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The price of clean water: environmental, economic, and human health implications of PFAS treatment

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Abstract

This thesis examines the use of innovative technologies to treat PFAS-contaminated water and assesses the associated environmental, economic, and human health costs.

First, a literature review and a meta-analysis of published studies are conducted. The meta-analysis consolidates current knowledge on environmental and economic performance of innovative technologies used to treat PFAS-contaminated water. The analysis aims at offering new insights into the climate impacts per gram of PFAS treated and the annual capital and operational costs per volume water treated. The results show large variability in climate impacts, ranging from 0.1 to 70,190 kg CO_2 eq./gram of PFAS treated, largely driven by differences in raw water concentrations. The economic analysis shows that operational costs span from \$0.03/m^3 to \$28/m^3, while capital expenditures range from \$0.01 to \$0.51/m^3 of water treated and exhibit some economies of scale.

From the literature review, methodological limitations in published LCA studies are identified and critically examined to inform and support the development of more robust and comprehensive future LCA studies on PFAS treatment. The analysis specifically examines the extent to which toxicity-related impacts were addressed and characterized in the reviewed LCA studies. Results show that many studies on PFAS treatment do not fully capture the entire life cycle, often overlooking key fate and exposure pathways. Technologies like granular activated carbon (GAC) filters and ion exchange (IEX) resins mainly transfer PFAS to waste streams rather than eliminate them, yet LCAs frequently assume complete destruction and ignore residual emissions and disposal impacts. Additionally, the lack of appropriate characterization factors (CFs) for PFAS limits accurate toxicity assessment, leading to inconsistent methods or omissions. As a result, current LCAs tend to underestimate the true environmental and health impacts of PFAS treatment systems.

Second, a case study is conducted. Five innovative treatment trains (TTs) for PFAS removal from drinking water in Sweden are assessed and compared regarding their environmental and human health impacts. Each treatment train was divided into: (1) pre-treatment (for technologies used previously to the one designated for PFAS removal); (2) PFAS removal (for technologies capable of transferring PFAS from the contaminated water to either solid media or a waste stream, such as GAC, membranes, IEX and FF, including transportation of used media); (3) post-treatment (to ensure that the final drinking water of all TTs has the same characteristics and quality, remineralization and pH adjustment are implemented when needed in addition to disinfection); and (4) PFAS destruction (for technologies that utilize either high temperature alone or a combination of high temperature and pressure to mineralize PFAS compounds, such as reactivation of GAC, and incineration of spent media or waste). The analysis is intended to support informed and transparent decision-making in the design and implementation of PFAS removal strategies in drinking water treatment, and to guide policies on PFAS restrictions. The results indicate that contributions to environmental impact categories vary across the TTs depending on the technologies applied. However, PFAS removal step is the main driver of environmental impacts overall, largely due to its requirements for chemicals, materials, and energy. Post-treatment processes can also contribute significantly, while PFAS destruction generally has a smaller impact, except when GAC reactivation is performed.

Finally, a net human health benefit (NHHB) approach is developed and tested in the case study to serve as a methodological contribution and a complement to the traditional LCA. NHHB is applied to assess whether the avoided PFAS-4–related human health impacts achieved through water treatment outweigh the life cycle impacts induced by DWTP operations, thereby determining whether PFAS

removal provides a net human health benefit relative to a no-treatment scenario. This knowledge could not be captured by the traditional LCA. In the NHHB analysis, to enable a more comprehensive evaluation of the potential range of impacts from direct PFAS ingestion through drinking water, two characterization factors (CFs) are calculated and applied based on effect factors (EFs) derived from (1) repeated-dose rodent studies and (2) epidemiological data extrapolated to non-cancer human lifetime equivalent.

The NHHB results are highly sensitive to the CF choice. With rodent-based cancer and non-cancer human toxicity CFs, treatment burdens outweigh benefits, while epidemiological non-cancer human toxicity CF indicate net health benefits. Although treatment reduces exposure, it shifts impacts upstream via energy, chemicals, and materials production. These findings suggest that prioritizing policy changes focusing on preventing problems by enforcing comprehensive restrictions on PFAS may be more sustainable than relying on downstream water treatment.

List of publications

Paper I

Altmeyer Mendes, S., Aggarwal, R., Svanström, M., & Peters, G. (2025). Review of water treatment technologies for PFAS from a life cycle perspective, with meta-analysis of financial costs and climate impacts. *Resources, Conservation and Recycling*, *223*, 108524. https://doi.org/10.1016/j.resconrec.2025.108524

Paper II

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Other related publications:

Publication A

Altmeyer Mendes, S., Svanström, M., Aggarwal, R., & Peters, G. (2024) Life Cycle Assessment of Innovations in Water Treatment for PFAS Removal: What Do We Know? Extended abstract at the SETAC Europe 26th LCA Symposium. Available at: https://www.lifecyclecenter.se/wp-content/uploads/2024-5-Extended-abstracts-of-the-26th-SETAC-Europe-LCA-Symposium-v2.pdf.

Publication B

Altmeyer Mendes, S., & Peters, G. (2025) Life cycle assessment of emerging PFAS removal technologies in drinking water treatment in Sweden. The 12th Australian Conference on Life Cycle Assessment: Next-generation LCA to support sustainability transitions. Abstract available at: https://www.lcaconference.com.au/_files/ugd/9ffc42_549d3bfc54e54cbc846aa2dc2d08e087.pdf.

Dream.

Believe.

Work.

Achieve.

Acknowledgments

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And to my family back home: mãe e mana, obrigada por sempre apoiarem meus sonhos, mesmo quando isso significa eu ter que me mudar para um outro país, a muitos kms de casa. Ter vocês em minha vida faz essa caminhada ser ainda mais significativa.

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1. Introduction

Access to safe and clean drinking water is essential to public health and has been formally recognized as a human right since 2010 (WHO, 2023). Despite this recognition, drinking water quality and availability face significant challenges both now and in the future. Among the most pressing concerns is the contamination of water systems by persistent per- and polyfluoroalkyl substances (PFAS) (Wang et al., 2022). According to the Organisation for Economic Co-operation and Development (OECD, 2021), PFAS are generally defined, with a few specific exceptions, as any chemical compound that contains at least one fully fluorinated methyl group ($-CF_3$) or fully fluorinated methylene group ($-CF_2-$). Under this definition, PubChem, the largest open chemical data collection with 116 million compounds, now contains over 7 million chemicals classified as PFAS (Schymanski et al., 2023).

These synthetic chemicals have been produced since 1940s and have become a focus of concern due to their extensive use in industrial and consumer products, their persistence in the environment, bioaccumulation potential, toxicity, and widespread occurrence (Ahrens & Bundschuh, 2014). Because of their distinctive physicochemical properties, including both hydrophilic and hydrophobic behavior, PFAS have been widely used in a broad range of consumer and industrial applications for their water and oil repellency, non-stick properties, and resistance to heat and stains (Ellis et al., 2023; Wee & Aris, 2023).

PFAS enter the environment through both point and diffuse sources. Point sources include wastewater treatment plants (WWTPs) (Tisler et al., 2025), landfills (Feng et al., 2021) and firefighter training facilities (Ahrens et al., 2015), while diffuse sources encompass water run-off (Zhao et al., 2013) and atmospheric deposition (Podder et al., 2021). Once released, PFAS persist and accumulate in the environment, including soil, water, air, wildlife, and human tissues, largely due to their high water solubility combined with strong lipid affinity and thermal stability (Ellis et al., 2023; Wee & Aris, 2023).

Around the globe, many water reservoirs have been contaminated with PFAS in the past and present years. For example, in the city of Uppsala, Sweden, drinking water has been contaminated with PFAS since at least 1996, primarily due to the use of aqueous film-forming foams (AFFF) at firefighting training sites, leading to elevated PFAS levels in the blood of citizens (Gyllenhammar et al., 2019; Gyllenhammar et al., 2015). Another case became known in 2013, when approximately 127,000 people in the Veneto Region of Italy were exposed to PFAS through their drinking-water. The contamination resulted primarily from the industrial emissions of a chemical plant in the area that produced these substances. As a result, groundwater, surface water, and drinking water supplies were contaminated across 21 municipalities (WHO, 2017). In Rastatt, Germany, approximately 1,105 hectares of soil and around 490 million m^3 of groundwater have been contaminated with PFAS. The pollution is thought to have resulted from compost blended with PFAS-containing paper sludge, which was applied to agricultural land in the central Baden region between 2006 and 2008 (Stadtwerke Rastatt, 2025).

In Sydney, Australia, the Adams Creek and Medlow Bath drinking water catchments in the Blue Mountains have been contaminated with PFAS, despite being in the middle of a national park. The most likely sources of contamination are the historical use of firefighting foams during two petrol tanker fires in 1992 and 2002, as well as fire system testing activities and uncontrolled environmental releases from a local Rural Fire Brigade station (Parliament NSW, 2025). Across Brazil, PFAS have been detected in many water bodies, with studies conducted over recent decades consistently

identifying PFOS and PFOA as some of the most prevalent compounds in surface waters. This pattern persists despite regulatory efforts aimed at curbing their production and use. The primary sources of PFAS contamination are associated with the leather industry, urban wastewater discharges, and industrial effluents, particularly from chemical, food and beverage, and textile manufacturing sectors (Souza-Araujo et al., 2026; V.M. Starling et al., 2024; Stefano et al., 2023; Quinete et al., 2009).

Alongside food intake, dust ingestion and indoor air inhalation, drinking water represents one of the primary PFAS exposure pathways (EFSA, 2020; Gyllenhammar et al., 2015). Exposure to PFAS has been associated with various adverse health effects in humans, including immune system dysfunction, developmental delays, reproductive problems, hormonal disruptions, liver damage, and certain types of cancer, especially kidney cancer (Wee & Aris, 2023; ATSDR, 2021; Temkin et al., 2020). As awareness of the toxicological risks associated with PFAS has grown, so too have efforts to mitigate their presence in drinking water. This has led to increasingly stringent legal limits for PFAS concentration in drinking water. For example, for 2026, Sweden's Drinking Water Directive sets limit values for PFAS-4[1] and PFAS-21[2] at 4 ng/L and 100 ng/L, respectively (Swedish Food Agency, 2022). Previously, the recommended action level in drinking water was 90 ng/L for PFAS-11[3] (Swedish Food Agency, 2016).

Other countries have different regulations. For instance, Australia's guidelines set limit values for PFOS, PFOA, PFHxS and PFBS at 8 ng/L, 200 ng/L, 30 ng/L and 1000 ng/L, respectively (Australian Government, 2025). The U.S. Environmental Protection Agency (2022) established maximum contaminant levels of 4 ng/L for PFOS and PFOA, and 10 ng/L for PFHxS, PFNA, and HFPO-DA. The Government of Canada (2024) has set a limit of 30 ng/L for the sum of 25 specific PFAS[4], and Denmark presents the lowest limit level of 2 ng/L for PFAS-4 (Danish Environmental Protection Agency, 2025).

However, conventional drinking water treatment methods are often ineffective at removing PFAS (Wee & Aris, 2023; Franke et al., 2021; Belkouteb et al., 2020; Appleman et al., 2014). As a result, research efforts are focused on developing innovative technologies to remove and destroy PFAS from contaminated waters. Treatment approaches currently under investigation include granular activated carbon (GAC) filtration, ion exchange (IEX) resins, membrane filtration, and foam fractionation (FF). Although these methods can effectively separate PFAS from water, they do not destroy the compounds. Instead, they produce PFAS-containing residuals or waste streams, such as used media, that require further handling or treatment. Achieving complete PFAS destruction—defined as the breakdown of PFAS into less harmful end products such as carbon dioxide, water, and fluoride ions—requires more aggressive treatment approaches. Technologies that operate at high temperatures, either alone or in combination with elevated pressure, are necessary to mineralize PFAS compounds, with incineration being a common example. In addition, emerging destruction technologies, including electrochemical oxidation (EO), enhanced contact plasma (ECP), and hydrothermal carbonization, are

[1] Sum of the following four substances: PFOS, PFOA, PFNA and PFHxS. The complete names of the substances can be found in the Supporting Information (SI).
[2] Sum of twenty-one substances: PFBA, PFPA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFUnDA, PFDoDA, PFTrDA, PFBS, PFPS, PFHxS, PFHpS, PFOS, PFNS, PFDS, PFUnDS, PFDoDS, PFTrDS, 6:2 FTS.
[3] Sum of eleven substances: PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFBS, PFHxS, PFOS, 6:2 FTSA.
[4] Sum of twenty-five substances: PFBA, PFNA, PFPeS, 6:2 FTS, PFMBA, PFPeA, PFDA, PFHxS, 8:2 FTS, NFDHA, PFHxA, PFUnA, PFHpS, HFPO-DA, 9Cl-PF3ONS, PFHpA, PFDoA, PFOS, ADONA, 11Cl-PF3OUdS, PFOA, PFBS, 4:2 FTS, PFMPA, PFEESA.

actively being studied for their potential to destroy PFAS (Altiparmaki et al., 2026; Kulkarni et al., 2025).

Despite growing concern over PFAS contamination and the need to remove these substances from drinking water to protect human health, significant knowledge gaps remain. In particular, the environmental and human health impacts and financial costs of innovative treatment processes are not yet well understood, nor are the ecological and health benefits of PFAS removal when assessed from a life cycle perspective. Addressing these gaps is essential to inform decision-making for future drinking water treatment strategies and for guiding policies on PFAS restrictions.

2. Research questions

To address the identified research gaps, this study evaluates the environmental, human health, and financial costs of PFAS remediation in drinking water production from a life cycle perspective. Based on this objective, four research questions are formulated to guide the analysis.

RQ1: What do published studies on life cycle assessment and cost analyses say about the environmental and financial performance of innovative PFAS treatment technologies?

This research question focuses on synthesizing the current state of knowledge on life cycle assessment (LCA) and cost analyses of PFAS treatment technologies. It aims to consolidate insights into the environmental and economic impacts associated with PFAS-contaminated water treatment, with particular attention to the role of the functional unit, and to annual capital and operational costs normalized per volume of water treated.

In addition, RQ1 seeks to identify the dominant contributors to environmental burdens and costs for different treatment trains (TTs). A further objective is to list the technologies that have been assessed to date. Thereby, RQ1 establishes a foundation for identifying research gaps and informing the subsequent research questions.

RQ2: To what extent do published studies fall short in representing the life cycle environmental impacts of innovative PFAS-contaminated water treatment?

This research question aims to identify and critically examine methodological limitations in published LCA studies of PFAS-contaminated water treatment systems. By systematically mapping these shortcomings, RQ2 seeks to clarify where existing frameworks fail to capture the complete life cycle environmental impacts of emerging PFAS treatment technologies.

The insights gained are intended to inform and support the development of more robust and comprehensive future LCA studies. Ultimately, this work aims to improve the evaluation of PFAS treatment systems, leading to assessments that are more balanced, transparent, and representative of real-world conditions.

RQ3: What are the impacts on human health and the environment of applying innovative technologies to treat PFAS-contaminated water in Sweden?

The purpose of this research question is to assess and compare the impacts of innovative TTs designed and tested for PFAS removal from drinking water in Sweden. The analysis focuses on systems developed to comply with the Swedish Drinking Water Directive limit value of 4 ng/L for PFAS-4.

Beyond evaluating PFAS removal efficiency, RQ3 extends the assessment to include the environmental and human health impacts associated with the operational life cycle of the treatment systems, applying a more robust LCA approach raised from gathered knowledge from previous RQs.

RQ4: Does removing PFAS from drinking water in fact result in net human health benefit when assessed from a life cycle perspective?

This research question focuses on understanding better the human health burdens and benefits achieved through treatment that could not be captured by the traditional LCA. RQ4 aims at assessing whether the avoided PFAS-4–related human health impacts achieved through water treatment outweigh the life cycle impacts induced by DWTP operations, thereby determining whether PFAS removal provides a net human health benefit relative to a no-treatment scenario. To support policy development, it is

meaningful to ask whether PFAS removal in fact results in a net human health benefit when assessed from a life cycle perspective.

For that, a net human health benefit (NHHB) approach is developed and tested to account for the health effects of residual PFAS in treated water and to capture health impacts not reflected in the traditional LCA. This approach provides a scientific contribution to the methodological development of assessing the burdens and benefits of innovative treatment systems for PFAS-contaminated water.

