93	92	87	86	82	81	69	97	86	87
Trimethoprim	97	75	96	94	89	82	72	64	93	100	99
Venlafaxine	89	87	92	90	89	78	64	70	92	98	99

Removal (%)
0
10
20
30
40
50
60
70
80
90
100

Figure 20. Concentrations of organic micropollutants in the Kivik WWTP influent and effluent. When the effluent concentration was below the limit of quantification (LOQ), a concentration of 0.5 x LOQ was assumed.



Concentration in GAC

			3000 BV		6000 BV				9000 BV			12000 BV
17α-etinylestradiol	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
17β-estradiol	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Acetamiprid	nd	nd	nd	3.0	5.7	5.6	3.0	3.7	nd	nd	nd	nd
Atenolol	414	45	33	0.2	4.9	13	24	51	66	45	45	17
Azithromycin	31	9.9	28.3	8.0	16	52	42	24	28	17	11	6.7
Benzotriazole	301	155	181	114	175	253	292	252	506	236	297	1107
Bisphenol A	<10.0	<10.0	<10.0	<10.0	<10.0	<10.0	<10.0	<10.0	nd	nd	nd	<10.0
Carbamazepine	95	50	46	33	158	296	267	174	249	83	134	34
Ciprofloxacin	76	77	55	40	61	90	32	44	157	106	28	27
Citalopram	219	55	87	78	126	178	284	196	151	38	38	33
Clarithromycin	nd	nd	nd	nd	nd	nd	1.9	5.4	nd	8.7	4.4	1.0
Diclofenac	837	538	673	509	658	963	767	709	1324	411	234	244
Erythromycin	nd	nd	0.9	nd	1.9	7.3	100	4.3	22	23	0.8	<1
Estrone	0.5	<0.1	0.3	nd	nd	0.1	0.5	0.2	0.1	0.1	0.2	0.3
Fluconazole	50	43	87	14	152	175	245	84	55	38	1.2	19
Furosemide	554	187	354	114	544	564	983	311	271	77	52	149
Hydrochlorothiazide	764	584	463	600	766	973	753	1280	688	417	322	242
Ibuprofen	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Imidacloprid	2.0	2.2	1.7	5.1	8.2	16	11	5.3	7.1	2.1	1.5	1.1
Irbesartan	336	171	145	195	332	593	512	549	140	41	53	30
Ketoconazole	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Losartan	305	36	63	97	153	158	255	62	73	38	56	99
Methotrexate	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Metoprolol	1478	488	745	430	607	651	1733	1031	784	549	311	128
Naproxen	47	1.6	15	2.4	3.9	5.9	6.1	2.1	<25.0	19	8.6	12
Oxazepam	612	271	311	374	356	499	548	509	531	221	202	130
Paracetamol	35	24	21	14	2.9	9.7	4.7	28	32	16	17	12
PFOA	<3.0	<3.0	<3.0	<3.0	<3.0	3.0	4.3	4.1	4.1	3.9	<3.0	<3.0
PFOS	<3.0	nd	nd	nd	<3.0	nd	nd	<3.0	nd	nd	nd	nd
Propranolol	71	29	33	49	50	50	85	73	42	17	14	15
Sertraline	46	33	37	98	63	98	323	211	102	58	14	15
Sulfamethoxazole	73	48	36	7.3	247	557	56	41	294	35	20	29
Thiakloprid	1.5	1.1	3.4	<1.0	<1.0	<1.0	<1.0	<1.0	nd	nd	nd	nd
Tiamethoxam	<0.1	0.1	<0.1	0.2	<0.1	0.6	0.6	0.3	nd	nd	nd	nd
Tramadol	256	164	146	201	173	250	388	480	1811	956	559	776
Trimethoprim	46	22	19	6.4	106	140	73	57	63	8.8	7.2	12
Venlafaxine	868	383	521	522	612	683	983	803	620	329	227	257
Zolpidem	2.5	0.8	0.9	0.7	1.2	1.8	3.0	2.1	2.5	<1.0	<1.0	<1.0
	2020-12-17	2021-02-02	2021-03-09	2021-04-20	2021-05-18	2021-06-15	2021-07-20	2021-08-17	2021-09-14	2021-10-12	2021-11-16	2021-12-14

Concentration effl GAC1

			3000 BV		6000 BV				9000 BV			12000 BV
17α-etinylestradiol	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
17β-estradiol	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Acetamiprid	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Atenolol	26	2.9	2.4	0.6	1.1	2.5	4.4	11	8.9	6.8	11	4.9
Azithromycin	7.0	nd	nd	nd	nd	1.9	2.6	2.0	nd	2.0	nd	1.7
Benzotriazole	24	1.6	1.6	3.9	12	14	21	21	28	19	32	93
Bisphenol A	nd	nd	nd	nd	nd	nd	<10.0	nd	nd	nd	nd	nd
Carbamazepine	9.7	2.8	3.0	3.4	27	33	39	32	28	17	26	10
Ciprofloxacin	nd	nd	nd	nd	8.9	0.4	0.8	0.5	nd	nd	nd	nd
Citalopram	36	1.7	2.5	4.8	15	11	28	20	5.9	5.4	3.1	4.6
Clarithromycin	nd	nd	nd	nd	nd	nd	nd	nd	nd	1.2	1.2	<1
Diclofenac	83	56	86	54	149	147	120	124	186	85	72	89
Erythromycin	nd	nd	nd	nd	nd	nd	19	nd	4.5	12	nd	1.2
Estrone	0.4	nd	nd	nd	nd	nd	nd	nd	0.4	nd	nd	nd
Fluconazole	5.6	5.1	17	5.4	43	42	53	27	17	8.7	2.8	9.2
Furosemide	26	15	28	13	99	77	118	53	31	24	12	47
Hydrochlorothiazide	69	21	31	52	124	151	144	278	137	50	69	42
Ibuprofen	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Imidacloprid	0.4	nd	0.1	0.4	1.1	1.5	1.2	1.0	1.0	0.4	0.4	0.2
Irbesartan	65	20	26	22	84	109	75	110	12	20	10	15
Ketoconazole	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Losartan	36	6.5	16	15	52	42	66	24	21	15	32	62
Methotrexate	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Metoprolol	215	18	32	34	85	65	215	153	75	72	64	26
Naproxen	3.6	nd	nd	nd	nd	nd	nd	nd	nd	<25.0	nd	<25.0
Oxazepam	63	22	33	39	76	79	99	98	76	54	60	52
Paracetamol	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
PFOA	5.6	<3.0	<3.0	<3.0	<3.0	<3.0	<3.0	<3.0	<3.0	<3.0	<3.0	<3.0
PFOS	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Propranolol	12	0.4	0.3	1.8	3.6	2.5	5.9	6.7	2.0	2.0	1.0	1.1
Sertraline	43	2.2	1.5	2.6	3.5	4.0	15	17	7.9	4.4	2.7	1.7
Sulfamethoxazole	7.2	5.4	6.6	3.0	56	104	12	13	43	9.9	5.5	15
Thiakloprid	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Tiamethoxam	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
Tramadol	17	9.6	11	14	28	30	55	76	261	173	122	260
Trimethoprim	5.8	<1.0	<1.0	<1.0	12	11	7.6	7.7	7.3	1.3	1.4	2.9
Venlafaxine	67	36	63	60	131	103	183	148	77	60	58	80
Zolpidem	<1.0	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd	nd
	2020-12-17	2021-02-02	2021-03-09	2021-04-20	2021-05-18	2021-06-15	2021-07-20	2021-08-17	2021-09-14	2021-10-12	2021-11-16	2021-12-14

Conc. (ng/l)
2000
1500
1000
750
500
250
100
75
50
25
0

Removal, GAC1

			3000 BV		6000 BV				9000 BV			12000 BV
17α-etinylestradiol	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
17β-estradiol	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	nd	N/A	N/A
Acetamiprid	N/A	N/A	N/A	100	100	100	100	100	N/A	N/A	N/A	N/A
Atenolol	94	93	93	<0	77	81	82	78	86	85	75	71
Azithromycin	77	100	100	100	100	96	94	91	100	88	100	75
Benzotriazole	92	99	99	97	93	94	93	92	95	92	89	92
Bisphenol A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Carbamazepine	90	94	93	90	83	89	85	82	89	79	80	70
Ciprofloxacin	100	100	100	100	85	100	97	99	100	100	100	100
Citalopram	84	97	97	94	88	94	90	90	96	86	92	86
Clarithromycin	N/A	N/A	N/A	N/A	N/A	N/A	100	100	N/A	87	72	48
Diclofenac	90	90	87	89	77	85	84	82	86	79	69	63
Erythromycin	N/A	N/A	100	N/A	100	100	81	100	79	50	N/A	N/A
Estrone	20	N/A	100	N/A	N/A	100	100	100	<0	100	100	100
Fluconazole	89	88	80	60	72	76	78	68	69	77	<0	51
Furosemide	95	92	92	89	82	86	88	83	89	70	77	68
Hydrochlorothiazide	91	96	93	91	84	84	81	78	80	88	79	83
Ibuprofen	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Imidacloprid	77	100	91	93	86	90	89	81	86	80	69	76
Irbesartan	81	88	82	89	75	82	85	80	92	51	81	50
Ketoconazole	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Losartan	88	82	75	84	66	74	74	62	71	60	43	38
Methotrexate	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Metoprolol	85	96	96	92	86	90	88	85	90	87	79	79
Naproxen	92	100	100	100	100	100	100	100	N/A	33	100	<0
Oxazepam	90	92	89	89	79	84	82	81	86	75	70	60
Paracetamol	100	100	100	100	100	100	100	100	100	100	100	100
PFOA	<0	N/A	N/A	N/A	N/A	50	65	63	63	62	N/A	N/A
PFOS	N/A	N/A	nd	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Propranolol	83	99	99	96	93	95	93	91	95	88	92	93
Sertraline	7	93	96	97	94	96	95	92	92	92	80	89
Sulfamethoxazole	90	89	82	59	78	81	79	68	85	72	73	49
Thiakloprid	100	100	100	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Tiamethoxam	N/A	100	N/A	100	N/A	100	100	100	N/A	N/A	N/A	N/A
Tramadol	93	94	93	93	84	88	86	84	86	82	78	67
Trimethoprim	87	98	97	92	89	92	90	87	88	85	80	76
Venlafaxine	92	91	88	89	79	85	81	82	88	82	74	69
Zolpidem	80	100	100	100	100	100	100	100	100	N/A	N/A	N/A
	2020-12-17	2021-02-02	2021-03-09	2021-04-20	2021-05-18	2021-06-15	2021-07-20	2021-08-17	2021-09-14	2021-10-12	2021-11-16	2021-12-14

Removal (%)
0
10
20
30
40
50
60
70
80
90
100

Figure 21. Concentrations of organic micropollutants in the Kivik GAC1 influent and effluent. When the effluent concentration was below the limit of quantification (LOQ), a concentration of 0.5 x LOQ was assumed.

