



DEGREE PROJECT IN TECHNOLOGY,
SECOND CYCLE, 30 CREDITS
STOCKHOLM, SWEDEN 2023

The Potential of Dissolved Air Flotation for PFAS Reduction in Norrvatten's Future Waterworks

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Abstract

Norrvatten is one of the largest drinking water producers in Sweden. With the need to increase their production capacity and new regulations, they are planning to improve their treatment process. One aspect that requires extensive investigation, is the treatment approach to reach the new limit from the Swedish Food Agency of four PFAS chemicals. PFAS is a group of chemicals that have been humely produced since the 1950s, which are persistent and hazardous to human health. Their environmental stability causes them to accumulate in nature and ends up in foods and drinking water. To fulfill the new regulations, Norrvatten has to decrease the PFAS4 concentration in their drinking water from approximately 5.7 ng/L to 4 ng/L. The removal is planned to be achieved through either activated carbon filters or a combination where dissolved air flotation is included. Activated carbon has a proven separation and destruction possibility for PFAS4 but recently, DAF has appeared to be a promising alternative. However, there is limited research on its efficiency, which establishes a need for investigations on the potential of DAF for PFAS removal. To navigate Norrvatten through the decision between exclusively applying GAC filters or implementing a combination with DAF, this thesis intends to provide a basis of its efficiency and favorability.

Therefore, this thesis investigated the removal efficiency that was obtained in one existing DAF basin in Norrvattens waterworks. By measuring the PFAS4 concentration in the inlet, outlet, and sludge flow, a removal efficiency of 19-38% was acquired for different flow levels. In addition, the sludge concentration was measured to approximately 200-700 ng/L in the water phase. Presence of sources of errors could be identified in the measurements as the mass balance did not add up despite the addition of the measurement uncertainty in the analysis. Also, two samples were analyzed twice as the results differed from the expected results, which provided significant variations. Although some sources of error were identified, a full explanation for the differing results could not be disclosed. However, through some consistency, it was possible to conclude that the removal efficiency and sludge concentration could provide an accurate interval to illustrate the reality. In addition, the results indicate that DAF cannot be applied alone to reach the concentration goal, but has to be combined with additional GAC filters to reach the target.

DAF provides a separation of PFAS into a sludge stream that requires treatment to remove the chemicals from the environmental cycle. To investigate how the sludge could be managed, several treatment methods were identified. The comparison included GAC, AIX, membranes, oxidation processes, sonolysis, incineration, landfill, soil cleaning, DAF, foam fractionation, treatment at a wastewater facility, and reuse for soil improvement. The methods were compared based on proven robustness, cost, and applicability. It was concluded that a viable alternative was to transfer the sludge to a second stage of DAF, incinerate the concentrated sludge, and treat the permeate with AIX.

From the sludge management proposal, the implementation of DAF was analyzed based on maximum PFAS destruction, economics, carbon emissions, and energy requirements. It was found that the energy requirement was larger for the process including DAF but the carbon emissions are smaller. However, the delimitations on the carbon emissions may have been in favor of the DAF process as the GAC reactivation emissions are included but emissions due to incineration or energy production are excluded. The PFAS destruction was approximately 1-2% larger for the process that excluded DAF as an additional stream containing PFAS would be released into the environment after AIX, which is not present when only GAC is implemented. The economic outcome depended on the concentration limit, where implementation of DAF was more expensive for a treatment target of 4 ng/L in the drinking water and more economical for 3 ng/L. Lastly, it was found that the volume of the concentrated sludge that would be sent for incineration had a large impact on the operational cost. Therefore, if DAF is implemented optimizations are of relevance to minimize the volume of that stream.

Keywords

PFAS4, drinking water production, dissolved air flotation, granular activated carbon filter, sludge management

Sammanfattning

Norrvatten är en av Sveriges största dricksvattenproducenter. Med ett ökande behov av en större produktionskapacitet och nya regleringar, planerar dem att förbättra sin reningsprocess. En aspekt som kräver omfattande undersökning är val av reningsmetod för att nå den nya gränsen från Livsmedelsverket för fyra PFAS kemikalier. PFAS är en grupp kemikalier som är mänskligt producerade sedan 1950-talet, vilka är persistenta och farliga för människors hälsa. Deras stabilitet i miljö leder till att de ackumulerar i naturen och hamnar i mat och dricksvatten. För att uppnå de nya regleringarna måste Norrvatten minska sin PFAS4 koncentration från cirka 5.7 ng/L till 4 ng/L. PFAS-separationen är planerad att genomföras via antingen aktivt kol eller en kombination där flotation även implementeras. Aktivt kol har en bevisad möjlighet att separera och destruera PFAS4, men nyligen har indikationer på en potential hos flotation uppenbarats. Dock är forskningen kring dess effektivitet begränsad vilket medför ett behov av att utvärdera dess användbarhet. Denna rapport syftar till att förse Norrvatten med ett underlag på effektiviteten och gynnsamheten med flotation inför beslutet att implementera flotation.

Av den anledningen undersöktes separationseffektiviteten som uppnås i en befintlig flotationsbassäng i Norrvattens vattenverk. Genom att mäta PFAS4 koncentrationen i inflödet, utflödet och slammet kunde en bortagningseffektivitet på 19-38% bestämmas för olika belastningar. Slamkoncentrationen mättes till ungefär 200-700 ng/L. Felkällor kunde identifieras i mätningarna eftersom massbalansen inte stämde överens trots att mätosäkerheten från analysen adderades. Dessutom gjordes en omanalys av två prover som skilde sig från det förväntade resultatet, vilket gav avsevärda skillnader i resultatet. Trots att några felkällor kunde identifieras kunde inte en fullständig förklaring till avvikelserna fastställas. Trots detta kunde slutsatser dras att den uppmätta separationseffektiviteten och slamkoncentrationen kunde förse ett korrekt intervall. Dessutom indikerade resultaten att flotation inte kan appliceras enskilt för att nå koncentrationsmålet utan bör kombineras med kompletterande kolfilter för att nå gränsvärdet.

Flotation ger upphov till en mer koncentrerad ström av PFAS som kräver behandling för att eliminera kemikalierna från miljöns kretslopp. Flera potentiella behandlingsmetoder identifierades för hur slammet kan hanteras. Jämförelsen inkluderade aktivt kolfilter, jonbytare, membran, oxidationsprocesser, sonolys, förbränning, deponi, jordrening, flotation, skumfraktionering, behandling på reningsverk och återanvändning för jordförbättring. Metoderna jämfördes utifrån bevisad robusthet, kostnad och applicerbarhet. Ett möjligt slamhanteringsalternativ visade sig vara ett sekundärt steg med flotation, förbränning av det uppkoncentrerade slammet och behandling av permeatet med jonbytare.

Med den föreslagna slamhanteringen kunde implementering av flotation utvärderas baserat på maximal PFAS destruktion, ekonomi, koldioxidutsläpp och energiförbrukning. Det visade sig att energiförbrukningen är större för processen som inkluderar flotation men koldioxidutsläppen är mindre. Dock är det möjligt att avgränsningarna för koldioxid kan ha gynnat flotationsprocessen eftersom utsläpp från reaktivering av kolfilter inkluderades men inte utsläpp från förbränning eller energiproduktion. PFAS destruktionen blev ungefär 1-2% större för processen med exklusivt kolfilter eftersom flotationsprocessen innehåller ytterligare en ström som släpper ut PFAS efter behandling med jonbytare. Det mest ekonomiska alternativet visade sig bero på koncentrationsgränsen där implementering av flotation var dyrare för en PFAS4 koncentration på 4 ng/L i dricksvattnet, men billigare vid en gräns på 3 ng/L. Slutligen upptäcktes det att volymen av sekundärt slam som beräknades att skickas till förbränning har en stor påverkan på driftkostnaderna. Därför bör optimeringar genomföras för att minimera den sekundära slamvolymen om flotation implementeras.

Nyckelord

PFAS4, dricksvattenproduktion, flotation, granulärt aktivt kolfilter, slamhantering

Acknowledgements

This essay concludes a five-year study in the program Chemical Engineering with the Master of Chemical Engineering for Energy and Environment at KTH, Royal Institute of Technology. I would like to begin by expressing my deepest gratitude to my supervisor at Norrvatten, Lisa Wähländer. My success in completing this project would not have been possible without the support and guidance that she provided. I would also like to extend my thanks to Daniel Hellström at Norrvatten for providing me with this project. I am deeply thankful for the opportunity to work on this project in collaboration with Norrvatten. Furthermore, I want to express my appreciation to my examiner at KTH, Henrik Kusar, for his support, insightful comments, and assistance during this project.

I am also extremely thankful to Måns Lundh who has provided me with valuable advice, helpful contributions, and practical suggestions. I would like to acknowledge the assistance from Eurofins which have performed analysis on the samples gathered in this project with an especial thanks to Torbjörn Synnerdahl and Fanny Karlsson for their attempts to identify possible errors in the analysis results. I must also thank Ewelina Basiak-Klingspetz, Frida Hugg, Ida Sekizovic, and David Heldt for their helpful contributions of information about the Norrvatten process. I very much appreciate the opportunity to participate in the kick-off of the SID-Water project with the invitation from Lutz Ahrens. That meeting really gave me an insight into the extent of the PFAS challenge, the complexity of drinking water treatment, and the opportunity to share knowledge from several experts including Philip McCleaf who carried out many research projects within the PFAS challenge. Lastly, I want to express my sincerest gratitude to all of those who have supported my effort in this thesis including my family, partner, and friends.

Acronyms

AIX Ion Exchange

DAF Dissolved Air Flotation

EBCT Empty Bed Contact Time

EFSA European Food Safety Authority

GAC Granular Activated Carbon

NF Nanofiltration

NFWP Norrvatten's Future Water Production

PFAS Poly- and Perfluorinated Alkyl Substances

PFHxS Perfluorohexyl Sulfonate

PFNA Perfluorononanoic Acid

PFOA Perfluorooctanoic Acid

PFOS Perfluorooctane Sulfonic Acid

POPs Persistent Organic Pollutants

RO Reverse Osmosis

SAFF Surface Active Foam Fractionation

SCWO Super Critical Water Oxidation

TS Total Solids

WHO The World Health Organization

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1 Introduction

Clean and safe drinking water is fundamental to human health and plays a critical role in supporting basic survival, sanitation, and hygiene [1]. Access to clean drinking water is a basic human right and is essential to reach sustainable development [2]. The World Health Organization (WHO) remark that contaminants in drinking water pose an extensive threat to human health as it has been related to several diseases. Drinking water provides a significant exposure route for chemical exposure, which makes it crucial to ensure a high quality of drinking water [1]. Healthier drinking water may be the most efficient target to minimize overall human exposure, which can be achieved through treatment processes and upstream measures [1].

As Sweden's fourth largest drinking water producer, Norrvatten supplies Arlanda Airport, several hospitals, and approximately 700 000 people in 14 municipalities in northern Stockholm. Their waterworks, Görvälnverket, is located in Järfälla, produces 1600 liters of drinking water every second with Mälaren as a water source [3]. With the increasing population in northern Stockholm, Norrvatten plans to expand their production capacity. Thus, a supplementary facility will be constructed adjacent to Görvälnverket. However, in order to initiate the construction process several years of planning, work, and decisions will be required [4].

One aspect that requires extensive investigation before implementing a new treatment process is the removal of Poly- and Perfluorinated Alkyl Substances (PFAS). PFAS is a large group of chemicals that have been produced since the 20th century due to their extraordinary characteristics that are utilized in various products [5]. The chemicals are non-degradable in nature and consequently, large amounts end up in foods and drinking water. PFAS is stored for a long time in the human body, which can cause severe impacts on health, particularly for fetuses and children [6]. For that reason, the Swedish Food Agency has set a new limit for PFAS in drinking water at 4 ng/L for four of the most common PFASs to ensure safe and healthy drinking water [5]. This limit leads to extra treatment requirements for Norrvatten's waterworks.

In this project, the new supplementary facility, called Norrvatten's Future Water Production (NFWP), will be analyzed with a focus on ensuring a sufficient reduction of PFAS in accordance with the limit from the Swedish Food Agency. Activated carbon filters and Dissolved Air Flotation (DAF) have been proposed as alternative methods to reduce PFAS in the planned process. A system analysis is required to determine the most optimal alternative with regard to waste stream management, carbon emissions, energy demand, and cost. Thereupon, the most desirable decision can be made to favor both Norrvatten, the environment, and the citizens in northern Stockholm. As a result, it would also support a sustainable future and promote the sustainable development goals set by the United Nations. For example, goal 6: 'Clean water and sanitation' is promoted since PFAS removal improves the quality of drinking water. Also, more responsible management of PFAS from water courses could be achieved, which is a part of goal 12: 'Responsible Consumption and Production' [7].

1.1 Aim

This thesis intended to investigate the potential of DAF for PFAS reduction in Norrvatten's waterworks. Therefore, DAF and activated carbon filters are analyzed for two degrees of purification at 3 ng/L and 4 ng/L to reach the PFAS4 limit of 4 ng/L set by The Swedish Food Agency. It will be disclosed if the inclusion of DAF benefits the purification process. The comparison will regard a holistic perspective with focus on maximum PFAS destruction, minimal operating costs, climate impact, and residual stream management.

1.2 Research Questions

To address the problem and answer the aim presented above, the following three research questions are answered in this report.

- What separation efficiency is reached with DAF and what are the concentration flows after purification in the DAF outlet and residual streams?
- What PFAS treatment techniques or disposal possibilities could be suitable for managing the sludge produced from DAF?
- What consequences do inclusion of DAF entail in terms of carbon dioxide emissions, electricity requirement, reactivation frequency of activated carbon filters, and cost for one sludge management proposition?

1.3 Delimitations

This report focused on the proposed process for Norrvatten's supplementary water works. The separation efficiency that could be obtained by DAF was measured in one of Görvålverket's basins, called line 5. The scope of the study was limited to the examination of PFAS4 concentration and no other PFASs were concerned. The study did only consider the operation phase of NFWP's treatment process. It did not consider the implementation phase, the existing water treatment process, or any future perspectives such as PFAS4 concentration changes in Mälaren. The analysis focused on direct costs and direct emissions connected to the operation phase of the treatment process. Thus, only emissions and costs for the waterworks were included with an exception for carbon emissions related to the reactivation of activated carbon filters. There were not any optimizations of the separation or treatment methods, meaning that the analysis was based on existing conditions. In addition, the comparison between the inclusion of DAF in the process or only implementing GAC was made from one sludge management route. The chosen management route was limited to methods where a reliable cost estimation could be identified.

2 Background

The background will provide an overview of the most essential elements that are considered in the report. PFAS is described with its physical and chemical properties and its health effects are presented. Also, the four PFAS that are limited by the new limit from the Swedish Food Agency, PFAS4, is specified. Thereafter, the most common PFAS removal methods that exist are presented. Lastly, the water treatment process at Görvånverket and the plans for the new supplementary treatment facility are explained.

2.1 PFAS

PFAS is a group of synthetic fluorinated chemicals that have been produced and used since the 1950s [8, 9]. The chemicals consist of a hydrophobic alkyl chain of 4 to 16 carbons and a hydrophilic end group (see Figure 1). The alkyl chain can be fully or partially fluorinated, which affects the chemical stability and characteristics of the chemical. The end group on the chemicals can be sulfonates (SO_3^-), carboxylates (COO^-), phosphates (OPO_3^-), quaternary ammonium (NR₄⁺), or hydroxides (OH) [10].

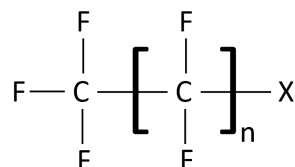


Figure 1: General structural formula of PFAS, where X is a functional group.

Baresel et al. [8] state that almost five thousand known chemicals belong to the PFAS family according to the Organization for Economic Cooperation and Development [8]. Over two thousand of those are present on the global market [11]. The extensive number of chemicals illustrates the broad usage and spread of PFAS. They have been used in the production of commercial, household, and consumer products including nonstick cookware, water-repellent clothing, stain-resistant fabrics, firefighting foam, and cosmetics. The chemicals are also widely used in industrial processes as for example metal electroplating [8, 9, 12, 13].

PFAS can bioaccumulate, avoid biodegradation, and persist in nature for an extended period of time due to the stable structure of the molecules. This explains why the chemicals can be globally found in numerous places in the environment including surface waters, groundwater, air, waste, soil, ice sheets, plants, animals, and humans. Globally, the chemicals are introduced to nature from contaminated land areas and untreated wastewater from industry and municipalities. There is no PFAS production in Sweden meaning that PFAS enters the country by importation of consumer and industrial products or through atmospheric deposition. Runoff from firefighting foam constitutes a large source of the PFAS in water courses, and soil in Sweden [8]. Due to contaminated water sources and inadequate PFAS removal, PFAS has been discovered in drinking water, which has been identified as one of the major PFAS exposures to humans [8, 9].

PFAS is rapidly emerging as a threat to public health and requires extensive work to remove from the water supply with efficient and economical solutions [14]. Therefore, as part of an EU directive from 2006, restrictions were set on one of the most common PFASs, called Perfluorooctane Sulfonic Acid (PFOS). In the directive, which applied from 2008, the marketing, and use of PFOS in new products were regulated. Perfluorooctanoic Acid (PFOA), another common PFAS, was also included in the directive for risk assessment [10]. Additionally, from the end of 2020, PFOA was prohibited in 182 countries by the United Nations to increase the protection of human health and the environment [15]. Despite the regulations, PFAS will remain in nature for an extended period of time and will therefore continue to cause problems [8]. There is also an additional issue raised by Baresel et al. [8], who suggest that the regulations do not consider false substitution, where chemicals are replaced with substances that have similar environmental and health effects.

2.1.1 PFAS4

Lindfeldt [16], stated that the majority of the exposure to drinking water comes from four PFAS chemicals. These four chemicals are PFOA, PFOS, Perfluorohexyl Sulfonate (PFHxS), and Perfluorononanoic Acid (PFNA), which are classified together in a group called PFAS4. Those chemicals are also the most studied PFASs and have been phased out in several countries [12]. The four chemicals differ in length and functional group according to Figure 2. PFOS contains eight completely fluorinated carbons and has a sulfonate group at one end of the chain. PFOA is an eight-carbon organic acid that is completely fluorinated. PFNA has one more carbon than PFOA, while PFHxS is similar to PFOS but only has six carbons.

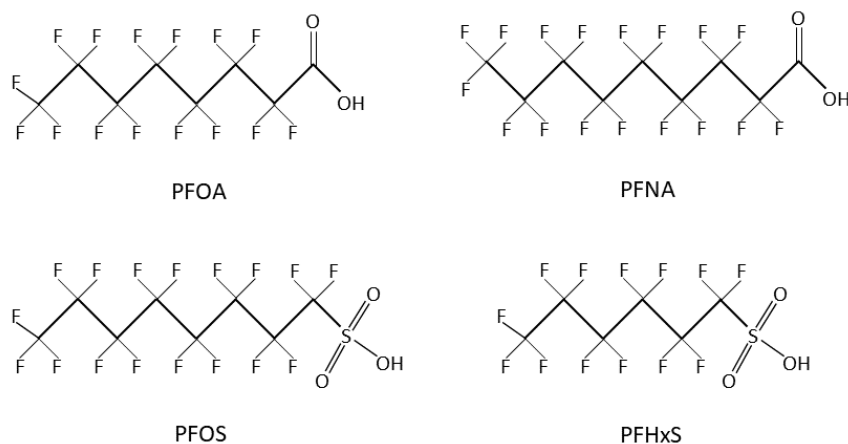


Figure 2: Structural formula of the chemicals included in PFAS4

PFOA and PFOS are historically two of the most studied and utilized chemicals for numerous applications [8, 9]. Therefore, the main part of the existing knowledge about PFAS originates foremost from PFOA and PFOS [17]. According to the European Food Safety Authority (EFSA) [10], the concentration of PFOS is generally higher than PFOA in the environment and food samples. As a result of their eight carbons, they exhibit unique chemical properties, including thermal and acid resistance, water and oil resistance, and high surface activity. They are stable, non-volatile, and very soluble in water. Also, they are very mobile in the environment which facilitates them to disperse in nature [11].

The limit value from the Swedish food agency is based on a tolerable weekly intake calculated by EFSA for PFAS4, which was determined to be 4.4 ng/kg body weight and week. The limit concerns the sum of PFAS4 as they have displayed similar toxicokinetics, have comparable effects in animals and have shown the highest concentration in human blood. The risk evaluation has been determined with the assumption that all PFAS4 have identical toxicity and toxicokinetics. Based on the tolerable weekly intake from EFSA, exposure from intake of food, and an allocation factor of 20%, it was found that drinking water exposure should not exceed 4 ng/L. The allocation factor was used since the exposure has more than one source (i.e., both food and water), where the standard for allocation of drinking water for the total daily intake is 20% [18].

2.1.2 Physical and Chemical Properties

The characteristics of PFASs vary tremendously between the different chemicals [8]. However, some characteristics are joint for all PFAS due to the C-F bond that appears in all PFASs. The strong covalent bonds between fluorine and carbon make PFAS highly resistant to degradation and are chemically and thermally stable [8, 10]. Therefore, the chemicals obtain a long half-life that allows them to persist in nature for a very long time, varying from a few months to several years within the PFAS family [13]. Moose [19], present that PFOA has a half-life of 92 years in water, while PFOS has a 41-year half-life.

A PFAS can be classified as long or short depending on its carbon chain. Long carbon chains have eight or more carbons if the chemical is polyfluorinated (i.e. not completely fluorinated). For perfluorinated PFAS (i.e. completely fluorinated carbons), long chains have six or more carbons. The short carbon chains of PFAS are gaining attention since they are more mobile in nature [19]. Some short PFAS are volatile and can spread by atmospheric transport, which explains why they have been found in the Arctic and Antarctic. The shorter chain PFAS are less bioaccumulative but have the potential to be as toxic as longer chained PFASs [19]. Due to their higher instability and the potential to transform into longer PFAS such as PFOS or PFOA, they are referred to as precursors, which explains why concentrations of longer PFAS in some cases increase in, for example, water treatment processes [8].

Long perfluorinated chemicals are both lipid and water-repellent, which enables them to act as a surface-active agent. The functional group in PFAS locates itself in the water phase and the carbon chain is in another phase such as air [10]. The water solubility is dependent on the length of the chain where a long chain leads to a lower solubility. For PFAS that can bind or release a hydrogen ion, the water solubility is also dependent on the environmental pH, where the anionic form has higher solubility and lower sorption potential. PFOA and PFOS are in their anionic form at environmental pH and will therefore have a high water solubility [8]. However, electrostatic interactions from the sulfonate group will make ionized PFAS more prone to sorb on particles [19]. Therefore, PFOS is often detected in both the soluble phase and sorbed on particles. Long PFAS with more than 11 carbons can only be detected in sediment due to their low water solubility [8].

2.1.3 Health Effects

Humans are exposed to PFAS via a number of routes including ingestion, inhalation, and dermal contact [10]. In the human body, the chemicals are adsorbable on proteins and circulate through the blood [8]. They are not metabolized, which means that they accumulate in the body and can thereby reach toxic concentrations over time [8, 11]. Since humans are high up in the food chain, people are exposed to high concentrations through foods from animals such as meat and dairy products due to bioaccumulation [8]. EFSA has determined that humans can intake 4.4 ng per kg body weight and week of PFAS4 without risking effects on the health [20]. However, in higher amounts, PFAS has a documented toxicity and can lead to multiple health issues, which was the basis of the limit value of 4 ng/L from the Swedish Food Agency [8, 13].

According to Baresel, studies have shown associations between PFAS exposure and health effects including liver disease, insulin disorder, worsen immune and thyroid function, negative effects on development, reduced fertility, kidney disease, and some cancer forms. Camdzic et al. [13], also mention metabolism disruptions that can cause weight gain in especially women as a potential effect from PFAS exposure. They can be transferred through the placenta and be passed on to fetuses, leading to a decreased birth weight [8, 10]. While long-chain PFAS such as PFOS and PFOA have the most toxic effects, short-chain PFAS can also negatively impact human health and the environment in time due to their persistence [8].

Baresel et al. [8], point out that the health effects are hard to determine due to a number of uncertainties. Many of the results are determined through animal studies, but the toxic effects could differ tremendously between species. This is especially true for PFAS since they have significant differences in elimination speed in various species. Moreover, there are very few PFASs that have documented health effects, which leads to a lack of toxicity data for the majority of the PFASs in society. Lastly, the information on the consequences of cocktail effects from exposure of several different chemicals is limited for both different PFAS mixtures and PFAS substitutions. As a result, there is a huge blind spot in the health issues associated with PFAS [8].

2.2 PFAS Removal Techniques

Baresel et al. [8], describe that there are several techniques that can be applied for PFAS removal in drinking water treatment. Some techniques have not progressed further than the developing stage, leaving the most prominent techniques to those that utilize membrane separation, Ion Exchange (AIX), and adsorption on Granular Activated Carbon (GAC). Furthermore, the article highlights surface active foam fractionation or flotation as a promising technique for PFAS removal. Baresel

stated that all of the mentioned methods are more efficient when treating PFAS at high concentrations [8]. To select the most appropriate method for a specific stream, a complete water profile including water analysis, contaminant targeting, evaluation of other technologies on-site, space limitations and volume are necessary to make a justified choice [14, 19]. In the following section, there is a summary of the mentioned removal methods.

2.2.1 Granular Activated Carbon

Activated carbon is a broadly applied method in drinking water treatment for the adsorption of compounds that influence the taste or odor of the water. In addition, it has been used for decades to remove synthetic and natural organic chemicals that may be present in drinking water. Activated carbon functions as an efficient adsorbent because of its porous properties, providing a large surface area where particles can adsorb. The material can be produced from either fossil coal or renewable sources, as for example wood. Activated carbon is commonly used in a granular form in filters in water treatment but is also present in its pulverized form. The pulverized activated carbon could allow for micro compounds to be transferred to the sludge. The presence of micro compounds in sludge is problematic in Sweden as the sludge is commonly used for different purposes including farming applications. Thus, the applicability of pulverized activated carbon is limited [8].

GAC can remove several hazardous micro compounds, such as drug residues, pesticides, and solvents from drinking water. The material has also been extensively researched for the removal of PFOS and PFOA, and has been shown to provide a removal to undetectable levels when applying fresh GAC [11]. The separation efficiency is increased for long PFASs but is also affected by charged functional groups. In addition, high concentrations of dissolved organic compounds in the water source can worsen the removal [8, 19]. The Empty Bed Contact Time (EBCT), which is defined by the bed volume per flow or the filter bed depth per surface load, is relatively short for GAC giving a time-efficient process, but the carbon footprint is more extensive than for some other options [8]. The removal rate is reduced with time, and will eventually require thermal reactivation to raise the efficiency of the material, which is carbon intensive [19].

2.2.2 Membrane Separation

Membrane separation is a commonly applied technique in drinking water treatment, especially Nanofiltration (NF) or Reverse Osmosis (RO) [19]. Membrane processes utilize high pressure to force water through a permeable membrane while retaining dissolved ions or molecules [14, 19]. In RO, the osmotic pressure, that aims to reduce concentration differences, needs to be exceeded to force water through the membrane. Therefore, the applied pressure that is required in RO will be dependent on the concentration in the water stream, which leads to RO being more energy intensive than NF. All PFASs can be removed from the stream when using the RO method, in contrast to NF due to its smaller pore size. However, when decreasing the pore size of a NF the PFAS separation is increased but also the energy demand is enlarged [14]. Because of the generally higher removal efficiency of RO was a basis for Moose [19] to declare RO as the preferred membrane separation technique. However, RO comes with a higher operating cost than NF [19].

One major drawback of membrane separation is the water losses that end up in the retentate with high PFAS concentrations. Additionally, there can be issues with its high capital cost and energy demand. Moreover, the membranes are exposed to fouling where particles accumulate on the membranes, which can make contaminant removal problematic at high levels of fouling. Therefore, the stream that should be treated may require extensive pretreatment to reduce clogging of the membranes. The separated PFAS will require disposal, which can be costly and usually requires either hazardous landfill or incineration according to Moose [19]. Baresel et al. [8], suggest activated carbon as an alternative to clean the concentrate, since it has a higher capacity at higher pollution concentrations. The activated carbon can be sent for reactivation after treatment of the concentrate to destroy the collected PFAS.

2.2.3 Ion Exchange

AIX is a technique that utilizes synthetic resin media that can be applied for the removal of PFAS from drinking water. The resin media are small, porous plastic beads with charged functional

groups that are specifically designed to adsorb a specific contaminant [8, 14]. Each functional group on the resin media can capture one molecule by releasing a hydrogen ion or hydroxide group. PFAS typically exist in water as anions, which benefits the removal by AIX. AIX is more selective than GAC, but has a larger pressure loss over the filter and can require more extensive pretreatment to avoid clogging [8]. The technique is capable of removing both short and long-chained PFASs at an exceptionally short EBCT [14]. However, the removal efficiency and the reduction speed are dependent on several factors including the PFAS concentration, the presence of competitive ions, the treatment design, and the ions used in the beads [8]. In addition, short-chained PFASs have usually a lower sorption ability resulting in a worsened removal efficiency. After usage, the media is typically sent for thermal destruction. Some AIX materials can be regenerated, but the regeneration process does not destroy the PFAS resulting in PFAS being transferred to a secondary pollution stream that requires treatment [14].

2.2.4 Dissolved Air Flotation

For over a century, DAF has been used to remove suspended pollution from water streams [21]. The separation technique is applicable for micro-scale-sized particles with low density. It utilizes tiny air bubbles that can collect particles and rise to the surface to be separated from the water. The generation of microbubbles is achieved by mixing air and water in an air saturator under high pressure at usually 300-800 kPa. The supersaturated water is released through a pressure reduction nozzle that is subject to atmospheric pressure. The pressure drop causes the air to precipitate as microbubbles in the flotation tank. The microbubbles collide with suspended particles in the water, adhere to them, and rise to the surface of the water due to the buoyancy of the microbubbles. On the surface, the bubbles and pollution construct a sludge blanket that can be mechanically removed resulting in a more clean water stream [22].

The surface active properties of PFASs force the chemicals to be placed on the surface between the water and air bubbles. The carbon chain is present in the air and the functional group in the water according to Figure 3. The position causes the PFASs to follow the air bubbles into the sludge that is produced, resulting in a concentrate of PFAS in a smaller, more manageable volume. The sludge will require treatment in terms of additional removal and destruction methods. Usually, the sludge obtains a concentration of 10 to 20 times the inlet concentration when used for drinking water treatment. However, the PFAS separation efficiency varies depending on the chemical structure of the substance, as all PFAS have different surface properties affecting the ability to attach to the microbubbles. In general, PFAS compounds with six or more perforated carbons can be almost completely separated by DAF at optimal conditions. On the contrary, the chemicals with less than four perforated carbons are barely separated [8].

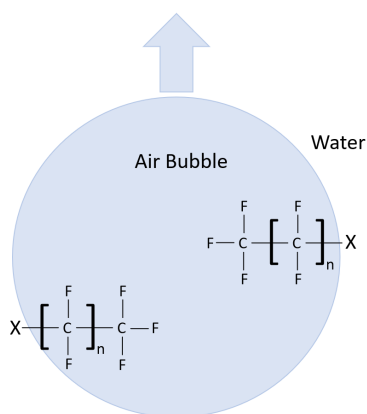


Figure 3: Principle of PFAS removal by DAF.