In addition, RQ4 aims to enable a more comprehensive evaluation of the potential range of impacts from direct PFAS ingestion through drinking water. For that, two characterization factors (CFs) based on effect factors (EFs) derived from (1) repeated-dose rodent studies and (2) epidemiological data extrapolated to non-cancer human lifetime equivalent are calculated and applied in the NHHB approach. The findings of this research question are intended to support informed and transparent decision-making in the design and implementation of PFAS removal strategies in drinking water treatment, and to guide policies on PFAS restrictions.

Together, these four research questions serve the overarching aim of this thesis: to critically evaluate the environmental, human health and financial costs of innovative treatment technologies for PFAS-contaminated water from a life cycle perspective. By integrating insights from life cycle assessment, costs and NHHB analyses, the thesis provides decision-relevant knowledge to support strategic and practical planning in the water industry, to assist utilities and stakeholders in making informed investment decisions for PFAS treatment in real-world applications, and to inform the development of more effective and evidence-based policies for managing PFAS restrictions and contaminations.

3. Theoretical background

The papers underpinning this thesis, and the thesis itself, are situated within the scientific field of environmental systems analysis, a discipline rooted in industrial ecology, and within the theoretical systems perspective of life cycle.

In the environmental systems analysis field, systems-oriented approaches are applied to evaluate environmental and human health impacts of technologies and services. A central methodological framework in this field is life cycle assessment (LCA), a standardized approach (ISO 14040:2006 and 14044:2006) for quantifying environmental impacts associated with resource use and emissions across the life cycle of a product or system (Hauschild et al., 2018; Guinée et al., 2004).

The application of LCA reflects a theoretical systems perspective, in which technologies are understood as embedded within interconnected socio-technical and environmental systems (Hauschild et al., 2018). Rather than focusing on isolated processes, this perspective considers upstream and downstream processes linked by the economic supply chain, enabling the identification of trade-offs and burden shifting between life cycle stages (Baumann & Tillman, 2004; Guinée et al., 2004).

LCA modelling further relies on a set of methodological and theoretical assumptions, including the definition of system boundaries, the use of a functional unit, and the characterization of emissions into potential impacts through cause-effect chains (Hauschild et al., 2018; Baumann & Tillman, 2004). LCA can be contrasted with other analytical tools available for inventorying and analyzing the environmental aspects of a given object. For example, risk assessment is a tool used to assess the adverse effects of an activity on human safety and ecosystems. However, the analysis always focuses on the risks that a particular human activity poses to the surrounding area and is therefore always site-specific (Guinée et al., 2004). While risk assessment is well suited for evaluating the health risks associated with PFAS exposure through drinking water, LCA provides a broader perspective by accounting for environmental and human health impacts across the entire life cycle of treatment technologies. These approaches are complementary but reflect different theoretical perspectives: risk assessment emphasizes localized, exposure-driven impacts, whereas LCA emphasizes system-wide trade-offs and indirect effects.

Within the same life cycle perspective, life cycle costing (LCC) complements LCA by assessing the economic dimension of systems, accounting for costs incurred over their entire lifespan, including capital and operational expenditures. By incorporating discount rates to reflect the time value of money, LCC enables the evaluation of investments in terms of their net present value (Hunkeler et al., 2008; Rödger et al., 2018). This is particularly relevant for water treatment technologies, where high costs can influence implementation decisions (Valladares Linares et al., 2016). The combined use of LCA and LCC reflects an integrated systems perspective, where environmental and economic trade-offs can be evaluated consistently across the life cycle.

3.1 Life cycle assessment (LCA)

LCA is a systematic and comprehensive method used to evaluate the potential environmental impacts and the resources used throughout a product, process, or service entire life cycle (ISO, 2006a). This method takes a life cycle perspective, e.g., covers a product system from the extraction of raw materials to the end-of-life treatment. The assessment is based on the study of resource flows from technical activities and their effects on the environment.

An LCA is carried out in four main steps (Baumann & Tillman, 2004; ISO, 2006a), as shown in Figure 1. First, the goal and scope are defined. This includes specifying the purpose of the study and the methodological choices that follow from it. The system boundaries, the alternatives to be analyzed, and the functional unit are established. The functional unit represents the function of the system under study and provides the basis for comparison both among the analyzed alternatives and with other systems delivering the same function. In addition, the environmental impact categories to be assessed are selected. These may range from a broad set of categories to a limited number of key indicators, and can include midpoint indicators (i.e., indicators located along the cause-effect chain, such as climate change, mineral resource depletion, or freshwater eutrophication) and/or endpoint indicators (i.e., indicators representing damage to areas of protection, namely human health, resource availability, and ecosystems, thereby aggregating multiple midpoint impacts).

Next, the inventory analysis is carried out by compiling data on all relevant activities and flows within the system. These flows include inputs and outputs of materials and energy, products, emissions, waste streams, and other physical inputs such as land use. In the impact assessment phase, each flow is evaluated in terms of its contribution to the selected impact categories. The potential environmental impacts of the product system are then calculated by aggregating these contributions within each category. Finally, the interpretation phase is conducted, where the results are analyzed in relation to the study's goal and scope, as well as the methodological choices made (Baumann & Tillman, 2004; ISO, 2006a).

The assessment is conducted iteratively, with insights from each stage informing subsequent methodological choices. For example, the analysis of preliminary results may prompt additional data collection for processes with significant contributions or lead to the inclusion of sensitivity analyses for selected input parameters (Peters & Svanström, 2019).

The conclusions from an LCA can support decision-making processes in selecting a more environmentally friendly process, in product design and development as well as in public policy. This is achieved by identifying improvement opportunities and environmental hotspots across the life cycle, or by comparing alternatives that deliver the same function (Baumann & Tillman, 2004).

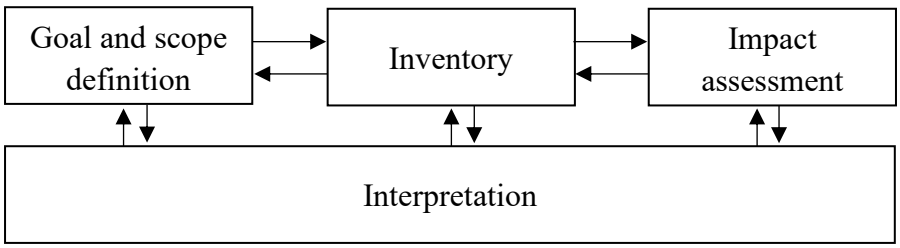


Figure 1 - The LCA procedure (ISO, 2006a)

3.1.1 Attributional and consequential LCA

There are two main life cycle inventory (LCI) modeling frameworks: attributional and consequential LCA. While there are different definitions of the two types, similar characteristics can be seen in different definitions (Ekvall et al., 2016). The purpose of attributional LCA can be described as quantifying impacts of existing product systems or existing systems projected into some stable but

different (geographical, temporal or other) context. This implies that average data of the processes is used. Multifunctional processes are handled by partitioning the environmental burden between the products (Hauschild et al., 2018).

Consequential LCA instead aims to quantify how the system might change due to a certain decision (Ekvall, 2020). In terms of data, it should reflect the changed flows and therefore marginal data should be used. Multifunctional processes are dealt with by substitution. Both attributional and consequential studies can be retrospective or prospective (Arvidsson et al., 2018).

4. Applied methods

This section summarizes the methodologies used in the papers and in this licentiate thesis.

Figure 2 presents an overview of the applied approach for this thesis. Section 4.1 describes the method used to select, review, and integrate data from published studies for providing new insights into the environmental impacts as well as the annual capital and operational costs of treating PFAS-contaminated water. Section 4.2 presents the method used for understanding the shortcomings of published LCAs on innovative PFAS treatment technologies. Section 4.3 shows the LCA methodology applied for calculating human health and environmental impacts of applying innovative technologies to treat PFAS-contaminated water in Sweden, as well as the developed NHHB approach.

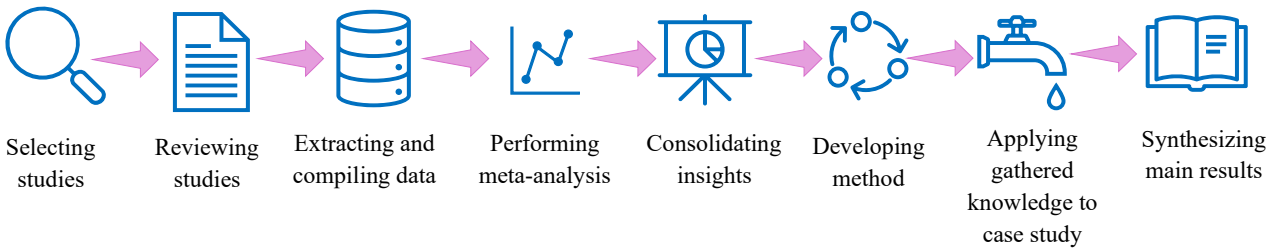


Figure 2 - Overview of applied approach for this licentiate thesis

4.1 Compilation of data

To understand what published LCA studies and cost analyses report about the environmental and financial performance of emerging PFAS removal technologies, a scoping review (Sutton et al., 2019; Munn et al., 2018; Arksey & O'Malley, 2005) was conducted. In total, 17 articles published between 2018 and 2024 were selected. Eight of these articles focused on LCA. Among them, four also included an economic assessment of PFAS treatment. Two applied LCC, while the other two evaluated specific cost components, such as capital investment and operation and maintenance expenses.

Of the selected seventeen published papers, one study focused exclusively on LCC to compare the costs of different PFAS treatment technologies over a defined payback period. Because only a limited number of LCC studies were identified, the review also includes eight articles in which the authors analyze only the price of water treatment (Table 1).

4.1.1 Synthesizing information on environmental impacts of PFAS treatment technologies

Results from multiple LCA studies were synthesized through quantitative methods. A meta-analysis was conducted by collecting data from the LCA studies to achieve a more comprehensive understanding of the environmental impacts associated with PFAS treatment. In total, seven published LCA studies provided sufficient data for the meta-analysis.

Since contributions to climate change (CCC) was the sole environmental impact category examined across all the LCA studies reviewed, it was selected for the meta-analysis to consolidate data on CO_2 equivalent emissions generated during treatment of PFAS-contaminated water.

In total, 30 TTs were extracted from the LCA studies and included in the meta-analysis. The functional unit selected as a basis for comparing the results was one gram of PFAS treated[5], as it directly reflects the primary function of TTs designed for PFAS-contaminated water. This unit represents the total load of PFAS removed from the contaminated source. Thus, the results from the meta-analysis were expressed in kg CO_2 per gram of PFAS treated.

[5] "Treated" refers to the concentration of PFAS that has been removed and/or destroyed, depending on the applied technology, and scaled according to the treatment efficiency.

Table 1 - Overview of identified and selected studies for the literature review and meta-analysis

References	LCA	LCC	Cost analysis	Selected for literature review	Included in the meta-analysis of CCC	Included in the meta-analysis of costs
Zhu et al., 2013	✓					
Bixler et al., 2021	✓			✓	✓	
Boyer et al., 2021	✓			✓	✓	
Maga et al., 2021	✓			✓	✓	
Li et al., 2022a	✓			✓	✓	
Mohamed et al., 2023	✓					
García-Castrillo et al., 2024	✓					
Morales et al., 2024	✓					
Feng et al., 2021	✓	✓		✓	✓	✓
Ellis et al., 2023	✓	✓		✓	✓	✓
Emery et al., 2019	✓		✓	✓	✓	✓
Laramay & Crimi, 2020	✓		✓			
Moeini et al., 2022	✓		✓	✓		✓
Li et al., 2022b	✓					
McNamara et al., 2018			✓	✓		✓
Quinnan et al., 2023		✓		✓		✓
Belkouteb et al., 2020			✓	✓		✓
Franke et al., 2021			✓	✓		✓
Murray et al., 2021			✓	✓		✓
Kanchanapiya & Tantisattayakul, 2022			✓	✓		✓
Medina et al., 2022			✓			
Ersan et al., 2023			✓			
Ling et al., 2023			✓	✓		✓
Malovanyy et al., 2023			✓	✓		✓
Murray et al., 2023			✓			
Riegel et al., 2023			✓			
Ophorst et al., 2024			✓			
Jiang et al., 2024			✓	✓		✓

4.1.2 Synthesizing information on capital and operational costs of treating PFAS-contaminated water

13 studies were reviewed to evaluate the costs of PFAS treatment technologies, from which 59 distinct treatment scenarios were identified and included in this analysis. The economic assessment follows a life cycle cost (LCC) framework, with costs categorized into capital expenditure (CAPEX) and operational expenditure (OPEX). Decommissioning costs were excluded, as they were not addressed in the reviewed literature.

CAPEX represents the upfront investment required for project development and construction. This typically includes costs related to equipment procurement, infrastructure installation, and construction activities. OPEX encompasses the recurring expenses incurred over the operational lifetime of the treatment system, including operation and maintenance costs. These may involve electricity consumption, replacement of adsorbents or treatment media, chemical inputs, transportation, disposal of spent materials, and, in some cases, labor costs. Among the 59 treatment scenarios analyzed, 17 reported capital investment costs, while all scenarios provided data on operation and maintenance expenses.

Both CAPEX and OPEX were annualized based on the project lifetimes specified in the respective studies. Equation (1) was used to calculate annualized CAPEX. When a study did not specify a project lifetime, a 20-year time horizon was assumed for CAPEX amortization, consistent with information from other studies included in the review. Equation (2) was applied to determine the annual CAPEX per cubic meter of PFAS-contaminated water treated. Annual treatment volumes were calculated using data reported in the selected studies. Additionally, the discount rates applied in the LCC analyses—used to estimate the present value of future costs over the technology lifecycle—were extracted and recorded.

Eq. (1)

$$\text{Annual CAPEX} = \frac{\text{Total CAPEX}}{\text{time horizon set for payment}}$$

Eq. (2)

$$\text{Annual } \frac{\text{CAPEX}}{\text{m}^3} \text{ of water treated} = \frac{\text{Annual CAPEX}}{\text{volume of water treated per year}}$$

For OPEX, most studies report operational costs on an annual basis. Equation (3) was used to determine the annual OPEX per cubic meter of treated water. When studies provided the volume of PFAS-contaminated water treated on a daily or hourly basis, continuous operation was assumed (24 hours per day and 365 days per year), unless an alternative operating schedule was explicitly specified.

Eq. (3)

$$\text{Annual } \frac{\text{OPEX}}{\text{m}^3} \text{ of water treated} = \frac{\text{Annual OPEX}}{\text{volume of water treated per year}}$$

4.1.3 Statistical analysis

To provide insights into different variables that influence climate impact and financial aspects, regression analyses were conducted on parameters that could be extracted from the literature. The strength and nature of the correlations were assessed using the coefficient of determination (R^2), while p-values ($p < 0.05$) were used to indicate statistical significance. Specifically, these analyses explored the extent to which climate impact and annual treatment costs for PFAS-contaminated water are driven by inlet concentration, PFAS concentration reduction, outlet concentration, volume of water treated per year, plant scale and the age of the study. In addition to these factors that suited regression analysis, the choice of technology was considered, and a hotspot analysis was performed to identify the parts of the process life cycle that cause the most environmental damage.

4.2 Understanding the shortcomings of published LCAs

According to ISO 14044, LCA practitioners should select impact indicators that appropriately reflect both the environmental issues associated with the system under study and the overall goal of the assessment. Consistent with this guidance, the reviewed LCA studies collectively evaluated 13 impact categories: climate change, ozone depletion, photochemical ozone formation, acidification, eutrophication, fossil fuel depletion, ecotoxicity, respiratory effects, cumulative energy demand, carcinogenic and non-carcinogenic disability-adjusted life years (DALYs), ionizing radiation, and resource use.

Among these categories, toxicity-related indicators—particularly human toxicity (cancer and non-cancer effects) and ecotoxicity—are especially relevant for technologies designed to remove hazardous substances such as PFAS. Because these treatment systems are intended to mitigate toxicological risks to both human health and ecosystems, this analysis specifically examines the extent to which toxicity-related impacts were addressed and characterized in the reviewed LCA studies.

4.3 Life cycle assessment case study

The aim of the case study is to understand the environmental and human health costs of treating PFAS-contaminated groundwater using innovative technologies to achieve a PFAS-4 concentration below 4 ng/L. The LCA goal is to inform utility managers, analysts and technology developers about the key hotspots for these emerging technologies to support informed decision-making in drinking water treatment.