The lowest removals (< 0%) occurred when influent concentrations were low. When the influent concentration is low, the removal calculation is sensitive to small variations in the effluent concentration, and a low removal can thus occur even though the effluent concentration is notably low, for example, for atenolol in the sample from 2021-04-20.

The concentrations in the influent varied throughout the year, with higher concentrations observed in July–August (Figure 20), likely explained by the annual flow variations. For all measured compounds, the average removal throughout the treatment plant was > 80%—which is the criterion in the proposal for a new urban wastewater directive (EU, 2022/0345/COD)—except in November and December 2021.

The removal decreased until December 2021, after which it rose (after approximately 15 000 BVs) (Figure 20). It is not possible to draw any conclusions on what caused this shift, but one potential explanation is that maturation of the biofilm resulted in the biological degradation of certain compounds. One possible compound is diclofenac, for which the effluent concentrations in April and May 2022 were notably lower than in April and May 2021, although the influent concentrations were similar (Figure 20). Despite the previously reported recalcitrance of diclofenac to conventional biological wastewater treatment (Joss et al., 2006), its biological degradation has been confirmed in certain biofilm processes (Betsholtz et al., 2021; Falås et al., 2012). The removal also increased for compounds that are known to be persistent to biological wastewater treatment, such as carbamazepine (Falås et al., 2013; Joss et al., 2006), and thus other factors than biological degradation, such as the varying flow and EBCT, must have caused their increased removal.

In addition to variations in flow and EBCT, which affect the adsorption of the compounds, the varying influent concentrations (Figure 20) may explain the temporal removal variations. Bioregeneration—a process in which the biofilm degrades previously adsorbed organic molecules and thus increases the adsorptive capacity of the GAC—could also have contributed to the removal patterns, as discussed by Baresel et al. (2019) and El Gamal et al. (2018).

Implications for wastewater reuse

A potentially concerning uptake of some pharmaceuticals have in previous studies been reported for plants irrigated with wastewater (Malchi et al., 2014; Riemenschneider et al., 2016). The concentrations of these compounds in the wastewater that was used for the irrigation (ciprofloxacin: 300 ng/L; carbamazepine: 20–1700 ng/L) were generally higher than those that have been measured in the Kivik WWTP effluent (ciprofloxacin: ≤ 16 ng/L, carbamazepine: ≤ 45 ng/L).

The concentrations of the analyzed organic micropollutants in the GAC effluents met the limits in the Australian guidelines for water recycling (2008), Reungoat (2012), and the Swedish drinking water criteria (LIVSFS 2022:12, PFAS 4 and PFAS 21 were not analyzed) (Table 4 and Figure 21). However, the limits in the Australian guidelines for water recycling (carbamazepine: 100 000 ng/L; ciprofloxacin: 250 000 ng/L) are notably higher than the concentrations reported by Malchi et al. (2014) and Riemenschneider et al. (2016), discussed above, regarding plant uptake after irrigation with treated wastewater.

This creates insecurities regarding what constitutes safe levels in reused wastewater. To complement the guidelines and limits, effluent concentrations were therefore compared with values in the literature of concentrations from measurements in Swedish lakes (Table 4). Many of the compounds in the GAC effluents were present at concentrations that were on the same order of magnitude as in the Swedish lakes, whereas others, such as diclofenac, furosemide, metoprolol, oxazepam, tramadol, and venlafaxine, during certain months were 10–100 times higher than in these lakes (Table 4 and Figure 21).

The metal concentrations met the criteria for irrigation (US EPA, 2012) (Table 3). They generally fulfilled the Swedish drinking water criteria, except for arsenic (8.6 µg/L) and manganese (59 µg/L), the levels of which exceeded the limits (5.0 µg/L arsenic and 50 µg/L manganese) on one of four occasions in one of the two GAC effluents.

5.2 Disinfection with ultraviolet radiation

In the Kivik WWTP effluent, the concentrations of chemical and microbial contaminants were low overall, but those of *E. coli* and total coliforms exceeded the limits for drinking water, and in approximately half of the cases, *E. coli* concentrations exceeded the limit for quality class A for irrigation (Paper I). Thus, UV disinfection of the GAC effluent from Kivik WWTP was investigated, and compared with UV disinfection of the GAC effluent from S:t Olof WWTP (Paper II). Both of the WWTPs comprise GAC filtration, but with different treatment upstream of the GAC filters (MBR at Kivik WWTP, and CAS + sand filtration at S:t Olof WWTP). Further, the removal of microbial contaminants through the treatment plants was analyzed (Paper II). Sampling and UV experiments were performed on three occasions.

The effects from UV disinfection varies depending on microorganism—*E. coli* and other coliform bacteria are generally inactivated more easily (Jacangelo et al., 2003; Mezzanotte et al., 2007; Sommer et al., 2000) compared with *Clostridium perfringens* (Carabias et al., 2023; Hijnen et al., 2006) and microfungi, such as yeast (Spotte and Buck, 1981). Further, the UV efficiency is affected by water quality in terms of absorbance and suspended solids content (Carré et al., 2018; Crittenden et al., 2012; Qualls et al., 1983) and thus is influenced by the upstream wastewater treatment (Venditto et al., 2022).

5.2.1 Removal of microbial contaminants by GAC filtration

The LRV of *E. coli* and total coliforms was higher for the MBR + GAC process at Kivik WWTP (6.8 and 6.7, respectively) than the CAS + sand + GAC process at S:t Olof WWTP (3.9 for both), with the highest removal observed for the MBR, likely primarily due to the UF (Figure 22).

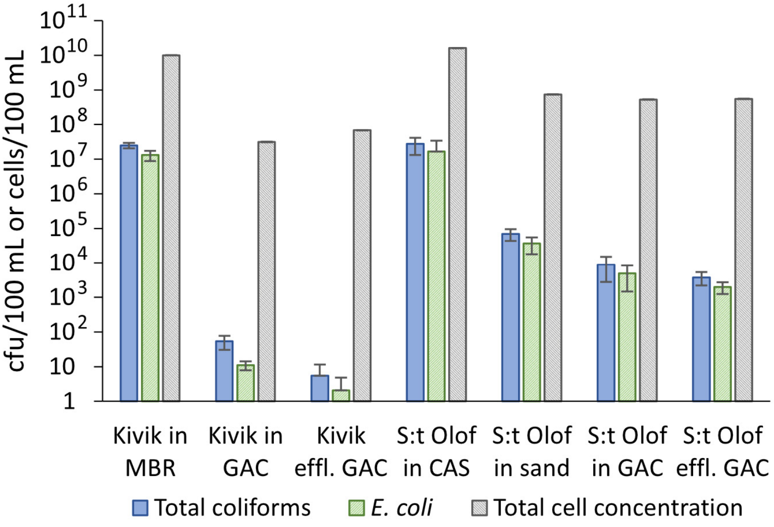


Figure 22. Total coliform, *E. coli* and total cell concentration through Kivik and S:t Olof WWTPs.

The TCC was decreased by the MBR at Kivik WWTP, and by the CAS at S:t Olof WWTP, but remained constant or increased after the GAC and sand filters at both WWTPs.

The GAC filters slightly lowered the concentrations of *E. coli* and total coliforms, as observed in previous samplings (Paper I) and research (El-Zanfaly et al., 1998; Hijnen et al., 2010; Spit et al., 2022). The LRV was 0.7 and 1.0 for *E. coli* and total coliforms, respectively, over the Kivik GAC filters and 0.4 for both *E. coli* and total coliforms over the S:t Olof GAC filter. The GAC filters on average lowered also the concentrations of *Clostridium perfringens* and intestinal enterococci, a pattern that was most prominent for the Kivik GAC filter (Figure 23).

