PFAS – how can Swedish wastewater treatment plants meet the challenge?
Compilation of knowledge and guidance for water/wastewater actors regarding PFAS
Report number: B2478

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Preface

Per- and Polyfluoroalkyl Substances (PFAS) are a group of chemicals that has been in discussion for some time now but appears still to be a challenge for many water sectors. What exactly is the problem? How are we affected? What can we do and what should we do? Significant work by organization such as Svenskt Vatten (Swedish Water and wastewater Association) and Swedish Chemicals Agency (KEMI) has shed light on various aspects. However, there remains still ambiguities on organized knowledge of PFAS, and what roles and responsibilities the Swedish wastewater treatment plants have in this societal challenge.

At the initiative of several water operators, the idea was raised on compilation of knowledge and guidance on the PFAS problem with a focus on the areas of the wastewater treatment plant operations. The work has included a comprehensive review of the PFAS issues to understand the problem, to assess the environmental impact, how to analyze and map the work that are carried out, what role wastewater treatment plants has as a transport route, sludge management, and possible remediation measures including various treatment techniques.

The report provides an overview of the state of knowledge, compiled by the IVL Swedish Environmental Research Institute in collaboration with Lutz Ahrens from SLU Swedish University of Agriculture and Sahar Dalahmeh from Uppsala University. Our ambition was to produce a consolidated and concise description of the situation, but the report has become more comprehensive than the project group anticipated. This is mainly due to the complexity of the PFAS challenge, but also because there is a lot of useful knowledge that needs to be passed on to Swedish Water sectors and more. During the work, it has become clear that there are gaps in knowledge that need to be addressed and investigated. The report also highlights the rapid developments in the area, for example in new regulations and technological developments. Given the challenges we face with PFAS in our society the entire compilation is a valuable reading. To facilitate understanding and to promote knowledge spread, the project group also intends to produce a summary brochure from the report and hold presentations.

The project group will especially express gratitude to the foundation SIVL, Stiftelsen Institutet för Vatten- och Luftvårdsforskning, and SVU, Svenskt Vatten Utveckling, as the main financiers of the project. We would also like to thank the participating organizations that contributed with their time and financial means, but above all, with their knowledge.
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Christian Baresel, Project Leader
Summary

Per- and Polyfluoroalkyl Substances (PFAS) are everywhere around us in society, found in commercial and industrial products, the atmosphere, waste, water (waste, surface, drinking, and ground), soil, plants, animals, and even in our bodies. The use and spread of PFAS is a global societal challenge, affecting even the most remote places on Earth. One of the reasons why PFAS has been an attractive component in many products and industrial applications is their extreme chemical and thermal stability. However, these same properties allow for the persistence of PFAS in the environment, whereby even low PFAS emissions over time can be accumulated and pose a high risk of negative health and environmental effects. Today there are thousands of known and unknown PFAS with widely varying properties and toxicity, which makes both risk assessments and management of this growing environmental problem difficult.

According to a national mass balance for PFAS emissions from products and atmospheric deposition are the major sources of PFAS in Sweden. The amount of PFAS which are environmentally dispersed via wastewater and sewage sludge can be considered a minor part. However, due to the persistence of PFAS, measures to minimize their addition via these pathways may be necessary to reduce the total environmental load.

Due to their toxicity and persistence, PFAS have recently become heavily regulated, with many regulatory agencies lowering the accepted PFAS environmental level ranges. Many PFAS have already been banned in Sweden or the EU and assessment grounds or action limits have been defined for various PFAS (e.g., surface water bodies, groundwater and drinking water) to initiate measures to reduce the spread of the substances. Already announced and stricter regulations will further increase the need for measures to minimize human exposure to PFAS and their dispersal in the environment. Regardless of which measures are implemented, PFAS will remain in the environment for a long time, even if a global ban of the chemicals is implemented. Long-term management of PFAS is thus necessary to removal from the cycle gradually. The focus of mitigation actions should primarily be on heavily contaminated land and landfill leachate.

The review of existing data from Swedish wastewater treatment plants WWTP and receiving recipients shows that today’s treatment processes do not remove PFAS. At some WWTP, however, an effective separation of certain PFAS is observed,
which should be investigated further. Perfluorooctanoic sulfonic acid (PFOS) levels in many of the investigated inland surface waters receiving effluent from treatment plants and PFAS from other sources/pathways exceed existing limits. In many cases, however, analysis limitations prevent an assessment.

Ongoing activities around various treatment and destruction techniques for PFAS show that there are currently no techniques that achieve a far-reaching PFAS removal from municipal wastewater without significant resource consumption and related costs. For the continued use of sludge as a fertilizer, upstream mitigation is needed, with e.g., disconnection or treatment of PFAS-contaminated leachate. However, several ongoing projects indicate that a certain part of PFAS in wastewater can be removed as a side-effect of advanced treatment for pharmaceutical removal.

This report provides guidance to stakeholders on how the PFAS problem can be tackled. It also demonstrates the need to improve and spread PFAS knowledge, particularly those involved with PFAS measurement data, treatment techniques and PFAS in sludge.
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1 Background: What is PFAS?

Per- and polyfluorinated alkyl substances (PFAS) are a versatile group of synthetic fluorinated compounds. Due to their unique surfactant properties and high chemical and thermal stability, PFAS have been widely used in various industrial processes and in products that we use in our daily lives. They provide properties such as non-stick, water-repellent and dirt/grease-repellent to many household products (e.g., frying pans, cleaning agents, paint, textiles, beauty products, etc...). Their extensive use in long-life household products, such as carpets, furniture, and paints, means that they exist and remain in the environment long after production. PFAS have also been found globally in surface water, sediment, air, soil, sludge, and ice sheets, as well as in plants, animals, and humans.

Based on the Organization for Economic Cooperation and Development (OECD 2021), there are currently ca. 4,730 known substances that belong to the large PFAS family. The United States Environmental Protection Agency (EPA) has made an inventory of PFAS and listed over 10,000 PFAS in its EPA CompTox Chemicals Dashboard (Williams et al., 2017). The most studied and used PFAS in industry is Perfluorooctanoic acid (PFOA) (CAS no. 335-67-1) and Perfluorooctanesulfonic acid (PFOS) (CAS no. 1763-23-1).

PFAS have received increased attention due to their potential to bioaccumulate (resulting from their stable molecular structure), their environmental persistence (being easily dispersed via stormwater from contaminated land areas and effluents from municipal and industrial wastewater), and toxicity documented for some PFAS. As PFAS are not removed using conventional treatment techniques, they end up contaminating surface and groundwater, sources often used for drinking water. A recent article by researchers from Stockholm University concluded that PFAS have exceeded a planetary boundary for novel entities and therefore requires immediate counteractive measures (Cousins et al., 2022).

1.1 Nomenclature

PFAS are a broad group of chemical substances that contain several fully fluorinated (per fluorinated) or partially fluorinated (polyfluorinated) carbon atoms. PFAS are divided into different categories depending on the structures of the molecules and functional groups (Figure 1.1). Figure 1.1 also depicts precursors/precursor compounds that include PFAS, which can be degraded via various processes into other PFAS such as Per fluorinated alkyl acids (PFAA).

As previously mentioned, there are several thousand different PFAS and their
properties vary greatly. Properties of the most common PFAS are summarized by the Swedish Chemical Agency (KEMI, 2021a).

The sum of several most common PFAS such as PFAS4, PFAS11 and PFAS20/21 are usually used to describe certain PFAS that are classified as a group in regulations (see section 3.2). The sums are also used as a designation in the analysis of PFAS substances, where the determined sum means that the levels of the various PFAS have been added up. The designation "sum PFAS" is also often used in various publications and usually refers to the combined concentration/quantity of the PFAS substances analyzed for the specific project.

The most common PFAS groupings are:

- **PFAS4**: PFOA, PFNA, PFHxS and PFOS
- **PFAS7**: PFPeA, PFHxA, PFHpA, PFOA, PFBS, PFHxS, and PFOS
- **PFAS11**: PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFBS, PFHxS, PFOS, 6:2 FTSA (6:2 Fluorotelomer sulfonic acid, even 6:2 FTS used in occasions)
- **PFAS20**: PFAS11 except 6:2 FTSA; and PFUnDA, PFDoDA, PFTrDA, PFPS, PFHpS, PFNS, PFDS, PFUnDS, PFDoDS and PFTrDS
- **PFAS21**: PFAS20 + 6:2 FTSA
- **PFAS24 equivalent RPF for EQS**: PFHxS, PFOS, PFOA, PFNA, PFBS, PFHxA, PFBA, PFPeA/PFPA, PFPeS/PFPS, PFDA, PFDoDA/PFDoA, PFUnDA/PFUnA, PFHpA, PFTrDA, PFDS, PFPeDA, PFHpS, PFHxS, PFHxA, PFODA, HFPO-DA/GenX, ADONA, 6:2FTOH, 8:2FTOH, C6O4
- **PFAS Total**: Some laboratories already offer PFAS Total analysis that includes 50 PFAS: PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFDA, PFUnDA, PFDoDA, PFTrDA, PFTeDA, PFHxDA, PFODA, PFPS, PFBS, PFPeS, PFHxS, PFHpS, PFOS, PFNS, PFDS, PFUnDS, PFDoDS, PFTrDS, 4:2 FTSA, 6:2 FTSA, 8:2 FTSA, 10:2 FTSA, EtFOSAA, MeFOSAA, HFPO-DA (GenX), FOSA, MeFOSA, EtFOSA, MeFOSE, EtFOSE, 6:2 CI-PFAES, 8:2 CI-PFAES, DONA, PFHxSA, PFMPA, HPFHpA, HFEHS, PFMBA, PFEESA, NFDHA, 3:3 FTCA, 5:3 FTCA, 7:3 FTCA and PFBSA

They are used in the new drinking water directive, where it is also stated that technical guidelines for the parameter need to be first drawn up.

- **ΣPFAS (sum PFAS)** i.e., ΣPFAS4, ΣPFAS11, ΣPFAS21 etc.: The sum of the concentrations for the substances included in the PFAS grouping. Most often, only concentrations above the reporting limit (detection or quantification limit in the analysis) are included.
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It is not essential to know the names of all PFAS. The importance is to understand the general structure of the most common PFAS covered by the current regulations and how they are referred to. Figure 1.2 shows structural formulas for three common PFAS. What these three molecules have in common is that they include a carbon chain with six fully fluorinated carbon atoms. The first molecule (PFHpA) has a carboxyl group (-COOH) at one end and belongs to the group of perfluorinated carboxylic acids (PFCA). The second molecule (PFHxS) has a sulfonyl group (-SOOOH) at one end and therefore belongs to the PFSA group (perfluorinated sulfonic acids). The groups PFCA and PFSA are collectively referred to as perfluoroalkyl acids (PFAA). The length of the carbon chain of PFAA is indicated in the name by the Greek numbers from at least four atoms (butane, letter B in the abbreviation, e.g., PFBA) to up to 13 carbon atoms (tridecane, TrD in the abbreviation, e.g., PFTrDA). Short-chain PFCA are written as C_nF_{2n+1}COOH with fewer than seven fluorinated carbon atoms, in addition to the carboxyl group. Long-chain PFCA have the corresponding formula with seven or more fully fluorinated carbon atoms in addition to the carboxyl group, i.e., n ≥ 7. Short-chain PFSA can be described with C_nF_{2n+1}SOH where is n < 6, whereas long-chain PFSA with C_nF_{2n+1}SOH but with at least six fully fluorinated carbon atoms. It is worth noting that PFHpA and PFHxS are different numerals despite the same length of
fluorinated carbon chains (six fluorinated carbon atoms). This is because PFHpA also contains one carboxyl group, and all carbon atoms are counted instead of only fully fluorinated carbon atoms. Hepta in PFHpA, a total of seven carbon atoms and hexa in PFHxS, a total of six carbon atoms, Figure 1.2). The last molecule in Figure 1.2 is a fluorotelomer acid that also has two non-fluorinated carbon atoms the closest to the sulfonyl group. Telomeric acids are named with numbers of the quantity of fluorinated carbon atoms and non-fluorinated carbon atoms, separated by a colon (e.g., 6:2 FTSA). All 21 PFAS covered by current regulations (see chapter 3.2) are included in these three groups of PFAS (PFCA, PFSA and fluorotelomer acid (FTSA)) and their names are built according to the rules described above.

![PFAS structural formulas](image)

**Figure 1.2 Examples of PFAS structural formulas.**

### 1.2 PFAS properties

Many PFAS molecules contain both a water-soluble hydrophilic part: the functional group and a water-repellent hydrophobic part: the fluorinated carbon chain. The fluorinated carbon chain can either be linear or branched, of which the chain length determines its water solubility (i.e., higher solubility with longer chains).

If the functional group consists of an acid or another group that can bind or release a hydrogen ion, the electrical charge of PFAS molecule changes with the pH value of the water. The charge can change from negative (anion) to neutral to positive...
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(cation). Some PFAS can also be zwitterionic, i.e., one part of the molecule is positively charged, while another part is negatively charged. In natural water, all PFAAs and FTSAs are negatively charged.

Less stable PFAS precursor compounds can form other PFAS such as PFCA and PFSA through abiotic or biotic transformation. Precursors such as N-ethyl-N-(2-hydroxyethyl) perfluorooctylsulphonamide (EtFOSE), N-ethylperfluorooctane sulfonamidoacetic acid (EtFOSAA) and fluorotelomer alcohol (FTOH) can be converted to PFOS or PFOA (Kannan 2011; Kunacheva et al., 2011; Mejia Avendaño and Liu, 2015). It is important to consider this conversion of precursors to PFCA and PFSA to understand, for example, why the levels of PFOS and PFOA can increase through wastewater treatment process (Bossi et al., 2008; Kannan, 2011; Kunacheva et al., 2011; Yang et al. 2014). Moreover, some volatile precursors such as EtFOSE can commonly spread in the atmosphere during aeration in the sewage treatment process. Larger PFAS-molecules such as polymers can even be formed in WWTP plant through a single chain PFAS being bound together by the biological treatment processes (Ahrens et al., 2011; Rhoads et al., 2008).

In treatment plants, PFAS are distributed between water, suspended particles and sediment. The distribution of PFAS in treatment plants was investigated in a sorption study by Ahrens et al. (2010). Short-chain PFCAs were detected exclusively in the dissolved phase in water, while long-chain PFCAs, PFOS, Perfluorooctanesulfonamide (FOSA) and Perfluorohexane sulfonate (PFHxS) were detected both in the dissolved phase as well as bound to suspended particles. Long-chain PFCAs (C >11) and Perfluorodecane sulfonate (PFDS) were exclusively detected in sediments (Ahrens et al., 2010). Similar results were observed in a study conducted by Higgins and Luthy (2006), who illustrated the carbon chain length increased the ability of PFAS molecules to bind to sludge. This can be referred to as the term log Koc, a measure of the sorption potential of a chemical in relation to the length of the carbon chain.

There is also a link between PFAS and microplastics, although this has not been studied in detail. PFAS can for example occur in fluorine-containing plastics such as polytetrafluoroethylene (PTFE, also known as Teflon), polyvinylidene fluoride (PVDF) and polyvinyl fluoride (PVF). However, these substances are generally not dangerous as long as they are not bioavailable. Microplastics containing PFAS precursors that are released from textiles with washing can also be a source of PFAS in wastewater (Schellenberger et al., 2019). PTFE and PVF can occur in the environment as secondary microplastics (fragmentation of larger plastic debris or
pieces of plastic) or as intentionally produced primary microplastics (Bergmann et al., 2017; Ebnesajjad 2013). Due to their high density, PTFE and PVF microplastics are more likely to end up in the sludge of sewage treatment plants. Microplastics and PFAS are often found together in the environment and recent research have shown that microplastics can increase PFAS toxicity, by, for instance, facilitating the transport of PFAS into organisms (Pramanik et al., 2020; Le Bihanic et al., 2020; Sobhani et al., 2021).

1.3 Health and environmental effects

Reports on the health and environmental effects of PFAS have increased and reported, not only in the scope of scientifically reviewed literature. These effects are linked both to toxic effects and the persistency of PFAS. Today, there are risks of toxic effects mostly of long-chain PFAS where a specific toxicity has been established, such as PFOS and PFOA.

Other PFAS, including short-chain PFAS, are often considered less prioritized because the known toxicity is lower than for long-chain PFAS. However, they can also lead to negative health and environmental effects in the long run due to their persistence (Gomis et al., 2018; Neuwald et al., 2022). They never disappear, or at most, break down very slowly. They therefore pose a risk for toxic effects in the environment when accumulated and their concentration increases.

A review by Fenton et al., (2021) shows that much of the available toxicity data for PFAS currently applies to a few chemicals, primarily the well-used PFAAs such as PFOA and PFOS. Epidemiological studies have revealed associations between exposure to specific PFAS and different health effects, including altered immune and thyroid function, liver disease, lipid and insulin disturbances, kidney disease, adverse reproductive and developmental health outcomes, and some forms of cancer. Many of these effects are verified and correspond to data results from animal experiments. The health effects have only been documented for a few PFAS compounds so far, and therefore the majority of the PFAS found today lacks toxicity data. Therefore, it can be stated that knowledge on the health effects of PFAS is still limited, including the knowledge on PFAS substitutes and PFAS mixtures (cocktail effects).

The PFAS levels found in relatively low background levels in food and drinking water do not cause acute health problems. However, persistent PFAS exposure can accumulate in the body for a long time and if exposed daily, it can lead to health risks in the long term. Drinking water that contains high amount of PFAS for a
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long time is suspected to increase the risk of adverse health effects. In food, it is mostly PFOS and PFOA that are found in the highest quantity. According to the European Food Safety Authority (EFSA) (2018), food contributed to 67 – 84 % of the total median PFOA intake and 88 – 99 % of the total PFOS intake. Similarly, the relative median contribution from drinking water ranged between 0.57 – 0.68 % for PFOS and 9.1 – 11 % for PFOA, respectively.

Available epidemiological studies suggest associations between human exposure to PFAS and several adverse health effects (e.g., Barry et al., 2013; Lopez-Espinosa et al., 2011; 2016):

- hypertension due to pregnancy (high blood pressure) /poisoning (PFOA, PFOS)
- liver damage, as evidenced by increases in serum enzymes and decreases in serum bilirubin levels (PFOA, PFOS, PFHxS)
- increases in serum lipids, especially total cholesterol, and low-density lipoprotein (LDL) cholesterol (PFOA, PFOS, PFNA, PFDeA)
- increased risk of thyroid disease (PFOA, PFOS)
- reduced antibody response from vaccines (PFOA, PFOS, PFHxS, PFDeA)
- increased risk of asthma (PFOA)
- increased risk of reduced fertility (PFOA, PFOS)
- testicular and kidney cancer (PFOA)
- lower sex hormone levels and growth factor in children (PFOA)
- delayed puberty (PFOS, PFOA for girls) and (PFOS for boys)
- small reductions in birth weight (PFOA, PFOS).

The toxicity of PFAS, especially PFOA and PFOS, has been evaluated in extensive studies in humans and laboratory animals. However, the knowledge of toxic effects in other species cannot be simply applied to humans as there are significant differences in the rate of elimination of PFAS between species. In addition, unlike many other classic environmental toxins which accumulate in body fat, PFAS binds to protein and circulates with blood. The types of proteins occurring in the metabolism can vary between species, and therefore the toxic effects will be different between different species. As an example, the half-life of PFOA in the human body is several years, whereas a few hours in female rats. Although many epidemiological studies have investigated the potential of perfluoroalkyl compounds to induce adverse health effects, it has not been made clear how the PFAS adverse health effects are. Malovanyy et al. (2021) have proposed factors for conversion to PFOS-equivalent content, based on various scientific studies on half-lives in humans for the most common PFAS (Table 1.1). A similar methodology for
compensation for different toxicity of different substances is also proposed in WFD
Circa (2021), which is discussed in more detail in chapter 3.2.3.

Table 1.1 Half-life of different PFAS in human blood represented by numbers of years (Malovanyy et al., 2021).

<table>
<thead>
<tr>
<th>PFAS</th>
<th>PFOS</th>
<th>PFOA</th>
<th>PFPeS*</th>
<th>PFHxS</th>
<th>PFHpS*</th>
<th>PFNA</th>
<th>PFBS</th>
</tr>
</thead>
<tbody>
<tr>
<td>Half-life (yr)</td>
<td>5.7</td>
<td>3.3</td>
<td>0.82</td>
<td>9.9</td>
<td>3.1</td>
<td>2.8</td>
<td>0.11</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>PFAS</th>
<th>PFBA*</th>
<th>PFHxA*</th>
<th>PFHpA*</th>
<th>6:2 FTSA*</th>
<th>PFPeA*</th>
<th>PFDA*</th>
<th>PFUnA</th>
<th>Gen-X</th>
</tr>
</thead>
<tbody>
<tr>
<td>Half-life (yr)</td>
<td>0.009</td>
<td>0.059</td>
<td>0.44</td>
<td>0.080</td>
<td>0.1</td>
<td>5.6</td>
<td>5.7</td>
<td>280</td>
</tr>
</tbody>
</table>

* = substances included in PFAS11.

The International Agency for Research on Cancer (IARC 2017) concluded that
PFOA is possibly carcinogenic to humans (Group 2B) and the EPA (2016a, b)
concluded that there was substantial evidence for the carcinogenic effect of PFOA
and PFOS in humans. Increases in testicular and kidney cancers have been
observed in humans with high exposures (ATSDR, 2018).

A study by the Nordic Council of Ministers (Goldenman et al., 2019) estimated
health-related cost caused by PFAS exposure to be 52 – 84 billion Euro annually in
Europe. This cost was assumed to be an underestimate, as it only includes costs
associated with a restricted number of health effects and a small number of PFAS.