To this end, an attributional cradle-to-gate LCA was conducted for five TTs applied to treat PFAS-contaminated groundwater for drinking purposes in Uppsala, Sweden. Each TT includes four main steps: pre-treatment, PFAS removal, post-treatment, and PFAS destruction (Figure 3). The TTs are primarily distinguished by their PFAS removal technologies:

- TT 1: granular activated carbon (GAC) filters and off-site thermal reactivation of used GAC performed in Germany[6].

[6] No off-site thermal reactivation facilities in Sweden are currently equipped to receive the spent GAC. Consequently, the drinking water treatment plant (DWTP) transports the material for treatment at a plant in Premnitz, Germany.

- TT 2: closed-circuit membrane filtration (CCMF, with 89% recovery rate followed by concentrate treatment using a pellet reactor (PR) for softening, then foam-fractionation (FF) and incineration of foam .
- TT 3: two-stage nanofiltration (NF, 80% recovery rate) followed by concentrate treatment with foam-fractionation (FF) and incineration of foam.
- TT 4: two-stage nanofiltration (NF, 80% recovery rate) followed by concentrate treatment with GAC, and off-site thermal reactivation of spent GAC performed in Germany.
- TT 5: two-stage nanofiltration (NF, 80% recovery rate) followed by concentrate treatment with ion exchange resin (IEX) and incineration of spent resin.

Although the specific processes differ between TTs, each treatment train includes steps to condition the raw water and protect downstream processes and reduce water hardness. For TT 1, the pre-treatment steps encompass aeration, softening (using a fluidized bed pellet reactor), and rapid sand and anthracite filtration. During the softening process, approximately 3,340 tonnes of calcium-sand pellets are generated yearly, from which 75% is reused as a pH-adjustment product for lakes and 25% are treated as waste and is sent to landfill. It is important to note that softening is a core treatment step in TT 1. However, for the purpose of classifying the treatment processes, softening is grouped to pre-treatment step solely because it precedes the PFAS removal stage.

For TTs 2 to 5, aeration, biofiltration, and a 5 µm pre-filter are used as pre-treatment steps, mainly for helping prevent membrane fouling. In TTs 2 to 5, additional hardness reduction occurs during the PFAS removal stage, as CCMF and NF also remove hardness from the water. The untreated groundwater is classified as hard water (hardness ranging from 16 to 19 °dH). After treatment, the water is classified as soft for all TTs, making them comparable.

Within post-treatment, remineralization and pH adjustment are implemented in membrane filtration scenarios (TT 2–TT 5) to ensure that the final drinking water has the same characteristics and quality as the treated water in TT 1. For that, slaked lime (calcium hydroxide, $Ca(OH)_2$), formed by the reaction of calcium oxide with water ($CaO + H_2O \rightarrow Ca(OH)_2$), is added to the finished water to provide stable water to protect the water network from corrosion. Additionally, disinfection with sodium hypochlorite is applied across all TTs. For the PFAS destruction step, off-site thermal reactivation is considered for used GAC, and incineration for spent IEX, used membranes and PFAS-laden residuals (e.g., foam from FF).

Inventory data was collected from various sources. Foreground data, such as the quantity of chemicals, electricity and the media used, were obtained from: (1) operational and production data from 2024 and 2025 from full-scale drinking water treatment plants (DWTPs), assuming that conditions will remain representative for 2026; and (2) pilot studies conducted between 2019 and 2025. When needed, additional data was sought from relevant literature, and from personal communications with experts from the drinking water industry. Background data was sourced from the Ecoinvent database (version 3.11, content version 2025.1). The remaining concentration of PFAS-4 in the drinking water (<4 ng/L) was included in the assessment.

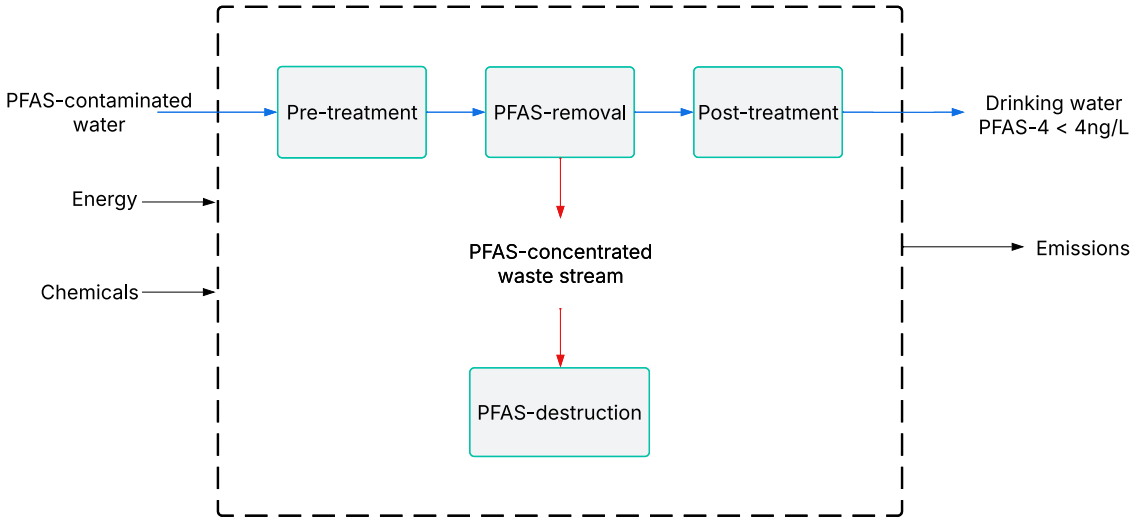


Figure 3 - General system boundaries of PFAS-contaminated water treatment trains

The systems investigated were modeled using LCA for Experts software (Sphera, 2026). The following impact categories were chosen for presentation and discussion of results with the intention of covering the main possible environmental and human health impacts originating from the PFAS-contaminated water treatment processes: human toxicity, ecotoxicity, climate change, photochemical ozone formation, terrestrial acidification, and eutrophication. Both the human toxicity and ecotoxicity impact categories reflect the release of chemicals to the environment and the pathways of exposure to humans and biota, whereas acidification and eutrophication reflect environmental pollution. Photochemical ozone formation was included to account for emissions of NO_x, VOCs, and CO arising from energy use, chemical production, and transport associated with PFAS treatment. Climate change was included to quantify greenhouse gas emissions and to reflect national Swedish reduction targets.

For calculating the environmental and human health toxicity related impacts, Usetox 2.14 method was used. For accounting for PFAS-4 flows leaving the system, the freshwater compartment midpoint characterization factors (CFs) for ecotoxicity provided by Aggarwal et al. (2024) was used in addition to the midpoint human health PFOA CFs from Holmquist et al. (2020). As CFs for human health are available only for PFOA, the flows of PFNA, PFOS, and PFHxS were converted to PFOA equivalents to represent total PFAS-4 flows, applying the relative potency factor (RPF) approach described by Zeilmaker et al. (2018).

In this approach, RPFs are calculated using liver toxicity after oral exposure in the male rat as the common effect. The liver is one of the main target organs of PFOA and other PFAS because exposure in rats generally results in liver hypertrophy (hepatocellular, centrilobular) and accompanying liver enlargement (i.e., absolute and relative liver weight increases). At this point in time, this endpoint is considered one of the most data-rich choices for derivation of RPFs, and PFOA is chosen as the index compound because it is one of the best-studied perfluoroalkyl acids (PFAA) (Zeilmaker et al., 2018).

Converting PFAS flows into PFOA equivalents ensures that differences in toxic potency are accounted for and enables the inclusion of multiple PFAS in the impact assessment when substance-specific CFs are unavailable or cannot be derived by the LCA analyst due to data or time limitations. However, this approach does not fully reflect differences in chemical risk, as the applied RPFs are based on relative

toxic potency for a single endpoint (rat liver toxicity) and do not account for substance-specific variations in toxicokinetics, environmental fate, exposure pathways, or broader health effects embedded in CFs.

For calculating climate change, photochemical ozone formation, terrestrial acidification, and eutrophication impact categories, ReciPe 2016 method at midpoint (H) level was used. The contributions of the TTs to climate change were subsequently expressed as CO_2-equivalent emissions per gram of PFAS treated, enabling comparison of the case study results with findings from the previously conducted meta-analysis.

4.3.1 Net human health benefit (NHHB) analysis

In addition to performing a traditional LCA, a NHHB approach is developed and tested to understand better the human health impacts and benefits achieved through treatment that could not be captured by the LCA. Previous LCAs generally do not account for the health effects of residual PFAS in treated water, either because the end user is not specified or because concentrations are reduced to legally "safe" levels (Ellis et al., 2023; Li et al., 2022; Emery et al., 2019).

The NHHB approach addresses this gap by comparing the avoided PFAS-4–related human health impacts achieved through water treatment with the life cycle impacts induced by DWTP operations. This enables an assessment of whether PFAS removal provides a net benefit relative to the no treatment (null) scenario.

The NHHB is developed from the net environmental benefit (NEB) analysis by Godin et al. (2012) and addresses the following question: what net human health benefit is provided by a DWTP designed to comply with stringent PFAS-4 drinking water regulations (<4 ng/L)? Another way to put it: are the environmental and human health costs of reducing the risks associated with ingesting PFAS-contaminated water justified?

NHHB represents a novel adaptation of the NEB framework, originally developed for wastewater treatment systems. While it builds on the core NEB equation, it differs in both scope and formulation. In particular, NHHB shifts the focus from wastewater to drinking water, from general environmental impact categories to human health toxicity expressed in DALYs, and to general contaminants released to the environment to PFAS ingestion through drinking water. It also refines the underlying calculations to better represent human exposure pathways and incorporates PFAS characterization factors derived from both rodent and epidemiological data to capture PFAS-related health effects.

To this end, endpoint impact categories expressed in DALYs (disability adjusted life years) were calculated. The midpoint human toxicity cancer and non-cancer LCA results were converted from CTUh (comparative toxic units) to DALY applying the conversion factors of 11.5 DALYs and 2.7 DALYs, respectively, derived from Huijbregts et al. (2005). The impacts of climate change, fine particulate matter formation and freshwater consumption on human health were also calculated, but applying ReciPe 2016 (H) method. These endpoint impact categories were then used for calculating the NHHB, as described below.

First, a null option and a water treatment option are defined. The null option assumes that consumers are exposed to the PFAS-4 concentration in the raw groundwater prior to the implementation of any additional PFAS removal treatment, representing a scenario without PFAS removal. This scenario

establishes a baseline against which the advantages and disadvantages of the proposed DWTP are evaluated. The water treatment option accounts for the operation of PFAS removal and destruction technologies, as well as for people exposed to the treated water quality. The NHHB is therefore defined as the difference between avoided and induced potential human health impacts (PI) associated with the DWTP and is calculated for the FU using Equation (4):

Eq. (4)

$$NHHB = (PINO - PITW) - PILCTW$$

where PINO = potential impact from null option (treating water without PFAS removal and destruction steps), PITW = potential impact from the PFAS-treated drinking water and associated wastewater streams (i.e., PFAS-laden residuals generated during drinking water treatment), and PILCTW = potential induced impact of the resources extracted, and the emissions generated by the life cycle of the water treatment steps applied for PFAS removal and destruction.

Although the primary function of drinking water production is human consumption, only a small fraction of the produced drinking water is ultimately ingested, meaning that only a percentage of PFAS remaining in the drinking water is ingested. Even when assuming a high-daily per-capita water intake for drinking and cooking of 2.5 L/person.day (EFSA, 2010), empirical data indicate that only about 1.2% of produced drinking water is actually ingested, on average. The remaining 98.8% is assumed to enter the wastewater system or other pathways and to be discharged into freshwater.

Most wastewater treatment plants (WWTPs) rely on conventional treatment technologies (e.g., biological treatment, oxidation, coagulation–flocculation), which have been shown to be largely ineffective at removing PFAS (Liu et al., 2026; Kim et al., 2024; Müller et al., 2023). As a result, the majority of PFAS present in drinking water is eventually discharged back into freshwater bodies via treated wastewater.

For calculating PINO and PITW for toxicity related human health impacts, the following equations were applied.

Eq. (5)

$$PINO = B_{NO} + [(PFAS_{rw} * 98.8\%) * (PFAS\ CF_{fw})] + [(PFAS_{rw} * 1.2\%) * (PFAS\ CF_{di})]$$

where B_{NO} = burdens of DWTP operation without PFAS removal/destruction steps, $PFAS_{rw}$ = total PFAS load per FU in raw water, PFAS CF_{fw} = human health midpoint characterization factor for PFAS from emissions to freshwater, and PFAS CF_{di} = PFAS CF calculated for the directly ingested portion of water produced.

Eq. (6)

$$PITW = \{[(PFAS_{dw} * 98.8\%) + (PFAS_{tww})] * (PFAS\ CF_{fw})\} + [(PFAS_{rw} * 1.2\%) * (PFAS\ CF_{di})]$$

where $PFAS_{dw}$ = total PFAS load per FU in treated drinking water, and $PFAS_{tww}$ = total PFAS load per FU in treated wastewater.

For calculating PILCTW, Equation (7) was used.

Eq. (7)

$$PILCTW = B_{PFASr} + B_{PFASd}$$

where B_{PFASr} = burdens of PFAS removal step, and B_{PFASd} = burdens of PFAS destruction step.

The PFAS CF_{fw} applied for human toxicity cancer and non-cancer are the midpoint human health PFOA CFs (1.26×10^{-5} and 4.68×10^{-3} cases/kg emitted, respectively) taken from Holmquist et al. (2020). These CFs correspond to emissions to the freshwater compartment and are based on the USEtox framework, i.e., CF = fate factor (FF) × exposure factor (XF) × effect factor (EF), linking emissions to human health impacts (Fantke et al., 2017; Rosenbaum et al., 2011). The effect factors are based on USEtox norms using rodent laboratory toxicological test results (Rosenbaum et al., 2011).

At this point in time, many LCAs do not consider the residual PFAS in the treated water as having any health effects, either because the user of the water is not specified, or because a legally "safe" concentration is reached by the treatment (read more in Altmeyer Mendes et al., 2025). So, to additionally account for human toxicity from direct ingestion of PFAS-4 via drinking water, the cancer and non-cancer related effect factor (EF) for PFOA reported by Holmquist et al. (2020) was taken to calculate PFAS CF_{di}. These EFs are based on the ED_{50} values (effective dose affecting 50% of exposed individuals for a defined end-point) derived from repeated-dose rodent studies compiled in their work. In the PFAS CF_{di} calculations, fate and exposure factors were set to 1, assuming that the emitted PFAS-4 mass is directly and completely ingested by humans, with no environmental processes in between. This is a novel approach that isolates the human-health toxicity associated with direct oral intake while avoiding double-counting with the freshwater emission pathway. The resulting cancer PFOA CF_{di} (EF_{rodent}) is equal to 2.45×10^{-2} cases/kg emitted, and the non-cancer PFOA CF_{di} (EF_{rodent}) is 9.09 cases/kg emitted.

The PFAS-4 limit value of 4 ng/L established by the Swedish Food Agency is based on a report from the European Food Safety Authority (EFSA, 2020) that established a tolerable weekly intake (TWI) of 4.4 ng per kg body weight per week for the sum of PFAS-4, reflecting the importance of long-term accumulation. To explore implications of this high potential toxicity, epidemiological data roughly extrapolated to non-cancer human lifetime equivalent (PFOA ED_{50} of 1.30×10^{-5} kg/lifetime) reported in Holmquist et al. (2020) was used to derive another characterization factor for the non-cancer human toxicity impact of direct ingestion of water, PFOA CF_{di} (EF_{human}), which resulted in 3.85×10^{4} cases/kg emitted. Applying these CFs in the human health toxicity assessment enables a more comprehensive evaluation of the potential range of impacts from PFAS ingestion through drinking water.

After applying the NHHB method in the case study, an evaluation was conducted to assess its effectiveness. The purpose of this evaluation was to determine whether the proposed approach successfully addressed RQ4, as well as to examine the level of difficulty involved in its application.

5. Results

Sections 5.1 to 5.3 present a selection of results from Paper I, while section 5.4 provides a summary of results from Paper II and compares them with the findings from Paper I. Further details can be found in the respective papers.

5.1 Overview of PFAS treatment technologies

Innovative treatment technologies are necessary for treating water contaminated with PFAS as conventional treatment methods are proven inadequate in eliminating them. These technologies can be classified as PFAS removal or PFAS destruction technologies. For PFAS removal, processes such as granular activated carbon (GAC) filters, ion exchange (IEX) resins, nanofiltration (NF) membranes, and foam-fractionation (FF) can be applied. These technologies may be combined in multiple configurations, resulting in distinct TTs. They can also generate PFAS-laden residuals or wastes that require further treatment or disposal.

