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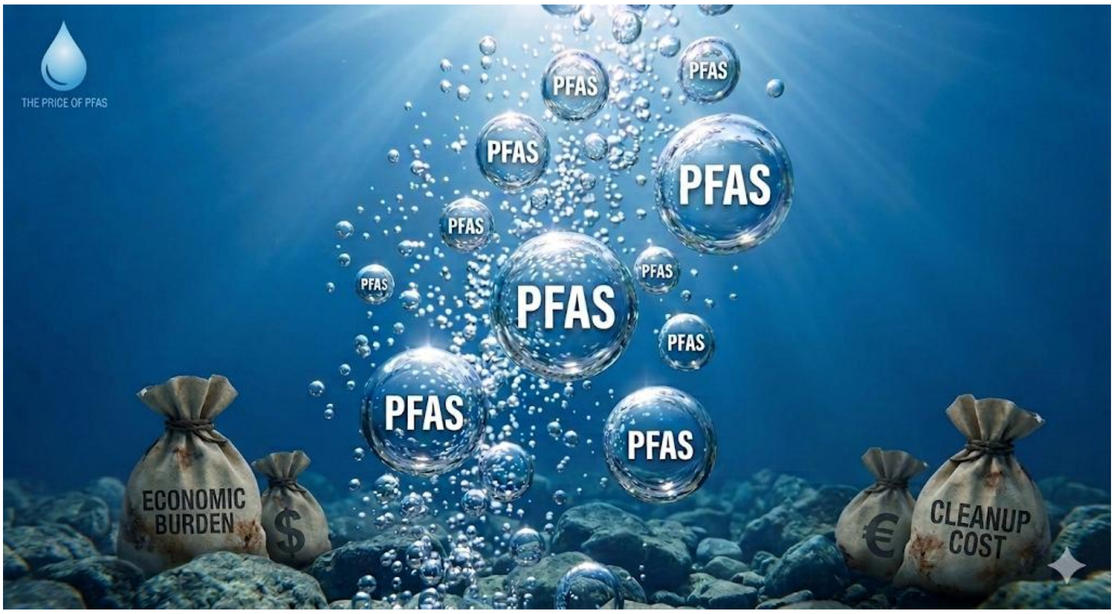

Economic Burdens of PFAS in Residual Streams from Drinking Water Treatment

Master’s thesis in Environmental and Infrastructure Engineering

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Division of Water Environment Technology
CHALMERS UNIVERSITY OF TECHNOLOGY
Gothenburg, Sweden 2026
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MASTER'S THESIS ACEX30

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Master's Thesis in the Master's Programme in Infrastructure and Environmental Engineering

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Treatment

Master's thesis in the Master's Programme Master's Programme Name

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Abstract

Per- and polyfluoroalkyl substances (PFAS) are highly persistent environmental pollutants linked to adverse health effects, with drinking water serving as a primary human exposure route. While previous studies often focus on treatment efficiency and operational costs of PFAS removal, the economic burdens of PFAS transferred to residual streams generated within drinking water treatment processes remain understudied. This study develops a conceptual framework to investigate the fate of PFAS compounds transferred into residual streams and to assess the associated economic impacts. Considering both local recirculation into raw water systems and release into surrounding aquatic environments, the framework was applied to three Swedish drinking water treatment plants with distinct hydrological conditions.

The results demonstrate that hydrological conditions heavily dictate PFAS transport pathways. Closed hydrological systems exhibit high recycling rates and dominant local recirculation costs. In contrast, open systems show limited recirculation but significant downstream transport, resulting in substantial environmental burden costs. Sensitivity analysis revealed that water residence time and time horizon significantly impact these patterns, while higher recipient removal efficiency reduces environmental costs. Conversely, increasing the number of recipient GAC filter units elevates environmental costs.

Furthermore, even small annual PFAS masses can generate substantial downstream treatment burdens due to the high cost of removing diluted pollution. Ultimately, the study highlights the necessity of source-oriented management strategies, showing that minimizing PFAS transfer to residual streams is critical to reducing future environmental disadvantages and costly downstream treatment.

Key words: PFAS, economic burdens, residual streams, drinking water treatment, hydrological pathways, recirculation, release, local recirculation cost, downstream, environmental burden cost

Ekonomiska Bördor av PFAS i Restströmmar från Dricksvattenrening

Examensarbete inom mastersprogrammet infrastruktur och miljöteknik

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Sammanfattning

Per- och polyfluorerade alkylsubstanser (PFAS) är mycket persistenta miljöföroreningar kopplade till negativa hälsoeffekter, där dricksvatten utgör en primär exponeringsväg för människor. Medan tidigare studier ofta fokuserar på reningseffektivitet och driftskostnader för PFAS-avskiljning, är de ekonomiska bördorna av att PFAS överförs till restströmmar inom dricksvattenreningen fortfarande bristfälligt studerade. Denna studie utvecklar ett konceptuellt ramverk för att undersöka PFAS-föreningars slutgiltiga öde när de migrerar till restströmmar, samt för att bedöma de tillhörande ekonomiska effekterna. Med hänsyn till både lokal recirkulation till råvattensystem och utsläpp till omgivande vattenmiljöer tillämpades ramverket på tre svenska dricksvattenreningsverk med olika hydrologiska förhållanden.

Resultaten visar att de hydrologiska förhållandena i hög grad dikterar transportvägarna för PFAS. Slutna hydrologiska system uppvisar höga recirkulationsgrader och dominerande lokala återföringskostnader. Däremot uppvisar öppna system begränsad recirkulation men omfattande nedströmstransport, vilket resulterar i betydande miljökostnader. Känslighetsanalysen visade att vattnets uppehållstid och den valda tidshorisonten påverkar dessa mönster avsevärt, medan en hög avskiljningseffektivitet i recipienten minskar miljökostnaderna. Omvänt ökar ytterligare GAC-filterenheter i recipienten miljökostnaderna.

Dessutom kan även små årliga PFAS-massor generera betydande behandlingsbördor nedströms på grund av den höga kostnaden för att avlägsna utspädda föroreningar. I slutändan belyser studien behovet av källorienterade hanteringsstrategier och visar att en minimering av PFAS-överföringen till restströmmar är avgörande för att minska framtida miljöskulder och undvika kostsam nedströmsbehandling.

Nyckelord: PFAS, ekonomiska bördor, restströmmar, dricksvattenrening, hydrologiska vägar, recirkulation, utsläpp, lokal recirkulationkostnad, nedströms, miljöbelastningskostnad

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Preface

This thesis has been prepared as part of the Infrastructure and Environmental Engineering master's program at Chalmers University of Technology and comprises 30 credits. The work was carried out between January and June 2026 by Amna H M Alhitta.

First and foremost, I wish to express my sincere gratitude to my supervisor, Jesper Knutsson, at the Division of Water Environment Technology. His continuous support, valuable feedback, and insightful guidance throughout this project have been invaluable. His expertise, encouragement, and our constructive discussions truly shaped the direction of this work. I am also grateful to my examiner, Thomas Pettersson, at the Division of Water Environment Technology, for his time, interest, and final evaluation of this thesis.

Sincere appreciation is extended to all the contact persons and representatives at the drinking water treatment plants involved in this study. Thank you for your time, cooperation, and willingness to share the essential data that made this project possible.

Furthermore, a special thank you goes to Mattias Modén at Jacobi Group. His provision of supplier quotations for the conceptual recipient treatment systems was vital for the economic calculations and constituted a cornerstone of this study. A special thank you goes to Stephan Köhler and Emma Risén for providing an excellent report that is not widely available yet.

Finally, my deepest gratitude goes to my family, whose love, support, and encouragement have been my greatest source of motivation throughout this project and my entire academic journey. I am also thankful to my friends and everyone who has supported me along the way; your belief in me has been invaluable.

Gothenburg, June 2026

Amna H M Alhitta

1 Introduction

This chapter provides background information on PFAS substances and their potential hazards. It concludes by outlining the report's objective, research questions, and limitations.

1.1 Background

Per- and polyfluoroalkyl substances (PFAS) are a group of synthetic fluorinated organic chemicals. Since the mid-20th century, these substances have become widely used in a variety of applications, including fire-extinguishing foams, textiles, food packaging, cosmetics, coatings, and many others (Kurwadkar et al., 2022; McCleaf et al., 2017; Swedish Chemicals Agency, 2025). The strong carbon-fluorine bond in their chemical structure gives them high resistance to thermal, chemical, and biological degradation, and therefore, they are very stable in the environment (Buck et al., 2011; Crone et al., 2019; Franke et al., 2017). As a result, PFAS compounds are frequently detected in surface water, groundwater, and drinking water sources worldwide. Due to their impact on the environment and health, these substances are classified as globally important pollutants. Numerous health effects have been reported in association with long-term exposure, including immune suppression, metabolic effects, developmental impacts, and certain cancers (Crone et al., 2019; Säve-Söderbergh et al., 2025; Xu et al., 2023).

Furthermore, their tendency to bioaccumulate and their long biological half-lives mean that even low concentrations in drinking water can lead to significant accumulation in the human body over time (Rosato et al., 2024). Human biological studies have revealed the detectability of these compounds in the blood of a large segment of the population. An exposure study of PFAS in a sample of Swedish adolescents showed the prevalence of exposure in the country and revealed the relationship between exposure to these compounds and drinking water and food, especially seafood (Nyström-Kandola, 2024). According to a meta-analysis of human biomonitoring of exposure to these compounds across different socioeconomic groups. The income levels influence exposure, as high income groups tend to be more exposed to these substances due to their dietary habits (Buekers et al., 2018).

PFAS compounds have been detected in drinking water sources in Sweden, and some Swedish municipalities have experienced significant contamination incidents in their water supplies (Banzhaf et al., 2017; Goldenman et al., 2019). Since water is the most important source of life, and everyone has the right to clean and safe drinking water (Globala målen, 2024; World Health Organization, 2022), setting limits and regulations, as well as monitoring mechanisms, such as establishing maximum permissible concentrations of these substances in drinking water to mitigate their risks, is essential. Regulations for these substances, as defined by the Swedish Food Agency, have set limits of 4 ng/l for PFAS4 and 100 ng/l for PFAS21, based on European risk assessments, to reduce the risks associated with long-term exposure to these substances (Livsmedelsverket, 2022).

Due to high resistance of PFAS to degradation, conventional drinking water treatment processes, such as coagulation, sedimentation, and sand filtration, are generally ineffective. Therefore, specialized treatment techniques are required. A comprehensive review of PFAS presence and treatment techniques shows that the most common removal methods are activated carbon, ion exchange resins, and high-pressure membrane systems (Crone et al., 2019). From a lifecycle and sustainability perspective, the cost and environmental impact of PFAS treatment technologies vary considerably. Recent lifecycle analysis of the technologies has shown that membrane systems are the most efficient at removing PFAS, but they also consume the most energy and produce the highest greenhouse gas emissions, while adsorption technologies, such as activated carbon and ion exchange, have relatively lower environmental impacts but require periodic media replacement and generate waste (Altmeyer Mendes et al., 2025).

Economic factors undoubtedly have a significant impact on decision-making regarding PFAS management. Many studies demonstrate that the cost of inaction far outweighs the cost of action. According to the findings of a socio-economic study conducted by the Nordic Council of Ministers, the cost of inaction in managing these substances is significantly higher than the cost of action. In other words, inaction in PFAS management results in substantial losses, including healthcare expenditures, environmental degradation, and cleanup efforts (Goldenman et al., 2019). Furthermore, the cost of removing PFAS compounds after they have become widespread in the environment can be extremely high, according to cost-scale analysis (Ling, 2024). This underscores the importance of early control and remediation of PFAS emissions and dispersal.

The problem of PFAS cannot be understood from a purely engineering perspective; it encompasses social, economic, and regulatory dimensions. Engineers, water and wastewater utility operators must determine the optimal level of investment in PFAS removal while fully complying with legislative requirements regarding the use of taxpayer funds. This investment must be justified by regulatory and benefit-based criteria, as well as being cost-effective. The level of investment should be assessed considering factors such as efficiency, economic viability, climate change impact, healthcare cost savings, and socioeconomic effects (Altmeyer Mendes et al., 2025; Goldenman et al., 2019; van der Donk & Llavero- Pasquina, 2025). In general, the challenge for water and wastewater companies is to determine the level of investment in PFAS removal that is both socially and economically viable.

1.2 Aim & research questions

This study aims to investigate how PFAS transferred to drinking water treatment residual streams may recirculate back to the raw water system or be released into the surrounding environment. Furthermore, it examines how these pathways generate additional economic burdens, specifically in the form of local recirculation costs and environmental burden costs. However, this project will address the following research questions:

- What potential economic benefits may result from reducing PFAS transfer to residual streams in drinking water systems?

- What types of economic burdens are generated by PFAS transferred to residual streams?
- How do hydrological residence time and receiving system type influence PFAS fate?

1.3 Limitations

The scope of this study had several limitations. The study focuses exclusively on PFAS compounds associated with residual streams from drinking water treatment. Other environmental components and pathways, including wastewater, soil, sediment, and other residual stream types, are excluded to maintain a clear and specific focus. Because it is not possible to cover all PFAS compounds in the analysis as there are more than 10,000 different PFAS substances. The focus was only on PFAS4 and PFAS21 as they are the only PFAS groups currently associated with threshold values established by the Swedish Food Agency.