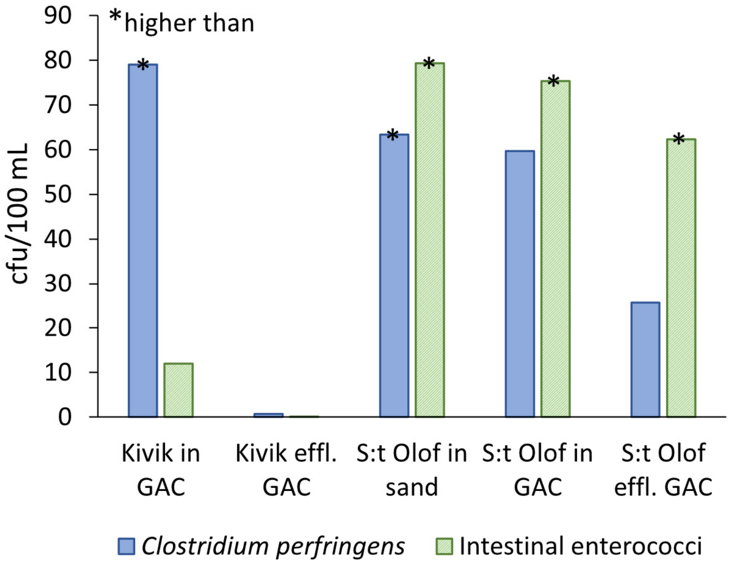


Figure 23. Concentrations of *Clostridium perfringens* and intestinal enterococci before and after the GAC and sand filters at Kivik and S:t Olof WWTPs. When one or more of the concentrations exceeded the upper LOQ (100 cfu/100 mL), the LOQ was used for calculation of the average. These average values are marked with "*higher than" than in the figure.

The increase in TCC and the declines in *E. coli*, total coliform, intestinal enterococci, and *Clostridium perfringens* concentrations could be due to a biofilm growth that results in the selection of a new bacterial community, as supported by the sequencing results (Figure 19) in Section 5.1.1. Indicator bacteria were removed to a higher extent by the Kivik GAC filter versus the S:t Olof GAC filter, potentially implying that that the capacity for the removal of microbial contaminants could be improved in GAC filters if they are preceded by an MBR, compared to GAC filters that are preceded by CAS and sand filtration. This may be due to differences in the concentrations and compositions of nutrients, DOC, suspended solids, and bacteria in the filter influents, likely affecting the microbial communities in the filters. However, the Kivik GAC filter had been in operation longer and had treated more BVs, which could also have affected the biofilm community.

The concentration of *E. coli* in the Kivik effluent (on average 2 cfu/100 mL) met the quality class A criteria in Regulation (EU) 2020/741 on all sampling occasions. In previous analyses, this limit was not met consistently (Paper I). The average *E. coli* concentration in the S:t Olof effluent was 1967 cfu/100 mL and thus met quality class D standards. The *E. coli*, total coliform, *Clostridium perfringens*, intestinal enterococci, and microfungi (sum of mold fungi and yeast) levels exceeded the limits in the Swedish drinking water criteria (Table 2, Paper II). Consequently, UV disinfection of the Kivik and S:t Olof effluents was performed.

5.2.2 Removal of microbial contaminants with UV disinfection

The Kivik and S:t Olof effluents were subjected to UV disinfection at three UV fluences (200 J/m^2, 400 J/m^2, and 700 J/m^2) in laboratory experiments on three occasions. UV inactivates *E. coli* and total coliforms efficiently (Jacangelo et al., 2003; Mezzanotte et al., 2007; Sommer et al., 2000), whereas higher UV fluences are generally required for *Clostridium perfringens* and microfungi (Carabias et al., 2023; Hijnen et al., 2006). This pattern also arose in the UV experiments. *E. coli*, total coliforms, and enterococci were easily inactivated in both WWTP effluents, whereas *Clostridium prefringens* and microfungi generally remained in higher concentrations in the UV treated effluents (Table 6).

Table 6. Concentrations of microbial parameters in the GAC effluents before and after UV treatment, versus Swedish drinking water criteria (LIVSFS 2022:12).

		Clostridium perfringens	**Enterococci**	**Microfungi**	***E. coli***	**Total coliforms**
Limit or indicator value:		Present (cfu/100 mL)	Present (cfu/100 mL)	100 cfu/100 mL	Present (cfu/100 mL)	Present (cfu/100 mL)
Kivik	2023-06-12	<1	<1	4	<1	1
effl GAC	2023-06-26	2	<1	5	5	12
	2023-10-02	<1	1	37	1	3
Kivik	2023-06-12	<1	<1	5	<1	<1
200 J/m^2	2023-06-26	<1	<1	27	<1	<1
	2023-10-02	<1	<1	550	<1	<1
Kivik	2023-06-12	<1	<1	1	<1	<1
400 J/m^2	2023-06-26	<1	<1	8	<1	<1
	2023-10-02	<1	<1	53	<1	<1
Kivik	2023-06-12	<1	<1	<1	<1	<1
700 J/m^2	2023-06-26	<1	<1	4	<1	<1
	2023-10-02	<1	<1	3	<1	<1
S:t Olof	2023-06-12	14	45	19	2700	5300
effl GAC	2023-06-26	49	>100	129	2000	3700
	2023-10-02	14	42	304	1200	2200
S:t Olof	2023-06-12	7	<1	4	<1	<1
200 J/m^2	2023-06-26	11	<1	45	<1	<1
	2023-10-02	4	<1	107	<1	<1
S:t Olof	2023-06-12	6	<1	2	<1	<1
400 J/m^2	2023-06-26	4	<1	8	<1	<1
	2023-10-02	2	<1	78	<1	<1
S:t Olof	2023-06-12	<1	<1	4	<1	<1
700 J/m^2	2023-06-26	<1	<1	3	<1	<1
	2023-10-02	1	<1	30	<1	<1

The *Clostridium perfringens* concentration in the Kivik (MBR + GAC) effluent was low, and a UV fluence of 200 J/m^2 was sufficient to obtain < 1 cfu/100 mL. The limit for microfungi (100 cfu/100 mL) was exceeded on one occasion in the Kivik effluent treated at a UV fluence of 200 J/m^2, although the high concentration (550 cfu/100 mL) is difficult to explain, given its concentration of 37 cfu/100 mL in the GAC effluent (Table 6). It is likely that this peak was caused by contamination during sample transport or analysis or by continued fungal growth between sampling and analysis.

The concentrations of the parameters in the UV-treated Kivik effluent fulfilled the Swedish drinking water criteria at all UV fluences, other than the high microfungi concentration. One Kivik 400 J/m^2 sample was therefore selected for further extensive chemical analysis to evaluate chemical drinking water quality. A UV fluence of 400 J/m^2 is commonly used for drinking water production in Sweden (Saguti et al., 2022), and thus corresponds to a standard scenario.

5.2.3 Chemical drinking water quality

The chemical analysis of the selected sample (Kivik GAC effluent, treated at a UV fluence of 400 J/m^2) comprised 114 parameters from the Swedish drinking water criteria, including metals, PFASs, trihalomethanes (THMs), polycyclic aromatic hydrocarbons (PAHs), and pesticides. A selection of the concentrations is presented in Table 7.

Table 7. Concentrations of selected chemical parameters in the Kivik GAC effluent, treated at a UV fluence of 400 J/m^2.

Parameter	Unit	Concentration, Kivik 400 J/m^2	Limit (LIVSFS 2022:12)
Sum THMs	µg/L	<4.0	100
Sum PAHs	µg/L	<0.10	0.10
Sum PFAS 4	ng/L	2.9	4.0
Sum PFAS 21	ng/L	19	100
Sum pesticides	µg/L	Not detected	0.50

All concentrations met the drinking water criteria, with the only exception being nitrate (Table S4, Paper II). Subjected to a biological treatment process with optimized denitrification, which is common at larger WWTPs, the nitrate concentration would likely be sufficiently low.

The PFAS 4 level (2.9 ng/L) neared the limit of 4.0 ng/L. With large flow fluctuations, as observed at Kivik WWTP (Section 5.1 and Paper I), the concentration could exceed this threshold during times of lower flow. However, the sample was collected during a period with low flow (Figure S1, Paper II), suggesting that this risk is low. PFAS 4 should be evaluated over a longer sampling period to ensure that its concentrations consistently meet the criteria. Further, there are no limits on pharmaceuticals in the Swedish drinking water legislation, and to implement potable reuse, the need for such limits should be evaluated.

5.3 Biological degradation of organic micropollutants in GAC filters

In addition to the increased TCC and the lower concentrations of indicator bacteria in the GAC effluents (Papers I and II), we observed signs of biological degradation of some compounds, including diclofenac, at Kivik WWTP (Paper I). This phenomenon was studied further in laboratory-scale incubation experiments with media from one of the Kivik GAC filters (GAC1), media from the sand filter and the two GAC filters at Degeberga WWTP, and media from the pilot-scale GAC filter at Svedala WWTP (Paper III).

In the GAC filters at Kivik and Svedala WWTPs—operated with high respective low influent oxygen concentration—degradation was studied over time. In the two GAC filters at Degeberga WWTP—both operated with high influent oxygen concentration and comprising different materials—degradation was studied with GAC media from different depths.

5.3.1 Temporal development with varying oxygen concentrations

Figure 24 shows the ^{14}C partitioning between the wastewater (liquid phase) and the CO_2 trap (gas phase). The ^{14}C activity decreased in the liquid phase (blue lines) in all incubations, interpreted as adsorption to the GAC media or as degradation by the GAC-bound biofilm. In the incubations with GAC media from Kivik WWTP (GAC filter operated with high oxygen concentrations; DO > 8 mg/L), $^{14}CO_2$ formed from ibuprofen and diclofenac (green lines), whereas in those with GAC media from Svedala WWTP (GAC filter operated with low oxygen concentrations; DO = 0.5–1 mg/L), $^{14}CO_2$ formation was observed only from ibuprofen. The formation of $^{14}CO_2$ originated from the biological mineralization of the labeled moieties of the compounds (the carboxyl moieties, Figure 15), and the results are supported by previous studies that have reported mineralization of these moieties in diclofenac (Jewell et al., 2016; Wu et al., 2020, 2019) and ibuprofen (Falås et al., 2018; Löffler et al., 2005). No $^{14}CO_2$ formation from carbamazepine, which is known to be persistent to biological wastewater treatment (Joss et al., 2006), was observed in any of the incubations.