In a more comprehensive study of PFAS in the environment, Kärrman et al. (2019)
showed that PFAS were found in samples from animals (including seabird eggs,
fish, marine mammals, and terrestrial mammals), water, and air. Documented
knowledge of the ecotoxicological effects of PFAS is often dominated to PFAA, and
above all to PFOS and PFOA. The PFAA are not assessed as acutely toxic
substances to aquatic organisms, but they have caused liver damage,
immunotoxicity, birth defects and hormonal disturbances in test animals such as
birds and mammals (KEMI 2021a).

It is also problematic when PFAS accumulate in living organisms and become more
concentrated as the food chain progresses to higher levels. Through consumption
of animals such as meat and dairy products, people are at the top of the food chain
and are therefore highly exposed. This has been taken into consideration in setting
the limit values found in the EU Water Framework Directive.

Studies show that PFAS can disrupt soil enzyme activity as well as change
microbial availability and damage their cellular structure (Cai et al., 2019, 2020).
There is an indication of a direct correlation between PFAS concentrations in soil
and bioaccumulation in plants (EurEau 2022). At the same time, some have
concluded that soils fertilized with municipal sewage sludge are unlikely to be a significant source of PFAS exposure in humans (Blaine et al., 2014).

Svenskt Vatten (2014) investigated PFOS and PFOA in soils and sugar beets fertilized with a combination of sewage sludge and mineral fertilizers and found that none were found in the concentration above the detection limits of the analytical methods. However, these detection limits were as high as 0.010 mg/kg TS. This study is currently being repeated with improved analytical methods and lower detection limits. The study illustrates the challenges in assessing when the soil is contaminated, with analytical methods that resulted in the absence of a PFAS baseline or background levels in soils due to excessively high detection limits.

2 Aim and purpose

Despite many interesting projects and PFAS-related studies that have been conducted, there is a substantial lack of compiled knowledge on PFAS. This shortfall of knowledge also extends to WWTPs in terms of as a transport route and its possibilities to remove PFAS, to prevent PFAS from being released into the surrounding environment.

The purpose of this project is to compile knowledge and produce guidance for the PFAS problem with focus on municipal WWTPs and the broader area of their operations. This compilation and guide included analysis and mapping, understanding the problems, the environmental impacts, the role of wastewater treatment plants as a transport route, sludge treatment, as well as actions including various treatment techniques and upstream work.

Specific goals of the project were:

▪ To compile existing measurement data from Swedish WWTPs and summarize knowledge from those projects that are ongoing and recently completed.
▪ To compile and discuss different PFAS treatment techniques with focus on WWTPs but also techniques related to effective upstream work or sludge management.

Based on the above compilation of existing data and knowledge base:

▪ To develop guidelines for WWTPs for handling the PFAS problem.
▪ To identify gaps in knowledge and to address challenges that need to be overcome.
The project intends to equip different stakeholders with more knowledge to understand the importance of making a collective assessment of the current pathways of how PFAS are spread locally and regionally, to be able to prioritize the right measures and actions. The project has focused on municipal wastewater treatment, but knowledge of other relevant routes of PFAS spreading have been included to the extent that these can contribute to an understanding of how the wastewater sector can tackle the PFAS problem. Target groups include both experts with good knowledge in treatments of micro pollutions and people without expert knowledge. Available data and practical knowledge used in the project are from Swedish WWTPs from the last ten years.

3 PFAS in society and the environment

3.1 Sources and transport routes

To understand possible PFAS sources and transport pathways in Sweden, an example of the mass balance for PFAS and PFOS has been developed in the project (see Figure 3.1). It depicts possible PFOS flows in Sweden based on various studies that describe potential sources. As no PFAS is manufactured in Sweden (Svenskt Vatten 2022), all PFAS comes via imports, most commonly via consumer- and industrial products. In addition, it is estimated that there is a large atmospheric deposition of PFAS. PFAS are mobile pollutants and are spread in water, soil, and air. PFAS can therefore disperse into the environment, away from where they are produced and used. They can also spread via indirect sources that receive PFAS-contaminated water or waste (e.g., WWTPs and landfills). PFAS can also be spread via runoff from contaminated sites such as fire drills and airports. Due to its environmental persistence, PFAS can accumulate in water bodies, biota, animals, and soil.

In Figure 3.1, the estimated outflows of PFOS via surface water is greater than sum of inflows via products and atmospheric deposition. This is explained by PFOS leaching from “internal” sources such as landfills, contaminated areas, and land. It is almost impossible to quantify the various PFAS flows in the society and the environment, and the figure should thus only be used as a possible scenario that does not reflect all sources and transport routes. As shown below, the flows of PFOS from outgoing water and sludge via SSTPs are approximately 22 kg/year and thus relatively small. PFAS leakages from contaminated land areas must be handled more urgently, but it is important to note that the outflow from
wastewater and sludge together with leachate constitutes the few of many PFAS flows in society where a continuous monitoring is possible.

A reduction in both the import of PFAS (via products, use in industries and operations) and the atmospheric deposition would lead to a reduction in export via surface water. However, it is uncertain how large the reductions would need to be, even if all inflows were to cease. This uncertainty is in large part associated with the amount of PFAS that has already accumulated in contaminated soil and water and the continued emission dynamics and duration of these accumulated PFAS. The following chapter sections describe various known sources and pathways of PFAS.

### 3.1.1 Products

The total annual usage of PFOS in Sweden was estimated to be 190 kg by KEMI (2015). Currently, there are many different PFAS in a wide range of applications and products, and there is no comprehensive mapping of the substances and their functions in different applications. Glüge et al. (2020) and Svenskt Vatten (2022) have made a broad overview of PFAS types which have been used in various products as well as those imported by Swedish companies. In addition to common
consumer goods such as paper, textiles, and fire-fighting foam, PFAS are also included in many products and activities which are not usually considered as contributing factors to the PFAS problem (e.g., ammunition, climbing ropes, guitar strings, artificial turf etc). Hanson et al. (2020) and KEMI (2021b) showed that PFAS are also used in products in cosmetics, skin care, hair care and toiletries. There is currently no information on the actual amount of various PFAS that are imported and circulated, as they are difficult to estimate. Long-chain PFAS such as PFOS and PFOA have been phased out within the European Union and United States since 2000, after studies demonstrating their toxicity; however, they are still produced and used in other parts of the world and are therefore found in various products.

3.1.2 Atmospheric deposition

Atmospheric transport of PFAS can take place both over short and long distances as shown by the presence of PFAS in remote areas such as the Arctic and Antarctic. There are four main sources of PFAS to the atmosphere:

1. Direct emissions from manufacturing (Dreyer et al., 2009; Prevedouros et al., 2006), which can be considered negligible in Sweden,
2. Direct emissions of volatile PFAS during product lifetime,
3. Formation of PFAS in the atmosphere via degradation from volatile precursors such as fluorotelomer alcohols (FTOH) (Armitage et al., 2006, 2009), and
4. Water-to-air transfer via aerosols from the sea (Sea Spray Aerosol -SSA; Johansson et al., 2019; Reth et al., 2011; Sha et al., 2022) or other sources such as WWTPs and landfills (Ahrens et al., 2011).

Although comprehensive measurements lack in Sweden, analyses of precipitation and estimations of PFAS added through the precipitation showed that atmospheric deposition contributes to in the range of ca 650 - 1,000 kg/year of the sum for nine different kinds of PFAS, and 23 - 310 kg/year for PFOS (Filipovic, 2015; Dreyer et al., 2009; Hansson et al., 2016). Filipovic et al. (2013) estimated the PFOS content in precipitation to be 1.07 ± 0.195 ng/L. Sweden, with a land area of approximately 530,000 km² and approximately 650 mm precipitation in the year 2021, is estimated to have a PFOS deposition of 370 kg per year. The atmospheric deposition of PFOS is thus greater than or equal to the estimated import of PFOS to Sweden via products (Figure 3.1).

Based on a study regarding PFAS background levels in Swedish soil, the total accumulated amount of PFOS and ΣPFAS28 in the top 10 cm soil layer was
estimated to be $2.7 \pm 2.4$ tons and $16 \pm 14$ tons within Sweden (Sörengård et al. 2022), of which atmospheric deposition may be the main cause. This indicated that soil is a pitfall for PFAS and a potential source of releases to the surrounding soil and water environment. The atmospheric deposition is a significant annual net addition to the Swedish PFAS cycle, given that Sweden does not manufacture PFAS or is allowed to emit relevant PFAS into the atmosphere (see 3.1.3 below). Contributions to PFAS atmospheric deposition from sea aerosols are difficult to assess.

### 3.1.3 Wastewater treatment

Wastewater treatment plants are not producers, nor do they use PFAS chemicals in their operations. As with other pollutants, they are recipients of these chemicals and a transport route and link between society and environment. But because wastewater treatment plants are a collection point and pathway for emissions from society (e.g., households, industries, and leachate from landfills), they constitute one of the well-quantified transport routes of PFAS into the environment, the other being atmospheric deposition. It should be noted that, according to current regulations (Naturvårdsverket, The Swedish Environmental Protection Agency’s regulations on the treatment and control of wastewater from densely built-up areas, NFS 2016:6), industries are generally required to treat their wastewater before it can be released into municipal WWTP and to prevent the release of leachate into Revaq-certified wastewater treatment plants (Revaq: sludge certification system in Sweden for sludge application as fertilizer). Infiltration of storm- and groundwater can also result in the contamination of wastewater with PFAS if these supplemental waters come from, for example PFAS-contaminated land areas. PFAS release that contaminate ground and surface water can also contaminate drinking water, which again ends up in WWTPs. Filipovic and Berger (2015) showed that drinking water can be one of the largest avenues to spread PFAS into wastewater. In Bromma WWTP (a Swedish WWTP in Stockholm), the spread via drinking water contributed over 40% of certain PFAS. As such, PFAS knowledge in drinking water is important for the purpose of understanding PFAS load in wastewater treatment plants (see section 3.1.10).

Conventional treatment techniques used in WWTPs (mechanical, biological, and chemical treatments) are generally ineffective in removing PFAS (Banzhaf et al., 2017). There are studies which show that PFOS can be "formed" in wastewater treatment plants through the breakdown of other PFAS, i.e., from the PFOS precursors, during the biological treatment step (Arvanii and Stasinakis, 2015; Becker et al., 2008; Eriksson et al., 2017; Schultz et al., 2006).
Hanson et al. (2016) estimated the total annual emissions of 15 PFAS via effluent from 336 Swedish WWTPs with a load of over 2,000 population equivalents (p.e.) to be approximately 70kg. PFOS and PFOA were identified as the dominant PFAS in effluent water, of which PFOS was estimated to be about 20 kg/year. For sludge, the total emissions from the same 336 Swedish wastewater treatment plants were estimated to be 8 kg per year, of which 2 kg consisted of PFOS. However, there are knowledge gaps regarding PFAS in sludge, partly because sampling and analysis in sewage sludge are less reliable than in water. Anaerobic and aerobic treatment of sewage sludge have shown to increase the concentration of PFAA compared to untreated sludge (Yu et al., 2009; Guerra et al., 2014). Possible causes include degradation of precursors, the reduction of volatile solids during digestion (anaerobic treatment), and increased sorption capacity of the digested sludge (Guerra et al., 2014).

The contribution of WWTPs to PFAS emissions via aerosols in the processing air has been investigated in some studies (e.g., Ahrens et al., 2011; Vierke et al., 2011). They showed seven times higher in levels of PFOS (113 ±39 pg/m³) in the air in Ontario WWTP in Canada, compared to the reference points. The highest concentrations were measured at the aerated process. Elevated levels could also be found downstream in the wind direction from the plant. Shoeib et al. (2016) showed that PFOS was the dominant PFAS (47 – 81 %) of all PFAS investigated in air around the wastewater treatment plant.

PFAS are found in all types of wastewaters. In addition to WWTPs, private sewage systems also contribute to the total PFAS load to the environment. As there is a lack of data from individual drains, it is difficult to assess the magnitude of their load to the environment.

### 3.1.4 Fire drill sites and fire extinguishing

The release of film-forming fire-extinguishing foam (Aqueous Film Forming Foam; AFFF) from fire drills is one of the most noticed pollution problems with PFAS, both in Sweden and internationally (Cousins et al., 2016). In Sweden, it is primarily the fire and rescue services that use fire drills sites, which can lead to the leakage of extinguishing foam into the environment and result in elevated levels of PFAS. In Sweden, the first report on PFAS in the environment came in 2001 whereby the Rosersberg fire drill site was identified as an important local source point (Järnberg, 2001).
According to information presented by Hansson et al. (2016), the Swedish Public Safety and Emergency Management Authority (Myndigheten för samhällsskydd och beredskap, MSB) have conducted and continues to conduct the most extensive fire training drills at Revinge, Sandö and Skövde and Rosersberg. These repeated training exercises should constitute as sources of PFAS to the environment.

It is difficult to map which training sites belong to industries and civil or municipal fire training sites, and their potential use and leakage of PFAS-containing fire foam. But given the large number, these sites can be concluded to be a significant source (Hansson et al., 2016). The same mapping also shows that fire stations that may have been used or are being used for fire drills could constitute as a PFAS source. This suggests that fire stations could be point sources that are scattered throughout the country. For example, the annual ΣPFAS11 emissions from Arlanda Airport in Stockholm released to Lake Mälaren, where PFAS mainly originates from fire foam, has been estimated to be 5.3 kg with the main share from PFOS being 2.6 kg/year (Ahrens et al., 2015).

The actual use and leakage of PFAS-containing fire foam during extinguishing work is not possible to quantify. Hanson et al. (2016) calculated with some speculations, that a total of 660 kg of PFAS may have been released in connection with fire extinguishing work between the years 1998 and 2014. PFAS containing fire foam are used today only for the purpose of extinguishing or preventing fires in flammable liquids, which can occur at traffic accidents to avoid ignition where people are rescued from vehicles. According to the Great Stockholm Fire and Rescue Service (SSBF), an on-site assessment is always made when other extinguishing methods are insufficient to stop an ongoing fire or to establish appropriate protection (Erika Wester, personal communication, 2022).

SSBF still uses PFAS-containing firefighting foam, but the need has decreased as various alternatives have been developed and awareness of the foam’s environmental impact has increased, and work is ongoing to find alternatives that do not contain PFAS. Existing stock of PFAS-containing fire foam must be destroyed, and foam tanks cleaned. SSBF’s ambition is to phase out PFAS containing foam liquid in the coming year. The Great Gothenburg Fire and Rescue Service states that only PFAS-free fire foam is used since 2021, including fire foam for fuel fires (Helena Grimm, personal communication, 2022). No direct disadvantages of PFAS-free fire foam have been reported. However, costs for purchasing PFAS-free fire foam are higher, and the ammonium-based alternatives
can damage fire-fighting equipment such as nozzles, as these are made of light metal.

KEMI has recently published a knowledge compilation of the studies and literature available on foam liquids, with special focus on PFAS based foam liquids (KEMI 2022b). As it is also shown there, for example, water mist can be used as a good substitute especially if the mist can be finely divided more effectively with the additional use of ammonium chloride.

### 3.1.5 Airports

There is also a risk of PFAS leakage into the environment at airports, which is primarily due to the historical use of PFAS containing fire foam during fire drills carried out at airports. However, use of PFAS in hydraulic fluids in the aviation industry is also a potential source. It is not known how many of all Swedish airports emit PFAS to the environment as measurements have only been carried out at a few airports where they are known to be used as fire drill site. The pollution around Landvetter (Gothenburg, Sweden) and Arlanda (Stockholm, Sweden) airport were investigated in detail already almost two decades ago (Woldegiorgis et al., 2006; Woldegiorgis and Viktor, 2008). After the measurements at Arlanda, a fishing ban was introduced in the nearby lake Halmsjön to reduce the risk of human exposure via food (Norström, 2013). Elevated levels of PFAS have also been measured in connection with other commercial service airports such as Bromma, Visby, Kiruna, Umeå, Sundsvall and Malmö (Swedavia, 2014). It should be noted that Swedavia stopped purchasing PFAS containing fire extinguishing agents already in 2002 (Hansson et al., 2016). Swedavia is also conducting extensive work to investigate PFAS sources and pathways in soil and water to be able to clean up PFAS from all their airports in Sweden (Henriksson, J. personal communication Aug 31, 2023).

For some of the airports, it was possible to estimate the load to the environment (Table 3.1). Assessing the quantities used at these airports, the total PFOS emissions were estimated to be ca. 1,000 – 3,500 kg, in comparison to the environmental load of 25 – 120 kg per year from all 27 known fire drill sites at airports (Hansson et al., 2016).
Table 3.1 Selection of relevant investigations related to PFAS at airports.

<table>
<thead>
<tr>
<th>References</th>
<th>Descriptions</th>
</tr>
</thead>
<tbody>
<tr>
<td>Norström et al., 2013</td>
<td>The spread of PFAS from fire drills at Landvetter and Arlanda Airport including assessment of effects and risks with the spread of PFAS. Implemented within the RE-PATH project.</td>
</tr>
<tr>
<td>Rosenqvist et al., 2017</td>
<td>The spread of PFAS in fields, surface, and groundwater around fire drill sites at Arlanda Airport. Implemented within RE-PATH II the project.</td>
</tr>
</tbody>
</table>

3.1.6 Operations of the Swedish Armed Forces

PFAS containing fire extinguishing foam has also been used by the Swedish Armed Forces (Försvarsmakten), mainly at several military air bases. The Swedish Armed Forces is accused of having contaminated drinking water with PFAS in Botkyrka, Båstad, Halmstad, Uppsala, Ängelholm, Östersund and Ronneby (Svenskt Vatten, 2022). A complete overview or quantification of used amounts of PFAS containing fire-fighting foam does not exist, however the estimate of used PFAS by the Swedish Armed Forces was summed to be 540 kg of between the years 1970 – 1990 in connection with fire drills at F18 Tullinge/Södertörn (Försvarsmakten, 2015).

3.1.7 Waste incineration

Waste incineration plants can be seen as removing PFAS from the cycle as the high temperatures can break down PFAS. It is required for incineration to reach at least 850 degrees for non-hazardous waste, and at least 1100 degrees for hazardous waste, (Industrial Emissions Directive 2010/75/EU). Since Swedish waste incineration plants are equipped with advanced flue gas treatment, it is generally assumed that only low emissions of PFAS occur from them.

Investigations by Sandblom et al. (2014) have shown low PFAS levels in flue gas condensate as well as in ash and slag (Table 3.2). The amount of PFAS via ash and slag deposited has been estimated to be around 7.5 kg/year for ΣPFAS9 and 0.83 kg/year for PFOS. In an investigation that included 27 of 38 incineration plants existing in Sweden, detectable levels of PFAS27 were found in the bottom ash, fly ash and in the condensate water at several of the plants (Strandberg et al., 2021).

The study also found that high combustion temperatures or a high proportion of a certain type of waste does not guarantee PFAS destruction. In contrast, Lundin and Jansson (2017) concluded that the investigated PFAS including PFOS were already destroyed in ordinary municipal waste incineration plants with a temperature of 850 degrees and that the risk of conversion to other PFAS was low. Degradation of PFAS such as PFOS has been shown to occur already at temperatures of 350 °C with the addition of calcium hydroxide (Wang et al., 2015a). These studies thus
show that it is not entirely clear how reliable the combustion temperature is for the destruction of PFAS and that further studies are required to determine the right conditions for a thermal destruction of PFAS.

The ash from incineration is usually deposited in landfill, and leakage of PFAS gathers in leachate. PFAS found in flue gas condensate can be released in the environment, as usually only metals and nitrogen are removed from condensate. This condensate is released to the recipient or to a wastewater treatment plant. In some cases, Reverse Osmosis (RO) are used for the treatment of the condensate, and then the fraction containing PFAS is sent back into the incineration boiler.

Waste is often stored in incineration plants to cover the heat demand during certain periods. In a project at Umeå University, investigations were done to see to what extent the stored waste could release PFAS (Björklund et al., 2021). Björklund et al. (2021) collected stormwater runoff from a waste storage for six months in 2019 and analyzed for approximately 30 commonly occurring PFAS. The sample showed same types and concentrations as those found from landfill leachate.

<table>
<thead>
<tr>
<th>References</th>
<th>Descriptions</th>
</tr>
</thead>
<tbody>
<tr>
<td>Strandberg et al., 2021 (Link)</td>
<td>Mapping of PFAS releases to the environment via bottom ash, fly ash and condensate water at Swedish final disposal facilities. 27 of the totals of 38 incineration plants in Sweden were included in the project.</td>
</tr>
<tr>
<td>Lundin and Jansson (2017) (Link)</td>
<td>Mapping the degradation efficiency of some persistent organic compounds incl. PFAS in three different types of incineration plants; municipal 850-degree waste incinerators, hazardous waste incinerators (HWI) and incinerators used in the manufacture of cement.</td>
</tr>
<tr>
<td>Sandblom et al., 2014 (Link)</td>
<td>Research on waste incineration plants in Sweden can be a source of PFAA to the atmosphere and to the environment in general. Four different combustion plants were included in the sampling of slag, fly ash, condensate, and wastewater from the flue gas treatment. The main conclusion of this study was that Swedish waste incineration is not a significant source of PFAA to the atmosphere.</td>
</tr>
<tr>
<td>Avfall Sverige 2020 (Link)</td>
<td>Development and validation of analytical methods for PFAS11 in slag and flue gas condensate as well as application to several samples from two incineration plants in Sweden.</td>
</tr>
<tr>
<td>Björklund et al., 2021 (Link)</td>
<td>Investigation of PFAS leakage from waste storage at the Umeå incineration plant.</td>
</tr>
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</table>

3.1.8 Landfills and waste facilities

Although household waste is no longer allowed to be deposited in Sweden, various industrial waste is deposited as well as ashes and slag from waste incineration. In addition to the waste which may contain PFAS, sludge from WWTPs containing PFAS is also used as a sealing and covering layer in landfills.
There are several thousand landfills in Sweden, some of which are closed and some which are still active. Information on the types of waste deposited is usually not available especially for the older and smaller landfills. Older and smaller landfills also often lack sealing layers or collection and treatment of leachate, which can mean that these leak PFAS into the environment. Sewage sludge can also be a possible spreading route when the sludge is used for landfill cover, as PFAS levels in sludge like those found in many other waste streams.