The framework is based on a static conceptual model and therefore does not simulate long-term accumulation dynamics, temporal changes in PFAS concentrations, or future regulatory developments. Consequently, the results represent annualized conceptual burden estimates rather than long-term environmental trajectories.
Human exposure pathways and health-related impacts are not explicitly modelled within the framework. The study instead focuses primarily on hydrological fate pathways and associated economic burden mechanisms.

The estimated environmental burden costs are based on conceptual recipient treatment scenarios derived from supplier quotations rather than measured downstream treatment systems. The study focuses on three selected water treatment plants (WTPs) representing different hydrological conditions. Water treatment plant A represents a closed receiving system (lake), while water treatment plants C and D represent an open receiving system (river discharging into the sea). Other investigated treatment plants are excluded due to their limited suitability for the recirculation-release modelling framework.

2 Theory

This chapter provides a description of both PFAS properties and behavior, their effects on human health and the environment, along with an economic assessment It also highlights current technologies used for removing or even reducing these compounds in water. Furthermore, it addresses the Swedish regulatory framework for PFAS in drinking water.

2.1 Chemical Characteristics and Environmental Behavior of PFAS

PFAS are a group of thousands of synthetic fluorinated organic compounds that contain at least one fully fluorinated carbon atom in a methyl or methylene group. The chemical structure of PFAS consists of a carbon-fluorine bond, one of the strongest chemical bonds, with dissociation energies ranging from 488 to 536 kJ/mol (Fang et al., 2024; Gulver et al., 2026; Naturvårdsverket, 2025). This bond makes PFAS chemically and thermally stable, allowing them to survive in the ecosystem (Buck et al., 2011; Crone et al., 2019; Franke et al., 2017). Due to the extreme stability of PFAS substances, these substances are therefore known as “forever chemicals” (Cordner et al., 2021; van der Donk & Llavero-Pasquina, 2025).

PFAS are generally classified into perfluoroalkyl substances (fully fluorinated carbon chains) and polyfluoroalkyl substances (partially fluorinated). These substances comprise three main classes: perfluoroalkyl acids (PFAAs), the most environmentally sensitive compounds, including perfluoroalkyl carboxylic acids (PFCAs) and perfluoroalkyl sulfonic acids (PFSAs), as well as other PFAA precursors such as fluorinated polymers and fluorinated polyethers. Chain length significantly influences their environmental fate: long-chain PFAS generally exhibit greater bioaccumulation potential, while short-chain PFAS are more water-soluble and more mobile in groundwater systems (Buck et al., 2011; Crone et al., 2019; Domingo & Nadal, 2019; Fang et al., 2024; Rahman et al., 2014). Due to their amphipathic structure, consisting of a hydrophobic fluorinated carbon tail and a hydrophilic functional head group, PFAS exhibit characteristics of surfactants. In aquatic environments, these compounds are highly soluble and resistant to hydrolysis, photolysis, and biodegradation (Franke et al., 2017; Kucharzyk et al., 2017). Short-chain PFAS compounds exhibit limited uptake in soils and sediments, leading to increased mobility in drinking water sources (Kucharzyk et al., 2017). These physicochemical properties pose significant challenges to conventional drinking water treatment processes.

EU regulatory frameworks often focus on specific PFAS compounds considered to pose the greatest risk to human health and the environment. In this context, particular attention is given to PFAS4 and PFAS21. PFAS4 comprises four individual PFAS compounds, while PFAS21 includes these four plus 17 additional PFAS compounds. Figure 1 illustrates the compounds included in each group (Sveriges geologiska undersökning (SGU), 2024). The four substances included in PFAS4 are among the most studied PFAS compounds. The European Food Safety Authority (EFSA) has classified these compounds as posing significant health risks, particularly through exposure to drinking water (Livsmedelsverket, 2026b). To cover a broader range of

PFAS compounds commonly found in drinking water, regulatory limits have also been established for PFAS21.

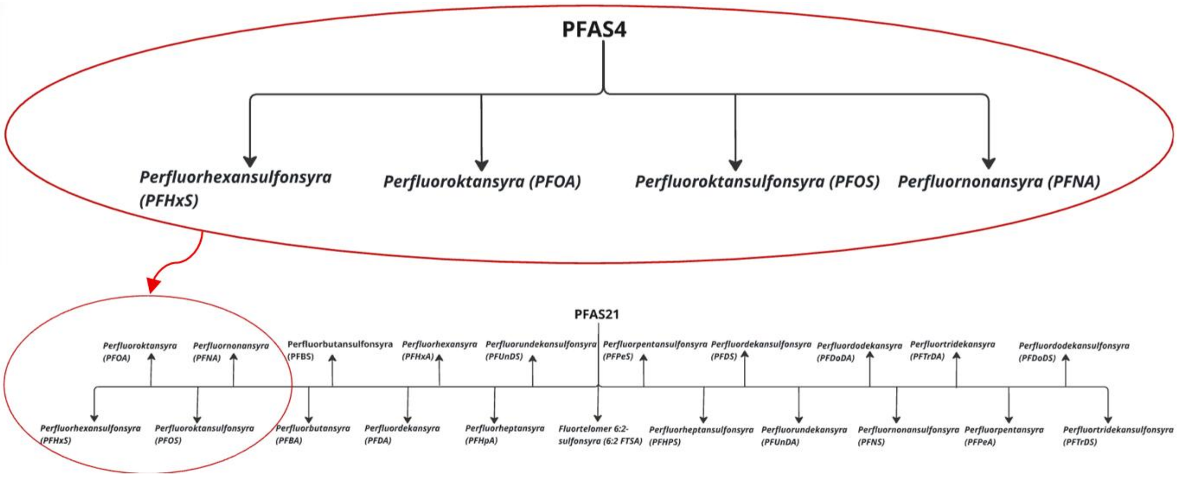


Figure 1: The different PFAS substances in PFAS4 and PFAS21 groups.

2.2 Human exposure and public health impacts

Human exposure to PFAS substances is widespread and well-documented in numerous biomonitoring studies conducted throughout Europe and North America. This widespread exposure and the bioaccumulation of PFAS compounds are demonstrated by measurable concentrations detected in the blood of the general population (Buck et al., 2011; Crone et al., 2019). Drinking water has also been identified as a major route of PFAS exposure in contaminated areas. This route is particularly important in areas contaminated by industrial activities and the use of firefighting foam (Banzhaf et al., 2017; Goldenman et al., 2019; Svenskt Vatten, 2022). Other routes of exposure include the consumption of contaminated foods, especially seafood, as well as indoor dust and consumer products such as cosmetics and paints (Livsmedelsverket, 2026a; Nyström-Kandola, 2024; O'Leary et al., 2016). Once ingested, PFAS compounds are readily absorbed and diffuse into the bloodstream, where they primarily bind to proteins. Due to their slow elimination rate, some long-chain PFAS compounds, such as perfluorooctanoic acid (PFOA) and perfluoro octane sulfonate (PFOS), can bioaccumulate in the body over many years. This means that even with exposure to drinking water containing low levels of these substances, their levels can slowly accumulate in the body (Domingo & Nadal, 2019; Fan et al., 2020; Post, 2020; Rahman et al., 2014; Rosato et al., 2024).

Extensive epidemiological evidence suggests an association between exposure to PFAS and a variety of adverse health effects. These include elevated cholesterol levels, increased risk of type 2 diabetes, kidney and testicular cancer, impaired lipid metabolism, and immune system suppression, including reduced antibody response to vaccination. Other health effects associated with PFAS include thyroid dysfunction, hypertension during pregnancy, low birth weight, and an increased risk of polycystic ovary syndrome (PCOS) and uterine fibroids (Crone et al., 2019; Hammarstrand et al., 2021; Säve-Söderbergh et al., 2025; Wee & Aris, 2023; Xu et al., 2023). Although some uncertainty remains regarding the health effects of PFAS, the consistency of evidence across different exposure groups has led to the establishment of protective concentration levels in drinking water. These low PFAS levels aim to reduce cumulative health risks by minimizing long-term exposure to these compounds, which are a significant health hazard. These low PFAS levels are based on the relationship between the concentration of these substances in drinking water and the resulting health burden (Livsmedelsverket, 2022; Svenskt Vatten, 2022). This relationship forms the conceptual basis for assessing the societal benefits of PFAS treatment.

2.3 Environmental Impacts of PFAS

The environmental behavior of PFAS compounds is primarily governed by their exceptional chemical stability. The strong carbon-fluorine bond makes these compounds highly resistant to hydrolysis, photolysis, microbial degradation, and other natural dilution processes. Consequently, PFAS compounds persist in the environment for extended periods and are often referred to as "persistent organic pollutants" of increasing concern (Crone et al., 2019; Franke et al., 2017; Liu & Mejia Avendaño, 2013). Once released, PFAS compounds are not removed but are redistributed among water, soil, sediments, and living organisms.

In aqueous environments, many PFAS compounds, particularly short-chain PFAS, exhibit high water solubility and limited adsorption onto soil and sediment particles. This increases their mobility and facilitates their long-distance transport through groundwater and surface water systems (Crone et al., 2019; Narasimhappa et al., 2025; Rahman et al., 2014). Long-chain PFAS compounds such as PFOA and PFOS tend to be more adsorbed on organic matter; however, they remain environmentally stable and can accumulate in sediments (United States Environmental Protection Agency (EPA), 2017). Contaminated soils may act as long-term secondary sources, gradually releasing PFAS into the surrounding groundwater over decades (Brusseau et al., 2020; Hunter Anderson et al., 2019). The transport of volatile primary materials from PFAS through the atmosphere contributes to their global spread, leading to pollution even in remote and polar regions (Ahrens & Bundschuh, 2014; Vento et al., 2012).

In addition to their stability and mobility, PFAS compounds exhibit bioaccumulation in wildlife. Unlike many conventional hydrophobic pollutants that accumulate in adipose tissue, PFAS compounds preferentially bind to proteins in the blood and liver. Long-chain PFAS compounds have demonstrated significant bioaccumulation in animals, and biomagnification has been observed in aquatic food webs (Ahrens & Bundschuh, 2014; Wen et al., 2023). High concentrations in top predators indicate potential ecosystem-wide risks. Experimental and field studies have linked PFAS exposure to reproductive toxicity, developmental disorders, endocrine disruption, and immunotoxin effects in animals and aquatic organisms (Nyström-Kandola, 2024; Savoca & Pace, 2021). PFAS can also be taken up by plants through contaminated soil and irrigation water, allowing entry into terrestrial food chains (Adu et al., 2023). This leads to the creation of interconnected exposure pathways across environmental components, reinforcing the systemic nature of PFAS pollution.

2.4 PFAS-Related Environmental and Treatment Burdens

Exposure to PFAS through contaminated drinking water contributes to significant social and environmental burdens. In addition to potential health effects, PFAS contamination increases the long-term costs associated with drinking water treatment, environmental monitoring, and resource management and remediation (Cordner et al., 2021; European Commission et al., 2026; Goldenman et al., 2019). Because PFAS compounds are stable and diffuse in aquatic systems, they can migrate between water bodies and persist in the environment for extended periods. This poses challenges for drinking water treatment systems and may facilitate the transfer of PFAS contamination between local water systems and surrounding aquatic environments.

In addition to the direct treatment costs in drinking water systems, PFAS contamination can generate external environmental burdens related to receiving water treatment and pollution management. These impacts are often discussed in the context of damage costs and abatement costs in environmental economics theory. Damage costs refer to the societal and environmental consequences associated with the contamination, while abatement costs represent the costs of reducing, preventing or removing pollutants from the environment through treatment technologies and remediation measures (Tietenberg & Lewis, 2012). Within the present study, these concepts are applied through the environmental burden cost framework, where downstream treatment requirements and environmental management efforts associated with released PFAS are conceptualized as environmental burden costs.

Environmental economics theory by Tietenberg & Lewis (2012) dictates that pollutants should ideally be treated at the source where intervention is most efficient, thereby avoiding the exponentially higher costs associated with widespread dispersion and downstream remediation. As contamination spreads, remediation becomes technically complex and economically demanding; socioeconomic analyses confirm that delayed intervention significantly inflates cleanup costs (Goldenman et al., 2019; Ling, 2024). The estimated annual health costs of PFAS exposure in the Nordic countries range from €2.8 billion to €4.6 billion. In addition, non-health costs in Sweden, such as environmental remediation and management, are estimated at between €18 million and €5.1 billion over 20 years (about €0.9–€255 million annually), depending on the level of contamination (European Commission et al., 2026; Goldenman et al., 2019). However, these macroeconomic numbers only reflect the major regional impacts affecting approximately 3% of the total population, meaning they likely underestimate the full societal burden. Therefore, this analysis provides a very limited picture of how the PFAS cost burden is managed at the local level, particularly in drinking water treatment plants.