Beyond the properties of the chemicals that should be removed, there are some important parameters in a DAF system, such as air dissolution, bubble size distribution, and the hydrodynamics of the system. Air dissolution plays a vital role in the DAF system in two ways. Firstly, it supplies air

that is precipitated as microbubbles and separates the suspended particles. It affects the removal efficiency as more dissolved air leads to more microbubbles, which in turn increases the number of collision opportunities between bubbles and particles [22]. Lundh et al. [23], stated that an increased air flow would increase the separation due to an increased bubble surface. However, the water amount in the foam would also increase, leading to a lower concentration of the contaminant in the foam [23]. The second reason is related to economics as the air saturators constitute about 50% of operating costs [22].

The microbubble size distribution affects the effectiveness of a DAF system. Smaller-sized microbubbles increase the likelihood of collision with suspended particles, thus improving the system's performance. The hydrodynamics of the flotation tank is another aspect of the DAF system that influences its performance. Hydrodynamics refers to the bubble rising velocity and flow patterns inside the flotation tank [22]. The size of the microbubbles impacts the velocity at which they rise, which can affect the contact time of the bubbles as well as the rate of collision with particles. According to Fanaie and Khiadani [22], the size of microbubbles in the contact zone ranges from 40-80 μm , while the size in the separation zones ranges from 50-150 μm . Lundh et al. [23], presented that a longer contact time increases the separation of a contaminant, which can be achieved by smaller air bubbles since they rise more slowly to the surface.

Usually, there is no need for chemical addition if the water does not create too much or too little foam, which would complicate the PFAS separation. However, the addition of surface-active substances as for example Fe^{3+} or Al^{3+} has shown to improve the efficiency [23]. DAF alone may not provide a complete PFAS removal, thus additional techniques may be required to meet the concentration demands [8].

2.2.5 Foam Fractionation

A similar method to DAF is foam fractionation. Foam fractionation is traditionally used for wastewater treatment to remove suspended solids, oil, and grease from industrial effluents. Also, both technologies use microbubbles to float impurities to the surface for removal. However, there are several differences between the two, including the process mechanism and microbubble generation. In foam fractionation, the air is mixed with a surfactant to generate a stable foam that can float impurities to the surface [24, 25]. Another difference between DAF and foam fractionation is the size of the bubbles. For DAF the bubbles are in the order of magnitude of 100 μm while foam fractionation generates bubbles in millimeter sizes [23]. Foam fractionation can have a higher efficiency for removal of low-density particles in comparison to DAF. However, that method has a limited amount of available surfactants that can be used in foam fractionation and can be energy-intensive [24, 25].

2.2.6 Previous Flotation Investigations in Literature

The PFAS removal efficiency from coagulation followed by sedimentation or DAF was evaluated by Appleman [26]. Their goal was to evaluate PFAS removal in different processes in full-scale water treatment. The process included coagulation where the added coagulants included aluminum sulfate and polymer, aluminum sulfate, and polyaluminum chloride. Coagulation was followed by an ozonation step before either DAF. The water inlet to the process contained a concentration of 11 ng PFOS/L and 2 ng PFNA/L. They discovered that DAF provided a 49% removal of PFOS and 29% removal of PFNA. All short-chained PFASs that had been detected had a minimal removal by DAF [26].

In a study by Thompson et al. [27], the PFAS concentration was measured in a water reclamation plant in Australia containing a DAF step. Before the flocculation and DAF, the treatment process contained a de-nitrification and ozonation step. The inlet concentration of PFAS4 was 10.9-12.6 ng/L. The results indicated a relatively constant concentration of PFOA and PFHxS throughout the entire treatment process. In contrast, the removal of PFOS and PFNA could be observed as primarily related to DAF and biologically activated carbon filtration. It appears as PFNA was reduced by at least 27% and PFOS was reduced by around 50% over the flocculation, DAF, and sand filtration in the process. There was no information given about the contact time, air content

or the dimensions of the basin [27].

McCleaf et al. [28] suggested that foam fractionation could be used on PFAS concentrate from NF with a concentration of approximately 400 ng PFAS per L. They stated that foam fractionation is usually most efficient at high PFAS concentrations such as landfill leachate and therefore should be appropriate for NF concentrate as well. It appeared as foam fractionation had a PFAS removal efficiency of 90% without co-surfactants. When the cationic co-surfactant Montaline C40 was added, the removal efficiency increased to 94% at an airflow of 4 L/min, yielding a foam with over 3000 ng PFAS/L. McCleaf et al. suggested that for example electrochemical oxidation, chemical reduction, or incineration could be used to destruct the PFAS in the concentrated foam. Those methods would have an increased efficiency after foam fractionation due to the resulting low volume and high PFAS concentration. However, they raise a concern about the fate of the co-surfactant as it may be present in both the foam and the permeate and would entail problems if released into the environment.

In addition to these studies, several trials have been executed where landfill leachate is treated with DAF. Landfill leachate usually has a very high concentration of PFAS. In a study by Robey et al. [29], the removal effectiveness of DAF on landfill leachate from central Florida was investigated. The leachate had an initial mean concentration of 950 ng PFOA/L, 100 ng PFOS/L, 60 ng PFNA/L, and 2000 ng PFHxS/L. During the investigation, the air was bubbled through the landfill leachate at about 2.6 L/min. The effectiveness varied between each compound but for PFAS4 a removal efficiency of over 90% was achieved resulting in a foam concentration of 5700 ng PFOA/L, 270 ng PFOS/L, 200 ng PFNA/L and 10 600 ng PFHxS/L. Robey et al. found a positive correlation between the number of carbons in the carbon-fluorine chains and interfacial partitioning, which suggests that longer PFAS compounds would have a higher removal efficiency. The investigation also showed a lower efficiency for the removal of the largest and smallest PFAS compounds. Lastly, it appeared as the technique was similarly effective at removing sulfonates and carboxylate PFAS compounds [29].

In an experiment by McCleaf et al. [30], a removal efficiency greater than 90% could be achieved for PFOA, PFOS, and PFHxS using foam fractionation on landfill leachate. PFNA reached a removal of 50-80 %. In the study leachate was collected from Hovgården landfill. The leachate was added into a column where air was introduced at the bottom. Samples were taken over a time period of 20 or 60 minutes. They found that an increased airflow increased the efficiency and that the maximum PFAS removal efficiency was achieved at 10 minutes.

2.2.7 PFAS Destruction

After PFAS have been separated from the water, it is essential to destroy the chemicals to prevent them from circling back to the environment. Destruction of PFAS entails a complete defluorination of the carbon chain. Because PFAS are persistent, they are difficult to remove from drinking water since they often are relocated into another medium rather than destroyed [14].

Thermal destruction is most frequently applied, where either incineration or pyrolysis is used. Usually, incineration is used to permanently destroy PFAS but can entail environmental issues from the high energy consumption and carbon emissions. Incineration is also problematic because normal incineration temperatures are not high enough to degrade PFAS, causing the chemicals to enter the air instead [14]. According to Moose [19], a temperature greater than 1000 °C is required to completely destroy PFAS because of the thermal stability of the chemicals. However, the required temperature is dependent on the length of the carbon chain and the concentration in the media that should be destroyed [8].

Reactivation and regeneration of spent activated carbon is discussed by DiStefano et al. [31] where only reactivation has the potential to destroy adsorbed contaminants. In regeneration, pyrolysis is used to partially oxidize the carbon at 900 °C in an oxygen-poor environment [8]. In reactivation elevated temperatures are applied in furnaces with addition of a selective oxidant absent from oxygen. The high temperature restores GAC close to its virgin state, which reinforces its adsorption capacity [32]. DiStefano et al. [31], found that a PFAS destruction greater than

99.99% could be achieved through carbon reactivation. Therefore, is reactivation of used GAC a special case of thermal methods to destruct PFAS [8].

2.3 Current Water Treatment Process at Görvånverket

When Norrvattens waterworks Görvånverket was built in the end of the 1920s the treatment process consisted of a rapid filtration and chlorination. With an increased demand for drinking water and new requirements, the treatment process has advanced and the water must now undergo several process steps until it becomes drinking water [20, 33]. The steps in the treatment process are illustrated in Figure 4 and are described below.

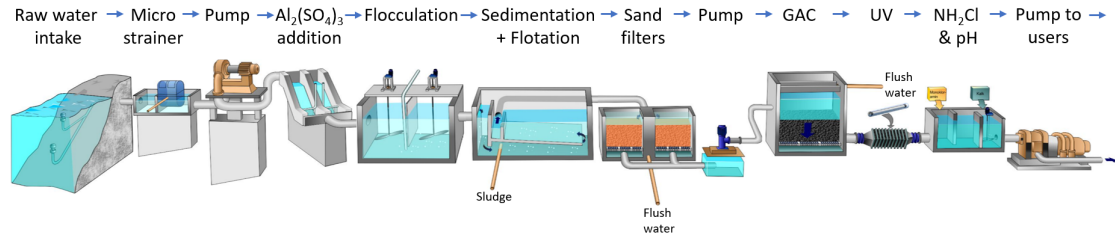


Figure 4: Flow scheme of the water treatment in Görvånverket. From (Norrvatten, *Reningsprocessen*).

Raw water is collected from Görvån fjärden, which is filtered from large matter, such as fish and algae, by a large micro-filter called a basket belt strainer. The water is pumped into the purification process and is mixed with aluminum sulfate, which induces flocking. The water continues to the flocculation chamber where the water is mixed to create flocs. In the flocs, the coagulants bind themselves to humus substances (decomposed plant residues), clay particles, and microorganisms. Sodium silicate allows the flocs to attach to each other since larger flocs are easier to remove.

To remove the flocs the water is divided into five sedimentation basins, called lines, where the flocs settle, sink to the bottom due to gravity and form sludge [33]. The produced sludge is removed from the bottom of the basin using scrapers. The sedimentation basins have recently been converted into a hybrid basin combining simultaneous DAF and sedimentation to enlarge the production capacity [34]. The DAF is dimensioned to be used during the mean to maximum production flow to enable an increased capacity at high production [35]. In DAF, water is saturated with air in high-pressure pumps at 5-6 bars, before it is released through dispersion valves in the flotation basin [35]. Air bubbles are formed, which produces a foam on the water surface that is removed by either a scraper (basin 1) or a waltz (basin 2-5). The foam is transferred outside of the facility and is mixed with sludge from the rest of the treatment process and is dewatered using sedimentation and centrifugation (see Figure 5) [34].

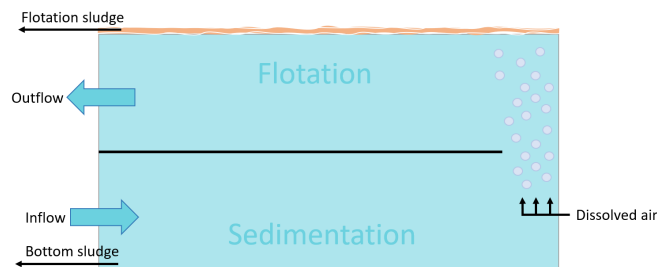


Figure 5: Schematic Figure of a hybrid basin.

In the water treatment process after the sedimentation/DAF basins, the water runs through rapid sand filters to remove the last remains of the flocks that were not separated with sedimentation or flotation. To further improve the quality of the water GAC filters are used as biological filters due to the lack of reactivation, which removes smaller particles that affect the smell and taste of the water. Then the water flows to UV reactors that disinfect the water. Lastly, chemicals are added to the

water. Lime water is used to adjust pH to be slightly alkaline, which reduces the risk of corrosion. Finally, a low dose of monochloramine is added to prevent bacterial growth in the pipe system [33].

The volumetric flow is measured in several parts of the water treatment process. There are two different methods used to measure the volumetric flow in the process, namely by weir or by magnetic inductive flow meters. Weir allows the water to flow through a rectangular or triangular geometry and by measuring the depth of the water the total flow can be decided. By automatically measuring the water level, the volumetric flow can be calculated for the process [36]. Magnetic inductive flow meters can measure the volumetric flow in a pipe. It is based on electromagnetic induction that is defined by Faraday’s law. A magnetic field is produced over the cross-sectional area of the pipe, a sensor generates an electric current that is directly proportional to the flow speed in the pipe. Magnetic inductive flow meters can reach an accuracy of $\pm 0.5\%$ [37].

The PFAS that needs to be removed from the drinking water produced in Görvälnverket has its origin in Mälaren. The concentration in the raw water differs slightly, where Norrvatten has measured the PFAS4 concentration to be between 2.2-7.7 ng/L in the years 2013-2020 [20]. However, due to legal restrictions and upstream retribution in recent years, the PFAS4 concentration has decreased in Mälaren. Thus, in the last five years, the concentration of PFAS4 has been measured to 3.6-5.7 ng/L in the raw water. The average concentration for each PFAS4 between 2018-2022 is shown in Table 1.

Table 1: Average PFAS concentrations in raw water samples 2018-2023.

PFOA	PFOS	PFNA	PFHxS	PFAS4
1.1 ng/L	2 ng/L	≤ 0.3 ng/L *	1.3 ng/L	4.5 ng/L

* Lower concentration than the detection limit of 0.3 ng/L for 20 out of 30 measurements.

Norrvatten has also measured the PFAS4 concentration in the drinking water after the treatment process in Görvälnverket. The measurements have shown concentrations between 3.4-5.9 ng/L [38]. Beyond the new limit at 4 ng/L from the Swedish Food Agency [5], Norrvatten has determined an internal guideline that declares that substances with harmful features should not be released in more concentrated forms compared to the background levels in Mälaren [39]. Therefore, the flotation sludge will require treatment to reduce the PFAS4 concentration to the same concentration as in the raw water.

2.4 Planned Process for Norrvatten’s Future Water Production

Despite the increase in population in northern Stockholm, the water production has historically remained relatively constant due to the development of new water-saving technology [40]. However, since 2014 the water consumption has increased by almost 20%, which causes Görvälnverket to reach its production capacity limit in some parts of the year [40, 41]. It is estimated that it should be possible to increase the maximum capacity of Görvälnverket to 220 000 m³/d by implementing DAF basins, which should meet the capacity requirement until 2030. Thereafter, Norrvatten has to extend their production lines to meet the demand [20]. This will be done through a supplementary facility next to Görvälnverket, with the project name NFWP [41].

The treatment process in NFWP will differ from the current process in Görvälnverket due to an increased demand for chemical substance protection, worsen water quality in Mälaren, and an inadequate microbial barrier effect. Therefore, the treatment will be improved to ensure safe and healthy drinking water, with an additional microbial barrier and a barrier for substances that are harmful to health [20]. Currently, the planned process scheme is according to Figure 6.

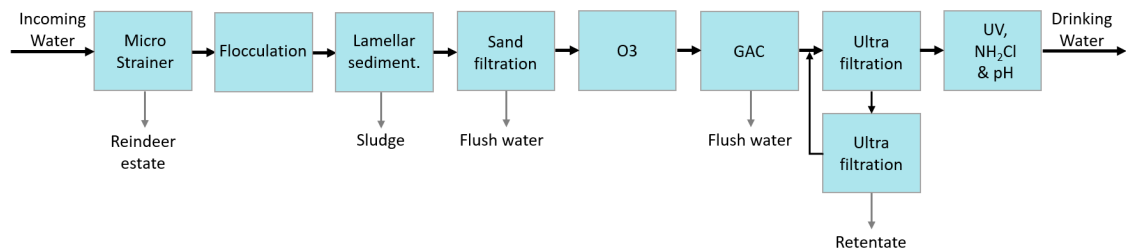


Figure 6: Flow scheme of the proposed water treatment process in NFWP.

The initial part of the process is relatively similar to the existing process in Görvålnerket. The process will contain a micro strainer to provide a rough cleaning. Then, flocculation will enable removal of particles where precipitation chemicals will be added, similarly to the process in Görvålnerket. Thereafter, lamellar sedimentation will be used where the flocks will settle on the bottom of the basin and be removed. The sludge that is produced in lamellar sedimentation will be pumped out of the process and treated in a similar manner as the current dewatering technology at Görvålnerket. Rapid sand filters will further remove particles that did not settle in the sedimentation basin.

Thereafter, the process will provide microbial and chemical barriers to protect against harmful substances. The process will be prepared for an ozone step. Ozone may be added to ensure a barrier for unknown chemicals in the future and optimize the lifetime of the GAC filters. Initially, some space will be kept to make room for ozonation but ozone generators will not be constructed [42]. GAC filters will be implemented to remove compounds that affect smell and taste. Also, they will remove micro substances as the activated carbon will be reactivated more often compared to the GAC used in Görvålnerket. Thereafter, ultrafiltration has been added into the process, which increases the removal of infectious agents. This will be done in two steps to minimize water losses as shown in Figure 6. Lastly, there is a UV treatment and monochloramine addition to disinfect the water and pH will be adjusted before the water enters the supply networks [34, 43].

Some residual streams may be recirculated back into the process before the flocculation step to make use of the water content of those streams. This is not illustrated in Figure 6 as it is not decided what streams that should be recirculated. It is possible that the water used in backwash of sand filters may be recirculated. Also, the retentate from the second ultrafilter may also be recirculated to the flocculation step. Both of those options would increase the water production by about 5000 m³/d [43]. However, the calculations performed in this investigation are based on the flows presented in Figure 6.

2.4.1 Flotation in Norrvattens Future Water Production

Norrvatten is currently considering including a DAF step in NFWP. The reason for implementation of DAF is to support the PFAS cleaning to reduce the reactivation need of GAC filters and possibly reduce cost and environmental impact [23]. DAF would be implemented in addition to lamellar sedimentation, similar to the present solution at Görvålnerket. Thus, it would not act as a substitute due to the challenges to handle large amounts of PFAS-contaminated sludge. The prevailing process proposal is to place the flotation on top of the sand filters to avoid increasing the facility area. The water is led to a contact zone where the dissolved air will be added, which creates microbubbles that rise to the water surface. The flotation sludge is formed on the surface and is removed by a waltz into a sludge chute to transport it to sludge treatment. The water is separated from the air phase and is led through the sand filter by gravitation. Thereafter, the water will continue to subsequent process steps [23]. A preliminary schematic of the configuration is illustrated in Figure 7.

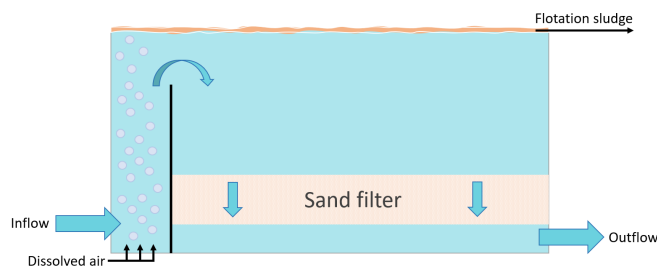


Figure 7: Schematic figure of DAF over sand filters in NFWP.

The sludge produced from the DAF will contain more concentrated PFAS than the water. Therefore, the sludge will require additional treatment of the clear phase or the dewatered sludge to ensure that PFAS is not recirculated into the environment. What treatment that is appropriate will be discussed later in this report. GAC filters will be required regardless of the separation efficiency of DAF to reduce smell and taste as well as remove other micro substances. It is also possible that GAC will be required as a complement to DAF to reach Norrvatten’s maximal PFAS4 concentration demand. The flow scheme of the process in NFWP when DAF is implemented is presented in Figure 8.

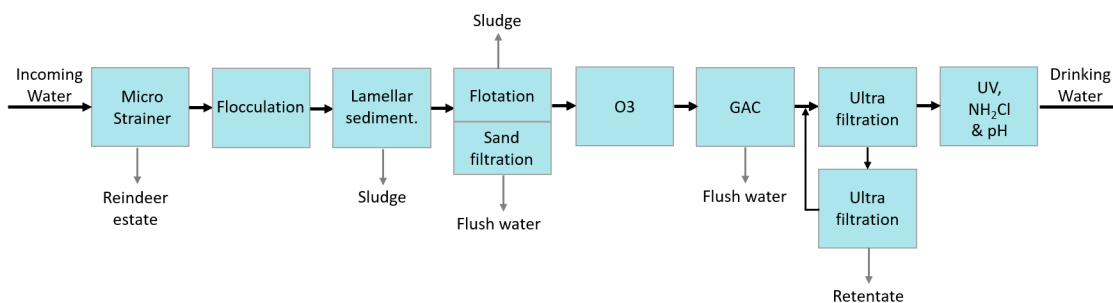


Figure 8: Flow scheme of the proposed water treatment process in NFWP including DAF.

2.4.2 Activated Carbon Filter Configuration in Norrvatten’s Future Process

Norrvatten has carried out two pilot trials at Görvålnverket to evaluate PFAS reduction in activated carbon filters. These investigations have acted as a basis for the planned GAC filter configuration in NFWP [44]. The prevailing solution consists of 32 parallel connected GAC filters in groups of four, which divides the process into eight lines. The GAC filters should be implemented with an EBCT of 20 minutes for the mean volumetric flow in the year 2050, resulting in a carbon volume of 102 m³ per filter [42, 44]. The EBCT was calculated for NFWP when disregarding the previous ozonation step that may be included in the process [44]. It is estimated that 1-2 filters will always be out of order due to reactivation [42]. The dimensioning of the GAC filters has been done based on the mean concentration of PFAS4. The mean concentration in the raw water has, according to Ramboll, provided a required degree of reduction at 25%, which means that an average breakthrough of 75% can be allowed [44].

The activated carbon filters will be based on gravitation where the contaminants are captured in the first part of the filter bed and decrease with depth. The top of the filter is saturated first and thereafter the contaminants spread downwards in the mass transfer zone. The mass transfer zone moves downwards in the filter as the saturation zone increases and when the mass transfer zone finally reaches the end of the filter, PFAS will be released. When the concentration exceeds the permitted limit, breakthrough has occurred [45]. The breakthrough pattern of PFAS4 is unknown as the first pilot trial ended before total breakthrough was achieved and the second trial is still ongoing [44]. However, through assumptions, Ramboll calculated the frequency of reactivation to 3 years, but to compensate for uncertainties 1-3 years was established to not exceed the permitted PFAS4 concentration.

Norrvatten has considered GAC filters connected in a series of two steps to increase the possibilities for operating strategies to reduce the reactivation frequency, which could have economic and environmental benefits when less carbon is managed. Series-connected filters enable a maximal use of the activated carbon without risking a worse separation of the contaminant. One major benefit with series-connected filters with reversible flow is that reactivation is not necessary until the activated carbon is fully saturated. This is because the limit concentration is often exceeded before the lower part of the filter is saturated in a one-stage filter and the coal needs to be sent for reactivation. In series-connected filters, the carbon filters can be replaced when they are fully saturated as the other filter can compensate for the poorer separation ability. However, when many parallel filters are applied it is possible to allow some filters to exceed the limit value and be fully saturated, while other carbon filters have a higher degree of separation [45].

3 Method

The method consists of a literature investigation, an experimental section, and an explanation of the calculations performed. The literature investigation aimed to gather data that could be used for comparison of the results, review possible sludge management techniques, and collect data needed in the evaluation of DAF. The experimental section had the purpose of determining the performance of DAF and sludge PFAS concentration in the Norrvatten facility. Lastly, the calculations were performed to enable an evaluation of the potential of implementing DAF in NFWP. The entire process in Görvålverket and NFWP is not regarded in this report, which is presented in the system boundary.

3.1 System Boundary

There are three modules of the planned processes for NFWP that will be regarded in this report as illustrated in Figure 9. First, the PFAS4 reduction in DAF was analyzed as well as the outlet concentration to estimate the need for additional GAC filters and the requirements of sludge treatment. This investigation was performed in the sedimentation/DAF basins in Görvålverket. Secondly, possible treatment routes for sludge management were compared to provide a possible management approach in NFWP. Finally, calculations were made within the system boundary to estimate if DAF is more beneficial than exclusively applying GAC in terms of reactivation frequency of GAC filters, carbon dioxide emissions, energy consumption, and costs. In all assessments in this report, all treatment steps except DAF, GAC, and sludge treatment were disregarded.

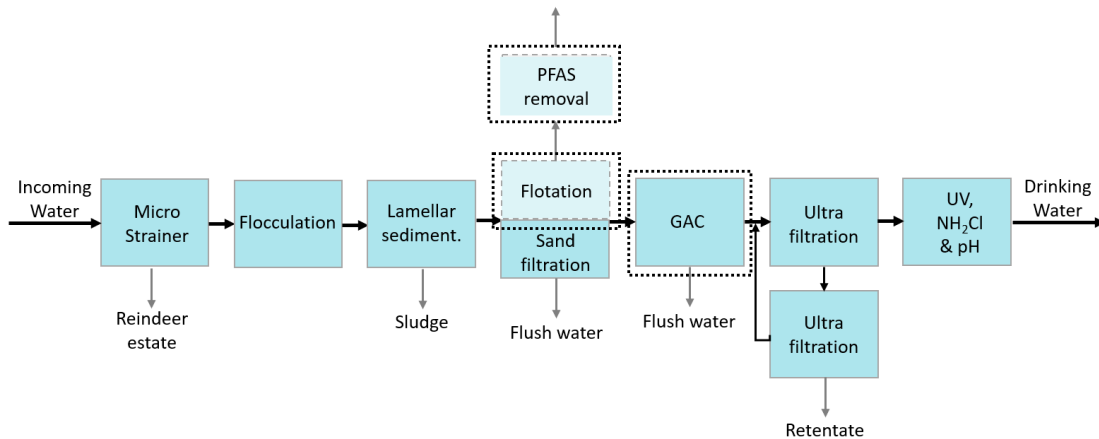


Figure 9: System boundary in process.

3.2 Data Collection

The removal efficiency that was obtained from the experimental procedure was compared with previous investigations performed at Görvålverket in line 1, where three samples had been analyzed for each stream of the sludge, inlet, and outlet in the basin. In addition, the results were compared with other previous literature where studies have been performed on DAF with similar inlet concentrations as are applied in Görvålverket.

To assess different disposal possibilities and treatment techniques for the residual streams produced from DAF in NFWP, a literature review was performed. The review involved all identified techniques for either the entire sludge stream or methods that can be applied after phase separation. The techniques that were included were activated carbon, ion exchange, membranes, oxidation processes, sonolysis, treatment at a wastewater plant, flotation, reuse, incineration, landfill, and soil cleaning. The techniques were identified through literature, by consulting Norrvatten and Svenskt Vatten, and through a kickoff meeting of the SIDWater project, which aims to research new approaches to removing PFAS.

After the identification of different management techniques, advantages and disadvantages were gathered to enable a comparison of the applicability of the methods in NFWP. From the collected

data the alternatives were compared based on feasibility, cost, and PFAS destruction, where four treatment strategies could be identified to be promising options. The final management route was selected based on the presence of reliable data on costs, efficiencies, and operation conditions.

3.3 Experimental Procedure

The experimental procedure consists of a sampling procedure to analyze the PFAS removal efficiency in Görvålnverket, line 5. Also, the experimental procedure includes a centrifuging trial, where sludge was centrifuged to enable PFAS analysis on solid particles.

3.3.1 Sampling for Analysis of PFAS Removal Efficiency

To determine the PFAS removal efficiency that can be achieved by DAF, measurements in one hybrid basin at Görvålnverket were performed. The measurements were taken in line 5, which has a similar construction as lines 2-4. To assess the PFAS removal efficiency in line 5 in the sedimentation/DAF basin, sampling was made on the inlet, outlet, and sludge stream. For the sludge sample, a flow-proportional sample was sought to ensure that the sample could represent all sludge. Therefore, the entire runoff occasion should be included in a sample, where the volume in the samples should be proportional to the total flow in a collective sample [46, 47]. In addition, collective samples were gathered to reduce the possible fluctuations in PFAS concentration or other error sources. Collective samples with 6 hourly sub-samples were gathered from the raw water intake, the flotation sludge in line 5, and the outlet from line 5. 100 mL of raw water was collected from a sampling tap at the raw water inlet. A sample of the water outlet from basin 5 was collected in a 1 L glass flask just below the water surface, where 100 mL was kept. 500 mL of sludge was collected in each sub-sample from an entire pumping sequence.

For the collection of sludge samples, it was decided to collect sludge at the pipe outlet before the sludge tank. That sampling point enabled a more flow-proportional sample and a more constant relation between the foam and water in the samples than other possible sampling points such as in the chute that the sludge is scraped down into before removal. Sampling from the chute would entail uncertainties due to insufficient mixing of the sludge, which would yield uncertainties on the water-foam proportion that would have been collected in each sample.

Line 5 in Görvålnverket consists of two basins with different loads, where the sludge from both basins is removed through the same pipe from the sludge chute (see Figure 10). The waltz are operating in sequences, where the sludge is removed when the waltz is in operating mode. It was decided to ensure that sludge was removed from both basins to achieve a sample that is representative for the entire line 5. Therefore, the settings were adjusted to ensure that both waltzes were operating simultaneously. When the waltz is operating, the sludge is filled into the sludge chute and for a certain height, the pumps in the bottom of the chutes are started, which removes the sludge through pipes to the sludge tank. To ease the sampling and ensure that the sample would contain sludge from the entire sludge blanket, a single operation sequence was desired. Therefore, the pump was run manually for the first collective sample. The sludge was collected in the chute for the entire operating time of the waltz while the pump was turned off. After the sludge had been collected, the pump was run for 1 minute until all sludge had been pumped away. During the pumping, sludge was collected at the pipe outlet in 1 L glass bottles at the beginning, middle, and the end of the sludge removal. The sub-samples were mixed into one sample and were sent off to Eurofins.

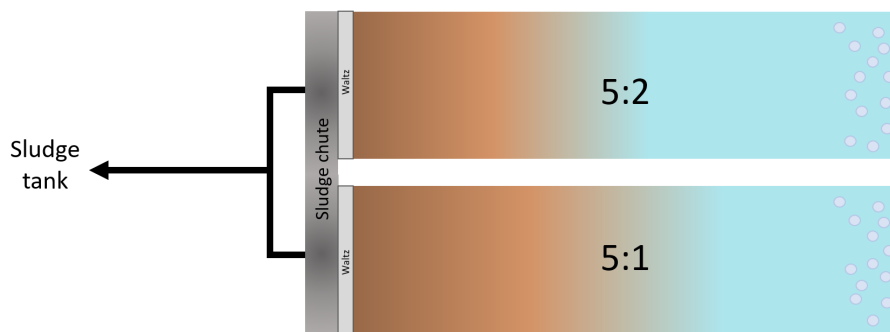


Figure 10: Schematic over line 5 in Görvålnverket from a birds-eye-view.

The procedure was changed for the following samples to improve the efficiency regarding the number of people needed during the sampling. Therefore, the start level for the pump was changed to a higher height, which meant that the chute was automatically filled to a higher level before the pump started and one continuous pumping sequence could be achieved. However, for some samples, the pump stopped before the waltz and the sludge level did not reach the start level for the pump in the remaining time. Therefore, some of the sludge was left in the chute until the next sludge removal point. During the time when the pump ran, sludge was collected at the pipe outlet in a 1 L glass bottle, where the bottle was filled every 20 s until the pump stopped. The sludge was mixed and a sub-sample of 500 mL was collected. All sub-samples were mixed, at the end of each day to obtain the final collective sample, which was sent off to Eurofins.