For PFAS destruction, technologies that utilize either high temperature alone or a combination of high temperature and pressure to mineralize PFAS compounds into carbon dioxide, water, and fluoride ions, such as reactivation of GAC, and incineration of spent media or waste, are usually applied. More recent technologies, such as electrochemical oxidation (EO), enhanced contact plasma (ECP) and hydrothermal carbonization, are being studied as potential methods for PFAS destruction (Altiparmaki et al., 2026; Kulkarni et al., 2025).

From the published LCA studies, 30 treatment scenarios were identified and analyzed. GAC and IEX resins were the primary technologies evaluated, appearing in 62% and 75% of the studies, respectively. This highlights their status as the most commonly applied ex situ adsorbents for removing PFAS from water streams, as also noted by Kulkarni et al. (2025).

5.2 Environmental and economic performance of PFAS treatment technologies in earlier studies

5.2.1 Climate impacts of PFAS treatment technologies

Across the analyzed scenarios, the median climate impact of PFAS treatment was 88 kg CO_2 eq. per gram of PFAS treated. When disaggregated, removal accounted for a median of 80 kg CO_2 eq./g (Figure 4), while destruction contributed 23 kg CO_2 eq./g (Figure 5). This indicates that PFAS removal generally imposes a substantially higher climate burden than destruction, likely due to the more energy- and resource-intensive nature of separation processes compared to the treatment of concentrated residuals.

In 53% of cases, innovative TTs emitted between 10 and 1,000 kg CO_2 eq./g PFAS treated. GAC typically showed relatively low emissions (1–80 kg CO_2 eq./g PFAS removed), although notable outliers occurred at very low influent concentrations (<0.7 µg/L) combined with high removal efficiencies (>86%), where emissions increased dramatically (up to 13,321 kg CO_2 eq./g). IEX systems exhibited a wider range of greenhouse emissions (0.4-462 kg CO_2 eq./g), while membrane filtration processes generally showed a narrower range (82 to 122 kg CO_2 eq./g). Small-scale point-of-use

(POU) systems (usually applied under the kitchen sink) combining GAC, IEX, and reverse osmosis showed the highest impacts, reaching 57,900-70,190 kg CO_2 eq./g PFAS removed.

For destruction processes, EO generated emissions between 65 and 1,896 kg CO_2 eq./g PFAS destroyed, whereas incineration of spent media or PFAS-containing residuals showed a much broader but generally lower range (0.02-142 kg CO_2 eq./g).

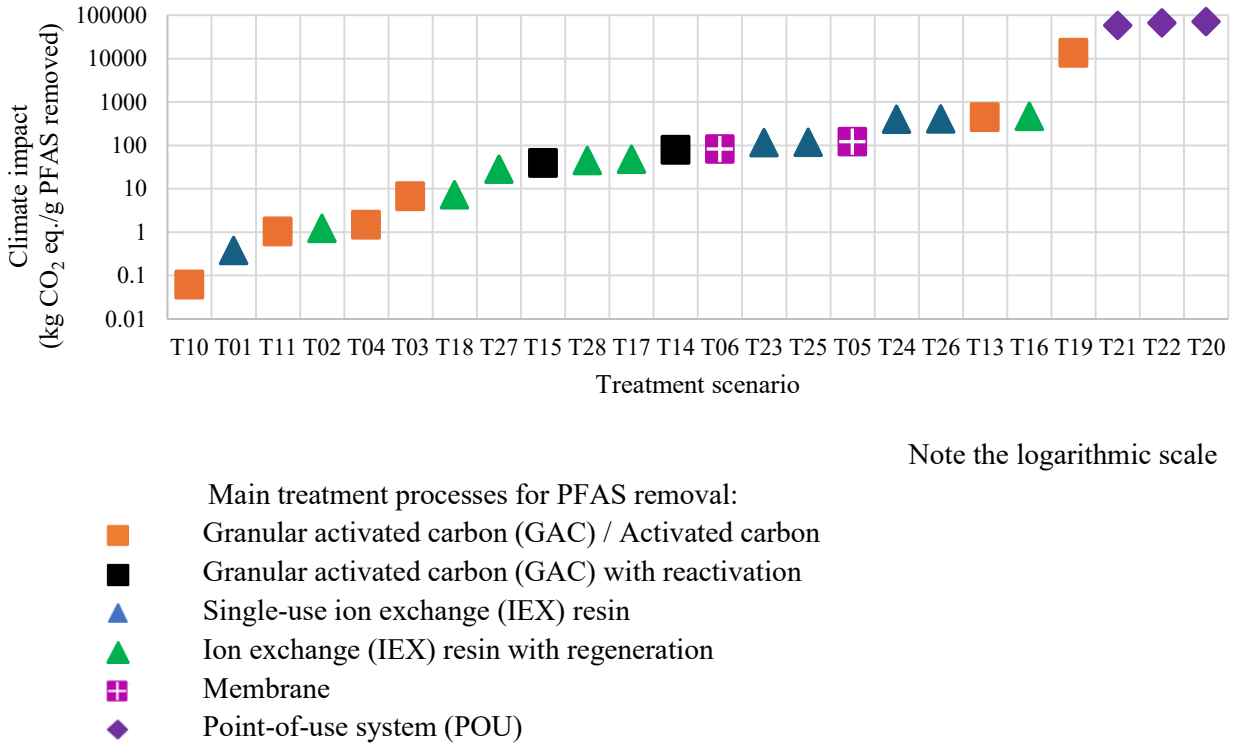


Figure 4 - LCA meta-analysis results: CO_2 eq. emissions per gram of PFAS removed

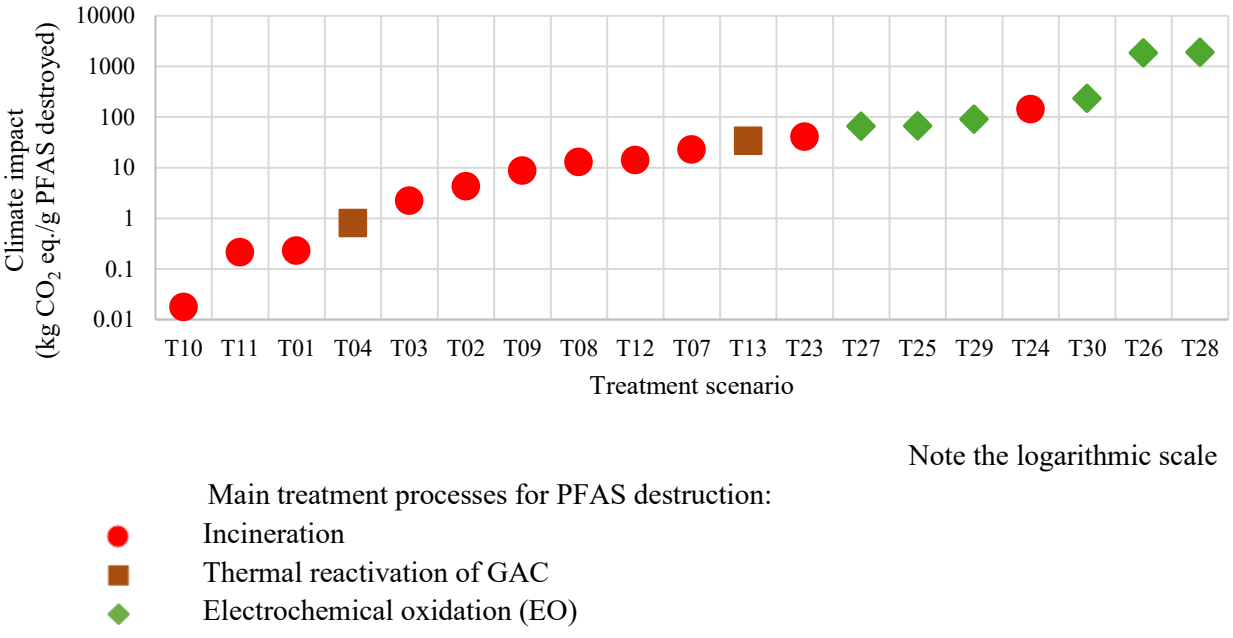
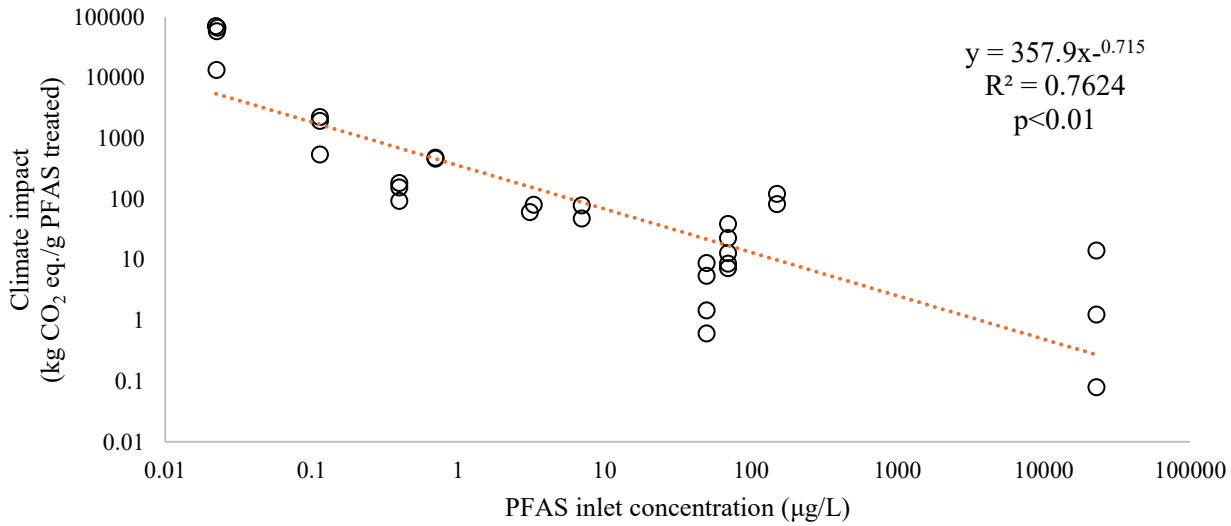


Figure 5 - LCA meta-analysis results: CO_2 eq. emissions per gram of PFAS destroyed

A key factor influencing emissions across all scenarios is the initial PFAS concentration in the water (Figure 6). Higher influent concentrations and greater concentration reductions are associated with lower emissions per gram of PFAS treated. This reflects an "entropic penalty" when treating dilute contaminants: lower concentrations require more energy and resources per unit of PFAS removed. In practical terms, higher PFAS concentrations enhance mass transfer and improve the efficiency of technologies such as GAC and IEX. However, pre-treatment remains important to remove competing substances (e.g., total organic carbon), which can reduce adsorption efficiency.

(A)



(B)

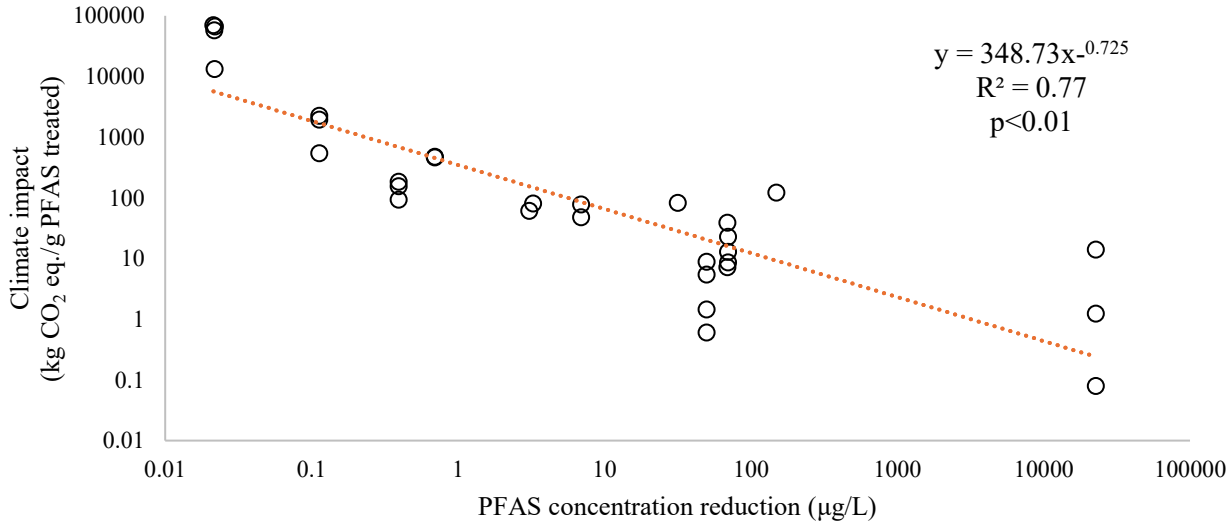


Figure 6 - Relationship between climate impact and (A) PFAS inlet concentration and (B) concentration of PFAS treated on a double logarithmic scale

5.2.2 Economic performance (CAPEX & OPEX)

Over the 15 to 30 years lifetime of innovative technologies for treating PFAS-contaminated water, operational costs typically exceed the initial capital investment. On average, operating expenses (OPEX) are about 20% higher than capital expenditures (CAPEX) in annual life cycle costing (LCC; Figure 7). Taken together, the median annual cost is \$1.52/m^3, with 68% of reported cases ranging between \$0.04 and \$1.77/m^3.

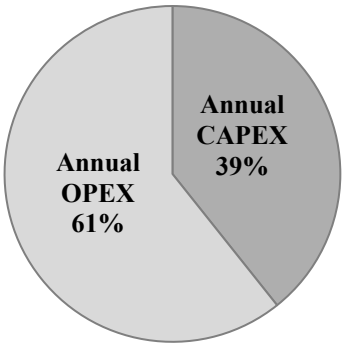
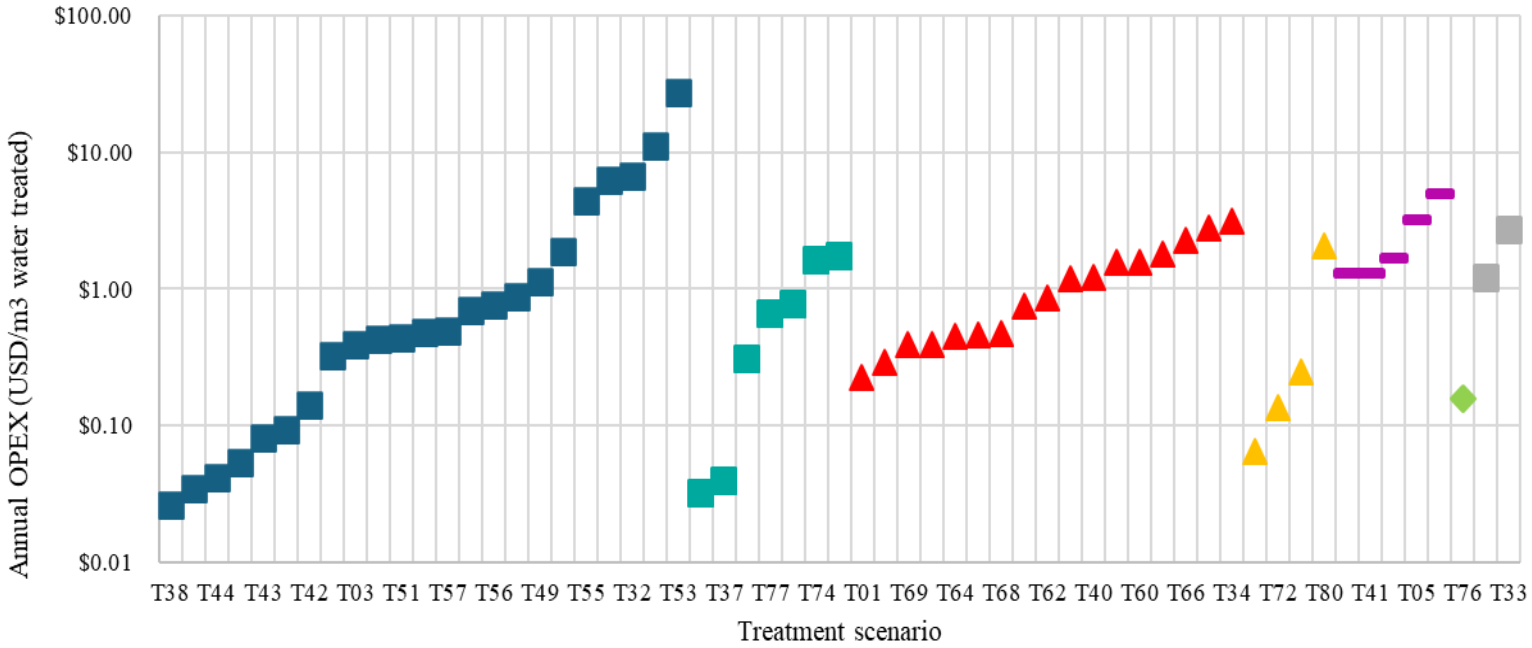


Figure 7 - Components of life cycle cost

Annual CAPEX varies by technology (Figure 8). IEX and GAC systems have similar capital costs, ranging from \$0.01 to \$0.45/m^3 of treated water. Membrane technologies, such as reverse osmosis and nanofiltration, are generally more expensive, with CAPEX between \$0.40 and \$0.51/m^3. Although there is some overlap, membranes tend to have higher average capital costs. These investments are typically evaluated over a 15-30 years payback period, assuming discount rates of 3-7%.