This study fills this gap by developing a conceptual model to assess the cost of local recirculation, as well as the cost of environmental burden caused by PFAS leaching from drinking water treatment plants into adjacent water sources. In the developed conceptual model, the specific environmental treatment cost ($\mathrm{Cost}_{\mathrm{env,\ specific}}$) can be compared to the treatment burden incurred to remove diluted PFAS concentrations from downstream receiving systems. The latter illustrates how delaying downstream treatment becomes significantly more costly than previous source-oriented processing and recycling strategies.

2.5 PFAS Treatment Technologies

PFAS treatment technologies can be categorized into separation technologies (adsorption and membranes), concentration technologies (reverse osmosis and nanofiltration), and destruction technologies (electrochemical oxidation, plasma, and thermal processes). Most currently implemented drinking water systems rely on separation rather than mineralization, meaning PFAS are transferred into secondary waste streams rather than destroyed.

Granular activated carbon (GAC) is one of the commonly applied used technologies for removing PFAS in drinking water treatment plants. In the application of GAC, PFAS is removed from the water via adsorption on the surface of the carbon using hydrophobic and electrostatic interactions. Extensive trials have demonstrated the effectiveness of GAC in removing long-chain PFAS with a removal efficiency exceeding 90%, although its effectiveness decreases with increasing carbon saturation. Effectiveness also decreases with short-chain PFAS. Long-term trials have also shown that high flow rates reduce removal efficiency due to the shorter contact time (Belkouteb et al., 2020; Köhler et al., 2026; Roest et al., 2021).

Ion exchange resins (IEX) offer a variety of adsorption techniques, where negatively charged PFAS molecules bind to positively charged molecules. IEX techniques are particularly effective at removing short-chain PFAS compounds from water, especially compared to granular GAC. However, these resins may be more sensitive to different water compositions (Crone et al., 2019; Roest et al., 2021).

Dissolved air flotation (DAF) technology relies on microscopic air bubbles to separate pollutants in water. These bubbles absorb PFAS compounds through their surface activity, then migrate to the water's surface and accumulate in the floating sludge. PFAS removal efficiency depends primarily on chain length, with longer-chain compounds being removed more efficiently than shorter-chain compounds. The PFAS4 removal efficiency typically recorded with DAF ranges from 10% to 40%, with higher efficiency achievable for some longer-chain PFAS compounds. Due to its relatively low removal efficiency, DAF is often used combined with other methods, such GAC, to increase efficiency. The main operating costs of DAF are related to the energy consumption required for air saturation, compression, and microbubble generation (Köhler et al., 2026; Norrvatten, 2026).

High-pressure membrane processes, including nanofiltration and reverse osmosis, rely on separating PFAS compounds from water and other solutions based on particle size and charge. High-pressure membranes have proven highly effective in processing a wide range of PFAS compounds. However, this technology is energy-intensive, resulting in high capital and operating costs, as well as generating significant amounts of waste that must be disposed of (Altmeyer Mendes et al., 2025; Crone et al., 2019).

Newly developed destructive technologies target the breakdown of the compound instead of its separation. These technologies include supercritical water oxidation

(SCWO), advanced oxidation technologies, and electrochemical technologies. Conference and pilot studies have demonstrated that SCWO can destroy PFAS under extremely high temperature and pressure conditions. However, these technologies are still energy-intensive and not yet implemented on a full scale (Jama et al., 2020).

2.6 Regulatory framework for PFAS in drinking water

In Sweden, PFAS are strictly controlled through a combination of national legislation and EU regulations, as well as the efforts of numerous agencies involved in monitoring, assessing, and reducing exposure to these substances, particularly in drinking water sources, due to their potentially serious effects. The regulatory framework within the EU is governed by REACH regulations, which regulate the use and handling of chemicals (Kemikalieinspektionen, 2015)

At the national level, responsibility for monitoring these substances is shared among several agencies. The Swedish Chemicals Agency is responsible for regulating the use of PFAS in products and industries and for developing coordinated national action plans to reduce emissions and exposure to these substances (Kemikalieinspektionen, 2025; Swedish Chemicals Agency, 2025). The Swedish Food Agency also plays a key role in regulating the use of PFAS in drinking water. Based on long-term exposure risk assessments, the agency develops guidelines and regulations that are binding on drinking water producers. The current limit values are 4 ng/L for PFAS4 and 100 ng/L for PFAS21. Drinking water producers are required to monitor and measure the concentrations of these substances in the produced water to ensure compliance with these limits (Kemikalieinspektionen, 2025; Livsmedelsverket, 2022)

The Swedish Agency for Marine and Aquatic Environment Management is responsible for assessing the environmental status of surface and groundwater, including environmental quality standards for PFAS. These standards include limit values for PFOS in water and organisms, as well as an index for estimating PFAS concentration in raw water sources used in drinking water production. The agency also provides guidance on water protection zones and environmental monitoring programs. In addition, the Swedish Environmental Protection Agency provides guidance on contaminated sites and PFAS risk assessment under the Environmental Act. In a spirit of cooperation and mutual support, this agency works with other entities to provide information on the presence of PFAS compounds in the environment and supports national monitoring programs (Kemikalieinspektionen, 2025).

3 Methodology

To conduct the socioeconomic analysis of PFAS treatment in Swedish waterworks, the methodology was divided into different key components that will be described in this chapter. These components include:

3.1 Literature search

For the literature reviews, Google and Google Scholar databases were primarily relied, along with additional references provided by the project supervisor. Searches were conducted in peer-reviewed journals, scientific reports and information obtained from credible and authoritative web sources. During the search process, keywords were used to achieve the various study objectives. Keywords that were generally used included "PFAS" together with other specific ones based on the research topics and methods. Examples of keyword combinations used include:

- "Socioeconomic analysis" AND "PFAS"
- "PFAS" AND "drinking water" AND "Sweden"
- "PFAS" AND "dricksvatten" AND "Sverige"
- "PFAS" AND "drinking water" AND "treatment technologies"
- "PFAS" AND "hydrological residence time"
- "PFAS" AND "Regulatory framework" AND "Sweden"

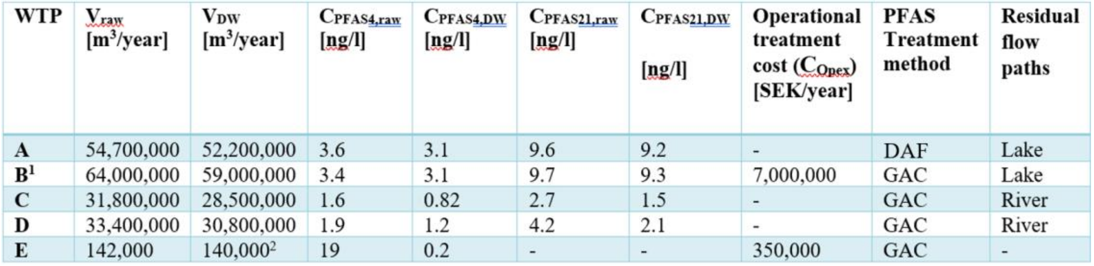
To improve the relevance and comprehensiveness of the literature search, snowballing techniques and citation tracking were applied. This included examining reference lists and citations within the initially identified publications to identify additional relevant sources.

3.2 Data collection and operational cost estimation

Data regarding PFAS concentrations and their associated treatment costs were collected from various Swedish water treatment plants. The identities of the facilities have been withheld to ensure anonymity. The data collected from year 2025 are summarized in Table 1 and include the following components:

- Annual raw water withdrawal (V_{raw})
- Annual drinking water production (V_{DW})
- Average PFAS concentrations in raw water ($C_{PFAS,raw}$)
- Average PFAS concentrations in treated drinking water ($C_{PFAS,DW}$)
- Annual Operational treatment cost
- Energy use
- Treatment techniques
- Residual flow paths

Table 1: Data collected from water treatment plants for PFAS4 and PFAS21.



WTP	V_{raw} [m^3/year]	V_{DW} [m^3/year]	$C_{PFAS4,raw}$ [ng/l]	$C_{PFAS4,DW}$ [ng/l]	$C_{PFAS21,raw}$ [ng/l]	$C_{PFAS21,DW}$ [ng/l]	Operational treatment cost (C_{Opex}) [SEK/year]	PFAS Treatment method	Residual flow paths
A	54,700,000	52,200,000	3.6	3.1	9.6	9.2	-	DAF	Lake
B[1]	64,000,000	59,000,000	3.4	3.1	9.7	9.3	7,000,000	GAC	Lake
C	31,800,000	28,500,000	1.6	0.82	2.7	1.5	-	GAC	River
D	33,400,000	30,800,000	1.9	1.2	4.2	2.1	-	GAC	River
E	142,000	140,000[2]	19	0.2	-	-	350,000	GAC	-

Additional information regarding mobile GAC treatment systems is obtained from supplier quotes, in contact with Mattias Modén at Jacobi Group, which is shown in Table 2.

Table 2: Data regarding mobile GAC treatment systems

WTP	Number of filters (N)	Vessel configuration	Rent [SEK/year/filter]	Carbon weight (W_c) [kg/filter]	Carbon price (P_c) [SEK/kg]	Shipping cost ($Cost_{ship}$) [SEK/filter]	Service interval[3] (SI) [year]
C	6	3 parallel lines×2 in series	9500	9000	21	40,000	1
D	4	2 parallel lines×2 in series	9500	9000	21	40,000	1

Only WTPs C and D were subjected to the recipient downstream treatment scenario. WTP A was excluded because it represents a closed hydrological system with negligible downstream discharge, making the recipient treatment scenario conceptually less relevant within the applied framework. WTPs B and E were also excluded because mass balance calculations showed zero transfer of PFAS to residual streams ($m_{PFAS,rest}$= 0), indicating that the removed PFAS mass was retained entirely within the internal treatment system. Consequently, these two plants were excluded from the entire study.

For WTPs (A, C and D) where energy cost and annual operational treatment cost data related to PFAS treatment are not available, independent calculations were performed, Table 3 shows summary of those calculations. WTP (A) utilizes flotation as a treatment method for PFAS compounds. Because this method increases electricity consumption, it primarily impacts energy cost. To estimate this cost, data were utilized from a 2026

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[1] The reported raw water intake and drinking water production are 63 and 59 million m^3/year, respectively, a flow of 60 million m^3/year is used in the PFAS mass balance calculations, corresponding to the flow passing through the GAC filters. According to the water utility, approximately 3 million m^3/year was returned before PFAS treatment and 1 million m^3/year after treatment.

[2] The reported drinking water production is 140,000 m^3/year, a value of 142,000 m^3/year is used for both raw water intake and treated water production in the mass balance calculations. According to the water utility, the difference represents internally reused treated water for filter backwashing, ion-exchange regeneration, and online analysis.

[3] Time before the carbon replacement.

technical report by Norrvatten, a WTP that also employs DAF treatment. The report provides information on the energy requirements of a similar DAF process, including approximate electricity prices and specific energy consumption per cubic meter of treated water. By applying these reference values to the reported annual drinking water production of Plant A, the annual energy cost was estimated using Equation (3.1):

$$C_{\text{annual}} = V_{\text{DW}} \times E_{\text{spec}} \times P_{\text{el}} \tag{3.1}$$

$$C_{\text{annual}} = 52{,}200{,}000 \times 1.2 \times 0.025 \approx 1{,}566{,}000\ SEK/year$$

Where:

C_{annual} = Annual energy cost *[SEK/year]*

V_{DW} = Annual drinking water production *[m^3/year]*

E_{spec} = Specific energy consumption *[kWh/m^3]*

P_{el} = Approximate electricity price *[SEK/kWh]*

The annual operational treatment costs for Plants C and D were estimated using reference data from WTP B. Even though this facility has been excluded from the study, it provides a reasonable and scalable baseline for operational treatment unit costs since all three facilities employ identical GAC treatment technology. The operational treatment unit cost per m^3 of plant B is estimated with Equation (3.2):

$$c_{Opex,unit} = \frac{c_{Opex,B}}{v_{DW,B}} \tag{3.2}$$

$$c_{Opex,unit} = \frac{7{,}000{,}000}{59{,}000{,}000} \approx 0.12\ SEK/m^3$$

Where:

$c_{Opex,unit}$ = Operational treatment unit cost *[SEK/m^3]*

C_{Opex}= Annual operational treatment cost *[SEK/year]*

This unit cost is subsequently multiplied by the annual drinking water production of WTPs C and D to estimate their corresponding annual operational treatment costs, according to Equation (3.3).

$$C_{Opex} = V_{DW} \times c_{Opex,unit} \tag{3.3}$$

$$C_{Opex,C} = \ 28,500{,}000 \times 0.12 \approx 3{,}325{,}000\ SEK/year$$

$$C_{Opex,D} = \ 30{,}800{,}000 \times 0.12 \approx 3{,}593{,}333\ SEK/year$$

Table 3: Summary of calculated operational treatment cost/ energy cost in the investigated water treatment plants.