To examine the separation of PFAS4 in relation to the total dispersion quantity, three flow levels in line 5 were investigated. During the first 3 samples the maximal flow (490 l/s) was set, for the 3 following samples normal flow during the day was used and for the last 3 samples the minimal flow (310 l/s) was set. A time schedule, settings, and deviations to the experimental procedure are presented in Table 2. The flow into line 5 was possible to keep constant for the entire sampling period for some samples. However, on some days the flow varied significantly due to other operations in the waterworks. To achieve a sample that could be representative for a single flow, the sampling was adjusted to when the wanted flow was applied. Therefore, the flow is presented in three different ways, either as a mean flow during the entire sampling period, as the mean flow at the sampling points, or as the specific flow at the sampling (see Appendix A). Thus, the flows in the table are the flows that are representative for the measurement results.

It was decided to analyze the samples with Eurofins analysis package PLWAZ containing analysis of PFAS4 with a detection limit of 0.1 ng/L and an uncertainty of 39%. Another alternative that was under consideration was a similar analysis package with a detection limit of 0.3 ng/L and an uncertainty of 29%. However, since previous studies have indicated that the concentration of PFNA usually is closely below 0.3 ng/L, it was assumed that a larger uncertainty may be acceptable if the PFNA concentration could be measured and included in the results.

Table 2: Sampling specifications.

Date	Time	Inlet flow [l/s]	Waltz settings	Comment
2023-02-14	10:00-14:00	520 ^a	Operation: 8 min Pause: 4 min	Sludge: 4 sub-samples Raw water & Outflow: 5 sub-samples. Manual operation of the pump.
2023-02-15	10:00-15:00	480 ^a	Operation: 7 min Pause: 4 min	1 st sub-sample at the manual operation of the pump.
2023-02-17	10:00-15:30	470 ^a	Operation: 7 min Pause: 4 min	No sample at 13:00.
2023-02-21	09:30-15:00	360-380 ^b	Operation: 7 min Pause: 4 min	5 sub-samples. Varying flow during the day.
2023-02-22	08:55-14:15	370-410 ^b	Operation: 7 min Pause: 4 min	
2023-02-24	08:30-14:15	380/440 ^c	Operation: 7 min Pause: 4 min	No sampling at 12:30-13:30 due to low flow.
2023-03-07	10:50-16:00	310 ^a	Operation: 7 min Pause: 4 min	5 sub-samples. Changed start level to 1.6 for the pump for 3 sub-samples due to lower flow of sludge. Irregular sludge flow during several sub-samples.
2023-03-08	09:30-15:00	315 ^a	Operation: 7 min Pause: 4 min	5 sub-samples, no sampling 11:00-13:00. Changed start level for the pump to 1.7 for 2 sub-samples. Irregular sludge flow during several sub-samples.
2023-03-09	09:00-14:00	320 ^a	Operation: 7 min Pause: 4 min	Changed start level for the pump to 1.7. Irregular sludge flow during several sub-samples.

^a Mean flow during the sampling period.

^b Measured volumetric flow of sludge at the sampling point.

^c Four samples were taken at a mean flow of 380 l/s and two samples at 435 l/s.

3.3.2 Centrifuging for Analysis of PFAS on Solid Particles

To assess if PFAS is present in the water phase or on the solid particles in the DAF sludge, an attempt was made to centrifuge the sludge to achieve a phase separation and enable a separate analysis. An initial sludge sample was collected from the sludge chute, which was centrifuged in four 750 mL plastic beakers with a Multifuge 3 s-r/3 l-r, legend rt 120v (see settings in Appendix B). Because of the maximum allowed rotation speed for the beaker, the rotation speed was set to 4000 rpm.

It was observed that the suspension had begun to settle before centrifuging. After 15 minutes of centrifuging, the majority of the solid particles had collected at the bottom, but the clear phase still contained small flocks and had a yellow color. Therefore, the centrifuging ran for another 35

minutes when a complete separation had been achieved.

Due to the successful phase separation in the first centrifuging attempt, collective samples were gathered to send the separated phases to Eurofins for analysis. Initially, 3 L of the sludge collected on 2023-03-09 was kept overnight for centrifuging. The 750 mL beakers were filled with sludge and centrifuged for 60 minutes. It was observed that the separation was not complete as flocks could be recognized in the clear phase. Therefore, the sludge was centrifuged for another 60 minutes. No difference could be observed, thus it was decided to cancel the centrifuging, about 600 mL of the clear phase was removed from each beaker and was collected for analysis at Eurofins. The remaining sludge was then gathered into one beaker that was filled to the top with more fresh sludge. The centrifuge ran for another 60 minutes, where the same poor separation was observed. As much clear phase as possible was removed without losing any large particles of solids. The solids were dried in an oven at 35 °C for three days and sent off to Eurofins for analysis. At Eurofins, a weighting of 1 g was chosen for analysis, which resulted in a higher than expected detection limit of 0.36 $\mu\text{g}/\text{kg}$ TS, which corresponds to approximately 0.8 ng/l sludge (see calculation in Appendix B).

To confirm the analysis results, two grab samples were taken on 2023-03-15 within one hour apart. The flow into line 5 was 470 l/s during the sampling. The grab samples were taken during an entire pumping sequence and should therefore be applicable for all sludge. It was assumed that collective samples would not be necessary as it was assumed that the distribution of PFAS between water and solid particles is constant. The samples were centrifuged in a similar manner as the previous sample but with differing operating times to investigate if a satisfactory phase separation could be achieved. The first grab sample had an initial operating time of 3 h, where a poor separation was observed. After removal of the clear phase, transferring all solid sludge to one beaker, and refilling with fresh sludge, an operation time of 3 h where set before the sludge was dried at 35 °C for four days. The second grab sample was centrifuged overnight for 17 h, which yielded a better phase separation but not as satisfactory as in the assessment trial as small flocks and yellow color still could be observed. The secondary centrifuging consisted of 3 h and the sample was then dried for three days. Both the clear phase from the initial centrifugation and fresh sludge was sent off to Eurofins. For all centrifuged samples, the TS content in the sludge was calculated according to Section 3.4.2 *Total Solids Content in Sludge*.

3.4 Calculations

In this section, the calculations that were performed are explained. The calculations included removal efficiency, contact time, solid concentration for comparison with the sludge, sludge flow in NFWP, GAC reactivation, carbon emissions, costs, and energy consumption. All calculations that are time-dependent are determined on a yearly basis.

3.4.1 Removal Efficiency of PFAS4

The PFAS4 removal efficiency that was obtained by DAF was calculated for the collective samples gathered in line 5. The removal efficiency was decided based on the difference between the inlet and outlet concentration of PFAS4 according to Equation 1.

$$\text{Removal efficiency} = \left(1 - \frac{c_{out}}{c_{in}}\right) \cdot 100\% \quad (1)$$

where c_{out} is the PFAS4 concentration in the outlet stream from DAF (ng/L) and c_{in} is the PFAS4 concentration in the raw water intake (ng/L).

An attempt to verify the analysis results was made by constructing a mass balance over the DAF basins. Therefore, the concentration was recalculated to mass by multiplying the concentration with the average volumetric flow per day based on the flow that was measured during the sampling (see Equation 2). This was done for the raw water inlet, DAF sludge, and outlet from line 5. It was assumed that the volumetric flow for the raw water was identical to the outlet from DAF to only concern the fraction of water that entered line 5.

$$\dot{m} = c \cdot \dot{V} \quad (2)$$

The mass balance was constructed where the outlet mass of PFAS4 was calculated by the law of conservation of mass. The value from the mass balance was compared to the analytical results. To verify the results, the difference between the amount of entering and exiting PFAS4 was compared. The difference in percentages was also calculated according to Equation 3.

$$\frac{\dot{m}_{entering} - \dot{m}_{exiting}}{\dot{m}_{entering}} \quad (3)$$

where $\dot{m}_{entering}$ is the amount of PFAS in the raw water inlet (mg/d) and $\dot{m}_{exiting}$ is the sum of the amount of PFAS in the outlet and in the sludge (mg/d).

3.4.2 Total Solids Content in Sludge

The Total Solids (TS) content is an important parameter to understand the physical properties of a sample and is of great importance when investigating sludge management strategies. According to Hamilton and Zhang [48], the TS content should be determined by drying the sludge in an oven for 24 hours at 103 °C. Then the quota of the mass of the dried sample and the original mass gives the TS content in percentage according to Equation 17.

$$TS = \frac{\text{Mass of solids}}{\text{Original mass}} \cdot 100\% \quad (4)$$

However, after the sludge had been centrifuged the drying temperature was set to 37 °C to avoid interference with the PFAS content in the sample. This was decided after consultation with the analysis company Eurofins [49]. The difference of TS content at 103 and 60 °C has been estimated in previous literature to 2% [50], which can give an estimation of the possible error from the lower temperature used in this investigation.

3.4.3 Contact Time

The contact time to the air bubbles in DAF was calculated for both line 5 and NFWP. In the DAF basins, the contact time can be calculated from the contact volume, and the total volumetric flow, which is the sum of the water flow and dispersion flow (see Equation 5).

$$CT = \frac{V_{contact}}{\dot{V}_{tot}} = \frac{V_{contact}}{\dot{V}_{water} + \dot{V}_{disp}} \quad (5)$$

In the equation, CT is the contact time (s), $V_{contact}$ is the contact volume (m³), \dot{V}_{tot} is the total volumetric flow (m³/s), which is equal to the sum \dot{V}_{water} that is the volumetric flow of water entering the basin, and \dot{V}_{disp} that is the volumetric flow of dispersion air.

The basin can be divided into different zones, which have different impacts on the contact time. The zones are presented in Figure 11. The contact zone is the area where the air bubbles are introduced and PFAS can attach to the bubbles. The separation zone is the area where the bubbles are separated from the water and a sludge is formed. Just below the sludge blanket, it has been shown to be a thin layer of microbubbles [51].

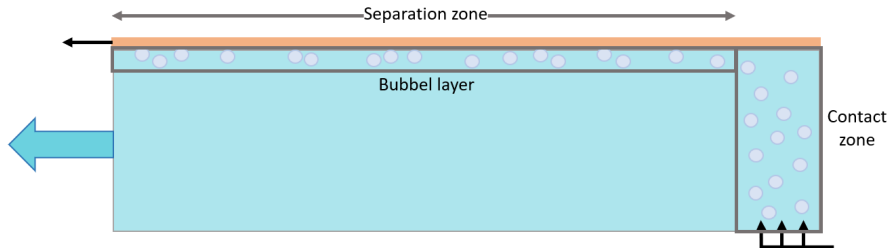


Figure 11: The different zones in the DAF basins.

The volume of the contact zone is determined from the area at the inlet for dissolved air in the basin multiplied by the distance between the inlet for the dissolved air to the water surface. From

the dimensions of the different basins that are analyzed in this project (see Table 3), the total contact volume could be calculated [51]. The dimensions are gathered from Lundh et al. who stated that there is a slope in one of the walls in the DAF basin in line 5 leading to an interval of the dimensions [23]. In Table 3, the flows are specified in both liters per second and cubic meters per day as both units are presented in this report in different calculations.

Table 3: Dimensions and volumetric flows in the different basins when assuming 44 running basins in NFWP.

Parameter	Line 5	NFWP
Dimensions contact zone [m]	0.78-1 x 8.6 x 2.4	0.55 x 5.15 x 3
Area DAF basin [m]	16 x 8.6	8.35 x 5.15
Minimum flow [l/s]/[m ³ /d]	310 / 26 784	34 / 2951 *
Average flow [l/s]/[m ³ /d]	400 / 1400	53 / 4561 *
Maximum flow [l/s]/[m ³ /d]	490 / 42 336	72 / 6205 *
Dispersion amount	44 l/s	10%

* Dimensioned values.

It was assumed that there is an additional horizontal contact time due to the horizontal flow through the layer of bubble water below the sludge. The presence of bubbles in the water should affect the reaction time and is therefore included in the contact time. It was estimated by Lundh et al. [23], that the thickness of the bubble water layer is about 25 cm over the entire basin. For the planned basins in NFWP the water will gradually drain from the surface into the sand filters leading to a variation of the contact time with the bubble layer. Therefore, a mean horizontal contact time for the entire water flow was estimated as half of the longest contact time possible. The horizontal contact time is calculated in a similar manner as the vertical contact time. The contact volume will correspond to the volume for the layer over the entire basin with the area above the nozzles excluded (see Appendix C).

3.4.4 Sludge Flow in Norrvatten's Future Water Production

The amount of sludge in NFWP in 2050 was estimated by determining the amount of sludge that is produced depending on the inlet flow of water. In the first attempt, the average flow of incoming water and outgoing sludge was determined based on data from the collection tool at Norrvatten called Acurve. For line 5, the flows were gathered for a time period of 35 days when DAF was in operation. By assuming a linear relationship between the amount of incoming water and sludge production, the volume of sludge was calculated for each line in NFWP assuming 44 running flotation basins and the average flow presented in Table 3.

In a second calculation, the incoming flows and sludge flow from the date 2023-03-14 were concerned when all five lines ran in DAF mode simultaneously. This calculation was implemented to investigate the applicability of the sludge flow in other basins than specifically line 5. Based on the same mean flow for 2050 and a linear relationship between the incoming water flow and sludge flow, the sludge flow was calculated. Lastly, to establish a linear relationship between the inflow and production of sludge, the proportion of sludge was calculated for each sludge flow that was obtained during the sampling for the different flow levels into the basin.

3.4.5 Evaluation of Dissolved Air Flotation

After the identification of a sludge management route, the process in NFWP could be analyzed regarding the favorability to include DAF. First, the reactivation frequency of the activated carbon filters was calculated for different scenarios that regarded the treatment targets, two reactivation time consumptions, and the choice to include DAF. The reactivation calculation was based on data from a pilot investigation performed at Görvålverket that evaluated the removal efficiency of different activated carbon materials. Thereafter, the energy requirement was calculated, where it was assumed that most of the power could be assumed to be equal for the processes except for the additional energy to run DAF. The energy requirement of DAF was based on a measurement performed in Görvålverkets line 5, which was linearly scaled for the inlet flows in NFWP.

Operational costs were determined for the GAC filters, DAF, sludge management, and transportation. The calculations included costs for reactivation at Chemviron in Belgium, where the cost

included both the cost of reactivation and the addition of fresh GAC due to losses. The energy price was gathered from the Swedish energy authority for the year 2022. The cost for transportation was determined for the GAC filters to Chemviron, for purchase of AIX material, and transport for used AIX materials and sludge to incineration. The cost for transportation to incineration was gathered from the transport company Foria, which was based on time consumption resulting in a need for an estimate of the time that is required to drive, fill, and empty the truck. The other costs of transportation were collected from the manufacturer of AIX and Chemviron. Lastly, the present currency exchange was used to recalculate the international costs stated in Euros.

Estimations of the carbon emissions were calculated in amounts of carbon dioxide per year. The calculation included transportation and reactivation as it was assumed that the energy consumption consists of 100% fossil-free energy and does not yield any net emissions. The emissions of transportation were estimated based on data from The Swedish Transport Administration for either a small truck or a truck with a trailer depending on the amounts that required transport. In addition, the distance of the transports was determined through Google Maps. The emissions from reactivation were based on data from the literature, which included the emissions for both reactivation and the addition of fresh GAC. Also, the amount of PFAS being destructed in the two process options was calculated based on the exiting streams from the process that contained PFAS.

Lastly, a sensitivity analysis was performed as the calculations performed to compare GAC with the process including DAF involve many uncertainties and assumptions. Therefore, an investigation was made regarding the influence of different parameters on the results. It was decided to perform the sensitivity analysis based on reasonable fluctuations of the most influential factors. It would be of interest to investigate what parameter that could have the largest impact on the results as well by changing the factors equally. However, due to limitations in time, it was decided to base the sensitivity analysis on real scenarios instead. In addition, as this project aimed to investigate the potential of DAF, it was considered that reasonable fluctuations would be more valuable than the effects of the different parameters contra each other.

3.4.6 Activated Carbon Filter Reactivation Frequency

The reactivation frequency of GAC in the water treatment process was calculated based on the method presented by Lundh et al. [45] that utilizes a saturation curve for a specific GAC material. The calculations were performed for two PFAS4 purification targets of 4 ng/L and 3 ng/L to illustrate the difference if Norrvatten aims to reach towards the limit set by the Swedish food agency or if Norrvatten decides to have a margin to the limit due to expectations on more strict regulations in the future. The calculations were made for both the process including and excluding DAF.

The plant in NFWP is currently planned to have eight groups of carbon filters, where each group consists of four GAC filters. The water from two groups is then mixed in a mixing point before the next step in the process (see Figure 12). To ensure a PFAS4 concentration below the limit value in the drinking water, the concentration of PFAS4 in the mixing point needs to achieve the target [45]. Therefore, the calculation was based on the concentration after mixing the flow through eight parallel filters.

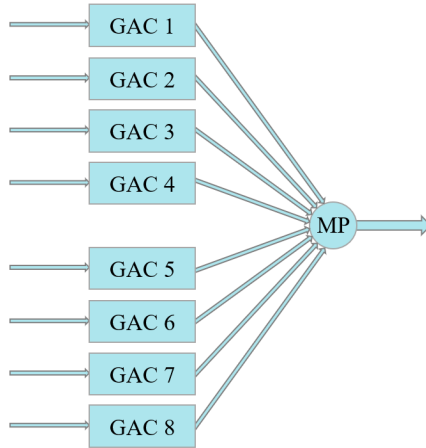


Figure 12: Flow through GAC filters in NFWP. 8 filters correspond to 2 lines, MP is the mixing point

A redundancy of N-5 was applied, which means that five filters can be out of operation simultaneously without affecting the treatment. It will allow for, for example, two GAC filters to fail, two filters that are being maintained, and one filter that is reactivated in the entire process. Therefore, when a filter is taken out of operation, the water is divided into the remaining filters, which need to provide sufficient PFAS removal to clean the water to the regulated concentration. The redundancy was simulated by having one of the eight GAC filters put out of service in the calculations to illustrate repair or maintenance.

The calculations were also made based on two alternative time consumption scenarios for exchanging old GAC with reactivated GAC. The reactivation was assumed to take approximately 1-1.5 months based on experience from Norrvatten and Uppsala Vatten, where the time consumption for each activity are gathered from Forsberg [42] and is summarized in Table 4. The lead time for the supplier corresponds to the time it takes for the supplier to reactivate the GAC and transport it back to Görvålverket. This time consumption can be reduced to zero if Norrvatten has an additional batch of carbon at either Görvålverket or at the supplier that can be added immediately after emptying the used GAC. Therefore, calculations were made on both a total of 1 month and 1.5 months to illustrate two alternative courses of action regarding the GAC exchange.

Table 4: Time consumption during the exchange of used GAC.

Activity	Number of days
Shutdown and emptying	5 days
Lead time before collection by supplier	5 days
Supplier lead time	14 days
Commissioning backwash	2 days
Commissioning 1st filtration	14 days

Norrvatten has executed a pilot investigation on the efficiency of different activated carbon materials from different manufacturers at Görvålverket since 2021. The results from the pilot trial were used to construct a saturation curve that could be utilized in the GAC reactivation calculations. The pilot investigation included various GAC materials including Norit 1240, Norit 830, Filtrasorb 300, Filtrasorb 300HA, and Filtrasorb 400. The results suggested that Filtrasorb 400 had the best reduction capacity after the largest number of bed volumes that had been passed through the GAC filter. As Filtrasorb 400 also has a reputation of being a highly productive material it was decided to estimate the GAC reactivation frequency based on that material. Filtrasorb 400 was operated with an EBCT of 20 min, which is the same EBCT that will be applied in NFWP [52].

The results from the pilot investigation presented the degree of reduction for PFAS4 for a number of different bed volumes. The results need to be scaled up to be applicable to the flow and bed volume that will be applied in NFWP. From the dimensioning of NFWP, the GAC filter bed volume has been decided to be 102 m³ in NFWP. By recalculating the number of bed volumes that correspond to a specific saturation state of the GAC filter to time consumption in NFWP, a

saturation curve could be gathered. The amount of water that is passed through the GAC filter for a certain amount of bed volumes could be determined. By dividing the amount of water with the average flow through each filter in NFWP, the time consumption from the application of fresh GAC, which corresponds to a certain degree of reduction could be determined (see Equation 6). The time was plotted against the degree of reduction and a logarithmic curve was fitted to the points in Excel. By extrapolating the fitted curve, the calculations on the regeneration frequency could be performed.

$$\Delta t(NFWP) = \frac{\Delta BV(pilot) \cdot V_{BV}(NFWP)}{\dot{V}(NFWP)} \quad (6)$$

In the equation $\Delta t(NFWP)$ is the time consumption that corresponds to a specific saturation state (months), $\Delta BV(pilot)$ is the number of bed volumes that were measured in the pilot trial for a specific saturation state, $V_{BV}(NFWP)$ is the bed volume (m^3), $\dot{V}(NFWP)$ is the flow to the filters in NFWP ($m^3/month$).

In Excel, 7 parallel GAC filters are assigned to the degree of separation from the saturation curve for the current month. The eighth filter was assumed to have no flow of water to it as it was simulated to be out of service. The outlet concentration for each filter was determined for a time period of 2 years based on the saturation curve and the highest PFAS4 concentration in the raw water during the last 5 years (5.7 ng/L) (see Equation 7). By applying the highest concentration, no margin to the limit should be required. The average of all outlet concentrations except for the eighth filter were calculated, which will correspond to the concentration in the mixing point as it is assumed that the flow will be equal into each filter (see Equation 8).

$$c_{out}(n) = DoS(\Delta t) \cdot c_{in} \quad (7)$$

$$c_{MP} = \sum_{i=1}^n \frac{1}{n} \cdot c_{out}(n) \quad (8)$$

$c_{out}(n)$ is the concentration out from one filter (ng/L), DoS is the degree of separation for that filter which is dependent on the time the GAC filter has been in operation (Δt), $c_{in}(n)$ is the incoming concentration (ng/L) to the filter, c_{MP} is the concentration in the mixing point and n is the number of filters that are connected to the mixing point except for the filter being maintained ($n: 1 \rightarrow 7$).

The reactivation was simulated continuously over a time period of two years. One of the seven GAC filters in operation was reactivated when the sum concentration in the mixing point reached towards the treatment target of either 4 or 3 ng/L. After a filter had been reactivated, the GAC filter would obtain a removal efficiency of 100% (see Figure 13). Thereafter, the filter is gradually saturated with PFAS, leading to a lower reduction efficiency. In Figure 13, the blue areas correspond to the reactivation time consumption when a filter is removed for reactivation.

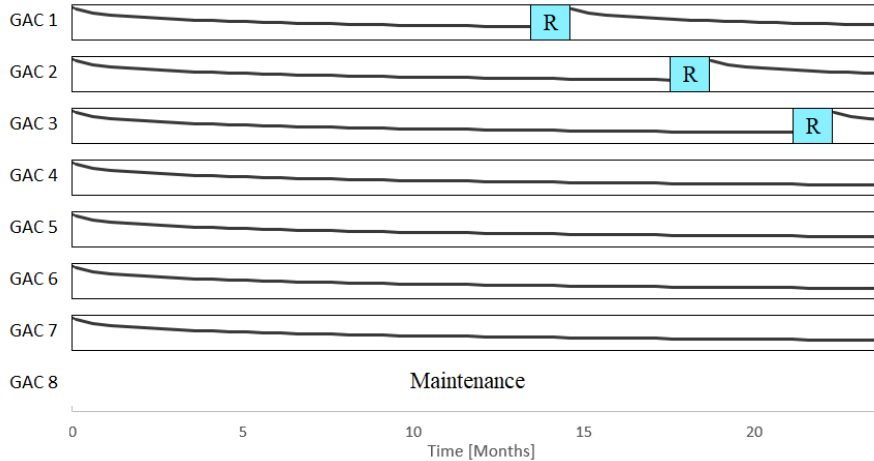


Figure 13: Illustration over how the GAC reactivation frequency was determined for 8 filters. The saturation curves show the removal efficiency of PFAS4 for a duration of 2 years after fresh GAC has been applied.

The yearly number of reactivations could be calculated by summarizing the blue areas over a time period where a repeated pattern could be detected, which could illustrate steady state. Then the time duration for the time period of steady state was identified where full cycles of reactivations were included. Full cycles mean the time period was set to begin with the initiation of reactivation and end just before another reactivation. The number of reactivations per year was calculated with Equation 9 to then be compared for each limit concentration, GAC exchange time scenario, and inclusion or exclusion of DAF.

$$\textit{Reactivation frequency} = \frac{\textit{Amount of reactivations in steady state}}{\textit{Time duration in steady state}} \quad (9)$$

4 Experimental Results

In this section, the results from the experimental procedure are presented. First, the PFAS reduction efficiency that was measured in line 5 is shown and discussed. Secondly, the PFAS content on solid particles, TS content is presented, and the PFAS distribution between solid and aqueous phases. Lastly, the accuracy of the results was discussed based on the mass balance over the process and possible sources of errors.

4.1 PFAS Reduction with Flotation

The analyzed PFAS4 concentration in the raw water and outlet from line 5 is presented in Table 5. It is assumed that the PFAS content is constant in the process from the raw water inlet until the dissolved air addition, which has been confirmed through previous measurements in Görvånverket without DAF [53]. Thus, the difference in PFAS concentration will be due to the separation of the sludge produced from DAF.

Table 5: PFAS4 reduction efficiency in DAF basin 5.

Date of sampling	Raw water [ng/L]					Outlet DAF [ng/L]					Water flow
	PFOA	PFOS	PFNA	PFHxS	Sum	PFOA	PFOS	PFNA	PFHxS	Sum	
14-feb	1.2	2.2	0.27	1.3	5	1.1	1.4	0.2	1.1	3.8	High
15-feb	1.2	1.8	0.19	1.1	4.3	1.1	1.3	0.13	0.97	3.5	High
17-feb	1.3	2.2	0.25	1.4	5.2	1.1	1.5	0.25	1.2	4.1	High
Average	1.1	1.4	0.2	1.1	3.8	1.2	2.1	0.2	1.3	4.8	High
21-feb	1.1	2.4	0.25	1.3	5.1	1.1	1.4	0.17	1.1	3.8	Mean
22-feb	1.1	2.7	0.22	1.3	5.3	0.94	1.4	0.18	1	3.5	Mean
24-feb	1	2.2	0.31	1.2	4.7	0.97	1.4	0.2	1.1	3.7	Mean
Average	1.0	1.4	0.2	1.1	3.7	1.1	2.4	0.3	1.3	5.0	Mean
07-mar	1.1	2.4	0.22	1.1	4.8	0.98	0.9	0.14	0.94	3	Low
08-mar	1.2	2.9	0.31	1.1	5.5	0.98	1.2	0.14	1.1	3.4	Low
09-mar	1.1	2.3	0.26	1.1	4.8	1	1.4	0.16	1.2	3.8	Low
Average	1.0	1.2	0.1	1.1	3.4	1.1	2.5	0.3	1.1	5.0	Low
Total average	1.1	2.3	0.25	1.2	4.9	1.0	1.3	0.18	1.1	3.6	-

As can be observed in Table 5, the analyzed concentrations are relatively constant despite the measurement uncertainty of 39%. As mentioned previously, the PFAS4 concentration in the raw water has been measured to 3.6-5.7 ng/L, which is consistent with these results. Although the concentration difference between the raw water and outlet is within the measurement uncertainty, the lower outlet concentration for almost every PFAS could be taken as proof that PFAS is reduced in the process. As sifting, flocculation and sedimentation have no proven effect on PFAS in water, it can be assumed that the concentration difference is due to separation through DAF [54].

The measured outlet concentration of PFAS4 is below 4 ng/L for all measurements except the sample 2023-02-17, which indicates that DAF may be sufficient to reach the Swedish Food Agency’s limit value of 4 ng/L especially if optimizations of the DAF would be done. However, as the concentration is close to 4 ng/L in many cases, the analysis uncertainty might play a significant role in fully determining if DAF can sufficiently reduce PFAS. Therefore, from these measurements, it is assumed that DAF has to be combined with GAC filters to reach the limit of 4 ng/L.

For the samples taken on 2023-03-09, the PFHxS concentration appears to be higher in the outlet compared to the inlet. It is possible for formation of PFAS in the process as the water also contains some smaller PFAS chemicals, which can react to increase the concentrations in water treatment processes according to Baresel et al. [8]. However, when all measurements are considered it is more probable that the negative reduction is due to the measurement uncertainty in the analysis, which will be further discussed in 4.3 *PFAS Mass Balance*. There was no evidence of PFAS formation in the other samples.

The reduction that is achieved by DAF was calculated by Equation 1. The reduction efficiency for each sample is presented in Table 6. The calculated reduction of PFAS4 shows an efficiency between 20-40%.

Table 6: PFAS4 reduction efficiency in DAF basin 5.

Date of sampling	Reduction					Water flow
	PFOA	PFOS	PFNA	PFHxS	Sum	
14-feb	8%	36%	26%	15%	24%	High
15-feb	8%	28%	32%	12%	19%	High
17-feb	15%	32%	0%	14%	21%	High
Average	11%	32%	19%	14%	21%	High
21-feb	0%	42%	32%	45%	25%	Mean
22-feb	15%	48%	18%	23%	34%	Mean
24-feb	3%	36%	35%	8%	21%	Mean
Average	6%	42%	29%	16%	27%	Mean
07-mar	11%	63%	36%	15%	38%	Low
08-mar	18%	59%	55%	0%	38%	Low
09-mar	9%	39%	38%	-9%	21%	Low
Average	13%	53%	43%	2%	32%	Low
Total average	9%	41%	31%	11%	26%	-

When the result from each chemical was regarded, it was observed that PFOS and PFNA had a significantly larger reduction. As PFOS is a perfluoroalkyl sulfonic acid and PFNA is a the perfluoroalkyl carboxylic acid, the results do not illustrate any clear dependency between the removal efficiency and the functional group on the chemical. As PFNA has the lowest inlet concentration, the difference in removal efficiency between the chemicals cannot be explained by the concentration either. However, from the molecular structure of the compounds it can be observed that PFNA and PFOS have longer carbon chains compared to PFOA and PFHxS, therefore it is probable that PFAS reduction with DAF is dependent on the carbon chain length as also has been discussed by Baresel et al. [8].

When the results were analyzed, an increase in reduction depending on the inlet concentration could be observed. As illustrated in Figure 14, this is true, especially for PFOS. It is reasonable that the reduction would increase with the inlet concentration as the contact opportunities between the microbubbles and chemicals. When the contact increases, more PFAS can be removed as the chemicals follow the bubbles to the sludge and are separated from the water stream. However, for the remaining PFASs, it appears as the inlet concentration has been relatively constant but the removal efficiency varies. Therefore, some other parameters must have influenced the removal efficiency as well.

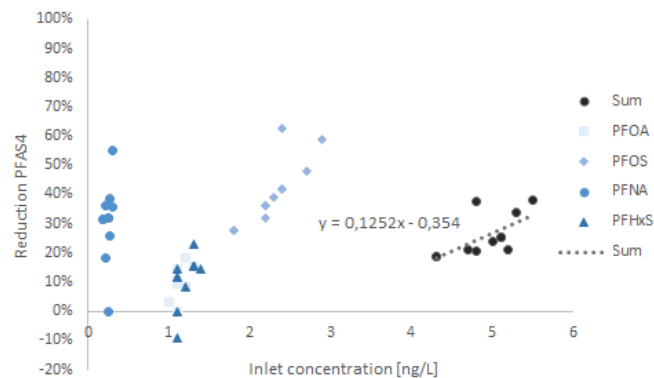


Figure 14: Reduction of PFAS4 depending on inlet concentration.