A study examined PFAS in leachate, sludge, groundwater, and wastewater at the Hovgården landfill in Uppsala (Bonnet, 2017). The ΣPFAS28 concentration in the leachate varied between 59 and 1500 ng/L and in the groundwater between 8.5 and 1,800 ng/L. Most groundwater was above the threshold for groundwater of 45 ng PFOS/L.

Ahrens et al. (2011) and Hamid et al. (2018) reported elevated concentrations of PFOS and PFAS in ambient air at landfills, suggesting that landfills may act as emission sources for atmospheric PFOS and other PFAS. Apart from the landfills, facilities for receiving and handling hazardous waste could also pose a risk for PFAS leakage.

### Table 3.3 Selection of relevant investigations related to PFAS at waste facilities.

<table>
<thead>
<tr>
<th>References</th>
<th>Descriptions</th>
</tr>
</thead>
<tbody>
<tr>
<td>Avfall Sverige 2018</td>
<td>Compilation of the state of knowledge about PFAS at waste facilities in Sweden. For various PFAS in water from waste facilities, maximum levels around 10,000 ng/L and average levels around 1,000 ng/L were observed. The median level of PFOS was 120 ng/L.</td>
</tr>
<tr>
<td>Bonnet, 2017</td>
<td>Mapping of PFAS at the Hovgården landfill in Uppsala including analyses in leachate, sludge, groundwater and wastewater.</td>
</tr>
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</table>

### 3.1.9 Water bodies

Leakage of PFAS usually stems from contaminated land into water sources, for example via WWTPs or via drainages. There is a reversible correlation between atmospheric deposition of PFAS (see section 3.1.2) and water bodies, (i.e., atmospheric deposition contributes to PFAS in water bodies while PFAS found in surface water can also be added to the atmosphere via aerosols). A yearlong field study in Lake Krycklan in Umeå showed that atmospheric precipitation was the most important transport route of PFOA and PFOS to the lake (Filipovic, 2015). The levels were higher in the rain and snow than in the sampled water mass. This indicated that a large part of PFOA and PFOS from rain or snow was transported further out to other groundwater and surface water bodies, thus increasing levels of PFAS. The average level of PFOS in Swedish rivers estimated by Nguyen et al.
(2017) was 3.5 ng/L (with variations between 0.04 – 6.9 ng/L). Transport of PFOS to the sea via Swedish waterways can thus be estimated to be around 600 kg/year, with the assumption that the total runoff amounts to about 5,600 m³/s for the whole of Sweden and that only a negligible part of 0.4 per-mille of the groundwater flows directly out to sea (the Swedish Meteorological and Hydrological Institute, SMHI 1996). Estuaries could act as sinks for PFOS due to the change in salinity and the consequent change in solubility (Pan and You, 2010; Wang, 2015b).

3.1.10 Drinking water

A survey of PFAS in raw water and drinking water carried out by the Swedish Food Agency in 2020 included 580 water operators in 257 municipalities (Livsmedelsverket, 2021). Most water operators had concentrations below 10 ng/L in the raw water. In two waterworks, 40 ng/L was measured on single sampling occasions, while lower levels were observed on other occasions. All these levels are below the current Swedish action limit of 90 ng/L for ΣPFAS11 and the limit value in the new drinking water directive of 100 ng/L for ΣPFAS20.

Investigations into Swedish groundwater and surface water with a focus on drinking water sources showed that the PFCA was the most found PFAS in groundwater and surface water (Table 3.4). The high proportions of C4–C8 PFCA indicated high mobility of the shorter PFCA (Gobelius et al., 2018). A total of 5 of 169 samples (3 %) exceeded the threshold value of 90 ng/L for ΣPFAS11.

<table>
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<tr>
<th>References</th>
<th>Descriptions</th>
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</thead>
<tbody>
<tr>
<td>Livsmedelsverket 2021 (Link)</td>
<td>Mapping of PFAS in Sweden's municipal raw and drinking water based on a survey sent out in 2020. 580 waterworks were included with individual measurements several years back. Detectable levels were found in the drinking water of 74 of the 154 waterworks where analysis was carried out. Drinking water distributed to approximately 2.2 million people had PFAS levels above 10 ng/L.</td>
</tr>
<tr>
<td>Naturskyddsföreningen 2021 (Link)</td>
<td>Compilation of PFAS levels in drinking water and comparison with EFSA’s new limit value, using the Swedish Food Agency’s methodology for the currently valid Swedish action limits.</td>
</tr>
<tr>
<td>Gobelius et al., 2018 (Link)</td>
<td>Mapping of 26 PFAS in Swedish ground and surface water with almost 500 samples. The highest ΣPFAS26 concentrations were detected in surface water (13,000 ng/L) and groundwater (6,400 ng/L).</td>
</tr>
</tbody>
</table>

3.1.11 Soil

Atmospheric deposition (see section 3.1.2), contaminated soil at fire drill sites (see 3.1.4 and 3.1.5) and the spread of contaminated water or sludge contribute to the PFAS load on terrestrial ecosystems. Different soil systems such as agricultural,
forest and other soil types also form a reservoir of PFAS that can leak into aquatic environments within a foreseeable time.

Sörengård et al. (2022) for instance estimated that approximately 2.7 tons of PFOS and 17 tons of ΣPFAS are stored in Swedish soil (excluding contaminated areas such as at fire drills). It should be noted that these are estimated amounts, with relatively high variation. The study found a strong north-south and west-east gradient with the highest amounts in the south and east.

Table 3.5 Selection of relevant investigations related to PFAS in soil.

<table>
<thead>
<tr>
<th>References</th>
<th>Descriptions</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sörengård et al. 2022</td>
<td>Mapping of PFAS levels in Sweden's forests. PFOS showed the highest detection rate with a distinct increase from north to south and west to east.</td>
</tr>
</tbody>
</table>

3.2 Legislation and guideline values/ limit values

Currently, there is no comprehensive legislation that covers all PFAS, but only specific PFAS and the substances that can be transformed into these PFAS are regulated. KEMI has compiled a review of laws and regulations concerning PFAS on its website (KEMI 2022a). Below is a brief overview of existing legislation with the limit values/guidelines relevant for this report.

PFOS, its precursors, compounds that can be broken down to PFOS, as well as PFOA and PFOA-related compounds, are included in the Stockholm Convention. The convention is incorporated into EU legislation via the Persistent Organic Pollutants, POPs Regulation (Regulation (EU) 2019/1021) which deals with persistent organic pollutants. PFHxS and PFHxS-related compounds are also included in the Stockholm Convention. In addition to the restrictions on PFAS in the POPs regulation, there are several PFAS that are regulated under REACH - Registration, Evaluation, Authorisation and Restriction of Chemicals, (the REACH regulation or the Chemicals Registration Regulation, EG no. 1907/2006). Ongoing work continues within the EU for further bans and restrictions, and the development of legislations are complex to follow. A summary table with limit values and guideline values in various matrices that are relevant to the water sector is shown in Table 3.6.

The guideline values in the form of Environmental Quality Standards, EQS (Miljökvaltetsnormer, MKN) are used as a baseline where levels can be temporarily exceeded, as long as the total load does not become too high. EQS that are used to form the basis for discharge conditions in a permit review are applied for Special Polluting Substances (SPS) that are included to assess the ecological
status. An EQS describes the quality level that a water body should reach at a certain time. The main rule is that all water bodies must achieve good ecological status in water management.

<table>
<thead>
<tr>
<th>Matrix</th>
<th>Limit value/reference value</th>
<th>Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>Drinking water</td>
<td>Minimum requirements limit values EU drinking water directive 2020/2184 ΣPFAS20: 100 ng/L; ΣPFAS total: 500 ng/L Boundary, guideline, and action limits in Sweden: Recommended action limit from 2014 ΣPFAS11 90 ng/L (although not legally binding) New regulations for drinking water LIVSFS 2022:12 decided 1st January 2023 and with compliance from 1st January 2026. Limit values: ΣPFAS21: 100 ng/L; ΣPFAS4: 4 ng/L</td>
<td>Limit values were set at EU level on 12 January 2021 and member states have 2 years to implement the same or stricter national limit values. The new regulation by the Swedish Food Agency’s LIVSFS2022:12 has enforced limits based on the requirement in directive 2020/2184 (ΣPFAS20plus 6:2 FTSA) and an additional limit value for ΣPFAS4 (PFOS, PFOA, PFHxS and PFNA) based on the European Food Authority’s (EFSA) guidelines based on tolerable weekly intake (TWI) of PFAS.</td>
</tr>
<tr>
<td>Surface water (inland/ coastal waters, transition zone)</td>
<td>Chemical Status: PFOS annual average (AA-MKN*): - 0.65 ng/L (inland surface water) - 0.13 ng/L (other surface waters), and - 9.1 ng/L MKN biota. PFOS maximum value (MAC-MKN): - 36,000 ng/L (inland surface water) - 7,200 ng/L (other surface waters) Ecological status, national assessment basis in Sweden: ΣPFAS11 maximum permitted concentration: 90 ng/L (inland and coastal water, transition zone) New proposed EQS for EU water framework directive for PFAS24 is 4.4 ng/L expressed as sum of PFOA equivalents.</td>
<td>PFOS is a priority substance with a limit value (MKN, HVMFS 2019:25) which is used to assess the chemical surface water status. The new EU Commission proposal for modification of three water management Directives: The Water Framework Directive (WFD, 2000/60), the Groundwater Directive (GWD, 2006/1198/EC), and the Directive on Environmental Quality Standards (EQSD, 2008/105/EC). The proposal includes setting groupwise EQS values for PFAS, and the relative potency factors-approach for assessing PFAS, the sum EQS expressed as PFOA equivalents. The value for ΣPFAS11 refers to water bodies with extraction for drinking water production that have been identified in accordance with Ch. 3. Section 2 of the Water Management Ordinance (2004:660). The value must not be exceeded in the water body at the point that is representative of raw water intake. (HVMFS 2019:25).</td>
</tr>
<tr>
<td>Ground water</td>
<td>New proposed EQS for EU water framework directive for PFAS24 is 4.4 ng/L expressed as sum of PFOA-equivalents (SGU-FS20231:1)</td>
<td>The water delegations decided in October 2018 on a basis for assessment (guideline value) regarding a maximum permitted concentration of 90 ng/L ΣPFAS11 in groundwater bodies and that measures to reverse an upward trend when the level exceeds 18 ng/L (20 % of the guideline value). SGI (2015) is based on the Swedish Food Agency’s action limit for drinking water. The preliminary assessment for PFOS was extended to include ΣPFAS7, but not ΣPFAS11. The new threshold value 4.4 ng PFOA equivalent/L replaced the previous national EQS.</td>
</tr>
<tr>
<td>Sewage sludge</td>
<td>No limit values in Sweden, but in Denmark apply: ΣPFAS4: 15 µg /kg BW (body weight) ΣPFAS22: 50 – 100 µg /kg BW Proposed limit values for PFOS for the spread of sludge to agricultural land: 0.07 mg/kg DS in 2015 0.02 mg/kg DS until 2030</td>
<td>The revision of the Sewage Sludge Directive is expected in 2024. Proposed regulations according to previous government assignments regarding the sustainable return of phosphorus (Naturvårdsverket 2013), see also Sewage sludge 3.2.4.</td>
</tr>
</tbody>
</table>

* AA- annual average; MKN - environmental quality standard

### 3.2.1 Drinking water and food

In May 2016, the US EPA (EPA 2016a, b) set a concentration of 70 ng/L for PFOA and PFOS (individually or combined) as the maximum limit value for exposure from drinking water. The Drinking Water Directive in the EU (2020) proposes a
"group limit value" for "Total PFAS load" of 500 ng/L or the limit value for the "sum of 20 PFAS" of 100 ng/L in drinking water. In Sweden, a national limit value for PFAS in drinking water came into force in January 2023, with PFAS 4.4 ng/L and PFAS21 to 100 mg/L. PFAS21 includes earlier PFAS20 with the addition of 6:2 FTAS, and substances in PFAS4 categories are included in the PFAS21. The new limit values are to be applied from January 1st, 2026 (LIVSFS 2022:12).

The drinking water action limit were calculated based on safe consumption levels of substances recommended by Swedish Food Agency (2014). The action limit of 90 ng/L was based on a tolerable daily intake (TDI) of 150 ng/kg of body weight for PFOS. The TDI describes the long-term maximum intake of substances in food or drinking water that can be consumed daily without risking adverse health effects. This value was developed by the European Food Safety Authority (EFSA) in 2008. Since then, this value has been re-assessed several times. The latest assessment done in 2020 is based on tolerable weekly intake (TWI) instead of TDI, and for sum of PFAS4 instead of PFOS only, which means at least reduction with factor of 240 times. The TWI with the latest EFSA assessment is 4.4 ng/kg body weight per week is presented for PFAS4.

For Swedish regulation, the Swedish Food Agency has used this TWI to calculate 4 ng/L for ΣPFAS4 to be safe (Dnr 2022/01733). This level should be safe for 95 % of all men and 75 % of women and children 12 years of age or older. For younger children, the contribution of ΣPFAS4 from the diet is so great that a safe limit value for drinking water could not be calculated. The content is proposed as an extra limit value in addition to the limit value for PFAS21 which is introduced following the adoption of the drinking water directive.

It is important to point out that both PFAS11 and PFAS21 contain PFAS with both short chain fluorinated carbon (i.e., PFBA with 3 fluorinated carbon atoms) and long fluorinated carbon chains (i.e., PFDA with 9 fluorinated carbon atoms). In general, the toxicity of PFAS with longer carbon chains is much higher compared to shorter PFAS. The bio accumulative properties are also different. When the values are summed up, a general misconception of all PFAS to be as toxic as PFOS can give a false picture. The risk of exposure to PFAS contaminated water can be simply overestimated and hence stricter reductions are imposed than necessary.

In the new proposed amendment of EU Water Framework Directive, this is accounted for and all of the PFAS are recalculated using relative potency factors (RPF) to PFOA equivalent concentration before they are summed up. A similar
approach is however not applied for drinking water. There are ongoing developments for documentation for the assessment of individual and grouped PFAS. However, it is important to point out that these methodologies are based on toxicity data that are known today. There are risks that the substances have an unknown effect on certain species or organs. The current view in some PFAS toxicities might indicate low risk which might ease the limitation actions of their emissions. However, many PFAS are very persistent and can eventually accumulate to the levels that can produce toxic effect that is yet unknown.

Denmark lowered the limit value for PFAS substances (e.g., PFAS4, PFOA, PFOS, PFNA and PFHxS) in water from 100 ng/L to 2 ng/L, a sharp reduction after the Korsör case where meat was contaminated with health-threatening and endocrine-disrupting PFAS chemicals. The cows had been drinking contaminated water from a fire drill over a long time. This low limit value was set so that the total intake of PFAS via both drinking water and consumption of meat will not be harmful to health.

3.2.2 Products

In June 2019, the European Council of Ministers (EC, 2019) highlighted the widespread occurrence of PFAS in the environment, products, and people. They called for an action plan to eliminate all use of PFAS that is not necessary in society. In August 2021, EU member states took the decision to ban 200 long-chain PFAS substances (C9 – C145) within the EU from February 2023. The decision came after the initiative from KEMI in Sweden together with the German Environmental Agency (UBA). Among the substances to be banned are PFNA and PFDA, which are included in ΣPFAS11 and ΣPFAS20; PFUnDA, PFDoDA and PFTrDA which are included in ΣPFAS20; precursors to these PFAS, and PFTeDA and its precursors. These PFAS have long carbon chains and relatively low water solubility. These substances are highly bio accumulative which justifies the ban, but they are also relatively easy to remove from water. PFOS, PFOA, PFHxS and substances that can be broken down to these are regulated within the EU. For example, the ban of use of PFHxS came into place in June 2022 (The Ministry of the Environment 2022). The substance is also in the ΣPFAS4 group regulated by the Swedish Food Agency, with low limit value. However, PFAS with short carbon chains are still allowed to be used. They are more difficult to separate from water even with the advanced treatment methods compared to the long chains. Regulation is being discussed for PFHxA and other substances that can degrade to PFHxA. In January 2023 new limit values for PFAS in food products were introduced in the EU (EU Commission regulation 2022/2388, later substituted with
Commission regulation 2023/915). The limit values were introduced for PFOS, PFOA, PFNA and PFHxS both separately and as a sum of the four substances.

In addition to the PFAS that are regulated at EU level, KEMI, the Swedish Chemical Agency, would like to see the EU to ban more PFAS in the future. Draft proposal includes prohibition of the use of PFHxA submitted on June 19th, 2023, its salts, and related substances in various application. KEMI, together with the chemical authorities from Germany, the Netherlands, Norway, and Denmark, presented a new proposal (ECHA, 2023. Annex XV) that goes beyond existing decisions. The proposal introduces a total ban on the use of all kinds of PFAS in certain products and certain areas of use, several of which are of importance for the PFAS content in wastewater. Proposal on the prohibition of the manufacture and sale of PFAS, applied to any substance containing at least one fully fluorinated methyl group (-CF3) or one fully fluorinated methylene group (-CF2-) without any hydrogen, chlorine, bromine, or iodine atom, and thereby applicable to more than 10,000 PFAS substances.

Moreover, the European Chemicals Agency (ECHA) has presented a proposal for an EU-wide restriction of all PFAS-containing fire-fighting foams (ECHA 2022). The proposed alternative would ban the sale, use and export of all PFAS in fire-fighting foam after sector-specific transition periods. These transition periods would allow time for the industry to replace PFAS-containing foams without compromising fire safety.

The limit values for PFOS were in the range of 0.3 – 35 µg/kg wet weight for different types of food products and 1.3 – 50 µg/kg wet weight for ΣPFAS4. These values are high compared with the limit values in drinking water. For example, the limit value in muscle meat of some fish species in case they are not intended for the production of food for infants and young children is 45 µg/kg wet weight as ΣPFAS4. This means that the maximum allowed intake of ΣPFAS4 from 1 kg of fish is equivalent to consumption of 11,250 liters of drinking water (limit value of 4 ng/L in Sweden), which is equivalent of approximately 10 years drinking water intake for an average person. The limit value in fish is also comparable to the limit value of ΣPFAS4 in wastewater sludge for agricultural application in Denmark, 15 µg/kg DS, which is equivalent to ~55 µg/kg wet weight).

### 3.2.3 Surface water and groundwater

There are environmental quality standards for PFOS for inland surface water, coastal water, and water in the transition zone. The annual average value must not
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exceed 0.65 ng/L for inland surface water, and 0.13 ng/L for coastal water and water in transition zones. The maximum value must not exceed 36,000 ng/L and 7,200 ng/L, respectively (HVMFS2013:25). The Water Directive environmental quality standard for PFOS is based on the assessment made in 2011 based on EFSA’s previous TDI (tolerable daily intake) and is set for fish (MKN biota) at 9.1 µg/kg wet weight (KEMI 2021a). This means that it is considered safe for humans to eat fish that contain lower levels of PFOS than the limit value. When these are converted to the content in surface water, the habitat of fish, the PFOS limit value then becomes 0.65 ng/L for inland waters and 0.13 ng/L for other surface waters.

In addition to the environmental quality standard for PFOS, there is also an assessment basis for ΣPFAS11 for inland surface water, when used as raw water for production of drinking water. The assessment basis should make it easier to interpret whether measured values are low or high compared to the average for the country or to original levels. The assessment basis is 90 ng/L ΣPFAS11 for the maximum allowed concentration and applies to raw water intake. The assessment basis has been added to the national general guidelines (HVMFS 2013:25) after the Swedish Food Agency determined the action limit for ΣPFAS11. This assessment basis functions therefore as a safeguard to ensure that the release of PFAS to surface water remains low, and the water operators do not have to introduce PFAS remediation of the raw water. It is possible that the new limit values for drinking water of 100 ng/L for ΣPFAS21, and 4 ng/L for ΣPFAS4, in similar way will be considered as basis for assessment in surface water in the national regulations.

A new threshold value of PFAS in groundwater has been applied since March 2023, through SGU’s (Geological Survey of Sweden) regulations on mapping, risk assessment and classification of the status of groundwater (SGU-FS 2023:1). This is based on the EU commission’s provision for revision of the Groundwater Directive (GWD, 2006/1198/EC). The new threshold value was regulated with the PFAS24 sum parameter at 4.4 ng PFOA equivalents/L.