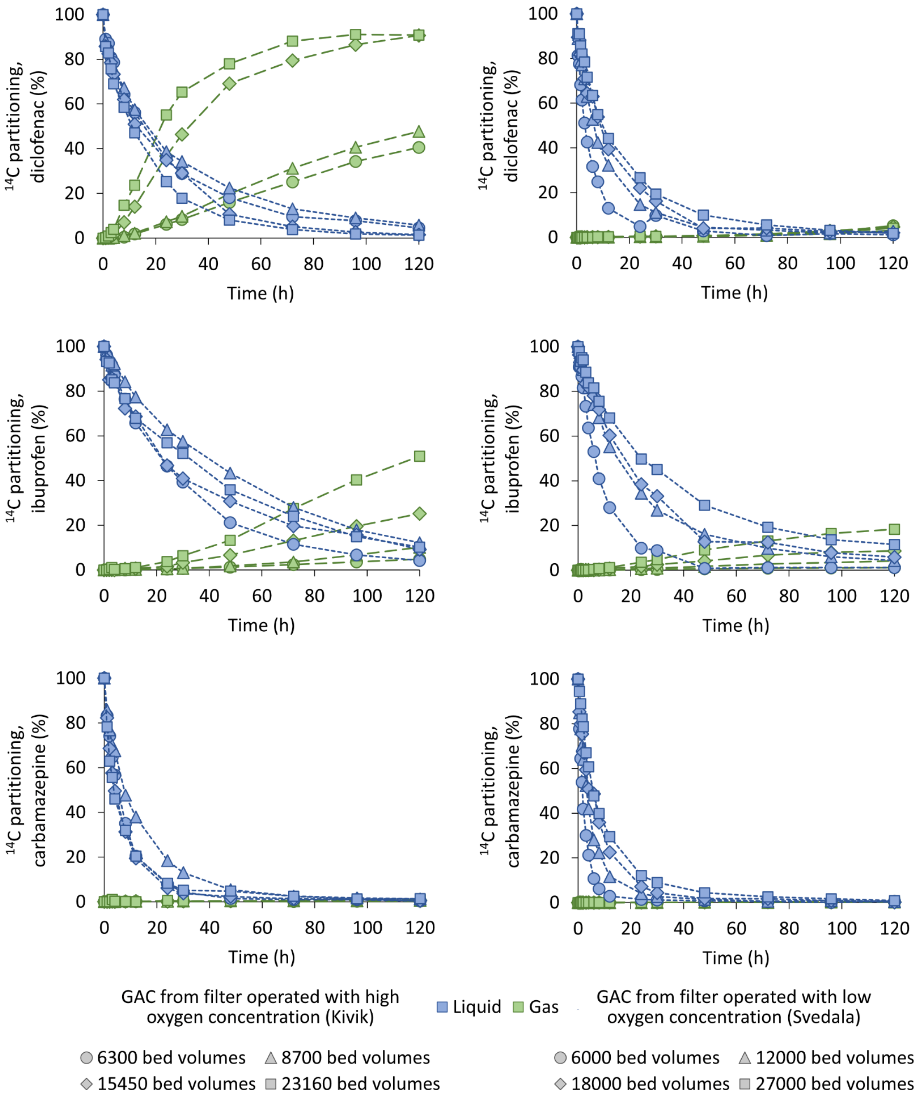


Figure 24. ^{14}C partitioning between the liquid (i.e., wastewater) and gas (i.e., CO_2 trap) phases in the incubation experiments with GAC media from filters operated at various oxygen concentration (left: high oxygen concentration, Kivik WWTP; right: low oxygen concentration, Svedala WWTP).

In the incubations with GAC from Kivik WWTP, the rate and magnitude of the $^{14}CO_2$ formation (i.e., biological degradation) from diclofenac was higher than from ibuprofen (Figure 24), in contrast to what is currently generally reported in the literature: that ibuprofen is more easily degradable (Abegglen et al., 2009; Joss et al., 2006; Langenhoff et al., 2013; Peng et al., 2019; Quintana et al., 2005). The ^{14}C-labeling of the compounds could not explain this discrepancy, since a faster $^{14}CO_2$

formation has been reported for ibuprofen versus diclofenac in experiments using compounds with the same labeling in incubations with biofilm carriers and activated sludge (Betsholtz et al., 2021; Falås et al., 2018). The rate and magnitude of the $^{14}CO_2$ formation from diclofenac and ibuprofen rose with increasing BVs (Figure 24). The biological degradation of diclofenac and ibuprofen also appeared to be affected by the oxygen concentration at which the filters were operated—the rate and magnitude of the $^{14}CO_2$ formation was notably higher in the incubations with GAC from the filter at Kivik WWTP, which was operated at high oxygen concentrations (Figure 24). These findings indicate that diclofenac can be degraded in GAC filters, and that elevated oxygen levels in the filters and prolonged operation times promote the biological degradation of organic micropollutants in GAC filters.

5.3.2 Spatial variations with various filter materials

Figure 25 shows the ^{14}C partitioning between the liquid phase and gas phase in the incubations with GAC media from three filter depths, in filters of different materials (bitumen-based and coconut-based GAC). Like those at Kivik WWTP, the GAC filters at Degeberga WWTP are operated with high oxygen concentrations, and, as with the GAC media from Kivik WWTP, diclofenac was degraded at a high rate and magnitude, especially with the bitumen-based GAC. Ibuprofen was also degraded, albeit to a lower extent than diclofenac (Figure 25).

The degradation rate of diclofenac slightly increased at greater filter depths, especially with media from the coconut-based filter, unlike ibuprofen (Figure 25). One explanation to this contrast could be that the biofilm at deeper layers is more specialized to degrade persistent substances, whereas more easily degradable compounds are degraded in the surface layer, as discussed for, for example, aquifer recharge (Alidina et al., 2014; Li et al., 2014). At the deeper layers, the adsorption was sometimes faster. This was the case with, for example, carbamazepine in the incubations with coconut-based GAC, for which the ^{14}C activity in the liquid phase (blue lines) decreased more quickly at greater depths (Figure 25). This pattern could in turn result in a faster mass transfer of the compound from the wastewater to the GAC and biofilm, potentially enabling a faster degradation. This explanation, however, is not consistent with the results of the experiments with the GAC from Kivik WWTP, in which the rate and magnitude of the CO_2 formation increased with increasing BVs, without a consistent trend in adsorption (Figure 24).

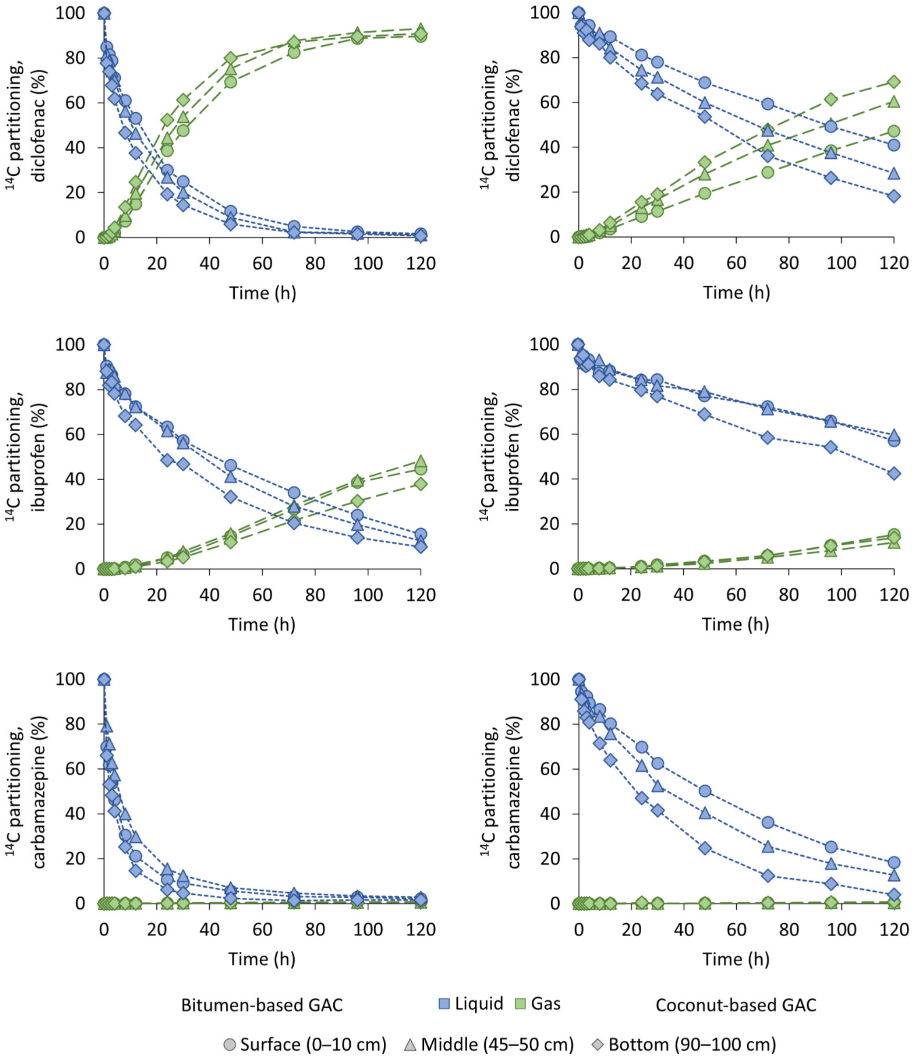


Figure 25. ^{14}C partitioning between the liquid (i.e., wastewater) and gas (i.e., CO_2 trap) phases in incubation experiments with GAC media from three filter depths (surface: 0–10 cm; middle: 40–50 cm; bottom: 90–100 cm) in filters of varying materials (left: bitumen-based GAC; right: coconut-based GAC).