The amount of dispersion is significant for the removal of contaminants with DAF, where a higher amount of dissolved air has shown to be beneficial [22]. As the dispersion amount to the basin is constant, the percentage of air will differ depending on the flow of water. For line 5, the dispersion

amount will differ between 9-14% for the flows that were applied during the investigation. In addition, with increasing flow, the contact time decreases as the water moves faster. Therefore, a higher flow should decrease the reduction efficiency.

In Figure 15, the average PFAS removal for the analyzed water flows is illustrated. It can be observed that the removal efficiency of PFAS4 decreased with increasing flow, which is reasonable as the percentage of added air and the contact time decreased. As the contact time and amount of dispersion addition have not been varied separately it is impossible to draw a conclusion about what factor affected the results. In addition, when the four chemicals were analyzed separately it was observed that PFOA and PFHxS do not show the same pattern. For those chemicals, the results are not consistently increasing or decreasing with the flow. It is possible that the lack of pattern could be due to measurement uncertainty as the removal of those substances was small. However, it is possible to conclude that a higher dispersion amount and/or a longer contact time increases the removal efficiency for PFOS and PFNA.

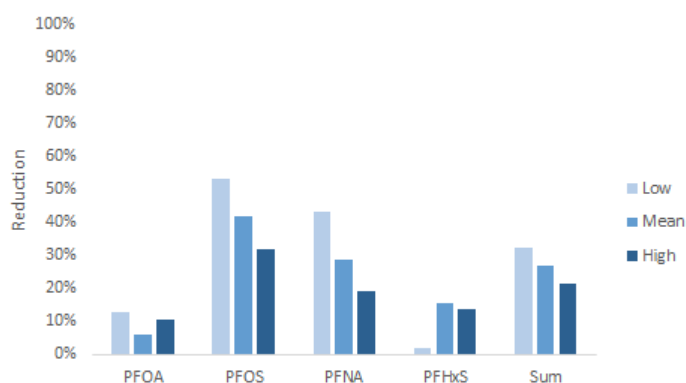


Figure 15: Average PFAS reduction for different water flows.

To compare the results with other studies on DAF, the amount of PFAS removed per amount of air is an important parameter. 44 l water per second was saturated with air at 5-6 bar. The solubility of air can be determined by Henry's law to 0.114 g/kg at 5 bar and 25 °C [55]. The water temperature is usually around 10 °C according to Acurve, which will lead to a lower amount of dissolved air compared to the value at 25 °C. However, due to the sources of errors in the measurement, it can be assumed to be an acceptable estimate. By dividing the average amount of PFAS4 per second by the amount of air per second, it was determined that 113 ng PFAS4 is separated per g of air.

The contact time can affect the PFAS removal efficiency as have been presented in the background. As these results should provide an estimation of the efficiency of DAF in NFWP, the difference in the contact time between Görvålnverket and NFWP is relevant to investigate. Therefore, the contact time was calculated for both line 5 in Görvålnverket and for NFWP based on the current process dimensions. It was found that the total contact time is significantly larger in NFWP compared to Görvålnverket. The vertical contact time was increased approximately 4-5 times, while the horizontal contact time increased approximately by a factor of 1.5.

Table 7: Contact times for minimum, maximum, and additional investigated flows.

Parameter	Line 5			NFWP	
	Max	400 l/s	Min	Max	Min
Vertical contact time [s]	34	41	51	108	250
Horizontal contact time [s]	51	77	96	64	148
Total contact time [s]	98	118	147	172	398

4.2 PFAS Concentration in Sludge

The PFAS concentration in the DAF sludge is presented in Table 8. The analysis cannot be made on a mixture of a liquid and solid particles because of analysis equipment limitations. Therefore, the samples were either centrifuged or decanted to remove the solids before analysis. Thus, these

results only reflect the concentration in the liquid phase. From these results, the total PFAS4 content cannot be decided as the solid particles are excluded.

Table 8: Analysis results on sludge from DAF basin 5.

Date of sampling	Sludge [ng/L]					Water flow	Sludge flow [l/s]
	PFOA	PFOS	PFNA	PFHxS	Sum		
14-feb	12	183	14	7.4	220	High	2.4
15-feb	23	320	21	14	380	High	2.3
15-feb Reanalysis	17	220	15	10	260	High	2.3
17-feb	≤10 *	130 *	10 *	≤10 *	140 *	High	2.3
Average	17	241	17	10	297	High	2.3
21-feb	37	606	44	23	710	Mean	1.8
21-feb Reanalysis	33	505	35	17	590	Mean	1.8
22-feb	20	193	16	12	240	Mean	1.8
24-feb	24	240	24	13	300	Mean	1.9
Average	29	386	30	16	460	Mean	1.8
07-mar	49	433	37	23	540	Low	1.6
08-mar	42	518	43	22	630	Low	1.4
09-mar	43	489	38	20	590	Low	1.3
Average	45	480	39	22	587	Low	1.4

* Not centrifuged before analysis giving a higher detection limit. Excluded in the average value.

The PFAS4 concentration varies by about 500 ng/L between the samples, which cannot only be explained by an analysis uncertainty of 39%. The sample that was gathered on 2023-02-17 showed a significantly lower concentration compared to all other measurements. That sample was not possible to concentrate before the analysis, which resulted in a higher limit of quantification at 10 ng/L. During the previous measurements performed in line 1, it was observed that the reported PFAS concentrations for unconcentrated samples were lower than for concentrated samples [53]. As the same pattern was illustrated in this investigation, it was decided to disregard that result. The issue was discussed with the laboratory but no explanation for the lower concentrations could be identified.

It was decided to reanalyze the samples gathered on 2023-02-15 and 2023-02-21 as the concentrations of especially PFOS and PFNA were significantly higher than expected. The results appeared to be more reasonable for the new analysis where the results differed by 14-44 % compared to the initial results depending on how the measurement uncertainty is applied. According to Torbjörn Synnerdahl [56], the measurement uncertainty rapidly decreases for higher concentrations, where the uncertainty is 39% for 0.1 ng/L and 29% for 0.3 ng/L. As the PFAS concentrations in the sludge are in the range of 10-600 ng/L, it would be expected to have significantly lower uncertainty and more consistent results. It was discussed with the laboratory if the matrix in the sludge samples could affect the analysis results, but the sludge is similar to the water samples except for the solids that are removed before analysis. The laboratory could not identify any deviations chromatographically or analytically, where the chromatograms were without disturbances, interferences, and linearity and the control samples were without remarks.

The sample 2023-02-14 was gathered at a different waltz setting with a break time of 8 min instead of 7 min as for the other samples. It can be observed that the concentration is lower compared to the 2023-02-15 value, which was taken at a similar flow. This suggests that when the sludge has a longer residence time on the water surface the concentration decreases, which may be due to PFAS re-entering the water. However, the different waltz settings were only present for that single sample. When regarding the measurement uncertainty and other sources of errors that could have affected the sample, it was decided that no conclusion about the waltz setting's impact on the results could be drawn from these measurements. Continuous studies on this parameter are of relevance to investigate in continuous studies to optimize the PFAS removal efficiency in the DAF basins.

Because of the absence of a sampling tap, it was decided to collect sludge at the run-off to the sludge tank. It was assumed that the sludge flow was constant throughout the entire pumping

sequence as sludge was collected continuously for the entire pumping time period. However, when observing the flow sheets in Appendix A, it was distinguished that the sludge flow varied throughout the pumping sequence. This could also be observed during the collection of sludge as the flow was irregular during the sampling. This would have led to that the sample did contain unrepresentative amounts of sludge compared to the flow. The sludge may differ in concentration throughout the sludge blanket, which could have affected the accuracy of the measurements.

The water flow and PFAS concentration in line 5 were investigated for all measurements except for the uncentrifuged sample and for the initial analysis of the samples that were analyzed twice. Figure 16 illustrates that the highest PFAS concentrations were found for the lowest flows, and the lowest concentrations were found for the highest flow. This would be expected since a lower flow means a higher proportion of dispersion and a longer contact time, yielding a better separation. As the sludge waltz is set to a specific height, the water surface height will determine the amount of water that is removed with the sludge. At a lower flow, the water surface will be lower, meaning that less water will be removed, and the sludge will be more concentrated. It would be expected that the sludge concentration would be lower in the samples taken at high flow as the sludge becomes more diluted. In addition, at high flow, a lower PFAS separation is expected as the contact time and the proportion of air to water is decreased. A pattern was identified where the lowest PFAS concentrations are obtained at high flows and the highest concentrations are at low flows. However, there are some deviations from the pattern especially for the samples gathered on 2023-02-15 and 2023-02-21.

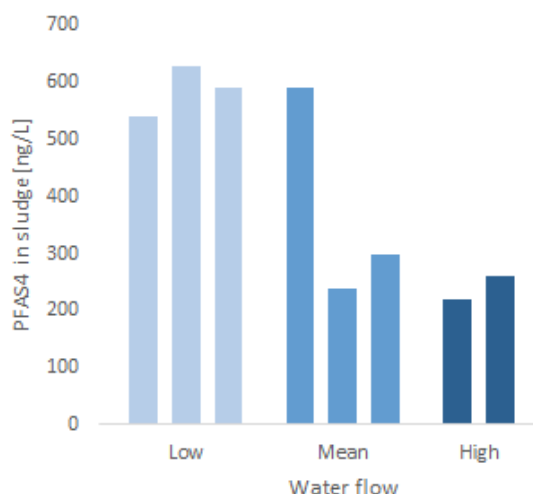


Figure 16: PFAS4 concentration in sludge depending on the flow of water.

The results for the mean flow in Figure 16 do not follow the expected pattern with widely varying concentrations. In Table 8 it can be observed that the sludge flow is equally large for all mean samples, which excludes the explanation that the sludge could be differently diluted. The sample gathered on 2023-02-21 received a significantly higher concentration than the rest of the samples gathered at the mean flow for both the initial and re-analysis. There is no known aspect that differed for that sample compared to the other samples. It may be possible that more sludge contra water was collected for that sample as the flow of sludge was poorly mixed. However, as the sample consisted of several sub-samples the probability of that explanation is small. Therefore, no answer for the differing results could be identified.

The solid particles in the sludge were separated from the sludge to enable analysis of their PFAS content. About 2.5-3 L sludge was centrifuged for three samples, which yielded approximately 6-7 g of completely dry particles. Thus, the samples had a TS content of 0.22%, 0.25%, and 0.23% respectively (see Appendix D). The analysis at Eurofins showed that the samples were 100% dry. Therefore, the TS content should be reliable despite the low temperature used during drying. The analysis results were presented in $\mu\text{g}/\text{kg}$ TS, which was recalculated to ng/L to enable comparison with the liquid samples by multiplying the results with the TS content in the centrifuged volume. The exact centrifuged volume of sludge was determined from the mass and on the assumption that

the sludge had the same density as water.

The PFAS content on the solid particles is presented in Table 9. The limit of quantification differs because different weights of solids were used in the analysis. A weight of 20 g would have yielded the optimal limit of quantification. However, the sludge was difficult to centrifuge with a low content of solid particles and the work was limited in time, only about 7 g could be produced for each sample. The samples were dried at low temperatures to avoid evaporation of PFAS from the centrifuged sludge. Thus, the PFAS that was in the liquid phase in the centrifuged sludge will have been included in the analysis.

Table 9: PFAS concentration on solid particles.

Sample	$\mu\text{g}/\text{kg TS}$					ng/L sludge				
	PFOA	PFOS	PFNA	PFHxS	Sum	PFOA	PFOS	PFNA	PFHxS	Sum
Collective 9/3	<0.36	4.4	<0.36	<0.36	4.4	<0.8	9.8	<0.8	<0.8	9.8
Grab 15/3(10am)	0.11	2.4	0.094	<0.087	2.6	0.27	5.9	0.23	<0.21	6.4
Grab 15/3(11am)	0.12	2.7	0.12	<0.088	2.9	0.28	6.2	0.28	<0.2	6.7

All results are in the same order of magnitude where no result differs significantly when the measurement uncertainty of 23% for solid analysis is considered. However, the PFOS concentration is higher for the collective sample compared to the grab samples. There are many explanations for that difference beyond the measurement uncertainty. Another explanation could be that plastic in the centrifuge beakers would contain PFAS, it is reasonable that they would release most contaminants for the initial centrifuging. However, according to the manufacturer, there is no known PFAS content in the plastic, although no analysis has been performed for confirmation.

Another explanation for the difference between the collective and grab samples may be because the peaks in the chromatograms from the analysis are manually measured at Eurofins. Therefore, the human factor may explain some of the differences as it is reasonable that the grab samples were analyzed simultaneously as they were sent together. Lastly, to ensure a higher limit of quantification for the grab samples, the separated phases were kept for over one week until confirmation of a higher weighting for the grab samples was obtained. According to Lutz [57], it is possible that PFAS can attach to the walls of a beaker. Therefore, it is possible that PFAS was lost to the beaker, which could explain the lower values. It would be unlikely to have collected very differing amounts of sludge contra water as also the grab samples were gathered several times during one pumping sequence.

To investigate the presence of PFAS in the different phases after centrifuging, the water phase was sent for analysis. The PFAS concentration in the clear phase that is obtained after centrifugation can simulate the real situation where the sludge contains large amounts of water. To confirm that the centrifuging did not affect the phase PFAS was present in, both the uncentrifuged and centrifuged sludge was sent for analysis. The clear phase was collected from the first step of centrifugation and was sent to Eurofins. The concentrations are presented in Table 10.

Table 10: Concentration of PFAS4 in the clear phase after centrifuging and in uncentrifuged samples.

Sample	Uncentrifuged samples					Clear phase from centrifuging				
	PFOA	PFOS	PFNA	PFHxS	Sum	PFOA	PFOS	PFNA	PFHxS	Sum
Collective 9/3	37	550	38	20	650	37	350	28	17	430
Grab 15/3(10am)	18	160	15	9.9	200	15	150	15	9.2	190
Grab 15/3(11am)	19	190	15	9.7	230	16	220	18	9.3	260

No systematic difference between the clear phase and uncentrifuged samples could be identified. Therefore, it is possible to assume that the centrifugation did not affect the concentration in the clear phase by for example either releasing or attaching PFAS to the solids in the suspension. However, it can be observed that the concentration in the collective sample is significantly higher than the grab samples. The difference can be explained by the low flow in the collective sample and high flow in the grab samples, which have been shown to result in higher PFAS concentrations in the sludge at low flow.

The distribution of PFAS4 between the solid particles and the water phase could be calculated from the measurements, which are presented in Table 11. For all measurements that did not exceed the detection limit, the concentration is counted as zero. It can be seen that the majority of the PFAS is located in the aqueous phase with less than 3% on the solid phase.

Table 11: Distribution of PFAS4 between solid (So.) and water phase (Aq).

Sample	PFOA		PFOS		PFNA		PFHxS		Sum	
	Aq	So.	Aq	So.	Aq	So.	Aq	So.	Aq	So.
Collective 9/3	100%	0%	98%	2%	100%	0%	100%	0%	98%	2%
Grab 15/3(10am)	98%	2%	96%	4%	98%	2%	100%	0%	97%	3%
Grab 15/3(11am)	98%	2%	97%	3%	98%	2%	100%	0%	97%	3%

However, the results presented in the table represent the division between the aqueous phase and a centrifuged phase as the solid phase contained 30-60 g water in each sample. If another approach than centrifugation is utilized in NFWP the amount of PFAS on exclusively the solids could be of relevance. To determine the amount of PFAS on the solid particles, the amount of PFAS that dried onto the solids from the water present after centrifuging must be excluded. The concentration in the water phase was assumed to be the average measured concentration in the uncentrifuged sludge and the clear phase. Thus, it was necessary to subtract the amount of PFAS4 that the water phase should have contributed, from the total amount of PFAS in the solid samples to estimate the PFAS on the solid particles per liter of sludge (see Appendix E). The calculations gave -2.6 ng/L for *Collective 9/3*, 4.3 ng/L for *Grab 15/3(10am)* and 3.8 ng/L for *Grab 15/3(11am)*.

The negative concentration is probably because only PFOS was measured on the solids in the collective sample resulting in a lower PFAS4 concentration on the solids than in reality. From the results, it can be concluded that the major fraction of PFAS is present in the water phase. Approximately 0-5 ng/L appeared to be attached to the solids, which corresponds to approximately 2 $\mu\text{g}/\text{kg}$ TS. Despite measurement uncertainties, all three samples provided concentrations in the same order of magnitude, which implies that concluding a maximal PFAS concentration of a few ng per liter is reasonable.

4.3 PFAS Mass Balance

As previously mentioned, the PFAS concentrations differed significantly for the measurements on the initial sludge samples. Some differences were expected as the raw water concentration differs and the removal degree depends on the water flow in the basin. To better understand the results, mass balances were constructed over the basin, where it was assumed that the only streams that will carry PFAS4, or affect the PFAS4 concentration, are the inlet stream, outlet stream, and sludge stream. The mass balance was then evaluated in two ways: first by comparing the analysis results and mass balance result with an uncertainty of 39% and secondly by comparing the amount of PFAS entering and exiting the basin.

In Figure 17, the percentage difference between the analyzed value of the outlet stream and the calculated value from the mass balance is shown as the difference between the inlet and sludge. In cases where the bars exceed the 39% uncertainty, it indicates that allocating the entire uncertainty to one of the streams is inadequate to ensure result consistency. In that case, it can be assumed that the measurement uncertainty has a significant impact on the result. As all three streams could have an uncertainty of 39%, this only provides a first approximation of the accuracy in the measurement.

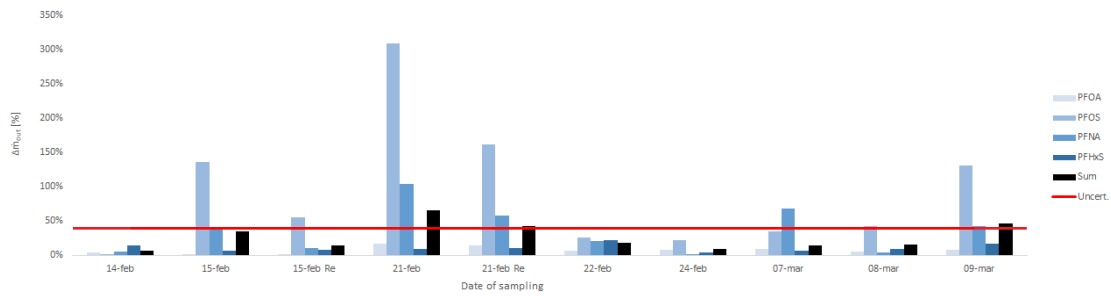


Figure 17: Percentage difference between mass balance and analysis results for the outlet. The red line corresponds to 39% measurement uncertainty.

As can be seen, 7 of 11 samples do not have consistent results when comparing the analysis data and mass balance despite 39% uncertainty. Especially the first analysis on the sample gathered 2023-02-21 had a tremendously large deviation of over 300%, where the difference had significantly improved in the reanalysis. Also, for the reanalysis of the sample 2023-02-15, the deviations between the mass balance and analysis result decreased greatly. This suggests that the reanalysis is more trustworthy than the initial results for both of those samples as the results better cohere with the mass balance. It would have been relevant to reanalyze the samples gathered in March as well as all results for PFOS exceeding the limit, which may suggest an issue with the calibration or some other unknown issue in the analysis.

In Figure 18, the amount of PFAS entering or exiting the basin through each stream is illustrated. It can be observed that when the sludge concentration is large, the amount of PFAS entering the basin is generally exceeded. In other words, when the sludge concentration is low the exiting amount of PFAS is generally lower or closer to the entering amount. When the sludge concentration increases, the exiting amount of PFAS is measured as significantly higher than the entering amount. The same relationship cannot be identified for the outlet concentration, as that concentration varies between the samples without leading to a systematic relationship between the existing and entering amounts of PFAS. This suggests that the reason that the mass balance does not add up is majorly due to sources of errors from the sludge concentration.

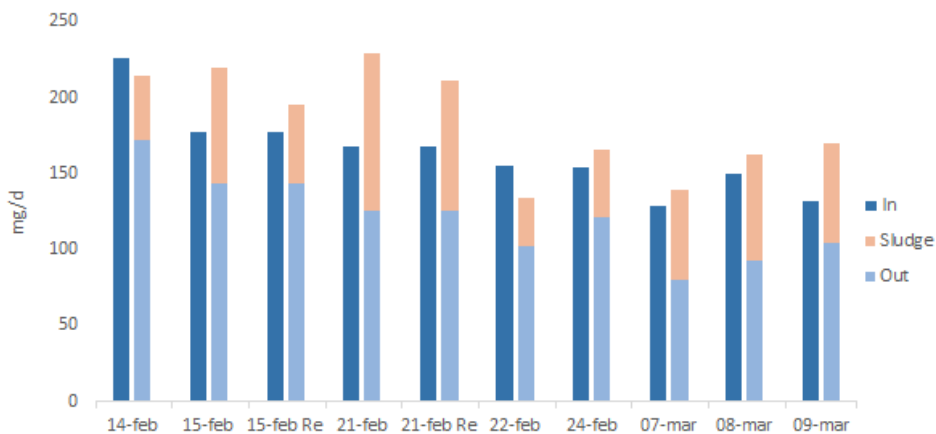


Figure 18: Difference in amount of PFAS entering or exiting by each stream.

An attempt was made to calculate the deviations between the inlet and the outlet in percentages, which is presented in Table 12. The error became significantly smaller than what was calculated in Figure 17. This is reasonable as the difference between the amount of PFAS in the outlet and the sludge differs, leading to the error in the sludge being lost when the higher amount in the outlet is applied. Therefore, the error is larger than what is presented in the table. As it is unknown how much the outlet and sludge stream contribute to the error, it is not possible to estimate the real error.

Table 12: Difference in amount of entering and exiting PFAS.

Difference in mass balance [mg/d]	Percentage difference in mass balance
12	5%
-43	-24%
16	8%
-61	-37%
20	13%
-11	-7%
-11	-8%
-13	-9%
-39	-30%

Therefore, the analysis was examined to identify possible sources of errors. Eurofins performed a second opinion on the chromatography to reduce the risk of human errors, which did not significantly change any results. Only the results from the second opinion were presented in this report as they should correlate more and have reduced human error. Also, they investigated deviations from internal standards in the form of interference and checked linearity, blank samples, and control samples to ensure the quality of each processing. No deviations or errors could be found, thus the explanation for the varying results and explanations for deviations from the mass balance are unidentified.

5 Literature Review of Management Technique for the Residual Streams

The sludge produced from DAF becomes waste that needs to be treated in a sustainable way [58]. The regulations on waterworks sludge are few as it does not have the same high priority as wastewater sludge by environmental authorities. However, it is assumed that the restrictions will become sharper in the future, which means that Norrvatten has to have a plan for how the sludge may be sustainably managed [59]. Generally for waste management the 5R principle (Reduce, Reprocess, Reuse, Recycle, and Recover) should be adopted to strive towards a circular economy [58, 60]. This is especially important with the increasing global population, lack of resources, and environmental pressure. Therefore, it is of large significance to identify the value of waste, implement a cradle-to-cradle perspective and apply industrial symbiosis [60]. On the contrary, increased material reuse challenges a non-toxic environmental cycle. Therefore, alternative disposals for hazardous waste are needed that provide a suitable treatment based on the hazardous characteristics of the waste [61].

One principle that can be implemented from the 5R is reducing the waste, as it is important to strive for a minimal amount of toxic sludge to obtain more manageable amounts to dispose of. Therefore, optimizations may be needed to reduce the water content in the sludge stream without increasing the PFAS concentration in the drinking water. Another relevant measure would be to reduce the amount of PFAS-contaminated sludge by separating the DAF sludge from the sedimentation sludge. Presently, the sludge from DAF is mixed with residual streams from the entire treatment process and is dewatered. The clear phase is returned to Mälaren and the solid phase is currently gathered by Biototal, which uses it for landfill cover or soil improvement [62]. However, in NFWP the sludge from DAF will be treated separately and thus contain high concentrations of PFAS [39].

Zhao et al. [58], presented possible waste management routes that could be useful for regular waterworks sludge. They stated that one resource that is relevant to reuse is the coagulants that are added to enhance the impurity removal efficiency in the treatment process. Usually, alum and ferric salts are used as coagulants in the treatment process, which produces sludge by the formation of agglomerated particles with impurities. It has been demonstrated that liquid waterworks sludge or dewatered sludge can be reused as a coagulant in the wastewater industry [58]. Furthermore, Zhao et al. [58], stated that the driving force in developing various reuse strategies for waterworks treatment sludge is the fact that most of the sludge is comparatively clean from harmful or toxic elements. The sludge produced from DAF contains high amounts of PFAS4 according to the measurements that have been done. Therefore, it is probable that waste with high amounts of PFAS should be treated as hazardous waste in the future, meaning that it should be sent off to destruction or landfill to minimize the risks of releasing the chemicals back into the environment [60]. Norrvatten has decided that no water can be returned to Mälaren that contains elevated levels of PFAS, which will require management of the water phase in NFWP.

Baresel et al. [8], discussed the various alternatives to destruct PFAS to protect the environment from returning PFASs. In their article, they show that thermal treatment has the possibility to reach a removal efficiency of 99% at 900 °C and 2 minutes operating time. Also, they present landfill as a hypothetical option if PFAS-contaminated leachate is managed [8]. In addition, foam fractionation was raised as an alternative at the SIDWater meeting by Philip McCleaf [63] and Anders Finnson at Svenskt Vatten [64] raised the possibility of transferring the sludge to a wastewater treatment plant. Other techniques such as chemical destruction techniques, including electrochemical treatment, sonolysis, and photolysis are known to be energy and time-consuming, have a hard-to-control efficiency, and have a high risk to form other PFASs as by-products. Also, PFAS degradation cannot be expected from biological degradation and supercritical wet oxidation has a very high cost exceeding 1000 SEK/m³. Baresel et al. [8], state that mechanochemical destruction has shown the potential to yield a high removal efficiency of PFOA and PFOS, which could be promising for sludge treatment but has a large knowledge gap due to a lack of research.

At Norrvatten, the waste management of sludge from the DAF can occur either before or after phase separation between the solids and water. Sludge treatment after phase separation would entail

additional energy requirements and operation costs due to centrifuging. However, it may bring new opportunities to treat the sludge if the phases are separated. In the following section, options for the sludge are presented. This involves that the entire sludge flow could possibly be treated with either foam fractionation or be transferred to a wastewater treatment plant. Conversely, the solid phase could be treated separately from the clear phase, where the solids could be incinerated, reused, or put in a landfill, while the clear phase could be treated with GAC, AIX, membrane, sonolysis, or oxidation.

5.1 Transfer to Wastewater Treatment Plant

One possible treatment route could be to transfer the sludge to a nearby wastewater treatment plant. Wastewater treatment plants receive PFASs from municipal wastewater, industrial wastewater, and stormwater leakage from contaminated land areas. Thereafter, the wastewater treatment plants function as a route of spread between the society and the environment unless the chemicals are removed from the water before it is released [8]. The treatment process at the closest wastewater treatment plant, Käppala, presently consists of a mechanical, biological and chemical treatment [65]. Those treatment techniques are overall inefficient for the removal of PFAS, which would make the transfer unsuitable as the PFAS would not be captured or destroyed [8].

Käppala has a need to extend their treatment process due to pharmaceutical residues according to their operating permit. To achieve a removal of pharmaceutical residues, ozone or activated carbon is considered [66]. This additional treatment could also function as a removal technique for PFAS. In addition, As Käppala is exposed to PFAS through several sources, it is possible that they will need to implement PFAS removal in the future due to increasing regulations. However, even if the wastewater treatment plant would have a PFAS treatment technology in place, it is uncertain if they would accept treating the sludge due to potentially high costs.

Baresel et al. [8], stated that Käppala currently receives 4.3 ng PFOA/L, 7.7 ng PFOS/L and 43.2 ng PFAS11/L. When Norrvatten implements PFAS removal in their process, the PFAS levels would decrease as the chemicals would be removed from the drinking water that later enters Käppala's process after usage. However, if the DAF sludge would be directed to Käppala, the concentrations should reach identical values as today since the PFAS only would take another route to the treatment facility. Therefore, the challenges Käppala is currently facing should not be increased in the future if the sludge is transferred to them if the issues with receiving a more concentrated stream could be managed [67].

5.2 Flotation

It may be possible to implement foam fractionation or a second DAF step to obtain a permeate that can be released to Mälaren. To estimate the efficiency of foam fractionation or DAF on the sludge, the literature study was made flotation applied on streams with higher PFAS concentrations. McCleaf et al. [28] used foam fractionation for NF concentrate with approximately 400 ng PFAS per L. They achieved a 90% reduction without co-surfactants and 94% reduction efficiency with a cationic co-surfactant. They used an airflow of 4 L/min, yielding a foam with over 3000 ng PFAS/L. Similarly, Robey et al. [29], achieved a removal efficiency of over 90% for PFAS4 by applying DAF on a stream of landfill leachate containing approximately 3300 ng PFAS4/L. Correspondingly, in another experiment performed by McCleaf et al. [30], where they applied foam fractionation on a stream with a concentration of approximately 6200 ng/L. They obtained a removal efficiency greater than 90% for PFAS4 except for PFNA, which had a removal of 50-80%. The similarities in the results from several studies implicate that foam fractionation or DAF could be an efficient alternative to concentrate sludge from DAF.

If foam fractionation would be applied to the DAF sludge in NFWP, the resulting concentration in the permeate would be about 30 ng/L based on the initial sludge concentration of 300 ng/L and a 90% removal efficiency, which is the most common efficiency presented in these research projects [28]. As mentioned, water is not allowed to be released to Mälaren if it contains more concentrated PFAS [39]. The raw water contains a concentration of 3.6-5.7 ng/L, thus the permeate from foam fractionation would require further cleaning before it could be released. As the permeate and foam would require separate treatment, it may be possible that foam fractionation cannot provide a

cost-efficient solution. However, further investigations are required before that could be concluded.

Envytech offers a solution where foam fractionation is applied in several steps, that goes by the name Surface Active Foam Fractionation (SAFF). The process utilizes air bubbles that pass through the water in a small water pillar that improves the removal efficiency. The concentrate is separated on the surface by a "spill over the edge" system and a vacuum system. They state that the water should be possible to be released to the recipient, while the concentrate continues to a similar fractionation step to further increase the concentration and decrease the volume. By removing the new concentrate with vacuum, a drier fraction can be obtained with a higher concentration. The concentrate goes through a final step of fractionation before the PFAS is destroyed with for example incineration, SCWO, or electrochemical oxidation. The final concentration reaches a concentration of 50 000 - 2 000 000 times the inlet concentration and the volumetric flow becomes less than 1-10 m^3 per 40 000 m^3 treated water depending on the characteristics of the water [68]. However, there have not been any investigations performed by this technique in this water making the estimations from the manufacturer uncertain.