The levels of the various PFAS were recalculated to a PFOA-equivalent before being summarized, unlike all PFAS levels in ng/L which were summed up in the Drinking Water Directive. The recalculation was done by multiplying the respective concentration by a factor that compensated for PFOA’s bioaccumulation potential (RBF, relative bioaccumulation factor). The value is then multiplied by another factor that compensated for the substance’s toxicity in relation to PFOA’s toxicity (RPF, relative potency factor). RBF were not reported in the proposal except for certain substances. The RPF for short-chain PFAS were usually very low.
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(for example for PFBA is 0.05; PFBS 0.001; PFHxA 0.01) and levels of these substances would therefore only marginally affect the environmental quality standard. This is thus very different from the Drinking Water Directive where all PFAS were equated, and their levels were summed without any factors that compensate for their varying harmful effects on the environment. A similar recalculation of PFAS levels to PFOS-equivalents has been proposed by Malovanyy et al. (2021). There, each substance’s half-life in human blood were used to compensate for different bioaccumulation potentials of the substances. The conversion factors for short-chain PFAS according to that methodology were also in the range 0.01 – 0.1 g PFAS/g PFOS.

3.2.4 Sewage sludge

In March 2023, the Danish Environmental Agency set an indicative limit value for ΣPFAS4 for sewage sludge used on arable land to 15 µg/kg DS (dry solids) and 50 – 100 ng/kg DS for of ΣPFAS22. In Sweden, there is currently no limit value for PFAS in sludge that is applied to arable land. A so-called low risk level (LRN, lågrisknivå) in sludge was calculated by Sternbeck et al. (2013) based on the toxicity for terrestrial organisms and impact on surface water. It was 0.12 mg/kg DS for PFOS, thus significantly higher than the Danish limit value which was 0.01mg/kg DS for the sum of PFAS4. This calculated LRN has no legal importance but is still used today for investigation (“tolerable investigation”) at Revaq-certified treatment plants, to determine the impacts from the receiving leachate on the quality of the sludge.

In its report “Sustainable recycling of phosphorus” (2013), the Swedish Environmental Protection Agency produced a proposal for a limit value for PFOS. The limit value was set at 0.07 mg/kg DS in 2015, which was proposed to be reduced to 0.02 mg/kg DS by 2030. The proposal is partly the result of the work of Sternbeck et al. (2013). Inspired by the Ministry of Environment of Denmark (Danish EPA) Environmental Project no.2232, Svenskt Vatten is currently investigating PFAS4 and PFAS22 in Revaq-certified sludge in Sweden in an effort to benchmark and manage PFAS in sludge for agricultural use (A. Finnsson, personal communication, August 25th, 2023).

3.3 Possible treatment requirements

The current treatment requirements for wastewater treatment plants are determined on a case-by-case basis during the review of their permit. The basic principle in environmental assessment is that discharges of treated water should
not degrade the status of the nearest water body downstream from the point of discharge. As discussed in Chapter 3.2.3, there is a limit value for PFOS in inland surface waters and coastal waters, with an annual mean at 0.65 ng/L and 0.13 ng/L, respectively. In addition to the limit values, there is an assessment basis for ΣPFAS11 of 90 ng/L, but this only applies to surface water used as raw water for drinking water production.

The status of a water body is assessed at one or more representative sampling points, while measures and environmental monitoring of point sources are carried out through surveillance and environmental monitoring of activities. In practice, this means that the conditions for the discharge of PFOS (and, where applicable, ΣPFAS11) with effluent should be chosen according to the specific conditions of the water body concerned. The two factors influencing the amount of discharged PFOS/ΣPFAS11 are the degree of dilution in the water body, which is dependent on the volume and turnover time, and the amount of PFOS/ΣPFAS11 coming from upstream sources into the receiving water body. In principle, the smaller the amount of PFAS added from upstream sources, the greater the margin WWTPs have as discharge levels. However, it is rarely the case that discharges from an activity, such as a WWTP, are allowed to use all the capacity within the environmental quality standard. It is also common that PFOS levels upstream of a treatment plant are already higher than the limit value. Emissions from all major sources therefore need to be reduced to achieve a good ecological status of the water body in the long term. The requirement for treatment should be set proportionate, and reduction requirements are placed according to the emission risk posed to the water body by each operator. However, it is considered not likely for treatment plants to introduce measures to meet the assessment baseline for ΣPFAS11, as the level of ΣPFAS11 in most surface water bodies is well below 90 ng/L.

Conditions or requirements for treatment must be chosen in accordance with the Environmental Code, as to which requirements are reasonable and appropriate. The assessment should be based on the overall picture of the load on the water body and the possibilities of the various operators to reduce their current emissions. If a water operator already has low PFAS emissions, the cost is high to reduce the emissions further. In this case, addressing actors with high PFAS emissions is more reasonable. In other cases, it is more interesting to regulate the total amount discharged over a longer period, rather than concentrations. For example, a quantity requirement could be applied as it is with phosphorus at some wastewater treatment plants today. This applies, for example, to substances that
accumulate in sediments, or water bodies that have increasing concentrations of pollutants or long turnovers.

In terms of surface water, there is much uncertainty how the upcoming limit value will be designed, which substances will be covered, and at what level the limit values will be. In the long term, quality standards developed at the EU level will guide surface water limit values. In the shorter term, the new proposed limit values for drinking water may be applied to surface water at the point of raw water intake.

4 Analysis of PFAS

The analysis of PFAS is a major challenge due to the large number of PFAS with different properties, sizes (from small molecules to polymers), different PFAS precursors, and degradation products. It is important to choose analytical methods which are relevant to the study conducted, based on various factors such as what matrix to be measured, what questions to be addressed, what substances are relevant, and what detection limit is applicable for the purpose.

4.1 Analytical methods

Depending on the information required, analytical methods may be necessary for the quantification of individual PFAS as well as for a wide range of PFAS. Due to the large grouping of PFAS, it is impossible to have a targeted and quantitative analytical method for every substance. Various analytical strategies have therefore been developed, ranging from the accurate determination of a few PFAS that have low detection limits using qualitative screening methods, to broad-spectrum analyses that reflect the amount of organically bound fluorine or total fluorine present in the sample.

Figure 4.1 provides a schematic overview of the analytical methods described in this report. The most common is a targeted analysis of several PFAS. For example, 11 PFAS substances that have an action limit in drinking water can be analyzed thus providing the basis for surface water status. If the quantification should include PFAS precursors, extractable organofluorides, or all chemicals containing fluorine, broad-spectrum analyses are needed. Examples of some of these analysis types include total oxidizable precursors (TOP), extractable organic fluorine (EOF) or total organic fluorine (TOF), also called total fluorine (TF) if the sample may contain inorganic fluorine. There are also suspect- or non-target screening methods, which can identify different PFAS without quantifying them.
The number of PFAS that can be analyzed has constantly increased, with universities, commercial laboratories and other operators offering various analysis packages. Historically, PFAS analysis has focused on acid-type PFAS (PFAA such as PFSA and PFCA) and initially measured mostly PFOS and PFOA. For a long time, the 11 PFAS (see section 1.1) recommended by The Swedish Food Agency have been the common standard PFAS to be analyzed. However, this has subsequently been extended to include a few additional substances. New EU Directives will be implemented to increase the number of substances to be measured as PFAS20 (see 3.2.1).

The large-scale use of other types of PFAS, such as ether-based PFAS (Gen-X, F-53B), telomer alcohols (FTOH), sulfonamides (MeFOSA, EtFOSA), etc... (Refer to review by Glüge et al., 2020), means that they may be present at higher concentrations in water samples than PFAA at present. Whether it is preferable to measure the substances that are regulated or to highlight the risks with respect to the overall PFAS occurrence, is a choice that needs to be decided upon. In addition, choosing the appropriate PFAS analytical method, and the preparation of the sample for analysis is equally important.

4.1.1 Sampling and sample preparation

To generate reliable analytical results, it is important to maintain high-quality work in all steps when handling samples. This includes all steps from the planning of sampling to its actual execution, as well as transport, sample storage, sample processing, chemical analysis, quality assurance and reporting. A general review of aspects related to sampling and sample preparation with respect to
micropollutants can be reviewed in Baresel et al. (2015a). When sampling PFAS in wastewater, representativeness is achieved best by flow-proportional sampling over one day or one week. Samples can be stored in standard plastic bottles without preservation, but they should be stored refrigerated.

Sample preparation is especially important to detect low concentrations of PFAS, where the sample is concentrated (e.g., a 500 mL sample can be concentrated to a final volume of less than 1 mL). Sample preparation also involves the cleaning of the sample from other impurities, such as particulates and other substances which may cause interference within the analytical instruments, thus affecting the results. For some broad-spectrum analyses, sample preparation is not required, other than weighing of the sample (see section 4.1.3).

Since different PFAS are distributed differently between the water and sludge phases in a wastewater treatment plant, sampling and sample preparation can affect the results of an analysis. For example, the analysis of a sludge sample may give different answers depending on whether the particulate phase is separated from the water and each fraction is analyzed separately (as two samples) or whether both the water and particulate phases are analyzed together as one sample.

For sludge samples with low suspended solids content, the type of analysis may have to be discussed with the analytical laboratory. This is for example to avoid the laboratory filtering out particulates to avoid interference in the sample preparation. In cases where particulates hinder sample preparation, the solid phase can be considered as a separate sample from the water phase thus analyzing two samples. However, this usually involves a higher cost.

Most commonly, solid phase extraction (SPE) columns are used for water sample preparation, whereby PFAS substances will remain on a solid phase within the column while the excess liquid flows through the column. At the end of a solid phase extraction, PFAS are eluted. This means that the column is washed with a solvent so that the PFAS will detach from the SPE. It is this fraction that is analyzed. Solid phase extraction can be done with columns packed with different types of materials. The choice of column type is important, as it determines which PFAS are obtained at the end of the extraction (e.g., anion exchangers are effective in reprocessing anionic PFAS but do not work for other PFAS). Sample particulates can be problematic in solid phase extraction as they can clog the columns.

Extraction of PFAS from a more solid sample such as sludge may require the use of
microwave or ultrasonic assisted extraction methods. The solvent used for sample preparation influences which PFAS are concentrated. For example, to extract long-chain or polymeric PFAS from a SPE column or sludge sample, different solvents may be needed from those releasing short-chain PFAS.

Automated sample preparation linked to a liquid chromatography (LC; see section 4.1.2) system has become increasingly common (e.g., by having a two-dimensional liquid chromatography system with an integrated trap column). Automation reduces much of the manual time-consuming work during sample preparation. In these systems, the PFAS-containing fraction of a sample is concentrated and purified in the trap column and then transferred to the analytical separation column for the actual LC-coupled to mass spectrometry (LC-MS/MS) analysis (see section 4.1.2).

4.1.2 Target screening of specific PFAS

Most of the data available for PFAS from different types of samples have been obtained by LC-MS/MS, with SPE columns used for sample preparation (see section 4.1.1). With LC, substances are separated according to different properties. This is done with a separation column that has different properties which allow different substances to pass at different rates. In a mass spectrometer, the separated substances are ionized and then the formed ions are separated according to their charge to mass ratio. The ions are then detected and for each substance a fingerprint is obtained, which can be compared with a reference. Historically, PFAA have received the most attention and LC is the most appropriate separation technique for these substances. For some more volatile PFAS, e.g., FTOH, which are more difficult to analyze in LC-MS/MS, gas chromatography coupled to mass spectrometry is the most appropriate analytical technique (GC-MS). This report however focuses on the LC-MS/MS technique, as it is most suitable for the PFAS substances commonly found in water and sludge.

Method developments, such as using separation columns with smaller particles and pumps that are capable of higher back pressures, have led to shorter analysis time and an increase in the number of substances which can be reviewed per analysis. This is due to greater separation efficiency and separation capacity in the new generations of instruments.

After the separation column, between the LC and the MS, a process called electrospray ionization is used in the LC-MS/MS analysis. This is performed either with positive or negative ionization, which disfavors anionic PFAS (e.g., acids) in
positive ionization, and cationic PFAS in negative ionization. Efficiency improvements with which substances are ionized and transported through the mass spectrometer have meant that detection and quantification limits have been continually lowered with each new generation of instrument.

Tandem mass spectrometry (MS/MS) has long been the dominant technique for the targeted analysis of specific organic compounds, especially when the analysis requires low detection limits. This is due to high selectivity that reduces "background noise" in the analysis and enables the possibility to obtain a very specific signal from the substances to be measured. The instrument is highly selective, as it has three mass filters in a row that search for the PFAS substance of interest (Figure 4.2).

To ensure the detection of the correct substance, at least two combinations of a parent ion (molecular ion) and fragments are measured per substance, referred to as quantification and qualification ions. For a substance to be positively identified, the signal ratio between the quantification and qualification ions must be the same as the reference substance used for calibration and that measured in the sample, so that a positive identification can be made. They must also have the same retention time, which is the time that the substance remained in the column during chromatographic separation. The short-chain substance PFBA has only one distinct mass transition, which has historically led to the misidentification of signals and uncertain values in analyses.

The MS/MS analysis can also be performed with a combination of low-resolution mass filters and high-resolution mass spectrometry such as an orbitrap (e.g., Q-Exactive orbitrap) or time-of-flight mass spectrometers (e.g., Q-TOF). These instruments offer the possibility of a higher selectivity for the second dimension of the mass spectrometer. Different configurations of analytical instruments provide different advantages in terms of investment cost, robustness, selectivity, detection limits, etc.
PFAS standards containing different isotopes (i.e., same element but different mass) are used to quantify the substances present in the samples. These are referred to as either internal standards (IS) or surrogate standards. One advantage of quantification using the IS is that it compensates for losses during sample processing and effects in the analysis itself, such as signal suppression or signal amplification (so-called ion suppression or ion amplification).

Since PFAS can be found in solvents and materials used in the analytical equipment, background level checks with blank samples (solvent blanks, and process/extraction blanks) are required. To avoid the risk of background contamination from water or solvents used in the analysis process, a separate column is often used as reference to compare the retention times against the substances coming from the samples. Some manufacturers of analytical instruments offer materials that is not suitable for PFAS analysis.

4.1.3 Broad-spectrum analyses for PFAS

Modern LC-MS/MS instruments can analyze many compounds per analysis, but it has been shown that these targeted analyses can often measure only part of the fluorine-containing organics present (Schultes et al., 2018). In addition to the specific PFAS that can be analyzed by a targeted analysis, there are thousands of other PFAS and precursors that may need to be quantified in different contexts. For this purpose, a variety of broad-spectrum analyses are available.

4.1.3.1 Total oxidizable precursors (TOP)

The Total Oxidizable Precursors (TOP) analysis provides information on PFAS and precursors that are present and can potentially be converted into "classical" PFAS (Figure 4.3). These can be polymeric PFAS or other PFAS, such as sulphonates or phosphonates. A large proportion of the PFAS that are currently in use are of these types, and therefore valuable information can be generated via TOP analysis. The results can for example be used to explain the presence of classical PFAS even if no likely direct source of emissions of these exists. However, there are PFAS that do not convert to PFAA under the conditions used in a TOP analysis, e.g., Ester-based PFAS such as F-53B.

The main disadvantage of TOP analysis is that it does not provide any information on which precursors are present, but only tells what transformation products are formed. TOP analysis provides an answer to how much of the traditional PFAS is formed under oxidative conditions (i.e., with chemical reactions) from substances that are not measured by a targeted analysis. For example, if polymeric PFAS are
present in a sludge sample, they can often be degraded, releasing PFAS with short or medium chains.

A TOP analysis measures the PFAS content of a given sample before and after oxidation and compares the levels of the PFAS that can be measured with the targeted measurement method. Oxidation is often carried out under heating using Potassium Persulphate as the oxidizing agent, but other oxidizing agents can be used. It is however worth noting that there is some uncertainty in oxidation efficiency for all the thousands of PFAS available.

![Figure 4.3 TOP analysis workflow.](image)

### 4.1.3.2 Extractable Organic Fluorine (EOF), Total Organic Fluorine (TOF) and Total Fluorine (TF)

One technique that has been used to obtain a complete picture of the total PFAS content is Combustion Ion Chromatography (CIC). This is performed by burning an extract of a sample (in the case of Extractable Organic Fluorine, EOF) or a whole sample and collecting the fluorine ions released by combustion. The collection takes place in a scrubber where fluorine in the gas phase is collected in a water sample. The amount of fluorine is then quantified in the water sample by ion chromatography (Figure 4.4). The choice of extraction solvent will influence which PFAS will be extracted and thereby included in the fluorine analysis. In the EOF analysis, only an extract of the sample is incinerated, which means that the inorganic fluorine content does not accompany the extract and is therefore not analyzed. In the Total Fluorine (TF) analysis, the whole sample is incinerated, and the analysis result shows the total fluorine contents, both organic and inorganic. EOF analyses are performed on water samples instead of TF due to the fact that the incineration of the whole water sample volume is often not feasible. The TF analysis does not provide any information of PFAS occurrence or its precursors in water samples. Water samples often have a higher concentration of inorganic fluorine in the form of fluoride, compared to PFAS concentrations. In sludge
samples, the presence of inorganic fluorine is low, which means that most of the fluorine comes from the decomposition of organic substances. TF analysis performed on sludge is therefore often referred to as TOF analysis, although in fact inorganic fluorine content is also included. TF/TOF analyses can therefore provide useful information when analyzing sludge samples but should not be used for water samples. In addition, it is not possible to distinguish between fluoride originating from organic substances and inorganic fluoride after a sample has been incinerated and measured with CIC. Thus, the extraction itself determines the proportion of the total fluorine content analyzed. The method does neither provide information on which PFAS were originally present. Uncertainties on the levels of PFAS present in the sample will always exist, for example, one molecule of PFBA contains 7 fluorine atoms and one molecule of PFDoDA contains 23 fluorine atoms. Therefore, a targeted analysis of commonly occurring PFAS is a valuable complement to CIC. The comparison provides valuable information on the concentrations of PFAS and whether there may be PFAS substances that are not included in the targeted analysis. It should be noted that CIC analysis has a higher detection limit compared to LC-MS/MS analysis.

Recently, an analytical comparison was published containing results from three different laboratories that performed EOF and one laboratory that performed a targeted analysis (Kärrman, 2021). This comparison of results was fairly coherent among the different laboratories. It also showed that only a small fraction of the organically bound fluorine could be described by the targeted analysis. This clearly shows the importance of choosing the right analysis to answer the question at hand. However, there are currently no reference or limit values that can be used for interpretation of broad-spectrum analyses. Therefore, it is only when a relative comparison of the total amount of PFAS can provide additional knowledge, such as in mass balances.
The quantification of TF can be considered as the broadest analysis of PFAS (Figure 4.1) and is mainly used for solid matrices such as contaminated soil. The TF analysis indicates how much total fluorine (inorganic and organic) is present in a sample. The TF analysis can be used to obtain an overall picture of PFAS in certain matrices, but it needs to be combined with targeted analyses. TF can also be measured by Particle-Induced γ-ray Emission Spectroscopy (PIGE) and Instrumental Neutron Activation Analysis (INAA), in addition to CIC. The techniques have different advantages and disadvantages which have been discussed by Schultes et al. (2019).

4.1.4 Suspect and non-target screening methods

In suspect and non-target screening, high-resolution MS is used to find signals from chemical substances. They are then identified either via a list of expected mass numbers (suspect screening) or by the searching of databases of mass numbers (non-target screening). Some modern mass spectrometers give with relatively high certainty a signal which corresponds to a certain combination of elements (e.g., 498.9297u for the ion of PFOS - C8F17O3S). This is in part due to their high resolution and accuracy in the determination of mass number. Screening analyses usually search for a wide range of mass numbers compared to a target analysis, where the instrument searches for certain combination of mass numbers. This makes them less sensitive than the targeted analysis. The absence of reference substances also means that the analysis does not give quantitative data of the
sample, it only gives information on which substances are detected. The suspect screening method depends on available information to be detected on the suspected substances e.g., based on expected transformation products or physicochemical properties in the literature or databases (Joerss et al., 2022). Therefore, this analytical method may not be suitable for complex matrices.

Such prior information of the substances to be "detected" is not required in non-target screening. The method relies on various priorities such as signal intensity, frequency, and isotope pattern characteristics. A non-target screening is usually complemented with a suspect screening or target screening depending on the outcome. Both suspect and non-target screening methods are used mostly in research. As they are time consuming and costly, they are not a common analytical method applied to the PFAS-related work in the water sectors.

### 4.2 Choice of PFAS analyses

Analysis of raw and drinking water compiled by the Swedish Food Agency in 2021 showed that of the PFAS11 concentrations measured, a large part consisted of PFAS4. This is also in line with EFSA’s assessment for PFAS in water and biota (EFSA 2020). The EU Directive also defines a limit value for ΣPFAS20 of 100 ng/L, and the Swedish Food Agency new national limit for ΣPFAS21 including 6:2 FTSA, with limit value unchanged at 100 ng/L.

Commercial analytical laboratories offer several packages containing different PFAS, with the smallest containing only PFOS and PFOA. Many laboratories offer a package that includes all substances included in PFAS11, and among these substances are also the four substances in PFAS4. There are also analytical packages that include all PFAS included in PFAS21. Some older packages include 25 or 32 different PFAS or even more, but these sometimes do not include all substances included in PFAS21.

The number of substances included in different PFAS analytical packages is constantly increasing, however, there is no common understanding of how many substances should be measured. The number of substances that can be regularly included in analyses is limited by factors such as the commercial availability of standard substances. Also, maintaining standard mixtures and mixtures of corresponding isotope-labelled standards is costly for analytical laboratories.

In the future, it is possible that the drinking water limit will also apply to surface water, at the intake point of raw water for drinking water production. Therefore, it
may be useful for wastewater treatment plant operators to analyze PFAS21 already now, especially in cases where there is a risk of exceeding $\Sigma$PFAS21. The collection of this type of data over several years would be useful if the assessment basis in surface water should change to include more substances (e.g., possible extension from PFAS11 to PFAS21).