In the experiment with media from the sand filter at Degeberga WWTP, sampled from the surface layer, ibuprofen was degraded to a high extent, whereas diclofenac and carbamazepine were not degraded (Figure 26), indicating that the GAC media provides conditions for the degradation of diclofenac or for the growth of microorganisms that degrade it. According to the concentrations measured in the sand filter influent and effluent, ~35% of the diclofenac was, however, removed in the sand filter (Svahn and Borg, 2024). With the ^{14}C method that was applied in the experiments, biodegradation can be detected only from the formation of CO_2 from the labeled moiety, and no other transformations would be noticeable, potentially explaining the diverging results between the full-scale sand filter and the incubation experiments.

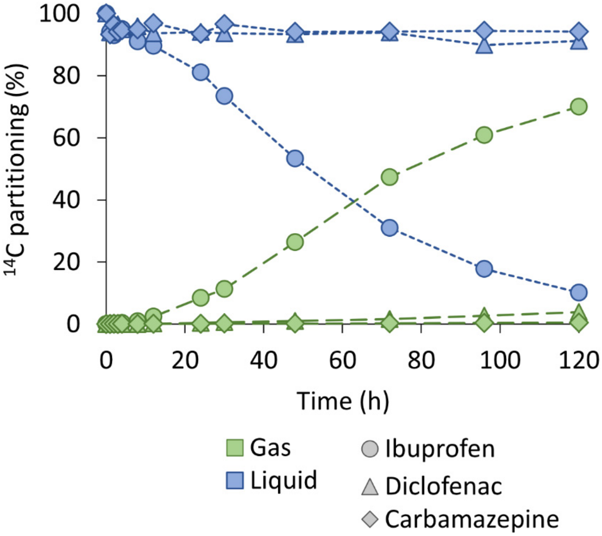


Figure 26. ^{14}C partitioning between the liquid (i.e., wastewater) and gas (i.e., CO_2 trap) phases in incubation experiments with media from the sand filter at Degeberga WWTP (Paper III).

5.3.3 Microbial communities in GAC and sand filters

Several factors affect the microbial communities in biofilters, such as the type of filter media (Piras et al., 2022; Sauter et al., 2023; Vignola et al., 2018), the upstream wastewater treatment (Piras et al., 2022; Torresi et al., 2018), and operation time (Paper I).

The PCoA clustering suggests that, despite differences in filter material (e.g., GAC versus sand) and time of sampling (i.e., number of BVs), the bacterial communities in filters at the same WWTP were more similar to each other (the dots are closer to each other in the PCoA plot) than those in filters at different WWTPs (the dots are further apart) (Figure 27). This pattern indicates that the WWTP the GAC or sand was collected from has a greater impact on the bacterial community than the type of filter media and operation time, supporting that the upstream wastewater treatment, and thus the composition of the influent water, affects the microbial community in downstream biological treatment processes.

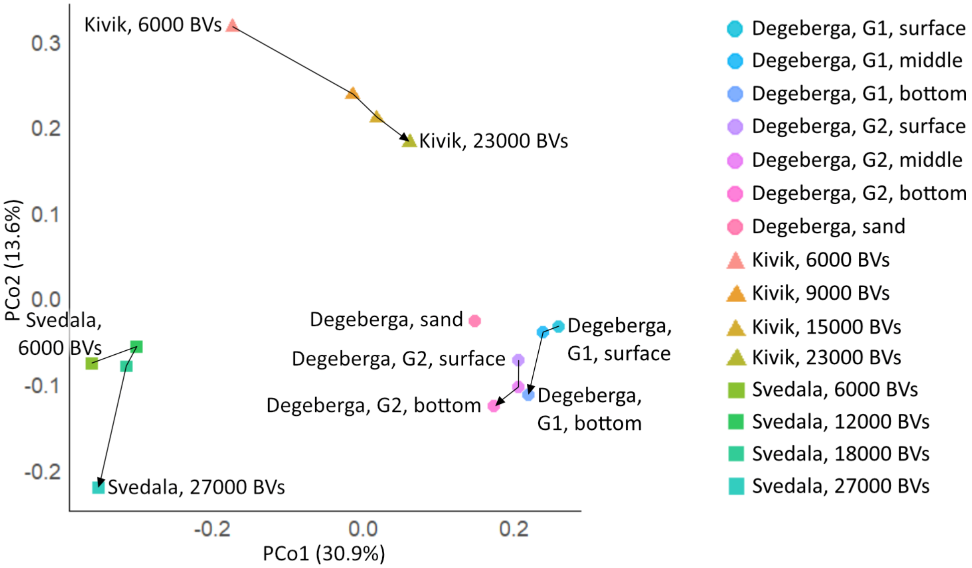


Figure 27. Principal coordinate analysis of bacterial communities on the GAC and sand media from five full-scale and pilot-scale filters at three WWTPs (Paper III). G1: bitumen-based GAC; G2: coconut-based GAC.

Although the degradation of diclofenac and ibuprofen differed considerably between filters with different types of media (high rate and magnitude of the degradation of diclofenac with the bitumen-based GAC versus no degradation of diclofenac with sand; Figure 25), the bacterial communities were relatively similar, as indicated by the PCoA (Figure 27). Conversely, the bacterial communities in the Kivik and Degeberga GAC filters were relatively different, but diclofenac was degraded to a high extent with media from both filters.

Adsorption of the organic micropollutants to GAC media, but not to sand, could potentially influence their degradation, and thus potentially explain the higher degradation of diclofenac with the GAC versus the sand media from Degeberga WWTP. However, the low degradation observed in the incubations with Svedala GAC, despite adsorption taking place, indicates that not all types of microbes in a GAC filter degrade diclofenac and that the filter conditions, such as oxygen availability, must be favorable for their growth. Moreover, despite similar adsorption rates, the degradation of diclofenac with the Kivik GAC media (operated with high oxygen concentrations) increased considerably between the second (9000 BVs) and third (15 000 BVs) samplings, suggesting the growth or selection of microbes that can degrade diclofenac.

5.4 Complexities of GAC filters and implications for wastewater reuse

GAC filtration affects the water quality, for example, through the adsorption of compounds to the GAC and through the biofilm growth on the granules. The biofilm decreases the adsorption rate and adsorptive capacity of the GAC and thus the treatment efficiency. It is, however, probably impossible to avoid biofilm growth in such an environment; thus, the biofilm growth is a factor that we likely need to accept. The biofilm in GAC filters can also have beneficial effects on wastewater treatment, due to a selection of certain bacteria in the biofilm, which likely contributes to the removal of some bacteria and the degradation of some organic contaminants.

Although the adsorptive capacity of GAC filters generally decreases over time, due to the blockage of pores and adsorption sites, the results from the study of Kivik WWTP showed a different trend: the removal declined until approximately 15 000 BVs, after which it increased. We confirmed the biological degradation of diclofenac and ibuprofen with GAC media from the filter (Paper III), likely explaining the greater removal of diclofenac but not, for example, carbamazepine. Fluctuating concentrations of the compound in the filter influent, varying EBCTs, and the potential bioregeneration of the filter are factors that could also have affected the removal.

The biofilm continues to develop over long periods (potentially for the entire operation life of the filter) and can shift the bacterial community in the water, raising concerns regarding the growth of opportunistic pathogenic bacteria in GAC filters. However, none of the indicator bacteria that were studied in this thesis increased in concentration; instead, lower concentrations were often measured in the GAC effluents versus influents. Microbial parameters, such as antibiotic-resistant bacteria and viruses were, however, not analyzed.

It is also apparent that not all GAC filters behave similarly—operation time, upstream wastewater treatment, oxygen concentration, and GAC material can affect the filter capacity. Thus, knowledge of the full-scale conditions of several WWTPs could enhance the possibility for process optimization. Oxygen concentration, for example, seems to be an important parameter for improving the prerequisites for micropollutant degradation, and upstream wastewater treatment processes seem to somewhat affect the removal of microbial contaminants by GAC filters. The removal of indicator bacteria was higher by the GAC filter downstream of an MBR process compared with that downstream of a CAS + sand filter, although little removal of microbial contaminants generally occurs in GAC filters, requiring additional processes (such as membranes or UV disinfection) for substantial removal.

6 Storylines and imaginaries of water

The novelty of wastewater reuse and desalination in Sweden, paralleled by its recent implementation in some regions, has created interesting cases for the study of their discourses. The discourses of the new desalination and reuse plants on Öland and Gotland were examined through interviews and a media analysis (Paper IV). Local storylines and imaginaries were identified and contrasted with general ones identified from the literature, described below.

6.1 General discourse

Water infrastructure—such as large-scale dams or irrigation projects—has, particularly during the 20th century, sometimes been connected to national visions of modernization, in which societal problems are solved through centralized and state-driven development and through engineering of nature to meet human needs (Flaminio, 2021; Randle and Barnes, 2018; Swyngedouw, 2014, 2013). These perspectives have been challenged due to ecological and environmental concerns and to growing quests for regional autonomy; in this context, desalination has sometimes been described as being decentralized and democratic and as a means of increasing local autonomy (Flaminio, 2021; Swyngedouw, 2014, 2013).

