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Onsite wastewater treatment by multi-module biochar filters (MmBF)

A technical assessment with institutional insights

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Abstract
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Onsite wastewater treatment systems (OWTS) are vital in rural and peri-urban areas, but conventional designs such as septic tanks with soil infiltration often provide insufficient treatment. This contributes to eutrophication, groundwater contamination, and the spread of emerging micropollutants. To address these challenges, this thesis investigates a biochar-based multi-module filter system (MmBF) and examines the institutional factors influencing the adoption of innovative OWTS technologies in Sweden.

Each of the triplicate MmBFs configurations comprised six sequential modules with aerobic and anoxic conditions to promote organic matter oxidation, nitrification, denitrification, and faecal bacteria attenuation. The MmBFs achieved consistently high removal, with chemical oxygen demand reduced by about 95%, inorganic nitrogen by 71%, and Escherichia coli by 1.8–2.3 log10. Pharmaceutical active compounds were also effectively treated, with overall concentrations reduced by more than 99%, most within the first aerobic stage. Phosphate removal remained limited, indicating a need for design improvements. To improve nutrient removal, sawdust and Polonite® were amended at 10% and 30% in two of the MmBFs, each applied to separate modules. The 30% sawdust amendment enhanced nitrate reduction by up to 83% for a short term, while 30% Polonite® achieved short-term phosphate retention of 31 ± 19%. Both effects declined over time, suggesting the need for periodic media replacement.

A qualitative study complemented the technical evaluation by examining the socio-institutional context of OWTS transitions. Interviews with municipal officers, regulators, developers, and knowledge intermediaries, analysed through a Multi-Level Perspective framework, showed that regulatory ambiguity, fragmented responsibilities, and risk-averse permitting hinder adoption. At the same time, enabling factors were identified, including reference demonstration units, stronger municipal engagement, and more accessible knowledge-sharing platforms. While developers and researchers drive technical innovation, municipalities and national authorities retain decisive roles in approval and diffusion.

The thesis concludes that advancing decentralised wastewater treatment in Sweden requires both technical and institutional progress. For biochar-based systems, further work should address media replacement, installation design, and potential investigations for nutrient recovery, to support market-ready products. At the institutional level, municipalities and national authorities must collaborate with innovators to create transparent approval pathways and strengthen legitimacy for novel systems.

Keywords: onsite wastewater treatment, biochar, multi-module biochar filter (MmBF), nitrogen, nitrification, denitrification, phosphorus, polonite, pharmaceutically active compounds (PhACs), socio-technical transition, qualitative analysis, multi-level perspective, innovation, Sweden.

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List of Papers

This thesis is based on the following papers, which are referred to in the text by their Roman numerals.

I. Shigei, M., Herbert, R.B., Persson, F., Sokolova, E., Dalahmeh, S.S. (2024). Efficient removal of organic matter and nitrogen from municipal wastewater in multi-module biochar filters for onsite wastewater treatment. *Environmental Technology* 1–13. https://doi.org/10.1080/09593330.2024.2402096

II. Shigei, M., Esfahani, B., Persson, F., Dalahmeh, S.S. (2025). Removal of pharmaceutical active compounds in multi-module biochar filter (MmBF) for post-septic tank treatment. *Water Science and Technology.* https://doi.org/10.2166/wst.2025.111

III. Shigei, M., Liu, Z., Herbert, R.B., Persson, F., Dalahmeh, S.S. (2025). Effect of sawdust and polonite addition to Multi-module biochar filters for the enhanced removal of nitrate and phosphorus. *Manuscript*

IV. Shigei, M., Karpouzoglou, T., Herbert, R.B., Dalahmeh, S.S. (2025). Institutional and technological transitions in onsite wastewater treatment − Exploring the role of municipalities and innovators in Sweden. *Manuscript*

Author contributions

The CRediT taxonomy[1] was used to define author roles for each paper. "Responsible" indicates that I (Makoto Shigei) carried out the role and had primary responsibility for it. "Shared" indicates a jointly undertaken role among all co-authors of the respective paper.

For **Paper II**, instrumental analysis was conducted by Bahare Esfahani. For **Paper III**, shaking tests for carbon release and the column experiment data were provided by Zhongxing Liu.

Paper	**I**	**II**	**III**	**IV**
Conceptualisation	Shared	Shared	Shared	Shared
Formal analysis	**Responsible**	**Responsible**	**Responsible**	**Responsible**
Investigation	**Responsible**	Shared	Shared	**Responsible**
Visualisation	**Responsible**	**Responsible**	**Responsible**	**Responsible**
Writing – original draft	**Responsible**	**Responsible**	**Responsible**	**Responsible**
Writing – review & editing	Shared	Shared	Shared	Shared

[1] Contributor Roles Taxonomy. Available at: https://credit.niso.org/

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Abbreviations

Biochar	Carbonised biomass via pyrolysis
CFU	Colony-forming unit
COD	Chemical oxygen demand
DO	Dissolved oxygen
E. coli	*Escherichia coli*
HLR	Hydraulic loading rate
HRT	Hydraulic retention time
MLP	Multi-level perspective
MmBF	Multi-module biochar filter
NH_4-N	Ammonium nitrogen (NH_4^+-N)
NO_3-N	Nitrate nitrogen (NO_3^--N)
NO_2-N	Nitrite nitrogen (NO_2^--N)
OWTS	Onsite wastewater treatment system
PhACs	Pharmaceutically active compounds
PO_4-P	Phosphate phosphorus (PO_4^{3-}-P)
Tot-N	Total nitrogen
WWTP	Wastewater treatment plant

Introduction

Sanitation infrastructure is essential for protecting human health and the environment. Through activities such as bathing, dishwashing, laundry, and toilet use, households generate wastewater that must be effectively treated to prevent pollution and reduce risks to both people and ecosystems. At the same time, population growth and intensified human activity are placing increasing pressure on water and sanitation systems (Ferdowsi et al., 2024). The global population reached 8 billion in November 2022 and is projected to expand to 9.7 billion by 2050 (United Nations, 2022). Climate change, with more frequent droughts, floods, and temperature extremes, is adding further strain on existing water management systems (Ferdowsi et al., 2024; Gosling and Arnell, 2016).

The weaknesses of prevailing sanitation systems are becoming more apparent worldwide. These systems may rely on centralised sewerage networks in urban areas or on basic facilities such as pit latrines and septic tanks in rural and peri-urban settings. In many low-income countries, inadequate wastewater treatment infrastructure remains a critical challenge (Montgomery and Elimelech, 2007). Rapid population growth, urbanisation, and limited financial and technical resources have resulted in widespread reliance on basic or unimproved sanitation. As of 2024, more than 3.4 billion people lacked access to safely managed sanitation services, and 419 million continued to practice open defecation (UNICEF and WHO, 2023; United Nations, 2025). While large-scale centralised sewerage systems can be effective in dense and resource-rich settings, they are often infeasible in rural and peri-urban areas. In such contexts, decentralised or onsite wastewater treatment systems (OWTSs) are increasingly recognised for their adaptability and cost-effectiveness (Capodaglio, 2017). Bernal et al., (2021) also argue that decentralised systems are not only viable but necessary to reduce environmental impacts, enable resource recovery, and expand access to sanitation in rural and peri-urban communities.

In high-income countries, including Sweden, which is the focus of this thesis, aging infrastructure is under pressure from demographic changes and climate extremes. Many centralised wastewater systems were designed decades ago under assumptions of continued growth and stability (OECD, 2015). Today, population decline and ageing demographics, which reduce the tax base,

challenge the viability of maintaining extensive sewer networks (Hedström et al., 2016). Climate-related events such as droughts, floods, and fluctuating precipitation are increasingly pushing these systems beyond their design capacity (Bernal et al., 2021; Ferdowsi et al., 2024; Svenskt Vatten, 2023). Over 90% of costs in centralised systems are associated with sewer network construction and maintenance (Garrido-Baserba et al., 2024). As the population served by these systems declines and fewer users contribute to operational funding, the cost per user rises, making the systems less financially sustainable. In northern Sweden, for example, many municipal utilities built in the 1960s–1970s now face depopulation and limited financial and technical resources (Hedström et al., 2016). In such settings, decentralised systems can reduce strain on existing infrastructure, minimise the need for expensive sewer extensions through modular system designs, and create opportunities for resource recovery (Garrido-Baserba et al., 2024).

Decentralised systems, often implemented as onsite wastewater treatment systems (OWTSs), can compartmentalise risk, as failure in one unit affects only a small number of users (Bernal et al., 2021). The conventional OWTS, typically a septic tank with soil infiltration, is the most widespread design in rural Sweden and elsewhere (Laukka et al., 2022). However, when OWTSs are deployed at scale, their limitations become apparent, including variable effluent quality and the potential for environmental pollution compared to centralised systems with tertiary treatment. For instance, a survey of 395 OWTSs in Finland and Sweden found that over 25% exceeded permitted phosphorus limits, and 10% exceeded nitrogen discharge thresholds (Kinnunen et al., 2023). Inadequate nutrient removal can harm aquatic ecosystems, while insufficient pathogen removal can contaminate groundwater and pose health risks by affecting drinking water wells (Schaider et al., 2016; Swedish EPA, 2022). These shortcomings explain why centralised systems remain the standard for pollution control. Nonetheless, OWTSs can be adapted in design and operation to match local soil types, water conditions, and community requirements. (Bernal et al., 2021).

One approach to improve OWTS performance is the integration of biochar as a filtration medium. Biochar is a carbon-rich material derived from biomass through thermal conversion, and has traditionally been used in agriculture as a soil amendment (Lehmann and Joseph, 2009). More recently, it has gained attention in environmental engineering for its dual role as a waste-to-resource product and functional adsorbent (He et al., 2022a). In water treatment, biochar offers a low-cost, sustainable alternative to sand- or soil-based media (Ahmad et al., 2014; Bimová et al., 2021). Its high porosity, large surface area, and diverse functional groups (e.g., hydroxyl, carboxyl, amino) enable removal of pollutants through adsorption and ion exchange (Bolan et al., 2023; Xu et al., 2020). Laboratory tests have shown its efficacy for emerging micropollutants as well (Purkait, 2022; Späth et al., 2021). Dalahmeh et al. (2018) reported >98% removal of carbamazepine in biochar filters, compared

to about 7% in sand under similar conditions. In addition to these physicochemical properties, biochar promotes biofilm formation on its surface, facilitating processes such as nitrification and denitrification and thereby functioning as a hybrid medium for both physical and biological treatment (Bolan et al., 2023). Beyond treatment performance, biochar contributes to circular economy goals. Produced from forestry or agricultural waste, it provides a pathway for valorising residual biomass. After use, nutrient-laden biochar could be re-applied as fertiliser, returning phosphorus and nitrogen to soils (Marcińczyk et al., 2022). Biochar also sequesters carbon in stable forms, contributing to climate change mitigation (He et al., 2022a; Karan et al., 2023). These co-benefits position biochar as an attractive component in resilient and innovative OWTS design, aligning with environmental protection and resource recovery objectives.

Meanwhile, transitioning from conventional to innovative systems involves institutional and cultural challenges. Conventional infrastructure is supported by decades of regulatory, technical, and social norms (Garrido-Baserba et al., 2024). Policies and standards often do not accommodate small-scale innovations, and decision-makers tend to favour established solutions. For such transitions to succeed, enabling policies and demonstrable pilot cases are essential. Hacker and Binz (2021) found that onsite water reuse diffusion in San Francisco required local actors to navigate institutional demands by adapting roles and building supportive structures. Similarly, Särkilahti et al. (2017) identified integrated planning and public acceptance as critical for scaling up source-separation technologies in Finland. In Sweden, McConville et al. (2017) observed that low legitimacy and fragmented support networks were major barriers to adopting source-separation systems.

In summary, the convergence of aging infrastructure and climate stress is prompting a rethinking of wastewater management in Sweden and other high-income countries. While centralised systems offer robust treatment, they may be less suited to shrinking or remote communities. Decentralised OWTSs can provide a flexible and resilient alternative, particularly when combined with innovations such as biochar filtration. This thesis investigates how such systems perform in meeting modern treatment standards and what institutional and governance factors are required to support their broader adoption.

Aim and objectives

The overall aim of this thesis is to advance our understanding of the technical performance of biochar-based treatment systems for OWTSs, and to explore how such novel technologies could be supported and adopted within society.

For the investigation of technical performance, the studies focus on evaluating the removal of organic matter, nutrients (phosphorus, P, and nitrogen, N), *Escherichia coli (E. coli)* as an indicator of faecal bacteria, and pharmaceutically active compounds (PhACs) using a multi-module biochar filter (MmBF) system. Although the MmBF system was installed and operated at a municipal wastewater treatment plant for pilot-scale experiments, it was designed for OWTS application to treat septic tank effluent. The MmBF operation was continued for over four years, with data collected during three distinct sampling campaigns, forming the basis of **Papers I, II, and III**.

Complementing the technical focus, this thesis also investigates stakeholder interactions that influence the adoption and implementation of new technologies for OWTSs in Sweden, with particular attention to the interplay between municipalities and technology developers.
The three main objectives of the thesis are as follows:

1. Investigating the performance of the MmBFs in removing organic matter, nutrients, *E. coli*, and PhACs from municipal wastewater. (**Papers I and II)**

2. Evaluating an updated module configuration of the MmBFs for enhanced denitrification and phosphorus removal. (**Paper III**)

3. Exploring institutional and regulatory factors that affect the adoption of new technologies for onsite wastewater treatment. (**Paper IV**)

These objectives are addressed through a combination of pilot-scale experiments (**Papers I–III**) and a qualitative interview study (**Paper IV**). A summary of the individual studies and their methodological approaches is provided below and illustrated in **Figure 1**.

Paper I evaluates the removal of organic matter, nitrogen, phosphorus, and *E. coli*, in a pilot-scale MmBF system using influent-effluent sample analysis including a tracer test to determine a hydraulic retention time of each func-

tional unit of the MmBF. **Paper II** extends this work by examining the efficiency of the MmBFs to remove different types of PhACs in municipal wastewater, analysing 25 substances through high-performance liquid chromatography coupled with mass spectrometry (HPLC-MS/MS). **Paper III** investigates enhancements to the MmBFs configuration by incorporating amended filter media to improve nitrogen and phosphorus removal. This involved both batch tests for selecting effective carbon sources, and experiments with updated MmBFs under varied operational conditions. **Paper IV** shifts focus to the socio-institutional dimension, exploring the regulatory and institutional factors influencing the adoption of new OWTS technologies. This qualitative study is based on semi-structured interviews with key stakeholders and applies the Multi-Level Perspective (MLP) framework for analysis.

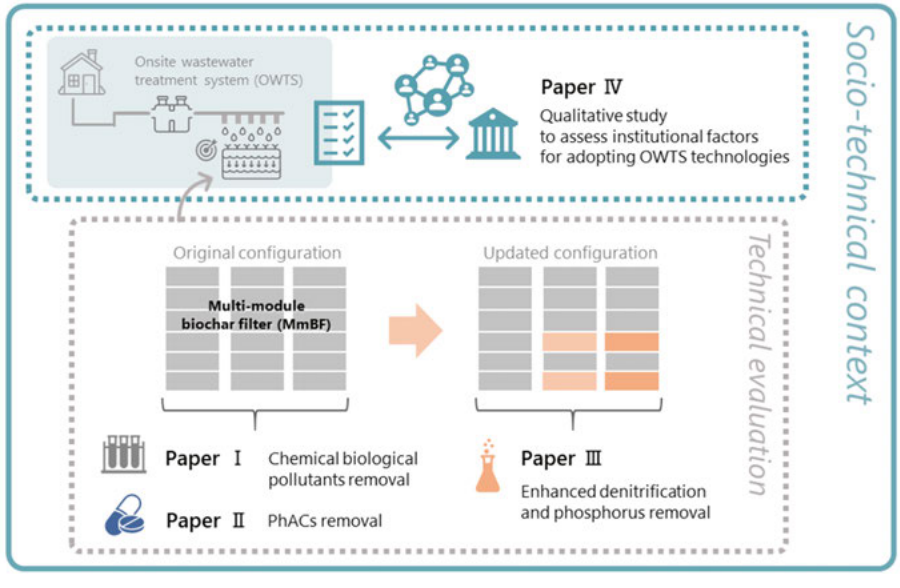

Figure 1. Conceptual overview of the thesis, integrating technical evaluation of the multi-module biochar filter (MmBF) system (**Papers I, II, and III)** and socio-technical context related to the adoption of new technologies for OWTS (**Paper IV**).

Background

From pollution control to circular sanitation: defining OWTS roles

The terms *decentralised* and *onsite* wastewater treatment system (OWTS) are often used interchangeably. In this thesis, OWTS is defined as a subset of decentralised systems that treat wastewater at or near its point of generation, typically at the household level (Figure 2). Hereafter, the term OWTS is used to refer specifically to such technical units, whereas decentralised is used for the broader societal and infrastructural context. In Swedish practice, decentralised water and wastewater systems are sometimes described as local facilities serving from a single household up to 2,000-person equivalents (PE), regardless of ownership or management model (VA-guiden, 2025). This usage is consistent with the upper threshold for agglomerations in the EU Urban Waste Water Treatment Directive (91/271/EEC) (European Parliament and Council, 2024), above which centralised collection and treatment obligations apply. In Sweden, OWTS is legally classified as *enskilt avlopp* and refers specifically to systems serving fewer than 200 PE (SwAM, 2016).