As mentioned earlier, the length of the fluorinated carbon chains can affect whether the PFAS compounds will be found mainly in wastewater or in sludge. Those substances with carbon chains of up to eight carbon atoms are found in the water phase (e.g., PFHxA and PFHpA with six and seven carbon atoms, respectively). If substances with longer carbon chains (e.g., PFUnDA with 11 carbons) are found in the water phase, it is a sign of either high contamination or sampling in proximity of a point source. For example, analyses of landfill leachate from several samples taken during a half year period showed that only PFPeS and PFHpS (five carbons and seven carbons, respectively) were detected apart from PFAS11 substances, and that $\Sigma$PFAS21 was only about 2 % higher than $\Sigma$PFAS11 (unpublished data, IVL). Similar PFAS occurrence can be assumed for wastewater. Therefore, analyzing only PFAS11 will probably capture most of the PFAS content in the water while PFPeS and PFHpS are overlooked.

The substances with longer per-fluorinated carbon chains and the large molecules from polymeric PFAS will mainly be found in the sludge. As such, PFAS21, which includes PFAS with longer carbon chains (9 – 13 carbons) that are normally not present in high concentrations in water as well as PFPeS and PFHpS, should be analyzed to investigate PFAS in the sludge phase.

If only PFAS11 or the 20 PFAS proposed by the EU are measured, then the precursors that can form PFPeA or PFHpA upon degradation would be overlooked. Side chain polymers or polymeric PFAS could give rise to shorter PFAS groups with carboxylic acid. Shorter PFAS could also be formed from substances with carbon chains that are not fully fluorinated such as telomer alcohols and 2:6-FTSA. Additionally, some sulphonic acids could be formed from precursors (e.g., PFPeS from sulphone-amid PFAS such as MeFOSA, EtFOSE or MeFOSAA).

It is recommended that PFAS measurements from the processing air in WWTP should at least include FTOH, FOSA and FOSE in the gas phase and PFAS11 in the particle phase (aerosols). A broader spectrum measurement via EOF or TOF may also be beneficial. As air sampling requires special equipment such as high-volume
air samplers or passive sampling (Ahrens et al., 2011; 2013), it must therefore be performed by a person with expert knowledge.

For unknown accidental releases such as those from construction sites, PFAS11 should be sufficient for an initial assessment to rule out PFAS contamination. Even a broader fluorine analysis such as EOF may be beneficial to gain a better idea on possible unknown PFAS.

Additional analyses are justified to complement the target PFAS analyses when a better understanding of all PFAS is needed. For example, to quantify PFAS flow across a WWTP or specific treatment processes, TOF (for sludge only), EOF or TOP in addition to PFAS11 or PFAS21 can provide information on the total amount of PFAS, precursors as well as unknown PFAS. With this information, not only the impact of the treatment processes on specific PFAS but also the total amount of PFAS can be investigated. PFAS with more than seven carbon atoms in the perfluorinated carbon chains are found in higher concentrations in sludge, as well as larger perfluorinated substances such as polymers with several fluorinated carbon chains. Therefore, PFAS21 is recommended for sludge. The presence of precursors is more likely in sludge, the TOP analysis may also be useful.

EOF analyses are recommended to confirm PFAS destruction, as potential PFAS destruction techniques, and not just a conversion of PFAS to other PFAS that cannot be detected by a limited targeted analysis. However, the high cost of these broad-spectrum analyses incurs an additional cost to targeted PFAS analyses that are usually needed. Other substances that may be worth including in a comprehensive study include fluorotelomer sulphonic acids (6:2 and 8:2 FTSA), sulphonamide PFAS (e.g., MeFOSA, EtFOSA, MeFOSE, EtFOSE). To look ahead, it may be even useful to include parameters for common substitute to PFAS such as Gen-X, phosphoric acid esters, mono- di- and tri-PAPs, F-53B, Me- and EtFOSAA, cationic or zwitterionic PFAS or mono- and di-PAPs. No studies have yet detected F-53B or Gen-X in wastewater in Sweden, but they are present in other countries and are used for example in electronic products.

The proposal of revised Framework Directive includes the limit value of ΣPFOA equivalent that is based on recalculation of concentrations of 24 different PFAS (see section 3.2.3 for more details). Even though many analytic packages cover more than 24 substances, none of the available packages offered by commercial laboratories in Sweden include all of the substances that are covered by the proposed Directive amendment. It is, however, expected that the package that
include all of the substances will be offered soon after the amendment is adopted. In that case it is recommended to analyze the treated water using this package, at least initially. Then, it will be possible to calculate ΣPFOA equivalent using all the 24 substances and compare it to ΣPFOA equivalent calculated from data of the more used packages, such as PFAS11.

In conclusion, it can be assumed that PFAS11 meets the needs for monitoring of the current regulatory framework for discharges of treated water. However, to adapt to future regulations, it may be useful to change to PFAS21 according to the new Water Directive (or PFAS24 as Water Framework Directive), or at least to increase the parameters for some samples of incoming and outgoing water, to create a database. The price of PFAS21 is only about 20% higher than that of PFAS11, so the extra cost could be justified.

4.3 Reporting limits and result follow up

The reporting limit for a PFAS analysis specifies the minimum concentrations of PFAS that can be identified by a specific analysis. Analytical results below the reporting limit are usually indicated by a less than symbol (<). Some analytical laboratories do not indicate whether the reporting limit corresponds to the lowest detectable level (i.e., limit of detection; LOD), or the lowest quantifiable level (i.e., limit of quantification; LOQ). LOQ describes substances with concentrations below the reporting limit which can be detected in the sample, but their specific concentration cannot be quantified. LOD is thus usually lower than LOQ. A common practice of LOQ and LOD, as used for example in pharmaceutical analysis, is that concentrations reported as “< LOD” are either set equal to 0 or equal to the LOD. Concentrations reported as “< LOQ” are usually set to LOQ/2 (i.e., half the limit of quantification) to indicate that the substance was detected but could not be quantified. The value of LOQ/2 should be above LOD. Ideally, both LOQ and LOD should be reported for each analysis batch. However, this becomes less important when lower reporting limits that can be achieved in the lab. LOD and LOQ limits may vary between different analysis batches for the same substance due to the sample matrix, sample processing, and variations to the analytical conditions vary. To facilitate smoother managing and reporting process, many commercial analytical laboratories have chosen to have fixed reporting limits for a given substance in each matrix, which then corresponds to highest reporting limit. However, in such cases, if the general reporting limits are set high, it may mean the concentrations that can be detected or quantified are concluded and reported as below the reporting limit.
PFAS analytical accuracy becomes important when monitoring environmental emissions and the possible exceedance of guidelines or limit values in the recipients. The analytical accuracy that can be achieved varies largely with the matrix to be analyzed (e.g., water or sludge) and potential interference by substances in individual sample matrices. An increase in analytical accuracy, which means lower reporting limit, may also mean a higher analytical cost, as it requires improved and more accurate sample processing and data analysis, and in some cases, re-analyses.

For sludge and water (including surface water, groundwater, wastewater, and leachate), there are LODs and LOQs that could be considered as standard for target analyses. Table 4.1 shows examples of LODs and LOQs for PFAS11 in both water and sludge phases that can be achieved with current analytical methods. For PFOS, for example, concentrations above 0.05 ng/L can be detected, which means that results can be compared with the limit value for PFOS in inland surface waters and coastal waters. This may be important for wastewater treatment plants with very low dilution in the recipients.

Table 4.1 Reporting limits in various matrices relevant to PFAS in water and sewage.

<table>
<thead>
<tr>
<th>PFAS-substances</th>
<th>Water phase (ng/L)</th>
<th>Sludge phase (ng/g TS)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>LOD</td>
<td>LOQ</td>
</tr>
<tr>
<td>PFBA</td>
<td>0.07</td>
<td>0.20</td>
</tr>
<tr>
<td>PFPeA</td>
<td>0.07</td>
<td>0.20</td>
</tr>
<tr>
<td>PFHxA</td>
<td>0.14</td>
<td>0.40</td>
</tr>
<tr>
<td>PFHpA</td>
<td>0.06</td>
<td>0.20</td>
</tr>
<tr>
<td>PFOA</td>
<td>0.06</td>
<td>0.20</td>
</tr>
<tr>
<td>PFNA</td>
<td>0.08</td>
<td>0.25</td>
</tr>
<tr>
<td>PFDA</td>
<td>0.08</td>
<td>0.25</td>
</tr>
<tr>
<td>PFBS</td>
<td>0.07</td>
<td>0.20</td>
</tr>
<tr>
<td>PFHxS</td>
<td>0.07</td>
<td>0.20</td>
</tr>
<tr>
<td>PFOS</td>
<td>0.05</td>
<td>0.15</td>
</tr>
<tr>
<td>6:2 FTSA</td>
<td>0.07</td>
<td>0.20</td>
</tr>
</tbody>
</table>

Active monitoring of analytical results is also important. Some samples are difficult to analyze, and if the reporting limit is consistently too high, a discussion with the laboratory is necessary. Request for re-analysis to obtain a reasonable measurements or analysis by another laboratory must be considered. Unreasonable levels or reporting errors could still occur despite a laboratory’s quality check/control measures. It is recommended to have close and continuous follow-up with the laboratory where analyses are being carried out. Malovanyy et al. (2021) illustrated the difference between analytical results reported by different laboratories for the same leachate samples. Although the results were generally in
agreement among different laboratories, values of some PFAS and reporting limits largely varied. A regular inter-calibration of analytical laboratories could provide better certainty and reduce the risk that the results are influenced by the choice of analytical laboratories. In Sweden, Swedish Board for Accreditation and Conformity Assessment (SWEDAC) accredited analytical methods require regular participation in an intercalibration study or test comparison every 4 years, or analysis of certified reference materials (CRM). However, it may be difficult to find intercalibrations or CRMs for certain substances and sample types. Analytical methods may in such case be accredited for some sample matrices and some of the substances included in the method, but not all. A discussion should be held with contracting laboratories regarding which reference samples are to be evaluated, the latest intercalibration, and test comparison.

5 PFAS Knowledge at Swedish wastewater treatment plants

PFAS knowledge regarding its presence in wastewater and sludge, treatment efficiency, and impact on recipients has been available by some Swedish wastewater organizations that have extensively worked with PFAS problems. In those WWTPs where more ambitious mapping campaigns were conducted, the effluents have generally been contaminated with PFAS (e.g., PFAS spread via drinking water, inflow of contaminated groundwater, or from landfills without leachate treatments).

However, the project team has not been able to locate a database or compilation that includes the PFAS analyses that have been carried out in connection with Swedish WWTPs. The available data is dispersed and difficult to access, making it challenging to collaborate and share with other stakeholders. In many WWTPs, PFAS analyses have not been carried out on influent and effluent water nor in sludge. It can therefore be said that PFAS knowledge regarding Swedish WWTPs and their impact of dispersion on soil and the environment is limited. The following section will give an overview of the complexity of the PFAS issue at Swedish WWTPs, which makes generalization of trends difficult.

5.1 PFAS mapping survey at Swedish wastewater treatment plants

This section provides a summary of major national and regional mapping campaigns compiled by the project team. It includes PFAS occurrence in influent
and/or effluent water from Swedish WWTPs, as well as in the sludge produced and the relevant recipient waters. Apart from these campaigns, single measurements or a series of measurements including analyses of PFAS have been carried out by a few WWTPs around Sweden. Table 5.1 provides an overview of some national and regional sampling campaigns that have mapped PFAS occurrence at Swedish wastewater treatment plants.

From 2019 to 2020, a sampling campaign was conducted to investigate organic micropollutants and their impact on the three largest lakes in Sweden: Lake Vänern, Lake Vättern and Lake Mälaren. The project was carried out by the Swedish University of Agricultural Sciences (SLU) whereby 13 PFAS were analyzed as micropollutants from 24 total treatment plants. Time- or flow-proportional daily composite samples were gathered from influent and effluent waters from the WWTPs. Four sampling occasions were done, however not all treatment plants participated on all occasions. A few recipient samples from the treatment plants were also included in the campaign (Malnes et al., 2021).

Within the National Environmental Monitoring Program, led by the Swedish Agency for Marine and Water Management (HaV) and the Swedish Environmental Protection Agency (NV), PFAS have been analyzed in treated wastewater since 2011 and in sludge since 2004. The substances analyzed have, however, varied over time. Nine wastewater treatment plants: Henriksdal, Ryaverket, Öns, Gässlösa, Ellinge, Nolhaga, Borlänge, Bergkvara, and Bollebygds, have been included in the environmental monitoring and flow-proportional weekly composite samples were taken every October. For the sludge, aggregated samples were obtained (Haglund, 2019).

Directed by the NV, SLU carried out a survey of PFAS in surface and groundwater in 2015. 26 PFAS were analyzed in 504 water samples, of which 13 samples were from effluents belonging to different WWTPs (Ahrens et al., 2016; Gobelius et al., 2018). Most of the wastewater samples were collected as grab samples (each sample was collected at one point of time from one location), providing only a snapshot of PFAS concentrations, while others were collected as time-proportional composite samples.

Pirzadeh et al. (2021) analyzed PFOS and PFOA at six WWTPs and in their associated recipient waters, located in the Skåne region of southern Sweden, during four sampling events in 2020. Measured PFOS concentrations exceeded the PFOS limit of 0.65 ng/L for inland water, during at least one sampling occasion at
Ekeby, Ellinge, Nyvång, Svalöv and Ekebro. However, the study did not include a mapping of other possible pathways to the WWTPs. Moreover, their analytical reporting limit of 3 ng/L for PFOS was relatively high. As the limit value was lower than the reporting limit, it is possible the concentrations above the limit value may have been neglected.

During 2018 and 2022, NV provided a grant to carry out preliminary studies and installations for the reduction of micropollutants at Swedish WWTPs. With this project funding, several wastewater organizations such as Käppala Association, Nordvästra Skånes Vatten och Avlopp - NSVA, Vatten & Miljö i Väst AB - VIVAB, Falun Energi och Vatten AB - FEVAB, Östersund Vatten and Miljö have also analyzed PFAS at WWTPs and in recipient waters. Sampling, the substances analyzed, and scope of the campaigns, however differed between the various projects. Some projects analyzed PFAS in some (and not all) samples collected. One of the most ambitious regional sampling campaigns granted was implemented in Skåne by NSVA. 13 WWTPs that belong to NSVA collected samples which were analyzed for PFAS11. Weekly composite samples of influent, effluent, and associated recipient waters (taken randomly) were collected and analyzed by IVL Swedish Environmental Institute.

Table 5.1 Selection of relevant national and regional sampling campaigns that have mapped the presence of PFAS at Swedish sewage treatment plants.

<table>
<thead>
<tr>
<th>References</th>
<th>Descriptions</th>
</tr>
</thead>
<tbody>
<tr>
<td>NSVA (2022)</td>
<td>Mapping of PFAS11 and other micropollutants at 13 of NSVA's wastewater treatment plants and in associated recipients based on four sampling occasions during 2021 - 2022 (ongoing).</td>
</tr>
<tr>
<td>Golovko et al., (2021) [Link]</td>
<td>Occurrence and removal of organic micropollutants including PFAS at 15 wastewater treatment plants (anonymized) and their impact on receiving recipients. However, the detection limits for PFOS and PFOA have been relatively high with &gt;6 ng/L and &gt;3 ng/L respectively, which makes an assessment difficult as the limit value was lower than the reporting limit.</td>
</tr>
<tr>
<td>Malnes et al., (2021) [Link]</td>
<td>Occurrence of organic environmental pollutants including PFAS in waterworks, sewage treatment plants and waterways around and in lakes Vänern, Vättern and Mälaren based on a sampling campaign carried out during the period June 2019 to July 2020. Calculated average values for PFOS, according to the study, often exceeded the environmental quality standard in the studied surface waters.</td>
</tr>
<tr>
<td>Pirzadeh et al., (2021) [Link]</td>
<td>Compilation of the presence of PFOS and PFOA (as well as other micropolllutants) in outgoing wastewater at seven Scanian ARVs (Rosendal (Tomelilla), Svalöv, Nyvång (Åstorp), Södra Sandby; Ellinge (Eslöv), Ekebro and Ekeby (Bjuv)) and in associated recipients. However, the quantification limit for PFOS and PFOA has been relatively high with &gt;= 3 ng/L, which makes an assessment difficult as the limit value was lower than the reporting limit.</td>
</tr>
<tr>
<td>Haglund (2019) [Link]</td>
<td>Compilation of the presence of PFAS and partial PFAS precursors as well as other organic substances in outgoing wastewater (from 2011) and/or sludge (from 2004) at nine Swedish sewage treatment plants (Henriksdal, Ryaverket, Ön ARV, Gäsölösa ARV, Ellinge ARV, Nolhaga ARV, Bollebygd ARV, Borlänge ARV and Torsås ARV.</td>
</tr>
<tr>
<td>Ahrens et al., (2016) [Link]</td>
<td>Compilation of the presence of 26 PFAS in 502 water samples from groundwater, surface water, wastewater, and leachate from landfills to determine background concentrations of PFAS in the Swedish water environment.</td>
</tr>
<tr>
<td>Gobelius et al., (2018) [Link]</td>
<td>Examining 26 PFAS in Swedish groundwater and surface water with almost 500 samples. Highest ΣPFAS26 concentrations were detected in surface water (13,000 ng/L) and groundwater (6,400 ng/L).</td>
</tr>
</tbody>
</table>
In addition to the studies shown in the table 5.1, there are some other regional measurement campaigns that included PFAS (or at least PFOS) analysis in surface water bodies. In these studies, however, the link between WWTPs and water bodies were not investigated. One example of such studies is by the County Council of Västra Götaland (2018, 2020), with a campaign that analyzed one to two grab samples from 50 surface waters, both from rivers and marine waters during the summers and autumns of 2017, 2018 and 2019. PFOS concentrations were often several times higher than the limit value (0.65 ng/L) in the examined inland waters samples. However, mapping of the possible PFAS sources was not included in the study. In those analyses where concentrations were below the limit value, the measured concentrations were close to the limit value. The same trends could be seen in the coastal water samples, although a higher dilution was assumed since the limit value of 0.13 ng/L was low.

The County Council of Jämtland (2020) also took water samples for analysis of PFAS substances in several surface water bodies in the county during 2018 – 2019. The results show that PFAS were present in several sample points and elevated concentrations were measured mainly in or near Östersund and northern Frösön (former Frösön air force bases: F4), as well as at Bräcke and Sveg. Results over the limit value for PFOS were observed at several sites, and they found three sites where the results exceeded the recommended action limit for drinking water for ΣPFAS11 of 90 ng/L. Further sampling of PFAS in surface water and fish is planned.

5.2 Treatment efficiency and PFAS transport in Swedish wastewater treatment plants

In addition to the survey by NSVA (2022), PFAS concentrations in the few other national and regional screenings are listed in Table 5.1. There are water organizations that have conducted or are in the process of conducting PFAS surveys at their wastewater treatment plants, using samples collected from influent and effluent waters. PFAS concentrations in water entering Swedish WWTPs naturally varies between plants depending on the composition and different sources upstream of PFAS. The proportion and variation of infiltration water leaking into the sewer system also affects the occurrence of PFAS. Depending on the different PFAS sources upstream, the composition of PFAS precursors and PFAS substances in the incoming wastewater may vary. This may affect the conversion of PFAS precursors into specific PFAS, which are quantified via analyses. The calculated treatment efficiency may therefore also be affected, since
the removal of PFAS from the water phase (i.e., the treatment effect) is concealed by the formation of PFASs from precursors.

### 5.2.1 Results of PFAS measurements at Swedish WWTPs

Table 5.2 shows results of some recent surveys mentioned in previous chapter, at various Swedish WWTPs of different sizes. It shows the incoming concentrations of PFOA, PFOS and ΣPFAS11 and reduction during the treatment process. The data compilation (Table 5.2) illustrates that there can be large difference in PFAS values and their treatment efficiencies among different WWTPs, but also variation over time is observed at within the same plant over time. This is despite all analyses were based on flow proportional weekly composite samples in which short-term variations are smoothed out. The treatment efficiency also varies between plants and between measurement events at the same WWTP. Data based on few measurements, or several measurements but over a limited period, should be processed with caution as these measurements can at most only give a first indication for the sampling WWTP. These tendencies where types of PFAS and precursors varies in water incoming to the wastewater treatment plant were also observed in a four-year study conducted in two large scale WWTP in Australia (Nguyen et al., 2019).

Based on the wide range of data reviewed, it is not recommended to generalize PFAS concentrations nor the capacity to reduce PFAS at Swedish WWTPs. However, the reduction of PFOA, PFOS and ΣPFAS11 generally appears to be moderate or negative (increasing concentrations over the treatment process) at most wastewater treatment plants (Table 5.2). The reduction here refers only to the change in water phase concentration over the treatment process. This, however, does not imply biodegradation but rather a transfer of PFAS from the water to the sludge phase. As previously discussed, precursors in incoming wastewater may contribute to the formation of PFCA/PFSA and the treatment effect therefore appears to be low or non-existent for the specific PFAS analyzed. It is difficult to explain why for example the Stadskvarn WWTP in Skövde with slightly high PFOS concentration in influent has a very good reduction. It could possibly be due to large variations in the PFAS load to the WWTP. It can also be due to the specific composition of the PFAS load with different sources such as untreated landfill water, industrial effluents and probably soil water from one of Sweden’s largest contaminated fire training sites that also influence PFOS formation in the WWTP. Alternatively, there is an unknown reason that PFOS is transferred to a greater extent to the sludge phase at this WWTP, but this cannot be assessed without access to sludge analyses. It can be noted that the Himmerfjärdsverket achieves a...
very good removal of PFOS in the new treatment process (MBR) compared to the current activated sludge process. At the same time, the old conventional activated sludge process has better reduction of ΣPFAS11 than in the MBR process. Other WWTPs such as Sjölunda, Kungsängen, Vikverket and Järpen also show a higher reduction of both PFOA and PFOS compared to other plants. Even for these WWTPs, it is difficult to pinpoint the cause, although at Sjölunda, the final flotation process could be the contributing factor. Without a more detailed mass balance including the PFAS content in sludge (both targeted and broad-spectrum analyses), it is difficult to assess why the treatment efficiency is anomalously high at some WWTPs. Also, analytical uncertainties cannot be excluded as a contributing factor to deviating results.