Operation and maintenance costs show considerable variability, ranging from \$0.03 to \$28/m^3. For example, a laboratory-scale GAC system treating groundwater with 1.92 µg/L PFOS and PFOA achieves 50% removal at very low cost (\$0.03/m^3), partly due to simplified assumptions. In contrast, another GAC system treats lower influent concentrations (0.57 µg/L PFAS-11) to a much stricter effluent target (0.004 µg/L), resulting in significantly higher costs due to expenses such as virgin GAC, regeneration, transport, and additional energy use. This highlights that even within the same technology, costs can differ substantially depending on treatment goals and system design. Overall, reported OPEX ranges are \$0.03-28/m^3 for GAC, \$0.06-3.20/m^3 for IEX, and \$1.30-4.90/m^3 for membrane technologies.



Note the logarithmic scale

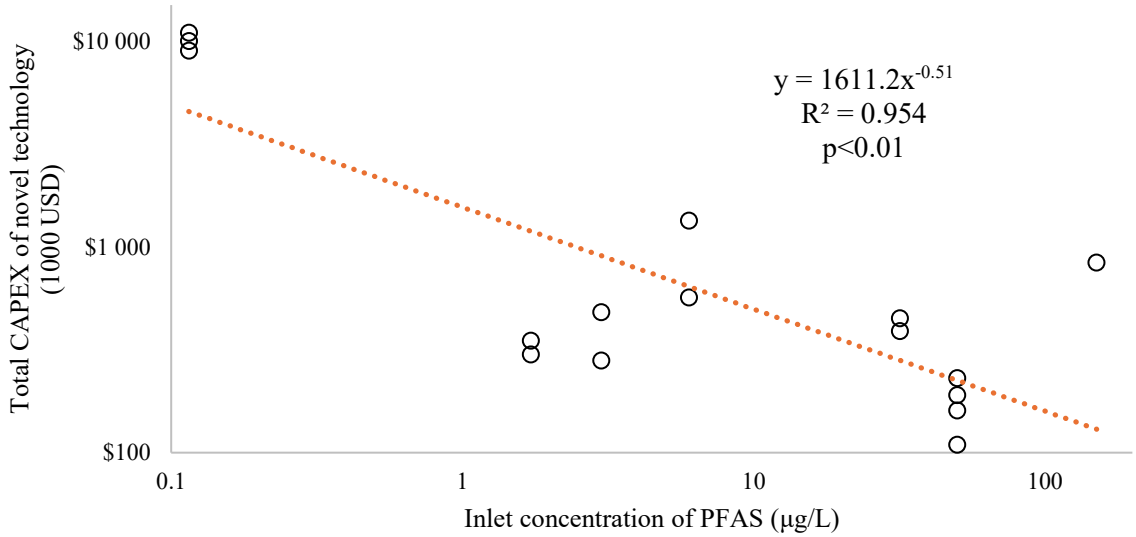
Main treatment processes:

- GAC / Activated carbon + Incineration / Reactivation
- GAC / PAC + Landfill disposal
- Ion exchange (IEX) + Incineration
- Ion exchange (IEX) + Landfill disposal
- Membrane filtration
- Foam-fractionation
- AquaPRS™ (carbon-based micro-adsorbent suspension + ceramic membrane filter)

Figure 8 - Annual OPEX of innovative treatment technologies

Capital and operational costs for PFAS treatment are driven by distinct sets of parameters. CAPEX is primarily influenced by influent PFAS concentration, required removal efficiency, and treatment scale. Regression analysis indicates a strong negative correlation between total CAPEX and both inlet concentration and concentration reduction, meaning that more dilute influent streams and stricter treatment targets require higher capital investment (Figure 9). From the analysis, CAPEX also shows a moderate correlation with the volume of water treated, with a less than linear exponent (~0.63) on the volume term, suggesting the existence of economies of scale in the treatment processes. This makes sense given that the underlying drivers of CAPEX would typically include, for example, the construction of large vessels or basins for contact between contaminated water and GAC or IEX resins.

(A)



(B)

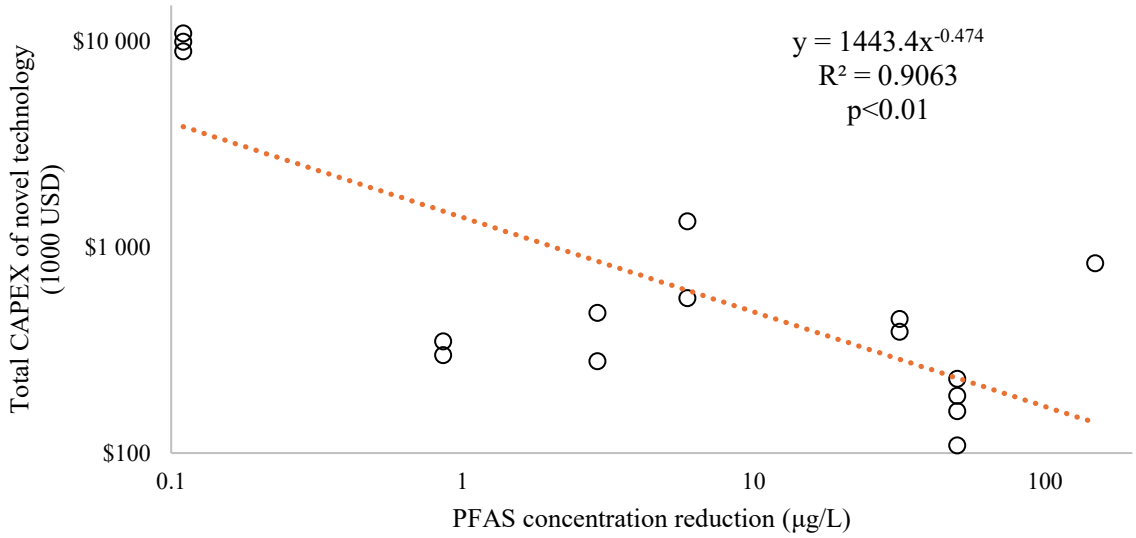


Figure 9 - Relationship between CAPEX and (A) PFAS inlet concentration and (B) concentration of PFAS treated on a double logarithmic scale

In contrast, OPEX is not significantly correlated with influent concentration, removal efficiency, or treatment volume. Instead, OPEX is governed by process-specific and site-dependent factors. Key contributors include adsorbent or resin costs (e.g., $1.25-3.20/kg for GAC and $8.43-17.60/kg for IEX), as well as media replacement frequency, which is determined by breakthrough capacity and treatment targets. Stricter effluent requirements increase media consumption and regeneration frequency, with GAC typically requiring more frequent replacement than IEX. Energy demand is another major driver, particularly for pumping and pressure-intensive processes such as membrane filtration. Additionally, local conditions such as water chemistry, electricity prices, and plant configuration, introduce variability in operating costs. Residuals management further contributes to OPEX uncertainty, with disposal costs ranging from approximately $90/tonne for incineration of non-hazardous waste to $870/tonne for hazardous waste.

5.2.2.1 CAPEX of representative PFAS treatment scenarios in the UK

As mentioned earlier in the introduction of this thesis, different countries have different regulations regarding guidance values for PFAS in drinking water. And as more is understood regarding the toxicological risks associated with PFAS exposure, the efforts to mitigate their presence in drinking water have grown. In this context, the UK Environment Agency has recently published results from a project where thresholds for managing PFAS in the aquatic environment were developed. The report establishes a health-based biota threshold of 77 ng/kg in fish for the combined concentration of PFAS-4, derived from the European Food Safety Authority's tolerable weekly intake. This reflects a protection objective based on cumulative exposure rather than individual substances. Corresponding equivalent freshwater concentrations of 0.015 ng/L for PFOS, 0.4 ng/L for PFOA, 0.2 ng/L for PFHxS, and 0.3 ng/L for PFNA were derived using bioaccumulation factors (BAFs) and are applicable for scenarios where a single PFAS is present (Environment Agency, 2026).

In cases where multiple PFAS occur simultaneously, the report recommends the use of a hazard index (HI) approach to account for additive effects. This involves calculating hazard quotients for each compound by dividing measured concentrations by their respective thresholds and summing these values; an HI greater than one indicates a potential exceedance of the threshold and associated risk. According to the Environment Agency (2026), this methodology ensures that mixture toxicity is appropriately considered, avoiding the underestimation of risk that would result from evaluating substances independently.

Although these PFAS-4 thresholds are not currently enforceable environmental quality standards (EQS), they may inform future regulatory frameworks. Their potential implementation raises important questions regarding the economic implications of achieving such stringent limits in comparison with the current guidance imposed by the Drinking Water Inspectorate (DWI) of 100 ng/L for the sum of 48 named PFAS (DWI, 2025). For instance, reported concentrations of PFOS in UK freshwater bodies range between 11 and 19 ng/L (Junqué et al., 2026; Kurwadkar et al., 2022). Applying the cost relationships derived in the economic meta-analysis presented earlier (Figure 9), and assuming an average influent concentration of 15 ng/L for PFOS, achieving a target concentration of 0.015 ng/L would require an estimated CAPEX of approximately USD 12.1 million for treatment infrastructure of one DWTP with an average treatment capacity of 20 L/s.

A comparable but more extreme scenario is illustrated by the findings of Megson et al. (2024), who analyzed water samples impacted by industrial discharges from a fluorochemical plant in Thornton-

Cleveleys (Northwest UK). In this case, 43 PFAS were detected, with an average total concentration of approximately 38,000 ng/L. Considering treatment to comply with the current DWI guidance value of 100 ng/L for the sum of PFAS, and applying the same cost relationships derived in the economic meta-analysis, the required CAPEX would be approximately two orders of magnitude lower than that estimated for achieving the much more stringent Environment Agency threshold. This contrast showcases the strong dependence of treatment costs on both target concentration levels and regulatory stringency, as well as the significant economic implications of adopting more precautionary PFAS standards.

5.3 Limitations of existing LCAs

5.3.1 Incomplete representation of PFAS fate, exposure, and life cycle impacts

Existing LCAs of PFAS treatment technologies often fail to adequately capture the fate of PFAS and the resulting exposure pathways, leading to incomplete life cycle coverage and underestimation of toxicity impacts. Most treatment technologies, such as GAC and IEX resin, function as separation processes that transfer PFAS from water to secondary waste streams (e.g., spent media or brines) rather than destroying them. However, many studies assume complete PFAS destruction during incineration or regeneration, despite significant uncertainties regarding destruction efficiency and the potential formation of harmful byproducts. In some cases, disposal pathways such as landfilling are included without considering long-term emissions. Additionally, residual PFAS in treated water or reject streams are frequently neglected or assumed to be harmless once regulatory thresholds for treated water are met. These simplifications result in an incomplete accounting of PFAS flows, overlook downstream environmental releases and exposure pathways, and ultimately limit the ability of LCAs to accurately represent real-world conditions and associated human and ecological health impacts.

5.3.2 Lack of characterization factors (CFs)

Accounting for ecological and human health impacts of PFAS in LCA is limited by the lack of appropriate characterization factors (CFs), particularly for key compounds such as PFOS and PFOA in commonly used methods like USEtox. This gap leads to incomplete representation and potential underestimation of toxicity-related impacts. To address this limitation, studies have adopted several workaround approaches. These include the use of proxy substances (e.g., PCBs) to approximate PFAS impacts (Feng et al., 2021), application of PFAS-specific near-field risk assessment methods to evaluate direct human health effects (Bixler et al., 2021), and, in some cases, exclusion of toxicity impacts under simplifying assumptions (e.g., compliance with regulatory thresholds) (Emery et al., 2019).

While these strategies enable approximate evaluations, they introduce uncertainty and reduce comparability across studies. Recent studies have produced PFAS-specific CFs for human toxicity and ecotoxicity (Aggarwal et al., 2024; Aggarwal & Peters, 2024; Holmquist et al., 2020), which offer improved opportunities for more accurate and consistent LCA assessments in future research.

Synthesizing these methodological observations, it becomes evident that practitioners facing data gaps in toxicity assessment during LCA use several options to address or bypass the lack of CFs. First, in the context of consequential LCA, one may assume that all compared options influence toxicity to the

same extent, allowing it to be excluded from comparative results—though this assumption should be justified and used cautiously. Second, practitioners may turn to near-field risk assessment approaches, which estimate exposure and risk directly, outside the LCA framework, offering a more localized and context-specific evaluation of toxic effects. Third, "read-across" methods can be used to approximate missing CFs by borrowing data from structurally or functionally similar substances, leveraging existing knowledge to fill in gaps. Finally, when no adequate proxies exist, new CFs can be generated using fate, exposure, and effect modeling tools. Although this is data- and resource-intensive, it can significantly enhance the robustness of the toxicity-related impact assessment. These strategies represent a spectrum from simplified assumptions to detailed modeling, and each of the first three were applied in the LCA studies reviewed.

5.4 Environmental and human health impacts of PFAS treatment in Sweden

5.4.1 Technology comparison and main contributors to impacts

Figure 10 compares the life cycle environmental impacts of five TTs for PFAS-contaminated groundwater. For each impact category, results are expressed as scaled characterization values relative to the system with the highest impact (100%), allowing comparison across systems despite differing impact units (e.g., kg CO_2-eq for climate change and CTUe for ecotoxicity).

Across the five systems, TT 2 exhibits the highest environmental impacts in most categories, followed by TT 1 and TT 5, while TTs 3 and 4 show comparatively lower impacts. The different treatment steps influence overall results (Figure 11). In TT 1, the pre-treatment accounts for 50% of climate change and 80% freshwater eutrophication, and requirements associated with water softening done by fluidized bed pellet reactors and the off-site treatment of wastewater and sludge generated after sedimentation of rapid sand filter backwash water are the main contributors of impacts, showing that downstream processes can have high influence on certain impact categories.

In TT 1 softening process, calcium oxide is used from which approximately 3,340 tonnes of calcium-sand pellets are generated yearly, needing truck transportation either for being reused, as for example a pH-adjustment product for lakes (75%; accounted as avoided calcium carbonate production), or for being disposed in a landfill (25%). For ecotoxicity, and human toxicity cancer and non-cancer in TT 1, this generation of calcium-sand pellets and its reuse as a pH-adjustment product produces environmental credits that outweigh the impacts associated with subsequent PFAS removal and destruction, resulting in net environmental benefits for these categories.

In contrast, photochemical ozone formation and terrestrial acidification are mainly driven by the PFAS removal step, particularly the annual transport of approximately 960 m^3 of GAC to and from an off-site thermal reactivation facility in Germany, as well as by the production of virgin GAC required to compensate for the 10% loss of media during reactivation.

In TTs 2 to 5, the PFAS removal step is the dominant contributor to environmental impacts across the majority of assessed categories, followed by post-treatment, primarily due to sodium hypochlorite production and use in disinfection (Figure 12). This indicates that the environmental footprint of PFAS treatment systems can be largely driven by the PFAS removal process, highlighting the importance of optimizing removal technologies to reduce overall environmental burdens.

When evaluating the PFAS removal step in isolation (Figure 13), CCMF emerges as the most detrimental PFAS removal technology across all categories, primarily due to hydrochloric acid production for consumption to reduce feed water pH and mitigate membrane scaling. However, this chemical requirement is site-specific and, in this case, reflects the relatively high hardness of Uppsala's groundwater (16–19 °dH), which increases the need for scaling control when CMMF is run with 89% recovery. This suggests that alternative antiscalant methods or more efficient chemical dosing should be considered.