In the context of OWTS, three interrelated functional goals can be distinguished (Figure 2). The first is pollution control, which focuses on protecting human health and the environment by meeting discharge standards. Representative technologies include conventional system such as septic tanks, soil infiltrations and package treatment plants. In this thesis, the pilot-scale evaluation of the MmBF is considered within this goal, as the primary focus is on its pollutant removal performance. The second is water reclamation and reuse, which aims to produce effluent suitable for non-potable applications such as irrigation, toilet flushing, or industrial use, thereby contributing to local water conservation (Kesari et al., 2021). Examples include membrane filtration systems, sand filters, and constructed wetlands (Ventura et al., 2024). The third is resource recovery, which seeks to recover energy and nutrients (P/N) for use as fertiliser or other resources, contributing to a circular economy; examples include urine-diverting toilets, anaerobic digestion, and struvite precipitation (Aliahmad et al., 2023; Almansa et al., 2023; Taddeo et al., 2018). While the Figure 2 assigns each technology to the goal it most directly supports, many of these systems have the potential to contribute to multiple goals. For example, a constructed wetland designed for nutrient removal may also produce water suitable for reuse.

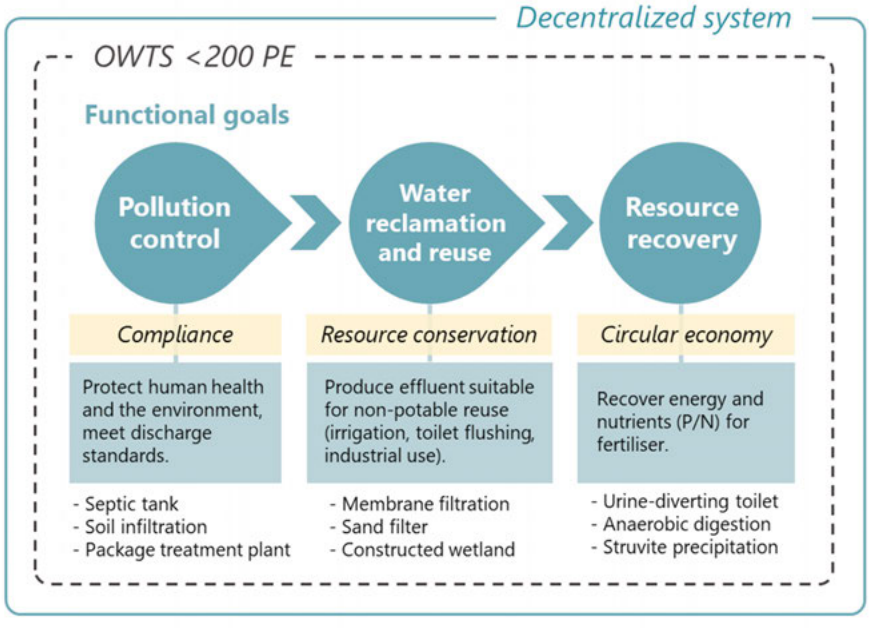

Figure 2. Functional goals of decentralised treatment systems. The three goals of pollution control, water reclamation and reuse, and resource recovery, are interrelated, with pollutant removal forming the foundation. Example technologies could cover more than one goal.

These categories provide a useful basis for categorising both conventional and emerging OWTS technologies. In the following sections, the discussion focuses first on the treatment processes and performance by conventional OWTSs, then it will discuss the application of biochar in wastewater treatment.

Pollutant removal in conventional OWTS

OWTSs are commonly used worldwide, particularly in rural and peri-urban areas that are not connected to centralised sewers. In high-income countries, such as the United States, Australia, and parts of Europe, between 15–30% of households rely on OWTSs (Beal et al., 2005; US EPA, 2002; Vedachalam et al., 2015). System design and performance are strongly influenced by local climate and hydrogeological conditions. This thesis focuses on Sweden as a representative case of a high-income country with a predominantly temperate climate.

OWTS numbers and requirements in Sweden

In Sweden, there are approximately 860,000 properties with an OWTS, and studies have estimated that more than half of these systems do not comply with national environmental regulations (Laukka et al., 2022; Statistics Sweden, 2023; Swedish EPA, 2022). Many older houses in sparsely populated areas and smaller communities have inadequate sewage treatment facilities, often consisting only of sludge separation in a septic tank. Houses built in the last 30 years are generally equipped with additional purification methods and therefore meet the requirements for treatment (Table 1) beyond sludge separation (Swedish EPA, 2008). This treatment requirement for BOD, Tot-N, and Tot-P is given by the Swedish agency for marine and water management (SwAM) for either normal and high protection levels which are determined by the environmental/geographical condition of the site (SwAM, 2016). According to Laukka et al. (2022), the most common type of OWTS in Sweden is a septic tank with soil infiltration (33%), followed by a septic tank without further treatment (27%) and a septic tank with a sand filter (16%).

Table 1. Treatment standard for BOD, Tot-P, and Tot-N in normal and high protection area, from the OWTS guidance by SwAM (2016)

	Normal protection level	High protection level
BOD_7	90% removal, 30 mg L^{-1}	90% removal, 30 mg L^{-1}
Tot-P	70% removal, 3mg L^{-1}	90% removal, 1 mg L^{-1}
Tot-N	-	50% removal, 40 mg L^{-1}

Characteristics of influent wastewater

Raw wastewater (influent) to OWTS, primarily originating from small residential settings, is also described in the literature as domestic wastewater. In this context, it refers to the combined flow of urine and faeces from toilets together with wastewater from bathrooms, kitchen sinks, dishwashers, hand basins, and laundry facilities (Lowe et al., 2009). Pollutants in the domestic wastewater include organic matter indicated by biological oxygen demand (BOD) and chemical oxygen demand (COD), nutrients (N/P), and substances like pharmaceuticals and personal care products (PPCPs) (Agoro et al., 2018; Munavalli et al., 2022; Pishgar et al., 2021; Zhou et al., 2023).

Nitrogen in domestic wastewater occurs in both inorganic and organic forms. Inorganic nitrogen includes ammonia (NH_3), ammonium (NH_4^+), nitrite (NO_2^-), nitrate (NO_3^-), and dinitrogen gas (N_2), while organic nitrogen consists mainly of amino acids, proteins, nucleic acids, and urea. The balance between NH_3 and NH_4^+ is governed by pH, with the proportion of free NH_3 increasing at higher pH levels (Rout et al., 2021). Nitrite is relatively unstable and readily oxidised to nitrate under aerobic conditions, whereas organic nitrogen is mineralised to NH_3 through microbial processes.

Phosphorus occurs as both particulate and dissolved forms, including orthophosphate (PO_4^{3-}), hydrogen phosphate (HPO_4^{2-}), dihydrogen phosphate ($H_2PO_4^-$), phosphoric acid (H_3PO_4), polyphosphates, and organically bound compounds such as phospholipids and phosphoproteins. The dominant phosphate species in solution depends on pH: $H_2PO_4^-$ predominates under mildly acidic to near-neutral conditions (pH ~4–6.5), whereas HPO_4^{2-} is more abundant under alkaline conditions (pH ~7.5–12). Polyphosphates hydrolyse to orthophosphate, which is the primary bioavailable form for aquatic organisms (Rout et al., 2021).

Although most microorganisms in domestic wastewater are non-pathogenic, a variety of pathogens may also be present, including bacteria (e.g., *E. coli*, *Salmonella*), viruses (e.g., hepatitis A, norovirus), and helminths such as hookworm. Heavy metals such as lead (Pb), mercury (Hg), and cadmium (Cd) are also present in domestic wastewater, largely due to detergents and body care products. Common PPCPs detected include ibuprofen, caffeine, acetaminophen, and carbamazepine (Clyde et al., 2021). These compounds differ in solubility and hydrophobicity, which influences their fate in treatment systems. For example, carbamazepine has a low aqueous solubility of about 17 mg L^{-1} and moderate hydrophobicity, with $\log K_{ow}$ 2.45, while ibuprofen is more hydrophobic, with a $\log K_{ow}$ of around 3.97 and slightly more soluble at 21 mg L^{-1}. Caffeine, in contrast, is highly soluble in water (>20 g L^{-1}) and has a low $\log K_{ow}$ (−0.07), making it more mobile in aquatic environments. Concentrations of PPCPs in domestic wastewater can range from tens of ng L^{-1} to several µg L^{-1}, depending on compound type and household consumption patterns (Clyde et al., 2021; Muoghalu et al., 2023). Influent quality can vary substantially with region, residents' age, household activities, time, with raw wastewater generally showing higher concentration ranges than septic tank effluent (Lowe et al., 2009; Shaikh and Ahammed, 2020). Table 2 presents example concentrations of physico-chemical and biological pollutants in domestic wastewater, based on literature values.

After primary treatment in a septic tank, the resulting septic tank effluent still contains measurable amounts of pollutants, although solids and BOD are reduced compared with raw wastewater. Reported concentrations typically range from 49–84 mg L^{-1} for total suspended solids (TSS), 201–944 mg L^{-1} for COD, 44–833 mg L^{-1} for BOD_5, 27–119 mg L^{-1} for Tot-N, and 0.2–33 mg L^{-1} for Tot-P (Lowe et al., 2009). Trace organic contaminants such as caffeine, triclosan, ibuprofen, and naproxen often remain at levels comparable to those in raw wastewater. The discharge flow from the septic tank is largely governed by household water use patterns (Lowe et al., 2009).

Table 2. Concentration ranges and mean ± standard deviation values of physico-chemical and biological parameters in domestic wastewater reported in literature. Superscripts: a = BOD_3; b = Total Kjeldahl nitrogen; c = Tot-P. A dash (–) indicates no data reported.

TSS (mg L^{-1})	COD (mg L^{-1})	BOD_5 (mg L^{-1})	NH_4^+-N (mg L^{-1})	Tot-N (mg L^{-1})	PO_4^{3-}-P (mg L^{-1})	Source
22-1690	139-4584	112-1102	1.6-94	9-240	0.2-32[c]	Lowe et al. (2009)
-	70-300	45-200	15-100	20-100	-	Zhou et al. (2023)
1653 ± 1174	567 ± 181	1160 ± 350	99 ± 19	100 ± 56	39 ± 28	Pishgar et al. (2021)
-	17-394	-	0.06-112	-	0.02-5.12	Agoro et al. (2018)
220 ± 15	298 ± 47	160 ± 25[a]	-	60 ± 4[b]	-	Munavalli et al. (2022)

Treatment mechanisms of septic tank with soil infiltration

Septic tanks followed by soil infiltration (Figure 3) are the most common OWTS in Sweden and other high-income countries (Laukka et al., 2022; US EPA, 2002). This configuration combines a buried, watertight sedimentation unit (septic tank) with a soil infiltration field, both connected to households for wastewater collection and treatment (Beal et al., 2005). In the septic tank, solids settle to form sludge, while fats, oils, and grease rise to form scum. Anaerobic microorganisms degrade the organic matter within the sludge, scum, and liquid phases (Toor et al., 2020; US EPA, 2002). Effluent then flows to the infiltration field, where it percolates through the soil. During this passage, pollutant removal occurs through adsorption, microbial activity, chemical precipitation, and pathogen die-off. Sedimentation units require periodic maintenance, including pumping to remove accumulated sludge and scum, which in Sweden is, typically transported to large-scale wastewater treatment facilities for energy recovery and nutrient recycling.

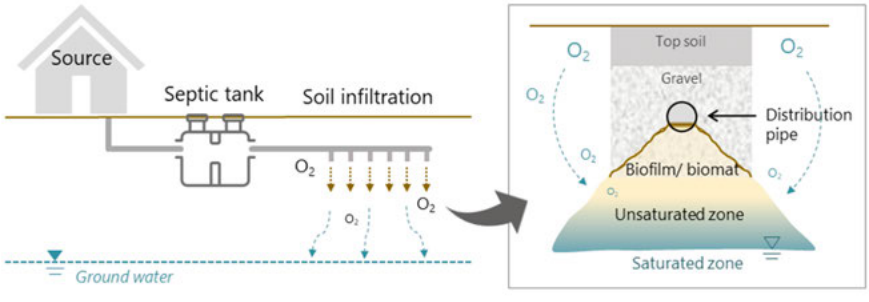

Figure 3. Schematic and cross-sectional view of a septic tank with soil infiltration system.

The treatment process in a soil infiltration field is similar to that of a biological wastewater treatment plant. Figure 3 and Table 3 provide an overview of pollutant removal mechanisms, while Table 4 presents a more detailed account of processes and challenges. Bacteria colonise soil surfaces, forming a biofilm (biomat) beneath and around infiltration trenches (Criado Monleon et al., 2022). This biofilm layer plays a central in removing soluble organic matter, particulates, and pathogens (Sehar et al., 2016). However, if treatment performance is insufficient, nearby wells and surface waters may become contaminated (Schaider et al., 2016). When the septic tank is functioning properly, the majority of solids are retained in the tank, and the infiltration field's primary function is to further reduce remaining pathogens and nutrients to environmentally safe levels.

Oxygen availability in the treatment zone is a key determinant of system performance (Criado Monleon et al., 2022). Sufficient oxygen supports aerobic bacteria, which degrade organic matter and produce CO_2, while oxygen-limited zones enable anaerobic processes that are essential for denitrification. However, if anaerobic conditions dominate, slower-growing microbes can produce methane and form dense biofilms, increasing the risk of clogging and reduced infiltration (Gerba et al., 2023).

Table 3. The treatment mechanisms occurring in soil infiltration system. OM: organic matter, SS: suspended solids.

Location	Treatment mechanism	Removal of:
Top soil	Mechanical Filtration	Large particles and debris
biofilm	Biological Degradation	OM, N, PPCPs, bacteria
unsaturated zone	Settling	Heavier particles
unsaturated zone	Sorption/Adsorption	P, N, PPCPs, and PhACs
Throughout the infiltration	Physical Filtration	SS, particulate matter
Throughout the infiltration	Chemical Precipitation	P and N
Throughout the infiltration	Dilution	Bacteria and chemicals

Domestic wastewater typically contains 73% organic nitrogen (organic-N) and 24% ammonium (Lowe et al., 2009). Most organic-N is mineralised to ammonium in the septic tank (Heatwole and McCray, 2007). Nitrification converts ammonium to nitrate in aerobic zones, usually within the upper 30 cm of unsaturated soil (Laak, 1982). Nitrate can leach to groundwater or be denitrified to nitrogen gas under anoxic, saturated conditions in deeper layers (> 60 cm), provided suitable electron donors such as organic carbon or sulphur (Laak, 1982). Thus, the balance between aerobic nitrification near the surface and denitrification in deeper, anoxic zone is central to nitrogen removal in soil infiltration systems.

Phosphorus in wastewater occurs mainly as orthophosphate species (e.g. phosphate, hydrogen phosphate), with smaller fractions of polyphosphate and organic forms. Around 85% of septic tank effluent phosphorus is present as orthophosphate and only 20–30% is removed by settling in the septic tank (Heatwole and McCray, 2007; Lowe et al., 2009). In the infiltration field, organic phosphorus is mineralised to orthophosphate, which is readily available for plant uptake. Leaching of phosphate can be reduced through adsorption onto aluminium (Al), iron (Fe), and manganese oxides (e.g., MnO_2) and hydroxides (e.g., $Mn(OH)_3$), as well as clay minerals (Brady and Weil, 2016). Sorption capacity is limited by competition with organic acids, which can also form chelates with Al and Fe, reducing their availability for phosphate binding. Organic carbon can immobilise phosphate in organic-bound forms, although organic soils are generally less effective for phosphorus removal (Brady and Weil, 2016).

PPCPs in domestic wastewater are only partly removed during septic tank passage, mainly through sorption to settled solids under anaerobic conditions, with removal efficiency varying according to compound hydrophobicity (Conn et al., 2010). In the subsequent unsaturated infiltration zone, PPCPs are further attenuated through a combination of sorption to soil organic matter and biofilms, and aerobic biodegradation by indigenous microorganisms. Readily degradable compounds such as caffeine often achieve > 90% removal within the first 60 cm of percolation, provided that unsaturated flow and adequate residence time are maintained. More persistent compounds, such as ethylenediaminetetraacetic acid (EDTA, a chelating agent commonly used in detergents and personal care products), show limited removal due to low sorption affinity and resistance to biodegradation, certain transformation products, such as nonylphenol from surfactants, may appear at intermediate depths before further breakdown occurs (Conn et al., 2010).

The effectiveness of an OWTS also depends on correctly sizing the infiltration field according to soil type and wastewater volume, typically estimated from household occupancy. Soil permeability, usually assessed through a percolation test, is critical, with denser soils requiring larger fields to ensure adequate treatment and infiltration (US EPA, 2002). Where suitable soils are lacking or groundwater tables are high, alternative systems may be necessary, often at higher cost and with greater maintenance needs (Beal et al., 2005).

Table 4. A summary of the treatment processes identified in the literature, which are responsible for the removal of different pollutants in soil-based infiltration systems. Temp: temperature, HRT: hydraulic retention time, HLR: hydraulic loading rate, DO: dissolved oxygen, OM: organic matter, UV: ultra violet light.