WTP	Annual Operational Treatment Cost/ Energy Cost *[SEK/year]*
A	1,566,000
C	3,325,000
D	3,593,333

3.3 Mass balance model

The PFAS mass calculations for raw water and drinking water in grams per year [*g/year]* are established as Equations (3.4) and (3.5):

$$m_{PFAS,raw} = C_{PFAS,raw} \times V_{raw} \times 10^{-6} \tag{3.4}$$
$$m_{PFAS,DW} = C_{PFAS,DW} \times V_{DW} \times 10^{-6} \tag{3.5}$$

Where C is the PFAS concentration in nanograms per Liter [n*g/l]*, while V is the annual water volume in cubic meters per year [m^3*/year]*.

The mass balance was calculated in two steps. The first step involved calculating the PFAS mass removed during treatment, defined as the difference between the PFAS mass in the raw water and the PFAS mass in the treated drinking water, using the following Equation (3.6):

$$m_{PFAS,removal} = m_{PFAS,raw} - m_{PFAS,DW} \tag{3.6}$$

The second step involved calculating the PFAS mass transferred to residual streams ($m_{PFAS,rest}$) which was determined by the difference between the PFAS mass removed during treatment and the PFAS mass retained in solid phases (e.g., GAC or sludge) and calculated with Equation (3.7):

$$m_{PFAS,rest} = m_{PFAS,removal} - m_{PFAS,retained} \tag{3.7}$$

The PFAS mass retained in solid phases is calculated with Equation (3.8) for plants B-E:

$$m_{PFAS,retained} = m_{PFAS,raw} \times \eta_{WTP} \tag{3.8}$$

Where:

η_{WTP} = removal efficiency of the water treatment plant

For plant (A), Equation (3.8) was not applied because the PFAS removal was achieved through DAF. In flotation systems, only a portion of the removed PFAS mass is retained in the flotation sludge, while a significant portion may remain in the water phase and be transferred to residual streams. Therefore, directly applying the overall treatment removal efficiency could overestimate PFAS retention. Consequently, the PFAS retention data reported by (Norrvatten, 2026) for a similar DAF-based PFAS treatment process were used as a representative estimate.

3.4 Hydrological model

The conceptual model distinguishes between two pathways for PFAS in residual streams: recirculation to the raw water source and environmental release. Recirculated PFAS contributes to local recirculation costs, while released PFAS contributes to environmental burden costs as shown in Figure 2. The key parameters used with those pathways are the recirculation coefficient (f) and the released coefficient(f_{release}).

The recirculation factor (f) represents the fraction of PFAS substances in residual streams that return to the raw water system over time, and is calculated with Equation (3.9) as follows:

$$f = 1 - e^{-t/\tau} \tag{3.9}$$

Where:

- t = time horizon (selected to be 30 years),
- τ = hydrological residence time from the discharge point back to the raw water intake.

A 30-year time horizon was chosen to reflect the long-term persistence of PFAS compounds and is consistent with typical system lifetimes (15–30 years) reported in PFAS treatment technology lifecycle assessment studies (Altmeyer Mendes et al., 2025; Song et al., 2024). Residence time τ depends on the type of receiving system. For open systems, such as a river flowing into the sea, τ is virtually infinite because the water is removed from the local hydrological circulation. As the discharge point is located downstream of the intake for WTPs C and D, τ is set to 4000 years. Since the average residence time in the sea is several thousand years, it is impossible for these substances to return to the raw water intake within the study's timeframe (30 years), resulting in a low recirculation coefficient ($f \approx 0$) (Broecker & Peng, 1982). In contrast, for closed and/or semi-closed systems such as lakes, τ reflects the internal circulation of the water body; these times are typically shorter than those of the sea, which produces a higher recirculation coefficient, often approaching 1 (Wetzel, 2001).

The annual PFAS mass returning to the raw water source $m_{PFAS,returned}$ [$g/year$] is calculated with Equation (3.10) as:

$$m_{PFAS,returned} = m_{PFAS,rest} \times f \tag{3.10}$$

The release factor (f_{release}) represents the proportion of PFAS substances present in residual streams that leave the local raw water system and are transported further downstream in the environment instead of returning to the original raw water intake. f_{release} can be calculated with Equation (3.11) as follows:

$$f_{release} = 1 - f \tag{3.11}$$

The annual mass of PFAS released to the system $m_{PFAS,released}$ *[g/year]* is calculated with Equation (3.12) as:

$$m_{PFAS,released} = m_{PFAS,rest} \times f_{release} \tag{3.12}$$

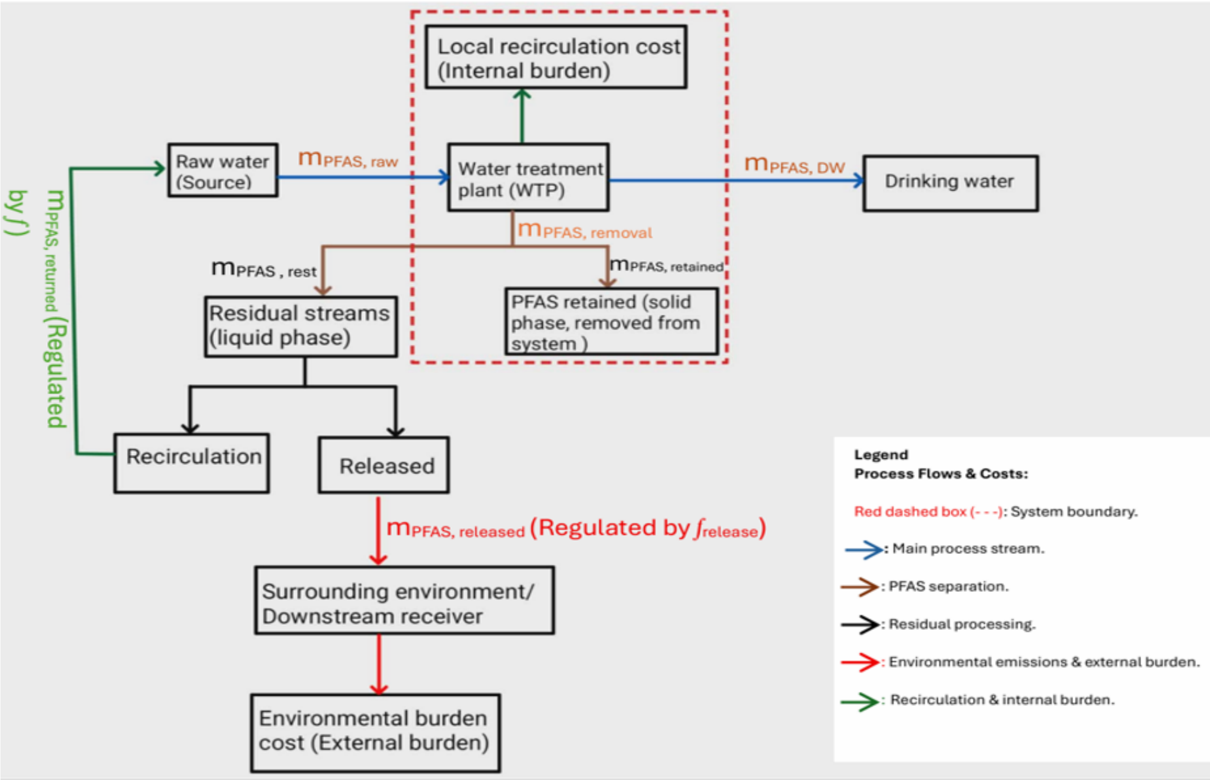


Figure 2: Conceptual model of PFAS mass transport, residual stream routing, and internal vs. external economic burdens.

Table 4 provides a summary of the mathematical expressions for all PFAS masses that have been identified and applied within the scope of this work.

Table 4: Summary of PFAS mass equations.

Mass component	Symbol	Equation	Unit
Raw water mass	$m_{PFAS,raw}$	$m_{PFAS,raw} = C_{PFAS,raw} \times V_{raw} \times 10^{-6}$	g/year
Drinking water mass	$m_{PFAS,\mathrm{DW}}$	$m_{PFAS,\mathrm{DW}} = C_{PFAS,\mathrm{DW}} \times V_{DW} \times 10^{-6}$	g/year
Removal mass	$m_{PFAS,removal}$	$m_{PFAS,removal} = m_{PFAS,raw} - m_{PFAS,\mathrm{DW}}$	g/year
Residual stream mass	$m_{PFAS,rest}$	$m_{PFAS,rest} = m_{PFAS,removal} - m_{PFAS,retained}$	g/year
Retained mass	$m_{PFAS,retained}$	$m_{PFAS,retained} = m_{PFAS,raw} \times \eta_{WTP}$	g/year
Returned mass	$m_{PFAS,returned}$	$m_{PFAS,returned} = m_{PFAS,rest} \times f$	g/year
Released mass	$m_{PFAS,released}$	$m_{PFAS,released} = m_{PFAS,rest} \times f_{release}$	g/year

3.5 Cost allocation framework

The total annual cost ($Cost_{total}$) *[SEK/year]* is defined as the sum of local return cost and total annual environmental cost and calculated with Equation (3.13):

$$Cost_{total} = Cost_{local,return} + Cost_{env,tot.} \tag{3.13}$$

3.5.1 Local recirculation cost

The local return cost $Cost_{local,return}$ [*SEK/year*] reflects the cost of annual internal burden associated with PFAS returning to the raw water system and is calculated with Equation (3.14) as:

$$Cost_{local,return} = Cost_{treatment} \times f \tag{3.14}$$

The specific local cost $Cost_{local,specific}$ *[SEK/kg]* represents the treatment cost normalized to the annual PFAS mass returning to the raw water system, and calculated with Equation (3.15):

$$Cost_{local,specific} = \frac{Cost_{treatment}}{m_{PFAS,returned}} \tag{3.15}$$

Where:

$Cost_{treatment}$ is the total annual operational cost of the waterwork's internal treatment system *[SEK/year]*. Table 5 summarizes the mathematical framework and specific equations used to calculate each element of the local recirculation costs.

Table 5: Summary of cost equations for local recirculation cost.

Cost Component	Symbol	Equation	Unit
Local Return Cost	$Cost_{local,return}$	$Cost_{local,return} = Cost_{treatment} \times f$	SEK/year
Specific Local Cost	$Cost_{local,specific}$	$Cost_{local,specific} = \frac{Cost_{treatment}}{m_{PFAS,returned}}$	SEK/kg

3.5.2 Environmental burden cost

The total annual environmental cost ($Cost_{env,tot.}$) represents the conceptual annual burden associated with PFAS released from the local water system. The framework combines two categories of environmental cost: the first includes the annual technical treatment cost required to remove a fraction of the emitted PFAS compounds ($Cost_{abatement}$), while the second represents the external environmental cost associated with the remaining untreated PFAS fraction ($Cost_{env,ext}$). Together, these components represent both the remediation effort and the residual environmental burden associated with PFAS release. The total annual environmental cost is estimated in *[SEK/year]* and expressed in Equation (3.16), where the underlying components are further defined below:

$$Cost_{env,tot.} = Cost_{abatement} + Cost_{env,ext} \tag{3.16}$$

Substituting the definition of the $\mathrm{Cost}_{\mathrm{env,ext}}$ from Equation (3.21) into Equation (3.16) yields Equation (3.17):

$$Cost_{env,tot.} = Cost_{abatement} + (\frac{Cost_{abatement}}{\eta_{recipient}} \times (1 - \eta_{recipient})) \qquad (3.17)$$

Simplifying Equation (3.17) gives Equation (3.18):

$$Cost_{env,tot.} = \frac{Cost_{abatement}}{\eta_{recipient}} \qquad (3.18)$$

$\eta_{recipient}$ is the downstream recipient removal efficiency, and it is applied to 90%. $Cost_{abatement}$ *[SEK/year]* represents the annual technical cost of operating the conceptual downstream treatment system associated with PFAS released from the local water system into the receiving environment, and it is calculated using Equation (3.19):

$$Cost_{abatement} = N \times Rent + \frac{N \times (W_{\mathrm{c}} \times P_{\mathrm{c}} + Cost_{\mathrm{ship}})}{SI} \qquad (3.19)$$

$Cost_{abatement}$ is estimated from supplier quotations based on conceptual GAC treatment systems dimensioned according to the equivalent annual treatment volume and flow conditions. Supplier quotations were requested using the study's input assumptions, such as volumetric processing capacity (m^3*/year* and m^3*/hour*), contact time, and assumed concentration levels, while the number of filters, carbon price, carbon weight, carbon replacement interval, and transport requirements were derived from the received supplier quotations. Consequently, the estimated annual treatment cost is scaled with the conceptual treatment volume and associated system sizing assumptions (such as GAC filter bed dimensions and vessel configurations).