5.3 Activated Carbon

The clear phase that would be produced after phase separation of the sludge will require treatment before it is released to Mälaren. The GAC adsorption technology has been widely investigated and is well established with an advantageous cost efficiency when removing PFOA and PFOS from water. Franke et al. [69], describes that GAC is the most frequently used removal technique to remove PFAS in drinking water treatment processes. A high removal efficiency has been reported in several studies, where a PFOA removal effectiveness of 99% could be achieved where the adsorption efficiency was even higher for PFOS. The PFOS removal efficiency is consistent through literature, which implies that GAC is a robust and reliable method to use. There may be a possibility to coordinate transports and reactivations with the treatment process in NFWP, which also will contain a GAC filter step. The EBCT for PFAS removal is typically at 10-20 minutes according to the American Water Works Association, which is relatively short.

Additionally, the spent GAC can be reactivated, which minimizes the amount of waste that is produced. The process entails exhaust carbon during production and thermal reactivation, but it can be argued the environmental benefits of removing PFAS extends the shortcomings in its environmental impact. However, GAC has a lower adsorption to short-chain PFAS probably due to a less favorable partitioning on the solid phase and as they have a lower hydrophobicity, which may be involved in the adsorption mechanism. Also, the selectivity in GAC is poor, which affects the reactivation requirement as other compounds than the desired ones may be adsorbed as well [70]. However, given the impending integration of GAC into the NFWP process, it may be worth considering its application for the clear phase as well, as it offers the opportunity for coordinated logistics and leverages the existing knowledge within the facility.

5.4 Ion Exchange

AIX has high efficiency due to the high affinity between the positively charged resins and negative PFAS ions. AIX has a higher removal efficiency of short-chain PFAS, smaller space requirements, and a shorter contact time compared to GAC [70]. According to Franke et al. [69], the removal efficiency varied between 42-97% for PFAS4. AIX is a cost-effective, and mature technology for water treatment with a small footprint. However, the possibility to be a satisfactory solution will depend on the AIX material as the cost and removal efficiency differs between the materials. AIX resins have been applied in several water and wastewater treatment processes for other purposes, however concerning PFAS it is not as extensively studied as GAC [70].

AIX possesses reactivation abilities after saturation, where an 80% reactivation efficiency could be achieved using NaCl. However, a highly saline aqueous solution with concentrated PFAS was obtained as waste that required additional destruction or disposal [70]. That issue must be solved unless it is decided to use the material one single time and thereafter dispose of it. Also, Yadav et al. [70] mentions that the adsorption is highly sensitive to several factors such as resin properties, pH, competing substances, and PFAS structure. The media has a higher initial cost than GAC. Lastly, Malovany et al. [71] mention that a common issue with AIX is clogging where the columns

are difficult to backwash due to the low density of the material and the small columns. Therefore, this supports the need to separate the solids before this method could be used on the sludge.

5.5 Membrane Processes

In recent times, membrane processes have been under consideration as they can function as both a chemical and microbiological barrier and have an exceptionally effective removal with NF and RO [69]. The membrane technology is mature and proven for wastewater treatment with high efficiency for all PFASs. Membrane processes have the highest proven removal efficiency according to Yadav et al. [70]. NF has demonstrated a PFAS removal efficiency of over 90%, which has been more efficient than RO at high concentrations. Fouling of NF membranes increases with higher initial flux and applied pressure, which may cause problems with the separation and disrupt the removal efficiency. On the contrary, the PFAS removal rate increases, which is assumed to be due to the formation of a cake on the membrane that functions as a secondary membrane. The increased PFAS removal could also be explained by a cake-enhanced concentration polarization. RO membranes have presented a PFAS removal efficiency over 99% or to below the detection limit. Membranes do not require reactivation but have a life expectancy of two to five years [70].

At high levels of fouling, pretreatment may be necessary leading to high operational costs and an increase in the complexity of the system. The capital and operational costs are also high due to the energy-intensive nature of membranes and additional treatment of the waste stream with highly concentrated PFAS [70]. Membranes offer a separation of PFAS but do not provide any following destruction of the concentrated PFAS retentate. Therefore, membranes may become an unnecessary expensive intermediate step unless there is an efficient destructing technique that requires high concentration or a smaller more manageable volume. Additional destruction techniques for the waste stream have demonstrated a high degradation efficiency of PFOS and PFOA in aqueous solutions. Especially, electrochemical oxidation, supercritical oxidation, and sonochemical decomposition have displayed a promising potential to degrade PFAS. Nevertheless, the methods require extreme and specific operation conditions and have a large energy and chemical demand. Consequently, the method becomes less practical and economical on larger scales leading to a non-cost competitive alternative in relation to adsorption processes [70]. However, if an efficient destructive technique would be identified and a sufficient phase separation could be achieved before the membrane steps it could act as an effective separation technique.

5.6 Chemical Oxidation

Many new developments are made in this field and several new methods are presented. Multiple methods have been investigated to chemically oxidize PFAS. Complete degradation of PFAS by chemical oxidation would yield the end products CO_2 and F^- . Free hydroxyl radicals (OH^\cdot) are strong oxidation agents that can be created from for example hydrogen peroxide in combination with UV-light or divalent iron. Precursors can be oxidized into stable PFASs but despite the strong oxidation characteristics of hydroxyl radicals, they cannot oxidize perfluorinated carbon chains. Another strong oxidation agent is persulphate ($\text{S}_2\text{O}_8^{2-}$), which has shown a potential to degrade carboxylic acids at 60–70 °C, where an efficiency of over 90% was reached for PFOA. However, the reduction of PFOS and PFHxS was lower than 20%. In addition, ozonation trials have been performed for PFAS destruction but led to low removal efficiencies with large variances between different facilities. Thereupon, identification of an oxidation agent that can degrade a larger number of PFASs is necessary for chemical oxidation to be a useful technique. In addition, oxidation has a moderate to high cost and a lack of full-scale applications for PFAS destruction by oxidation [72].

The oxidation method that has shown the most potential is electrochemical oxidation. By applying electricity in PFAS-contaminated water, the chemicals can be degraded by either indirect oxidation with formed hydroxyl radicals, or through direct electrochemical oxidation. A removal efficiency of over 90% has been reached where the degradation was higher for long-chained PFASs. It has been found that the amperage and treatment time are more significant parameters than the concentration of electrolyte and initial concentration of PFAS. There are several benefits mentioned by Pettersson et al. [72], including its possibility to destruct PFAS, the robustness in the

performance, and the ability to adjust to various compositions in the treated water.

However, there are some challenges with electrochemical oxidation including that the process is energy-intensive, leading to being most applicable for highly contaminated concentrate due to high costs. The method does have a need for pretreatment to raise the PFAS concentration and reduce the treated volumes to have a promising application strategy in reality as well as lowering costs. Additionally, there is a possibility to create unwanted by-products depending on the composition of the water including hydrogen fluoride, chlorine gas, bromate ions, and chlorate ions. Formation of dangerous by-products would require additional treatment, which would decrease the economic feasibility of the technique. Thus, further investigations are required to determine the composition of the outlet stream after the degradation of PFAS with electrochemical oxidation. In addition, there is an issue that laboratories that investigate removal efficiency do usually apply extreme operating conditions. Therefore, there is a need for trials with more environmentally relevant operating conditions to evaluate its actual potential. Also, Yadav et al. suggest that the electrodes consisting of heavy metals may slowly break down, resulting in toxic release into the environment [72].

Another alternative to destroying PFAS is Super Critical Water Oxidation (SCWO). A supercritical liquid is obtained when water reaches 374 °C and 221 atm, where non-polar substances become soluble and polar substances have a low solubility opposite to the usual properties of water. By the addition of oxygen to a supercritical liquid, organic substances are converted, which theoretically could lead to complete degradation of PFAS [72]. In a study presented by Pettersson et al. [72], complete degradation of PFOA and PFOS was reached at a temperature of 650 °C and a retention time of 30 seconds. SCWO is beneficial as the reaction speed is extremely rapid, it has a high destruction efficiency and low energy consumption. However, the method entails some challenges including corrosion in the equipment and precipitation of salts. Also, the knowledge about the robustness of the process is limited and the method is immature. Therefore, more research is required to confirm its potential to destroy PFAS [72].

5.7 Sonolysis

An additional option to destruct PFAS in water is to apply ultrasound, which creates pressure in the water that either pushes molecules together or pulls them apart. The pressure leads to the formation of small microbubbles that grow until an unstable diameter is reached, which produces cavitation when they collapse. Thereby, a high temperature that is sufficient to pyrolyze PFAS is reached. In addition, hydroxyl radicals are formed, which can oxidize organic compounds. According to Pettersson et al. [72], the temperature and pressure can exceed 3 700 °C and 1000 atm at a frequency of 20 kHz. It has been reported that a frequency of 358 kHz gives the most efficient degradation of PFAS4. However, a combination of different frequencies could increase the degradation rate further by synergistic effect [72].

It is possible to combine ultrasound technologies with reducing agents such as sulfide, iodine, or ferrous iron. The combination would produce highly reactive species with the capability to degrade chemicals or mineralize them to fewer toxic substances [70]. The method has high security and does not produce any hazardous secondary by-products. However, it is energy-intensive, and the reaction speed varies depending on current density, pH, type of PFAS, and inorganic substances. Another major drawback with the technology is that there is presently no existing full-scale application, resulting in uncertainties about its robustness on a larger scale. Therefore, the technique needs to be evaluated on the contaminated water at Norrvatten to optimize the degradation before it could be implemented in a treatment process [72].

5.8 Reuse of Solid Sludge

The cheapest application for the centrifuged sludge would be for usage in soil improvements, as Norrvatten is presently doing. However, the sludge from DAF becomes contaminated with PFAS, making it uncertain if the sludge could be applied for that application as either the entire sludge stream or only the solids from the centrifuged sludge. There are few regulations on how PFAS-contaminated sludge is allowed to be used. In Sweden, there is no limit on PFAS sludge applied on arable land. However, from the toxicity of PFOS on terrestrial organisms and surface water

impact, a low-risk level has been calculated to 0.12 mg/kg TS, which does not have any legal meaning but is used by for example REVAC-certification for wastewater sludge. In Denmark, there are limits on both PFAS4 and PFAS22 for wastewater sludge at 0.01 mg/kg TS and 0.4 mg/kg TS respectively. Also, The Swedish Environmental Protection Agency has proposed a limit for PFOS at 0.02 mg/kg TS in the year 2030 [8].

Applying solid sludge in the environment becomes a question of contaminated soil and how PFAS moves over time. If the concentration is lower compared to the limits proposed by Denmark and the Swedish Environmental Protection Agency, the dewatered sludge could be applied for reuse on for example arable lands without causing risks to the environment. In addition, it may be possible to mix it with sludge from sedimentation, which would yield a lower concentration, especially if the concentration is significantly lower than the limits presented above. However, as the sludge contains small amounts of particles, this option may not be beneficial.

5.9 Incineration

Highly contaminated sludge may be sent off to incineration. Waste incineration has been extensively researched from its energy production perspective but its role in detoxing waste is not as investigated [61]. Belleza and Youhanan [61], stated that high-temperature combustion had the potential to destruct PFAS with close to a 100% efficiency. Maraschin et al. [73] analyzed the flammability of aluminum-precipitated waterworks sludge and found that the solids were non-flammable. Therefore, co-combustion with other hazardous waste may be an alternative. Sludge incineration is not applied broadly in Sweden but with the need to remove PFAS from the environmental cycle it may be raised as an alternative for contaminated streams. This is because the knowledge basis on alternative treatment methods for PFAS-contaminated sludge is limited.

The main benefits of incineration of DAF sludge are that it is well researched to have the capacity to destruct PFASs, which can ensure an elimination of PFAS from the environmental cycle. However, despite the advantages, it has two major shortcomings that may need to be addressed. A common issue with sludge incineration is that desirable resources such as nutrients or trace elements go to waste, which is problematic to reach a circular and sustainable society [8]. However, waterworks sludge generally does not contain many desirable resources [62]. Therefore, it can be argued that incineration would not disrupt a circular economy approach [8].

Swedish waste incineration plants are equipped with advanced flue gas purification to decrease emissions to the atmosphere. However, after incineration, the ash is usually landfilled, which can result in leakage if a collection and treatment of PFAS is not in place. Additionally, depending on the recipient of the flue gas condensate, it is possible that PFAS can enter the environment. Usually, the flue gas is sent to a wastewater treatment plant or is directly released into the environment, which would result in contamination of the environment unless there is a PFAS treatment on the wastewater plant [8]. However, in the case of a leakage the net amount of PFAS would decrease due to the destroyed amount of PFAS during the incineration, so even in that case it can be argued that it is beneficial.

There is one rotor boiler close to Norrvatten that can reach sufficiently high temperatures to destroy PFAS located at Fortum waste solutions in Kumla [74]. They receive hazardous waste and destroy it at 1100 °C in their facility [61]. The regulation on Persistent Organic Pollutants (POPs) will yield new limits on when PFAS-contaminated waste must be treated to irreversibly destroy the chemicals [75]. The new limit will be decreased from 50 mg/kg to 1 mg/kg, therefore Fortum expects to receive increased volumes of PFAS-contaminated waste. Fortum accepts waste with lower concentrations than what is regulated by the POPs regulation to offer sustainable waste solutions to companies that do not wish to minimize their pollution [76]. Fortum has stated that they could accept 10 m³ of foam every second day for incineration at Fortum Waste Solutions.

5.10 Landfill

An alternative could be to put the sludge in a landfill. In Sweden, household waste is not allowed to be deposited but industrial waste and ashes from waste incineration are commonly put in landfill. More modern landfills can contain sealing layers or collection and treatment of leachate to prevent

pollution of the environment. Elevated levels of PFAS have been detected in the air surrounding landfills, which suggests that they are a collection point of PFAS in society [8]. Baresel et al. [8], suggest final storage on a closed landfill with leachate management as a solution to handle PFAS until there is more knowledge about thermal destruction. Out of the 6700 landfills registered in Sweden, there are only a few percent that investigate PFAS but the interest in the industry has been increasing during the last years [8].

PFAS treatment of landfill leachate has been investigated by the waste management company Ragn-Sells at their landfill Högbytorp, north of Stockholm. They used ultrafiltration in combination with membrane filtration [77]. The CEO of Chromafora, Johan Seijmer, stated in an article by Strid [77] that a decrease of over 45% has been achieved for PFAS11 but it is estimated that over 90% PFAS removal should be possible. In addition, Gustafson [74], stated that Ragn-Sells in Högbytorp has implemented a new facility called Ash2Salt that primarily treats fly ash from waste incineration from their landfill for hazardous waste. Ragn-Sells has an internal classification of hazardous waste, where all waste with a higher PFAS content than 0.003 mg/kg TS should be placed on their landfill for hazardous waste. Their non-hazardous landfill does not have any treatment of leachate water, thus PFAS content in waste that is placed on that landfill will be released into the environment. In Ash2Salt, leachate from the hazardous waste landfill will be used as cleaning water. The last treatment step for the washing water is an activated carbon filter that will remove PFASs [74].

Högbytorp does accept PFAS-contaminated soil, but there may be issues with their willingness to accept the concentrated sludge depending on its consistency. Ragn-Sells are currently planning to implement DAF on their landfill leachate. Therefore, they are investigating different solutions to the challenge on management of the sludge they will produce. It may be possible to mix the sludge with the sludge from Ragn-Sells for simultaneous treatment. Currently, Ragn-Sells are investigating the possibility of immobilizing PFAS in cement that could be placed on the landfill for hazardous waste. As the leachate from the landfill is used as cleaning water in their Ash2Salt facility, possible leakage from the cement would be collected in the GAC filters. However, this would entail many means of transport unless the sludge is concentrated in a smaller volume. Also, the technique is in an early research stage. Therefore, with the present methods at Högbytorp's landfill, the PFAS in the sludge cannot be efficiently treated [74].

5.11 Soil Cleaning

A relatively new process that can be applied to PFAS-contaminated soil is soil washing under the name SourceZone. It is a well-established method to clean soil from metals and some organic substances but the process is presently being designed to be applicable for PFAS removal. It is argued that soil washing can reach a 99% PFAS separation for both silty material and clay, but the results are not consistent [72]. Pettersson et al. [72], explain that a significant part of the treated soil masses should be reusable if the soil washing should be an economically sustainable option. As guidance on allowed PFAS concentrations for reused soil is absent, permission for reuse can be problematic. Also, they present a risk for high future requirements that cannot be reached through soil wash. An additional concern is that there is a significant need for development of the process and verification of the efficiency as well as minimizing the PFAS in residual streams [72].

Pettersson et al. [72], present thermal treatment as an option to clean soil from PFAS by heating the soil to high temperatures to evaporate PFAS. The volatility varies depending on the alkyl chain and the number of fluorine atoms attached to the carbon atoms. Data claim that at least 350 – 450 °C is necessary to vaporize PFAS but there are results indicating a significantly higher temperature is required. However, the temperature is not high enough to destroy PFAS. Therefore, the vaporized chemicals need to be captured in an air filter, commonly constituting of activated carbon. The method entails a high energy demand giving a negative impact on its cost efficiency and environmental burden. Also, the method has presently many uncertainties related to it including required temperatures and treatment times, what substances are formed from incomplete degradation of PFAS, and what happens with precursors and short-chain PFASs. Therefore, more research is required to increase the knowledge about thermal treatment of PFAS-contaminated materials [72].

Another alternative to prevent the spread of PFAS from soil is to add a sorbent with high affinity to PFAS, which would drastically increase the soil's capacity to bind PFAS. The mobility of the sorbent will then affect the possibility for PFAS to leak from the media making it possible to immobilize PFAS in soil. As the method is mature and cheap, large volumes can be treated with this method. However, the addition is usually difficult or impossible for fine-grained soil materials. Also, this method entails uncertainties related to the endurance and function over time of the sorbent [72]. Lastly, these methods may only be applicable for sludge with a high TS content, which would leave a highly concentrated clear phase without a sustainable waste management technique.

In summary, all presented soil cleaning options are relatively immature and have not been applied in full scale for waterworks sludge [72]. There are many knowledge gaps in both soil washing, PFAS vaporization, and sorbent addition. Also, it leaves a highly concentrated clear phase if the entire sludge flow cannot be treated. One drawback of applying these methods to DAF sludge is that the solid content is produced in small amounts resulting in a possibly negligible amount of recovered materials. Also, as the waterworks sludge contains limited amounts of usable substances, the use of clean solid materials may be limited. However, as more investigations are made and the knowledge gaps are filled, these techniques may have the potential to provide a sustainable alternative to save energy and minimize carbon emissions.

6 Application of Sludge Treatment Techniques in NFWP

In this section, the presented treatment strategies are evaluated to answer the second research question about PFAS treatment techniques that could be suitable for the DAF sludge in NFWP. First, the amount of sludge from DAF produced in NFWP, and its concentration are calculated. Thereafter, treatment strategies are discussed based on methods that are appropriate for the sludge. To estimate the final treatment strategy that is presented, some calculations on the efficiency of different routes were performed.

6.1 Amount of Sludge and Concentration

The production of sludge from DAF in NFWP was estimated based on present data from Görvålverket. The incoming flow was 420 l/s and the sludge flow was 2 l/s during the time period 2023-02-12 to 2023-03-18 in line 5, Görvålverket. This resulted in 0.48 % of the water inlet becoming sludge.

The sludge volume per inflow was also calculated by the same linear relationship but based on data from 2023-03-14 when all 5 DAF lines in Görvålverket were run simultaneously. This was done to capture eventual differences in sludge production between different lines and give the most representative estimation for NFWP. The mean incoming flows and sludge flows were gathered in Acurve for the time period 09:15-15:15 (see Table 13). By dividing the total sludge flow by the total inlet of water, the production of sludge was obtained to be 0.47 % of the inlet volume. This calculation showed that the sludge production can be applied for several basins and is not exclusively applicable to line 5.

Table 13: Mean water inflow and outgoing sludge flow 2023-03-14.

Line	Incoming flow [l/s]	Sludge flow [l/s]
1	57.5	1.79
2	217	1.15
3	440	0.63
4	319	0.83
5	322	1.92
Total l/s	1355	6.32
Total m³/d	117 101	546

Lastly, to investigate the assumption of linearity between the inlet flow of water and sludge production, the sludge production during sampling was calculated as different flows were applied for the various samples. If the sludge production is equally large for the different flows it can be assumed that the linear relationship is verified. The calculated flows are presented in Table 14.

Table 14: Sludge production during sampling per inflow of water.

Line	Incoming flow [l/s]	Sludge flow [l/s]	Sludge production [%]
14-feb	520	2.4	0.44
15-feb	480	2.3	0.48
17-feb	470	2.3	0.49
21-feb	381	1.8	0.47
22-feb	400	1.8	0.45
24-feb	380	1.9	0.5
07-mar	310	1.6	0.51
08-mar	315	1.3	0.41
09-mar	320	1.4	0.44

It can be seen that the percentage of sludge flow is approximately 0.4-0.5% for all different flows. There is no clear pattern that could be identified that the proportion of sludge would be affected depending on the inlet volume. Therefore, the linear relationship was considered to be verified and the proportion of sludge in the NFWP was calculated based on 0.5% of the inflow. As stated in Table 3, the mean flow for NFWP will be 4561 m³ per line and day, which corresponds to 200 665 m³/d (53 l/s) for the entire process. Thus, the sludge volume could be determined to be 1000 m³/d.

The concentration of PFAS in the sludge produced at NFWP was estimated based on the measurements performed in Görvålnverket, line 5. The concentration in the sludge differed between 220-710 ng/L. The concentrations for the mean flow in line 5 provide an average of 380 ng/L and a median value of 300 ng/L. As has been discussed, the measurements of the PFAS concentration in sludge differed significantly and did in many cases present higher sludge concentrations than what was reasonable according to the mass balance. Therefore, it was decided to do the calculations based on the median value for the mean flow on the sludge, which is 300 ng/L.

6.2 Comparison of Techniques for Sludge Treatment

The PFAS removal technique that is implemented must be dependable and well-evaluated to ensure a process that can reach the required reduction. The removal technique must also have a reasonable construction and operational cost. The PFAS-contaminated residual steam that has been separated from the solids in the sludge can be treated by using several different methods that have been presented previously. The most significant advantages and disadvantages mentioned are summarized in Table 26 together with some additional aspects collected from Pettersson et al. [72] and Yadav et al. [70]. Also, the removal efficiency and qualitatively stated costs are presented in Table in the table.

A substantial parameter in the selection of treatment technique for the liquid phase was the removal efficiency. The PFAS concentration in the sludge should be reduced from 300 ng/L to 5 ng/L to avoid release of elevated PFAS levels to Mälaren. This is in accordance with the internally set limit by Norrvatten that regulates the PFAS concentration in water that should be returned to the lake. To reach the reduction target, a removal efficiency of >98% is required, which leads to substantial demands on an effective treatment technique.

Table 15: Removal efficiency, cost, advantages and disadvantages for clear phase treatment techniques

Technique	PFAS removal efficiency [70]	Cost [70]	Advantages	Disadvantages
GAC	>80%	Low to Moderate	Highly researched technology for PFAS removal Robust with constant removal efficiency through literature Cost efficient Possible coordination with already existing treatment process in NFWP	Low adsorption of short-chain PFASs Poor selectivity Requires thermal reactivation that generates exhaust carbon
AIX	50–90%	Moderate to High	Capability to remove both long and short-chain PFAS Small space requirements Small footprint	Regeneration do not offer PFAS destruction, resulting in a need to incinerate the material after usage Highly sensitive to system circumstances
Membrane	>90%	High	Capable to remove all PFASs No reactivation is required Small space requirement	Receives a relatively large amount of water with highly concentrated PFAS that needs further treatment Is exposed to fouling May require pretreatment of incoming streams High operational cost Energy intensive
Oxidation	10-50%	Moderate to High	Ability to degrade almost all organic compounds Environmental compatibility Rapid and efficient degradation	Excessive energy consumption High initial cost Potential formation of hazardous by-products that require additional treatment No existing full-scale application
Sonolysis	>50–90%	High	No production of secondary by-products	Energy intensive Highly sensitive to process circumstances No existing full-scale application

To meet the required target concentration in the water that should be released into Mälaren and to provide a functioning removal, implementation of a two-step technique may be necessary. The treatment route will therefore require an initial phase separation to enable the application of the treatment techniques that are applicable to treat water streams. The other stream containing the solids will require an additional disposal route. The solid stream will likely contain significant amounts of water, which raises the PFAS contamination in that stream. Therefore, it is of importance that both streams after phase separation are exposed to a method that can offer PFAS destruction to remove the chemicals from the environment.

Based on present knowledge, four possible treatment routes for the sludge were identified for NFWP based on the literature review on possible management techniques. These options are presented in Figure 19. The method that contains the least uncertainties regarding operating conditions, robustness, efficiencies, and data in terms of costs and energy demands is sought. This is to be able to propose as reliable an estimate as possible. For all processes that include phase separation, it has been decided to include incineration for the solid phase. The choice is based on the fact that combustion is the only present method that effectively can permanently destroy PFAS.

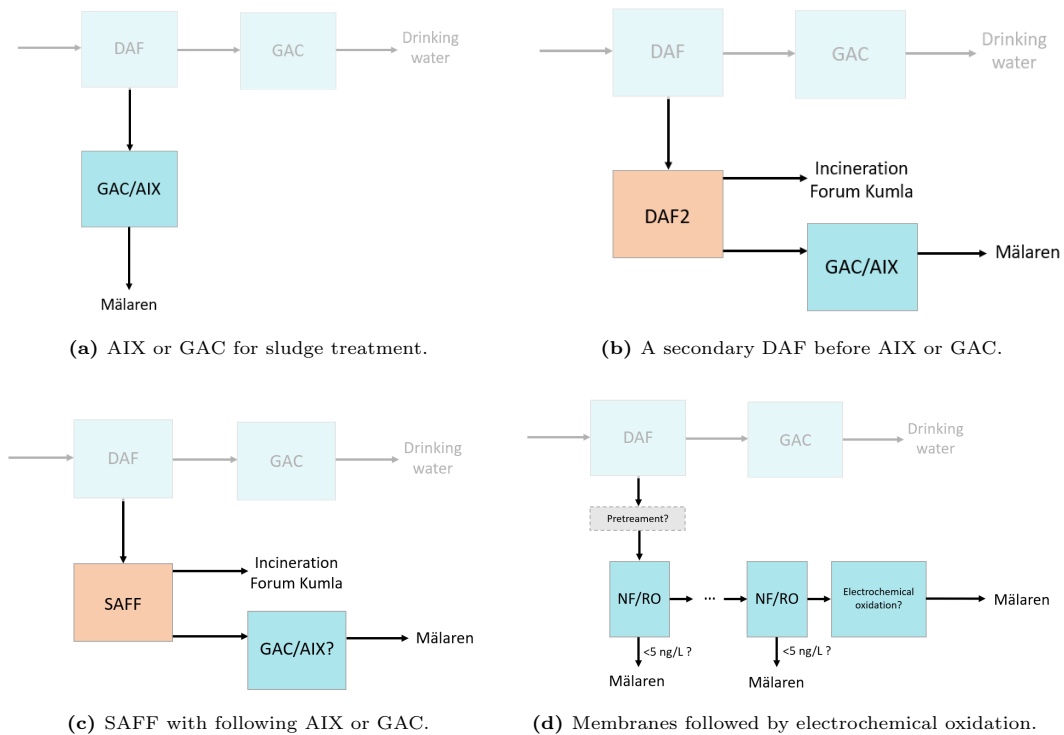


Figure 19: Proposed sludge management routes.

One possibility could be to transfer the sludge immediately to AIX or GAC before the water is released into Mälaren, which is illustrated in Figure 19a. However, based on the literature study on present techniques, solids can cause issues in many techniques aimed for water treatment. All membranes and filters including GAC and AIX can get issues such as clogging and reduced lifespan leading to larger costs and a less economical solution. Therefore, it was assumed that an intermediate step is required in terms of flotation, centrifuging, or filter media to remove the solids prior to the PFAS destruction.

Therefore, another possibility presented in Figure 19b, would be to lead the sludge to a second step of DAF. The secondary step would provide a phase separation with an increased concentration in the solid phase. Thereby, the water phase could be sent to GAC or AIX while the solids could be incinerated at high temperatures at Fortum Waste Solutions in Kumla. DAF2 is likely to have a higher separation efficiency than what has been measured in Görvålnverket due to the higher inlet concentration from the sludge. This efficiency could be approximated from the measurements taken at lower concentrations as well as literature on DAF in highly concentrated streams. The volume of concentrated sludge could be approximated from Görvålnverket to provide an estimate

of the amount that requires incineration.

The secondary DAF could be exchanged with foam fractionation for optimization of the volume of concentrated sludge and to increase the removal efficiency that would decrease the demand on AIX. In that case, SAFF could be of interest to reach a maximal concentration and minimal volume in the final sludge stream, as illustrated in Figure 19c. However, before this technique could be implemented more research is needed regarding the outlet concentration of the treated water from every step to estimate any cleaning requirements on those streams. Also, the cost of the SAFF process in relation to the other proposed techniques is important to evaluate.

Another possibility is to implement membranes to achieve a separation of the sludge, according to Figure 19d. In that case, RO and NF have both shown to have the potential for separation of PFAS. However, more investigations are required to determine what membrane that offers the most cost-efficient solution. Also, the need for pretreatment is of relevance as the solids may clog the membranes. According to Måns Lundh [78], the usage of a filter to remove the solids may be enough. Mixing the solids with the dewatered sludge from sedimentation should be acceptable since it has been shown that they carry close to an insignificant amount of PFAS. A suitable treatment technique for the retentate needs to be identified. It may be possible to use electrochemical oxidation, but that method entails uncertainties in the form of efficiency at the operating conditions and possible formation of other hazardous by-products. In addition, multiple membranes may be required to reach a reasonable volume of the retentate to achieve an economical solution. Lastly, it is of relevance to confirm that the permeate reaches a sufficiently low concentration to enable releasing the stream to Mälaren to ensure the benefits of the technique.

Based on this reasoning, it was decided to continue the calculations based on the sludge management route presented in Figure 19b where a second step of DAF is implemented before the concentrated sludge is sent to incineration and the aqueous phase is either AIX or GAC filters. The amount of concentrated sludge is approximated to be 0.5% of the inlet flow, resulting in a volume of 5 m³/d. Robey et al. [29] performed DAF on landfill leachate with a concentration of about 3000 ng PFAS4/L and achieved 90% removal efficiency with a foam concentration of 17 000 ng/L. Similarly, McCleaf [30] performed an investigation on landfill leachate with a concentration of 6200 ng/L, which reached a 90% removal for all PFAS4 except PFNA that reached 50-80%. The concentration used in the investigation by Robey et al. and McCleaf is significantly higher than the sludge concentration of 300 ng/L. It is reasonable that a higher concentration would yield a higher removal efficiency. Therefore, it was assumed that DAF on 300 ng/L could reach a 70% efficiency, which would yield a concentration of approximately 90 ng/L for the permeate. The concentration in the concentrated sludge could thereby be calculated as a concentrate from the inlet (see Equation 10). The calculation yielded a sludge concentration of 42 µg/L.

$$\frac{210 \text{ ng/L} \cdot 1000 \text{ m}^3/\text{d}}{5 \text{ m}^3/\text{d}} = 42 \text{ } \mu\text{g/L} \quad (10)$$

A schematic of the second DAF process on the primary sludge produced in NFWP is illustrated in Figure 20, specifying the concentration and volumetric flows over the DAF basin. In the figure, the naming of the streams that will be used in the following calculations is presented. The concentrated sludge will be assumed to be sent off to incineration and the permeate will require treatment before being returned to Mälaren. However, it is uncertain what technique that is most suitable for the permeate, therefore in the following sections, calculations are made on the material requirement for AIX and AGC filters.