Type of pollutants	Treatment process	Factors affecting the treatment	Challenges/ problems	Sources
TSS, particulate solids	filtration and straining	particle size, wastewater volume, pH, temp, HRT	clogging, biofouling, hydraulic overloading, soil type	Al-Mawla et al. (2023)
Organic matter	mineralization, microbial oxidation or degradation	wastewater composition, BOD, COD, HRT, pH, temp	clogging, biofouling, hydraulic overloading, soil type	Al-Mawla et al. (2023) Saeed and Jihad Miah (2021)
NH_4^+-N	oxidation, biological assimilation in biofilm, adsorption	influent concentration, pH, alkalinity, temp, DO, HRT, carbon source, biomass concentration	low pH, high influent concentration, limited cation exchange capacity, low microbial activity, cold temperature	Oakley et al. (2010) Wigginton et al. (2018)
NO_3^--N	denitrification	influent concentration, pH, temp, DO, carbon source, biomass concentration	high influent concentration, low microbial activity, low organic carbon content, high DO	Wigginton et al. (2018)
Tot-N	nitrification and denitrification	influent concentration, pH, alkalinity, temp, DO, HRT, carbon source, biomass concentration	high influent concentration, low cation exchange capacity, low microbial activity, low organic carbon content, inadequate RT, lack of oxygen, cold temperature	Wigginton et al. (2018)

PO_4^{3-}-P	adsorption, Biological assimilation, chemical coagulation	influent concentration, pH, coexisting ion, chemical coagulants	high influent concentration, limited phosphate sorption capacity, low microbial activity, low pH, limited cations, competition with organic acids	Akunna et al. (2017) Rout et al. (2021)
Tot-P (including particulate)	sedimentation of particulate	influent concentration, pH, coexisting ion, chemical coagulants	high influent concentration, limited phosphate sorption capacity, low microbial activity, low pH, limited cations	Brady and Weil (2016) Rout et al. (2021)
Bacteria	sedimentation, filtration, natural die-off, UV radiation, Ozone, Chlorination	temp, moisture content, pH, HRT, HLR, presence of OM, bacterial characteristics	inadequate RT, limited filtration, low microbial activity, low-oxygen condition, limited organic carbon, cold temperature	Akunna et al. (2017) Kadir and Nelson, (2014)
PPCPs	adsorption onto soil OM and biofilm, aerobic and anaerobic biodegradation, transformation to intermediate products	compound properties (e.g., hydrophobicity, biodegradability), soil texture and organic matter content, redox conditions, pH, temp, HRT, HLR, microbial community composition	variable removal efficiency between compounds, persistence of poorly degradable compounds, formation of more toxic transformation products (e.g., nonylphenol from surfactants), breakthrough under high loading rates or reduced unsaturated flow	Conn et al. (2010) Ejhed et al. (2018)

Removal efficiency in OWTSs

Table 5 summarises findings from recent literature on the performance of OWTS in temperate-climate countries. Removal of organic matter is typically high, with reductions of > 92% (based on BOD reduction) in systems combining septic tanks or sand filters with soil infiltration fields and, in some cases, phosphorus (P) filters (Laaksonen et al., 2017; Pishgar et al., 2021; Vidal et al., 2018). P filters are additional treatment units installed after septic tanks to enhance phosphorus removal, commonly using Polonite®, an engineered silica–calcite material derived from opoka (a natural silica–calcite rock) with high phosphorus-binding capacity (Vidal et al., 2018).

OWTS effectively remove organic matter through filtration and biological degradation. Pishgar et al. (2021) reported median removal efficiencies of 94% for BOD_5 and 93% for TSS in retrofitted septic tanks equipped with SludgeHammer®, an air-driven column with a microbubble diffuser that promotes aerobic bacterial growth. This retrofit enhanced organic matter removal prior to effluent discharge to the infiltration field and reduced TSS concentrations (195 ± 206 mg L^{-1}), thereby lowering the risk of clogging.

Nitrogen removal varies considerably across OWTS types (Laaksonen et al., 2017; Pishgar et al., 2021). In a 10-month monitoring study, Laaksonen et al. (2017) found that sand filter systems achieved effective nitrification but struggled with overall nitrogen reduction due to a lack of anoxic zones and insufficient carbon for denitrification. Phosphorus removal in these system ranged from 68–90%, whereas nitrogen removal remained limited (Laaksonen et al., 2017), performing similarly to aerobic treatment units but less effectively than municipal wastewater treatment plants (WWTPs) (Garcia et al., 2013).

Herrmann et al. (2016) examined phosphorus and bacterial removal in rural Swedish OWTSs. Sand filters generally reduced total phosphorus, but performance was sometimes limited by dilution effects. P filters showed variable results, with some achieving high levels of phosphorus reduction. Vidal et al. (2018) found that most sand filters failed to meet phosphorus standards due to clogging and dilution, whereas six of nine P filters performed well. They also observed a significant correlation between phosphorus removal and filtrate pH, suggesting pH may serve as an indicator for media replacement needs (Vidal et al., 2018).

Conventional OWTSs primarily target solids, organic matter, and nutrients, but pathogen removal is also important. Sedimentation removes some pathogens (Ergas et al., 2021), while filtration provides physical straining, with finer media grains enhancing removal through greater surface area and adsorption potential (Jenkins et al., 2011). Natural die-off and predation further reduce microbial loads (Wang et al., 2021). Free-living ciliate protozoa,

for example, efficiently prey on bacteria (Wand et al., 2007). However, filtration alone may not fully remove pathogens, especially viruses and resistant forms such as bacterial spores or protozoan cysts.

Kauppinen et al. (2014) studied pilot-scale sand filters for microbial and nutrient removal in Finland. *E. coli* reduction ranged from 1.4–5.4 $\log_{10}$ depending on filter design and season. Sand filters enriched with biotite achieved the highest removal (5.0 ± 0.7 $\log_{10}$), likely due to reduced permeability, which increased contact time for microbial binding. Standard sand filters also performed well (4.3 ± 0.8 $\log_{10}$) but showed lower virus removal in winter (Kauppinen et al., 2014). Smaller sand filters initially had lower *E. coli* removal but improved after summer, suggesting biofilm maturation enhanced capture. Nutrient removal remained relatively stable across seasons.

To further reduce pathogens in OWTS effluents, disinfection units such as UV radiation, ozone, and chlorination can be applied (Boutilier et al., 2009; Sengupta et al., 2011). These methods substantially improve effluent quality and reduce health risks in reclaimed water (Kadir and Nelson, 2014; Nguyen et al., 2015).

Table 5. Treatment performance presented as removal (%) of BOD, TOC, N species, P, bacteria, viruses, and PPCPs in OWTS as reported in the literature. A dash (–) indicates no data reported.

Type of pollutants	Location	Type of OWTS	Influent concentration (mg L^{-1} if not stated)	Removal efficiencies	Sampling	Source
TSS	Canada	Septic tank + drain field	1653 ± 1174	92 ± 6%	n=9	Pishgar et al. (2021)
BOD_5	Canada	Septic tank	1160 ± 350	92 ± 5%	n=9	Pishgar et al. (2021)
BOD_7	Sweden	+ drain field	1.0-18.8	98%	n=9	Vidal et al. (2018)
BOD_7	Finland	P filters (polonite bag)	189	97-98%	n=18	Laaksonen (2017)
		Septic tank + sand filter				
TOC	Sweden	P filters (polonite bag)	8.2-39.8		n=9	Vidal et al. (2018)
NH_4-N	Canada	Septic tank + drain field	99 ± 19	55 ± 31%	n=9	Pishgar et al. (2021)
NO_3-N	Canada	Septic tank + drain field	1.28 ± 0.12	-	n=9	Pishgar et al. (2021)
Tot-N	Canada	Septic tank	100 ± 56	50 ± 25%	n=9	Pishgar et al. (2021)
	Finland	+ drain field	38-116 g day^{-1}	10-85%	n=9	Laaksonen (2017)
		Septic tank + sand filter				
Tot-P	Sweden	P filters (polonite bag)	1.6-14.8	10-98%	n=9	Vidal et al. (2018)
	Finland	Septic tank + sand filter	6-16 g day^{-1}	68-90%	n=25	Laaksonen (2017)

PO_4-P	Canada	Septic tank + drain field	39 ± 28	67 ± 28%	n=9	Pishgar et al. (2021)
Particulate P	Sweden	P filters (polonite bag)	0-0.8	13-98%	n=9	Vidal et al. (2018)
Bacteria						
Fecal coliforms	Minnesota, USA	Constructed wetland	5.48 log_{10}CFU/100 mL	1.6-1.7 log_{10}	n=2, winter	Pundsack et al. (2001)
Escherichia coli	Minnesota, USA	Sand filter	5.43 log_{10}CFU/100 mL	3.3-3.4 log_{10}	n=2, winter	Pundsack et al. (2001)
	Finland	Sedimentation tank + Sand filter	6.52 log_{10}CFU/100 mL	1.4-5.4 log_{10}	n=4	Kauppinen et al. (2014)
Viruses						
Norovirus genogroup II	Finland	Sedimentation tank + Sand filter	3.36 log_{10} genome copies mL^{-1}	0.8-3.5 log_{10}	n=4	Kauppinen et al. (2014)
PPCPs						
Diclofenac	Sweden	Package plants[a]	0.15-13 µg L^{-1}	< 10%	n=2	Vidal et al. (2023)
Ibuprofen	Sweden	Package plants[a]	9.6-69 µg L^{-1}	< 40%	n=2	Vidal et al. (2023)
	North Carolina, USA	Septic tank + drain field[b]	74 µg L^{-1}	> 79%	n=4	Del Rosario et al. (2014)

a. Trickling filter and activated sludge with phosphorus removal by coagulation, treating septic tank effluent.

b. Influent concentrations refer to septic tank samples; removal efficiency was calculated from reported influent and effluent values.

Challenges in OWTS and alternatives

Many infiltration systems act as diffuse sources of pollutants to the environment (Herren et al., 2021; Lapointe et al., 2017). In Sweden, more than half of OWTSs fail to meet current standards (Laukka et al., 2022), contributing to nutrient leaching that threatens the ecological health of the Baltic Sea (Hamisi et al., 2022; SwAM, 2016). Gao et al. (2019) detected organic contaminants, including pharmaceuticals, personal care products, plasticisers, and flame retardants, in groundwater affected by OWTS infiltration.

Climate pressures add further challenges. In recent years, prolonged dry summers have led to irrigation bans in Sweden (Svenskt Vatten, 2023). These recurring water shortages emphasise the need for sustainable alternative water sources. Properly treated OWTS effluent could be reused for irrigation, toilet flushing, and other non-potable applications (Kesari et al., 2021). To realise this potential, treatment systems must consistently produce effluent with low microbial loads, minimal P and N concentrations, particularly in environmentally sensitive areas, and negligible pharmaceutical residues (Kesari et al., 2021).

A further operational challenge is that pollutant removal in infiltration fields depends heavily on oxygen availability in the upper soil layers. Limited aeration can restrict nitrification, while deeper anoxic zones may not support complete denitrification, particularly when organic carbon is scarce (Criado Monleon et al., 2022). Seasonal changes in temperature and soil moisture can exacerbate these oxygen constraints, reducing overall treatment efficiency and contributing to nutrient leaching (Kauppinen et al., 2014).

The design and operation of current OWTSs are also be constrained by local geological conditions. Systems that are less dependent on soil characteristics would be safer and more widely applicable. As an alternative to soil-based filtration, biochar has gained attention for its versatility and pollutant removal capacity. Other engineered substrates, such as Polonite® have also shown promise for phosphorus removal in decentralised systems (Karczmarczyk et al., 2017). The following section discusses biochar's potential applications in OWTS.

Biochar-based wastewater treatment technologies

Source and properties of biochar

Biochar is a carbon–rich material produced through pyrolysis, the high–temperature thermal conversion of organic matter such as wood or agricultural residues, under limited oxygen conditions (He et al., 2022b). Its highly porous structure and large surface area provide strong sorption capacity for nutrients and pollutants. Typically, near–neutral in pH, biochar can improve soil structure, retain nutrients, and sequester carbon over long timescales (Chagas et al., 2022). Apart from its role as a soil amendment, biochar is increasingly applied

in environmental remediation, where it can remove organic and inorganic contaminants from soil and water (Enaime et al., 2020). Early research focused on its physicochemical properties, contaminant sorption, effects on soil productivity, and microbial interactions (summarised in Lehmann et al., 2015; Li and Chen, 2020). Since the 2010s, interest has expanded to include applications in water and wastewater treatment (He et al., 2022b; Wang et al., 2020; Xiang et al., 2020).

Beyond its treatment function, biochar used in onsite wastewater treatment can be part of a circular resource pathway (Figure 4): once saturated with nutrients, it can be recovered and applied to agricultural soils as a fertiliser or soil conditioner, thereby recycling phosphorus and nitrogen while retaining its carbon sequestration benefits (Marcińczyk et al., 2022). Such reuse not only diverts residual biomass from waste streams but also supports nutrient recovery goals in decentralised sanitation. However, the suitability of spent biochar for land application should be carefully evaluated with respect to the types and levels of pollutants it has adsorbed, to avoid transferring contaminants to soils or crops (Marcińczyk et al., 2022).

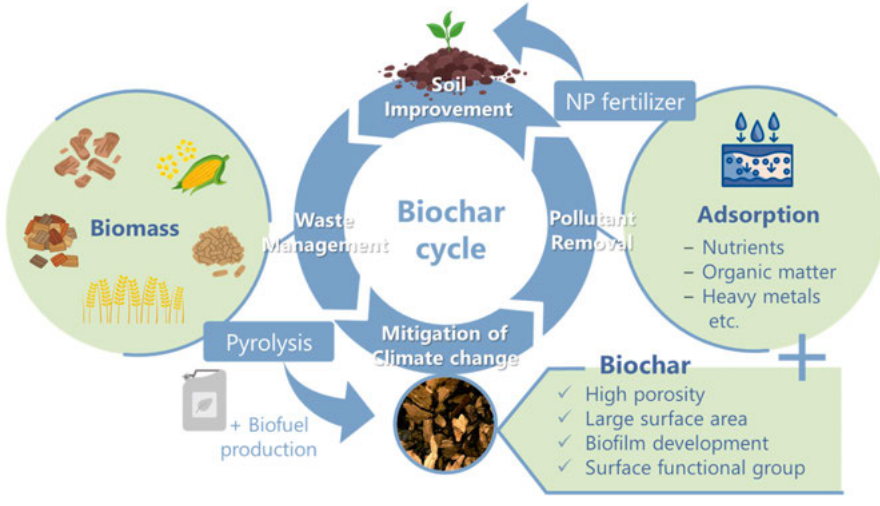

Figure 4. The benefits and environmental cycle of biochar applied as an effective adsorbent for wastewater treatment. Modified from Tan et al. (2015).

Biochar properties depend strongly on both feedstock and pyrolysis conditions. Ippolito et al. (2020) demonstrated that feedstock type has the greatest influence: wood–based biochar tends to exhibit the highest specific surface area, whereas crop– and grass–based biochar generally has higher cation exchange capacity, favouring nutrient retention. Pyrolysis temperature also affects composition and stability. Higher temperatures (> 600 °C) generate more

recalcitrant carbon with longer environmental persistence and potential to mitigate nitrous oxide emission, as such biochar can adsorb mineral nitrogen and promote more complete denitrification to N_2 (Liu et al., 2014; X. Zhang et al., 2023). By contrast, lower temperatures yield biochar with shorter lifespans. The type of pyrolysis (fast vs slow) appears to have less influence (Anand et al., 2023). Nutrient content also varies with feedstock: manures and biosolids produce biochar with higher concentrations of Tot-N, P, sulphur (S), calcium (Ca), and magnesium (Mg), though some nutrients volatilise at higher temperatures. Ash–associated elements such as Ca and P typically increase with increasing pyrolysis temperature (Ippolito et al., 2020).

Biochar performance can be enhanced through physical or chemical modification (Muoghalu et al., 2023). Physical activation methods such as steam or CO_2 treatment increase porosity and surface area (Kołtowski et al., 2017). Chemical activation introduces surface functional groups via acid, alkali, or oxidising treatments (Shen and Zhang, 2019). Impregnation with metals, metal oxides, hydroxides, or nanoparticles can enhance removal of specific contaminants (Dalahmeh et al., 2020; Liyanage et al., 2020). These modifications aim to increase adsorption capacity, tailored surface chemistry, or catalytic activity.

Biochar applications and treatment mechanisms

In onsite and municipal wastewater treatment, biochar filters have been shown to reduce COD, suspended solids, and nutrients (Almanassra et al., 2021; Manyuchi et al., 2018), while also supporting microbial biofilms development for nitrification and denitrification (Deng et al., 2021; Liang et al., 2020). In conventional OWTS, nitrogen removal is often constrained by insufficient anoxic zones and limited carbon availability for denitrification (Laaksonen et al., 2017). The Porous matrix of biochar can improve conditions by providing a large surface for microbial colonisation and retaining organic compounds, thereby enhancing coupled nitrification–denitrification processes (Dalahmeh et al., 2019).

Phosphorus, typically present as mobile orthophosphate in septic effluent, can be immobilised through electrostatic attraction, precipitation, and complexation with biochar's mineral phases. Modification with magnesium, calcium, aluminium, or iron substantially enhances phosphate binding, with capacities reported above 600 mg PO_4-P g^{-1} (He et al., 2022a; Jung and Ahn, 2016).

Biochar is also effective in removing PhACs such as carbamazepine, ibuprofen, and diclofenac, which are often poorly removed in soil infiltration systems. Removal occurs via hydrophobic interactions, hydrogen bonding, π–π interactions, and pore filling (Cheng et al., 2021). Lab studies have reported >90% removal of certain PhACs in biochar filters compared to <10% in sand (Dalahmeh et al., 2018).

While research on biochar modification and regeneration is extensive, these were not a focus of this thesis. Instead, the MmBF system evaluated here

focuses on long-term performance in removing organic matter, nitrogen, phosphorus, *E. coli*, and PhACs under operational conditions relevant to OWTS applications. Table 6 summarises the key factors affecting removal efficiency of biochar-amended OWTS.

Table 6. Summary of the key factors influencing the performance of biochar-amended OWTSs in removing different pollutants. Upward arrows (↑) indicate increased removal associated with that parameter, and vice versa. *N/A: information not available. pH_{pzc}: pH at the point of zero charge. Modified from Maleki Shahraki and Mao (2022).