At an 80% recovery rate, CCMF operation does not require hydrochloric acid, as antiscalant alone prevents membrane fouling. This reduces the environmental impacts of PFAS removal by 29% to 76% across most categories compared to the baseline, although climate change impacts increase by about 8% due to higher energy use from lower recovery. Despite these improvements, CCMF remains the most environmentally burdensome PFAS removal technology.

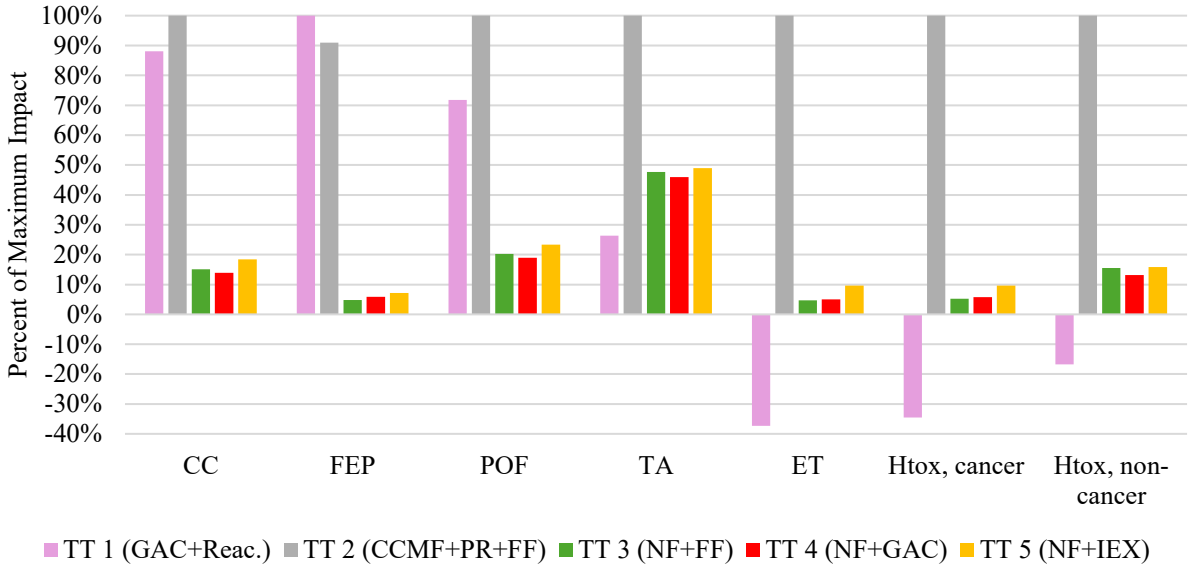


Figure 10 - Total life cycle environmental impacts for TTs 1 to 5. Impacts are normalized to values estimated for the most environmentally impactful treatment system in each impact category. CC - Climate change; FEP - Freshwater eutrophication; POF - Photochemical ozone formation, human health; TA - Terrestrial acidification; ET - Ecotoxicity; Htox, cancer - Human toxicity, cancer; Htox, non-cancer - Human toxicity, non-cancer.

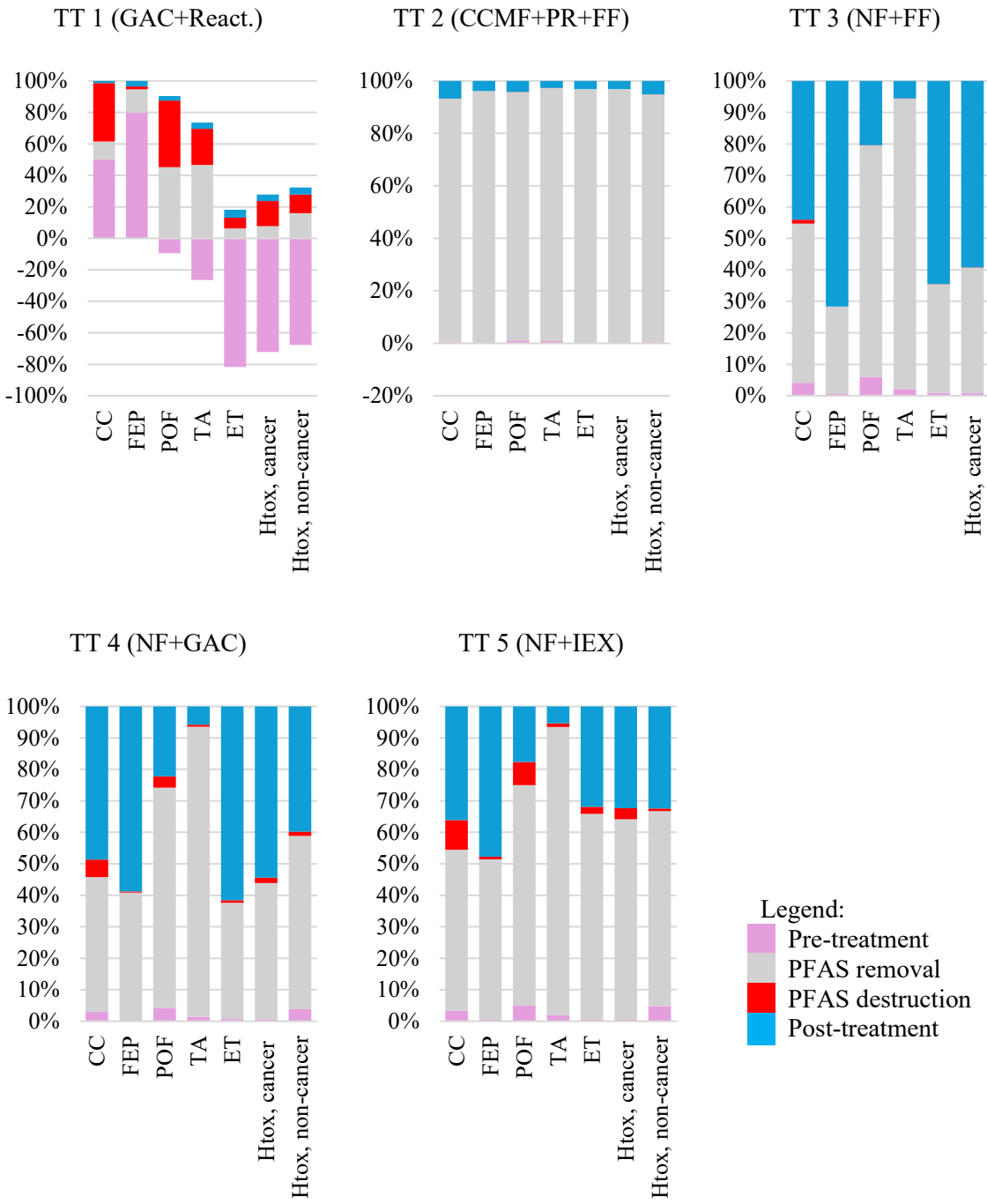


Figure 11 - Relative contributions of pre-treatment, PFAS-4 removal, and PFAS destruction to environmental and human health impacts. CC - Climate change; FEP - Freshwater eutrophication; POF - Photochemical ozone formation, human health; TA - Terrestrial acidification; ET - Ecotoxicity; Htox, cancer - Human toxicity, cancer; Htox, non-cancer - Human toxicity, non-cancer.

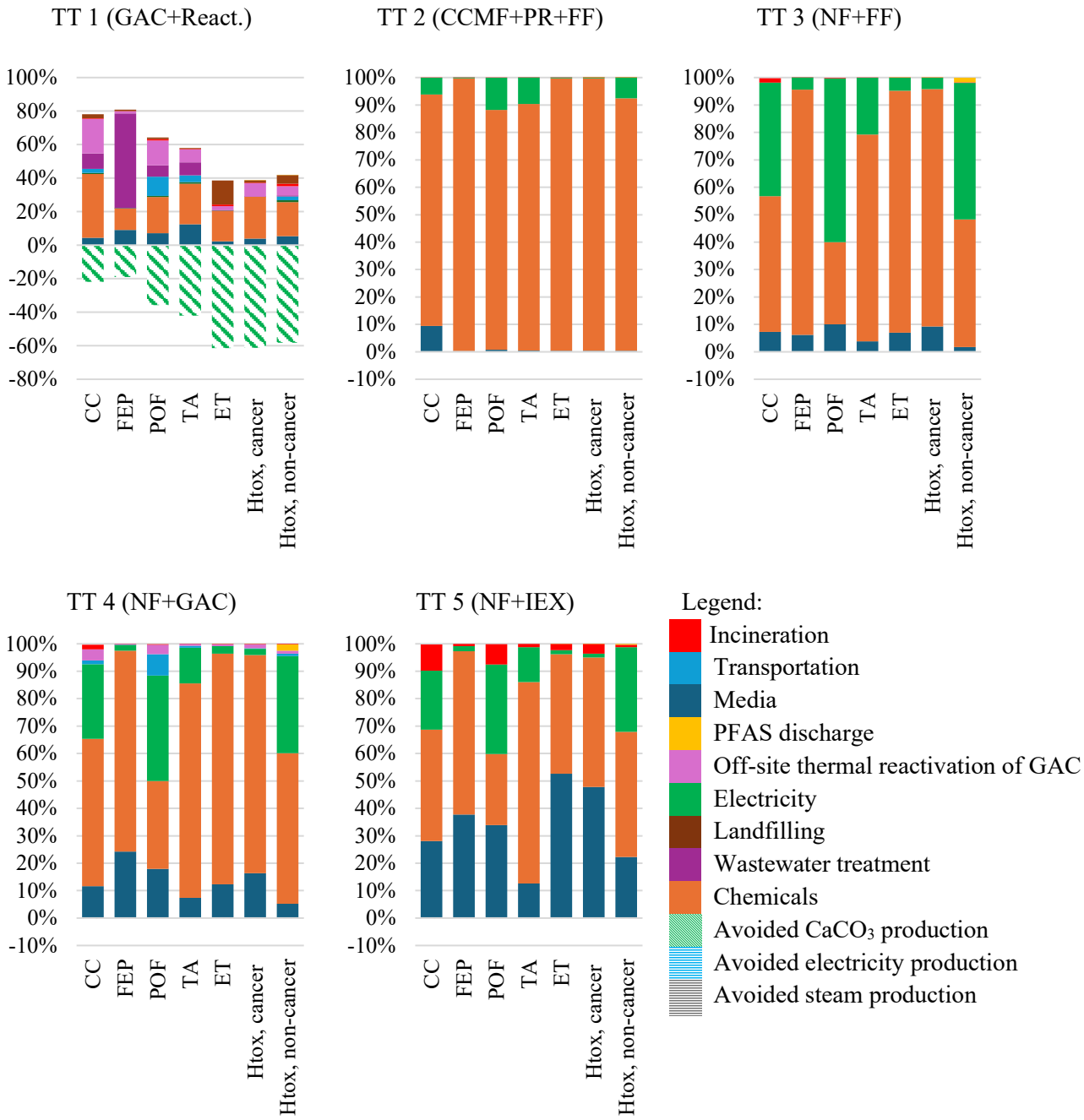


Figure 12 - Percent contributions of unit processes to the environmental and human health impacts. CC - Climate change; FEP - Freshwater eutrophication; POF - Photochemical ozone formation, human health; TA - Terrestrial acidification; ET - Ecotoxicity; Htox, cancer - Human toxicity, cancer; Htox, non-cancer - Human toxicity, non-cancer.

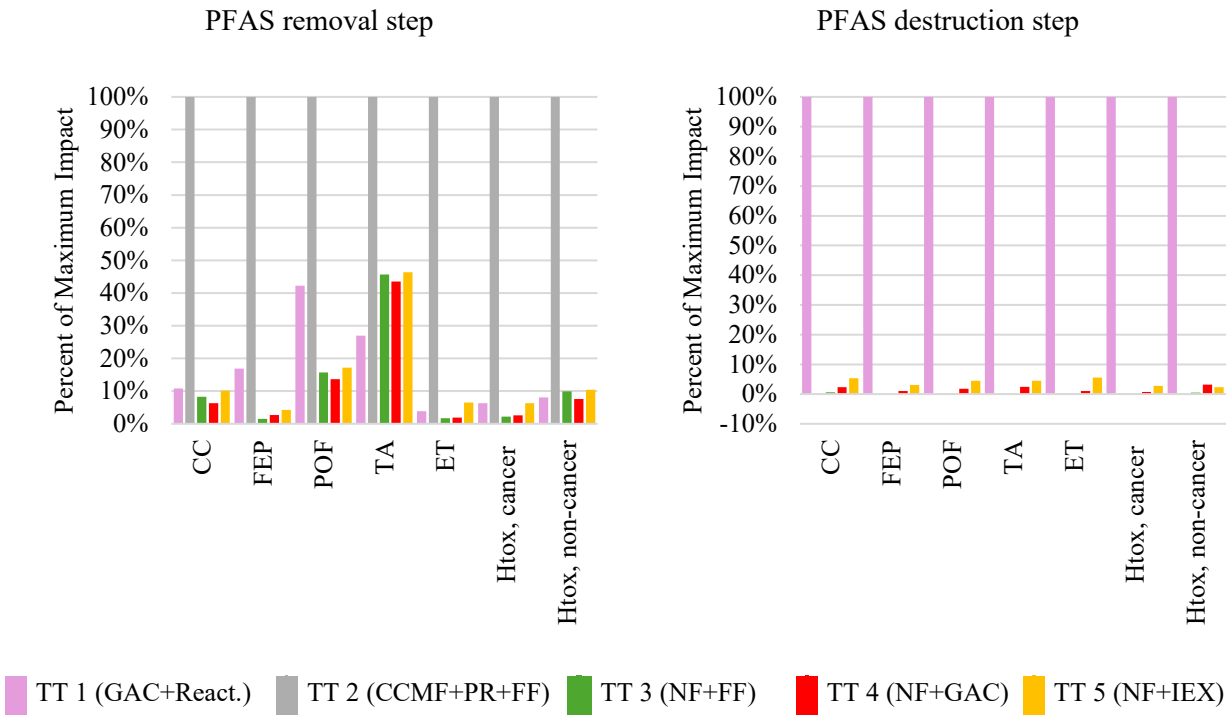


Figure 13 - Life cycle environmental impact comparison for PFAS-4 removal and PFAS destruction steps. Impacts are normalized to values estimated for the most damaging treatment system in each impact category. CC - Climate change; FEP - Freshwater eutrophication; POF - Photochemical ozone formation, human health; TA - Terrestrial acidification; ET - Ecotoxicity; Htox, cancer - Human toxicity, cancer; Htox, non-cancer - Human toxicity, non-cancer.

Overall, NF+IEX (TT 5) ranks as the second most impact-intensive PFAS removal technology, largely driven by the production of anionic resin used to replace single-use IEX media after PFAS-4 breakthrough, and the use of antiscalant in NF. Its environmental burden could be reduced through resin regeneration; however, the benefits depend on the type and amount of chemicals used. In some cases, the chemical demand of regeneration may offset or even exceed its advantages, making its overall performance highly context-dependent.

In TTs 3 (NF+FF) and 4 (NF+GAC), the NF process dominates the removal step, driven mainly by antiscalant production used to control fouling, electricity production to run the filtration process and membrane production for replacement. The post-treatment stage further adds to the overall impacts in TTs 3 to 5. The main drivers are the production of calcium oxide for remineralization and increasing the water pH after NF, and sodium hypochlorite used for final disinfection.

In TT 1, impacts from GAC filtration are mainly driven by the production of replacement GAC (due to 10% loss during reactivation) and the transport of spent GAC to Germany, with high water content increasing transport burdens. Varying GAC loss (5–15%) shows a moderate effect on impacts (±1.2–13%), with freshwater eutrophication and human toxicity most affected. Overall, reducing GAC loss improves environmental performance, while higher losses increase impacts.

When the PFAS destruction step is analyzed separately (Figure 13), off-site thermal reactivation of GAC in TT 1 stands out as the most impact-intensive option, mainly driven by the upstream provision of natural gas required for the reactivation process. If reactivation were carried in Sweden, it would

remain the most environmentally burdensome destruction option, largely due to the demand for natural gas and electricity, although transportation-related impacts would be substantially reduced (e.g., approximately 50% lower CO_2-equivalent emissions).