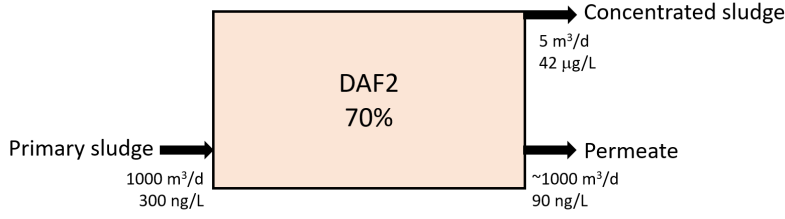


Figure 20: Second DAF on the sludge produced in NFWP.

6.2.1 Combination of DAF and Activated Carbon Filters

It was argued based on literature that GAC filters would be the most cost-efficient and sustainable solution for managing the permeate after DAF2. To investigate the suitability of GAC filters for the treatment of the aqueous phase, a calculation of the reactivation requirement was made. To perform the calculation, a saturation curve for the PFAS adsorption efficiency was necessary. The saturation state of the GAC filters will be dependent on the matrix in the water as various contaminants will be adsorbed on the filters because of the low selectivity of GAC filters. Therefore, it was considered that the most appropriate saturation data would be the data from the pilot trial performed at Görvålnverket. That argument was based on the fact that the pilot should have been exposed to similar contaminants that are present in NFWP because of the same raw water intake from Mälaren. However, the concentration of PFAS will be different, which was accounted for during the scaling up of the pilot.

The pilot was scaled up to the concentration, bed volume, and flow that will be applied in NFWP. To account for the difference in concentration, it was assumed that the time consumption for a specific saturation state was linear to the PFAS concentration in the inlet to the GAC filters. The GAC filters in the pilot investigation obtained an average inlet concentration of 3.9 ng PFAS4/L, while the filters in NFWP will receive a concentration of 90 ng/L. The bed volume was assumed to be 102 m³ consisting of the material Filtrasorb 400 as that material had provided the best results in the pilot investigation. The bed volume is the estimated volume for NFWP, where the same volume was chosen to ease comparison with the water treatment process. The removal efficiency was determined for a certain amount of bed volumes in the pilot. To estimate the time consumption in NFWP, the water volume for a given amount of bed volumes was calculated, which was divided by the average flow through the GAC filters in NFWP (see Equation 11).

$$\Delta t(NFWP) = \frac{\Delta BV(pilot) \cdot V_{BV}(NFWP)}{\dot{V}(sludge, NFWP)} \cdot \frac{c_{in}(NFWP)}{c_{in}(pilot)} \quad (11)$$

In the equation $\Delta t(NFWP)$ is the time consumption that corresponds to a specific saturation state, $\Delta BV(pilot)$ is the number of bed volumes that were measured in the pilot trial for a specific saturation state, $V_{BV}(NFWP)$ is the bed volume of 102 m³, $\dot{V}(sludge, NFWP)$ is the flow to the filters in NFWP consisting of approximately 1000 m³, and c_{in} is the PFAS4 concentration entering the GAC filters.

From the time consumption in NFWP that represents a certain reduction efficiency of the GAC filters, a saturation curve could be constructed. The curve was made by plotting $\Delta t(NFWP)$ towards the measured reduction efficiency in the pilot trial. The saturation curve was extrapolated to cover a time period from 0 to 12 months as illustrated in Figure 21 (see Appendix G).

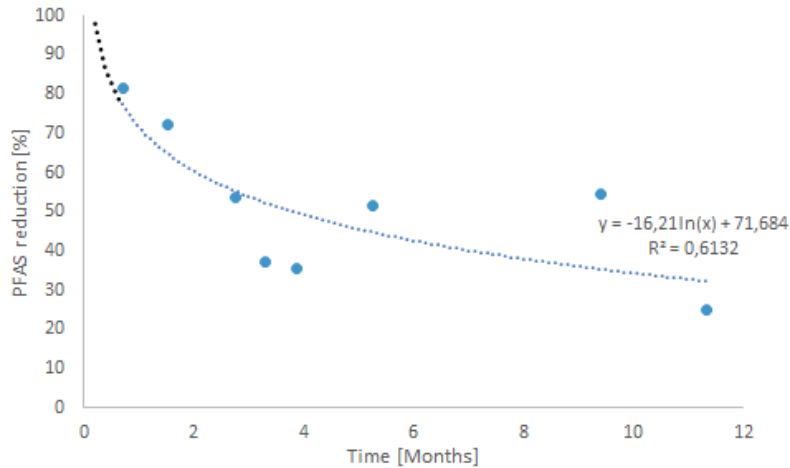


Figure 21: Saturation curve for inlet of 90 ng/L.

To reduce the concentration in the permeate after DAF2 from 90 ng/L to 5 ng/L an efficiency of about 94% is required. The time point where the removal efficiency decreased below the treatment target was sought to find the reactivation requirement. From the extrapolation, it was found that the reduction efficiency falls below 94% after 0.2 months when two filters of 102 m³ are used simultaneously. This means that 120 filters would have to be reactivated every year. However, it should be noted that the saturation curve produced data for a time duration from 2 months to 3 years. As the reduction efficiency is located in the values of the extrapolation, it is possible that the actual saturation could differ from the calculated value. Also, there are many other uncertainties, including the linearization of the concentration between the pilot and sludge, the analysis uncertainties in measurements in the pilot, an imperfect saturation curve illustrated by the R² value present in the figure, and deviations from the estimated DAF2 removal efficiency and sludge volume. Therefore, this calculation only provides a rough estimation of the reactivation requirement.

The amount of GAC filters that need reactivation cannot provide a competitive alternative despite consideration of the uncertainties in the calculations. Therefore, it was investigated if series-connected filters in two steps could reduce the reactivation frequency. The calculation assumed an initial removal of 67% to reduce the PFAS4 concentration from 90 ng/L to 30 ng/L. Thereafter, the second filter would require an 83% reduction efficiency to lower the concentration to 5 ng/L. It was found that the first filter would require 19.2 reactivations per year and the second filter 16 reactivations per year to reach their concentration targets. This yielded a total reactivation frequency of 35 filters per year. Although the decrease in the reactivation frequency was large through serie-connected filters, the reactivation frequency appeared to be non-competitive towards the treatment process in NFWP that excluded DAF. This could be determined as the reactivation frequency of the process excluding DAF had a significantly lower reactivation requirement, which will be presented in Section 7.1 *Reactivation of Activated Carbon Filters*.

6.2.2 Combination of DAF and Ion Exchange

As GAC was determined to be unsuitable for treating the water phase, it was examined if AIX could provide a more efficient alternative. The calculations were based on the material FLUORO-SORB [78]. In a study performed by Pannu and Plumlee [79], FLUORO-SORB 200 was evaluated on a water solution with a PFAS4 concentration of 52.2 ng/L. From their report, the bed volume could be calculated to be 0.83 dm³ and the flow rate to 18.4 m³/month. The sludge flow in NFWP will correspond to 30 000 m³/month, which is 1640 times higher than the flow used by Pannu and Plumlee. The actual filter size that is necessary could be determined from the flow and the EBCT. As the flow is significantly larger than the study, a larger EBCT of 6 minutes was assumed. By dividing the flow of 1000 m³/d with the EBCT, a bed volume of 4.17 m³ was determined. To have a redundancy of 2, the AIX filters would have to have a bed volume of 8.3 m³. The measurements were scaled up by accounting for the differing bed volume, flow, and concentration. This was done by multiplying the time consumption for a specific reduction efficiency according to Equation 12.

$$\Delta t(NFWP) = \Delta t(pilot) \cdot \frac{V_{BV}(NFWP)}{V_{BV}(pilot)} \cdot \frac{\dot{V}(pilot)}{\dot{V}(NFWP)} \cdot \frac{c_{in}(pilot)}{c_{in}(NFWP)} \quad (12)$$

In the equation, ΔT is the time fresh GAC has been used, BV is the bed volume, \dot{V} is the flow, and c is the concentration of PFAS4 in the water. Values for the reduction efficiency were gathered by reading a graph in the study. The graph illustrated the PFAS concentration in the outlet from the filters for a certain time consumption (ΔT). From that data, a saturation curve could be estimated (see Figure 22).

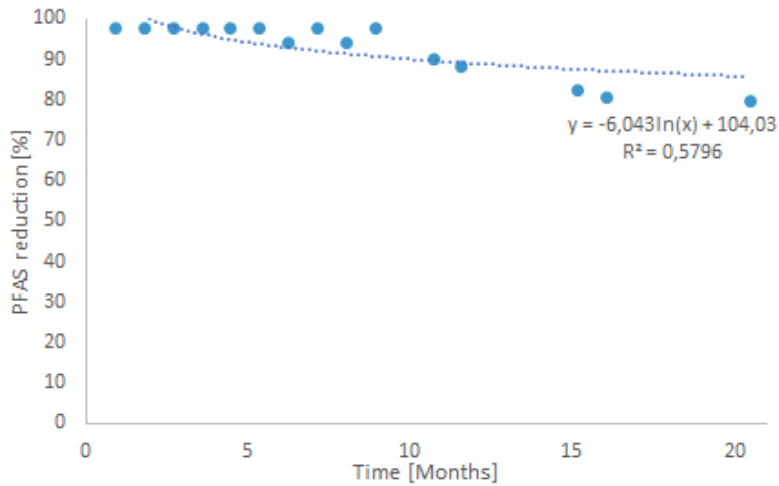


Figure 22: Saturation curve for AIX with an inlet concentration of 90 ng PFAS4 per L (see Appendix F).

The demanded reduction efficiency of 94% is still required for the permeate after the secondary DAF. By extrapolating the values for the fitted curve, it was estimated that two beds of 8.3 m^3 FLUORO-SORB 200 would need to be exchanged after 9 months. However, the calculations do include uncertainties, including deviations from the scale-up, other matrices, and varying concentrations of other PFASs than PFAS4 that may be present in the water as well as errors in the investigation and differences between the measurements and the fitted curve illustrated by the low R^2 -value. Therefore, it is possible that the calculated regeneration frequency could differ from the actual value, but it is uncertain in what directions the uncertainties will change the results. The significantly smaller amounts of AIX material compared to the amount of GAC indicates that it should be a more cost-efficient solution. Therefore, was it decided to continue the calculations based on AIX as a treatment for the DAF2 permeate.

7 Evaluation of the Potential of DAF in NFWP

To evaluate if DAF would be appropriate to include in NFWP, the process that only includes GAC filters was compared with the process containing DAF (see Figure 23). To evaluate the inclusion of DAF, the sludge management technique that provided the least uncertainties with confirmed operation conditions was included in the calculations. As has been presented, the technique that best fulfilled the criteria was the management process containing a secondary step of DAF where the permeate is treated with FLUORO-SORB filters before releasing the water to Mälaren. The concentrated sludge that should contain the solid particles was assumed to be sent off to incineration at Forum Waste Solutions Kumla.

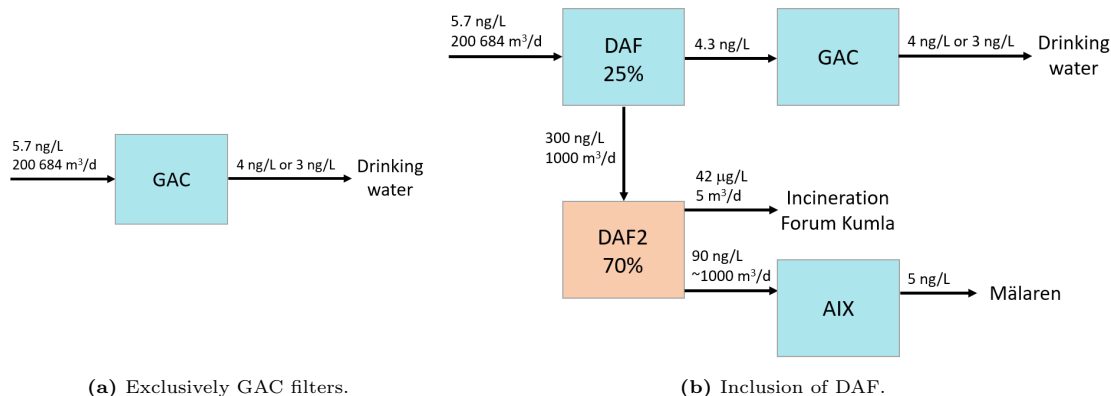


Figure 23: Flowcharts of the process proposals being compared.

In the calculations, it was assumed that the PFAS4 concentration in the raw water was the highest measured concentration during the last five years to avoid requirements of margins to the limit values. Similarly, to the previous calculations, it was also assumed that DAF2 would have a separation capacity of 70%, which is an assumption based on the measurements in Görvånverket and previous literature on DAF. In addition, the volume of sludge that would be produced in DAF2, was estimated to be 0.5% of the inlet water volume, which resulted in a concentrated sludge flow of 5 m³/d. Lastly, only the process modules within the system boundary were included in the evaluation as can be observed in Figure 23. The calculations involved a comparison of the reactivation frequency, energy requirement, operational expenses, and carbon emissions.

7.1 Reactivation of Activated Carbon Filters

The saturation curve that was constructed from the pilot trial at Görvånverket is illustrated in Figure 24 (see Appendix G). It was determined in the same manner as the calculations for the GAC filters for sludge but without any scaling by concentration. The saturation curve was produced as a logarithmic fitted curve based on the measurements from the pilot investigation performed at Görvånverket. By extrapolating the curve for a shorter and longer time period than measured in the pilot, an interval of two years could be used in the calculations.

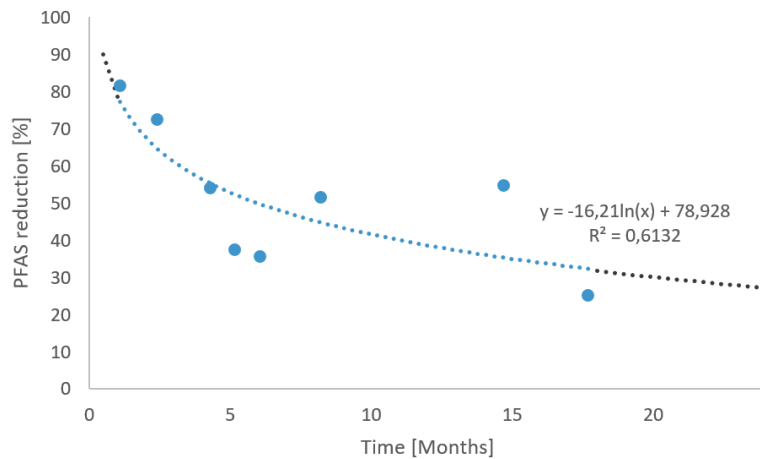


Figure 24: Saturation curve for Filtrasorb 400.

The R^2 value for the fitted curve shows a relatively large deviation between the saturation curve and the analytical results, which is also presented as the larger distance between the points and the curve. It is probable that the PFAS concentration analysis contained the same high analysis uncertainty that was obtained for the DAF investigation. Along with considerably few analysis points taken in the pilot, it is reasonable that the measurements deviate from the actual saturation curve. Nevertheless, the saturation curve had the expected behavior where it showed an almost 100% efficiency for fresh GAC and a significantly reduced removal efficiency after several months. The pilot trial on Filtrasorb 400 had only measurements for the first 18 months, which makes it uncertain how the material acts closer to its fully saturated state. Therefore, the extrapolation was not performed for a longer time period of 2 years to obtain a credible simulation.

The reactivation frequency is determined based on the outlet concentration from the mixing point in one group of 8 filters, which should not exceed the determined concentration limit. One filter is assumed to be out of service due to maintenance and there will be a total of four groups of filters. Thus, the average flow for 1/4 of the process, corresponding to one group, is divided into 7 filters. When one filter is reactivated, it is assumed that the partial amount of water that should receive treatment in that filter becomes untreated instead of being divided into the remaining filters in operation. From an optimization point of view, this may not be the most effective solution but did make the calculations manageable.

Figure 25a shows the outlet concentration from the mixing point for the optimal GAC reactivation frequency with the PFAS4 limit concentration of 4 ng/L and without DAF in the process. The calculations are based on a time consumption of 1 month for each reactivation. When a GAC filter is reactivated the outlet concentration decreases, resulting in the number of reactivation corresponding to the number of times the graph declines. It can be observed in the graph that no reactivation is necessary during the first 14 months and thereafter one reactivation is needed every 4-5 months for each group of eight filters. Thus, three reactivations will be required during the first two years for every group of eight filters. The dark dots in the figure represents the time that was included when determining the reactivation frequency per year, which is used to compare the different scenarios to each other. As the initial part of the graph will only be accurate for the start-up of the process it is not relevant for the yearly reactivation in a long-term perspective. Therefore, the start-up period was excluded from the calculations of the reactivation frequency. Also, to favor an equitable comparison, the largest amount of reactivation cycles in the range of steady state was included in the calculation.

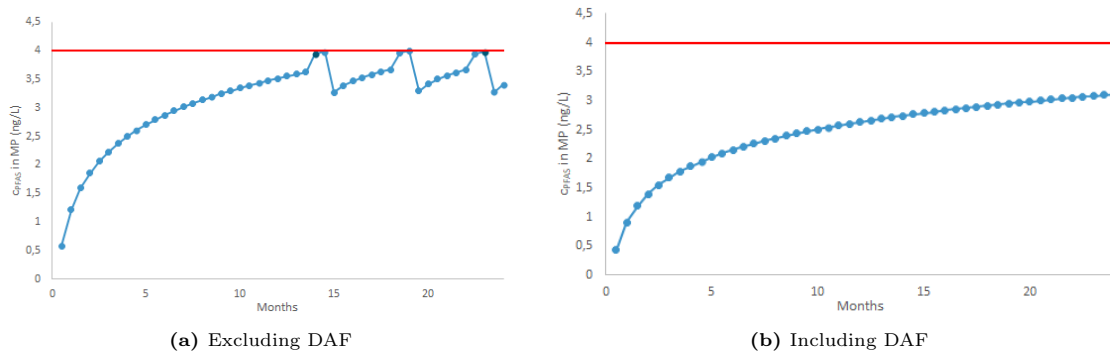


Figure 25: Outlet concentration from GAC filters with a concentration limit of 4 ng/L.

In Figure 25b, the outlet concentration in the mixing point is illustrated when DAF is included in the process. It was assumed that DAF could achieve a 25% PFAS4 reduction, thus the inlet concentration to the filter was set to 75% of the raw water concentration of 5.7 ng/L, which corresponds to 4.3 ng/L. After the saturation data had been applied for one group of eight filters for two years, the filters did provide removal of 27%, which is enough PFAS removal to achieve a limit of 4 ng/L. However, at some point, the GAC filters will be saturated and will require reactivation. Otherwise, the outlet concentration will reach the inlet concentration to the filters corresponding to 4.3 ng/L, which would result in the limit being exceeded. It is uncertain when the GAC filters will require reactivation as the saturation curve does not extend beyond 1.5 years. To have a realistic result that can be applied for more than 2 years, it was assumed that one filter will need reactivation during the two-year time period for each group of GAC filters. This assumption correlates to the assumption made by Lundh et al. [54], who stated that the reactivation should be done every second year to reach a sufficient PFAS reduction if DAF is included in the process.

In Figure 26a and 26b, the lower concentration limit of 3 ng/L was applied. When DAF is excluded in NFWP, a more frequent reactivation is necessary, which is due to the higher concentration of PFAS4 reaching the GAC filters. Reactivations are required after 3 months and thereafter every 1-1.5 months, where 16 reactivations would be required during the first two years. When DAF is included, no reactivation is required for 10 months. Thereafter, one filter needs to be reactivated about every third month for each group consisting of 8 filters resulting in a requirement of three reactivations during the initial two years.

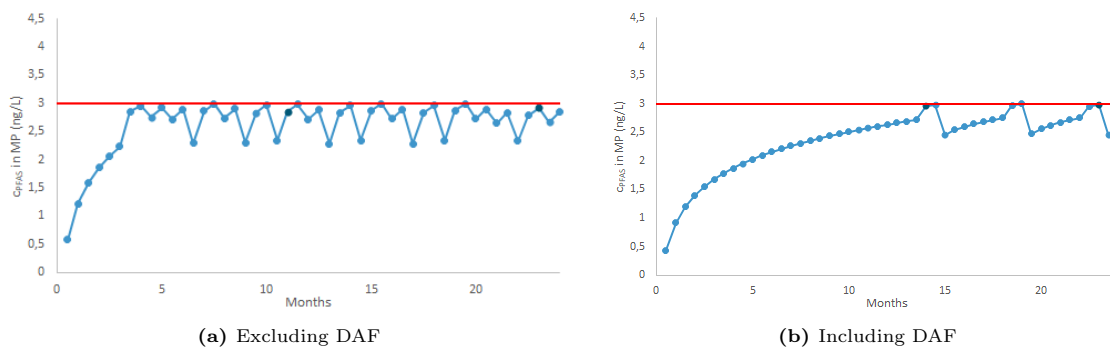


Figure 26: Outlet concentration from GAC filters with a concentration limit of 3 ng/L.

When the reactivation time consumption of 1.5 months was applied in the calculations, a similar behavior was obtained in most cases. For the concentration limit of 4 ng/L, the results were almost identical. When DAF was included, the reactivation time consumption did not have an impact as the concentration did not reach the limit for the first two years. The reactivation frequency when excluding DAF decreased slightly, where the reactivation had to be initiated 0.5 months earlier. For the concentration limit of 3 ng/L, when including DAF, the reactivation frequency also decreased similarly to when excluding DAF at a treatment target of 4 ng/L. The need to initiate the reactivation earlier was to not exceed the limit concentration as the time consumption was longer for the reactivation. In the calculation, the number of reactivation did not change. However, the

number of months included to calculate the reactivation frequency increased when the reactivation was initiated earlier.

With the concentration limit of 3 ng/L, excluding DAF, and with 1.5 months reactivation time consumption, the concentration limit could not be achieved. When more than 2 filters were reactivated simultaneously, the concentration increased rapidly. However, when only 1 or 2 filters were reactivated simultaneously, it was not possible to keep the concentration close to the limit. In Figures 27a and 27b, the two most optimal results are presented, which was when the reactivation is done once a month or when the reactivation is done every 1/2 month. In Figure 27a, only one filter is reactivated at a time, which resulted in 8 reactivations per year for 8 GAC filters, where the highest concentration became 3.05 ng/L. In Figure 27b, the reactivation of carbon filters is initiated while one other carbon filter has 0.5 months left until it can be put into operation. For that scenario, the outlet concentration from the filters increased to 3.13 ng/L but the reactivation frequency decreased to 12 times/year.

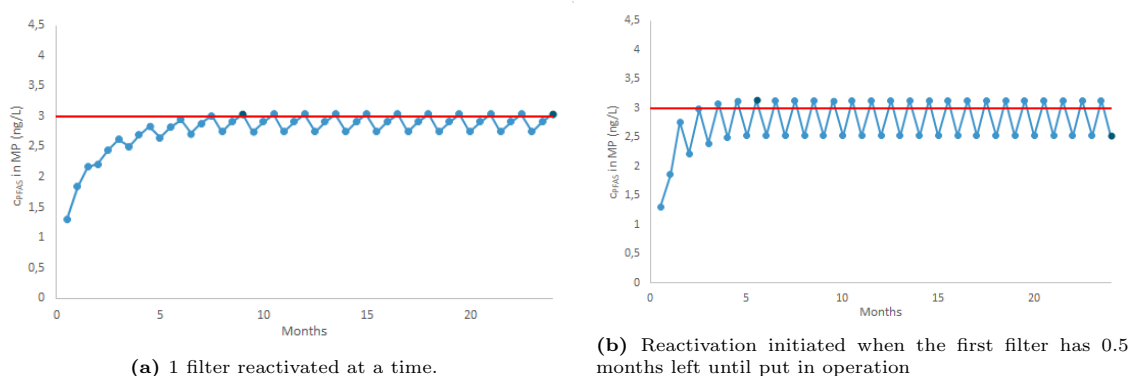


Figure 27: Outlet concentration from GAC with limit 3 ng/L, excluding DAF in the process when 1.5 months reactivation time is applied.

A summary of the reactivation frequency for varied limit values, reactivation time consumption, and choice of including DAF is presented in Table 16. The reactivation frequency in the table is presented for the entire process, which differs from the previous results that have been presented per group of 8 GAC filters. It can be observed that the chosen limit value and DAF has the largest impact on the reactivation frequency. It is reasonable that DAF decreases the reactivation frequency as the load of PFAS4 on the GAC filters decreases. For the limit of 4 ng/L, a reduction of 23% would be required, while for 3 ng/L a removal efficiency of 53% is necessary. As the required removal efficiency differ significantly, it is expected that the treatment target has a large impact on the reactivation frequency. The Excel sheets for each calculation are presented in Appendix H.

Table 16: Reactivation frequencies per year for varied limit values, time consumption for reactivation, and with exclusion and inclusion of DAF for the entire process.

Limit Time for reactivation	4 ng/L		3 ng/L	
	1 month	1.5 months	1 month	1.5 months
Excluding DAF	15.2	14.4	34.4	32 (3.05 ng/L)
Including DAF	2	2	15.2	14.4

7.2 Energy Requirement

The analysis of the DAF process revealed the energy consumption modules to be the pumping power required to transport the produced sludge from the basin to the dewatering facility, the DAF on the sludge, and the energy for running DAF, including compressing air, and utilizing waltzes and pumps. In the case of GAC filters, the only identified energy-consuming parts were the pumping power related to the addition of fresh GAC or removal of used GAC and the pumping of water to achieve the proper packing of the material. It was assumed that the pumping power for transporting the sludge to the dewatering facility and energy required for adding new AIX material was equal to the energy required for reactivating more GAC filters without DAF. Therefore,

the energy requirement for the process excluding DAF was assumed to be zero and the energy requirement for the process including DAF could be estimated from the energy for running DAF.

The energy consumption for DAF was determined based on the energy consumption of basin 5, which was measured between 2023-01-31 to 2023-02-07 to be 10 690 kWh/week [80]. During that week, the average flow in the basin was 426 L/s. This means that the energy requirement for the DAF basins corresponded to 41.5 Wh/m³. As presented in Table 3, the average flow in NFWP is dimensioned to be 53 L/s in 44 running lines. Also, DAF2 will be implemented for a sludge flow of 1000 m³/d. Thus, the energy requirement of DAF could be estimated by assuming that the energy for running DAF has a linear relationship to the inlet volume. The results are presented in Table 17. From the calculations, it can be concluded that the energy requirement will be over 3000 MWh larger every year when DAF is included in the process.

Table 17: Energy required to run DAF basins in NFWP.

Module	Energy requirement [MWh/y]
DAF in water treatment	3065
DAF for sludge treatment	15
Total energy for DAF	3080

7.3 Operational Expenses

The operational expenses were based on the cost of reactivation, energy requirement, transports, and sludge management. The reactivation cost was based on data from Chemviron, which produces and reactivates Filtrasorb 400 in Belgium. They assume that 10% of the GAC is lost and needs to be added as fresh GAC, which is a separate cost addition to the reactivation cost. The cost for the energy requirement was based on the energy price during 2022 for the highest consumption category from data from the energy authority presented by the Central Bureau of Statistics [81]. The cost for energy requirement does only include the power directly connected to DAF, similar to the energy requirement calculations. The cost for sludge treatment consisted of the energy requirement from the DAF2, FLUORO-SORB treatment on the water phase, and the cost for incineration at Fortum Waste Solutions for both the concentrated sludge and used FLUORO-SORB material.

For all transportation costs, the calculations were made based on the required volume that needed transportation instead of calculating the number of trucks that would be required for every delivery occasion, where some trucks would not be completely filled. In reality, to obtain the calculated cost, Norrvatten would need to order completely full trucks and keep excessive material on site between the delivery occasions. It is not probable that all material used in the process would be stored on-site to completely optimize the cost of transports due to additional costs related to storing on-site as well as limitations in space at the waterwork. However, it is probable that some optimizations will be made to reduce the number of deliveries that are required which would decrease the costs. What transportation Norrvatten chooses to optimize is uncertain, thus it was assumed that the fairest option would be to calculate the transportation requirement only based on the amount of material. Additional costs that could be related to storing the excessive material were not included in the calculations.

The material that should be sent off to incineration was assumed to be transported by Foria, which Norrvatten previously had used for transports. The price for the transport was given for a car with a trailer based on the A-price, which is the most expensive option as it does not include any potential negotiations or quantity discounts. It is assumed that the concentrated sludge is transported every week by one vehicle. For FLUORO-SORB, one filter consisting of approximately 4 m³ would be sent for incineration after usage, which also would require one car with a trailer for transportation. For both the concentrated sludge and FLUORO-SORB, it was assumed that filling and emptying takes 1 h each, which affects the cost of transportation as the price is based on time consumption. The time consumption to drive from a car garage in Sollentuna to Görvålverket with the final destination Fortum Waste Solutions and back to Sollentuna was determined based on Google Maps [82]. The input data for the cost calculations are summarized in Table 18.

Table 18: Input data for calculations of operational expenses.

Activity	Value	Ref.
Amount of GAC per filter in water treatment process	102 m ³	[45]
Cost of reactivation	10000 SEK/m ³	[83]
Cost for addition of fresh GAC	35 SEK/kg	[83]
Density GAC	540 kg/m ³	[84]
Price of electricity	0.9707 SEK/kWh	[81]
Energy requirement DAF	41.5 Wh/m ³	<i>Calc.</i>
Incineration at Fortum	7.34 SEK/kg	[76]
Transport cost for sludge	1900 SEK/h	[85]
Time to drive to Fortum, filling and emptying	5.3+2 h	[82]
Purchase of FLUORO-SORB	56 SEK/kg *	[86]
Transport cost for AIX	39 500 SEK/truck *	[86]
Load per truck	22 000 kg	[86]
Density FLUORO-SORB 200	740 kg/m ³	[87]

* Currency exchange: 11.29 SEK/Euro [88]

The material for GAC and AIX was needed in both volumes and amounts in the economics calculations, to convert between the two the density in Table 18 was used. The density will differ depending on if the material is dry or wet, which was not regarded in the calculations. Also, the cost of incinerating AIX was assumed to be the same as the cost to incinerate the concentrated sludge. The applicability of the assumption could be confirmed by the fact that the price for incinerating soil is the same as for foam, the same price was obtained for soil as for a thin foam. These assumptions were made due to limitations in time.

By inserting the number of reactivations and the need for transportation, the operational costs could be calculated for the different scenarios. The results are presented in Table 19. It should be noted that the results are based on current information, where several values may change in the future. A future perspective on the operational cost is partially covered by the sensitivity analysis, where the operational cost when changing different parameters was investigated. A more detailed presentation where the cost for each activity is stated can be found in Appendix I.

Table 19: Operational cost [MSEK/y].

	Module	4 ng/L	3 ng/L
Including DAF	GAC reactivation inc. transport	2.4	18
	Energy requirement	3	3
	Transportation excl. GAC	0.8	0.8
	AIX purchase and incineration	0.5	0.5
	Sludge incineration	13.4	13.4
	Total	20	36
Excluding DAF	GAC reactivation inc. transport	18	42
	Total	18	42

In the table, it can be seen that the major contributions to the cost come from the reactivation of GAC and incineration of the sludge. Therefore, it is the balance between those two parameters that will be deciding on what process suggestion is most economically favorable. For the more strict reduction target, the cost of reactivation exceeds the cost of incineration resulting in it being most economical to implement DAF. On the contrary, for the limit value from the Swedish Food Agency, it is more favorable to exclude DAF.