Factors	**Nitrogen**	**Phosphorus**	**Pathogen**	**PPCP**
Biochar source	**Wood-based↑** due to H/C and organic FG and improved zeta potential	**Sewage sludge↑** Due to cation content	**Wood-based↑** Due to wooden pore size 5-10 μm	N/A*
Pyrolysis temperature High (HT): 550-900 °C Low (LT): 250-450 °C	**Contradictory effect**; Specific surface area (HT↑), Organic functional group (HT↓)	**Contradictory effect**; Ratio of cation elements (HT↑), Polarity and aromaticity (HT↓)	N/A*	**Contradictory effect**; Specific surface area (HT↑), Residual carbon for partitioning (HT↓)
Hydraulic loading	**Better performance at higher HL** compared to sand systems due to increase of water capacity	N/A*	**No impact or better removal at higher HL** compared to sand systems	N/A*
pH	**Biochar helps in better N removal** due to providing alkalinity and buffering capacity	**Better P adsorption at pH < 8.5** due to more positive surface charge of biochar	Depends on the pathogen, Better removal at pH < 8.0 due to more positive surface charge of biochar	Depends on the PPCP, maximum adsorption at pKa compound < solution pH < pH_{pzc} of biochar

Institutional and socio-technical dimensions of decentralised sanitation

Governance and institutional context of OWTS in Sweden

The governance of OWTS in Sweden developed alongside advances in centralised wastewater management (Figure 5). Large-scale, centralised sewer networks began to appear in the 1880s, serving major cities (Swedish EPA, 2022). Public concern over water pollution, particularly from eutrophication in the Baltic sea, grew during the 1960s, prompting the establishment of the Swedish Environmental Protection Agency (Swedish EPA) in 1967. Expansion of large wastewater treatment plants in the 1970s substantially improved effluent quality (Swedish EPA, 2022).

In parallel, decentralised systems expanded to meet the needs of sparsely populated areas. The introduction of septic tanks in the 1950s marked the beginning of modern OWTS in Sweden. Regulatory requirements were strengthened in 1998 with the introduction of mandatory secondary treatment (SFS 1998:899). In 2011, national responsibility for OWTS guidance shifted from the Swedish EPA, which had overseen both centralised and decentralised systems, to the Swedish Agency for Marine and Water Management (SwAM; *Havs- och vattenmyndigheten*, HaV). This change created an authority dedicated to OWTS, separate from the governance of centralised systems (Figure 5). In the following text, this agency is referred to as HaV.

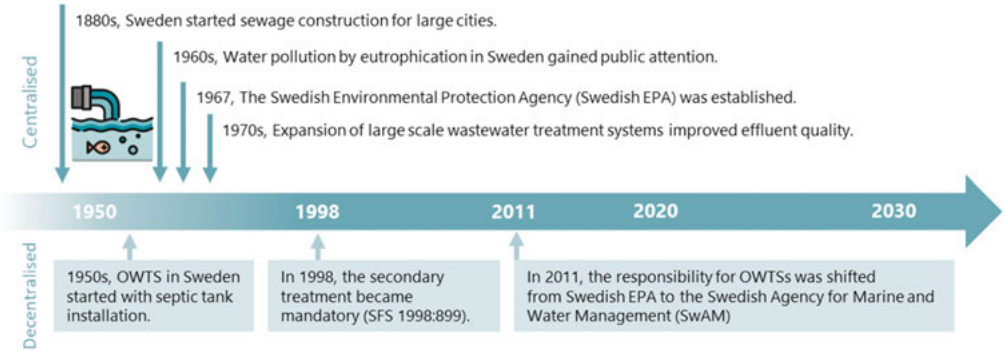

Figure 5. Historical development of centralised and decentralised wastewater governance in Sweden, showing key events influencing OWTS regulation and institutional responsibilities.

Today, municipal authorities hold primary responsibility for supervising and regulating OWTSs. They issue permits, conduct inspections, and enforce compliance with legal standards (Laukka et al., 2022). HaV supports these activities by providing national guidance, publishing recommendations, and

developing resources for local implementation (SwAM, 2016). County administrative boards (*länsstyrelserna*) function as regional supervisory bodies and serve as appeal authorities for municipal decisions.

OWTS operators are typically house owners, though some systems are jointly managed through wastewater associations. Operators are responsible for carrying out required operation and maintenance tasks in accordance with permit conditions, and many engage external service companies for technical servicing (Laukka et al., 2022). Governance practices vary among municipalities, resulting in differences in supervision, enforcement, and permitting procedures. Such variation can pose challenges in assessing and approving new treatment technologies, particularly where data on long-term performance is limited.

The legal framework for OWTSs includes the Swedish Environmental Code (*Miljöbalken*, SFS 1998:808; Regeringskansliet, 2000), the Ordinance Concerning Environmentally Hazardous Activities and Protection of Public Health (*Förordning 1998:899 om miljöfarlig verksamhet och hälsoskydd*), and HaV's general advice on onsite wastewater systems for domestic wastewater (*allmänna råd om små avloppsanordningar för hushållsspillvatten*, SwAM, 2016). Municipalities may impose additional requirements in high protection areas. Current policy priorities aim to accelerate the replacement of outdated systems and ensure compliance with environmental and public health objectives across the country (Laukka et al., 2022).

While this governance structure provides formal oversight, the adoption of sustainable OWTS innovations, those aiming not only for pollution control but also for water reuse or resource recovery, remains slow. Linking the governance context to the broader functional goals of OWTS (Figure 2) allows these ambitions to be examined through a socio-technical perspective that considers sanitation as an interplay of technologies, institutions, and actor networks.

Socio-technical transitions in sanitation systems

The socio-technical systems perspective emphasises that infrastructure transitions emerge from the co-evolution of technologies**,** institutions, and social practices (Geels, 2010). In urban water management, entrenched centralised systems are reinforced by technical standards, regulatory frameworks, and professional routines that resist change. Within this context, OWTSs occupy a niche space where innovation can occur but are influenced by the broader regime of urban water governance. As discussed in Hacker and Binz (2021a) and reflected in this thesis, the interaction between niche innovations that address not only pollution control but also broader sustainability goals (Figure 2) and the prevailing infrastructure shapes both the speed and direction of

change. Understanding these dynamics is essential for situating the development of innovative OWTS systems within wider efforts towards sustainable sanitation transitions in Sweden.

To analyse these processes, this study draws on the Multi-Level Perspective (MLP) framework (Geels, 2010, 2002). The MLP conceptualises transitions as interactions between three levels: Niches, where radical innovations are nurtured; Regimes, representing the dominant technical, institutional, and cultural arrangements; Landscape pressures, such as climate change, resource constraints, or societal value shifts. In the water sector, this framework has been applied to understand how decentralised systems with the goal beyond pollution control can respond to environmental pressures and complement centralised infrastructure (Särkilahti et al., 2017). In this thesis, the MLP provides a useful lens for analysing the institutional complexity encountered when introducing new OWTS technologies and for identifying potential leverage points for change.

Past studies have identified institutional barriers as critical constraints on new OWTS technologies adoption, often more significant than technical limitations (Reymond et al., 2020; Söderholm et al., 2023). Using an integrated framework, Hacker and Binz (2021b) outlined six key dimensions of such barriers: Equity, Knowledge and Capabilities, Financial Investment, Legal and Regulatory Frameworks, Legitimacy, and Market Structures. Barriers vary by application. For instance, greywater recycling may be hindered primarily by legal uncertainties and limited technical knowledge, whereas blackwater reuse for agriculture often faces legitimacy and market acceptance issues. Other studies (Eggimann et al., 2018; Pakizer et al., 2020) indicate that policy mixes are crucial: regulatory instruments must often be combined with procedural tools such as stakeholder participation to enable modular and decentralised systems. Scale-up experiences in Egypt and India (Reymond et al., 2020, 2018) further highlight the importance of enabling governance arrangements, clear roles and responsibilities, and pragmatic effluent standards. In Sweden, the dominance of centralised sanitation, combined with stringent environmental requirements, has limited space for experimentation with decentralised alternatives (Söderholm et al., 2023).

Beyond its technical merits for filtration, biochar is increasingly recognised for its potential role in climate change mitigation, particularly as a negative emission technology through long-term carbon sequestration. It also aligns with broader circular economy and sustainable resource management goals (He et al., 2022a). Nevertheless, widespread adoption has been slow, constrained by limited market awareness, uncertain profitability, and the absence of robust production and application standards (Hu et al., 2021). Stakeholder mapping by M. He et al. (2022b) showed the importance of policy support, standardisation, certification schemes, and integration into carbon trading mechanisms. While agricultural uses are well-established, the application

of biochar in OWTSs remains underexplored, representing both a technical and institutional frontier (Muoghalu et al., 2023).

The MmBF pilot developed in this thesis responds to these challenges by integrating biochar's treatment capabilities into a compact, modular OWTS design. The system aims not only to improve pollutant removal but also to provide flexibility for future adaptation to water reuse and resource recovery functions. Consistent with Garrido-Baserba et al. (2024), the modularity of the MmBF concept reflects emerging city-grid and hybrid infrastructure models, where decentralised units complement central networks. This thesis now turns to the Methods chapter, which details the technical design of the MmBF and the approach used to evaluate its performance under operational conditions.

Methods

This project integrates a technical evaluation of a biochar-based treatment system (**Papers I, II, and III**) with a qualitative investigation into the socio-technical dimensions that influence the adoption of new technologies in OWTS (**paper IV**). This chapter first presents the methods used in the pilot experiments with MmBFs conducted at the Uppsala municipal wastewater treatment plant, followed by the research design for the qualitative study. The same MmBF system was applied in **Papers I, II, and III** with internal module configurations updated in **Paper III**.

The following subsections describe the MmBF system setup, filter module configuration, and the sampling and analysis procedures. The final part introduces the qualitative study design presented in **Paper IV**, which addresses broader questions around the implementation of novel OWTS technologies beyond the MmBF system. It outlines the conceptual framework and case study design used to investigate institutional and regulatory factors, drawing on semi-structured interviews with key stakeholders and qualitative analysis to identify conditions that facilitate or constrain technology adoption.

MmBF pilot experiments

Pilot system setup and operation

To evaluate the performance of biochar-based systems for OWTSs, a pilot-scale MmBF system was utilised. Supporting infrastructure including influent supply lines, dosing and pumping systems, was designed and built. The layout of the overall system is shown in Figure 6. Although the MmBF was intended for post-septic tank treatment in decentralised contexts, access to actual household wastewater sources was not feasible. As a result, the pilot system was installed and operated at the municipal wastewater treatment plant in Uppsala, Sweden (Kungsängsverket), where it received pre-screened municipal wastewater.

The MmBF system comprises several key components: a dosing tank in which pre-screened wastewater circulates, a sedimentation tank (approximately 45 cm in diameter and 60 cm in height, with a maximum capacity of 100 L) where preliminary settling occurs, and three reservoirs used to distribute influent to each of the three MmBF units (MmBF 1–3). Timer-controlled

pumps were used to transfer wastewater from the dosing tank to each reservoir (Figure 6).

Each operational cycle began with the transfer of pre-screened wastewater from the dosing tank to the sedimentation tank (a process that took 1-2 hours for the daily flow amount required by the three MmBFs), where it was retained for approximately 16.5 hours to allow for primary sedimentation. After settling, the supernatant was pumped into three separate reservoirs, each connected to one of the MmBFs. The filters were operated using intermittent loading: each filter received five doses of 2 L every day (10 L day^{-1}), corresponding a hydraulic loading rate (HLR) of 50 L m^{-2} day^{-1} (Table 7). In the final three weeks of the operation, the influent volume was increased to 20 L day^{-1} per filter (HLR: 100 L m^{-2} day^{-1}) by doubling the dosing frequency to ten times per day.

The system was operated continuously over a period of 1,557 days (25 January 2021 – 30 April 2025). Details on the internal configuration of each filter are provided in the following subsection.

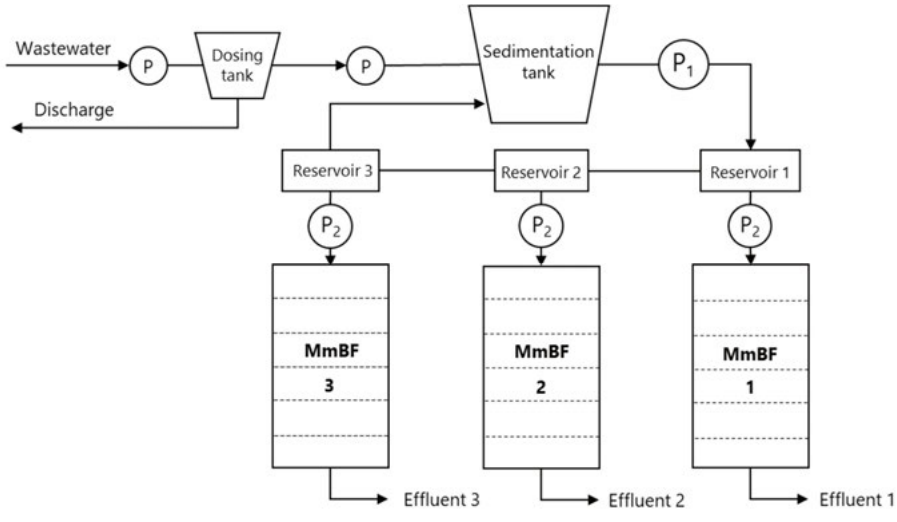

Figure 6. Schematic diagram of the pilot system setup using the multi-module biochar filters (MmBFs). Pump positions are indicated by letter P (This figure is modified from Paper I).

Table 7. Timer settings for Pump 1 and Pump 2 used to deliver influent water to each Multi-module Biochar Filter (MmBF) under two different hydraulic loading rates (HLR). Each dose corresponds to 2 L of influent per MmBF. Pump 2 settings are identical across all three units (MmBF 1–3) as illustrated in Figure 6. A dash (–) indicates no pumping scheduled for that slot.

	HLR 50 L m^{-2} day^{-1} (Day 1–1536)		HLR 100 L m^{-2} day^{-1} (Day 1537–1557)	
Dose	Pump1	Pump2	Pump1	Pump2
1	7:30-7:50	8:00-8:15	7:30-7:50	8:00-8:15
2	8:30-8:50	9:00-9:15	8:30-8:50	9:00-9:15
3	9:30-9:50	10:00-10:15	9:30-9:50	10:00-10:15
4	10:30-10:50	11:00-11:15	10:30-10:50	11:00-11:15
5	11:30-11:50	12:00-12:15	11:30-11:50	12:00-12:15
6	–	–	12:30-12:50	13:00-13:15
7	–	–	13:30-13:50	14:00-14:15
8	–	–	14:30-14:50	15:00-15:15
9	–	–	15:30-15:50	16:00-16:15
10	–	–	16:30-16:50	17:00-17:15

Module structure and flow configuration

Three identical MmBFs, each composed of six sequential modules, were constructed. Each module was built using plastic containers (polyvinyl chloride; external dimensions: 60 × 40 × 22 cm; internal dimensions: 56 × 36 × 20 cm; surface area in direction of flow in M1: 0.20 m^2) (Figure 7).

The modules were arranged to provide both aerobic and anoxic treatment zones within each multi-module system. Modules M1 to M3 operated under aerobic, unsaturated downflow conditions (Figure 7). Modules M4 and M5 were configured as anoxic units with saturated horizontal flow, maintained under limited oxygen conditions by sealing the containers with lids. The final module, M6, returned to aerobic operation, also with vertical unsaturated flow (Figure 7). This construction and flow configuration were consistent across **Papers I, II, and III**.

The MmBF system was designed with a modular layout to minimise footprint, requiring less space than conventional options such as soil infiltration fields. Its vertically arranged removable containers allow effluent to be sampled at different treatment stages, enabling assessment of pollutant distribution along the treatment process and offering potential for targeted biochar reuse. The modular configuration also facilitates routine maintenance, as filter media can be replaced easily when performance declines.

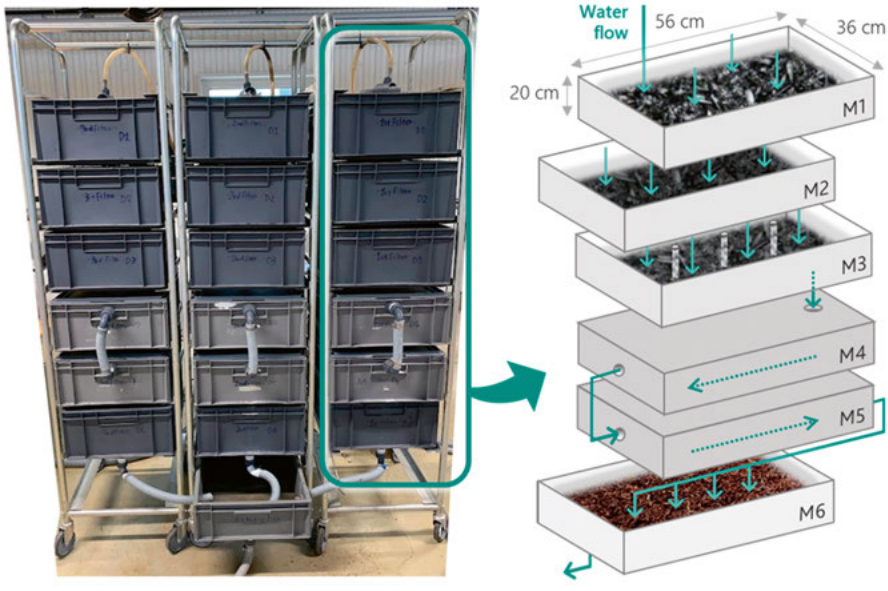

Figure 7. Photograph of the multi-module biochar filters (MmBFs) installed at the wastewater treatment plant and a schematic of a single filter module illustrating water flow and module configurations. The module dimension shows the internal size (cm).