State-driven visions can also be challenged by other prioritizations than those based on ecological sustainability, such as economic development. In Maharashtra, India, the aspiration toward a better life and upward social and economic mobility was described as a local community imaginary that legitimized overexploitation of water resources for cultivating water-intensive crops (Argade and Narayanan, 2019), which disputed a governmental project that promoted a sustainable use of water resources.

6.1.1 Perceptions of wastewater reuse and desalination

The following sections specifically describes the perceptions of wastewater reuse and desalination, summarized from the literature.

Sustainability

Wastewater reuse is often perceived as being environmentally friendly, especially compared with desalination, and can be associated with sustainability values, such as recirculation and reuse (Abdelrahman et al., 2020; Dolnicar and Schäfer, 2009; Williams, 2022). It generally uses less energy than desalination, the latter of which is thus considered more expensive (Garin et al., 2021; Harris-Lovett et al., 2015; Heck et al., 2018; Hou et al., 2020; López-Ruiz et al., 2021). Another environmental concern that is related to desalination is the release of brine (concentrated salt water) to the aquatic environment (Williams, 2022).

Legal aspects

Compared with desalination, wastewater reuse may on the other hand be perceived as being legally complicated, since legislation on water quality, responsibility, and ownership sometimes is lacking (Fuenfschilling and Truffer, 2016; Haldar et al., 2021; Lee and Jepson, 2020; Williams, 2022).

Risk

Wastewater reuse can be perceived as risky or simply disgusting. The perceived risks are affected by the trust in the water utilities and institutions that govern water production (Hartley, 2006; Peters and Goberdhan, 2016; Smith et al., 2018), and its acceptance can be increased with good communication, community education, and information programs (Hou et al., 2020; Nkhoma et al., 2021; Wade et al., 2021). Awareness of droughts and water scarcity can also promote the acceptance of wastewater reuse, despite cases of public opposition during droughts, such as in Toowoomba, Australia (Garcia-Cuerva et al., 2016; Segura et al., 2018; Smith et al., 2018).

The concerns over wastewater reuse tend to decrease with decreased body contact—i.e., the acceptance is likely to be higher if the water is reused for house cleaning, toilet flushing, or irrigation, compared with drinking and cooking (Abdelrahman et al., 2020; Akpan et al., 2020; Baghapour et al., 2017; Chfadi et al., 2021; Flint and Koci, 2021; Garcia-Cuerva et al., 2016; Peters and Goberdhan, 2016; Segura et al., 2018).

Desalination is sometimes described as a flexible means of providing a stable water supply (Liu et al., 2022). Conversely, desalinated water can be perceived as unhealthy, due to its low mineral content (Shlezinger et al., 2018; Spungen et al., 2013) or deterioration of the aquatic environment (the source water for desalination) from eutrophication or oil spills, for example (Heck et al., 2016).

Economy

Wastewater reuse and desalination could provide a new water source in regions that suffer from water scarcity and can be viewed as economically beneficial by contributing to general water security, improving agricultural yields, and promoting industrial development that would otherwise not have been possible (Akpan et al., 2020; Williams, 2022). A connection between desalination and certain industries is described in the literature—for example, mining and tourism that are supplied with desalinated water, and desalination plants that are constructed in coastal cities to compensate for the extraction of groundwater caused by mining activities (Williams, 2022). The economic growth of the water treatment industry itself can also be a driver or perceived benefit of wastewater reuse and desalination (Williams, 2022).

6.1.2 General storylines and imaginaries of water

Based on the arguments and views above, three general imaginaries concerning water management and infrastructure and six storylines concerning wastewater reuse and desalination were identified (Figure 28).

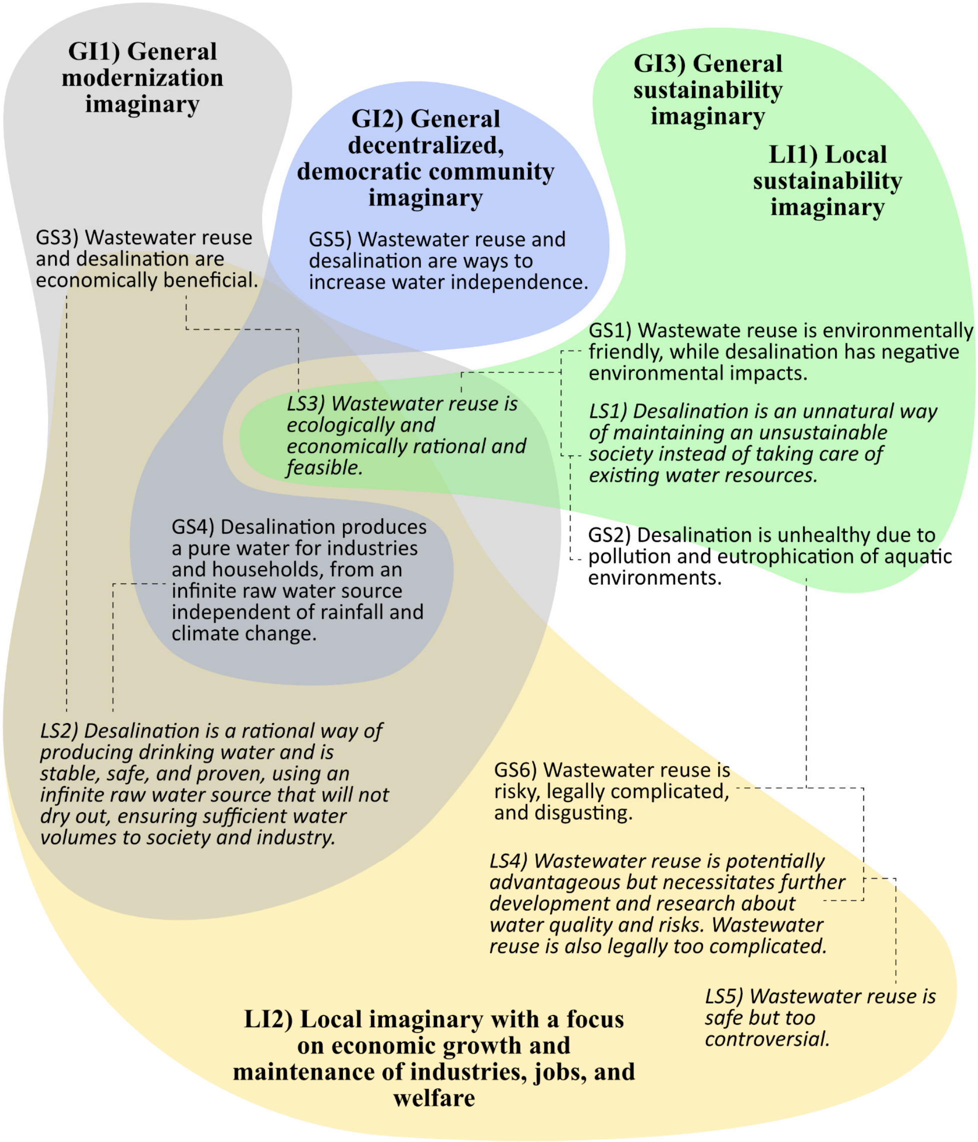


Figure 28. Visualization of general and local imaginaries and storylines. Connections or overlaps between storylines are indicated by a dashed line. The local discourse is italicized. GI: General imaginary, LI: Local imaginary, GS: General storyline, LS: Local storyline.

The following general imaginaries were identified:

1) The first imaginary, the "modernization imaginary," is based on the ambition and aspiration toward a better and more modern life and society. It can be large-scale and centralized, as well as local and individual, and can

legitimize a negative environmental impact and an overexploitation of water resources.

2) The second imaginary, the “decentralized, democratic community imaginary,” is focused on local rights and responsibilities, and can oppose large-scale, centralized modernization imaginaries. Desalination is described as decentralized and democratic, and as a means of achieving regional independence.
3) The third imaginary, the “sustainability imaginary,” is based on a vision of an ecologically sustainable future and society. Similar to the “decentralized, democratic community imaginary,” it can oppose large-scale modernization imaginaries (for example, large dam projects), if these result in a negative environmental impact.

These imaginaries in various aspects relate to the storylines on wastewater reuse and desalination, described below:

1) The first storyline argues that wastewater reuse is environmentally friendly, whereas desalination is expensive and has adverse environmental effects, due to high energy demand and harm of the aquatic environment. This storyline relates to the third imaginary (the sustainability imaginary) with regard to prioritizing ecological sustainability.
2) In the second storyline, desalinated water is described as unhealthy, due to its extreme purity and lack of minerals or due to its insufficient purity, resulting from pollution or eutrophication of the aquatic environment, which affects the source water for the desalination. This storyline overlaps with the sustainability imaginary, both of which highlight the pollution and eutrophication of aquatic environments, and shares the distrust towards desalination with storyline 1.
3) The third storyline characterizes wastewater reuse and desalination as economically necessary and advantageous, given the importance of water for local businesses and society and the financial revenue of the water sector. This storyline relates to the first imaginary, regarding modernization and an aspiration toward a better future.
4) The fourth storyline describes desalination as being advantageous, because it is a stable and proven technology that produces pure water for households and industries from an infinite raw water source that is not restricted by rainfall or climate change. Its stability and low risk can legitimize a negative environmental impact and higher costs. Components of this storyline relate to the second imaginary (the decentralized, democratic community imaginary) and its focus on water independence. The possibility of producing water to maintain or enable growth of industries and society is

also valued, and in this regard the storyline could also relate to the modernization imaginary.