Table 5.2 shows concentrations in incoming wastewater and treatment efficiency for a selection of PFASs. A detailed review of how these WWTPs differs in terms of process solutions and load remains to be done, so are the identifications of different trends and correlations, and how these may explain the differences.

### Table 5.2 Levels of PFOA, PFOS and ΣPFAS11 and treatment efficiency at some Swedish sewage treatment plants

Compilation of recent surveys at various Swedish WWTP of different sizes. It shows the incoming concentrations of PFOA, PFOS and ΣPFAS11 and reduction during the treatment process. The average, lowest and highest levels, and effects, respectively, are written in parentheses if the information is available.

<table>
<thead>
<tr>
<th>Sewage treatment plant (Sampling information) (reference)</th>
<th>Content in incoming wastewater ng/L</th>
<th>Reduction over the sewage treatment plant (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Average (min; max content)</td>
<td>Average (min; max reduction)</td>
</tr>
<tr>
<td>Getteröverket, Varberg (Weekly sample from w.37, 2020; w.15 and w.29, 2021) (Baresel et al., 2021a)</td>
<td>PFOA: 3.1 (2.7; 3.4) PFS: 3.2 (1.7; 4.6) ΣPFAS11: 12.7 (8.0; 20.4)</td>
<td>PFOA: -3 % (-12; 3) PFS: 26 % (-40; 48) ΣPFAS11: -12 % (-58; 7)</td>
</tr>
<tr>
<td>Stadskvarn, Skövde (Weekly sample from w.10, 25, 35 and 50, 2020) (Baresel et al., 2021b)</td>
<td>PFOA: 4.4 (&lt;3; 8.7) PFS: 11.3 (&lt;3; 25) ΣPFAS11: 36.6 (12; 71.4)</td>
<td>PFOA: -20 % (-117; 8) PFS: 72 % (41; 85) ΣPFAS11: -76 % (-312; -29)</td>
</tr>
<tr>
<td>Främby, Falun (Weekly sample from w.21, 24, 26 and 37, 2019) (Hedén et al., 2020)</td>
<td>PFOA: 18.3 (5.4; 30) PFS: 3.75 (3; 4.2) ΣPFAS11: 64.9 (41.4; 77.9)</td>
<td>PFOA: 33 % (-8; 67) PFS: 25 % (22; 29) ΣPFAS11: 57 % (51; 63)</td>
</tr>
<tr>
<td>Himmerfjärdsverket (AS-process) (7 Weekly samples from Oct 2020 - Mar 2022) (Baresel et al., 2022)</td>
<td>PFOA: 3.3 (2.6; 4.4) PFS: 2.83 (2.1; 4.4) ΣPFAS11: 45.4 (26.3; 107.9)</td>
<td>PFOA: 3 % (-33; 63) PFS: -13 % (-37; 9) ΣPFAS11: 16 % (-5; 60)</td>
</tr>
<tr>
<td>Himmerfjärdsverket (MBR-process) (7 Weekly samples from Oct 2020 - Mar 2022) (Baresel et al., 2022)</td>
<td>PFOA: 3.3 (2.6; 4.4) PFS: 2.8 (2.1; 4.4) ΣPFAS11: 45.4 (26.3; 107.9)</td>
<td>PFOA: -20 % (-57; 6) PFS: 84 % (59; 95) ΣPFAS11: 7 % (-20; 30)</td>
</tr>
<tr>
<td>Himmerfjärdsverket (1 Weekly sample from Nov 2014) (Allard och Wahlberg, 2017)</td>
<td>PFOA: 8.3 PFS: 11</td>
<td>PFOA: 58 % PFS: -11 %</td>
</tr>
<tr>
<td>Henriksdal* (4 Weekly samples, 3 from 2018, w.40, 2019) (Närhi et al., 2021)</td>
<td>PFOA: 4.4 (1.4; 12) PFS: 1.7 (&lt;0.1; 407) ΣPFAS11: 70</td>
<td>PFOA: 26.7 % PFS: -606 % ΣPFAS11: 38.5 %</td>
</tr>
<tr>
<td>Henriksdal (1 Weekly sample from Nov 2014) (Allard och Wahlberg, 2017)</td>
<td>PFOA: 7.3 PFS: 14</td>
<td>PFOA: 87 % PFS: 34 %</td>
</tr>
<tr>
<td>Öns WWTP (1 Weekly sample from Nov 2014) (Allard och Wahlberg, 2017)</td>
<td>PFOA: 5 PFS: 4.9</td>
<td>PFOA: &gt;97 % PFS: 19 %</td>
</tr>
<tr>
<td>Location</td>
<td>Sampling Details</td>
<td>PFOS</td>
</tr>
<tr>
<td>---------------------------</td>
<td>----------------------------------------------------------------------------------</td>
<td>------</td>
</tr>
<tr>
<td>Järpen WWTP</td>
<td>(Weekly sample from w.49, 2019; w.6, w.9 and w.32, 2020)</td>
<td>2.5</td>
</tr>
<tr>
<td>Vikverket</td>
<td>(Weekly sample from w.49, 2020)</td>
<td></td>
</tr>
<tr>
<td>Borlänge WWTP</td>
<td>(Weekly sample from w.45, 2021)</td>
<td>2.5</td>
</tr>
<tr>
<td>Öresundsverket</td>
<td>(Weekly sample from w.45, 2021)</td>
<td>1.4</td>
</tr>
<tr>
<td>Nyängsverket</td>
<td>(Weekly sample from w.45, 2021)</td>
<td>3.4</td>
</tr>
<tr>
<td>Kågeröd WWTP</td>
<td>(Weekly sample from w.45, 2021)</td>
<td>1.6</td>
</tr>
<tr>
<td>Lundåkraverket</td>
<td>(Weekly sample from w.45, 2021)</td>
<td>2.7</td>
</tr>
<tr>
<td>Sjölunda WWTP (Jonstrup et al., 2020)</td>
<td>(Daily samples from w.12, 2018 – w.11, 2019)</td>
<td>51.3</td>
</tr>
<tr>
<td>Järpen WWTP (Snis et al., 2020)</td>
<td>(Weekly sample from w.49, 19, 2019; w.6, w.9 and w.32, 2020)</td>
<td>6.1</td>
</tr>
</tbody>
</table>

*Compilation of knowledge and guidance for water/wastewater actors regarding PFAS*
5.2.2 Kungsängsverket example

To illustrate an example of PFAS flow in a WWTP, PFAS flows in the Kungsängsverket in Uppsala is shown in Figure 5.1. The PFAS flows in this figure are calculated as flow-weighted averages based on 12 monthly aggregated samples from daily samples for water and grab samples for sludge in 2021.

The reporting limit in the analysis reports from the laboratory were assumed to correspond to the LOQ and concentrations below this level were set to LOQ/2. According Figure 5.1, Kungsängsverket have slightly higher PFAS concentrations in incoming wastewater compared to a "general" Swedish WWTPs. Additionally, there was a small or negative reduction of most PFAS over the treatment processes. Furthermore, only a very small proportion of the analyzed PFAS ended up in the sludge, and the mass balance of PFASs analyzed does not add up. This could be due to the conversion of PFAS precursors to PFAS as well as the limited detection of other PFAS (i.e., only PFAS11 were analyzed).

There were no measured data available for the air emissions at Kungsängsverket, and the values in the Figure 5.1 are calculated based on the measurements by Ahrens et al. (2011), using estimate of PFOS air emission at 45 µg/year/PE (population equivalent). This estimate was calculated using a method used at a Canadian WWTP. The total PFOS emission via processing air at Kungsängsverket in 2021 is therefore calculated to be 7700 mg/year 0.0077 kg/year). This amount represents ca 4.3 % of PFOS that enters the treatment plant. The air emissions are however not included in Figure 5.1.
Of the PFAS reported, PFOS was the one that could be identified in sludge (Figure 5.2). The transport route via sludge was less than one tenth compared to the effluent from the plant for PFOA, PFOS, and ΣPFAS11 (Figure 5.2) when looking into the mass flow. This distribution of PFAS substances was also demonstrated in previous Swedish screening studies such as by Eriksson et al. (2015). It is important to note that several studies have shown that most PFAS precursors as well as many other PFAS, (PFAS11 not included) wind up in sewage sludge (e.g., Kärrman et al., 2021) and therefore, may contribute to PFAS contamination spread via sludge.
The same data from monthly samples in 2021 from Kungsängsverket could have large differences in the interpretation of the mass flow with regards to \( \Sigma PFAS_{11} \). Scenarios A-C in Figure 5.3 illustrate the problem which can arise if the reporting limits are not clearly defined in the analytical protocol and are interpreted in different ways. (See section 4.3 for reporting limits). The amount of PFOA and PFOS remains the same in this example as all concentrations analyzed exceeded the reporting limit. However, \( \Sigma PFAS_{11} \) included several other PFAS substances that were below the reporting limit.

Scenario A illustrates when analyses above the reporting limit were included, and the rest were set to 0 ng/L. Scenario B illustrates when the reporting limit is assumed to be the LOQ and concentrations below LOQ are set to LOQ/2. Scenario C illustrates when the reporting limit is interpreted as the LOD and concentrations below this were set to LOD. With this interpretation, a reduction of 30 % is seen.

These three scenarios give different results in reduction rates of \( \Sigma PFAS_{11} \) despite the same data used; an increase of to 95 % in effluent (negative reduction) in scenario A, whereas scenario B shows reduction of 15 %, and Scenario C reduction of 30 %.
If PFAS concentrations in effluent water to were set to 0 ng/L when they are below the reporting limit, the overall calculation of the removal of PFAS over the treatment process may be incorrect. High reporting limits also make it difficult to assess the actual treatment effect and the load on the recipient water. High reporting limits also make the assessment impossible to determine the impact on recipient waters, for example, the PFOS limit value of 0.65 ng/L for inland waters and 0.13 ng/L for coastal waters (see section 5.4). Reporting limits of around 3 ng/L for PFOS are found in several surveys, which makes it impossible to obtain relevant assessment of the environmental impact of the treatment plant. The reporting limit, whether it is interpreted as LOD or LOQ, would have less significance if the PFAS reporting limits are sufficiently low (see section 4.3). In such cases, most of the measured values in water or sludge will be higher than the reporting limit.

### 5.3 Trends for PFAS at Swedish WWTP

Hansson et al., (2016) and Haglund (2019) reported that although emissions varied between years, the emissions of 15 PFAS from WWTP effluents appeared to increase slightly between 2008 and 2013, except for PFOA and PFOS, where a downward trend was observed. For sludge, the total PFAS emissions of 18 PFAS showed a slight downward trend between 2004 and 2017, levelling off around 2008. However, for some WWTPs the trends look different. As illustrated in Figure
5.4. Henriksdal WWTP samples showed a clear downward trend except for 2013. Haglund reported similar trends (2019).

![Figure 5.4 Average emissions of PFAS via sludge in Sweden as well as for Henriksdal wastewater treatment plant during the period 2004 – 2013 (Hansson et al., 2016). PFAS levels below the detection limit were set to the LOD.](image)

There are a few WWTPs that have analyzed PFAS several times a year for several years. Kungsängsverket is one of these plants and Figure 5.5 shows PFAS concentrations in influent and effluent during the period of 2015 to 2021. Here, no clear trend for various PFAS could be seen over the years. This shows that treatment efficiency varies over the years without any specific trend being observed. Still, the WWTP reports a decrease of PFAS in the last two years in the incoming wastewater through an active management of PFAS-contamination upstream. The fact that these trends cannot be seen in Figure 5.5 could be explained by the much higher reporting limit of 100 ng/L which was used in several analyses for incoming water in 2021, mainly for PFBA and PFPeA. Using LOQ/2 for these values strongly affects ΣPFAS11, which shows the relevance of reasonable reporting limits to be used in analysis.
5.4 Recipient impact of PFAS from Swedish WWTPs

In contrast to pharmaceutical residues, which are released to recipient waters mainly via WWTPs, PFAS residues are commonly present at upstream locations of the discharge point for the treatment plant. PFAS concentrations in recipient surface water bodies, caused by discharges from WWTP, can be easily detected by increased PFAS concentrations in the recipient water downstream of the WWTP compared to upstream. Concentrations are usually lower in the recipient water than in the effluent due to a dilution effect. The dilution from the recipient water may have a too low PFAS concentration required for detection (limit value according to HVMFS 2019:25), whereas the effluent water represents a larger proportion of the total waterflow.

Figure 5.6 depicts water bodies which are affected by WWTP according to VISS Vatteninformations System Sverige (Water Information System Sweden, 2022) and those that do not achieve good ecological status compared the PFOS limit value. The figure also shows that many water bodies lack a PFOS classification. It should also be noted that VISS only included analyses from environmental monitoring that were also registered within VISS, therefore the actual status may be better than what can be seen on the map. Some of these analyses were carried out on biota (e.g., PFOS levels in fish tissue), and as such the status assessment is not necessarily based on values in water.
When looking at current projects, which include some form of recipient impact assessment with respect to PFAS emissions from wastewater treatment plants, there is a challenge to keep the concentrations below the PFOS limit value in recipient surface water bodies (Table 5.3). However, this is not primarily due to the emissions from the WWTPs, but rather to the additions from other sources upstream of the WWTP discharge point, which cause PFOS concentrations to be near or above the limit value.
However, even if analyses from the receiving water are not available, analytical data for PFAS in WWTPs can be used for an initial impact assessment. Table 5.2 shows that many WWTPs have PFOS concentrations in incoming wastewater of >2 ng/L and a negative reduction over the treatment process (i.e., increase in PFOS concentrations). This means that the effluent has PFOS concentrations exceeding the limit value for PFOS in inland waters (0.65 ng/L) by several 100 %. This in turn indicates that a dilution of at least the same order of magnitude is required in order not to exceed the limit value in the receiving water. In cases whereby there are already elevated PFOS concentrations upstream of the discharge point, an even higher dilution is required. WWTPs with a low dilution (<10 times) are therefore generally at risk of adversely affecting the recipient water. As such, PFAS mapping is recommended for these WWTPs to investigate the situation.

Figure 5.7 shows the PFOS measurements carried out by different measurement campaigns listed in Table 5.1 and Table 5.2. Most measurements were done regionally with an uneven data distribution. PFOS concentrations exceed the limit value in 146 inland water samples and 20 coastal water samples (Figure 5.7). Out of 270 measurements, 62 of them (about 1/5) measured PFOS concentration were below the limit value. However, actual concentrations for many of these points were close to the limit value (see the interactive map for more info on each monitoring point). For 42 of the measurements (about 1/6), it was not possible to

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Table 5.3 Recipient impact studies for PFAS at Swedish WWTP

<table>
<thead>
<tr>
<th>Year and reference</th>
<th>Description and comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>2022 NSVA (Contact)</td>
<td>Mapping of PFAS11 and other micropollutants at 13 of NSVA’s WWTP (Torekov, Perstorp, Oderljunga, Örkeljunga, Skånes Fagerhult, Ekebro, Ekeby, Nyvängsverket, Kågeröd, Lundåkraverket, Röstånga, Svalöv and Öresundsvetket) and associated recipients based on four sampling occasions during 2021 – 2022 (ongoing). The reporting limit for PFOS is generally lower than the limit value.</td>
</tr>
<tr>
<td>2021 – 2022 Kungsångsverket, Uppsala Vatten (Contact)</td>
<td>Recipient sampling that takes place once a month for a year. Analysis of PFAS11 and pharmaceutical residues in three points upstream and downstream respectively of the Kungsångsverket. In addition, an extra measuring station directly upstream of the discharge point from the Kungsångsverket, to investigate any impact of PFAS from stormwater in this position.</td>
</tr>
<tr>
<td>2021 Getteröverket, Varberg (Baresel et al., 2021)</td>
<td>Three sampling campaigns (w.37, 2020; w.15 and w.29, 2021) in seven points in surface water in connection with ARV incl. upstream. Aggregate sample based on several random samples taken at the same time. General low dilution of outgoing wastewater. The PFOS limit value is exceeded in all recipient samples and upstream samples. Reporting limit for PFOS is lower than the limit value.</td>
</tr>
<tr>
<td>2020 Stadskvarns WWTP, Skövde (Baresel 2021)</td>
<td>Four sampling campaigns (w.10, w.25, w.35 and w.50) at five points in watercourses in connection with WWTP incl. upstream. Aggregate sample based on several random samples taken over a week. General low dilution of outgoing wastewater, which is expected to decrease in the future in line with increased capacity. The PFOS limit value is exceeded in all recipient samples and upstream samples. Reporting limit for PFOS is lower than the limit value.</td>
</tr>
<tr>
<td>2019 Främby WWTP, Falun (Hedén et al., 2020)</td>
<td>Four sampling campaigns (w.21, w.24, w.29 and w.37) at two points in the surface water in connection with ARV incl. upstream. Sample. The contribution from ARV regarding PFOS is calculated to be about 10 % in the recipient. However, the measured concentrations in the receiver were generally higher than those calculated based on upstream concentrations and concentrations in the outgoing water, which indicates several PFAS sources. Reporting limit for PFOS is lower than the limit value.</td>
</tr>
</tbody>
</table>
assess whether the limit value was exceeded or not, as the LOD or LOQ was above the limit value. This means that data for 1/6 of all measurement points are meaningless, which could have been avoided by careful selection of analytical laboratories with sufficiently low LOD and LOQ before procurement.

Figure 5.7 Reported PFOS levels in surface water bodies according to the mappings shown in Table 5.1 and Table 5.2 as well as ongoing preliminary studies from Naturvårdsverket, Swedish EPA. The number in brackets indicates the number of measurement points for each grouping (Link to interactive map).
6 Possible measures of action

Simply reducing or stopping future PFAS emissions is not enough to protect humans and the environment from PFAS-related adverse effects. Because of its persistent property and its historical emissions, PFAS will remain in the environment unless additional measures are taken to remove or until their usage are banned. This may also lead to an increase in background levels of PFAS in parallel with what levels are considered acceptable in the environment, drinking water and food decrease. The longer PFAS remediation measures are postponed, the more challenging the work will be to remove PFAS. PFAS can bioaccumulate and spread in soil over time. To remove PFAS in larger masses and area will be more technically complex and costly. If the problem is not dealt with in a correct and timely manner, there is a high risk that large areas of land, many water bodies, and groundwater will become unusable for a long period.

An overall measurement priority with the waste hierarchy can be used to highlight a basis for work on PFAS (Figure 6.1). To adequately protect humans and the environment from the adverse effects of PFAS, a combination of prevention and remediation is needed. These can include but are not limited to preventive measures to avoid the release of new PFAS and treatment/remediation technologies that are both economically and environmentally sustainable to manage existing PFAS sources, pathways, and contaminations.

As shown in Figure 6.1, “Protection” (e.g., discouraging the consumption of fish from certain water bodies due to their excessive PFAS content) is considered the last and most undesirable measure in the action plan prioritization of PFAS removal. Moreover, only the destruction or inactivation of PFAS can remove PFAS from the cycle. Many remediation and treatment technologies do not guarantee PFAS destruction, however, they can be separated and concentrated from one contaminated matrix (e.g., wastewater) to another (e.g., Granulated activated carbon; GAC) so that the separated PFAS can then be sent for destruction. This is where WWTPs can contribute to PFAS remediation and treatment. Through effective upstream work (“Decontamination” in the diagram) and remediation (“Treatment” in the diagram) by containing and removing the existing PFAS from the water cycle through concentration and destruction.
6.1 Building and disseminating PFAS knowledge

All measures to remove PFAS from the natural environment, or at least reduce PFAS-related health and environmental risks, rely on the knowledge of PFAS and its management to be available and disseminated to all stakeholders who need to be involved in its removal plan. This can be achieved through various networks and collaborations at different levels and across different sectors of society. For example, the Swedish government’s coordination group (SamTox) with seven agency heads work together to detect and address new and potential chemical threats at an early stage, such as their work with PFAS. Another example is Swedish state PFAS network with 12 participating authorities who meet regularly to exchange information and updates.

Acquiring PFAS knowledge and its subsequent dissemination can lead to, for example:

- Improved understanding of PFAS toxicity and persistence to properly prioritize remediation efforts.
- The sharing of measurement data and other knowledge on PFAS, including from wastewater, sludge and in recipient waters.
- Improved information on PFAS content in goods and products.
Improved and transparent documentation from fire training sites including from defense and military facilities.

- Improved knowledge of the degradation and transformation products of PFAS.
- Improved understanding of how to remove PFAS from contaminated environmental matrices.
- An increase in knowledge exchanges regarding PFAS testing and investigations of treatment techniques and sludge management.
- Improved understanding regarding the relationship between PFAS and aspects in occupational health and safety.

6.2 Preventive measures

To completely remove PFAS from the natural environment, a ban on PFAS is necessary both in PFAS in products and use of PFAS. This would prevent the imports of PFAS, which in turn would reduce PFAS emissions into the society and the environment. However, since PFAS are also present in products produced abroad and imported to Sweden as well as in the atmosphere (Figure 3.1), a phase-out of PFAS at an international level is also required to eventually overcome the problem (see section 3.2).