Filter media composition and experimental configurations

While the structural configuration remained the same, the filter media compositions were modified between studies to enhance treatment performance. From here, we describe the specific media arrangements and materials used in each experimental phase. The initial configuration used across **Papers I and II** was designed prior to the start of this project, and the updated configuration was designed and operated for **Paper III**.

For the initial configuration used in **Papers I and II**, all the modules were filled with biochar, except for Module 6 (M6), which included pine bark in addition to biochar to aid in bacterial die-off by releasing tannin (Farha et al., 2020). The media layer in each module was 15-17 cm deep and the amount of material and particle size are shown in Figure 8. Biochar was obtained from Vindelkol AB (Umeå, Sweden), while the bark was purchased from Plantagen Sverige AB (Uppsala, Sweden). The biochar was commercially produced from hardwood—primarily Scots pine (*Pinus sylvestris*), with smaller portions of birch, alder, and aspen—through pyrolysis at 500 °C for 8–14 hours. Its surface morphology was examined using a Zeiss Supra 35-VP scanning electron microscope (SEM), and elemental composition was determined using an EDAX Apex 4 (Amtekh, Mahwah, USA) EDS detector for X-ray microanalysis, yielding 86.1% carbon, 13.2% oxygen, 0.2% potassium, and 0.4% calcium. Portable X-ray fluorescence (pXRF) analysis was also conducted

with a Bruker Tracer 5i (Bruker, Germany), reporting 0.5% Al_2O_3, 0.1% SiO_2, 0.2% K_2O, 0.9% Ca, 0.1% Mn, and 0.1% Zn.

Based on water flow and oxygen availability, each filter was divided into three functional units (Figure 8): the first for organic matter oxidation and nitrification (M1–M3), the second for denitrification (M4–M5), and the third for final filtration (M6).

In **Paper III**, amendments were introduced to MmBF2 and MmBF3 while MmBF1 remained as the control. These included the addition of sawdust (10 and 30 wt% respectively) and Polonite® (10 and 30 wt% respectively) (Figure 8). These amendments were selected to improve nitrogen and phosphorus removal based on findings from prior studies and from a supporting column experiment conducted as part of the **Paper III**. Sawdust was incorporated into the anoxic module M4 to supply an additional carbon source for denitrifying bacteria, chosen for its low cost and biodegradability. Polonite®, a calcium-carbonate-based sorbent containing silica, was mixed to the final aerobic module M6 to enhance phosphate adsorption. Two amendment levels (10 wt% and 30 wt%) were applied to simulate moderate and intensified configurations.

The particle density (ρ_s) of each filter material was determined, and these values were used to calculate the bulk density (ρ_b) and porosity (f) were calculated for each module across the three MmBFs (Table 8). For modules containing mixed materials (e.g., biochar and sawdust; biochar, bark, and polonite), porosity was calculated using the weighted average particle density of the individual components. Details of sampling campaigns and analytical procedures are provided in the following subsection.

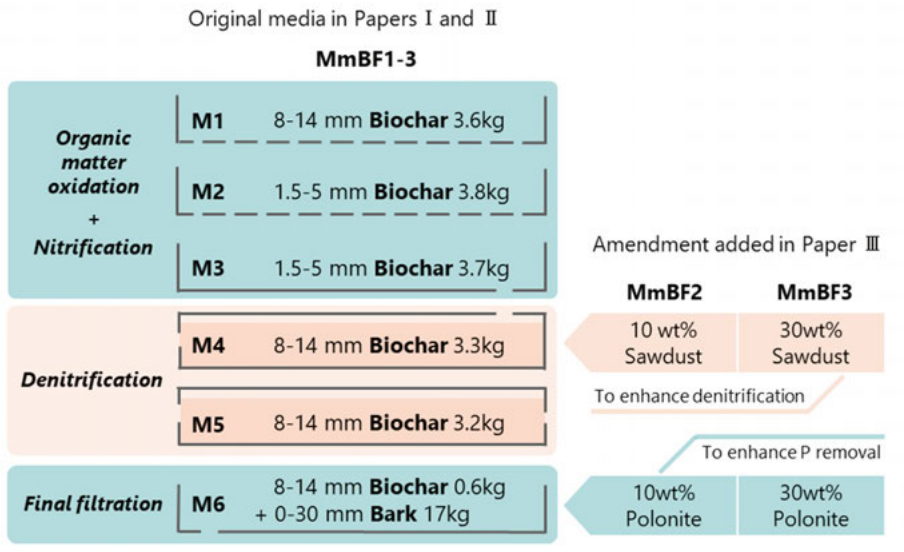

Figure 8. Filter media composition across three identical Multi-module Biochar Filters (MmBF1–3). The original configuration, applied in Papers I and II, included six modules filled with biochar (M1–M5) and bark (M6) In Paper III, MmBF1 remained unchanged while amendments were introduced to MmBF2 and MmBF3 to enhance denitrification and phosphorus removal. Modified from **Paper III**.

Table 8. Physical properties and media composition of each module (M1–M6) in the updated multi-module biochar filters (MmBFs). MmBF1 represents the original configuration, while MmBF2 and MmBF3 include modifications such as the addition of sawdust and Polonite to enhance nutrient removal. This table is modified from **Paper III**.

	MmBF1				**MmBF2**				**MmBF3**			
Module	Type of Material	Particle size [mm]	Bulk density [kg m^{-3}]	Porosity [cm^3 cm^{-3}]	Type of Material	Particle size [mm]	Bulk density [kg m^{-3}]	Porosity [cm^3 cm^{-3}]	Type of Material	Particle size [mm]	Bulk density [kg m^{-3}]	Porosity [cm^3 cm^{-3}]
1 (top)	Biochar	8-14	112	0.75	Biochar	8-14	111	0.75	Biochar	8-14	111	0.75
2	Biochar	1.5-5	117	0.85	Biochar	1.5-5	118	0.85	Biochar	1.5-5	118	0.85
3	Biochar	1.5-5	116	0.85	Biochar	1.5-5	116	0.85	Biochar	1.5-5	117	0.85
4	Biochar	8-14	101	0.78	Biochar + Sawdust (10%)	8-14 (biochar)/ <10 (sawdust)	112	0.77	Biochar + Sawdust (30%)	8-14 (biochar)/ <10 (sawdust)	132	0.80
5	Biochar	8-14	101	0.78	Biochar	8-14	100	0.78	Biochar	8-14	100	0.78
6	Biochar +Bark	8-14 (biochar) /0-30 (bark)	546	0.29	Biochar +Bark +Polonite (10%)	8-14/ 0-30/ 2-6 (polonite)	601	0.29	Biochar +Bark +Polonite (30%)	8-14/ 0-30/ 2-6	710	0.30

System monitoring, sampling and analyses

The MmBF system was operated from 25 January 2021 to 30 April 2025, corresponding to a total runtime of 1,557 days (approximately 4.3 years). Although the entire operational period represents the full timeline of the system, **Papers I, II, and III** each focus on distinct sampling period. For clarity, a separate day-counting system was applied for each period in the respective analyses (Table 9, Figure 9).

Two types of sampling were carried out. The first, *Influent–Effluent sampling* (Figure 9), involved collecting water from the sedimentation tank (influent) and the bottom outlet of M6 (final effluent). The second, *All Modules sampling*, included both influent and effluent from each individual module. Background monitoring was conducted throughout the operation, relying primarily on Influent–Effluent sampling to confirm stable system performance.

In **Paper I**, the performance of the original MmBF configuration was evaluated over two sampling periods: the first from 1 February 2021 (Day 1) to 7 April 2021 (Day 67), and the second from 27 October 2021 (Day 270) to 16 February 2022 (Day 381), as shown by black circles in Figure 9. Both Influent–Effluent samples (n=13 for Period 1; n=12 for Period 2) and All modules samples (n=4, Period 2 only) were collected during these periods.

In **Paper II**, the removal of PhACs was investigated in MmBF1 and MmBF2 using samples from Period 2. Both Influent–Effluent and All modules sampling were carried out (n=4 and 2, respectively), as indicated by cross symbols in Figure 9. Following these sampling campaigns, a tracer test was performed between 30 March and 12 May 2022 (Days 430–470 in Figure 9), and the results were presented in **Paper I**. Sample preparation for PhAC analysis (**Paper II**) also took place between May and June 2022 (approximately Days 470–540).

The sampling period for **Paper III** began after an operational pause between Days 1,139 and 1,290. The system was restarted on 7 August 2024 (Day 1,291 in Figure 9) and continued until 25 March 2025 (Day 1,521). Amendments of sawdust and Polonite® were introduced on 14 October 2024 (Day 1,359), which was defined as Day 1 in **Paper III** and marked the beginning of performance monitoring under the updated media configuration (see also Figure 8). During this period, all modules sampling was conducted from Day 1 to Day 178 (n = 16). Toward the end of the study, the hydraulic loading rate (HLR) was increased to 100 L m^{-2} day^{-1} between 10 and 30 April 2025 (Days 179–199 in **Paper III**), and system performance under higher flow conditions was assessed using All modules sampling (n = 3).

Table 9. Overview of the sampling periods across Papers I, II, and III during the full 1,557-day operation of the MmBF system. Each paper used a distinct day-counting system corresponding to its own experimental timeline.

Whole operation period in **Figure 9**	Operation in **Paper I and II**	Operation in **Paper III**
Day 1-7 (start-up)	-	-
Day 8-74	Period 1: Day 1-67	-
Day 276-388	Period 2: Day 270-381	-
Day 1359-1557	-	Day 1-199

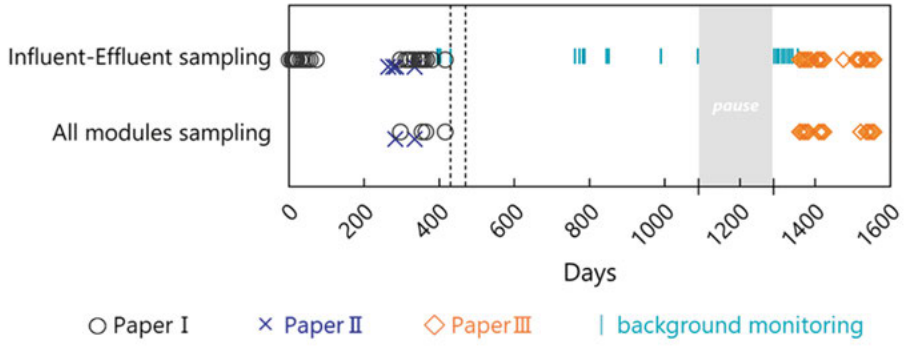

Figure 9. Overview of sampling points and analysis periods throughout the full operational timeline of the MmBFs. The dotted line shows the period when a tracer test was conducted (Day 430-470). The grey area marks an operational pause between Days 1139 and 1290.

For **Papers I to III**, the locations of each sampling point are illustrated in Figure 10. Effluent from Modules M1 and M2 was collected in four beakers placed beneath each module and then combined into a single 50 mL centrifuge tube. Effluent from Modules M3 to M5 was sampled using syringes and tubing at the outlet of each module, and each sample was transferred into separate Falcon tubes. For the PhACs analysis in **Paper II**, 400 mL of water sample was collected during Influent-Effluent sampling. After sampling, all samples were transported to the laboratory for analysis. After collection, all samples were transported to the laboratory. When immediate analysis was not possible, samples were stored at −20 °C until processing.

The hydraulic residence time (HRT) of the MmBF system was determined in **Paper I** by measuring the electrical conductivity at three points in MmBF1 (denoted by * in Figure 10). Based on the residence time distribution function E(t) and cumulative distribution function F(t) determined from the tracer test, the average HRT was calculated for each functional treatment unit (M1–M3, M4–M5, and M6; Figure 8).

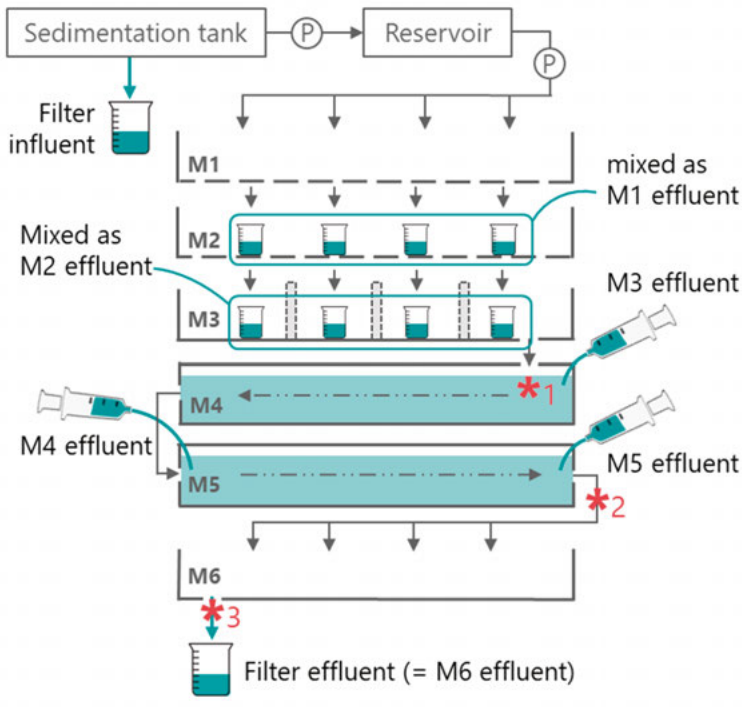

Figure 10. Schematic illustration of the sampling points used across all three MmBFs for Influent-Effluent sampling and All modules sampling. Red asterisks (*1–*3) indicate the points for electric conductivity measurement used during the tracer test conducted in MmBF1 only: *1 at the outlet of M3, *2 at the outlet of M5, and *3 at the outlet of M6.

The water samples collected throughout the experimental periods of **Papers I, II, and III** were analysed for chemical oxygen demand (COD), nitrogen species (NH_4^+-N, NO_2^--N, NO_3^--N, Tot-N), phosphate (PO_4^{3-}-P, Tot-P), and *E. coli*. In **Paper II**, the analysis also included 30 PhACs from various therapeutic classes. Ammonium (NH_4^+-N), nitrite (NO_2^--N), nitrate (NO_3^--N), phosphate (PO_4^{3-}-P) are hereafter referred to as NH_4-N, NO_2-N, NO_3-N, and PO_4-P for consistency with **Papers I to III**.

Chemical and microbiological analyses (**Papers I and III**) followed standard procedures (Baird et al., 2017). Concentrations of COD, NH_4-N, NO_2-N, NO_3-N, Tot-N, PO_4-P, and Tot-P were determined using Spectroquant® test kits (Merck KGaA, Darmstadt, Germany) and analysed colorimetrically using a Nova 60 photometer. Detection ranges for each parameter were as follows: COD (10–150 mg L^{-1} and 100–1500 mg L^{-1}), NH_4-N (2.0–150 mg N L^{-1}), NO_3-N (0.1–25.0 mg N L^{-1}), NO_2-N (0.007–3.28 mg N L^{-1}), Tot-N (0.1–25.0 mg N L^{-1}, measured via NO_3-N kit after digestion), PO_4-P and Tot-P (0.01–5.0 mg P L^{-1}, Tot-P was measured after digestion). Analytical quality

was assured using blank samples and standard solutions of known concentrations in each measurement series. Values below detection limits were conservatively presented in graphs as equal to the detection limit.

TSS and total volatile solids (TVS) were also determined according to Standard Methods 2540 D and 2540 E, respectively, in **Paper I** (Baird et al., 2017). *E. coli* enumeration was conducted by serial dilution (10–10^4) of wastewater samples in phosphate-buffered saline with Tween®20 (SVA, Sweden), plating 100 µL on Chromocult agar (Miclev, Sweden). After incubation at 37 °C for 24 h, colonies were counted and reported as CFU mL^{-1}. Counts exceeding 200 CFU mL^{-1} were considered too numerous to count due to the difficulty in identifying individual colonies.

In **Paper III**, dissolved oxygen (DO) was also measured for each sample using a portable optical DO meter (Orion Star A223; Thermo Fisher Scientific, USA). For Modules M4 and M5, DO was measured directly within the module rather than from collected samples.

A total of 30 PhACs, covering a wide range of therapeutic classes such as analgesics, antihypertensives, antidepressants, and antibiotics, were analysed in **Paper II**. Sample preparation involved filtration (0.7 µm), acidification, and addition of Na_2EDTA and isotopically labelled internal standards, followed by solid-phase extraction (SPE) using Oasis HLB cartridges. Extracts were eluted with methanol, evaporated to dryness, and reconstituted in a methanol–water solution. Instrumental analysis was performed by IVL Swedish Environmental Research Institute using HPLC-MS/MS with electrospray ionisation in both positive and negative modes. Quantification was based on a 7-point calibration curve (5–500 ng mL^{-1}), and analyte concentrations were corrected for recovery and blank subtraction. Limits of detection (LOD) and quantification (LOQ) were estimated as 3× and 10× the noise level in blanks, respectively. Five compounds with recovery < 60% were excluded, resulting in 25 PhACs used for final interpretation.