5) In the fifth storyline, wastewater reuse and desalination are viewed as means of increasing water independence and, thus, as local, decentralized, and democratic. This storyline relates to the second imaginary—the decentralized, democratic community imaginary.
6) In the sixth and last general storyline, wastewater reuse is perceived as risky, legally complicated, and controversial. Desalination is considered safe, easy to gain acceptance for, and as more compatible with current legislation, avoiding issues regarding water ownership and water quality requirements. With its focus on the legal situation, risk, and controversy, this storyline is not considered to be linked to any of the three identified imaginaries above, but it has some connection to storyline number 2, with regard to the focus on health risks of the water. The storyline could also be divided into two, one focusing on the legal situation and one on risk and controversy.

The storylines and imaginaries are not static, and neither are the connections between them. Although general storylines and imaginaries have been identified here, local ones can differ, as discussed for the cases in Sweden.

6.2 Local discourse

Sweden is a country with few implementations of desalination and wastewater reuse and has in general not experienced water scarcity historically. Nevertheless, the water accessibility varies across the country due to, for example, differences in climate. A survey study in a Swedish town (Knivsta) with good access to freshwater revealed a high acceptance (> 90%) of wastewater reuse for irrigation or toilet flushing but lower rates (~40%) for potable reuse (Gullberg et al., 2023). In contrast to Knivsta, the islands of Öland and Gotland have recently experienced water shortage and rapid implementation of desalination and wastewater reuse, resulting in one municipal combined direct potable reuse and desalination plant and three municipal desalination plants (Paper IV). The combined reuse and desalination plant is located in Mörbylånga (Öland) and uses brackish water from the Baltic Sea and wastewater from the poultry industry as raw water (Figure 2). The desalination part was commissioned in 2019, and the reuse part was commissioned in 2021. The desalination plants are located in Kvarnåkershamn and Herrvik on Gotland (commissioned in 2016 and 2019, respectively) and in Sandvik on Öland (commissioned in 2017) and use brackish water from the Baltic Sea as raw water (Figure 2).

6.2.1 Motives for wastewater reuse and desalination in Sweden

Water scarcity and droughts were mentioned as reasons for building the Swedish wastewater reuse and desalination plants. The winters of 2015/2016 and 2016/2017 were dry, as was the summer of 2018, which gained media attention, as evidenced by the number of results from a search of newspaper and debate articles in local media (search string "drinking water" + name of respective town) (Figure 29). Information campaigns and limitations on irrigation were instituted on both islands, and tanker trucks sometimes delivered drinking water to Öland from the mainland, creating broad awareness of the water shortage in local society.

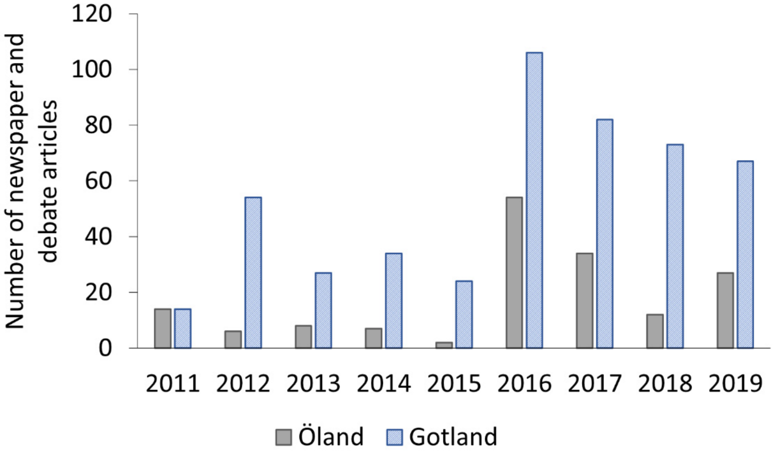


Figure 29. Number of results from media search on drinking water on Öland and Gotland (Paper IV).

Eventually, the formation of a local development company—which resulted in the first of the four plants, the Herrvik desalination plant—was motivated by the threat of a school closure, due to the decrease in population that was caused in part by the difficult water situation. Öland and Gotland are rural regions in which agriculture and tourism are important businesses, as are lime mining on Gotland and the poultry industry in Mörbylånga on Öland. Because it requires groundwater to be pumped into the Baltic Sea, lime mining on Gotland was mentioned in the interviews and media as a topic that was linked to the discussion on drinking water. This topic was described as controversial, and there had been previous conflicts regarding the expansion of mining activities and its potential effects on freshwater resources, such as the Ojnare case (Anshelm et al., 2018). Consequently, desalination was introduced into an already controversial discussion.

6.2.2 The desalination discourse

Desalination was suggested as a stable and proven means of producing drinking water to maintain industries, jobs and general welfare when the groundwater and lake water resources were not sufficient. It was highlighted that the desalination of water from the Baltic Sea is less energy-demanding and cheaper compared with desalination of water from, for example, the Mediterranean, due to the lower salt concentrations in the Baltic Sea.

Desalination was included in the municipal water plan in 2014 (Region Gotland, 2014) but received criticism for being expensive, energy-demanding, and unsustainable and for creating "dead water." Later, desalination was linked to mining in debate posts and described as a means of enabling more mining, among others, by Kingfors (2018) in Gotlands Allehanda:

> "At the same time, Cementa and Nordkalk are allowed to pump out groundwater to the Baltic Sea – when there is an irrigation ban and other people are told to save water. How is the acute need of water met? Through building desalination plants at many places on the island."

Desalination was sometimes described as unnatural and as a technological solution, whereas lake water and groundwater were viewed as natural and thus better, for example, by Heilborn and Wanneby (2014) in Gotlands Tidningar:

> "For Region Gotland, pure, natural water is a prerequisite for the future, even though you can technically succeed in producing distilled water from the Baltic Sea and get it classified as a drinkable."

Desalinated water was further sometimes considered less healthy than groundwater and surface water, due to its lower mineral content and sometimes due to a concern for harmful substances from the eutrophic and polluted Baltic Sea. Restoration of natural environments, such as previously drained wetlands, to increase groundwater levels was frequently broached as a more sustainable alternative to desalination. This was also mentioned as part of a wider vision of Gotland as a pioneer in sustainability:

> "It has been said, politically and structurally, that Gotland should be a pilot for renewable solutions and energy and things like that. I would like Gotland to be a pilot for sustainable water solutions as well." (Interview 8)

According to a water utility representative, the debate over desalination affected the development of the drinking water supply, through increasing the number of discussed alternatives, delaying the implementation of desalination plants, and resulting in fewer desalination plants:

> "Yes, to some extent the opponents were right. Desalination is more energy-demanding. We have advanced our understanding with these discussions, as well, and will look at this a bit differently in the future." (Interview 9)

6.2.3 The wastewater reuse discourse

Sustainability arguments affected also the discussion on wastewater reuse in Mörbylånga on Öland. The reuse of industrial wastewater was described as a sustainable approach to using resources while supplying the necessary water to industries and society. The drinking water that was produced from treated wastewater was perceived as easy to gain acceptance for:

> "It went much better [than we expected]; there was very little questioning of the water from the poultry industry." (Interview 3)

In Sandvik, on Öland, and on Gotland, where desalination was implemented, some water utility representatives considered wastewater reuse to be risky and the legal situation to sometimes be unclear. Others acknowledged that wastewater reuse is implemented worldwide and is a feasible method of producing drinking water but that it risks causing too much controversy. In contrast to wastewater reuse, desalination was viewed as an established and stable technology that was also legally less complicated than groundwater, because it does not affect the surrounding groundwater table and thus does not entail the need to establish a water protection area.

6.2.4 Local storylines and imaginaries

Visions, arguments, and opinions on wastewater reuse, desalination, and other alternatives, expressed in the interviews and in media, were grouped into storylines and imaginaries (Figure 28).

Two local imaginaries were identified: one that focused on ecological sustainability and one on economic growth and maintenance of industries, jobs, and welfare.

1) The first local imaginary focused on ecological sustainability and is similar to general imaginary 3, the "sustainability imaginary;" its aim is to create an ecologically sustainable future. Decreased pollution, climate mitigation, biodiversity, and conservation of freshwater resources should be achieved simultaneously—for example, through the restoration of wetlands. Certain industries, such as mining on Gotland, are criticized for being unsustainable.
2) The second imaginary, concentrating on economic growth and the maintenance of industries, jobs, and welfare, highlights the importance of

water for industries, businesses, population growth, and general welfare. Desalination is described as a feasible means of supplying society with sufficient volumes of good-quality water. Trust in technology as a solution to water scarcity is expressed, and sometimes a distrust in wastewater reuse, which is perceived as too risky, legally complicated, or controversial. In the distrust of wastewater reuse, the imaginary is related to general storyline 6, and with regard to economic growth and welfare, it also includes components from general storylines 3 and 4. The imaginary has similarities with general imaginary 1, the “modernization imaginary,” concerning trust in technology. However, it is not as focused on modernization, perhaps due to the Swedish context and because Sweden is generally considered modern and technologically developed.

On Gotland, the second imaginary conflicted with the first imaginary, wherein desalination was sometimes considered as enabling unsustainable industries and a means of solving water scarcity with technology instead of taking care of the natural systems. On Öland, components from the two imaginaries appeared to be more compatible. Industries, for example, were considered as being able to contribute to a more sustainable water supply through wastewater reuse, and at the same time to the maintenance of jobs.

In addition to the imaginaries above, five storylines were identified and compared with the general ones in Chapter 6.1.2.