6.3 Remediation of contaminated land

Today, there are two different techniques which are commonly used for soil remediation, where excavation and landfill are not an option. The first option is to use high temperature incineration for the solid phase (i.e., minerogenic and organogenic particles). The second option is an in-situ treatment of the water phase, either in groundwater pipes or in a treatment device, where PFAS are “washed” into a water phase which is subsequently treated. Recently, several other methods have been presented for the remediation of PFAS contamination in various environmental matrices (e.g., Niarchos et al., 2022; Sörengård et al., 2019; 2021). In general, measures at or near the source of contamination are more resource-efficient compared to actions in more dilute matrices such as wastewater. With an increased understanding regarding the complex nature of the PFAS problem, both the requirements for remediation techniques (e.g., removing PFAS from contaminated environmental matrices or understanding PFAS toxicity) and comprehensive screening (e.g., identification of degradation and/or transformation PFAS products) have improved. Improved understanding of PFAS toxicity is also critical to properly prioritize remediation efforts.
Phytoremediation is a method that involves cleaning or stabilizing the contaminated areas by planting certain plant species. The two main mechanisms of phytoremediation are contaminant stabilization in the soil, thus limiting leakage and spread, and pollutant uptake and concentration by the plant biomass, which can then be disposed of as hazardous waste. However, it is mainly the second mechanism which has been discussed as a possible point for PFAS remediation, as plants are more likely to uptake short-chained PFAS compared to long-chained PFAS. For example, bioconcentration factors (the difference in concentration between biomass and water) were evaluated for several plants regarding their uptake of PFPeA and PFOS (concentration by 175-fold in biomass respective 11-fold; Huff et al., 2020). Uptake of PFAS by different plant species grown in a contaminated area at Arlanda Airport has been evaluated by Ahrens et al. (2015). Experiments with different plants from leachate have also been performed at Nordvästra Skånes Rennhållnings AB in Helsingborg (Sorelius and Svensson 2021).

6.4 Treatment techniques for PFAS-contaminated water

PFAS are considered as organic micropollutants and some of the methods used for micropollutant removal such as pharmaceutical residue can be applied for PFAS removal. A detailed description of this and other methods used for micropollutant reduction can be found in various publications such as in Baresel et al. (2017a, b). Further specific methods for PFAS reduction in waterworks are described in Franke et al. (2017). This chapter section is a brief technical review including more detailed descriptions for techniques which are not used for micropollutants (e.g., foam fractionation).

Figure 6.2 provides a schematic overview of treatment processes and technologies available for the separation and destruction of PFAS in wastewater and sludge, which could be implemented by WWTPs. The figure also indicates whether the technologies are ready for broad implementation or whether technological development is ongoing or necessary. It should be noted that there are currently no references for the full-scale application of these techniques intended for PFAS treatment at Swedish wastewater treatment plants. Even if a technology is considered ready for broad implementation, it does not necessarily mean that there are or will be full-scale reference facilities available, since PFAS treatment has yet to be a focus at facilities such as WWTPs (see section 6.7). Although these technologies shown in the Figure 6.2 are mostly unproven in municipal wastewater treatment, they are often established in other areas, such as drinking water.
production, which can then be used as a basis for application assessment. A general aspect associated to the implementation of different technologies is that resource consumption and treatment efficiency will be lower for low PFAS concentrations per volume treated (e.g., in effluent water from WWTPs) compared to those with higher PFAS concentrations (e.g., leachate). The figure includes also possible residual streams containing PFAS which needs to be managed.

<table>
<thead>
<tr>
<th>Purification process</th>
<th>Purification technology</th>
<th>PFAS residual product (1)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Membrane separation</td>
<td>Reverse Osmosis (RO)</td>
<td>Retentate (10–30 %)</td>
</tr>
<tr>
<td></td>
<td>Nanofiltration (NF)</td>
<td>Retentate (10–30 %)</td>
</tr>
<tr>
<td>Adsorptive techniques</td>
<td>Activated carbon</td>
<td>GAK/PAK-sediment slurry</td>
</tr>
<tr>
<td></td>
<td>Ion Exchanger (ADX)</td>
<td>The ion exchange mass</td>
</tr>
<tr>
<td>Other techniques</td>
<td>Foam fractionation</td>
<td>Foam/sediment</td>
</tr>
<tr>
<td>Chemical destruction</td>
<td>Advanced Reducing Processes (ARP)</td>
<td>By-products</td>
</tr>
<tr>
<td></td>
<td>Ultrasound/cavitation</td>
<td>By-products</td>
</tr>
<tr>
<td></td>
<td>Electrochemical treatment</td>
<td>By-products</td>
</tr>
<tr>
<td>Thermal destruction</td>
<td>Combustion</td>
<td>No by-products (2)</td>
</tr>
<tr>
<td></td>
<td>Supercritical wet oxidation (VO)</td>
<td>No by-products (2)</td>
</tr>
<tr>
<td></td>
<td>Pyrolysis</td>
<td>Possible by-products</td>
</tr>
</tbody>
</table>

Legend: Standard technology | Technology development

Figure 6.2 Schematic view of existing treatment technologies for wastewater and sludge, technology development and any residual products and residual streams that contain PFAS that need to be managed.

### 6.4.1 Membrane separation with reverse osmosis and nanofiltration

Membrane treatment and membrane separation, either with reverse osmosis (RO) or nanofiltration (NF), are used in both industrial processes and drinking water production. The processes rely on the use of semi-permeable membranes through which water can pass, but not dissolved molecules or ions. Through osmosis, the water strives to equalize the concentration on both sides of the membrane. By applying an external pressure higher than the osmotic pressure, the flow can be
reversed so that the water leaves the more concentrated stream. The higher the concentration of solutes, the higher the pressure required. In theory, RO membranes do not allow salts and other dissolved substances to pass through, only water. In practice, on the other hand, some small polar molecules, ions, and/or dissolved gases can pass through the membrane. However, all PFAS are rejected by RO membranes. NF membranes are less dense than RO membranes, therefore NF membranes allow more small ions and molecules to pass through. NF membranes are characterized both by the degree of salt removal, usually NaCl or MgSO4, and by the size of molecules (expressed in units of Da) that is removed with 90%. For PFAS11, this means that over 99% of the compounds are separated with an RO membrane or a denser NF membrane (e.g., 90 Da, whereby 1 Da corresponds to a molecule size with an equivalent molecular weight of 1 g/mol). For thinner NF membranes (e.g., 270 Da), the removal of PFAS11 is at least 80 – 90% (Franke et al., 2019; 2021).

It is the molecular weight characterization that is most useful in assessing whether a membrane is suitable for the reduction of PFAS in PFAS-contaminated water. The size of the membrane should be less than 300 Da since the most common PFAS included in PFAS11 have a molecular weight of 214 – 514 Da. NF membranes that can effectively remove molecules with a weight of 90 Da would have a very high removal efficiency of all PFAS as well as salt (although to a lesser degree than an RO membrane would). Factors such as electrical charge and hydrophobic/hydrophilic properties would also affect removal. NF membranes with a higher cut off (molecular weight for the highest molecule which can pass the membranes) at 300 Da would, at least in theory, result in an increased removal of long-chained PFAS as well as a moderate removal of short-chained PFAS; however, it would allow most of the salt in the water to pass through. Thin membranes require less energy, however, the thinner the membrane is, the amount of rejected PFAS is lower.

The choice between RO and various dense NF membranes also depends on the use of the filtered water and the incoming salinity levels. RO membranes or dense NF membranes are more suitable for incoming water with high salinity (e.g., wastewater from coastal areas with high seawater inflow) and for filtered water which is to be reused for drinking water production or irrigation. Thinner NF membranes may be a more resource-efficient option where filtered water is reused as technical water (e.g., for use in machineries or cooling without any desired quality) or where the salinity of the incoming water is low. This would also apply for the treatment of high salinity water without a reuse purpose such as with
leachate. With this, the retentate (i.e., the concentrate that is retained) does not have a high salt concentration, which subsequently allows for less energy needed to force the water through the membrane.

In advanced membrane separation such as NF and RO, a concentrate stream is generated that contains all the impurities and salts removed from the treated water. The proportion of water passing through the membrane (recovery rate) and the corresponding volume of concentrate stream are determined based on the amount of fouling (the sorption of organic matter and biological growth) and scaling (the precipitation of salts) which can be allowed on the membrane surface. The risk of these problems becomes higher as the concentration degrees rise. The recovery rate is typically 70 – 90 % (retentate 10 – 30 % in Figure 6.2) and depends on the characteristics of the incoming water, types and availability of pre-treatments, and treatment targets. Thinner NF membranes have the potential to be reused compared to RO membranes since they allow for more ions pass through. When using RO or NF, a large volume of concentrated water (otherwise known as the concentrate) is formed, and this must be managed. As such, membrane filtration can be used to concentrate PFAS, however it cannot degrade it. For leachate treatment, the concentrate can theoretically be returned to the landfill. But this requires risk considerations and operating permits.

When NF or RO membranes are used for wastewater concentration, the process must be supplemented with a treatment for the concentrate, (e.g., carbon filters). With higher pollutant levels, the combination of NF/RO membranes and activated carbon may be beneficial and justified.

Advantages of advanced membrane separation for PFAS treatment include:

▪ All organic and inorganic pollutants larger than the membrane pores are removed. This includes, PFAS, pharmaceutical residues, pesticides, solvents, metals, arsenic, dissolved organic carbon and nutrients.
▪ Easy application on small and large scales.
▪ Easy monitoring and control of the process (e.g., control of fouling and removal efficiency).
▪ Stable and continuous removal (no decreasing removal efficiency or breakthrough as with sorption techniques).
▪ Production of reusable clean water.
Disadvantages of advanced membrane separation for PFAS treatment include:

- Generates a concentrate stream (10 – 30 % of the feed) containing PFAS which must be treated (e.g., with activated carbon).
- Pre-treatment is required to reduce the risk for fouling, required membrane cleaning frequencies and energy consumption.
- High energy consumption depending on the density of membranes and water matrix (e.g., RO with ultrafiltration (UF) as pre-treatment (UF/RO) requires about 0.3 – 0.8 kWh/m³).
- Generally high operational cost (about 5 – 10 SEK/m³).
- Not commonly used in the current municipal wastewater.

6.4.2 Adsorption with activated carbon

Activated carbon is commonly used to adsorb water contaminants (e.g., synthetic organic chemicals, natural organic compounds and other compounds that affect the taste and odor of drinking water) and has been used in Swedish drinking water plants for many decades. Adsorption is a physical chemical process where mass (atoms) is moved to a surface. Activated carbon, used either in granular form (GAC; often used for filters) or powder form (PAC; often added to the water phase), is an effective adsorbent for certain substances due to its highly porous nature and large surface area on which contaminants can adhere. Activated carbon is usually produced from fossil coal or renewable raw materials, such as coconut shell or wood. Yu et al. (2014) showed that adding PAC to a membrane bioreactor (MBR) could remove approximately 90 % of the PFCA tested at a dose of 100 mg PAC/L. However, depending on where in the process PAC is added, PAC dosing can result in a higher proportion of PFAS and other micropollutants being transferred to the sludge. Because of these micropollutant transfers, PAC has limited potential in Sweden as long as sludge is recycled to agricultural land etc.

Long-chained PFAS have been shown to be more efficiently removed when using activated carbon for PFAS treatment. The presence of charged groups, such as carboxyl and/or sulphone groups, also plays a role in the activated carbon reduction efficacy. A preliminary assessment has shown that activated carbon adsorbs poorly with carboxylic acids with a carbon chain shorter than eight (C8), and sulphonic acids with a carbon chain shorter than six (C6). This means that substances falling into this category will break through a carbon filter. For example, PFAS reduction would reduce to less than 80 % before 5,000 bed volumes (BV; filter volumes) treated. Although based on a preliminary assessment, it can be assumed that PFAS substances such as PFOS would break through a carbon filter much earlier than most pharmaceutical residues (see section 6.7).
Advantages of activated carbon adsorption for PFAS treatment include:

- Adsorbs not only PFAS but also other micropollutants such as pharmaceutical residues, pesticides, solvents, etc.
- The technology is scalable.
- Basic technical know-how is available at the level of the wastewater operators as the technology is based on known methods such as sand filters or dosing of coagulants.
- The possibility to destroy PFAS by thermal regeneration of carbon at high temperatures (GAC).
- PAC dosing, if chosen, can even be integrated into the existing treatment process, depending on the current process configuration.

Disadvantages of activated carbon adsorption for PFAS treatment include:

- Activated carbon has different characteristics and it is important to choose a carbon that is suitable for the intended application.
- Reduced treatment efficiency over time and rapid saturation for short chain PFAS such as PFBA (better performance for longer chains like PFNA and PFDA).
- Lower average adsorption capacity for PFAS (approximately 5,000BV) compared to most pharmaceutical residues (>20,000 BV) which implies a higher operational cost for replacing filter material.
- Extensive biological treatments and effective particle removal step prior to the adsorption process are necessary for stable operation (reduced risk of fouling but also less competition for adsorption surface).
- No easy indication of saturation. Periodic analyses of target substances are required.
- With the option of PAC dosing, EX classification (hazardous area) is required, and the destruction of sludge containing PAC, resulting in high operating costs.

6.4.3 Ion exchange (Advanced Ion Exchange - AIX)

Ion exchange (Advanced Ion Exchange; AIX) involves the use of resins, which are small porous plastic beads with charged functional groups, to treat leachate and in drinking water productions today. To each functional group, a charged impurity can bind and replace existing hydrogen cations (H+) or hydroxide ions (OH-). The reduction rate and efficiency of AIX depends on the initial concentration of the pollutant, the concentration of competing ions and solutes, the treatment design (i.e., flow rate and size of the beads) and the type of ions in the beads. Since most PFAS exist in water as dissolved anions (minus charged ions), anion exchangers are used for the treatment of PFAS-contaminated water. However, the ion
exchangers used for PFAS reduction does not remove all anions, but only specific ones. Ion exchange resins are more selective than GAC. Sorption performance, however, follows to some extent the same pattern observed for GAC - the shorter the PFAS carbon chain, the poorer the sorption and faster breakthrough. There is a large difference in the breakthrough of short and long-chained PFAS. For example, the treatment of PFBA could drop to 0 % after only a few thousand BV, while removal of PFAS could exceed 90 % remove rate at up to 34,000 BV (Malovanny et al., 2021).

As with activated carbon, there are many types of anion exchangers from different suppliers. Some ion exchange resins are marketed specifically designed for the treatment of PFAS-contaminated water (Purolite PFA694E, Lewatit TP108, etc.) while other suppliers recommend their resins for the removal of perchlorates (commercially produced salts which contain perchlorate ions) as well as for the treatment of PFAS-contaminated water (Amberlite PSR 2 Plus etc.). Like activated carbon, ion exchange resins can be used in filters. However, unlike their counterparts, ion exchange resins can result in a greater pressure drop across the filter and are required to shorten the contact time. Therefore, pressurized filters with high flow rate could be a better alternative compared to open filters.

**Advantages** of ion exchangers for PFAS treatment include:

- Replaces not only PFAS ions but also partially other organic pollutants present in the water in anionic form (e.g., certain pharmaceutical residues such as diclofenac).
- Relatively easy to apply on small (even disposable), medium and large scales.
- More water to be treated per filter media unit (factor 5 - 10 in leachate trials) due to higher and selective sorption capacity for PFAS in comparison to GAC.
- Destruction of PFAS with high temperature incineration of used resin.

**Disadvantages** of ion exchangers for PFAS treatment include:

- More effective pre-treatment to prevent clogging is required compared to those required for carbon filters. Often used as post-polishing.
- More difficult operation compared to GAC filters, especially backwashing when clogged.
- No easy indication of saturation. Periodic analyses of target substances required.
- Complicated regeneration involving flammable chemicals. Eluate material must be destroyed.
Untested in municipal wastewater treatment today (although it is an established technology in other areas such as drinking water production)

### 6.4.4 Surface Activation Foam Fractionation

Surface Activation Foam Fractionation separates PFAS from water by bubbling air through water to which PFAS can adhere to. At the water surface, a foam is formed which can be separated either by leading away the foam or via suction. The foam fractionation technique (as in the case of protein skimmers) has been used in water treatment for a long time to remove proteins, such as feed residues and waste from water in fish farms. More recently, the technique has been recognized for its ability to also remove PFAS. However, references for large-scale usage in municipal WWTP still do not exist.

PFAS can be separated with air bubbles because their molecules are polar, which means that the different ends of the molecules have different charges: one end hydrophobic (water-repellent) and the other hydrophilic (water-soluble). Also, PFAS molecules are surfactants, which means that they want to be at the interface between water and organic substances or solvents. When air bubbles are present in the water, the PFAS molecules position themselves so that their hydrophobic part (the fluorinated carbon chain) come into contact with the air bubbles, whereas the hydrophilic part (the SO3- and COO- groups) remain in the water phase (Figure 6.3). When the air bubble rises to the surface, the molecule follows. PFAS can then be separated by collecting this formed foam (air bubbles with a water film on top). For foam fractionation atmosphere air can be used, as well as ozone, which can create smaller bubbles.

![Figure 6.3 The principle of PFAS separation with air during foam fractionation.](image-url)
Foam fractionation is also a separation process where PFAS can be concentrated into a smaller volume of concentrate, which can then be handled. Typical concentration factors for foam fractionation are between 10 – 20 times, which means that the concentrate volume is about 5 – 10% of the incoming flow (OPEC Systems 2020; Sjöberg 2021). To further concentrate PFAS, foam fractionation can be done in several steps. OPEC Systems patented a multi-stage foam fractionation process (Phillips 2019). In the first stage, traditional foam fractionation is used where air is injected with injectors. In stage 2, foam fractionation is performed on the concentrate from the first stage. Then negative pressure is used to create "dry foam", which is drawn away as concentrate. Optionally, a third step can be used to treat the concentrate from step two. A two-stage foam fractionation practice allows for PFAS to be concentrated about 1000 times (OPEC Systems 2020). In a three-step foam fractionation, the volume can be further reduced by a factor of 100, resulting in only a very small volume of concentrate that needs to be sent for destruction (OPEC Systems 2020, Sjöberg 2021).

Foam fractionation separates different PFAS with different efficiencies. In general, PFAS with six or more fully fluorinated carbon atoms are almost completely removed. PFAS with four or fewer fully fluorinated atoms, on the other hand, are more challenging. Their low surface activity decreases their tendency to stay at the surface layer between the water and the air. For PFAS11 this means that 4 out of 11 substances are only separated to some extent (0 – 30%) by foam fractionation, while the remaining 7 out of 11 substances are separated by more than 90%. For foam fractionation to work, the water should tend to foam moderately. With treating water that does not foam naturally (e.g., groundwater with relatively low PFAS concentrations), surfactants are added to enhance foaming. When treating some leachates, foaming may be too strong. This results in a low concentration of PFAS in the foam, as more water follows with the foam. This can be solved either by filling the foam fractionation column less (higher freeboard level, space above the water surface) or by adding a small amount of defoamer (Malovanyy 2021).

Advantages of foam fractionation for PFAS treatment include:

- Simple and scalable technology (based on already tested applications with leachate).
- Very high and stable reduction of long chain PFAS.
- High concentration of PFAS can be achieved with several steps.
- The potentials to combine aeration in biological treatment steps with PFAS reduction.
- Addition of chemicals is required only at certain cases.
Disadvantages of foam fractionation for PFAS treatment include:

- Only partial purification from PFAS.
- It is not easy to control to obtain an appropriate foaming.
- The technique generates a residual fraction that needs to be managed (e.g., incineration).
- The technique is not established at WWTPs today. Few full-scale plants exist (leachate treatment and groundwater treatment with design flows of max. 15 m³/h).

6.4.5 Other treatment and complementary technologies under development

In addition to the treatment techniques described above, there are several techniques under development that can be used separately or supplementary to other techniques. For some of these techniques, their effectiveness in separating or destroying PFAS (including different types of PFAS) remains to be proven. Techniques that can be mentioned include:

- **Electrosorption**
  In electrosorption, an electrode is used to increase the movement of the ionic compounds, which can then be adsorbed using materials like activated carbon. Capacitive deionization (CDI) is one of the techniques involved here. The technique has not yet been tested at full scale and regenerating the electrodes presents challenges.

- **Electrocoagulation**
  Electrocoagulation releases metal ions from metallic electrodes that flocculate particles, mostly likely includes bound PFAS. Also, electrochemical oxidation can take place. Several experiments with highly PFAS concentrated water show significant PFAS reduction, but electrocoagulation can especially be suitable as a pre-treatment of the water before applying other techniques.

- **Electrochemical oxidation**
  Electrolysis occurring between two electrically charged electrodes can provide some degradation of PFAS through the formation of radicals in the process. However, the distances between the electrodes must be small, electrolytes may need to be added to increase conductivity and PFAS intermediates may be formed.

- **Complex formation and flocculation**
  PFAS are not separated when typical flocculants commonly used in WWTPs are added. However, there are special flocculants that bind PFAS into larger
complexes in dissolved form or flocs in solid form that can then be separated by mechanical filtration. Examples of such techniques are Chromafora Selpaxt (removal by ultrafiltration) and PerfluorAd (removal by sand filter).

- **Alternative sorbents**
  Instead of commercial activated carbon, adsorbents made from other materials or modified adsorbents can be used. Examples are activated carbon from sludge and magnetic activated carbon (expected easier removal with magnetic activated carbon). Different technologies are being developed and some of the most relevant are discussed in the next section (section 6.5).