Removal efficiency (%) was calculated in **Papers I to III**, from the influent and effluent concentrations, and the removal rate (g m^{-2} day^{-1}) was calculated based on the HLR of 50 or 100 L $m^{-2}d^{-1}$, depending on the experimental phase (Equation 1 and 2; where C_{in} is the influent concentration and C_{eff} is the effluent concentration). Additionally, removal per HRT cycle for each functional unit (M1–M3, M4–M5, M6) was estimated by multiplying the removal rate by the corresponding HRT of each unit in **Paper II**.

$$\text{Removal efficiency (\%)} = \left(1 - \frac{C_{eff}}{C_{in}}\right) \times 100 \quad (1)$$

$$\text{Removal rate (g m}^{-2}\text{ day}^{-1}\text{)} = (C_{in} - C_{eff}) \times HLR \quad (2)$$

Statistical analysis

For **Papers I–III,** statistical analyses were applied to assess treatment performance, compare filter configurations, and explore relationships between operational parameters and pollutant removal. Parametric tests, including t-tests (paired and unpaired) and analysis of variance (ANOVA), were used where appropriate, alongside correlation analyses and kinetic modelling in the evaluation of pharmaceutical removal. Assumptions of normality and homogeneity of variance were checked prior to analysis, with corrections such as Welch's adjustment or the Greenhouse–Geisser correction applied when necessary. The selection of each test was based on the study design, data structure, and research question. Full details of the statistical approaches for each dataset are provided in the individual papers.

Socio-technical context of decentralised system

Using the MLP framework to understand OWTS in Sweden

The qualitative component of this study (**Paper IV**) investigated the institutional and socio-technical conditions influencing the adoption of new OWTS technologies such as biochar-based filters in Sweden. To guide the analysis, the study employed the MLP framework (Geels, 2018, 2002), which conceptualises socio-technical transitions across three analytical levels: niches, where innovations emerge; regimes, representing dominant institutional and technical systems; and landscapes, referring to broader structural forces such as climate policy or societal expectations.

In this study, conventional OWTS refers to socio-technical systems centered around as septic tanks, sand filters, and soil infiltration units, which are widely used in Sweden. In contrast, alternative OWTS technologies, including source-separating systems, constructed wetlands, and biochar-based filters, have been developed but remain marginal in terms of widespread adoption. This study examines how such niche innovations can be better supported through interactions with the current regime of OWTS and its governance, which is shaped by national regulations, municipal enforcement and practice, and established industry practices (e.g., prevailing design standards, installation conventions, and certification routines). The analysis focuses on regime and niche dynamics, where institutional rules, actor relationships, and technological expectations either constrain or facilitate the adoption of alternative systems. In the Swedish OWTS context, these dynamics play out through municipal permitting practices, national environmental standards, and the expectations of technology suppliers and house owners.

While the MLP has been widely applied in studies of urban water and sanitation transitions (Helgegren et al., 2021; Karpouzoglou et al., 2023; Särkilahti et al., 2017), its application to OWTS remains limited. Previous studies have examined transitions at the level of the entire water sector, often treating

centralised systems as the regime and OWTS as niche innovations (Konrad et al., 2008; Särkilahti et al., 2017). In contrast, this study focuses specifically on the OWTS domain itself, which is conceptualised as a socio-technical subsystem that includes both niche and regime elements. It involves public regulation as well as private decision-making. The Swedish case provides an illustrative example of this complexity, with a multi-level governance structure involving national agencies, regional and municipal authorities, technology developers, contractors, and knowledge intermediaries (Figure 11).

Applying the MLP lens to this OWTS socio-technical context enable to identify cross-level coordination challenges, institutional frictions, and potential leverage points for policy and innovation.

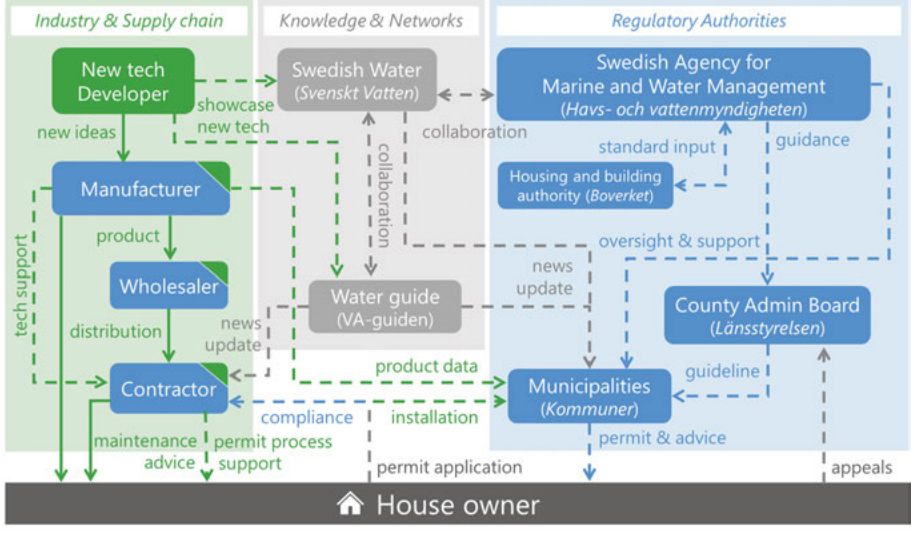

Figure 11. Actor interactions in the governance of onsite wastewater treatment systems (OWTS) in Sweden. Note that additional actors can be included in certain cases. Blue indicates regime-level actors, green denotes niche-level actors, and blended corners represent dual roles. Solid arrows show flows of products or services, and dashed arrows indicate information or institutional interactions (**Paper IV**).

Interviews and analysis

Interview recordings and written responses were collected through semi-structured interviews with 14 key stakeholders (12 oral interviews and 2 written responses) representing different actor types and institutional levels. Interviewees included municipal inspectors, national regulators, technology developers, researchers, consultants, and legal experts, covering both regime and niche levels of the MLP. Participants were selected to reflect a range of perspectives on OWTS governance, with anonymised identifiers (e.g., P01–P14) used for confidentiality. Interviews were conducted in person or digitally depending on availability, and audio-recorded with consent. All interviews were

transcribed, combining automated transcription using Whisper (large-v2 model; OpenAI, 2022; https://github.com/openai/whisper). The Whisper model was executed entirely offline within the secure UPPMAX high-performance computing environment at Uppsala University, ensuring that no audio files or transcripts were transferred to external or cloud-based servers.

After the automated step, the transcripts were carefully checked and corrected manually, with particular attention to passages affected by background noise or overlapping voices. The finalised texts were then imported into QualCoder (version 3.7; https://qualcoder.wordpress.com/), an offline, open-source package for qualitative analysis. Coding combined pre-defined categories created from the research framework, with new themes that emerged from the material. A codebook was iteratively developed and revised through coding rounds. The final coding structure is presented in Table 10. To guide the analysis, the research questions from **Paper IV** were each linked to a level of the MLP: landscape, regime, and niche. Interview questions were designed to probe these levels, for example by addressing national regulations and demographic trends (landscape), municipal enforcement and industry routines (regime), and technology development or adoption experiences (niche).

Thematic saturation was monitored throughout the process and determined to be reached after 14 interviews, as no substantially new themes emerged. Complementary document review (e.g., Swedish Environmental Code, EU laws, environmental reports) supported the analysis and provided context for understanding landscape-level dynamics such as regulatory trends and climate goals.

Table 10. Thematic coding structure and corresponding Paper IV research questions (RQs) (Paper IV).

Category	Sub code	Definition
RQ1. Which innovations (new technologies) are currently being adopted?		
RQ1	New technologies	Technologies mentioned by respondents as recently introduced, emerging, or under consideration for adoption in OWTS contexts. Includes both mainstream and experimental systems.
RQ1	OWTS regional context	How geographical, environmental, or socio-economic factors specific to a region influence the development or adoption of OWTS technologies.
RQ1	Awareness of biochar-based systems	The extent to which respondents are informed about or have engaged with biochar-based filtration systems in OWTS.
RQ1	Reactions to biochar	Affective or evaluative responses (e.g., enthusiasm, scepticism, interest) expressed by respondents when discussing biochar-based technologies.
RQ2. How do regulations and institutional policies (regime) affect the adoption of new wastewater treatment technologies (niche)?		
RQ2	Regulation	Comments related to official regulations affecting OWTS technologies and practices.
RQ2	Institutional policies	Broader organisational routines, or informal practices (e.g., municipal interpretations) that shape OWTS governance and adoption.
RQ2	Financial mechanism	Respondent insights into how OWTS innovations are financed, including subsidies, cost-sharing, or funding barriers.
RQ2	Adoption process/maintenance	Descriptions of how OWTS innovations are introduced, implemented, or maintained within institutional settings, including challenges related to long-term upkeep.
RQ3. How do different stakeholder groups interact in the adoption process, and what are the key barriers and facilitators in cross-level coordination?		
RQ3	Interaction	Descriptions of engagement or communication between different actor groups (e.g., municipalities and innovators).

RQ3	Collaborative	Positive examples of collaboration, partnership, or coordinated efforts among stakeholders in OWTS development or implementation.
RQ3	Challenging	Accounts of friction, misunderstanding, or difficulty in communication or coordination between actor groups.
RQ3	Barriers/problem	Identified systemic or situational obstacles that hinder OWTS innovation, adoption, or inter-actor cooperation.
RQ3	Facilitators	Factors identified as supportive of innovation, adoption, or improved interaction (e.g., clear guidance, flexibility).
RQ3	Improving point	Participant suggestions for how OWTS adoption or cross-actor interaction could be enhanced (e.g., through policy reform, technical guidance, or collaboration).
RQ3	Frustrating	Statements that convey strong emotional responses (especially irritation, stress, or disappointment) linked to barriers or interactions.
RQ3	Sadness	Expressions of concern, discouragement, or helplessness in response to perceived systemic issues or failures in OWTS governance.
RQ4. How do broader landscape-level changes impact the potential for integrating onsite wastewater treatment technologies into existing infrastructure?		
RQ4	Swedish laws influence	Commentary on how Swedish national laws, legal interpretations, or regulatory changes influence OWTS governance or innovation.
RQ4	EU laws influence	Observations on how EU directives or harmonisation processes affect OWTS-related policy or practice in Sweden.
RQ4	Sustainability goals influence	Reflections on how national or global sustainability frameworks (e.g., climate action, SDGs) influence or legitimise OWTS innovations.
RQ4	Other external influence	Broader external forces influencing OWTS transitions not covered by legal or sustainability frameworks (e.g., market forces, public opinion).

Main findings and discussion

This chapter presents and discusses the main findings of the thesis, moving from the technical evaluation of the MmBF system to the qualitative interview study. The technical section is organised to first describe the hydraulic retention time of the system (**Papers I–III**), followed by the removal performance for key pollutant groups: organic matter and faecal bacteria (**Papers I and III**), nitrogen with the effect of sawdust addition (**Papers I and III**), phosphorus including the effects of polonite amendments (**Papers I and III**), and pharmaceutically active compounds (**Paper II**). The final section presents the qualitative results from stakeholder interviews, examining the institutional and regulatory dimensions that affect the adoption of novel technologies for OWTS (**Paper IV**).

Hydraulic retention time

Tracer tests in **Paper I** showed that the average HRT was 168 h in the aerobic modules (M1–M3) and 135 h in the final module (M6). In contrast, the anoxic denitrification unit (M4–M5) had a measured HRT of 21 h, much shorter than the theoretical value of 152 h, indicating preferential flow pathways. Such uneven flow distribution could reduce contact between wastewater and biofilm, limiting denitrification efficiency. This interpretation is consistent with Wei et al. (2025), who found that improving hydraulic efficiency in a baffled constructed wetland significantly increased nitrogen removal by reducing short-circuiting. To address this issue, the internal piping was modified between the **Papers I, II** and **Paper III** periods by installing downward L-shaped connectors at the M4 outlets, M5 inlets, and M5 outlets (nine in total across the three MmBFs; Figure 10). These adjustments were intended to enhance flow distribution and reduce short-circuiting. Their effect on HRT was not evaluated within this project, but future tracer tests could confirm their impact on hydraulic efficiency.

Organic matter and faecal bacteria removal

Across both the original and amended MmBF configurations, the removal of organic matter remained consistently high (Figure 12 and 13). In **Paper I**, the mean effluent COD concentrations were within 36 ± 11 mg L^{-1}, corresponding

to mean removal efficiencies of 83 ± 4% during the sampling period 1 and 95 ± 2% in the period 2. The majority of COD attenuation occurred in the aerobic modules M1–M3, which together accounted for over 80% of the total removal, confirming that the aimed function for organic matter oxidation was working well (**Paper I**).

In addition to organic matter, **Paper I** also monitored faecal indicator bacteria (*E. coli*). The MmBF achieved reductions of 2.3 ± 0.9 $\log_{10}$ units during the sampling period 1, although performance declined to 1.8 ± 0.7 $\log_{10}$ in the period 2. This decline may reflect reduced tannin leaching from the bark over time, diminishing the antimicrobial contribution of those compounds (Arzola-Alavarez et al., 2020). Time-dependant decreases in aqueous tannin concentrations have also been reported (Earl et al., 2025). For comparison, conventional sand filters achieved 1.4–5.4 $\log_{10}$ *E. coli* removal (Table 5) (Kauppinen et al., 2014; Pundsack et al., 2001) , placing the MmBF within the higher end of reported ranges. However, as sampling from all modules for microbial analysis was not conducted, further work is needed to better characterise faecal bacteria removal dynamics across the MmBFs.

In the updated configurations tested in **Paper III**, the addition of sawdust to anoxic modules M4 and Polonite to the final module M6 did not alter the already efficient removal of organic matter. Removal consistently exceeded 90 % under the standard HLR (50 L m^{-2} day^{-1}), and remained above 85 % even when the HLR was doubled to 100 L m^{-2} day^{-1}. This stability suggests that biochar's high surface area and pore structure provided favourable conditions for microbial communities, supporting sustained degradation regardless of operational changes. During Day 2–199 of **Paper III**, the MmBF maintained a mean effluent COD of 22 ± 12 mg L^{-1}, which is on average below the BOD limit of 30 mg L^{-1} set by SwAM (2016), noting that BOD is always lower than or equal to COD for the same sample. This indicates that the system can meet national standards for organic matter removal in onsite wastewater effluent.

Nitrogen removal and the effect of sawdust amendment

In the original MmBF configuration assessed in **Paper I**, ammonium oxidation was consistently near complete across both sampling periods (Figure 12), with no significant difference between the three MmBFs ($p > 0.05$). Most influent NH_4-N was oxidised to nitrate within the aerobic modules M1–M3, supported by the long HRT of 168 h in this functional unit. Some biological assimilation into microbial biomass likely occurred, but adsorption to biochar, while possible (Dai et al., 2020; Liang et al., 2016), was considered minor compared to biological oxidation due to the NH_4-N conversion to the NO_3-N (Figure 12). Among the aerobic modules, M2–M3 showed the highest nitrification efficiency, likely due to the low COD/Inorg-N ratio, which aligns with the low carbon requirement of autotrophic nitrifiers (Navada et al., 2020) (Figure 12).

Denitrification efficiency was improved in period 2 compared to period 1 (41 ± 18% to 71 ± 7% by Inorg-N removal), lowering mean effluent NO_3-N from 24.1 ± 7.0 mg-N L^{-1} to 8.1 ± 3.7 mg-N L^{-1} ($p < 0.0001$). The improvement corresponded with a higher organic loading rate (36 g COD m^{-2} d^{-1} in period 2 compared with 12 g COD m^{-2} d^{-1} in period 1), which supplied more available carbon for denitrification. The COD/N ratio of 6 observed in M4 during period 2 was below the range of 8 to 10 that has been reported for effective denitrification in sequencing batch reactors (Keller et al., 2001; Puig et al., 2007). While the MmBF is structurally different from sequencing batch reactors, this benchmark suggests that the carbon supply was insufficient for optimal denitrification in the functional unit M4-M5 and that improved availability of biodegradable carbon would likely enhance nitrate removal.

Building on these results, **Paper III** tested sawdust amendments in M4 to provide an additional source of biodegradable carbon. During the first 24 days, the 30 wt% sawdust addition (MmBF3) achieved significantly higher nitrate removal (80%) than the control (49%, $p = 0.0115$). MmBF2 (10 % sawdust) reached 72% removal, though it was not statistically different from the control ($p = 0.0562$). The higher removal in MmBF3 was likely attributed to elevated COD/Tot-N ratios in M4 and M5 (12.7 and 6.1 in effluent, respectively), similar to the ratio (8-10) of efficient sequencing batch reactors (Keller et al., 2001; Puig et al., 2007). However, the effect of sawdust addition declined after three weeks. COD/Tot-N ratios in M4 effluent declined to averages of 1.3 (control), 1.5 (10 % sawdust), and 1.8 (30 % sawdust) during Days 52–199, consistent with depletion of readily bioavailable carbon in woody media, which typically exhibit an initial flush of labile dissolved organic carbon that drops sharply within the first weeks of operation (Abusallout and Hua, 2017; Warneke et al., 2011). These results imply that maintaining enhanced nitrate removal would require periodic replacement of sawdust.

While **Paper I** highlighted carbon limitation as a constraint on denitrification, DO measurements in **Paper III** further showed that the denitrification unit was not fully anoxic, with DO concentrations of 0.6-1.8 mg L^{-1} in M4 and M5 across all MmBFs, though values were lower than in the aerobic unit (3.5-5.5 mg L^{-1}). These conditions likely suppressed denitrification by supporting aerobic carbon oxidation, which competes with denitrifying processes (Wei et al., 2022). Even very low DO (0.1 mg L^{-1}) can reduce denitrification rates by 35% (Oh and Silverstein, 1999), and maintaining DO below 0.3 mg L^{-1} is critical to sustaining high denitrification efficiency (> 90%) (Urbini et al., 2015).

In the final bark and biochar module (M6), nitrate concentrations increased despite low ammonium (Figure 13), likely due to aerobic mineralisation of organic nitrogen followed by nitrification. This was reflected in the increase of the inorganic nitrogen fraction from 56–71% in M5 to 78–88% in M6, similar to findings in other bark-based aerobic environments (Romashkin et al., 2018; Taylor et al., 2013).