Across all TTs, environmental impacts are largely driven by a few key processes (mainly chemical and material production, transport, and electricity), while other contributions are minor. In Sweden, the low-carbon electricity mix limits climate impacts, but upstream processes (e.g., material production, uranium mining and enrichment, and small shares of imported fossil electricity) still contribute. In contrast, using a fossil-based energy mix such the one from Poland would increase climate impacts by up to 8.8 times, highlighting the energy-intensive nature of PFAS treatment.

The small amounts of PFAS-4 remaining in the treated effluents discharged to the environment (e.g., 30–177 ng/FU; treated process water generated after FF, IEX, GAC, or treatment in a wastewater treatment plant depending on the TT) and those remaining in drinking water (<4,000 ng/FU) contribute negligibly (<0.2%) to ecotoxicity and human toxicity across all TTs. However, for PFAS-4, human health toxicity impacts may be underestimated, as CFs are currently available only for PFOA. Furthermore, the focus on PFAS-4 may not capture the full extent of the risk, as other PFAS with potentially significant health effects are often present alongside them and warrant further research. To assess the influence of this limitation, ecotoxicity impacts from remaining PFAS-4 in drinking water were calculated for TTs 1–5 using both a PFOA-equivalent approach and compound-specific CFs for all four PFAS. The results show that using PFOA equivalents yields impacts that are one order of magnitude lower, which can also be the case for human toxicity. Nevertheless, given the overall minor contribution of PFAS-4 to ecotoxicity, applying specific CFs for PFOS, PFHxS, and PFNA instead of accounting for them as PFOA equivalents is unlikely to substantially affect the total human health toxicity results.

5.4.2 Results comparison from case study with findings from meta-analysis

By integrating the PFAS removal and destruction results from the case study into the previous meta-analysis, a direct comparison of findings can be made. The CO_2-equivalent emissions of TTs 3, 4, and 5 (considering only PFAS removal and destruction steps; 332–559 kg CO_2 eq./g PFAS treated) fall within the range identified as typical for innovative treatment systems, where emissions span from 10 to 1,000 kg CO_2 eq./g PFAS treated (see Section 5.2.1). In contrast, TTs 1 and 2 present values of 2,203 and 4,651 kg CO_2 eq./g PFAS treated, respectively, which are considerably higher but withing the range of results for centralized treatment systems in the published literature.

Furthermore, the case study results are consistent with previous findings indicating that PFAS removal generally imposes a substantially higher climate burden than PFAS destruction. The median climate impact for PFAS removal across TTs 1–5 is 472 kg CO_2 eq./g PFAS removed (Figure 14), whereas the corresponding median for PFAS destruction is 38 kg CO_2 eq./g PFAS destroyed (Figure 15).

However, TT 1 represents an exception to this trend. In this case, the PFAS destruction step exhibits a higher climate burden (1 684 kg CO_2 eq./g) than the preceding removal stage (519 kg CO_2 eq./g). This indicates that the treatment of concentrated residuals during GAC reactivation is more energy- and resource-intensive than the initial PFAS removal process carried out using GAC.

The LCA findings point to a burden-shifting effect: while PFAS removal reduces local exposure risks, it increases upstream environmental and health impacts due to higher energy and resource use. This

raises questions about the balance between benefits and broader environmental costs. Further NHHB analysis is therefore proposed and used to assess whether the avoided PFAS-4–related human health impacts from water treatment outweigh the life cycle impacts induced by DWTP.

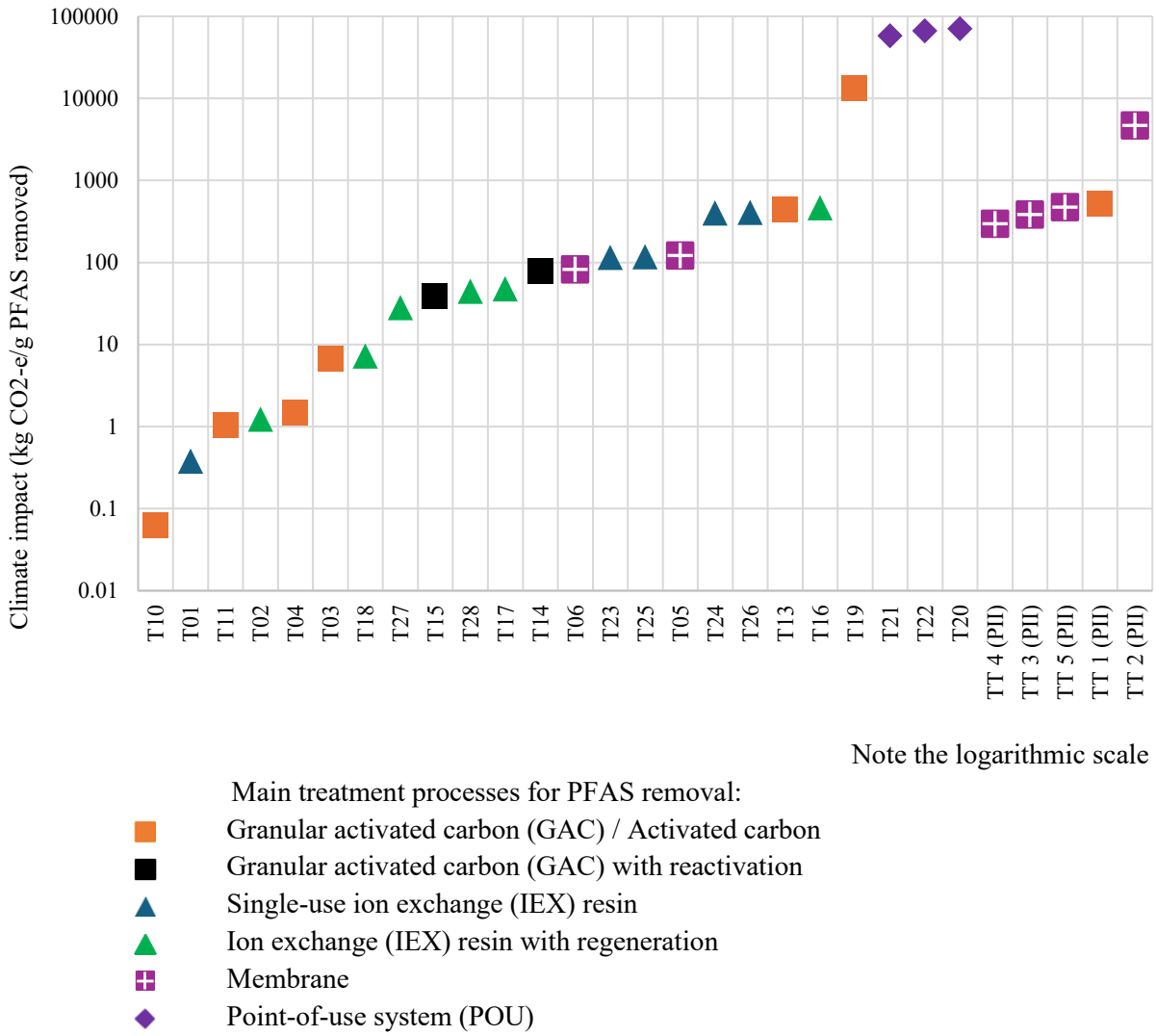


Figure 14 - LCA meta-analysis results: CO_2 eq. emissions per gram of PFAS removed

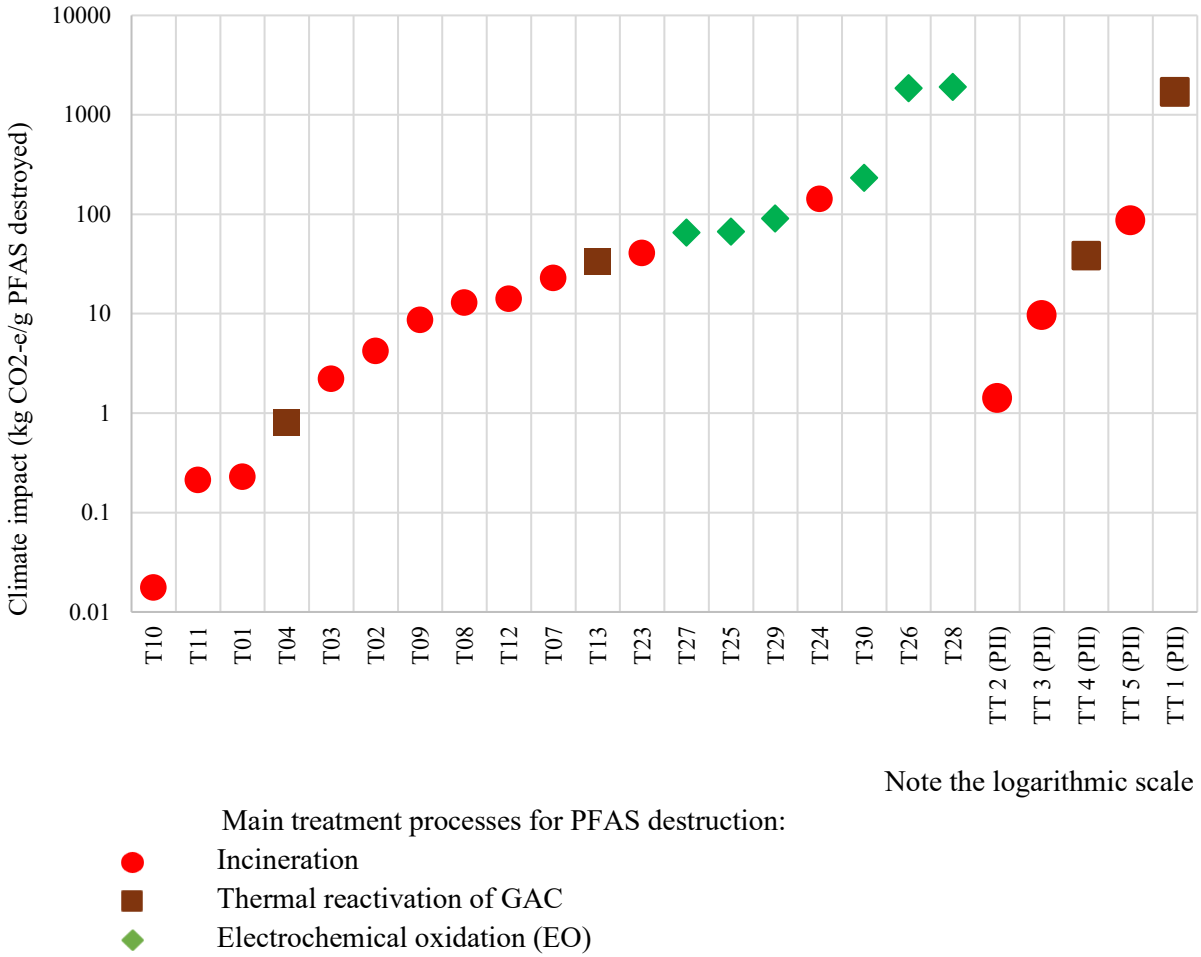


Figure 15 - LCA meta-analysis results: CO_2 eq. emissions per gram of PFAS destroyed

5.4.3 Net human health benefit (NHHB) analysis

The NHHB analysis determine whether the avoided PFAS-4-related human health impacts achieved through water treatment outweigh the life cycle impacts induced by DWTP operations. A positive NHHB value indicates that the TT provides an overall benefit in terms of human health impacts. In contrast, a negative NHHB value means that the life cycle impacts of operating the DWTP outweigh the benefits of removing PFAS-4, suggesting that the treatment process may result in a net human health burden. Table 2 present the NHHB results for each TT and CF applied, while Figure 16 presents the impact categories contributions to NHHB.

Overall, the potential benefits of removing PFAS-4 from drinking water are not overcome by the potential impacts generated by the DWTP operation life cycle, at least when LCA norms are applied, that is, when CFs derived from EF_{rodent} are applied to cancer and non-cancer related human toxicity. However, when non-cancer human health CF derived from epidemiological data (EF_{human}) is used, the benefit of treatment increases by factors of approximately 3 to 4 orders of magnitude, as epidemiology-based EF implies a much higher toxicity per unit intake compared to the rodent-based EF. Consequently, for the same emission or intake level, the estimated human health impacts of drinking contaminated water are much higher when epidemiological data are used. This showcases the potentially significant contribution of PFAS ingestion for non-cancer health related problems.

These results demonstrate that human toxicity outcomes are highly sensitive to the choice of EFs, reflecting a fundamental methodological trade-off. Using rodent-based EFs ensures consistency across emissions due to their broader availability, but may underestimate the human health relevance of PFAS exposure, particularly for non-cancer effects, where epidemiological data indicate much higher toxicity. In contrast, incorporating epidemiological EFs better represents the consequences of real-world human exposures but can overstate the relative benefits of PFAS removal, since comparable epidemiological data are lacking for many other emissions in the treatment life cycle. As a result, NHHB outcomes shift markedly depending on the EF approach, and should be interpreted as scenarios reflecting different assumptions about data completeness and representativeness, rather than definitive estimates. Therefore, it is advisable that LCA practitioners, and especially LCIA experts, carefully consider the differences between EFs when selecting approaches for CF calculation, particularly in the context of assessing human health impacts from direct exposure to contaminants.

The analysis focuses exclusively on PFAS-4, as these are the primary contaminants in the groundwater, and therefore does not account for potential co-benefits from removing other contaminants and substances (e.g., other PFAS, and pathogens such as giardia and cryptosporidium). Additionally, it is assumed that incineration or reactivation processes fully destroy PFAS under applied conditions. This assumption may underestimate toxicity impacts, as the complete destruction of PFAS is still uncertain. Incomplete treatment could lead to the formation and release of smaller, potentially harmful PFAS compounds, increasing overall toxicity.

These results indicate that as long as PFAS and PFAS-containing products continue to be produced and used, environmental releases will persist, adding to their known health and ecological impacts. These ongoing emissions require further water treatment, creating additional environmental burdens. Meanwhile, stricter regulations driven by growing scientific evidence demand more intensive, resource-heavy treatment methods. Therefore, this study suggests that preventive policies, such as restricting or banning PFAS, are likely more effective and sustainable than relying on downstream treatment alone.

Table 2 - Net human health benefit (NHHB) results

NHHB results									
TT 1 EF_{rodent}	TT 1 EF_{human}	TT 2 EF_{rodent}	TT 2 EF_{human}	TT 3 EF_{rodent}	TT 3 EF_{human}	TT 4 EF_{rodent}	TT 4 EF_{human}	TT 5 EF_{rodent}	TT 5 EF_{human}
-5.17×10^{-7}	7.72×10^{-5}	-3.21×10^{-6}	1.11×10^{-4}	-6.26×10^{-7}	1.10×10^{-4}	-6.35×10^{-7}	1.18×10^{-4}	-1.03×10^{-7}	1.18×10^{-4}

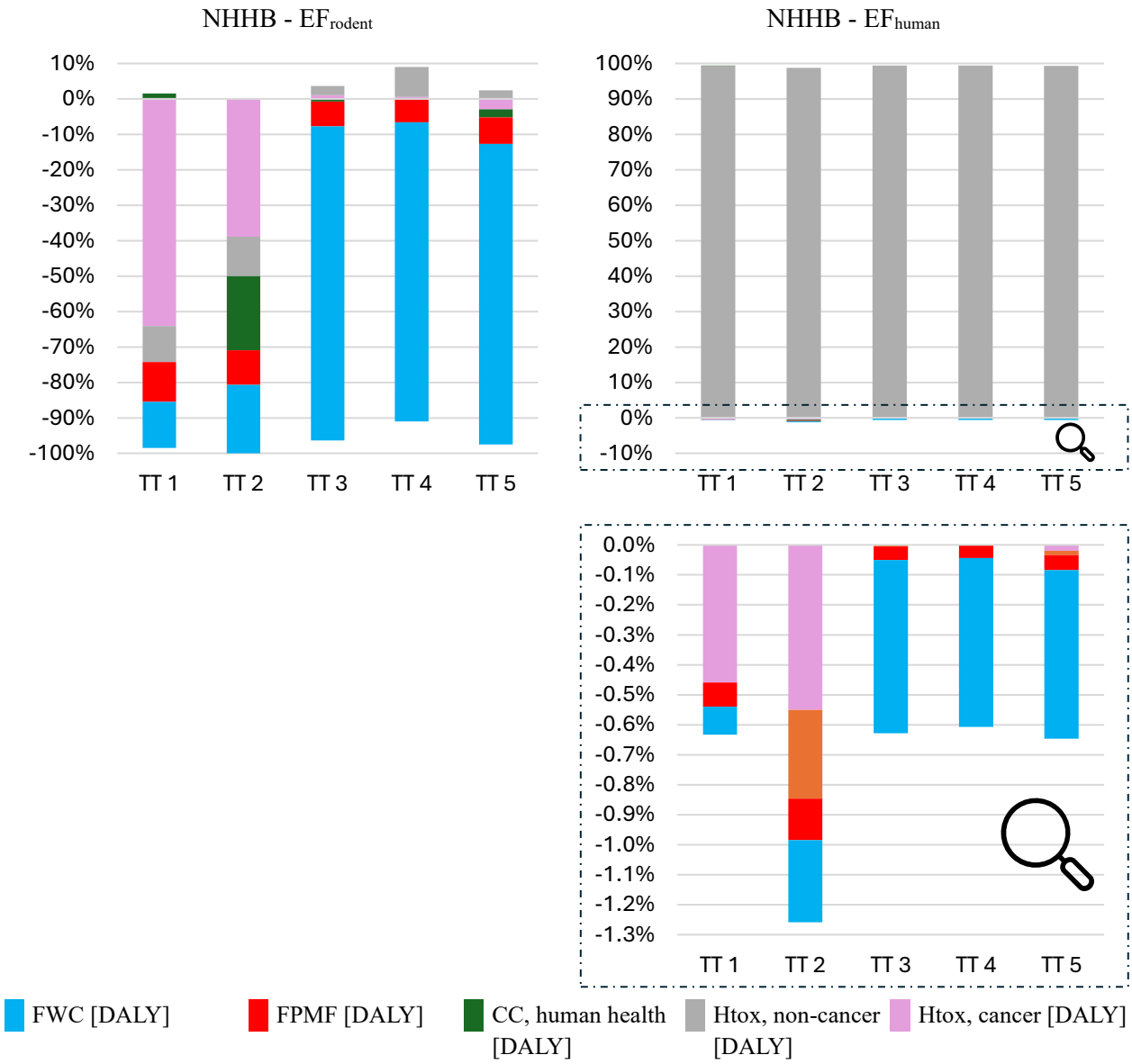


Figure 16 - Percent contributions of impact categories to total NHHB analysis. Htox, cancer - Human toxicity, cancer; Htox, non-cancer - Human toxicity, non-cancer; CC - Climate change, human health; FPMP - Fine particulate matter formation, human health; FWC - Freshwater consumption, human health. The dashed rectangles and magnifying glasses show a detailed view of the negative contributions of impact categories for the NHHB analysis when EF_{human} is applied.