For the higher concentration limit, it is more cost-efficient to exclude DAF but the opposite for the lower concentration limit. This is reasonable as the cost of sludge management is constant at 17 MSEK/y as the sludge production does not change depending on the limit of PFAS, which becomes dominant when the limit is low, and the reactivation frequency of the GAC filters is lower. However, when the concentration limit becomes stricter, the cost of reactivation exceeds the cost of sludge management. It is 16-27 MSEK/y more expensive to reach a concentration of 3 ng/L

in drinking water. However, it is possible that restrictions on more PFAS are applied and in that case, it may be relevant to investigate this option for those chemicals as well. The cost is not complete due to the exclusion of pumping the sludge and power needed during GAC reactivation. However, the results should be comparable with each other as the energy requirement is relatively small in all cases.

To verify the results on the AIX lifespan, the data on the sludge was sent to the manufacturer to compare the results with their estimation. Mitch Brandon at MineralsTech estimated that 13.3 m³ would have a lifespan of 338 days [89]. The results were compared by determining the amount of AIX that would be consumed every year, resulting in 11 m³/y for the pilot investigation and 14 m³/y based on the manufacturer’s estimation. The results are in a similar order of magnitude, but it is possible that the calculations have been slightly underestimated. It was found that the cost increased by 150 000 SEK/y and the carbon emissions increased by approximately 2 ton/y.

7.4 Carbon Emissions

The carbon emissions were calculated in the amount of CO₂ per year. The calculations assumed fossil-free energy, leading to zero carbon emissions. Therefore, the calculations on the emissions included emissions due to transportation and reactivation. All other emissions are excluded in accordance with the delimitations. The input for the calculations is summarized in Table 20.

Table 20: Input data for carbon calculations.

Activity	Value	Ref.
Emissions from transport small truck	0.15 kg CO ₂ /km	[90]
Emissions from transport truck with trailer	0.64 kg CO ₂ /km	[90]
Emissions for reactivation	2 kg CO ₂ /kg GAC	[91]
Density Filtrasorb 400	540 kg/m ³	[84]
Volume per truck	46 ton	<i>Assump.</i>

The emissions from transportation were estimated with data from the Swedish Transport Agency. It was assumed that the deliveries of fresh GAC and AIX from the manufacturer could be estimated based on the emissions from a heavy truck with a trailer. The transports of sludge and used AIX to incineration were estimated for a light truck (see Table 21). In Table 21, it is specified what transports have been included and the vehicle used in the calculations. The carbon emissions for transports to incineration are estimated for a small truck. However, the cost of those transports are estimated for the cost of a car with a trailer. The emissions were stated by the Swedish Transport Agency as both "Well-to-Wheels" or "Tank-to-Wheels", which determine how many phases from the life cycle of the fuel that is included [92]. As the carbon estimation is limited to the direct emissions, it was decided to use the value for the more limited "Tank-to-Wheels". The carbon emissions are calculated based on the volume of the material that needed transportation instead of the number of trucks that would be required, similar to the operational costs.

Table 21: Summary of transports included in the calculations.

Activity	Vehicle	Distance
Incineration of sludge	Small truck	442 km [82]
Incineration of AIX	Small truck	442 km [82]
Purchase of AIX	Truck with trailer	1539x2 km [93]
Transport of GAC for reactivation	Truck with trailer	1628x2 km [94]

In the calculations, the reactivation frequency is based on the reactivation time consumption of 1 month. The emissions from the reactivation of GAC were based on a study performed by Vilén [91]. During reactivation, a part of the activated carbon will be gasified leading to a mass loss. These losses are compensated by adding fresh GAC after the reactivation process. In the article written by Vilén [91], different calculations of the carbon emissions are summarized. Three of the estimations that were stated were given for reactivation where the addition of fresh material was included, which was used in this calculation. The different calculations yielded values of 1.2, 2, and 2.75 kg CO₂/kg GAC. The carbon emissions due to reactivation were therefore determined as

the average value of the stated emissions.

For the processes excluding DAF, only reactivation and transport yielded emissions and they were in turn determined based on the number of reactivations per year. The process that included DAF also had emissions due to the reactivation and transport of GAC and transport of the foam produced from DAF. The results are presented in Table 22 (see Appendix J).

Table 22: Carbon emissions [tons CO₂/y].

	Module	4 ng/L	3 ng/L
Excluding DAF	GAC reactivation	1674	3790
	Transportation	38	86
	Total	1712	3875
Including DAF	GAC reactivation	220	1674
	Transportation	10	43
	Total	230	1717

As illustrated, DAF significantly decreases the amount of CO₂ emissions. In addition, a stricter treatment target leads to a larger amount of emissions being released. The results were expected as the number of GAC filters that need reactivation changes accordingly and the emissions due to sludge treatment are small in comparison to reactivation. The delimitation may provide a larger difference between the scenarios as only direct emissions are included with an exception for the reactivations of GAC. However, the emissions due to the incineration of AIX and sludge are excluded, which may lead to it appearing as a more beneficial option compared to reality. However, this was not further investigated due to time limitations.

7.5 PFAS Destruction

Another parameter that was included in the final comparison between the process options was the amount of PFAS that is destroyed. With the emerging threat from PFAS, it is of large importance to remove PFAS from the environmental cycle, which only can be achieved through destruction. The amount of PFAS that is destroyed will be dependent on the amount of PFAS that enters the process and the amount of PFAS that exits the process (see Appendix K). In the process where DAF is excluded, PFAS will be released through the drinking water where the amounts depend on the treatment target. The process that includes DAF will have a second stream that releases PFAS. After treatment with AIX, the water is returned to Mälaren with a similar concentration as the concentration that is present in the lake. Therefore, by calculating the PFAS amount in the stream that does not lead to destruction, the difference in PFAS destruction between the process options could be decided (see Table 23).

Table 23: Comparison of the amount of PFAS being destroyed in the process options.

	4 ng/L	3 ng/L
Excluding DAF [g/y]	125	198
Including DAF [g/y]	123	196

The calculations revealed that more PFAS is destroyed for a higher treatment target and when DAF is excluded, which was expected. The difference between the process options was calculated to 2 g/y, which gives a percentage difference of 1-2 %. Even though the difference is almost insignificantly small, it is certain that the PFAS destruction will always be more efficient when DAF is excluded.

7.6 Sensitivity Analysis

A sensitivity analysis of different parameters' effects on the cost calculation was performed. The parameters applied in the sensitivity analysis were the ones based on assumptions, estimations, or that are likely to change in the future. The parameters that were investigated were the removal efficiency in DAF, which affects the cost as the frequency AIX has to be exchanged is changed, the energy price, the cost of incineration, the cost of reactivation, and the amount of concentrated sludge that is produced.

Two different removal efficiencies in DAF2 were investigated, 40% and 90%. It was decided that it was not of relevance to investigate removal efficiencies below 40% as that removal efficiency had been discovered for an inlet concentration of 5 ng/L, where the efficiency is likely to be significantly larger at an inlet concentration of approximately 300 ng/L. It was found that one filter has to be exchanged every month for a removal efficiency of 40% in DAF as the outlet concentration is enlarged, where the cost for the DAF process increased with 2 MSEK but did not impact which alternative that was most cheap (see Figure 28a). When 90% removal efficiency was applied, it was found that 4.3 m³ of AIX would sustain for 120 months, leading to a need to exchange one filter every 10 years. However, this number includes many uncertainties as the saturation curve based on the literature is not valid for extended periods of time. The impact on the costs, if one filter could sustain for 10 years, would not be of significance as the AIX only constituted 0.6 MSEK in the original cost scenario.

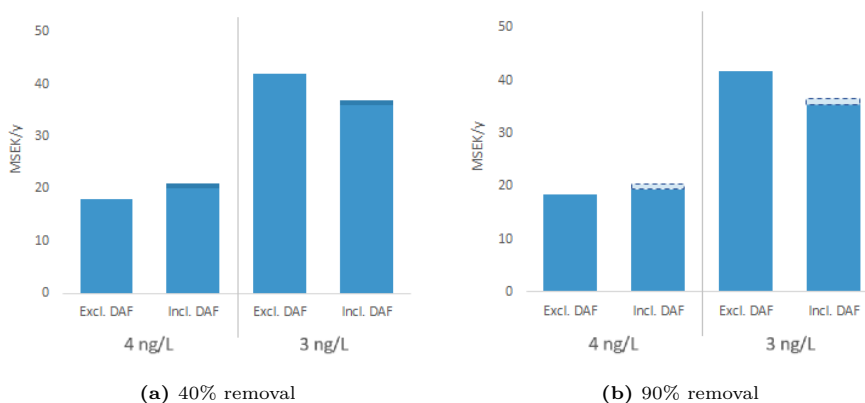


Figure 28: Removal efficiency in DAF2 affecting the operational cost.

The energy price used in these calculations was based on 2022. However, due to the recent pandemic and war in Europe, the energy price is presently higher than it used to be [95]. In 2017, the energy price was 0.367 SEK/kWh [81], which is approximately 1/3 of the energy price used in the calculations. When the electricity price of 2017 was used, the cost of the DAF process decreased to the same cost as the GAC process for 4 ng/L (see Figure 29.)

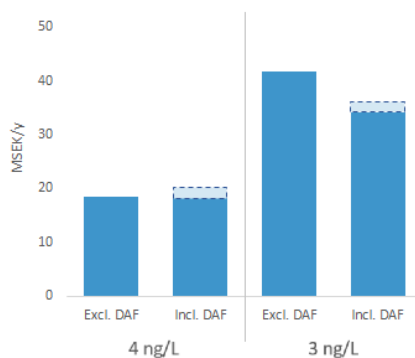


Figure 29: Electricity price 0.367 SEK/kWh affecting operational cost.

With the increase in regulations on PFAS-contaminated waste streams, Fortum expects that the demands on their facility will increase [76]. With the increased demand it is likely that the price would increase. If the cost would become twice as big, the cost for the DAF process would be significantly larger than the cost of the GAC process. It appeared that at an increase of approximately 3 SEK/kg, giving a total cost of 10.5 SEK/kg, the cost of the DAF process was identical to the cost of the GAC process at 3 ng/L (see Figure 30).

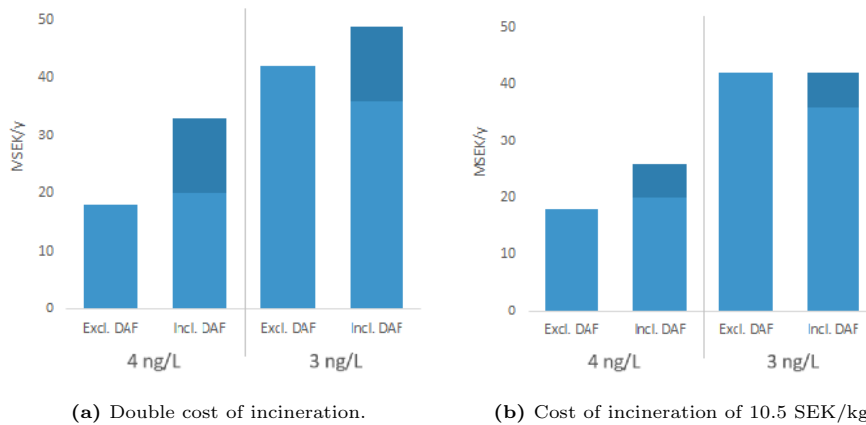


Figure 30: Cost of incineration affecting the operational cost.

The cost of reactivation may change due to differences in demand and new regulations. If the cost would decrease from 10 000 SEK/ m^3 to 7000 SEK/ m^3 , the GAC process becomes cheaper for both PFAS4 concentration limits. However, if the cost would increase to 12 000 SEK/ m^3 the DAF process would be cheaper for both concentration levels (see Figure 31).

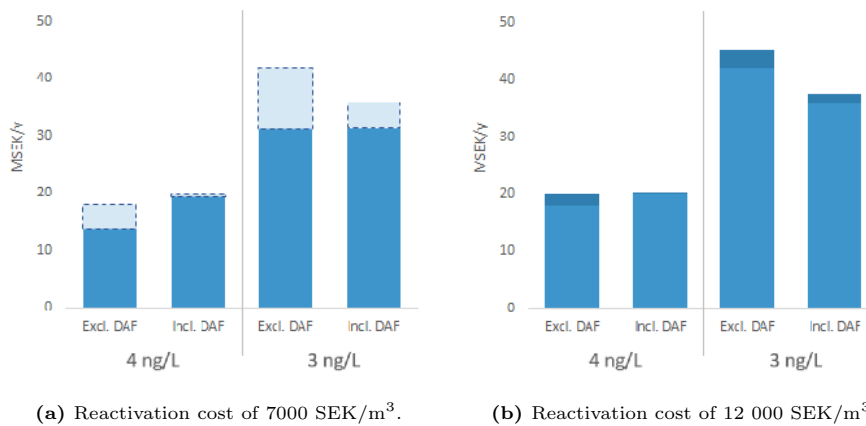
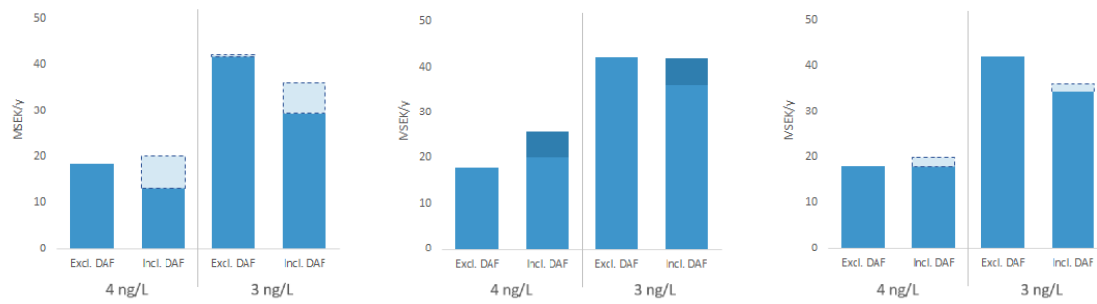


Figure 31: Impact from cost of reactivation.

Lastly, the amount of concentrated sludge that is sent off to incineration was investigated as it is based on a linear approximation based on Görvålverket's basins that produce 0.5% sludge from the inlet flow. If the amount of sludge would be reduced to half (2.5 m^3/d), the DAF process would be significantly more economical for both concentration levels. Already at a decrease of 1 m^3/d concentrated sludge, the DAF process becomes similarly costly as the GAC process at 4 ng/L. On the contrary, if the amount of sludge would increase, the DAF process becomes more costly than the GAC process at a volumetric flow of 7 m^3/d . The calculation did not include the differences in transportation demand. However, as the difference in volume is small, it is probable that the transportation would not have a significant impact (see Figure 32).



(a) Half sludge volume (2.5 m³/d).

(b) Sludge volume: 4.4 m³/d.

(c) Sludge volume: 7.2 m³/d.

Figure 32: Incinerated sludge volume effect on operational cost.

8 Discussion

A comprehensive assessment of the removal efficiency of DAF was assessed and the sampling performed allowed for a quantification of PFAS for the two phases in sludge. In addition, the potential to implement DAF in NFWP was disclosed with several considered key parameters. This section includes a discussion regarding the analytical results, the two process options where DAF is either included or excluded in the process, and the applicability of the results on NFWP. Although the experiments performed provided new insight into the potential benefits and drawbacks, there are some research gaps that still need to be filled before the final decision regarding the water treatment process is made.

8.1 Analytical Results

The analytical results indicate that a separation efficiency of 20-40% may be possible to achieve with DAF in the sedimentation/DAF basin with current operation conditions. The reduction follows the anticipated pattern where the reduction increases with an increased inlet concentration or a decreased water flow. Previously, there has been an investigation where three grab samples were analyzed in line 1 in Görvålverket. That analysis showed a possible removal efficiency of 20-30% [53]. In addition, the results can be compared with two literature studies that were identified to have similar inlet concentrations to Görvålverket.

In the study by Appleman [22], a removal efficiency for PFOS of 49% and PFNA of 29% was achieved for a total concentration of 13 ng/L for the two chemicals. Similarly, Thompson et al. [27] obtained a removal efficiency of 50% for PFOS and 27% for PFNA for an inlet concentration of approximately 11-13 ng PFAS4/L. They received a relatively constant concentration of PFOA and PFHxS. However, for some measurements, a decrease in the concentration of those chemicals could be observed. For comparison, in this study, the removal efficiency for PFOS of 41% and for PFNA of 31% was obtained. PFOA and PFHxS decreased by approximately 10%. However, the average concentration changed by 0.1 ng/L for both PFOA and PFHxS, therefore it can be argued that the results are consistent with all previous studies that have been identified. From the coherence between the investigations, it can be concluded that the provided removal efficiency is confirmed.

By assuming a 25% reduction (average removal rounded to the nearest 5), the outlet concentration of PFAS4 from DAF would be 4.3 ng/L for the highest inlet concentration to Görvålverket. As the limit by the Swedish Food Agency is set to 4 ng/L, DAF alone would not be sufficient to reach the limit. To reach the limit, a removal efficiency of 30% would be required for the highest concentration measured in the raw water. The measurements have shown that a reduction efficiency of 30% may be possible, thus with optimizations of the basin it might be possible to reach below 4 ng/L by only applying DAF. For the potential internal limit of 3 ng/L set by Norrvatten to prepare for future stricter PFAS limits in drinking water, the required reduction would be 48%. Based on the measurements, DAF alone cannot ensure a sufficient PFAS removal for the more strict treatment target for the drinking water with the current settings in line 5. Therefore, if DAF is implemented, it may need to be combined with other treatment methods.

According to Gyllenhammar et al. [96], it is probable that the limit values for PFAS chemicals will be broadened when there is more data present, which enables EFSA to perform a risk evaluation. Presently, there is no existing data on the PFAS chemicals beyond PFAS4, meaning that the risk the chemicals pose to health cannot be assessed. However, the lack of data is not an indication that the other chemicals are less toxic but a result of lagging research. Equal toxicity of the residual PFASs would mean that the limit would have to be decreased to not exceed the tolerable intake that EFSA has calculated. Therefore, if a larger number of PFAS are included in the regulation but the limit remains the same, the lower concentration limit of 3 ng/L may be of large relevance for the calculations that are made for the future waterwork NFWP [96].

All measurements have been gathered during one month, resulting in a lack of investigation of possible seasonal changes. Aspects such as water temperature and pH may affect the efficiency of DAF, which makes it relevant to investigate the efficiency over the seasons. Therefore, it may not be advisable to exclusively implement DAF to guarantee compliance with the limit value until

more investigations are made. The inlet concentration would also affect the removal efficiency. However, the PFAS concentration in the raw water has been measured previously where seasonal concentration changes have been disproved.

The measurements on the sludge revealed a PFAS concentration of 200-700 ng/L in the aqueous phase of the sludge. On the contrary, the mass balance suggested that the concentration should be in the range of 160-520 ng/L. Baresel et al. [8], stated that the sludge usually obtains a concentration that is 10-20 times higher than the inlet. When applying this to the measured raw water concentration, the sludge should have a concentration of 49-100 ng/L, which is significantly lower than the measured concentration. The difference to the study from Baresel et al. may be due to several factors including the amount of separated water with the sludge, water conditions, involved PFAS chemicals, or the settings used in the DAF basin. The analytically measured sludge concentration had a deviation of up to 140% from the mass balance values (see Appendix L), which indicates that the measurement uncertainty in the analysis is larger than expected or that other sources of errors might be present.

During sampling, several actions were taken to minimize the known possible sources of error. Collective samples were gathered over 6 h and sludge was collected during the entire pumping sequence to reduce the risk of receiving unrepresentative fractions of foam or water. The collective sample was carefully mixed before being sent off to the laboratory to ensure that the samples could represent the collected sludge. However, one misrepresentation was obtained due to sampling, which was identified after the completion of the measurements. The sludge samples were collected for the entire line 5 and the outlet sample was collected for only basin 5:1. Therefore, the results of the inlet and sludge correspond to the entire line 5, while 5:2 was excluded in the outlet sample.

It was observed that basin 5:1 had a higher load compared to 5:2 as more sludge was visible on the surface of that basin. Thus, it can be assumed that 5:1 receives a higher inflow of water than 5:2, resulting in a higher PFAS removal efficiency in 5:2 due to a larger air-to-water ratio. Thereupon, the outlet and sludge samples are representative for different streams, which can be taken as a safety margin as it should have provided a lower removal efficiency and a higher sludge concentration compared to reality. However, it could not explain the large deviations in the results of the performed reanalyses.

It was assumed that the volumetric flow of sludge is constant during one pumping sequence to support the assumption that the sample is representative of the entire sludge blanket. However, in the flow sheets, it could be seen that the volume changed during the pumping sequence, which should lead to a less optimal sample in terms of the representativeness of the reality. However, the errors should be decreased as several sub-samples were gathered. It was assumed that the collective samples would represent the entire concentration gradient during the entire sludge removal.

Another reason for the mass balance not being consistent may be because of other streams present in the process. The determined mass balance was based on the inlet, outlet, and sludge stream in line 5. However, as illustrated in Figure 33, there are several streams entering and exiting the basin that could carry PFAS. When the mass balance did not concur, the exiting amount of PFAS was higher than in the inlet. That effect could only be obtained from effects from the streams entering the basin. The inlet sample was taken at the raw water inlet, the water had therefore passed a flocculation step that was not considered in the investigation. However, Lundh et al. [54], stated that flocculation has small effects on the PFAS removal. Therefore, the raw water sample should be representative for the inlet to the basin as it does not pass any other process steps before the basin. This has also been verified in previous measurements in Görvålverket, which disclosed that flocculation and sedimentation do not affect the PFAS concentration in the water.

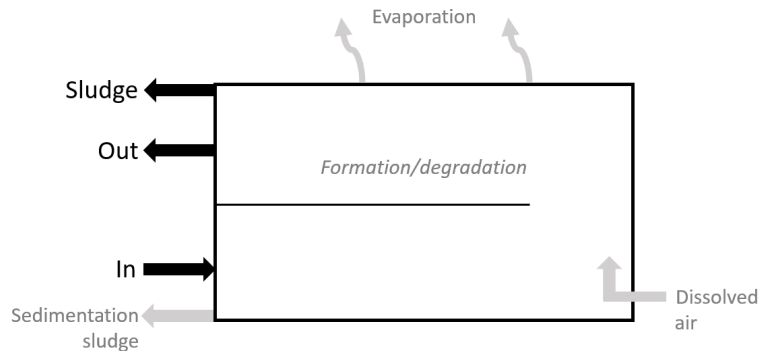


Figure 33: Schematic illustrating all streams that should have been included to have exact mass balance.

It is possible that the dissolved air could add some PFASs to the water. However, according to Jahnke and Berger [97], the PFAS concentration in air is in the order of magnitude of 10^{-12} g, which should not have a significant impact on the results. Lastly, it is possible for the formation or degradation of PFAS due to chemical reactions. Nevertheless, as previously demonstrated, PFASs exhibit high stability, and there are no identified factors within the process that would lead to increased reactions in the measured DAF basins. Consequently, it is improbable that potential reactions would significantly affect the results. In addition, all of these sources of errors should have given deviations in the same direction in the mass balance, which could not be identified in the results. Based on this reasoning, it is assumed that the additional streams in the basin should not have entailed significant errors larger than an order of magnitude of some picograms.

The differing results in the reanalyses indicate that there is a source of error in the laboratory that has a significant effect on the results in sludge concentration. On the contrary, the measurements on the water samples received a relatively constant concentration for all measurements. The more constant concentration indicates a smaller impact from the uncertainty for water samples compared to the sludge samples. The sludge should have a similar content to the water streams, except for a higher concentration of solid particles and PFAS. The solids were removed before the analysis at the laboratory, leaving no known difference between the samples that may impact the analysis. Therefore, there is a need for further investigations as no explanation could be identified for the larger deviations in the sludge analysis.

The analysis of the centrifuged samples showed that the majority of the PFAS is present in the water phase with approximately 2% on the solid particles. However, that proportion is for a solid sample where all four PFAS4 were not measured due to a high detection limit in the analysis. When the unmeasured PFAS was added with the value of the detection limit, the percentage did not change significantly. The low amount of PFAS on the solid particles is in accordance with a statement from EFSA [10], which declared that the PFAS4 chemicals have high aqueous solubility. The amount of solid particles in the sludge is small where the TS content is approximately 0.23%.

McCleaf [63], stated that the PFAS concentration is elevated in the highest layer of water directly below the sludge. Therefore, it is beneficial to remove a fraction of the water to maximize the separation. However, if a large portion of water is removed, a more diluted sludge is obtained, which leads to more extensive sludge treatment requirements. Sludge treatment entails a high cost according to the performed calculations, resulting in economic benefits when small amounts of sludge are produced. This leads to a balance between removing too little water and missing elevated PFAS levels below the surface or removing too much water and diluting the sludge. It is possible that the elevated PFAS layer is collected in line 5 as a relatively large amount of water appeared to be removed with the sludge. In addition, if the entire or a fraction of the layer would be lost at the lowest flow, it should be possible to identify a lower concentration for that setting. On the contrary, a significantly higher concentration was obtained because of less dilution of water, which supports the assumption that the PFAS layer is collected for all flows.

8.2 Final Comparison

From the calculations, it can be concluded that inclusion of DAF with the presented sludge management option would require a higher energy consumption, but less carbon emissions (see Table 24). The reactivation frequency of the GAC filters was decreased by approximately 14 filters per year, which leads to less transportation and emissions as can be seen in the results. Another parameter that was included in the final comparison was the amount of PFAS that is destroyed and thereby removed from the environmental cycle (see Appendix K). The difference in PFAS destruction is due to the additional stream that does not lead to PFAS destruction for the DAF process. The only way to affect the PFAS destruction is if the amount of primary sludge produced in NFWP would be decreased as that would lead to less water being released to Mälaren, or if the treatment targets for the drinking water or permeate that is transferred to Mälaren is changed. The PFAS destruction is obtained through the reactivation of GAC filters, incineration of concentrated sludge, and incineration of used AIX material.

Table 24: Comparison of the system with and without DAF.

	4 ng/L		3 ng/L	
	Excl. DAF	Incl. DAF	Excl. DAF	Incl. DAF
Reactivation frequency [MWh/y]	15	2	34	15
Energy requirement [MWh/y]	0	3080	0	3080
Carbon emissions [ton CO ₂ /y]	1712	230	3875	1717
PFAS4 destruction [g/y]	125	123	198	196
Operational cost [MSEK/y]	18	20	42	36

The most economical solution depends on the chosen limit value, where DAF is more beneficial at a lower treatment target in the drinking water, while GAC is more economical at a higher treatment target, as presented in Table 24. At a treatment target of 4 ng/L, the exclusion of DAF becomes more economically recoverable as the fixed cost for sludge treatment exceeds the cost for reactivation. However, when a more strict treatment target of 3 ng/L is applied, it becomes beneficial to implement DAF from an economic perspective. This is because the cost for the required amount of GAC becomes larger than the cost of sludge management through DAF.

In Table 24, the reactivation frequency was only stated for a reactivation time consumption of one month. In the results, the reactivation frequency decreased for a longer reactivation time consumption. This is because the reactivation had to be initiated earlier to meet the concentration goal. The frequency was calculated as the number of reactivations per the number of months after steady state (see Equation 9). The earlier initiation of reactivation for the longer reactivation time consumption led to a larger denominator, which resulted in a decrease in reactivation frequency. On the contrary, as the reactivation has to be initiated earlier, the longer time consumption is actually the unbeneficial alternative. Because of the choice to include full cycles of reactivations, the results could be misleading.

The reactivation time consumption may affect the possibility to reach the concentration target when the frequency of reactivation becomes smaller than the reactivation time consumption. For example, when DAF is excluded and a strict PFAS concentration limit is applied, it could become relevant to optimize the reactivation time consumption to reach the limit as the reactivation frequency becomes large in comparison to the reactivation time consumption. For the concentration limit of 3 ng/L when DAF is excluded, it was possible to conclude that the reactivation time consumption can become critical to optimize. As 3 ng/L probably would be an internally set limit, it could be decided to accept a small surpass of the limit. However, if the regulations on the PFAS limit would become more strict with a treatment target of 3 ng/L, it will be necessary to either implement another treatment technique or optimize the reactivation time consumption.

The results on the reactivation frequency can be compared with the similar investigation performed by Lundh et al. [45], who found a reactivation frequency of 1.3 times/month for the entire process (15.6 times/year) without DAF, with a limit of 3.8 ng/L and an inlet concentration of 5.2 ng/L. The results are in the same order of magnitude as the results presented in Table 16. However, comparing the results with the reactivation frequency of other waterworks may not be representative as the matrix in the water probably differs from the inlet water at Norrvatten. Also, the inlet

concentration, limit value, and GAC material are probable to vary from the data used in these calculations. For that reason and limitations in time, no such comparison was performed.

The concentration of PFAS4 never exceeded the concentration limits but approaches them with a difference in the order of magnitude 10^{-3} ng/L. When discussing concentrations with a certainty of 10^{-3} ng/L, it is of relevance to notice the large number of sources of errors that could have a significant impact on the results, where several are included in the saturation curve that is used as a basis for all GAC reactivation frequency. As discussed previously, the uncertainty when analyzing PFAS is usually around 30-40%, which is included in the measurement used to produce the saturation curve. Also, the saturation curve is an approximation based on eight measurements in a pilot trial, where the curve has a coefficient of determination of about 0.6. Therefore, it is probable that the actual saturation curve could differ from the approximated curve. The initial behavior of the GAC material before the first two months and after 18 months are not based on actual measurements as it only is extrapolated from the pilot results. It is possible that the behavior in its fresh and fully saturated condition could differ from the behavior shown in the measurements in its partially saturated state.

The calculations on the reactivation frequency also contain other sources of errors. The scale-up is done based on average flows that are dimensioned for NFWP. In reality both the flow to the process will vary, where a larger flow will require a reduction demand and increased reactivation frequencies. In addition, it is assumed that the reactivation will require exactly 1 month, which is not entirely true when regarding the reactivation time consumption presented in Table 4. Lastly, the inlet concentration was set to the highest raw water concentration measured during the last 5 years at Görvålverket. Therefore, no margin to the limit was applied, as it is probable that the inlet concentration most often will be lower leading to a lower reactivation requirement. Although the sources of errors could have a substantial impact, the results should be comparable between the different scenarios as all scenarios should have been impacted equally.

The numbers presented in Table 24 are only relevant for comparison between the alternatives because of the delimitations and assumptions made in the calculations. The energy requirement does not illustrate the total demand as it has been assumed that all power required beyond the DAF processes would be equally large for both process options. This is also valid for the cost calculations as the delimited energy requirements are included in the cost. The carbon emissions have been delimited to exclude all emissions that cannot directly be connected to Norrvatten's process except for the GAC reactivation. This delimitation may not provide an entirely true image of the reality, especially as the incineration of DAF sludge and AIX is excluded, which may entail large carbon emissions.