The external environmental cost *[SEK/year]* represents the conceptual annual cost associated with PFAS fraction that leaves the local water system and remains untreated within the downstream recipient treatment scenario. It is calculated with Equation (3.20) as following:

$$Cost_{env,ext} = m_{PFAS,released} \times Cost_{env,specific} \times (1 - \eta_{recipient}) \qquad (3.20)$$

Substituting the definition of the $\mathrm{Cost}_{\mathrm{env,specific}}$ from Equation (3.23) into Equation (3.20) yields Equation (3.21):

$$Cost_{env,ext} = \frac{Cost_{abatement}}{\eta_{recipient}} \times (1 - \eta_{recipient}) \qquad (3.21)$$

Where $Cost_{env,specific}$ *[SEK/kg]* is the specific environmental cost and is defined as annual technical cost of operating the conceptual downstream treatment system divided by the annual PFAS mass removed in the recipient treatment system and calculated with Equation (3.22):

$$Cost_{env,specific} = \frac{Cost_{abatement}}{m_{PFAS,abatement}} \qquad (3.22)$$

Substituting the definition of the $\mathrm{m_{PFAS,abatement}}$ from Equation (3.27) into Equation (3.22) yields Equation (3.23):

$$Cost_{env,specific} = \frac{Cost_{abatement}}{m_{PFAS,released} \times \eta_{recipient}} \tag{3.23}$$

$m_{PFAS,abatement}$ represents the annual mass of PFAS that can theoretically be removed within a conceptual downstream treatment scenario, estimated in *[g/year]* and is calculated with Equation (3.24):

$$m_{PFAS,abatement} = V_{abatement} \times \eta_{recipient} \times C_{raw} \tag{3.24}$$

Where $V_{abatement}$ represents the equivalent annual treatment volume *[m³/year]* required to contain the same PFAS mass released to the receiving environment at the corresponding raw water concentration and is calculated with Equation (3.25) as follows:

$$V_{abatement} = \frac{m_{PFAS,released}}{C_{raw}} \tag{3.25}$$

Although the concentration term is mathematically omitted during equation substitution, the equivalent annual treatment volume ($V_{abatement}$) is retained as an explicit intermediate parameter because it is used both to determine the theoretical treatment system size and to estimate the theoretically removable PFAS mass ($m_{PFAS,abatement}$) within the downstream treatment scenario. Since these physical dimensions of the conceptual GAC system dictate the supplier's pricing, retaining $V_{abatement}$ serves as a necessary computational step rather than merely a pedagogical representation.

Substituting $\mathrm{V_{abatement}}$ from Equation (3.25) into Equation (3.24) yields Equation (3.26):

$$m_{PFAS,abatement} = \frac{m_{PFAS,released}}{C_{raw}} \times \eta_{recipient} \times C_{raw} \tag{3.26}$$

Simplifying Equation (3.26) gives Equation (3.27):

$$m_{PFAS,abatement} = m_{PFAS,released} \times \eta_{recipient} \tag{3.27}$$

Environmental burden cost calculations are performed only for water treatment plants C and D, as these two plants showed the highest external burden and allowed for more reliable estimates. Table 6 summarizes the mathematical framework and specific equations used to calculate each element of the environmental burden costs.

Table 6: Summary of cost equations for environmental burden.

Cost Component	Symbol	Equation	Unit
Total Environmental Cost	$Cost_{env,tot.}$	$Cost_{env,tot.} = \frac{Cost_{abatement}}{\eta_{recipient}}$	SEK/year
Technical Downstream Treatment Cost	$Cost_{abatement}$	$Cost_{abatement} = N \times Rent + \frac{N \times (W_c \times P_c + Cost_{ship})}{SI}$	SEK/year
External Environmental Cost	$Cost_{env,ext}$	$Cost_{env,ext} = \frac{Cost_{abatement}}{\eta_{recipient}} \times (1 - \eta_{recipient})$	SEK/year
Specific Environmental Cost	$Cost_{env,specific}$	$Cost_{env,specific} = \frac{Cost_{abatement}}{m_{PFAS,released} \times \eta_{recipient}}$	SEK/kg

3.5.3 Sensitivity analysis

A multi-scenario sensitivity analysis is conducted to assess the impact of key parameters on the model results. The following parameters are varied:

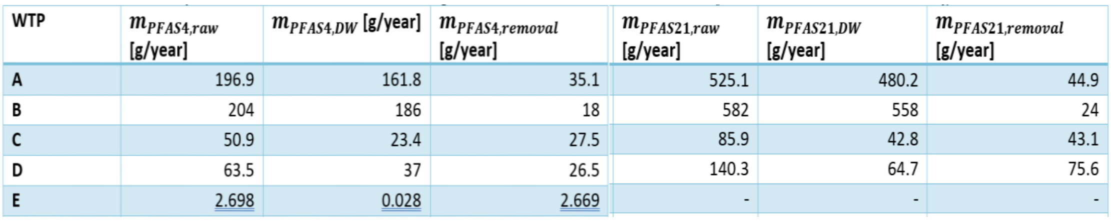
- Hydrological residence time τ
- Time horizon (t)
- Recipient removal efficiency $\eta_{recipient}$
- Number of recipient GAC filter units

Low and high residence time and time horizon scenarios are evaluated for both closed and open receiving systems. PFAS removal efficiencies of 70% and 99% are evaluated for recipient treatment scenarios (i.e., treatment plants C and D). Furthermore, the number of mobile GAC filter units is varied for treatment plant C and D to assess how changes in treatment capacity and equipment scaling impact the abatement costs, environmental external costs and specific environmental cost.

4 Results

Tables 7 below show the annual mass balance calculations for PFAS4 and PFAS21 in raw water, treated drinking water, and the mass of PFAS removed. The results indicate significant differences among the studied water treatment plants with respect to PFAS removal efficiency and resulting residual load. Because PFAS21 encompasses a wider range of PFAS compounds than PFAS4, it generally produced larger annual masses of PFAS. The lower mass of PFAS in treated drinking water compared to the mass of PFAS in raw water at treatment plants A, B, C, D, and E indicates the PFAS removal efficiency at these plants, with the highest removal efficiency at plant E.

Table 7: Annual mass calculations of PFAS4 and PFAS21 in raw water (raw) and drinking water (DW) and PFAS removal.



WTP	$m_{PFAS4,raw}$ [g/year]	$m_{PFAS4,DW}$ [g/year]	$m_{PFAS4,removal}$ [g/year]	$m_{PFAS21,raw}$ [g/year]	$m_{PFAS21,DW}$ [g/year]	$m_{PFAS21,removal}$ [g/year]
A	196.9	161.8	35.1	525.1	480.2	44.9
B	204	186	18	582	558	24
C	50.9	23.4	27.5	85.9	42.8	43.1
D	63.5	37	26.5	140.3	64.7	75.6
E	2.698	0.028	2.669	-	-	-

Table 8 presents the annual mass of PFAS compounds retained in solid phases and transferred to residual streams, revealing significant differences in how these masses are distributed between the two pathways. Since PFAS compounds in treatment plants B and E are retained in the treatment medium as a solid phase and are removed from the system without any mass transfer to residual streams ($m_{PFAS,rest} = 0$), they do not follow the recirculation-release model and are therefore excluded from this analysis. In contrast, WTPs A, C, and D exhibit significant PFAS transfer to residual streams, making them suitable for assessing the dynamics of local recirculation and environmental release. This indicates that the transfer of PFAS compounds to residual streams is influenced by their mass distribution within the treatment system, and not solely by their concentrations in the raw water or their total removal.

Table 8: The annual mass of PFAS retained in solid phases and transferred to residual streams (rest).

WTP	$m_{PFAS4,retained}$ [g/year]	$m_{PFAS4,rest}$ [g/year]	$m_{PFAS21,retained}$ [g/year]	$m_{PFAS21,rest}$ [g/year]
A	≈2[4]	33.1	2.56[5]	42.3
B[6]	18	0	24	0
C	24.8	2.71	38.2	4.95
D	23.4	3.12	70	5.46
E[6]	2.67	0	-	-

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[4] (Norrvatten, 2026)

[5] To estimate the retained mass of PFAS21 ($m_{PFAS21,retained}$), a proportional scaling approach was applied based on the known ratios of PFAS4. Specifically, the ratio of retained mass to removed mass for PFAS4 was determined to be approximately 6% (based on 2 g/year retained to 35.1 g/year removed) for PFAS4 was calculated. This proportion was then multiplied by the total removed mass of PFAS21 (44.9 g/year).

[6] Since the mass of PFAS transferred to residual streams is zero at facility B and E, these water treatment plants have been excluded from the study. However, the reported operating costs for plant B are considered representative and are therefore used as a basis for estimating the operating costs of water treatment plants C and D.

Comparing the three facilities, WTP A retains only a small fraction of the PFAS mass in solid form during treatment, with the majority passing into the residual streams. For PFAS4 and PFAS21, the masses transferred to residual streams are approximately 33.1 *g/year* and 42.3 *g/year*, respectively, while the retained masses are negligible. In contrast, WTPs C and D, which utilize granular activated carbon, retain larger quantities of PFAS in solid form compared to their residual streams. However, the remaining residual streams contribute to an increased likelihood of these compounds being released into the environment. Because the receiving water bodies are open systems where rivers flow into the sea, this untreated portion of the PFAS is carried by the current, placing an environmental burden on marine and downstream ecosystems.

The results in Table 9 and Figure 3 show the effect of hydrological conditions on the fate of PFAS substances (i.e., recycled locally or released into the surrounding environment). The type of receiving system affects the residence time ($\boldsymbol{\tau}$) and therefore significantly influences the recirculation factor ($\boldsymbol{f}$) and the release factor ($\boldsymbol{f_{release}}$) of the substances. For plant A, the closed lake system has a short residence time (2.8 *years*), resulting in a high recirculation factor ($\approx$1). This keeps the PFAS compounds circulating within the lake, resulting in a low release factor (2.2×10^{-5}). In contrast, plants C and D discharge into an open river system that extends to the sea with a long oceanic residence time (4000 *years*). The open nature of this system means that local recycling is virtually non-existent (7.5×10^{-3}), driving the release factor to nearly 100% (0.99). Thus, while plants C and D successfully capture a large mass of PFAS compounds on solids as shown in Figure 4, any untreated fraction entering their residual streams is almost entirely exported to the wider marine environment rather than being retained or recirculated locally. These results are the basis for the cost analysis in the following tables and figures.

Table 9: The receiving system, residence time, f and $f_{release}$.

WTP		**The receiving system**	**Residence time τ [years]**	$\boldsymbol{f}$	$\boldsymbol{f_{release}}$
A		Lake (Closed system)	2.8[7]	0.999	2.22×10^{-5}
C		River discharging to sea (Open system)	4000[8]	7.47×10^{-3}	0.993
D		River discharging to sea (Open system)	4000[8]	7.47×10^{-3}	0.993

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[7] (Sonesten et al., 2013)

[8] (Broecker & Peng, 1982)

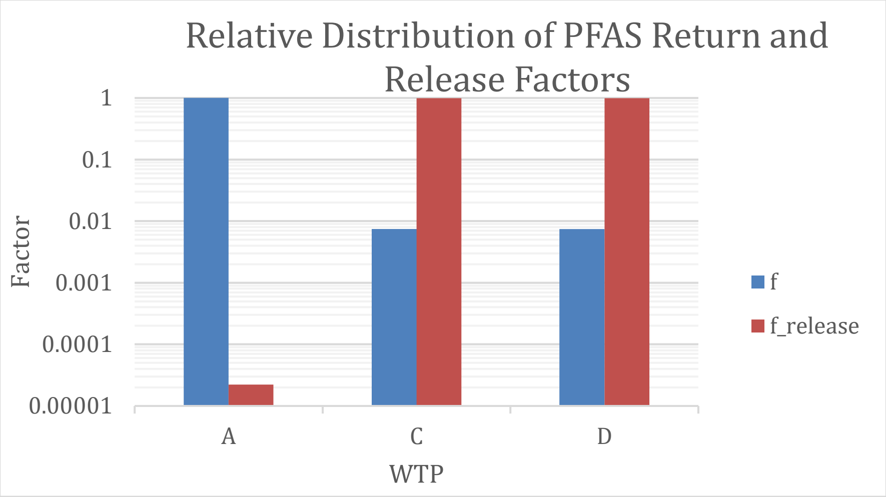

Figure 3: Logarithmic distribution of PFAS return (f) and release (frelease) factors for treatment plant A, C and D.

Table 10 and Figure 4 illustrate the predominant mass pathways of PFAS compounds associated with drinking water treatment residual streams. In WTP A, almost all PFAS compounds that migrate into the residual streams return to the local raw water system. For PFAS4 and PFAS21, the returned masses reach approximately 33.09 *g/year* and 42.3 *g/year*, respectively, while the released masses are negligible. This confirms that closed systems primarily generate internal recirculation burdens (i.e., returned PFAS pathways are predominant). In contrast, PFAS return masses are very low in WTPs C and D, while released masses are significantly higher. For PFAS4, the returned masses are 2.02×10^{-2} *g/year* at plant C and 2.33×10^{-2} *g/year* at plant D, while the released masses are 2.7 *g/year* and 3.1 *g/year* at plants C and D, respectively. These results demonstrate that open systems primarily contribute to downstream diffusion of PFAS compounds into the environment rather than local recycling (i.e., released PFAS pathways are dominant), leading to significant environmental burden costs.

Table 10: Dominant PFAS mass pathways associated with drinking water treatment residual streams for PFAS4 and PFAS21.