5.4.3.1 Effectiveness of NHHB approach

The application of the NHHB approach demonstrated that it can be used to address the research question of whether removing PFAS from drinking water yields a net human health benefit from a life cycle perspective. However, the results were not definitive and instead highlighted important methodological trade-offs. In particular, the human toxicity outcomes proved highly sensitive to the choice of CFs. Using rodent-based CFs offers consistency across emissions due to their wider availability, but may underestimate the human health relevance of PFAS exposure, especially for non-cancer effects. Conversely, incorporating epidemiological CFs better reflects real-world human

impacts, yet may overstate the relative benefits of PFAS removal because similar data are lacking for many other emissions in the system.

As a result, the NHHB outcomes varied significantly depending on the CF applied and are better interpreted as scenario-based insights rather than precise estimates. Despite this variability, the approach proves to be a useful and practical tool for assessing whether drinking water treatment technologies for PFAS removal deliver net human health benefits from a life cycle perspective. In addition, the rodent-based and epidemiology-based PFOA CFs developed within the NHHB framework can also be applied in case studies beyond Sweden, as they capture human health impacts from direct oral exposure while avoiding double-counting with impacts from freshwater emissions.

From a practical standpoint, the method was straightforward to apply and calculations were manageable. Nevertheless, because NHHB relies on outputs from a prior LCA, specifically for calculating PINO and PILCTW, it cannot be used as a standalone tool. This dependency makes the approach more time-consuming, particularly when multiple treatment technologies and sensitivity analyses are evaluated.

6. Discussion

This section discusses the answers to the research questions and outlines directions for future research.

6.1 Research questions revisited

RQ1: What do published studies on life cycle assessment and cost analyses say about the environmental and financial performance of emerging PFAS treatment technologies?

To answer this question, a scoping review and a meta-analysis of published life cycle assessment and cost analysis studies on PFAS treatment technologies were performed. Published studies show that emerging PFAS treatment technologies can reduce contamination, but they often come with substantial environmental burdens and variable financial costs. Life cycle assessment results from 30 treatment scenarios indicate a median climate impact of 88 kg CO_2-eq. per gram of PFAS treated. Removal processes generally account for the larger share of this burden, with a median of 80 kg CO_2-eq./g, compared with 23 kg CO_2-eq./g for destruction. This suggests that PFAS removal is usually more energy- and resource-intensive than destruction, largely because separation processes must handle large volumes of dilute water, whereas destruction is often applied to smaller, more concentrated residual streams. Among the technologies studied, GAC and ion exchange were the most frequently assessed, reflecting their current prominence in PFAS treatment practice. Their environmental performance, however, varies depending on influent concentration, removal targets, and system design. In general, dilute influent streams and stricter treatment requirements lead to much higher impacts per gram of PFAS treated.

The cost literature points to similarly wide variation. Across reported cost studies, the median annual cost was \$1.52/m^3 of treated water, with most cases falling between \$0.04 and \$1.77/m^3. Operational costs tend to exceed capital costs over the lifetime of the systems, by about 20% on average, showing that long-term operation and maintenance are a central financial concern. GAC and ion exchange generally have lower capital costs than membrane systems, while membrane technologies tend to be more expensive both in installation and operation. Operating costs are especially sensitive to process-specific factors such as media costs, replacement frequency, energy use, and residuals management, rather than to simple design parameters alone.

Overall, the published studies suggest that no single emerging PFAS treatment technology performs best across the environmental and financial criteria. Outcomes depend strongly on PFAS concentration, treatment scale, effluent targets, and local operating conditions. Therefore, the findings point out the importance of evaluating PFAS treatment technologies from both life cycle and cost perspectives, since technologies that appear effective in removal may still carry high climate impacts or long-term operating costs.

RQ2: To what extent do published studies fall short in representing the life cycle environmental impacts of innovative PFAS-contaminated water treatment?

The answer to this research question is based on the reviewed published LCA studies selected for the scoping review. Published studies on PFAS treatment often fall short in representing the full life cycle due to incomplete accounting of PFAS fate and exposure pathways. Many technologies, such as granular activated carbon and ion exchange resin, only transfer PFAS to secondary waste streams rather than destroy them. However, LCAs frequently assume complete destruction during subsequent

treatment (e.g., incineration and GAC reactivation) and neglect residual PFAS emissions from treated water or disposal pathways like landfilling. These simplifications overlook downstream releases and exposure routes, leading to an underestimation of environmental and health impacts.

A further limitation is the lack of suitable characterization factors for PFAS, which constrains the assessment of toxicity impacts. As a result, studies often rely on proxy substances, separate risk assessments, or omit toxicity altogether, reducing consistency and comparability. Overall, these methodological gaps mean that current LCAs do not fully capture the complexity of PFAS treatment systems and may underestimate their true impacts.

RQ3: What are the impacts on human health and the environment of applying innovative technologies to treat PFAS-contaminated water in Sweden?

To address this question, an attributional life cycle assessment is performed on five treatment trains designed and tested for the treatment of PFAS-contaminated groundwater in Uppsala, Sweden. The functional unit is delivery of 1 m^3 of treated drinking water with PFAS-4 concentration below 4ng/L.

The results provide new insights into environmental and human health impacts associated with treating PFAS-contaminated water with innovative TTs, which were divided into four main steps: pre-treatment, PFAS removal, post-treatment, and PFAS destruction. Overall, the case study demonstrates that the environmental and human health performance of PFAS treatment systems varies across treatment trains, with no clear answer to which TT provides better environmental performance. TT 2 (CCMF+PR+FF) presents the highest impacts, followed by TT 1 (GAC+Reactivation) and TT 5 (NF+IEX), while TT 3 (NF+FF) and TT 4 (NF+GAC) perform comparatively better. Across all systems, PFAS removal step is the dominant contributor to environmental burdens, driven primarily by chemical production, energy use, and material requirements such as membranes, IEX, and GAC. Post-treatment processes like remineralization (after membrane filtration) and disinfection also contribute significantly, whereas PFAS destruction generally plays a smaller role, except in the case of energy-intensive GAC reactivation.

Across all treatment systems, impacts are generally dominated by two or three key unit processes, such as the production of chemicals and media, transportation and electricity generation. Although Sweden relies on a relatively clean electricity mix, electricity still emerges as a significant contributor to overall impacts, highlighting the energy-intensive nature of PFAS treatment systems and indicating that impacts may be substantially higher in regions where electricity is generated from fossil fuels.

In contrast to these dominant contributors, the small quantities of PFAS-4 present in treated effluents discharged to the environment (e.g., treated process water generated after FF, IEX, GAC, or treatment in a wastewater treatment plant depending on the TT) and those remaining in drinking water contribute only minimally to ecotoxicity and human toxicity (<0.2%) across all TTs.

The application of innovative PFAS treatment technologies in Sweden reveals a clear burden-shifting effect between local benefits and broader environmental and human health impacts. While these systems effectively reduce PFAS exposure via drinking water and thereby protect consumers, they simultaneously generate impacts upstream through energy use, chemical production, material manufacturing, and transport. The PFAS removal step is the primary driver of these burdens, often requiring substantial resource inputs to treat dilute water streams. As a result, the environmental footprint is largely transferred from the point of consumption to other stages of the life cycle and to different geographical regions and people.

This redistribution of impacts raises a fundamental ethical question: whose well-being should take priority? While treatment protects individuals consuming contaminated water, it imposes environmental and health costs on other communities involved in the supply chain. Decisions about implementing PFAS treatment technologies therefore involve implicit value judgments about the distribution of risks and benefits across populations. These findings suggest that, although treatment is necessary to safeguard public health, relying solely on downstream solutions is insufficient. A more sustainable and equitable approach would prioritize reducing PFAS production and emissions at the source, thereby minimizing both direct exposure and the need for resource-intensive treatment.

RQ4: Does removing PFAS from drinking water in fact result in net human health benefit when assessed from a life cycle perspective?

To address this question, a net human health benefit (NHHB) approach is developed and tested to complement traditional LCA by assessing whether the health benefits of removing PFAS-4 from drinking water outweigh the life cycle impacts of operating the treatment plant, thereby determining whether the treatment of PFAS-contaminated drinking water results in an overall net health benefit relative to no-treatment. This approach provides a scientific contribution to the methodological development of assessing the burdens and benefits of innovative treatment systems for PFAS-contaminated water.

To enable a more comprehensive evaluation of the potential range of impacts from direct PFAS ingestion through drinking water, two characterization factors (CFs) based on effect factors (EFs) derived from (1) repeated-dose rodent studies and (2) epidemiological data extrapolated to non-cancer human lifetime equivalent are calculated and applied in the NHHB approach.

The results show that, overall, the potential benefits of removing PFAS-4 from drinking water are not overcome by the potential impacts generated by the DWTP's operation life cycle, at least when LCA norms are applied, that is, when CFs derived from EF_{rodent} are applied. This indicates that the additional impacts associated with the PFAS removal and destruction steps are greater than the impacts derived from the null option of not treating PFAS. In other words, implementing PFAS removal technologies generates more human health related impacts than treating the water without these additional processes, thereby increasing the total burden of illness in people (locally and even globally).

When epidemiological EFs are used, the benefit of treatment increases by factors of approximately 3 to 4 orders of magnitude, as epidemiology-based EF implies a much higher toxicity per unit intake compared to the rodent-based EF. Consequently, for the same emission or intake level, the estimated human health impacts of drinking contaminated water are much higher when epidemiological data are used. These results demonstrate that human toxicity outcomes are highly sensitive to the choice of EFs, reflecting a fundamental methodological trade-off that LCA practitioners and LCIA experts should therefore carefully consider when selecting approaches for CF calculation, particularly in the context of assessing human health impacts from direct exposure to contaminants.

Nevertheless, the NHHB proved to be a practical and valuable approach to understand weather the treatment of PFAS-contaminated drinking water results in an overall net health benefit relative to no-treatment. The developed PFOA CFs can be applied beyond the Swedish context as they account for the direct ingestion of the substance. However, since NHHB depends on prior LCA results, it cannot be used as a standalone method and may become time-intensive when applied to multiple TTs or sensitivity analyses scenarios.

6.2 Future research

This thesis provides a comprehensive overview of the environmental, economic, and human health implications of applying innovative technologies to treat PFAS-contaminated water. Future research could expand this work by applying life cycle costing to newly designed and tested treatment trains for PFAS-contaminated water, both in Sweden and globally, depending on data availability.

With respect to life cycle assessment, future studies could investigate how environmental and human health impacts vary under different PFAS treatment targets, for example in response to varying regulatory limits across countries. In addition, the net human health benefit analysis could be further developed to identify threshold PFAS concentrations at which treatment yields greater overall benefits relative to no-treatment.

Another way forward would be to focus on incorporating a broader range of PFAS toxicant flows and relevant characterization factors into life cycle assessment studies to more comprehensively capture toxicity-related impacts. Particular attention could be given to addressing methodological challenges associated with the selection of characterization factors (e.g., for PFOA) and to contributing to the development of a scientific consensus. Such efforts would help bridge the gap between rodent-based toxicity data and human-relevant risk assessments, thereby improving the robustness and comparability of future LCAs.

A different research direction could investigate the environmental and human health implications of PFAS destruction processes, with a specific focus on the potential formation of transformation products during incineration and reactivation. This would help to bridge the gap between total destruction of PFAS and the real-world conditions.

Additionally, it would be interesting to assess the consequences of a PFAS ban from a life-cycle perspective. It could be assessed by comparing continued-use and phase-out scenarios, accounting for both avoided PFAS emissions and the downstream changes in water-treatment requirements. In water treatment, this could include changes in energy and chemical use, infrastructure needs, treatment residues, destruction or disposal routes, and avoided human-health and ecotoxicity impacts. This approach would be helpful because it moves beyond simply reducing PFAS emissions and instead evaluates whether a ban actually lowers overall environmental and human health impacts across the entire system, including water treatment and potential trade-offs.

7. Conclusions

The thesis assesses the environmental, economic, and human health implications of applying innovative technologies to treat PFAS-contaminated water. By addressing the developed four research questions, the thesis contributes to the understanding of the main implications related to decreasing PFAS exposure through water treatment.

The findings show that while these technologies are effective in reducing PFAS concentrations in drinking water, they are associated with considerable environmental burdens and variable costs. Life cycle assessment results indicate that PFAS removal processes are the main contributors to climate impacts, often exceeding those of destruction, due to their energy- and resource-intensive nature. At the same time, life cycle costing reveals that operational expenses dominate overall costs. Together, these results highlight that no single technology performs optimally across all criteria, and that system performance is highly context dependent.

The thesis also identifies significant methodological limitations in existing life cycle assessments of PFAS treatment technologies. In particular, current studies often fail to fully capture PFAS fate, exposure pathways, and downstream emissions, while the lack of robust characterization factors limits the assessment of toxicity-related impacts. As a result, environmental and human health impacts may be underestimated, and comparisons across studies remain uncertain. Addressing these gaps is essential for improving the reliability and completeness of future assessments.

Most importantly, the results demonstrate a clear burden-shifting effect associated with PFAS treatment. While treatment reduces human exposure to PFAS in drinking water, it simultaneously transfers environmental and health burdens to other stages and people of the life cycle, including energy production, chemical and media manufacturing, and waste management. This redistribution of impacts raises important ethical considerations regarding whose well-being should be prioritized, as the benefits of reduced exposure for consumers may come at the expense of other populations affected by upstream processes.

Overall, the thesis underscores that, although advanced treatment technologies are necessary to mitigate PFAS contamination in drinking water, they are not a standalone solution. A more sustainable and equitable approach requires complementing treatment with upstream measures aimed at reducing PFAS production and emissions at the source. Such strategies would help minimize both direct exposure risks and the broader environmental and human health impacts associated with resource-intensive treatment systems

Future work is aimed at expanding both economic and environmental assessments by applying life cycle costing to emerging treatment trains and by refining life cycle assessment methodologies to better capture toxicity, PFAS fate, and exposure pathways. Together, these research directions can contribute to more comprehensive, accurate, and policy-relevant evaluations of PFAS treatment technologies.

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