From the calculations on the energy requirement, it can be concluded that the energy consumption will be over 3000 MWh/y larger when DAF is included in NFWP compared to when it is excluded. For comparison, Görvålverket consumed 22 400 MWh in 2021 [98]. Thus, the energy consumption from DAF in NFWP would consist of 14% of the total energy consumption of the current treatment process in Görvålverket. It is known that DAF is an energy-intensive process, thus it is expected that the energy consumption would constitute a significant part of the total energy requirement.

The energy consumption for the secondary DAF on the sludge is determined based on a linear estimation from the measurements on line 5 in Görvålverket. In the study performed by Robey et al. [29] who achieved a removal efficiency of 90%, a significantly higher percentage of air was used compared to the basins in Görvålverket. The study was performed on 750 mL of leachate with an airflow of 2.6 L/min. Assuming the same amount of air used in line 5 for the sludge yields an airflow of 2600 L/min for a sludge flow of 1000 m³/d, which corresponds to 694 L/min. An increased airflow could have ensured a higher removal efficiency. However, it is possible that the air addition would become prohibitively large relative to the water volume in a full-scale application. If the air addition would be increased, the energy requirement would become larger in NFWP compared to what has been calculated in this report. In that case, it would favor the option of excluding DAF in terms of costs and possibly carbon emissions if the energy is not entirely fossil free.

The carbon emissions do also entail several sources of errors. The emissions due to the transport of sludge are based on a small truck that runs on diesel, which is used every week to transport the sludge. It is possible that the transports will be performed less often or on another fuel, which would affect the carbon footprint. The same is also valid for the transport of GAC material, which is estimated as a truck with a trailer. Also, the emissions related to reactivations are based on literature and not actual measurements at Chemviron. Lastly, the emissions at Fortum are not included in the incineration. The carbon emissions were not investigated in the sensitivity analysis as the results differed significantly between the reactivation emissions and the emissions due to sludge treatment. In addition, due to the cursory calculations, a sensitivity analysis was not assumed to be of relevance. It is probable that the large difference may be due to the delimitation that may favor the DAF process. Because of limitations in time, this was not investigated further. However, the emissions can be affected in several ways in 2050 due to for example electric cars, carbon capture, or other new technology. In addition, it is possible that the locations of various manufacturers or waste management companies may change leading to different carbon emissions.

The choice of sludge management technique was based on the possibility of reaching 5 ng/l in combination with the available data. Because of the present data on removal efficiencies and volumetric flow division, it was possible to perform an estimation of the potential of that process when a second DAF step was implemented. As has been presented in the delimitations, the sludge treatment option presented in this report may not be the most suitable or efficient technique regarding cost, carbon footprint, or energy requirement. These techniques have been excluded due to a lack of data on either process conditions or economics. The sludge treatment techniques that have been identified as a potential alternative would require more investigation to determine its effectiveness in the process.

One aspect that was investigated was the possibility to use GAC for the permeate after DAF2. To provide a competitive alternative to the GAC process, the GAC reactivation frequency in sludge management has to minimally be less than the reactivation frequency in the GAC process. By comparing the difference in the reactivation frequency between the process including DAF and excluding DAF, it was found that approximately 12-14 filters per year are the maximum amount that can be used in the sludge management if the inclusion of DAF should have a chance to be competitive. It was therefore calculated how much lower the flow must be to require the reactivation of approximately one filter per month. It was found that the flow would have to decrease from 1000 m³/d to 240 m³/d, which is a significant decrease in the flow that cannot be achieved by optimization.

In addition, the required removal efficiency in DAF2 to reach a reactivation frequency of 1 filter per month was investigated. It was found that a removal efficiency of 93% would be required resulting in an inlet concentration of 21 ng/L to the GAC filters. Whether this efficiency is possible to reach must be determined with pilot investigations of either DAF or foam fractionation. However, GAC would still not offer a cost-competitive solution compared with FLUORO-SORB even if it reached a required reactivation frequency of 1 filter per month.

From the sensitivity analysis, it is not possible to see what parameters that gave the greatest impact on the operational cost. However, the process including DAF entails large fixed costs in terms of sludge incineration. The sensitivity analysis disclosed that the operational cost for DAF can be significantly decreased through optimizations, especially when the volume of concentrated sludge is decreased. With a minor decrease in sludge volume compared to what has been included in the initial cost calculations, the DAF process can become more economically favorable than the process excluding DAF.

8.3 Applicability of the Results in NFWP

The results on the removal efficiency by DAF should be applicable for NFWP as the new process will have the same raw water inflow and similar amounts of added dissolved air in relation to the treated water volume. However, there are two aspects that may differ between the processes that may affect the removal efficiency. Firstly, the DAF will be placed after lamellar sedimentation in-

stead of the regular sedimentation that is applied in Görvålverket. The change in sedimentation technique may affect the amount of solids that are present in the sludge. Lamellar sedimentation is a more efficient technique compared to conventional sedimentation. Therefore, there is a need to investigate if the solids have an impact on the separation efficiency. For example, it has been speculated that the solids may affect the stability of the DAF sludge, leading to reduced PFAS removal for a lower solids content in the water.

Secondly, the calculations on the contact time revealed that the planned process in NFWP will have a significantly longer contact time than what is present in line 5 in Görvålverket. A longer contact time should benefit the separation. However, if the contact time reaches very high values, it is possible that the PFAS will desorb from the microbubbles and return to the water. The calculations showed that it is majorly the vertical contact time that is increased, while the horizontal contact time is more similar to line 5. A significantly longer contact time in the horizontal dimension may provide a more negative impact on the removal efficiency. This is because an extended contact time in the separation zone may lead to a higher level of PFAS re-entering the water as the concentration of PFAS along the basin would be decreased. The lower concentration would provide new equilibrium relationships that could favor PFAS desorption.

Comparatively, a longer contact time in the contact zone could endure a higher probability of PFAS attaching to the bubbles and enhancing the removal. McCleaf et al. [30], found that the contact time had an optimum value where a very short or long contact time decreased the removal efficiency. Increased removal efficiency at longer contact time could also be identified in the analysis as the lowest flows gave the best removal efficiency. However, the amount of microbubbles in relation to water is also affected when changing the flow of water, making it uncertain how much the contact time affects the removal. The effect from the contact time depends on the time point where equilibrium occurs, which is presently unknown for the basins in Görvålverket and NFWP. Therefore, further studies are of relevance to ensure that the longer contact time does not compromise the removal efficiency.

In addition, the calculations and estimations have not been performed for recirculation from the ultra-filters and sand filters or an ozonation step, which may be included in NFWP. Ozonation should affect the PFAS concentration according to Lundh et al. [23], if it is implemented it could be of relevance to investigate if ozonation and DAF can reach the concentration limit and investigate the effects it may have on the reactivation frequency of the GAC filters.

Pettersson et al. [72], argue that there is a lack of methods to treat PFAS contaminated areas as well as handle soil and water with high PFAS concentrations. Therefore, large investments are made on both national and international levels to develop new approaches [72]. With the new regulations on the allowed PFAS limit in drinking water, the demands on several actors have increased. This is likely to increase the research frequency on PFAS removal techniques that can be applicable for these purposes. From more extensive regulations on several actors in society, it is reasonable that more sustainable disposal options may be present in the future. Therefore, there might be of relevance to investigate new treatment options before implementing the sludge treatment in NFWP. These new methods could be more efficient and economical than the present solutions and must be evaluated similarly to the presented sludge management options. In addition, there is a need for further investigation including optimization of the DAF, verification of the relationships including increased removal at higher inlet concentration, effects from differing conditions that can appear in the waterworks due to for example seasonal changes, and identification of an explanation to the deviations in the sludge analysis results to ensure trustworthy measurements in the future.

9 Conclusion

From the measurements, it could be concluded that the separation efficiency that could be achieved in basin 5:1 was between 19-38%. The removal yielded an outlet concentration of 3-4.1 ng/L for an inlet concentration of 4.3-5.5 ng/L. From the results, it could not be concluded that DAF alone could ensure reaching the limit from the Swedish Food Agency. The removal efficiency increased with the lower inflow of water and a higher inlet concentration of PFAS₄, which is consistent with literature. The sludge from line 5 obtained a concentration of 200-700 ng/L, with less than 5 ng/L on the solids. However, the mass balance implied that the measured sludge concentration was too high in some samples, and the measurement uncertainty illustrated large impacts on the reanalysis results.

A sludge management alternative was decided based on literature to be a secondary DAF to receive a concentrated sludge and a permeate with a lower concentration. The concentrated sludge could be incinerated at Fortum Waste Solutions in Kumla and the permeate could be treated with the AIX material FLUORO-SORB, which needs to be incinerated after usage. After treatment of the permeate, the water can be released to Mälaren in accordance with the internal regulations at Norrvatten. However, pilot investigations are necessary to completely evaluate the efficiency of a secondary DAF and FLUORO-SORB. The sludge management method should also be compared to other alternatives, including SAFF and membranes to ensure that the most sustainable solution is implemented.

The reactivation frequency of GAC decreases if DAF is included in the process by 12-19 filters per year depending on the concentration limit and time consumption for reactivation. When DAF is excluded and a strict PFAS concentration limit is applied, it can become relevant to optimize the reactivation time consumption to reach the limit. The energy requirement becomes approximately 3600 MWh larger per year for the process including DAF, while the carbon emissions are reduced by 1500-2100 ton/y depending on the PFAS limit. The amount of PFAS that is destroyed in the process is smaller when DAF is included. However, the difference was calculated to be less than 2%. It was found that the concentration limit is a significant factor when determining what process alternative that is most cost-efficient. To exclusively apply GAC becomes more economical with a limit of 4 ng/L but can be substantially more expensive with a limit of 3 ng/L, especially if the DAF process is optimized. The sensitivity analysis showed that several factors can impact the results on the economics. However, the amount of sludge that was sent off for incineration was an aspect that significantly affected the outcome, even for minor changes in flow. Finally, the results suggest that incorporating DAF into NFWP could offer potential benefits. Nevertheless, additional research is required to establish the extent of its positive impact.

10 References

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Appendices

A Sampling Flow Sheets

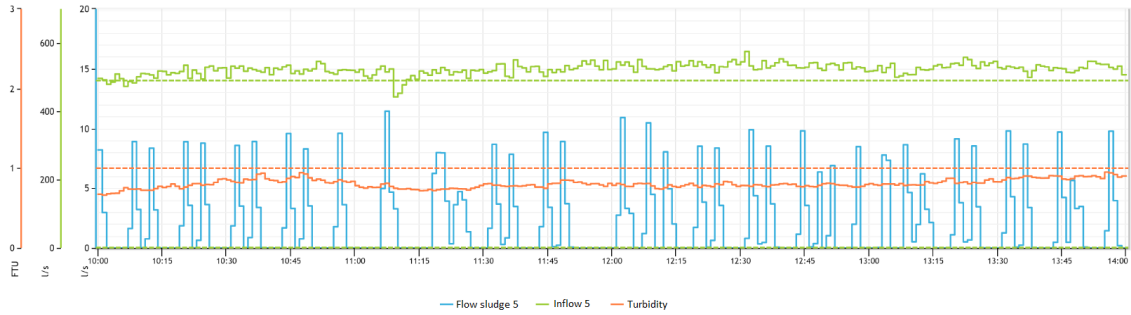


Figure 34: Flow during sampling 2023-02-14.

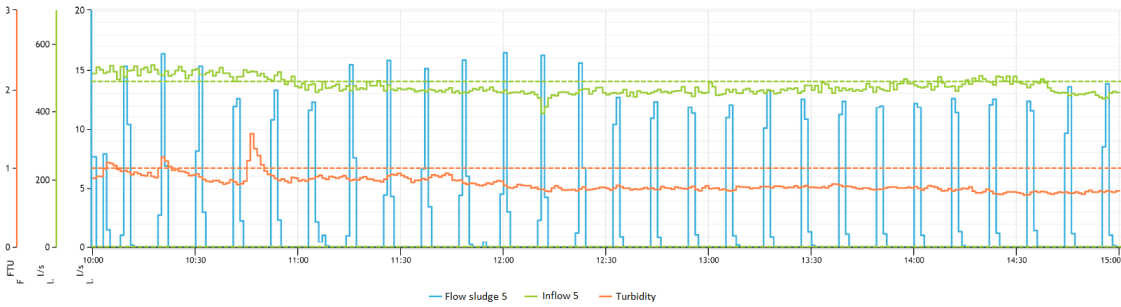


Figure 35: Flow during sampling 2023-02-15.

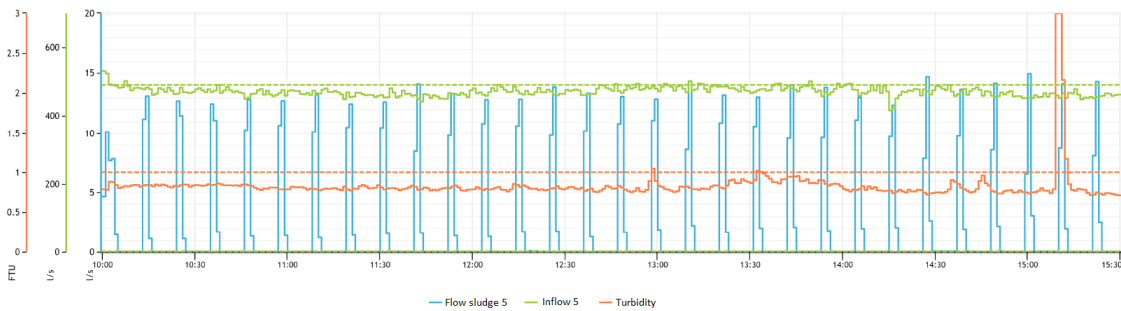


Figure 36: Flow during sampling 2023-02-17.

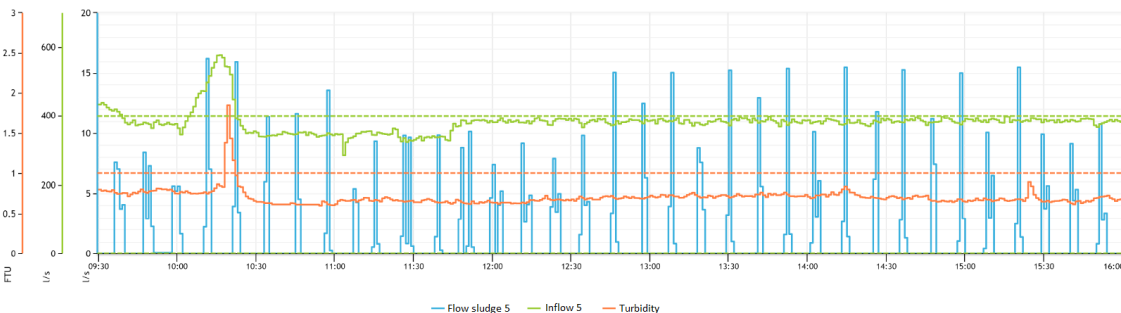


Figure 37: Flow during sampling 2023-02-21.

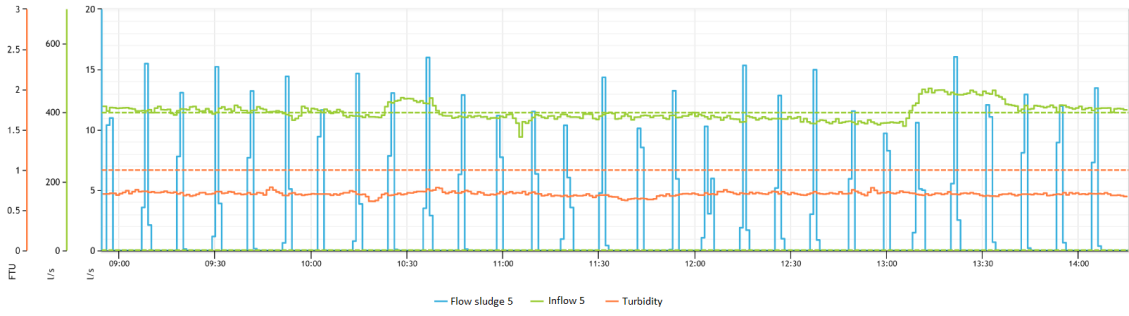


Figure 38: Flow during sampling 2023-02-22.

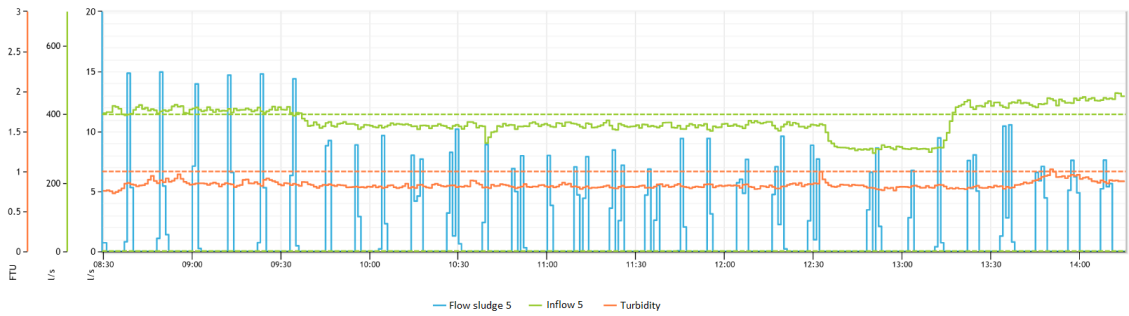


Figure 39: Flow during sampling 2023-02-24.

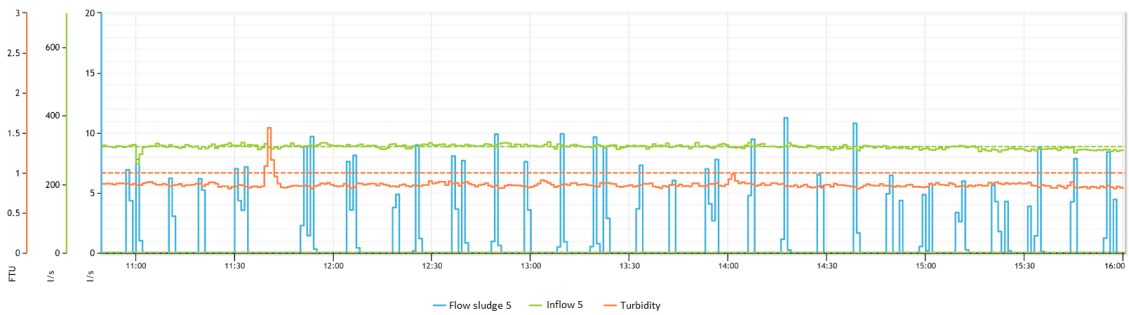


Figure 40: Flow during sampling 2023-03-07.

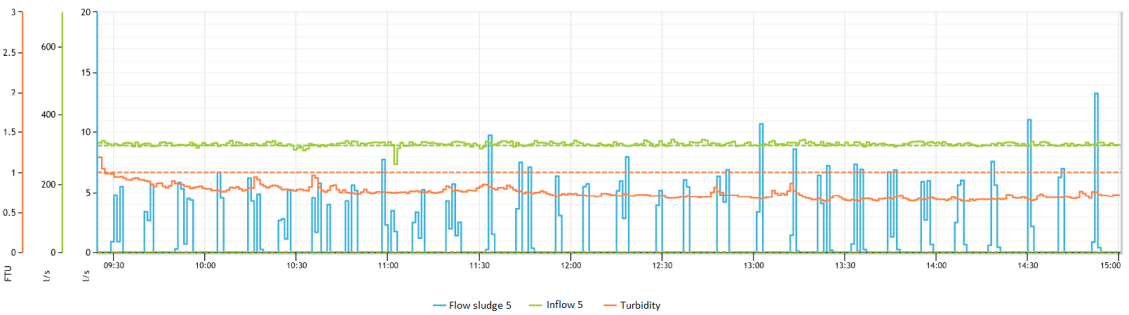


Figure 41: Flow during sampling 2023-03-08.

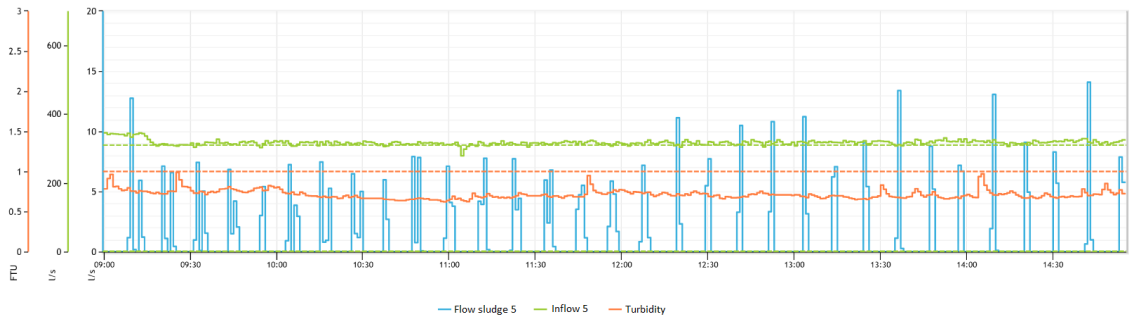


Figure 42: Flow during sampling 2023-03-09.

B Centrifuging Specifications

Table 25: Settings on Multifuge 3 s-r/3 l-r, legend rt 120v

Acceleration Profile	7
Deceleration Profile	9
Rotation speed	4000 rpm
Temperature	15 °C

Table 26: Specifications from centrifuging

	Collective 9/3	Grab 15/3	Grab 15/3
Amount of sludge [g]	2561	2784	2948
Mass after centrifuging [g]	66	75	72
Mass after drying [g]	5.7	6.9	6.8

C Calculation of Contact Times

The contact time in line 5 was calculated as the following: where A is the area in m² and h is the height, l is the basin length and w is the basin width in m.

$$CT_{tot} = CT_{horizontal} + CT_{vertical} \quad (13)$$

$$CT_{vertical} = \frac{A_{dispersion\ inlet}(h_{water\ surface} - h_{dispersion\ inlet})}{\dot{V}_{water} + \dot{V}_{dispersion}} \quad (14)$$

$$CT_{horizontal} = \frac{\text{Bubble layer thickness} \cdot \text{length} \cdot \text{width}}{\dot{V}_{water} + \dot{V}_{dispersion}} \quad (15)$$

In a similar manner, CT was calculated for NFWP with the difference that the horizontal contact time was divided by two as the water will gradually drain off down to the sand filter.

$$CT_{horizontal} = \frac{\text{Bubble layer thickness} \cdot \text{length} \cdot \text{width}}{(\dot{V}_{water} + \dot{V}_{dispersion}) \cdot 2} \quad (16)$$

D Total Solids Content in Sludge

The TS content is an important parameter to understand the physical properties of a sample and is of great importance when investigating the sludge management strategies. To determine the TS content, the sludge should be dried in an oven for 24 hours at 103 °C. Then the quota of the mass of the dried sample and the original mass gives the TS content in percentage according to Equation 17 [48].

$$TS = \frac{\text{Mass of solids}}{\text{Original mass}} \cdot 100\% \quad (17)$$

However, the drying temperature was set to 37 °C to not interfere with the PFAS content in the sample in consultation with the analysis company Eurofins [49]. The difference of TS content at 103 and 60 °C has been estimated to 2% [50], which can give an estimation on the error from the lower temperature used in this investigation.

E Calculation of PFAS on Solid Particles

Table 27: Calculation of PFAS content on solids excluding amount from dried water

	Collective 9/3	Grab 15/3(10am)	Grab 15/3(11am)
Water amount before drying [g]	58.66	30.416	34.537
PFAS concentration in water phase [ng/L]	540	195	245
PFAS amount in dried water [ng]	31.7	5.93	8.45
PFAS in dried sample [ng]	25.1	17.9	19.6
PFAS on solids [ng]	-6.6	12.0	11.2
Centrifuged sludge [kg]	2.56	2.78	2.95
PFAS on solids in sludge [ng/L] *	-2.6	4.3	3.8

* Assuming density of sludge of 1 kg/L

F FLUORO-SORB 200 Saturation Data

Table 28: Reduction efficiency for the time consumption in NFWP based on the study by Pannu and Plumlee.

Reduction	Time in NFWP [months]
0.981	0.29
0.981	0.58
0.981	0.87
0.981	1.16
0.981	1.45
0.981	1.74
0.943	2.03
0.981	2.32
0.943	2.61
0.981	2.90
0.904	3.48
0.885	3.77
0.828	4.93
0.808	5.22
0.799	6.67

Table 29: Extrapolation from fitted saturation curve for FLUORO-SORB 200.

Time [months]	Reduction degree [%]
2	99.8
4	95.7
4.5	94.9
5	94.3
9	90.8
10	90.1
12	89.0
14	88.1
16	87.3
18	86.6
20	85.9
22	85.4
24	84.8

G Filtrasorb 400 Saturation Data

Table 30: Extrapolation of saturation curve for drinking water treatment.

Time [months]	Degree of reduction [%]
0.5	89.9
1.0	78.7
1.5	72.1
2.0	67.4
2.5	63.8
3.0	60.9
3.5	58.4
4.0	56.2
4.5	54.3
5.0	52.6
5.5	51.0
6.0	49.6
6.5	48.3
7.0	47.1
7.5	46.0
8.0	45.0
8.5	44.0
9.0	43.0
9.5	42.2
10.0	41.3
10.5	40.6
11.0	39.8
11.5	39.1
12.0	38.4
12.5	37.7
13.0	37.1
13.5	36.5
14.0	35.9
14.5	35.3
15.0	34.8
15.5	34.2
16.0	33.7
16.5	33.2
17.0	32.7
17.5	32.3
18.0	31.8
18.5	31.4
19.0	30.9
19.5	30.5
20.0	30.1
20.5	29.7
21.0	29.3
21.5	28.9
22.0	28.6
22.5	28.2
23.0	27.8
23.5	27.5
24.0	27.1
24.5	26.8

Table 31: Extrapolation of saturation curve for sludge treatment.

Time [months]	Reduction degree [%]
0.2	97.8
0.4	86.5
0.6	80.0
0.7	77.5
0.8	75.3
1.0	71.7
2.0	60.4
3.0	53.9
4.0	49.2
5.0	45.6
7.5	39.0
10.0	34.4
12.0	31.4

I Calculation of Operational Costs

Excluding DAF. 4 ng/L			Excluding DAF. 3 ng/L		
Module	Activity	Cost [SEK/y]	Module	Activity	Cost [SEK/y]
GAC	Reactivation	15504000	GAC	Reactivation	35088000
	Addition of fresh GAC	2930256		Addition of fresh GAC	6631632
	TOTAL	18434256 SEK/y		TOTAL	41719632 SEK/y
		18 MSEK/y			42 MSEK/y

Including DAF. 4 ng/L			Including Flotation. 3 ng/L		
Module	Activity	Cost [SEK/y]	Module	Activity	Cost [SEK/y]
GAC	Reactivation	2040000	GAC	Reactivation	15504000
	Addition of fresh GAC	385560		Addition of fresh GAC	2930256
	DAF water treat.	Energy requirement		2975635	DAF water treat.
Sludge management	DAF energy requirement	14699	Sludge management	DAF energy requirement	14699
	Transport to incineration	721240		Transport to incineration	721240
	Incineration	13395500		Incineration	13395500
	AIX purchase	467857,6		AIX purchase	467857,6
	AIX incineration	60834		AIX incineration	60834
	AIX transport to NFWP	14886		AIX transport to NFWP	14886
	AIX transport to incineration	36986,66667		AIX transport to incineration	36987
TOTAL	20076212 SEK/y	TOTAL	36084908 SEK/y		
		20 MSEK/y			36 MSEK/y

J Calculation of Carbon Emissions

Only GAC, 4 ng/L			Only GAC, 3 ng/L		
GAC reactivation	number of filters	15,2	GAC reactivation	number of filters	34,4
	Emissions	1674432 kg CO2/y		Emissions	3789504 kg CO2/y
GAC transport	total number of trucks	18,2	GAC transport	total number of trucks	41,2
	Emissions	37927 kg CO2/y		Emissions	85834 kg CO2/y
TOTAL		1712359 kg CO2/y 1712 ton CO2/y	TOTAL		3875338 kg CO2/y 3875 ton CO2/y
DAF + DAF2 +incineration + AIX, 4 ng/L			DAF + DAF2 +incineration + AIX, 3 ng/L		
GAC reactivation	number of filters	2	GAC reactivation	number of filters	15,2
	Emissions	220320 kg CO2/y		Emissions	1674432 kg CO2/y
GAC transport	total number of trucks	2,4	GAC transport	total number of trucks	18,2
	Emissions	4990 kg CO2/y		Emissions	37927 kg CO2/y
DAF + DAF2	0 emissions: fossile free energy		DAF + DAF2	0 emissions: fossile free energy	
Incineration secondary sludge	number of trucks	52	Incineration secondary sludge	number of trucks	52
	Emissions	3448 kg CO2/y		Emissions	3448 kg CO2/y
AIX	transport new AIX	1252 kg CO2/y	AIX	transport new AIX	1252 kg CO2/y
	transport used AIX	176,8 kg CO2/y		transport used AIX	176,8 kg CO2/y
	Emissions	1429 kg CO2/y		Emissions	1429 kg CO2/y
TOTAL		230187 kg CO2/y 230 ton CO2/y	TOTAL		1717235 kg CO2/y 1717 ton CO2/y

K Calculation of PFAS Destruction Amount for the Different Processes

Table 41: Calculation of destructed PFAS for different scenarios

	4 ng/L		3 ng/L	
	Excl. DAF	Incl. DAF	Excl. DAF	Incl. DAF
PFAS4 in [g/d]	1.14	1.14	1.14	1.14
PFAS4 out [g/d]	0.80	0.60	0.80+0.005	0.60+0.005
PFAS4 destructed [g/d]	0.341	0.336	0.542	0.537
PFAS4 destructed [g/y]	125	123	198	196

L Sludge Concentration according to Mass Balance

Table 42: Mass and concentration of the sludge as the difference of the inlet and outlet. The deviation is the percentage difference between the mass balance and analytical concentration.

Date	m_{sludge} (mass balance) [mg/d]					c_{sludge} (mass balance) [ng/L]					Deviation [%]
	PFOA	PFOS	PFNA	PFHxS	Sum	PFOA	PFOS	PFNA	PFHxS	Sum	
14-feb	4.5	36.2	3.2	9.0	54.2	23.3	187	16.3	46.7	280	-0.2
15-feb	4.1	20.6	2.5	5.4	32.9	20.7	103	12.4	26.9	165	1.3
15-feb Re	4.1	20.6	2.5	5.4	32.9	20.7	103	12.4	26.9	165	0.6
17-feb	8.1	28.5	0.0	8.1	44.8	40.1	141	0.0	40.1	221	-0.4
21-feb	0.0	32.9	2.6	6.6	42.8	0.0	224	18.0	44.9	292	1.4
21-feb Re	0.0	32.9	2.6	6.6	42.8	0.0	224	18.0	44.9	292	1.0
22-feb	4.7	38.0	1.2	8.8	52.6	34.9	283	8.7	65.4	392	-0.4
24-feb	1.0	26.2	3.6	3.3	32.8	6.7	179	24.6	22.4	224	0.3
07-mar	3.2	40.1	2.1	4.3	48.1	29.4	367	19.6	39.2	440	0.2
08-mar	6.0	46.2	4.6	0.0	57.1	53.8	416	41.6	0.0	514	0.2
09-mar	2.7	24.7	2.7	-2.7	27.4	24.4	220	24.4	-24.4	244	1.4