WTP	$m_{PFAS4,returned}$ **[g/year]**	$m_{PFAS4,released}$ **[g/year]**	$m_{PFAS21,returned}$ **[g/year]**	$m_{PFAS21,released}$ **[g/year]**
A	33.09	7.36×10^{-4}	42.3	9.40×10^{-4}
C	2.02×10^{-2}	2.69	3.70×10^{-2}	4.91
D	2.33×10^{-2}	3.10	4.08×10^{-2}	5.42

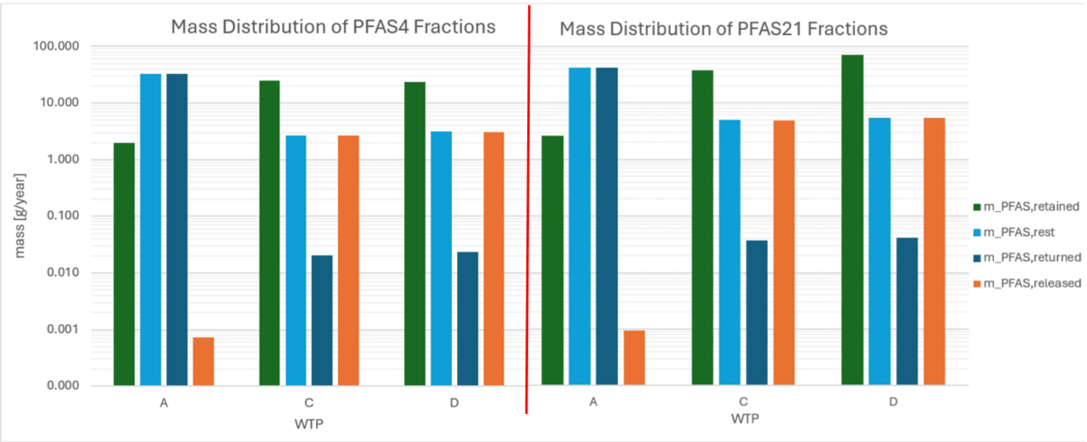

Figure 4: Annual mass distribution and pathways of PFAS4 (left) and PFAS21 (right) presented on logarithmic scale [g/year].

Table 11 presents calculations of the masses of PFAS compounds removed from WTPs C and D into the environment and reaching the receiving system. These masses are used to calculate the specific environmental treatment costs, and thus the environmental external costs.

Table 11: The calculations of abatement PFAS masses for PFAS4 and PFAS21.

WTP	**$m_{PFAS4,abatement}$ [g/year]**	**$m_{PFAS21,abatement}$ [g/year]**
C	2.44	4.46
D	2.81	4.91

Tables 12 and 13, and Figures 5 and 6, provide estimates of local recirculation costs, annual total environmental costs, and total annual costs associated with PFAS compounds for the treatment plants studied. Figure 5 presents a comprehensive economic summary for PFAS4 and PFAS21, simultaneously showing both the specific costs (SEK million/kg PFAS) and the total annual costs (SEK million/year). The light-colored bars represent the specific costs per kilogram of PFAS, while the dark bars represent the total annual costs associated with each burden category. The annual costs resulting from PFAS4 and PFAS21 were relatively similar across all studied systems, as shown in Figure 6. This is because the applied framework was primarily governed by the hydrological pathway's behavior and annual treatment conditions, rather than by differences between PFAS groups. Consequently, the burden allocation effect was dependent on the hydrological paths, not the specific PFAS classification itself

The results show that the main burden pathway is highly dependent on the hydrological characteristics of the receiving system. In the closed hydrological system represented by plant A, most of the PFAS compounds transferred to residual streams were continuously recirculated back into the raw water source, resulting in substantial local recirculation costs and minimal environmental burden costs. For plant A, the annual local return cost was approximately 1.57 *MSEK* for both PFAS4 and PFAS21, while the corresponding specific local costs exceeded 37 and 47 *MSEK/kg* of PFAS for PFAS4 and PFAS21, respectively. Due to the closed-loop design and the small mass of PFAS released, the environmental burden remained minimal for plant A. Furthermore, given the specific hydrological conditions and system design, it was deemed technically and conceptually inappropriate to apply the post-treatment concepts used in plants C and D, such as mobile downstream treatment systems. Therefore, plant A was excluded from the calculations of abatement, external environmental impacts, and environmental burden, which explains the absence of those bars for Plant A in Appendices 1 and 2.

In contrast, the open hydrological systems represented by plants C and D showed the opposite pattern. Due to the significantly high PFAS released, environmental burden costs became the dominant cost component, while local recirculation costs remained relatively low. Total annual environmental costs exceeded 2.3 *MSEK/year* for plant C and 1.52 *MSEK/year* for plant D, while local recirculation costs remained below 25,000 *SEK/year* for both plants. This disparity in burdens between closed and open systems is clearly illustrated in Appendix 2. The results also show that the release of relatively small quantities of PFAS annually can generate substantial environmental management costs when advanced treatment technologies are required. This is evident in the specific environmental costs, which exceeded 851 *MSEK/kg* of PFAS at plant C, as shown in Appendix 3. The figure also highlights that the dominant specific burden at plant A was local recirculation, while environmental burden costs dominated at plants C and D.

These high specific environmental costs primarily reflect a combination of relatively low annual PFAS emissions, high technical treatment requirements, and the persistence of PFAS pollution once it has spread into wider ecosystems. Therefore, the findings suggest that managing PFAS after it has spread into the environment can generate extremely large long-term remediation burdens. Appendix 1 provides a more detailed breakdown of the environmental burden costs, including the relative contributions of downstream abatement costs and the external environmental costs of plants C and D. The figure in the Appendix shows that downstream abatement costs constituted the dominant component of the environmental cost within the applied framework.

Table 12: local recirculation costs, total environmental cost and total costs for PFAS4

WTP	$\mathrm{Cost}_{\mathrm{local,return}}$ [SEK/year]	$\mathrm{Cost}_{\mathrm{local,specific}}$ [SEK/kg]	$\mathrm{Cost}_{\mathrm{env,ext}}$ [SEK/year]	$\mathrm{Cost}_{abatement}$ [SEK/year]	$\mathrm{Cost}_{\mathrm{env,tot.}}$ [SEK/year]	$\mathrm{Cost}_{\mathrm{env,specific}}$ [SEK/kg]	$\mathrm{Cost}_{\mathrm{total}}$ [SEK/year]
A	1,565,965	47,312,230	Very low	-	-	-	1,565,965
C	24,844	-	288,667	2,058,000	2,286,667	851,397,317	2,311,511
D	24,849	-	152,444	1,372,000	1,524,444	492,282,295	1,551,294

Table 13: local recirculation costs, environmental burden costs and total costs for PFAS21.

WTP	$\mathrm{Cost}_{\mathrm{local,return}}$ [SEK/year]	$\mathrm{Cost}_{\mathrm{local,specific}}$ [SEK/kg]	$\mathrm{Cost}_{\mathrm{env,ext}}$ [SEK/year]	$\mathrm{Cost}_{abatement}$ [SEK/year]	$\mathrm{Cost}_{\mathrm{env,tot.}}$ [SEK/year]	$\mathrm{Cost}_{\mathrm{env,specific}}$ [SEK/kg]	$\mathrm{Cost}_{\mathrm{total}}$ [SEK/year]
A	1,565,965	37,088,601	Very low	-	-	-	1,565,965
C	24,844	-	288,667	2,058,000	2,286,667	465,430,533	2,311,511
D	24,849	-	152,444	1,372,000	1,524,444	281,304,169	1,551,294

Figure 5: Economic summary of PFAS4- (left) and PFAS21 (right)-related costs for facilities A, C and D. Left axis shows specific costs in MSEK/kg (light green and light blue bars). Right axis shows annual total costs in MSEK/year (brown, red, dark blue, and dark green bars).

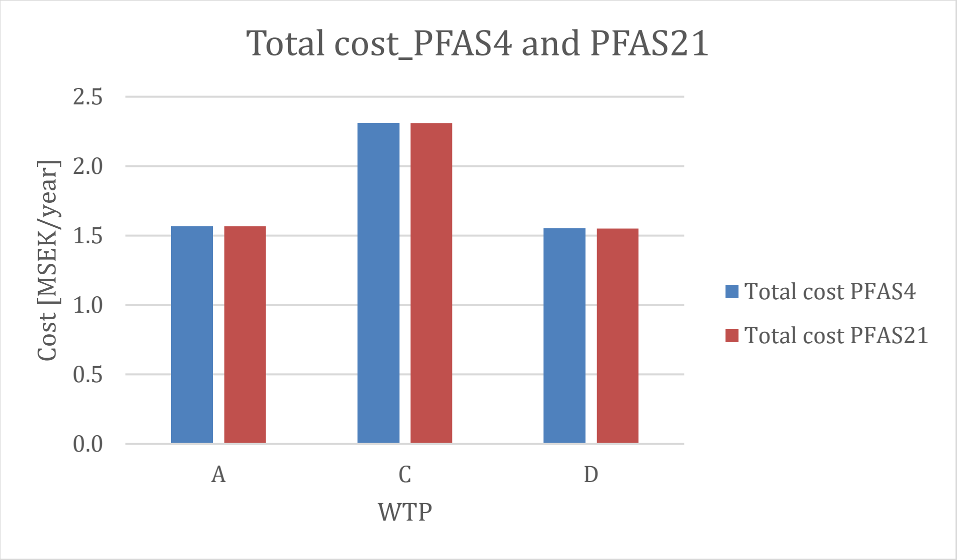

Figure 6: Annual total costs for PFAS4 and PFAS21 across water treatment plants A, C and D, expressed in millions of Swedish kronor per year [MSEK/year].

5 Sensitivity analysis

The sensitivity analysis was performed to evaluate how variations in hydrological residence time, time horizon, recipient treatment efficiency, and number of recipient GAC filter units influence PFAS burden pathways and the resulting economic burden distribution.

5.1 Hydrological residence time

The results shown in Tables 14 and 15 and Figure 7a demonstrate that hydrological residence time significantly impacts the fate of PFAS compounds within the system. Decreasing residence time increases local recirculation and reduces downstream release. Conversely, increasing residence time reduces the proportion of PFAS compounds returned to the raw water source, thus decreasing the local recirculation cost.
The environmental external cost ($\mathrm{Cost}_{\mathrm{env,\ ext}}$) remained constant with changes in residence time and time horizon because it depends on assumptions about the treatment

of the receiving water and its removal efficiency, rather than on the residence time and time horizon themselves.

Changes in specific costs were primarily driven by changes in PFAS mass as Figure 7b shows. At plant A, the local specific cost increased slightly with increasing residence time due to a decrease in the return mass of PFAS compounds. In contrast, at plants C and D, the environmental specific cost decreased with increasing residence time because the increased mass of released PFAS compounds resulted in the annual cost of receiving water treatment being distributed over a larger mass of PFAS compounds.

Table 14: The effect of different residence time scenarios (low, baseline, and high) on costs for PFAS4.

WTP	**Residence time (τ) [year]**	***f***	**$m_{PFAS,returned}$ [g/year]**	**$Cost_{local,return}$ [SEK/year]**	**$Cost_{local,specific}$ [SEK/kg]**	**$f_{release}$**	**$m_{PFAS,released}$ [g/year]**	**$Cost_{env,specific}$ [SEK/kg]**
A	1	≈1	33.10	1,566,000	47,311,178	9.36×10^{-14}	3.09×10^{-12}	-
A	2.8	0.999	33.09	1,565,965	47,312,230	2.22×10^{-5}	7.36×10^{-4}	-
A	5	0.998	33.02	1,562,118	47,428,742	2.48×10^{-3}	8.20×10^{-2}	-
C	1000	2.96×10^{-2}	7.99×10^{-2}	98,269	-	0.970	2.63	870,770,892
C	4000	7.47×10^{-3}	2.02×10^{-2}	24,844	-	0.993	2.69	851,397,317
C	10,000	2.99×10^{-3}	8.11×10^{-3}	9,960	-	0.997	2.70	847,574,637
D	1000	2.96×10^{-2}	9.22×10^{-2}	106,199	-	0.970	3.03	503,484,195
D	4000	7.47×10^{-3}	2.33×10^{-2}	26,849	-	0.993	3.10	492,282,295
D	10,000	2.99×10^{-3}	9.35×10^{-3}	10,764	-	0.997	3.11	490,072,001

Table 15: The effect of different residence time scenarios (low, baseline, and high) on costs for PFAS21.