1) In the first local storyline, desalination is described as an unnatural quick fix and an artificial method of maintaining an unsustainable society, instead of caring for existing freshwater resources. In the focus on ecological sustainability, it is related to the general and local sustainability imaginaries, and is similar to general storyline 1, associating desalination with a negative environmental impact. Distrust of technological and engineered solutions to sustainability problems is expressed, and desalination is criticized for its environmental impact and high costs. Desalinated water is also considered less healthy due to its low mineral content and due to the levels of harmful substances from the polluted Baltic Sea, as in general storyline 2. Natural processes, such as those in wetlands, are advocated.

2) The second storyline describes desalination as a rational method for drinking water production. Desalination is described as stable, safe, and proven, ensuring sufficient water volumes to society and industry. It relates to local imaginary 2, regarding its focus on the maintenance of industries, jobs, and welfare, and to general storylines 3, concerning the economic advantages for local businesses and society, and 4, describing desalination as stable and proven. Desalinated water is further considered safe, easy to gain acceptance for.

3) In the third storyline, wastewater reuse is depicted as ecologically and economically rational and feasible, because it enables a sufficient water supply to be provided to industries and society and concurrently recirculates limited freshwater resources. This storyline is similar to general storyline 1, associating desalination with a negative environmental impact and wastewater reuse with sustainability, and general storyline 3, describing wastewater reuse and desalination as economically advantageous. The ecological aspect of the storyline relates to the general and local sustainability imaginary. The economic focus relates to the general modernization imaginary and the local imaginary 2 that focuses on the maintenance of industries, jobs, and welfare.
4) The fourth storyline describes wastewater reuse as potentially advantageous but as an endeavor that necessitates further development and research on water quality and risks. Wastewater reuse is also considered legally complicated. This storyline is thus similar to general storyline 6, describing wastewater reuse as risky and legally complicated, and it shares the distrust toward wastewater reuse with local imaginary 2.
5) The fifth storyline describes wastewater reuse as safe but controversial. With regard to its distrust of wastewater reuse, it has similarities with local imaginary 2, local storyline 4, and general storyline 6.

Most general storylines and imaginaries had counterparts in the local discourse (Figure 28). However, no equivalents to global imaginary 2 (the “decentralized community imaginary”) or storyline 5 (on water independence, decentralization, and democracy) were identified in the local discourse, potentially due to the Swedish context, wherein Sweden is a democracy that has experienced long periods of peace. The storylines and imaginaries are not static and can change over time and between contexts.

7 Conclusions

This thesis examined technological and social aspects of wastewater reuse. In the work that focused on social aspects, discourses on wastewater reuse and desalination, in terms of storylines and imaginaries, were identified. These discourses appeared to be affected by several aspects. For example, the discourse of desalination was sometimes linked to the mining industry, which influenced the discussion and acceptance of desalination on Gotland in Sweden. The discourse of wastewater reuse was also affected by its association with various political questions, such as sustainability. Therefore, one should avoid drawing assumptions too quickly and simplistically of how the debate and discussion will evolve but should pursue early and transparent communication.

Based on the work that focused on technological aspects (Papers I–III), several conclusions could be drawn regarding factors that affected the five GAC filters that were studied:

- With increasing operation times, the adsorptive capacity of the GAC filters decreased, the bacterial community on the granules shifted, and the capacity for micropollutant degradation rose.
- The rate and magnitude of the biological degradation of certain organic micropollutants were higher in the GAC filters that were operated with high oxygen concentrations.
- Diclofenac was degraded with media from various GAC filters that were operated with high oxygen concentrations (DO > 8 mg/L), despite relatively large differences between the bacterial communities.
- Despite differences in the type of filter media (bitumen-based GAC, coconut-based GAC, sand), the bacterial communities on the media from filters at the same WWTP were relatively similar. Nevertheless, the biological degradation of certain organic micropollutants differed substantially: the rate and magnitude of the degradation of diclofenac, measured as $^{14}CO_2$ formation, were highest in the incubations with bitumen-based GAC from Degeberga WWTP, followed by the coconut-based GAC, while no degradation could be observed with sand filter media from this facility.

- The *E. coli*, TSS, and turbidity levels in municipal wastewater that was treated with an MBR followed by GAC filtration met the criteria for the irrigation of crops per quality class B standards of Regulation (EU) 2020/741.
- GAC filtration contributed marginally to the removal of microbial contaminants. The treatment processes that preceded the studied GAC filters (MBR with UF vs CAS followed by sand filtration) lowered the concentrations of microbial indicators to varying extents (with the highest removal observed in the MBR) and seemed to affect the removal of indicator bacteria in the downstream GAC filters. The GAC filter that was preceded by an MBR with UF removed indicator bacteria to a slightly higher extent than the GAC filter that was preceded by CAS and sand filtration.
- The concentrations of indicator bacteria in the MBR + GAC + UV effluent (UV fluence: 400 J/m^2) met the criteria for microbial drinking water quality per Swedish standards. In one selected sample, chemical drinking water criteria were also fulfilled, with the exception of a too high nitrate concentration.

Limits on many organic micropollutants, such as pharmaceuticals, are lacking in the legislation for irrigation and drinking water. These compounds are often present in higher concentrations in wastewater than in groundwater and surface water, and the implementation of potable reuse or wastewater reuse for irrigation will require the regulation of these compounds to be evaluated. These compounds were removed to a high degree in the MBR + GAC process, but WWTPs that are not equipped with GAC or other treatment processes to remove organic micropollutants are unlikely to attain such removal rates.

Although insecurities remain regarding the safe levels of certain compounds, such as pharmaceuticals, it is clear that the MBR + GAC + UV effluent nearly reaches drinking water quality, and the water could be a beneficial complementary water resource in times of drought or water shortage.

8 Future work

GAC filtration affect microbial and chemical water quality in various ways. Full-scale filters can behave differently than laboratory-scale and pilot-scale filters, necessitating long-term studies of more GAC filters for wastewater treatment. Viruses and antibiotic-resistant bacteria in the filter effluents must be investigated to ensure safe reuse of the treated water. Also, the toxicity of organic micropollutants, such as pharmaceuticals and their transformation products, should be further investigated, as should the need for limits on such compounds in wastewater that is reused for irrigation and drinking water production.

Further, to fully understand the long-term capacity of GAC filters, longer full-scale studies are needed. The results described in Paper I indicate that treatment capacity can increase over time, in contrast to the general expectation of decreased contaminant removal. This increased removal could have resulted from variations in influent concentrations or EBCT, from bioregeneration of the GAC filter, or from biological degradation of certain compounds. Degradation of some pharmaceuticals was studied in laboratory experiments, but the degradation of additional organic micropollutants beyond those in this thesis should also be examined.

GAC filters with different upstream wastewater treatment processes were compared (Papers II and III), and the results suggested a small effect on the removal of microbial contaminants but not on the degradation of organic micropollutants. To optimize microbial and chemical removal and prolong the regeneration intervals of the GAC, the effects of the upstream treatment processes on GAC function should be further evaluated.

Much research exists on the attitudes toward wastewater reuse and on factors that affect these. However, few studies have been performed in Nordic settings or in contexts without water scarcity. Sweden, with its varying climate and freshwater availability, would be an interesting subject to examine the attitudes toward wastewater reuse. However, the responses in a survey study will not necessarily mirror actual reactions to the implementation of wastewater reuse. Future potential implementations of municipal wastewater reuse in Sweden would provide an interesting opportunity for follow-up studies on attitudes and perceptions toward wastewater reuse.

The combination of technological and social perspectives in this thesis offers a nuanced picture of wastewater reuse. In the second part of the thesis (based on Paper

IV), the diversity of views, opinions and perceptions of wastewater reuse is discussed. This is not acknowledged in the first part of the thesis (based on Paper I, II and III), which mainly regards technological aspects. The second part thus adds important perspectives to the first part.

Further, combining experiences from different disciplines may effect research objectives that would otherwise not have been thought of, since researchers based in, for example, engineering generally have other types of experiences and knowledge of the water sector than social scientists. Therefore, collaborations between researchers from different disciplines could expand various research areas, and simultaneously increase the understanding between people with different backgrounds.

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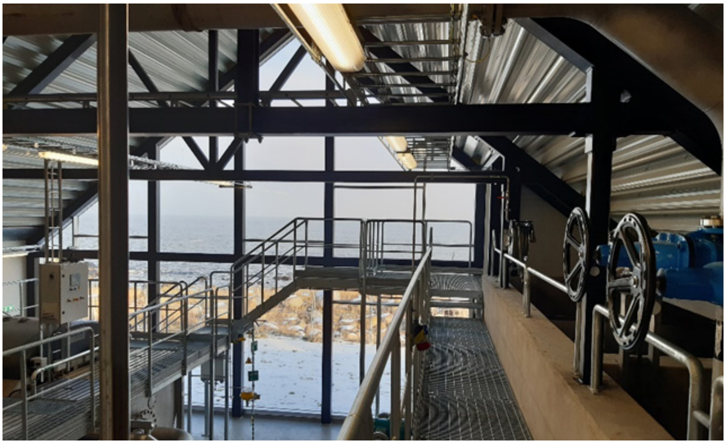

View from Kivik wastewater treatment plant.

The implementation of wastewater treatment for the removal of organic micropollutants at Swedish wastewater treatment plants, paralleled with an increased pressure on freshwater resources and changing precipitation patterns, have generated an interest in wastewater reuse in Sweden, where it has previously not been common practice. GAC filtration is one method to remove organic micropollutants. In this thesis, several aspects of GAC filtration were studied and the implications for wastewater reuse were evaluated. Since wastewater reuse is not only a question of technology, perceptions of wastewater reuse were also studied through interviews and a media analysis.

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