Overall, **Papers I and III** confirm that the MmBF consistently achieved near-complete nitrification but had limited denitrification in the absence of an additional carbon source and in the presence of residual dissolved oxygen. Although the denitrification unit was not fully anoxic, sawdust addition improved performance sufficiently to meet the Swedish Tot-N standard for on-site systems (50% removal or < 40 mg L^{-1}; SwAM, 2016), although the effect was temporary as labile carbon was depleted. Future applications may therefore require strategies such as media replacement, sawdust pre-treatment to extend carbon release (Hu et al., 2017), or alternative hydraulic control to improve flow distribution.

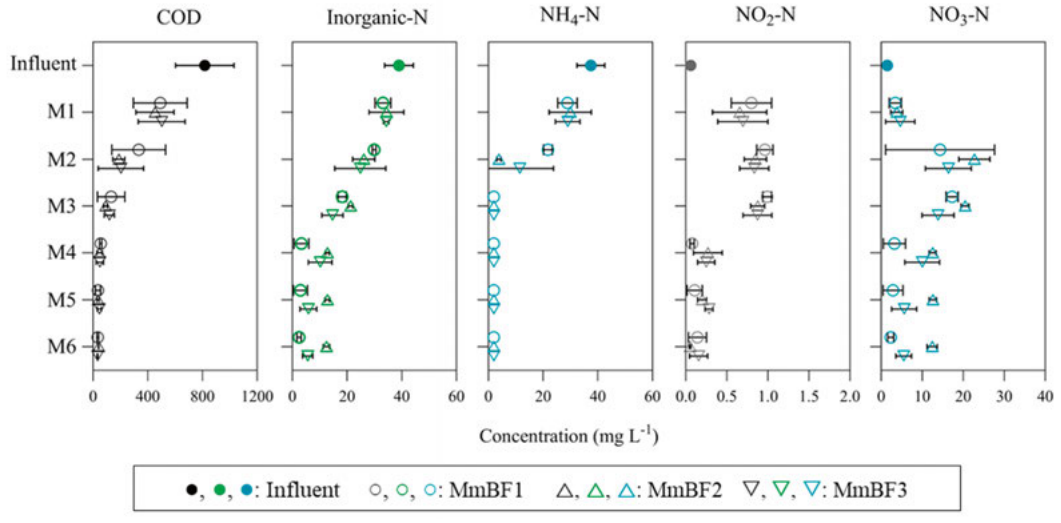

Figure 12. Concentrations (mg L^{-1}) of COD, Inorganic-N, NH_4-N, NO_2-N, and NO_3-N across modules in period 2 of **Paper I**. The data points and error bars for each module show the mean concentrations and standard deviations through the sampling occasions (n = 3 for Inorganic N, NH_4-N, NO_3-N, and NO_2-N, n = 4 for COD). For NH_4-N in modules M4–M6, concentrations were below the detection limit (2 mg L^{-1}); and values were plotted as the detection limit.

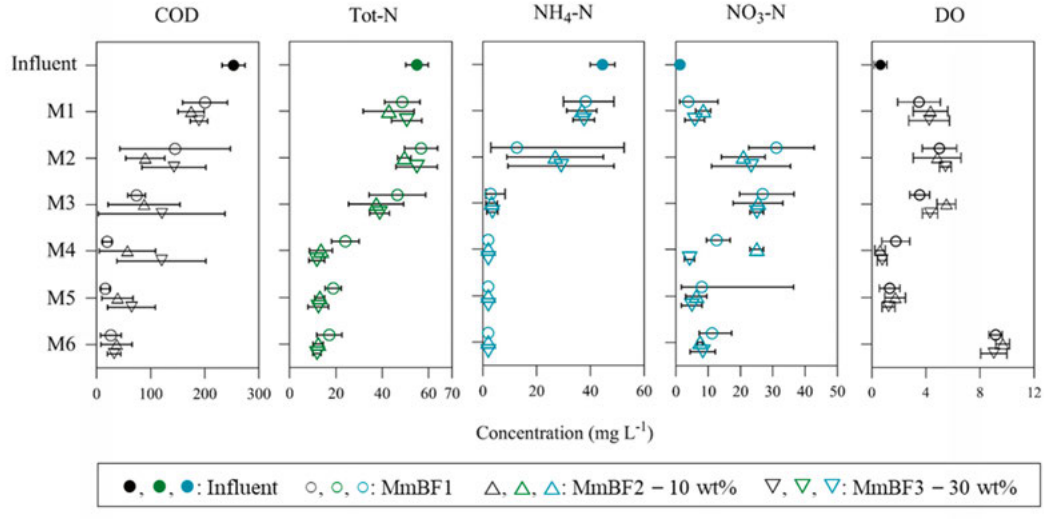

Figure 13. Concentrations (mg L^{-1}) of COD, Tot-N, NH_4-N, NO_3-N, Tot-N, and DO based on the all modules samplings during Days 2–24 (n = 5) of **Paper III**. For NH_4-N in modules M4–M6, concentrations were below the detection limit (2 mg L^{-1}), and values were plotted as the detection limit.

Phosphorus removal and Polonite effect

In the original MmBF configuration evaluated in **Paper I**, phosphorus removal was limited. Influent PO_4-P concentrations averaged 1.25 ± 1.63 mg L^{-1} in the period 1 and 4.04 ± 1.03 mg L^{-1} in the period 2 (Figure 14). Effluent concentrations often approached influent levels, and in some cases, higher effluent concentration was observed. This higher effluent was likely due to desorption from previously loaded biochar surfaces or the flushing of residual phosphorus from earlier testing before this project (see **Paper I** for details). The low phosphorus sorption capacity was attributed to the biochar's composition, which contained low amounts of calcium (0.9%), aluminium (0.5% as Al_2O_3), and iron (<0.1%), elements known to form insoluble phosphate precipitates (Dalahmeh et al., 2020).

Paper III addressed this limitation by amending the final aerobic module (M6) with Polonite, a calcium-carbonate-based material containing silica, derived from opoka rock, at the level of 10 wt% and 30 wt%. Polonite has a high calcium oxide content and demonstrated high phosphate sorption capacities in other studies (Karczmarczyk et al., 2017). Over 199-day monitoring period in MmBF, mean PO_4-P removal was 11.3 % in the control MmBF1, 17.2 % in the MmBF2 with 10 wt% Polonite, and 37.5 % in the MmBF3 with 30 wt% Polonite. During the early short period (Day 2–24, Figure 14), MmBF3 maintained enhanced removal, with post-hoc tests showing a significant difference between MmBF3 (31 % removal) and MmBF1 (2 % removal) ($p = 0.0373$), although the overall ANOVA among three MmBFs was not statistically significant ($p = 0.0585$). MmBF2 achieved 13 % removal, which was not significantly different from the control ($p = 0.528$). The efficiencies observed here were lower than those reported in earlier studies using Polonite in column experiments and field study. Gustafsson et al. (2008) reported over 95 % removal in saturated column experiments over 68 weeks, and Renman and Renman (2010) found an average of 89 % removal over nearly two years in a full-scale OWTS treating single-household wastewater. Both studies used only Polonite mixed with 10 wt% *Sphagnum* peat to reduce clogging from calcium carbonate precipitation. This corresponded to a considerably larger proportion of Polonite than was used in module M6 of the MmBF in the present study. Although the addition of bark and biochar in M6 may have reduced clogging risk, a higher Polonite fraction would likely have enhanced phosphorus removal performance.

The addition of Polonite did not adversely affect the removal of organic matter or nitrogen, suggesting that the integration of reactive media for phosphorus targeting could be implemented without compromising other treatment objectives. However, operational considerations would include potential pH elevation in the effluent, a known characteristic of Polonite (Renman and Renman, 2010), which can affect receiving water chemistry (Figure 14).

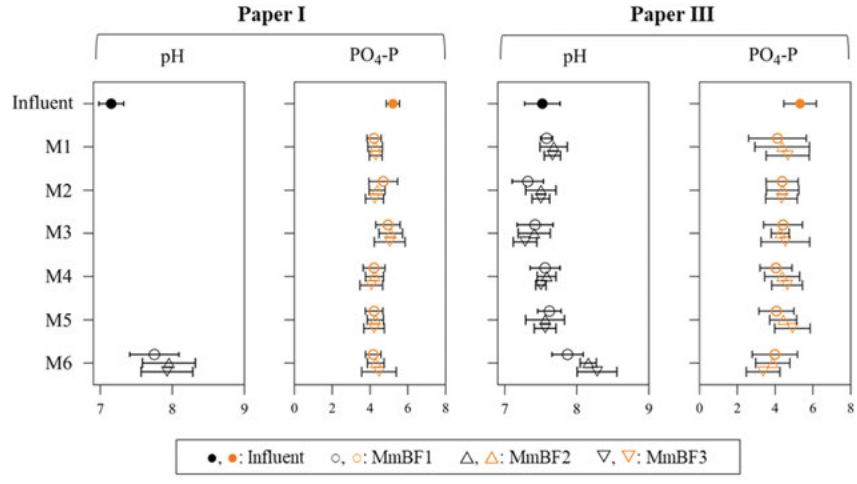

Figure 14. pH and phosphate (PO4-P) concentrations (mg L^{-1}) along the treatment modules (M1–M6) in period 2 of **Paper I**, and Days 2–24 (n = 5) of **Paper III**. Error bars represent standard deviations.

The results from **Papers I and III** together indicate that although the 30 wt% Polonite addition achieved significantly higher PO_4-P removal than the control, the performance still fell short of the Swedish standard for high protection areas, which requires 90% total phosphorus removal or effluent concentrations below 1 mg L^{-1} (SwAM, 2016). These findings suggest that optimising Polonite application in MmBFs may require alternative implementation strategies, such as using Polonite as the sole medium rather than mixing it with biochar, and adjusting media replacement intervals to achieve higher and more stable removal efficiencies.

PhACs removal

In **Paper II**, twenty-five out of thirty target PhACs were detected in the influent, with analgesics and antihypertensives as the predominant groups. Influent concentrations were broadly consistent with values reported for Swedish municipal WWTPs and OWTSs, as described by Falås et al. (2012) and Gros et al. (2017). The MmBFs achieved consistently high removal efficiencies for most compounds, with an overall average reduction of total PhACs (ΣPhACs) from 1.6×10^6 ng L^{-1} in the influent to 4.1×10^3 ng L^{-1} in the effluent, corresponding to a mean removal of 99.7 ± 0.3%. No significant difference in performance was found between the two MmBFs investigated ($p > 0.05$). Compounds such as paracetamol, naproxen, and ibuprofen exceeded 99% removal, with substantial retention in the biochar matrix, measured at 8.7×10^4 ng g^{-1} for paracetamol, 6.4×10^4 ng g^{-1} for naproxen, and 2.7×10^4 ng g^{-1} for ibu-

profen. For each compound removal, only five compounds had removal efficiencies below 95 % (fluconazole 94%, benzotriazole 93%, methotrexate 93%, furosemide 81%, hydrochlorothiazide 75%). These results were consistent across four Influent-Effluent samplings after one year of MmBF operation. This high removal after one year of operation without biochar replacement indicates long-term functional stability of MmBFs.

Analysis of module-specific performance showed that the first aerobic module, M1, accounted for 92.8 ± 3.9% of the total removal of ΣPhACs, reducing concentrations from 1.85×10^6 ng L^{-1} to 1.2×10^5 ng L^{-1} (Figure 15). The following aerobic modules, M2 and M3, provided smaller but measurable additional reductions, removing 46.6 ± 19.6% and 53.7 ± 24.0% of the incoming ΣPhACs, respectively. The high removal efficiencies are likely the result of multiple processes operating in combination. Adsorption is a key mechanism, particularly in early modules, facilitated by the biochar's porous structure, large surface area, and diverse surface functional groups (Maleki Shahraki and Mao, 2022; Muoghalu et al., 2023). Hydrophobic interactions and electrostatic attraction contribute strongly for compounds with high octanol–water partition coefficients (log Kow > 4) (Gao et al., 2019; Matamoros and Salvadó, 2013). Biodegradation also appears important for sustaining long-term performance, with microbial communities supported by the sequential aerobic–anoxic design enabling degradation under varying redox conditions (de Wilt et al., 2018; He et al., 2018). Aerobic degradation was likely dominant in the first modules (M1–M3), whereas anoxic processes may contribute in M4 and M5. Evidence from other biofiltration studies suggests that microorganisms can degrade compounds previously adsorbed onto media, freeing adsorption sites in a process known as bioregeneration, which could help extend the operational life of the biochar (Smolin et al., 2020; Zou et al., 2024).

Compound-specific results showed that analgesics and certain antibiotics were readily removed (> 94%), while a few hydrophilic and highly soluble substances, such as furosemide (81.2 ± 18.0% removal), hydrochlorothiazide (74.6 ± 50.6%), and benzotriazole (93.0 ± 2.5%), had lower removal efficiencies but still performed better than typical WWTP removal rates, which often remain below 50% for these compounds (Verlicchi et al., 2012). Metoprolol, generally considered less biodegradable, was removed at 99.1 ± 2.2%. Other compounds that are typically poorly removed in WWTPs (Verlicchi et al., 2012), such as tramadol, clarithromycin, trimethoprim, and propranolol, all achieved removal above 94%.

Compared to other decentralised treatment technologies, the MmBFs demonstrated superior performance; for instance, constructed wetlands often achieve 25–83% removal depending on compound and configuration (Ávila et al., 2015; Ilyas and van Hullebusch, 2020; H. Zhang et al., 2023), while slow sand filters have been reported to remove less than 10% of certain PhACs such as carbamazepine and gemfibrozil (D'Alessio et al., 2015). The cumulative PhAC load retained on biochar in **Paper II** (210 μg g^{-1}) was comparable

to the adsorption capacity achieved in a powdered activated carbon system tested by Kårelid et al., (2017), though the MmBFs were operating at much lower flow rates.

Overall, **Paper II** demonstrate that the MmBF system, operated under low-flow conditions and without media replacement for over a year, achieved high-level removal of a wide range of PhACs. The sequential aerobic–anoxic configuration, extended hydraulic retention, and use of biochar contributed to consistent performance across compounds with diverse physicochemical properties. These findings indicate strong potential for the MmBF concept as a long-term, low-maintenance option for decentralised wastewater treatment applications.

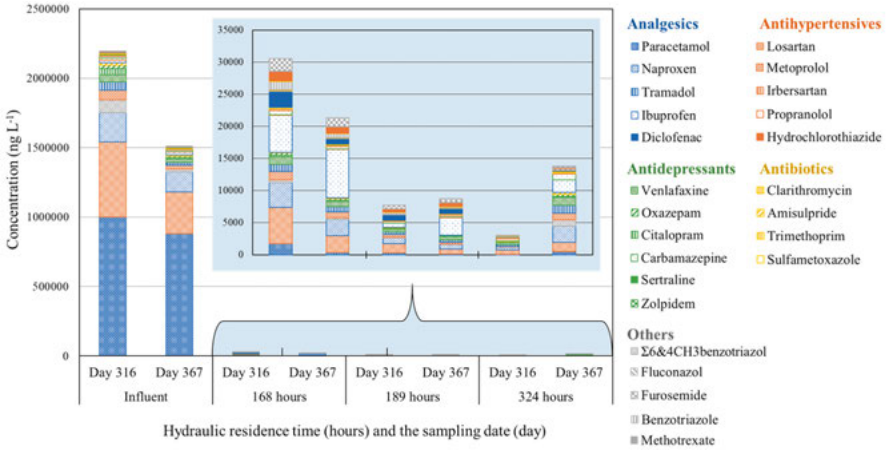

Figure 15. Effluent PhACs concentrations in each functional unit of the MmBFs, corresponding to the total hydraulic residence time until that unit (M1–M3: 168 h, M1–M5: 189 h, M1-M6: 324 h), based on averaged data of MmBF1 and 2 during two sampling events (Day 316 and Day 367). (**Paper II**)

Limitations and future study suggestions

The MmBF pilot-scale study provided valuable insights into system performance under controlled operational conditions. However, several limitations should be considered when interpreting the results and assessing their applicability to full-scale OWTS. The daily influent volume per filter (30 L day^{-1}) was lower than the average daily wastewater production of a typical household (170 L $person^{-1}$ day^{-1}) (SwAM, 2016), although the applied HLR was comparable to the surface loading rates (50 L m^{-2} day^{-1}) of conventional soil infiltration systems (US EPA, 2002). This operating choice reflected initial design recommendations and the resources available at the time. While the configuration was suitable for evaluating treatment processes, extrapolation to real-world conditions should consider potential variations in hydraulic patterns, peak flows, and seasonal influences.

The pilot-scale design also restricted the ability to fully characterise small-scale hydraulic dynamics within each module. Flow assessment relied on tracer-based HRT measurements and module-level water quality monitoring, which may not detect fine-scale preferential pathways or short-circuiting.

Future work could address these limitations by operating pilot systems at influent volumes closer to typical household wastewater production, ideally using real household wastewater, and evaluating long-term performance under a range of climatic conditions. Incorporating microbial community analyses from spatial sampling in M4 and M5 could provide valuable information for understanding the denitrification process. Such refinements would enable a more detailed understanding of media interactions and treatment mechanisms, ultimately informing the optimisation of MmBF design for wider application in decentralised sanitation.

From a broader OWTS perspective, the modular design of the MmBF allows potential flexibility in configuring modules to address site-specific phosphorus management needs. In highly sensitive catchments, higher proportions of Polonite or other reactive media could be applied, whereas in areas with lower sensitivity, reduced amendment levels could help minimise operational costs. This adaptability aligns with calls for more customised OWTS configurations that balance treatment performance, cost efficiency, and environmental protection (Garrido-Baserba et al., 2024).