WTP	**Residence time τ [year]**	***f***	**$m_{PFAS,returned}$ [g/year]**	**$Cost_{local,return}$ [SEK/year]**	**$Cost_{local,specific}$ [SEK/kg]**	**$f_{release}$**	**$m_{PFAS,released}$ [g/year]**	**$Cost_{env,specific}$ [SEK/kg]**
A	1	≈1	42.3	1,566,000	37,007,778	9.36×10^{-14}	3.96×10^{-12}	-
A	2.8	0.999	42.3	1,565,965	37,088,600	$2,22\times10^{-5}$	0.104	-
A	5	0.998	42.2	1,562,118	37,099,739	2.48×10^{-3}	0.105	-
C	1000	2.96×10^{-2}	0.146	98,269	-	0.970	4.80	476,021,421
C	4000	7.47×10^{-3}	3.70×10^{-2}	24,844	-	0.993	4.91	465,430,533
C	10,000	2.99×10^{-3}	1.48×10^{-2}	9,960	-	0.997	4.94	463,340,801
D	1000	2.96×10^{-2}	0.161	106,199	-	0.970	5.30	287,705,254
D	4000	7.47×10^{-3}	4.08×10^{-2}	26,849	-	0.993	5.42	281,304,169
D	10,000	2.99×10^{-3}	1.64×10^{-2}	10,764	-	0.997	5.44	280,041,144

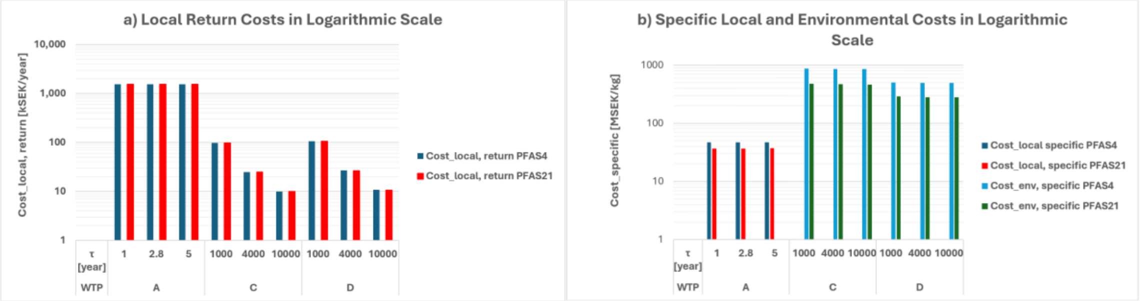

Figure 7: Sensitivity analysis of local recirculation and environmental costs to variations in residence time (τ) for PFAS4 and PFAS21 in logarithmic scale, showing (a) annual local return costs [kSEK/year] (left), and (b) specific local and environmental costs [MSEK/kg] (right) for water treatment plants A, C and D. (τ = 2.8 years for plant A; τ = 4000 years for plants C and D), while the adjacent bars show the effect of a shorted and an extended residence time, respectively.

5.2 Time horizon

The time horizon determines the amount of PFAS compounds that remain in the local water system versus the amount that flows downstream. Over a longer period, the mass of PFAS compounds recycled locally increased, which raises local recirculation costs but reduces the downstream environmental impact, as shown in Table 16 and 17 and Figure 8a. Changes in specific costs were primarily driven by changes in PFAS mass as Figure 8b shows. At plant A, the local specific cost decreased with increasing time horizon due to an increase in the return mass of PFAS compounds. In contrast, at plants C and D, the environmental specific cost increased with increasing time horizon because the decreased mass of released PFAS compounds resulted in the annual cost of receiving water treatment being distributed over a lower mass of PFAS compounds.

Table 16: The effect of different time horizon scenarios (low, baseline, and high) on costs for PFAS4.

WTP	Time horizon t [year]	f	$m_{PFAS,returned}$ [g/year]	$Cost_{local,return}$ [SEK/year]	$Cost_{local,specific}$ [SEK/kg]	$f_{release}$	$m_{PFAS,released}$ [g/year]	$Cost_{env,specific}$ [SEK/kg]
A	15	0.995	32.9	1,558,617	47,535,276	4.71×10^{-3}	0.156	-
A	30	0.999	33.09	1,565,965	47,312,230	2.22×10^{-5}	7.36×10^{-4}	-
A	45	≈1	33.1	1,566,000	47,311,183	1.05×10^{-7}	3.47×10^{-6}	-
C	15	3.74×10^{-3}	1.01×10^{-2}	12,445	-	0.996	2.70	848,210,556
C	30	7.47×10^{-3}	2.02×10^{-2}	24,844	-	0.993	2.69	851,397,317
C	45	1.12×10^{-2}	3.03×10^{-2}	37,197	-	0.989	2.68	854,596,051
D	15	3.74×10^{-3}	1.17×10^{-2}	13,450	-	0.996	3.11	490,439,693
D	30	7.47×10^{-3}	2.33×10^{-2}	26,849	-	0.993	3.10	492,282,295
D	45	1.12×10^{-2}	3.49×10^{-2}	40,198	-	0.989	3.09	494,131,819

Table 17: The effect of different time horizon scenarios (low, baseline, and high) on costs for PFAS21.

WTP	Time horizon t [year]	f	$m_{PFAS,returned}$ [g/year]	$Cost_{local,return}$ [SEK/year]	$Cost_{local,specific}$ [SEK/kg]	$f_{release}$	$m_{PFAS,released}$ [g/year]	$Cost_{env,specific}$ [SEK/kg]
A	15	0.995	42.7	1,558,617	37,183,073	4.71×10^{-3}	0.199	-
A	30	0.999	42.3	1,565,965	37,008,600	2.22×10^{-5}	0.104	-
A	45	≈1	42.9	1,566,000	37,007,782	1.05×10^{-7}	4.43×10^{-6}	-
C	15	3.74×10^{-3}	0.019	12,445	-	0.996	4.93	463,688,437
C	30	7.47×10^{-3}	3.70×10^{-2}	24,844	-	0.993	4.91	465,430,533
C	45	1.12×10^{-2}	0.055	37,197	-	0.989	4.89	467,179,175
D	15	3.74×10^{-3}	0.020	13,450	-	0.996	5.44	280,251,253
D	30	7.47×10^{-3}	4.08×10^{-2}	26,849	-	0.993	5.42	281,304,169
D	45	1.12×10^{-2}	0.061	40,198	-	0.989	5.40	282,361,040

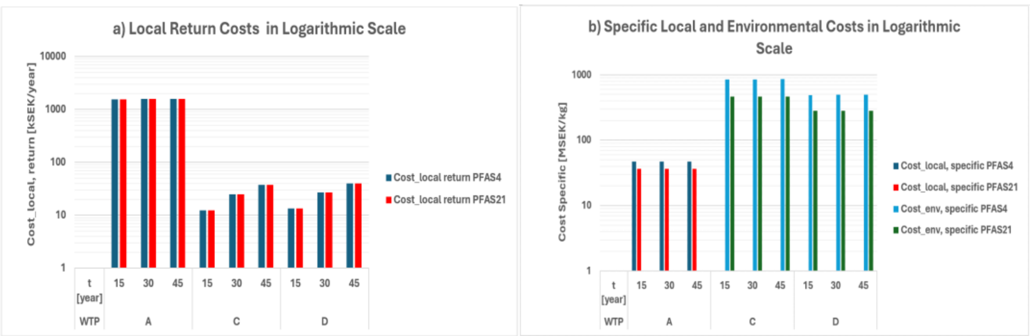

Figure 8: Sensitivity analysis of local recirculation and environmental costs to variations in time horizon (t) for PFAS4 and PFAS21 in logarithmic scale, showing (a) annual local return costs [kSEK/year] (left), and (b) specific local and environmental costs [MSEK/kg] (right) for water treatment plants A, C and D. The middle bar represents the baseline scenario (t= 30 years), while the adjacent bars show the effect of a shorted and extended time horizon, respectively.

5.3 Recipient removal efficiency

The sensitivity analysis further showed that recipient treatment efficiency substantially influenced the resulting environmental burden costs, as shown in Table 18 and 19 and Figure 9. Higher treatment efficiencies reduced the remaining untreated PFAS mass and therefore lowered the estimated environmental externalities costs. However, higher downstream removal efficiencies simultaneously required larger treatment efforts and operational costs, illustrating the economic challenge associated with downstream remediation of diluted PFAS contamination. The analysis includes only open-system configurations (plants C and D), as plant A operates as a completely closed loop with no direct discharge to an external recipient

Table 18: Lower, baseline, and higher downstream recipient removal efficiency [%] for PFAS4.

WTP	$\eta_{recipient}$ [%]	$m_{PFAS,abatement}$ [g/year]	$Cost_{env,ext}$ [SEK/year]	$Cost_{env,specific}$ [SEK/kg]
C	70%	1.89	882,000	1,094,653,693
C	90%	2.44	228,667	851,397,317
C	99%	2.68	20,788	773,997,561
D	70%	2.18	588,000	632,934,379
D	90%	2.81	152,444	492,282,295
D	99%	3.09	13,859	447,529,359

Table 19: Lower, baseline, and higher downstream recipient removal efficiency [%] for PFAS21.

WTP	$\eta_{recipient}$ [%]	$m_{PFAS,abatement}$ [g/year]	$Cost_{env,ext}$ [SEK/year]	$Cost_{env,specific}$ [SEK/kg]
C	70%	3.47	882,000	598,410,686
C	90%	4.46	228,667	465,430,533
C	99%	4.90	20,788	423,118,667
D	70%	3.82	588,000	361,676,788
D	90%	4.91	152,444	281,304,169
D	99%	5.41	13,859	255,731,062

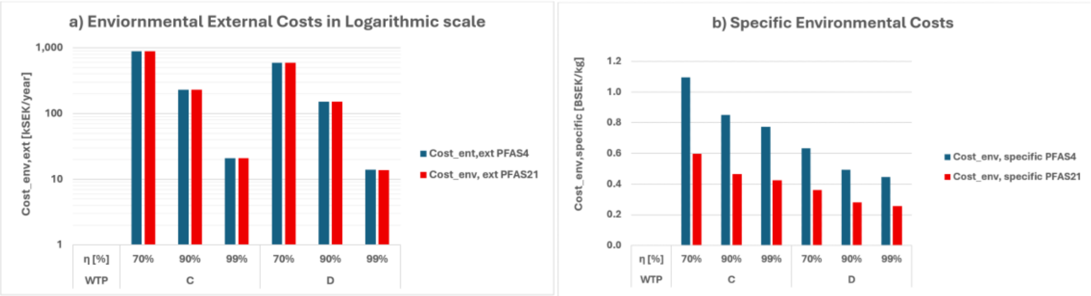

Figure 9: Sensitivity analysis of environmental costs to variations in recipient removal efficiency ($\eta_{recipient}$) for PFAS4 and PFAS21, showing (a) annual environmental external costs in logarithmic scale [kSEK/year] (left), and (b) specific environmental costs [BSEK/kg] (right) for water treatment plants C and D. The middle bar represents the baseline scenario ($\eta_{recipient}$ = 90%), while the adjacent bars show the effect of lower and higher removal efficiency, respectively.

5.4 Number of recipient GAC filter units

Sensitivity analysis showed that environmental costs are significantly affected by the number of filters assumed in the receiving water treatment system, se Tables 20 and 21 and Figure 10. Increasing the number of filters leads to higher estimated environmental costs, assuming a greater treatment capacity is needed to manage the PFAS released to the receiving water. Conversely, using fewer filters indicates lower treatment requirements and, consequently, lower environmental costs.

Table 20: The effect of changing the number of filters on environmental burden costs for PFAS4.

WTP	N	Cost$_{abatement}$ [SEK/year]	Cost$_{env,ext}$ [SEK/year]	Cost$_{env,\ tot.}$ [SEK/year]	cost$_{specific}$ [SEK/kg]
C	4	1,372,000	152,444	1,524,444	567,598,211
C	6	2,058,000	228,667	2,286,667	851,397,317
C	8	2,744,000	304,889	3,048,889	1,135,196,423
D	2	686,000	76,222	762,222	246,141,147
D	4	1,372,000	152,444	1,524,444	492,282,295
D	6	2,058,000	228,667	2,286,667	738,423,442

Table 21: The effect of changing the number of filters on environmental burden costs for PFAS21.

WTP	N	Cost$_{abatement}$ [SEK/year]	Cost$_{env,ext}$ [SEK/year]	Cost$_{env,\ tot.}$ [SEK/year]	cost$_{specific}$ [SEK/kg]
C	4	1,372,000	152,444	1,524,444	310,287,022
C	6	2,058,000	228,667	2,286,667	465,430,533
C	8	2,744,000	304,889	3,048,889	620,574,045
D	2	686,000	76,222	762,222	140,652,084
D	4	1,372,000	152,444	1,524,444	281,304,169
D	6	2,058,000	228,667	2,286,667	421,956,253

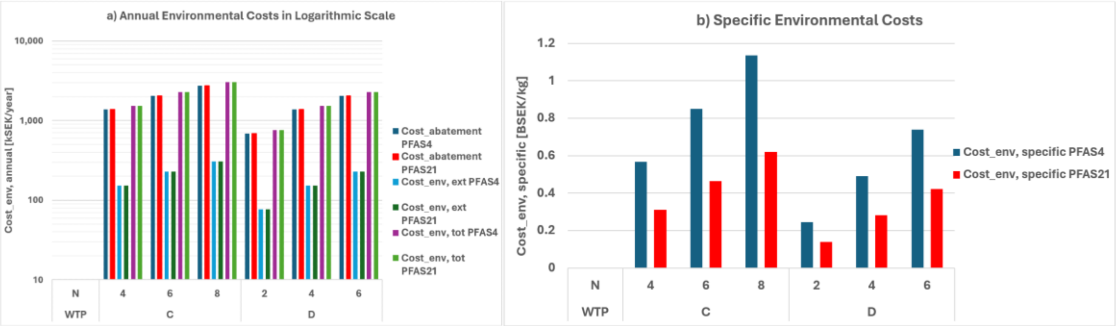

Figure 10: Sensitivity analysis of environmental costs to variations in the number of filters for PFAS4 and PFAS21, showing (a) annual environmental costs in logarithmic scale [kSEK/year] (left), and (b) specific environmental costs [BSEK/kg] (right) for water treatment plants C and D. The middle bar represents the baseline scenario (N = 6 for plant C, N=4 for plant D), while the adjacent bars show the effect of fewer and more filters, respectively.