A socio-technical perspective from qualitative interview

The following sections present a synthesis of interview findings from **Paper IV**, supported by selected direct quotations from participants. The excerpts have been lightly edited to remove filler words and redundancies, while preserving the original meaning and tone. Each quote is attributed to the participant's role and MLP level to provide context.

Interviewee information and distribution

Paper IV was based on semi-structured interviews with 14 stakeholders involved in the regulation, development, and implementation of OWTS in Sweden. Participants included municipal environmental officers, technology developers, researchers, and national authorities. The geographical distribution shown in Figure 16 reflects the regions associated with the interview content.

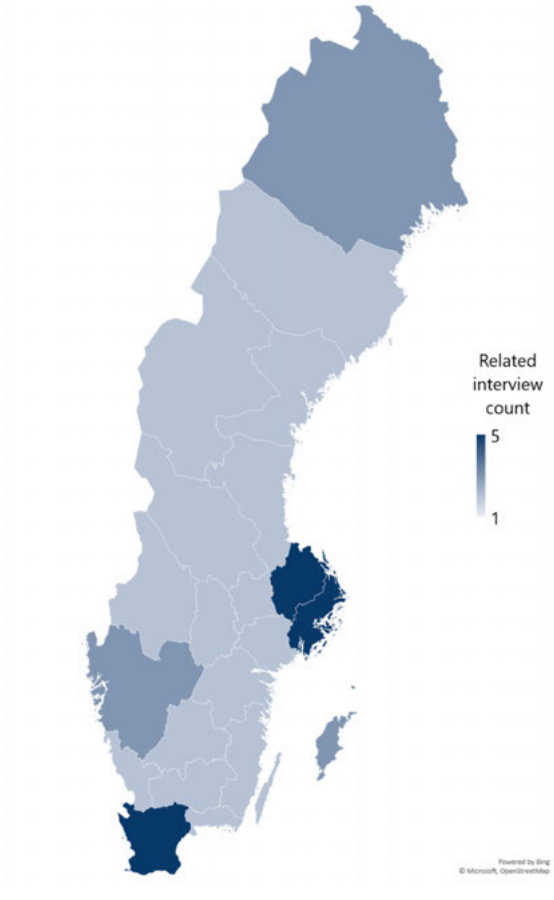

Figure 16. Geographic distribution of Swedish counties related to each interview. Shading indicates the number of interviews associated with each county. For actors with national-level roles (e.g., authorities or technology suppliers), one interview is counted for all counties to reflect their cross-regional relevance. The map represents regionally contextualised connections rather than individual interviewee locations.

An MLP socio-technical perspective

Applying the Multi-Level Perspective to Sweden's OWTS showed that innovation adoption occurs at the intersection of niche and regime dynamics, shaped by broader landscape influences. Niche-level actors, such as technology developers and researchers, focus on demonstrating the performance and viability of new treatment systems, while regime-level actors, including municipalities and national agencies, operate within established regulatory frameworks. The MLP lens revealed that these boundaries are fluid, with some regime actors engaging temporarily in niche activities, particularly during pilot projects.

Interplay of different niche and regime actors in the MLP

Interactions in the MLP between niche and regime actors are marked by both collaboration and tension. Technology developers often depend on municipalities for permitting and demonstration opportunities, while municipal officers rely on reference systems as and verified performance data to assess new technologies. Knowledge platforms, such as VA-guiden, act as intermediaries, facilitating information exchange across actor groups. However, decision-making authority remains fragmented, with municipalities differing in their willingness and capacity to engage in innovation. Contractors and manufacturers can operate in hybrid roles, sometimes supporting niche adoption through installation and maintenance of novel systems, yet defaulting to conventional solutions when faced with performance uncertainty.

> *"If I have a proven system as a reference, it's easy to build more. I just need something concrete to show the municipality."* (P03 – Niche – New tech developer/system reseller)

> *"Municipalities are generally not the main obstacle. Our role is to act as an intermediary between property owners and technology developers, focusing on ensuring that applications meet requirements rather than providing detailed advice.*" (P01 – Regime – Environmental inspector)

Barriers and potential facilitators of new technology adoption

Barriers identified include regulatory complexity, inconsistent interpretation of national guidelines across municipalities, limited funding for non-academic innovators, and discontinuity in pilot projects due to political or budgetary changes. Many municipal inspectors prefer familiar systems, viewing them as less risky within the constraints of their legal mandate. Access to credible performance data is often a prerequisite for acceptance, but structured knowledge transfer from trials is lacking. Potential facilitators include clearer and more flexible regulatory pathways for innovation, increased financial support for technology development outside academia, and mechanisms for systematic

dissemination of pilot project outcomes. Opportunities also arise when municipalities have the resources and interest to collaborate with developers, particularly if project costs are borne by the innovator.

> *"It's now extremely difficult to launch new products on the market. Many good ideas fail, not because of the technology, but because the conditions are not ready for them to succeed."* (P05 – Niche – Manufacturer/new tech developer)

> *"It's important to review past projects, learn from them, and make sure that knowledge is shared for future initiatives."* (P03 – Niche – New tech developer/system reseller)

What can we learn about biochar treatment from an MLP perspective on OWTS

The MLP analysis suggests that biochar-based systems currently operate as niche innovations with strong technical potential but face systemic barriers to scaling, including the absence of reference installations and a market-ready product. Their integration into the Swedish OWTS regime depends not only on demonstrated performance for nutrient and micropollutant removal, but also on alignment with institutional processes. Given the regime's cautious approach to novel systems, building legitimacy for biochar treatment requires market-ready designs that can be tested in real-field conditions, ideally through collaboration with all relevant actors from the early stages of design, installation, and monitoring. Starting with smaller-scale applications, such as retrofitting existing systems in phosphorus-sensitive catchments or piloting the removal of micropollutants in areas subject to stricter EU compliance requirements, may provide a more feasible entry point. Dedicating development efforts toward a fully operational MmBF product would facilitate regulatory approval and encourage municipal engagement. Over time, as technical performance is validated and maintenance procedures are established, these early niches could expand into broader adoption, supported by coordinated governance and targeted institutional support.

> *"Involve every stage of the actors from the beginning, find the municipality, find the end-user that we offer the system."* (P05 – Niche – Technology developer)

Conclusions and outlook

This thesis combined a technical evaluation of the MmBF with an institutional analysis of factors influencing the adoption of novel OWTS technologies in Sweden. The work addressed three main objectives: assessing the MmBF's performance in removing organic matter, nutrients, *Escherichia coli*, and pharmaceutically active compounds (PhACs); testing targeted media amendments for enhanced nutrient removal; and exploring institutional and regulatory contexts that affect technology adoption.

Technical conclusions

The long-term pilot-scale experiments demonstrated that the MmBF consistently met national standards for organic matter removal under surface hydraulic loading rates of conventional OWTSs. Over four years operation, COD removal remained above 90% during standard operation and above 85% under doubled HLR (100 L m^{-2} day^{-1}). Nitrification was robust, with near-complete ammonium oxidation in the aerobic modules (M1–M3).

In contrast, denitrification in the original configuration was constrained by the shorter residence time in the anoxic modules (M4–M5), limited carbon availability, and incomplete anoxic conditions due to residual dissolved oxygen. Incorporating sawdust into M4 improved nitrate removal in the short term (24 days), with the higher-level amendment (30 wt%) achieving 80% removal in the denitrification unit M4-M5 and meeting the Swedish total nitrogen performance requirement for high-protection areas.

For phosphorus, unamended MmBFs achieved only minimal removal due to its low content of calcium, aluminium, and iron. Amending M6 with Polonite increased removal to 31 ± 19% during Day 2-24 at the higher amendment level (30 wt%), but this still fell short of the 90% required in high-protection areas.

PhACs removal in MmBFs was consistently high, with mean total reductions of 99.7% maintained after one year of operation without media replacement. Removal was dominated by the first aerobic module, with additional contributions from subsequent stages. These results indicate that the MmBF is capable of delivering high-quality effluent for a broad range of pollutants, with scope for further optimization of nutrient removal.

Socio-technical insights

The qualitative study revealed that while technical performance is essential for legitimacy, adoption depends strongly on municipal regulatory interpretation, availability of reference systems, and pathways for knowledge transfer. Municipalities often default to conventional systems due to risk aversion and limited internal resources for assessing new technologies. Innovators face challenges in securing pilot sites, sustaining funding beyond initial trials, and navigating heterogeneous permit conditions. Knowledge platforms and collaborative pilots can bridge gaps between niche innovators and regime actors, but these opportunities remain sporadic. Clearer regulatory pathways, consistent criteria for technology evaluation, and financial mechanisms that support both research institutions and private developers were identified as key enablers.

Taken together, the findings confirmed that effective OWTS innovation requires both technical optimisation and institutional alignment. The MmBF offers a modular and adaptable platform for decentralised wastewater treatment, capable of reliably removing organic matter, faecal bacteria, nitrogen, and a wide range of micropollutants. Targeted amendments can enhance nutrient removal, but sustaining these effects will require scheduled media replacement and refined hydraulic design to minimise preferential flow.

From an institutional perspective, the results show that for OWTS innovations to gain traction, clear governance frameworks, streamlined approval procedures, and structured data sharing from pilot studies, are essential. Such measures can strengthen trust among municipal regulators and, in some cases, accelerate market introduction. These insights are transferable and can inform the broader understanding of institutional factors influencing the adoption of MmBF and other novel OWTS technologies.

Outlook

Future research and development priorities include:

- Microbial community analysis: identifying dominant functional bacteria in each module to better understand biological processes driving pollutant removal.
- Long-term nutrient removal performance with amendment (sawdust, Polonite): determining optimal application strategies and replacement intervals to sustain phosphorus and nitrogen removal at levels that meet regulatory requirements.
- Hydraulic optimisation: redesigning anoxic modules to improve flow distribution and contact time, thereby enhancing denitrification efficiency.

- Full-scale OWTS trials: implementing MmBFs at single-household or cluster scales to validate pilot findings under real-world loading patterns and maintenance conditions.
- Institutional mechanisms: establishing national protocols for evaluating novel OWTSs, creating databases of performance data, and piloting incentive schemes that support new technology adoption.
- Resource recovery integration: assessing the safe reuse of nutrient-laden biochar as soil amendment, aligning with circular economy and climate mitigation goals.

By advancing both technical configurations and enabling institutional conditions in parallel, biochar-based modular filters such as the MmBF could become a practical and sustainable option for decentralised wastewater treatment, contributing to improved water quality, resource recovery, and resilience in small-community sanitation.

Populärvetenskaplig sammanfattning

Varje hushåll producerar avloppsvatten genom vardagliga aktiviteter som matlagning, bad och toalettanvändning. Vattnet som försvinner ner i avloppet innehåller det organiskt material, näringsämnen, bakterier och läkemedelsrester som kan förorena miljön om det inte behandlas på rätt sätt. I städerna omhändertas avloppsvatten i stora reningsverk men på landsbygdsområden i Sverige är många hushåll beroende av små enskilda avloppsanläggningar. Cirka 860 000 fastigheter använder sådana system men mer än var femte uppfyller inte miljökraven. Näringsämnen från bristfälligt renat avloppsvatten bidrar till algblomning i sjöar och Östersjön, medan läkemedelsrester och bakterier innebär risker för både ekosystem och människors hälsa.

Denna avhandling undersöker en alternativ reningsmetod med användning av biokol, ett träkols-liknande material som framställs av restprodukter från skogs- eller jordbruk. Biokol är strukturellt mycket poröst och därmed kan binda föroreningar och ge en livsmiljö för mikroorganismer som bryter ner kväve och organiskt material. Det erbjuder även klimatnytta genom att lagra kol och kan efter användning återvinnas som gödningsmedel. Ett multimodulärt biokolfilter utvecklades och tre stycken filter drevs i mer än fyra år i Uppsala vattens reningsverk för att undersöka hur väl de kunde behandla riktigt avloppsvatten. Varje filter bestod av 6st moduler som vattnet sekventiellt rann igenom; först 3st aeroba för oxidation och nitrifikation, sen 2st anoxiska för denitrifikation, samt en sista för avskiljning av fekala bakterier med biokol och bark.

Resultaten från experimentet visade att filtren konsekvent avskilde mer än 95 % av det organiska materialet. Totalkvävereduktionen låg runt 71 % men genom att tillsätta sågspån som en långsamt nedbrytbar kolkälla för denitrifikation ökade reduktionen till cirka 83 % i den inledande fasen vilket uppfyller miljökraven. Filtren minskade *Escherichia coli* med 1,8–2,6 log10-enheter vilket motsvarar en reduktion på 98–99,7 %, vilket i sin tur minskar miljö och hälsorisker. Fosforreduktionen var begränsad i de initiala filtren men när Polonite® användes som filtermedium nådde fosforbindningen toppar på 95 %. Effekten avtog dock med tiden och filtermaterialet behövde bytas. Filtren var

även mycket effektiva mot läkemedelsrester: mer än 99 % av vanliga substanser som paracetamol, naproxen och losartan avlägsnades, till och med det svårnedbrytbara carbamazepin reducerades med över 96 %.

För att ett sådant filter ska vara användbart i allmänt bruk räcker det inte med att det fungerar utan att det måste även vara lätt att tillämpa i samhället. Avhandlingen undersökte därför även de institutionella arbetssätten och regelverken för att införa nya tekniker för små avloppssystem. Intervjuer med kommunala tjänstepersoner, produktutvecklare och nationella myndigheter visade på hinder som osäkra regelverk, varierande resurser mellan kommuner och begränsad finansiering för lokala initiativ. Samtidigt identifierades möjligheter som demonstrationsprojekt, kommunalt engagemang och kunskapsplattformar som bygger förtroende och underlättar erfarenhetsutbyte.

Sammantaget visar resultaten att biokolfilter kan ge en tillförlitlig rening av organiskt material, näringsämnen, bakterier och läkemedel och kan därmed utgöra ett framtida hållbart alternativ för hushåll utan anslutning till det kommunala avloppet. För att systemen ska få bredare spridning krävs dock att tekniska framsteg kombineras med stödjande styrning, flexibla regelverk och praktiska finansieringslösningar. Med rätt förutsättningar kan decentraliserad avloppsrening bli mer än en reservlösning. Det kan bli en viktig del i renare vatten, friskare ekosystem och mer robusta sanitetslösningar för framtiden.

一般向け要約 (Popular science summary in Japanese)

炊事や入浴、トイレといった、各家庭での日常的な水の利用により生じる生活排水には、有機物や栄養塩、病原性細菌、さらには医薬品残留物などが含まれており、適切に処理されない場合には水環境汚染を引き起こす可能性があります。スウェーデンでは、都市部では大規模な下水処理場が生活排水の処理を担っていますが、農村部では小規模な分散型排水処理システムに依存する家庭が多く、約 86 万戸のうち 5 戸に 1 戸以上が環境基準を満たしていません。不十分な排水処理は、放流先の湖沼やバルト海で藻類の異常繁殖を引き起こし、さらに医薬品残留物や病原性細菌は生態系や人の健康にリスクをもたらします。

本研究では、森林資源や農業残渣を高温処理することで得られる「バイオチャー（バイオ炭）」を用いた新たな排水処理方法を検討しました。バイオチャーは軽量で多孔質な構造を有し、汚染物質を吸着するだけでなく、窒素や有機物を分解する微生物の住処にもなります。さらに、炭素を長期間土壌に固定できることから気候変動対策としても期待され、使用後は肥料としての再利用も可能です。

本研究では、多層型バイオチャーフィルターを用い、ウプサラの下水処理場において実際の排水を対象に 4 年以上にわたりパイロット実験を行いました。フィルターは 6 層構造であり、上部 3 層は酸化および硝化反応を促す好気条件、下部 2 層は脱窒反応を促す嫌気条件、最下層はバイオチャーと樹皮片を組み合わせることで病原性細菌を除去する仕組みとし、その性能の検証を行いました。

実験の結果、有機物は 95%以上で安定して除去されました。全窒素は平均で 71%除去され、炭素源としておがくずを加えた場合には、初期の除去率が 83%まで向上し、スウェーデン国内の環境基準を満たしました。大腸菌については 98～99.7%が除去され、健康リスクの低減効果が示唆されました。医薬品残留物に関しても高い除去効果が確認され、広く普及しているパラセタモール、ナプロキセン、およびロサルタンは 99%以上、難分解性

とされるカルバマゼピンも 96%以上が除去されました。一方、リンの除去性能は従来のフィルターでは限定的でしたが、Polonite®を添加することで一時的には 95%まで除去可能であることが示されました。

本研究で検討した技術を含む分散型排水処理システムを社会に普及させるためには性能だけでなく、制度的・社会的な条件も重要です。そこで本研究では、自治体職員や技術開発者、国の海洋・水管理庁へのインタビューを通じて、〜〜の(本技術の？)適用可能性と普及に向けた課題を調査しました。その結果、曖昧な分散型処理システムのガイドライン、分散型処理を担当する人材に関する自治体間の格差、および資金不足といった課題がある一方、実証プロジェクトや導入過程への自治体の積極的な関与、知識を共有するための機会提供が新技術の導入を促進しうることが明らかになりました。

総じて、本研究はバイオチャーフィルターが生活排水中の有機物、栄養塩、病原性細菌、及び医薬品残留物を効果的に除去できることを示し、下水道未接続家庭にとって有望かつ持続可能な選択肢となり得ることを明らかにしました。普及にあたっては、技術的改良に加え、支援的なガバナンスや柔軟な規制、そして技術開発者の資金調達の仕組みが欠かせません。これらの条件が整えば、分散型排水処理は単なる代替策にとどまらず、より清浄な水、健全な生態系、そして強靭な衛生システムを実現する重要な手段となるでしょう。

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