6 Discussion

This study investigates how PFAS transferred to drinking water treatment residual streams may either be recirculated within local water systems or be released into surrounding environments, and how these pathways generate different local recirculation costs and environmental burden costs. The results demonstrate that hydrological conditions strongly influence the fate of PFAS compounds and therefore determine whether the dominant burden remains local or becomes externalized to downstream environments.

One of the most striking findings is the significant contrast between closed and open hydrological systems. In the closed lake system represented by WTP A, almost all PFAS transferred to residual streams are returned to the local raw water system, while only negligible masses are released into the surrounding environment. This results in higher local recirculation costs but minimal environmental burden costs. In contrast, WTPs C and D, which are discharged into open river systems connected to the sea, exhibit the opposite pattern. In these systems, virtually all untreated PFAS that transferred into residual streams are discharged downstream, resulting in very low local recirculation costs but significantly larger environmental burden costs. These findings demonstrate that receiving system characteristics are more important than treatment process alone in determining the long-term burden associated with PFAS-containing residual streams.

The results further show that even relatively small annual PFAS releases may generate substantial downstream treatment burdens. Although the annual released PFAS masses are only a few grams per year, the estimated environmental burden costs reach several million SEK annually. This occurs because PFAS removal from diluted environmental systems requires advanced treatment technologies with high operational costs. The very high specific environmental treatment costs obtained in this study therefore reflect not only the technical complexity of PFAS removal, but also the persistent and diffuse nature of PFAS contamination once released into wider environmental systems. Although the estimated annual environmental total costs in this study (~1–3 *MSEK/year* per treatment plant) are considerably smaller than the national non-health cost estimates reported by (Goldenman et al., 2019) (9.8 *million* to 2.5 *billion SEK* annually[9]), the results still illustrate the same underlying economic principle. Even relatively small and localized PFAS releases may generate substantial long-term management burdens due to the persistence, mobility, and treatment complexity of PFAS compounds. The present study therefore provides a micro-scale conceptual illustration of how diffuse PFAS emissions from individual drinking water systems may contribute to broader cumulative societal costs over time.

The persistence of PFAS compounds also has important implications when interpreting the results. Because PFAS substances are highly resistant to environmental degradation, continuous annual releases may lead to long-term accumulation within downstream aquatic systems. The present framework is static and evaluates annual burden estimate

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[9] Since the Nordic Council of Ministers report (Goldenman et al., 2019) estimated all costs in Euros (EUR), a conversion to Swedish Kronor (SEK) was made to facilitate a direct comparison with the empirical findings of this study. The conversion is based on the 2026 exchange rate of approximately 1 EUR =10.87SEK (Boliga Finans ApS, n.d.).

only. However, repeated yearly emissions may progressively increase environmental concentrations and future remediation requirements over time. Consequently, the actual long-term environmental burden associated with PFAS releases may be significantly larger than the annual costs estimated in this study. The findings also support the principle that preventive PFAS management is likely more economically efficient than downstream remediation. According to environmental-economic theory, pollutants should ideally be controlled where marginal abatement costs are lowest. Once PFAS compounds are released into larger environmental systems, treatment becomes increasingly difficult and costly due to dilution, dispersion, and persistence. The conceptual environmental burden framework developed in this study therefore illustrates how untreated PFAS transferred to residual streams may shift treatment responsibilities and future remediation burdens from local water utilities to downstream ecosystems and other societal actors. Another important observation is that the resulting annual costs became nearly identical for PFAS4 and PFAS21 within the applied framework, despite differences in PFAS mass. This indicates that the developed conceptual model is primarily governed by hydrological pathway conditions and treatment assumptions rather than by differences between PFAS groups themselves. The results therefore suggest that the hydrological fate of PFAS compounds may be more critical for long-term burden generation than the specific PFAS classification used in the analysis.

Several limitations affect the interpretation of the results. The developed framework represents a conceptual modelling approach and does not quantify actual ecological damage or human health effects caused by PFAS exposure. The downstream treatment costs were estimated by supplier quotations based on conceptual treatment assumptions rather than implemented full-scale systems. In addition, several assumptions regarding treatment efficiency, residence time, and treatment system sizing were required. The model is static and evaluates annual PFAS burden estimates only. It therefore does not simulate how PFAS concentrations, accumulation processes, environmental burdens, or treatment requirements may change over longer time periods. This may be particularly important for closed hydrological systems such as plant A, where continuous recirculation of PFAS-containing residual streams could potentially contribute to gradual PFAS accumulation within the local water system over time. Nevertheless, the framework provides a conceptual basis for understanding how PFAS transferred to residual streams may generate different burden pathways depending on hydrological conditions and treatment system configurations.

7 Conclusion

Reducing PFAS transfer to residual streams minimizes future environmental burdens and avoids the expensive remediation costs associated with widespread contamination, thereby ensuring long-term regulatory compliance and significant economic efficiency.

This study demonstrated that PFAS transferred to residual streams may create both local recirculation costs and downstream environmental burdens depending on the hydrological conditions of the receiving system. Closed hydrological systems with short residence times primarily recirculated PFAS back into the local water system, resulting in repeated local treatment costs. In contrast, open river systems transported PFAS downstream into surrounding aquatic environments, thereby externalizing the burden beyond the local water system.

The results further showed that even relatively small PFAS releases may generate substantial downstream treatment burdens due to the high complexity and cost associated with removing diluted PFAS contamination from environmental systems. These findings indicate that reducing PFAS concentrations before environmental release is likely more economically efficient than downstream remediation after environmental dispersion occurs. Overall, the study highlights the importance of considering not only treatment efficiency within drinking water treatment plants, but also the long-term hydrological fate of PFAS-containing residual streams when evaluating sustainable PFAS management strategies.

8 Future studies and improvements

The current study focused primarily on hydrological pathways of PFAS substances and their associated economic burdens, including local recirculation costs and downstream environmental burdens. However, several aspects could be developed further in future research to expand this conceptual framework.

A key improvement is the integration of human exposure pathways into the modeling framework. This study did not explicitly assess how recirculated PFAS compounds contribute to consumer exposure through drinking water systems, as the primary objective was economic analysis. Future research could investigate the proportion of PFAS from residual streams that potentially reach consumers. This would help clarify how these compounds contribute to long-term health impacts, which is crucial in areas with high PFAS concentrations in drinking water sources.

Another crucial development is the study of long-term accumulation dynamics within closed hydrological systems. Because the current framework is static and relies on annual steady-state estimates, the effects of temporal accumulation effects have not been quantified. Due to the highly persistent nature of PFAS compounds, continuous recirculation and repeated releases over long periods will gradually increase environmental concentrations, especially within closed lake environments. Therefore, future studies could refine this framework by incorporation site-specific conditions, including sediment interactions, bioaccumulation processes, and seasonal hydrological variations. While the fate of PFAS compounds in this study was largely governed by hydrological pathways, differences in specific costs associated with the two different PFAS groups, PFAS4 and PFAS21, were observed. Therefore, further research could be conducted to expand the scope of the study and investigate other PFAS groups and individual PFAS compounds.

Furthermore, the present study focused exclusively on PFAS transferred to residual streams from drinking water treatment. Future studies could expand the scope to include additional environmental components and pathways, such as sewage systems, soils, and sediments, providing a more comprehensive assessment of PFAS transport and associated burdens in the wider environment. Finally, the environmental burden cost estimates presented in this paper are based on conceptual recipient treatment scenarios, which were determined based on price quotes from suppliers. In future research, the efficiency of the developed framework can be improved by applying real-world treatment facilities, data, and efficiencies.

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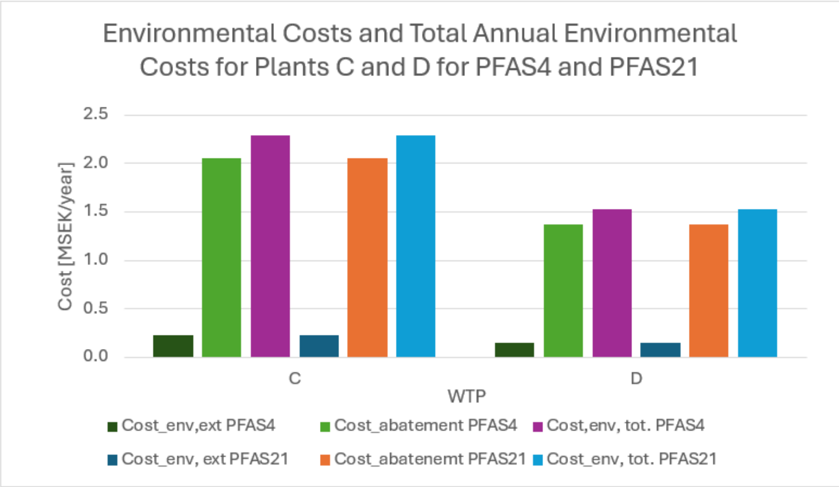
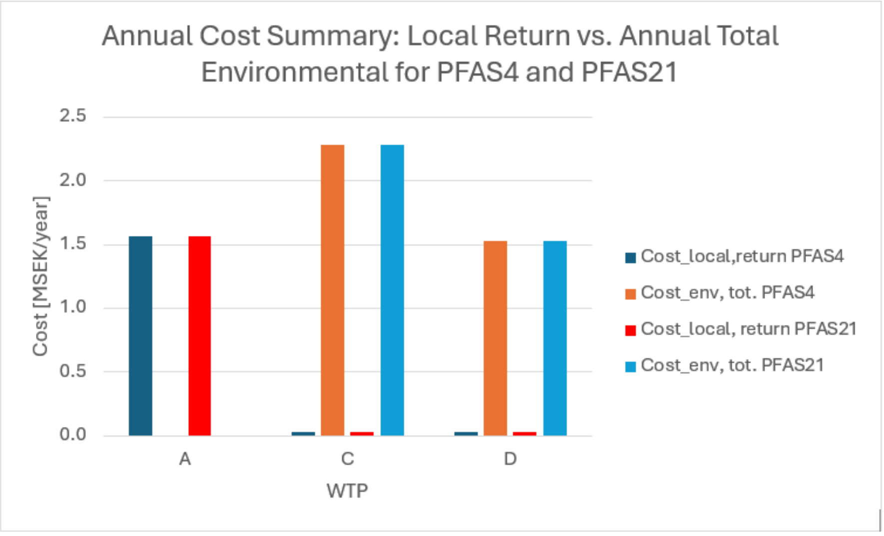
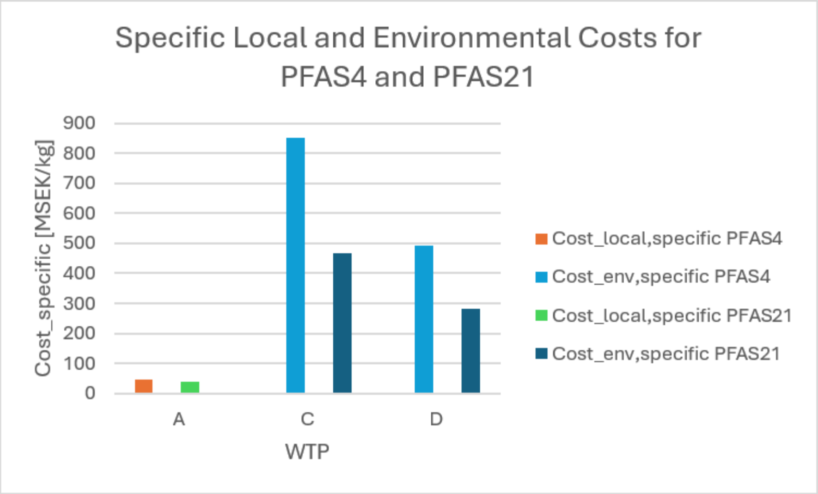
10 Appendices



Appendix 1: Annual environmental -related costs for water treatment plants C and D in millions of Swedish kronor per year [MSEK/year]. The blue and purple staples represent the total annual environmental cost (Cost$_{env, tot.}$), calculated as the sum of external environmental costs (Cost$_{env,ext}$) and abatement costs (Cost$_{abatment}$).



Appendix 2: Comparison of annual local return costs (Cost$_{local, return}$) and annual total environmental cost [(Cost$_{env,tot.}$), which is sum of Cost$_{abatement}$ and Cost$_{env,ext}$] for PFAS4 and PFAS21 across water treatment plants A, C and D, expressed in millions of Swedish kronor per year [MSEK/year]



Appendix 3: Comparison of dominant specific costs (local and environmental) [MSEK/kg] for PFAS4 and PFAS21 between the treatment plants.