- **Chemical destruction**
  Advanced reduction processes (ARP), electrochemical treatment and sonolysis (see section 6.5) can also be applied for the treatment of PFAS-contaminated water. ARP techniques such as electron beam, photolysis and ultraviolet (UV) radiation create reducing radicals which then sequentially defluorinate PFAS. Sonolysis and other techniques that create radicals and high temperatures through cavitation bubbles may also be able to destroy PFAS.

- **Fluorine coating for PFAS separation**
  Initial laboratory scale tests have shown that synthesized fluorine coatings or films can be used to capture PFAS from contaminated water. The separated matrix (film with captured PFAS) can then be thermally treated to destroy PFAS and to regenerate the coating (Sloand et al., 2021). However, this technology is still under development and is only available on a laboratory scale.

- **Technology combinations**
  Although some techniques, such as ozonation, are not suitable for PFAS removal (due to the strong carbon-fluorine bonds that remove the reactivity of ozone (Choe et al., 2020; von Gunten, 2003)), some combinations of techniques can achieve some PFAS treatment. One example is catalytic ozonation where activated persulfate or fenton is added to the wastewater (Arias Espana et al., 2015). However, such technology combinations require testing and verification before they can be applied at full scale.

Ozonation alone does not provide oxidation of PFAS due to the strong carbon-fluorine bonds that remove the reactivity of ozone (Choe et al., 2020; von Gunten 2003).

### 6.5 Destruction or inactivation of PFAS

Destruction or mineralization refers to the complete defluorination of the perfluoroalkyl chain and the PFAS molecule. Some treatment techniques for PFAS-
contaminated water discussed in the previous section only achieve a separation and concentration of PFAS without destroying the contaminants. If separated PFAS are not destroyed, there is a risk that they will re-enter the natural environment, for example via leachate. However, some treatment techniques for PFAS-contaminated water can also achieve complete or partial destruction of PFAS. These techniques and related aspects are therefore also addressed in this section. When looking into technologies for destroying PFAS and their precursors, it is important not to overlook the aspect of any undesirable degradation or by-products that may be formed from incomplete destruction. Whether PFAS-related degradation or by-products are created during treatment by different techniques has yet to be investigated to the same extent as, for example, in pharmaceutical treatment. This means that there is a risk that PFAS-related degradation or by-products that cannot be analyzed are generated during incomplete PFAS destruction. The environmental impact of possible PFAS degradation products has not been quantified but given their profile it is likely that they too are persistent and toxic.

The following description of PFAS destruction techniques should therefore be taken with some reservation, as further verification studies for each technique should be carried out before the technique can be considered as PFAS destruction method. Generation of PFAS-related degradation or by-products, and their treatment requirements, for example residues or streams, may prove to be unavoidable. However, an awareness of potential by-products may help to identify appropriate management of residual streams.

All destruction technologies are based on two basic mechanisms involving either molecular degradation via thermal destruction or sequential defluorination (Horst et al., 2020). The stability of the perfluoroalkyl group also means that alternative thermal destruction techniques to incineration require comparatively longer retention times to achieve mineralization, but other factors such as temperatures and the presence of oxygen are also significant.

### 6.5.1 Thermal treatment: incineration, pyrolysis, thermal hydrolysis

Incineration is considered to provide 99 % destruction of fluorinated organic toxic substances (Principal Organic Hazardous Constituents - POHCs) after only 2 seconds retention time at 900 °C (Taylor et al., 1990). For unimolecular thermal destruction where only the PFAS molecule participates in the reaction, 1440 °C for at least 1 second is estimated to be required (Tsang et al., 1998). A study by Kim Lazcano et al. (2019) showed that heating sludge to 650 °C is not effective for the
removal of PFAA. One study also showed that the required temperature for PFAS destruction increases with increasing chain length (Rayne and Forest, 2009). Solo-Gabriele et al. (2020) studied PFAS in different leachates from landfills where incinerator ash was placed. A statistically significant correlation between incineration temperature and the concentration of PFAS in the leachate from the landfilled ash was observed. The lowest PFAS concentrations were found in the leachate of ash from incineration at 930 and 980 °C. For incineration at lower temperatures, 760 to 870 °C, higher concentrations of various PFAS were found in the leachate. In addition, the results indicated that incineration at lower temperatures may produce shorter PFAS as degradation product instead of destroying PFAS.

It should be noted that complete PFAS degradation resulted in the production of hydrogen fluoride which is toxic and corrosive. Temperatures below 900 °C are likely to be insufficient for complete degradation of PFAS, although there are conflicting data in the literature. The contradictions may be due in part to differences in the material incinerated (e.g., solid or water phase) or to incomplete analysis of potential PFAS by-products (Horst et al., 2020). Complete degradation of PFAS was reported at incineration above 900 °C (Watanabe, 2016; 2018). However, quantifiable levels of PFAS in ash and flue gas condensate at Swedish incinerators suggested that complete oxidation does not necessarily occur at 900 °C and that various factors such as contact time, matrix, incineration technique, etc... affect final PFAS concentrations (Strandberg et al., 2021). In the study of Strandberg et al. (2021), it could not be demonstrated that high temperature incineration (>900 °C) resulted in a better destruction of PFAS compared to incineration at lower temperatures in conventional waste boilers (<900 °C).

Sludge incineration is a common method of sludge management in parts of Europe; in Netherlands and Switzerland, all sludge produced is incinerated, while in Germany only 55 % of sludge is incinerated (Fang et al., 2020). Sludge is usually incinerated in separate plants or together with waste or biofuels. Incineration as a destruction method is also relevant for the disposal of used activated carbon, ion exchange resins or other solid sorbents after PFAS remediation/treatment. Liquids, such as concentrates from foam fractionation or solutions from regeneration of ion exchange resins, can also be incinerated, but only in specific facilities capable of receiving and handling liquid waste.

EU Regulation 2019/1021 on persistent organic pollutants sets a 50 mg PFOS/kg waste limit for when waste containing PFOS must be destroyed or irreversibly
transformed so that the remaining waste or emissions do not exhibit persistent properties. Waste with concentrations higher than the set limit, must be destroyed. Whether incineration in conventional waste boilers for non-hazardous waste meets the destruction requirement is debatable. Estimated PFOS content in used GAC from leachate treatment is much lower than 50 mg PFOS/kg (Malovanyy et al., 2021), thus there are no legal barriers to incinerate the used activated carbon with other non-hazardous organic waste. The PFOS content in used ion exchange resins from leachate treatment is also usually lower than the 50 mg PFOS/kg limit, although the level may exceed the limit if leachates with high PFOS concentrations have been treated (Malovanyy et al., 2021). Used GAC or ion exchanger from municipal water treatment should also have concentrations lower than 50 mg PFOS/kg, as PFOS concentrations in municipal wastewater are much lower than in leachate. However, the fact that there are no legal barriers to incineration of PFAS-containing sorbent with non-hazardous waste should not be taken as an indication that such management is appropriate, given the uncertainties about incomplete degradation at low incineration temperatures as discussed above.

A special application of incineration is the reactivation of used GACs. The process is based on the partial oxidation of the carbon and the release of accumulated impurities from the material (desorption) at a temperature of about 900 °C in an oxygen deficient environment (pyrolysis). One of the major active carbon suppliers (Chemviron Carbon) asserts that PFAS are completely destroyed during reactivation (Calgon Carbon 2017). This claim, however, is based on analogies with other thermal destruction processes where similar temperatures are used. To the knowledge of the report authors, this assumption has not yet been confirmed by actual measurements in the current reactivation plant. Sending GAC for reactivation is a common method for handling activated carbon in drinking water treatment. For example, Uppsala Vatten och Avfall sends their activated carbon for PFAS reduction in drinking water production to Bäcklösa waterworks, a reactivation facility.

Pyrolysis, or anaerobic thermal treatment at high temperatures (300 – 900 °C), may also destroy PFAS, at least partially. There has been an increased interest in using pyrolysis for sewage sludge in Sweden to produce biochar and to harness its energy, but also to reduce the levels of micropollutants in the sludge. Thoma et al. (2022) investigated 41 PFAS in sludge and biochar produced in a commercial pyrolysis plant, however they could not detect any PFAS compounds in the biochar. In a laboratory study by Alinezhad et al. (2022), over 99 % of PFOS and PFOA were removed by pyrolysis of PFAS-contaminated soil at 400 – 500 °C for 30
minutes. However, the study also showed that the process produced fluorinated volatiles and non-polar degradation products of PFOA and PFOS, which was also reported in another study (Kundu et al., 2021). The planned test bed for sludge pyrolysis in Ellinge in Sweden (Sweden Water research, 2022) will hopefully provide more knowledge in the future. Hydrothermal treatment processes, including Hydrothermal Carbonization (HTC), involve treating sludge in an oxygen-free environment at high pressures and temperatures (up to 88 bar and 400 °C). Zhang and Liang (2021) showed that these treatment processes are not effective methods for PFAS destruction. This is explained by the thermal resistance of PFAS (stable compounds with high evaporation temperatures), and in accordance with the above discussion that temperatures below 900 °C cannot guarantee a complete PFAS destruction.

6.5.2 Chemical destruction

Chemical destruction refers to advanced reduction processes (ARP), electrochemical treatment and sonolysis. ARP techniques such as electron beam, photolysis and ultraviolet (UV) radiation create reducing radicals and thereafter sequentially defluorinate PFAS. Photolysis and UV use different UV wavelengths together with chemical reagents and catalyst materials to facilitate the formation of the radicals while electron beams use gamma radiation. ARP techniques for PFAS degradation are often energy and time consuming (several hours of retention time) and the efficiency is difficult to control, which poses a risk for PFAS by-product formation (Horst et al., 2020).

Another chemical treatment technique is electrochemical treatment based on electron transfer from an anode to PFAS. Different materials in the anode influence the efficiency. For example, boron-dipped diamond anodes seem to be more suitable than mixed metal oxide anodes due to their commercial availability, high reactivity, low adsorptivity and their ability to defluorinate many different PFAS (Horst et al., 2020; Smith et al., 2022). However, the risk of bromate formation must not be forgotten when bromide is present in the water. Another technique using electrochemical degradation is capacitive deionisation (CDI). The technique, which is mostly used for water deionisation, is based on electrochemical sorption of ions to carbon electrodes and is. A special design to the CDI technique was developed by KTH Royal Institute of Technology and Stockholm Water Technology, which allowed for the sorption of PFAS from water onto the electrodes but also to it being partially decomposed electrochemically (Carrillo et al., 2021).
Several studies have demonstrated that ultrasonic waves (sonolysis) at high frequencies (100 kHz) can destroy PFAS (Horst et al., 2020). The process requires only a moderate amount of energy for PFAS mineralization, and the risk of by-product formation is low. However, tests with leachates have shown that complex matrices with multiple contaminants can be challenging (Cheng et al., 2008). The destruction element of the process occurs when cavitation bubbles form and locally release enormous amounts of thermal energy upon their collapse; this destroys the PFAS. The cavitation also generates radicals, so that chemical destruction also occurs. Sidnell et al. (2022) compared ultrasonic waves with other destruction techniques and argue that the technique has proven to be the only treatment that completely degrades PFAS.

6.5.3 Biodegradation

Biodegradation of PFAS is generally not expected. Therefore, there are currently no accepted and practical biological mineralization techniques for PFAS, although several studies have investigated such techniques (Horst et al., 2020). For example, Huang and Jaffe (2019) were able to demonstrate promising PFOA and PFOS transformation to short-chain PFAA molecules via an autotrophic bacterium (Acidimicrobium sp. strain A6).

In a commercial application of biodegradation, the same aspects exist as for the use of chemical oxidants; PFAS precursors can be converted to PFAS, which are more easily spread. For example, Backe et al. (2013) showed that cationic and zwitterionic PFAS in some fire foams can be converted to anionic PFAA, such as PFHxS, PFOS and PFOA. How PFAS mobility can increase as a result of biological remediation methods has also been reported by Mejia-Avendano et al. (2016, 2017).

Composting is an example of biodegradation and is used for sludge management, among others. In composting, microorganisms decompose organic matter under aerobic conditions. The process emits heat, which leads to an increase in the temperature of the compost (Manga et al., 2021). There are only a few studies are available, which show how PFAS behave in sludge composting. According to Giovanoulis et al. (2019), compost treatment of the sludge from Henriksdal WWTP resulted in a slight decrease of PFAS concentrations after six months of composting, but this was mostly due to the release of volatile PFAS to the atmosphere (see section 6.6.5). Therefore, this is not a biodegradation but a chemical process.
6.5.4 Other destruction techniques

Supercritical Wet oxidation (SCWO) is similar to thermal destruction due to the high temperatures and pressures developed during the process. The WO process is based on the oxidation of all organic substances present in the water matrix, with pressures above 221 bar and temperatures above 374 °C. The high temperature and high pressure (supercritical state) give substances unique properties, which can best be described as the absence of a clear boundary between different phases. WO is usually performed on a residual product or treated water where PFAS levels are high. However, supercritical WO has also been tested for the destruction of PFAS in leachate (without concentration) and for the destruction of concentrates from foam fractionation (Malovanyy et al., 2021; Aquarden personal communication 2021). The experiments showed a PFAS reduction of over 99 % when concentrates from foam fractionation were treated and the degradation of PFAS was complete, which was confirmed by additional analyses and a calculation of fluoride balance. When large volumes without concentration were to be treated, the high treatment cost of more than 1000 SEK/m³ became a problem. However, destruction of concentrate streams with high PFAS concentrations (which was not tested) could be an interesting application of the technique, as the cost could be cheaper than high temperature incineration. The technique has also been tested for the destruction of solid materials (e.g., used carbon and ion exchangers from PFAS water treatment). The trials showed that it is technically possible to destroy solid material containing PFAS, but the destruction cost is higher than the high temperature incineration (personal communication Aquarden 2021).

Cavitation processes other than sonolysis (such as hydrodynamic cavitation) can be effective in destroying PFAS because both high temperatures and radical formation occur in a similar manner. However, no relevant studies could be identified where this method has been investigated in detail.

Advanced oxidation processes (AOP), sometimes used for the destruction of pharmaceutical residues, have so far given variable results with oxidation of different PFAS, but this is dependent on the present oxidant, catalyst, and PFAS being studied. For example, hydrogen peroxide in the presence of iron (III) has been shown to work well for PFOA (Mitchell et al., 2014). Heat-treated persulfate (S²O₈²⁻) has also provided an increased removal efficiency for PFOA, while generating short-chain PFCA at the same times during the reaction. The method was also found to be highly dependent on both chloride levels and pH and did not work at all for PFOS (Bruton and Sedlak, 2018). Many other advanced oxidation techniques such as ozone/H₂O₂, ozone/UV, and ozone/TiO₂ provided little to no
treatment effect due to the antioxidant properties of PFAS (Huang et al., 2016; Schröder and Meesters, 2005).

**Mechanochemical destruction** has been shown to be effective in destroying PFOA and PFOS in contaminated soils and is a promising technique for sludge remediation (Berg et al., 2022; Turner et al., 2021). This method involves processes such as grinding and shearing, (i.e., processes that alter the crystalline structure of the material), which leads to the emission of electrons, ions, and radicals. With the addition of potassium hydroxide KOH, for example, the electrons and radicals formed react with PFAS substances and resulting in de-fluorination. The production of other PFASs through the conversion of PFAS precursors and the emission of exhaust gases containing PFAS is a risk associated with this method.

Tang et al. (2022) recently presented a new destruction mechanism by the addition of dimethyl sulfoxide (DMSO) and boiling at 120 °C for 24 hours. Although this represents a new potential destruction option for PFAS, it requires large amounts of DMSO and a long reaction time. Whether the technique is more resource efficient than other destruction techniques remains to be demonstrated.

### 6.5.5 Final disposal in closed landfill with leachate management

A hypothetical option for PFAS management is to concentrate PFAS from water and store the concentrate as a liquid or sorbed into a solid, in a safe manner to minimize leakage. For example, used activated carbon or ion exchange resins could be deposited in this manner until more is known about the degradation of PFAS by thermal destruction. According to the Swedish Waste Regulation (Avfallsförordningen 2020:614), waste should be classified as hazardous based on the concentrations listed in Annex III of the EU Waste Regulation 2008/98/EC. As PFOS is toxic to reproduction, the limit set at 0.3 % (3,000 mg/kg) for the waste to be classified as hazardous. The PFOS content in an adsorbent is much lower than this limit and as such, there are currently no legal requirements to send used GAC to landfills with other non-hazardous waste. Organic materials such as ion exchange resins (polymer) are not permitted to be sent to landfills in Sweden. Although there are no hindrances to fill a landfill with used GAC and non-hazardous waste material, waste management companies do not want to accept material with high PFAS content, even for treatment as hazardous waste. This is because of the high risk PFAS poses if it leaches out and the associated costly cleanup treatment. Landfilling of PFAS in concentrated form is therefore not practiced today.
6.6 Relevant activities in Sweden

The following section presents the activities concerning PFAS that are relevant to the water sector and identified by the project team.

6.6.1 Knowledge building

Without knowledge, there is no possibility of resource-efficient measures to remove PFAS from society and the environment. Given that PFAS knowledge is still relatively limited, the dissemination of knowledge is therefore essential for future protective measures to be developed and implemented. Table 6.1 lists active knowledge-building initiatives.

Table 6.1. Selection of relevant projects with the aim of building and spreading PFAS knowledge

<table>
<thead>
<tr>
<th>Project Description</th>
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<tbody>
<tr>
<td><strong>Zero PFAS I and II - Baltic Sea PFAS Network (Coordinator IVL)</strong></td>
</tr>
<tr>
<td>A knowledge sharing platform to improve the understanding of PFAS in waste streams in the Baltic Sea region and to reduce emissions to the Baltic Sea. The network must ensure knowledge sharing and experience transfer in PFAS use and handling, reduction, environmental impact, as well as policy and regulations. The network organizes, among other things, workshops, and conferences. The network is open to anyone who is interested and free of charge to join. Flagship of EU Strategy for the Baltic Sea Region, Policy Area Hazard, under PA Coordinator Swedish EPA until 2021, now PA Coordinator HaV (<a href="#">More info</a>).</td>
</tr>
<tr>
<td><strong>The PFAS network and guide on PFAS (KEMI)</strong></td>
</tr>
<tr>
<td>The PFAS network consists of authority representatives, researchers, county administrative boards, municipalities, consultants, and companies and aims above all to share knowledge. The PFAS Guide has been produced by the authority network for PFAS issues, which includes the Chemicals Inspectorate, the Swedish Food Agency, the Swedish Environmental Protection Agency, the Swedish Geological Survey (SGU), the Swedish Geotechnical Institute (SGI), the Norwegian Maritime and Water Authority, the Swedish Defence Inspectorate for Health and the Environment and the County Administrative Board are included. The guide is updated in line with the work in the authority network (<a href="#">More info</a>).</td>
</tr>
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</table>

6.6.2 Upstream leachate treatment at waste facilities

There are pilot and full-scale projects for the treatment of PFAS contaminated leachate in Sweden (Table 6.2). Summary of some of the experiences are briefly shown in the following subchapter.
Table 6.2 Selection of relevant projects on PFAS treatment of leachate water.

<table>
<thead>
<tr>
<th>Project Description</th>
<th>Details</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Treatment of PFAS-contaminated water from waste facilities (IVL/Avfall Sweden)</strong></td>
<td>Practical trials at lab and pilot scale for the treatment of PFAS-contaminated leachate from waste facilities. Tested techniques include granulated activated carbon (GAC), ion exchangers, powdered activated carbon (PAC), ozonation, nanofiltration and foam fractionation. Treatment costs were calculated for all techniques. Foam fractionation, treatment with GAC and ion exchangers emerged as the most promising techniques and showed similar costs for the same degree of reduction. (Reference: Malovanyy et al., 2021, Direct Link.)</td>
</tr>
<tr>
<td><strong>Leachate treatment in innovative upstream filters - DynaSand (Lysekil)</strong></td>
<td>The project is carried out by IVL in collaboration with the waste company Rambo (Lysekil municipality) and the technology supplier Nordic Water and aims to test a new robust ion exchange filter for resource-efficient treatment of leachate for PFAS (More info).</td>
</tr>
<tr>
<td><strong>Remediation of contaminated areas with salix (Älvkarleby municipality)</strong></td>
<td>The salix plant (willow plant) at the Dragmossen landfill is irrigated with collected leachate. By felling the woody plant, the landfill can be cleaned up in the long run. The salix is transported to a heating plant for combustion and the flue gases are cleaned of impurities and the ash is taken care of (More info).</td>
</tr>
<tr>
<td><strong>Leachate water treatment with UF membrane and ion exchange filter with a focus on regeneration of ion exchangers (IVL/Ragn-Sells/Avfall Sweden)</strong></td>
<td>Several different commercial ion exchange masses are being tested on a pilot scale. Regeneration of the masses with different regeneration solutions and PFAS treatment after regeneration are also tested (More info).</td>
</tr>
<tr>
<td><strong>The potential of overflow surfaces to clean leachate from PFAS (Exam thesis SLU)</strong></td>
<td>At four waste facilities in Sweden, the potential of overflow surfaces as a treatment method for PFAS in leachate has been investigated. Plant and soil samples have also been taken in this study, which was carried out with the help of the company Water Revival Systems AB (WRS AB) and Waste Sweden. However, the potential through biomass harvesting was estimated at only 0.03 - 0.3 %, which does not justify an upscaling (More info).</td>
</tr>
<tr>
<td><strong>Evaluation of the treatment effect for PFAS in two full-scale plants with activated carbon (Avfall Sverige)</strong></td>
<td>Experiences from new full-scale plants at the Tagene and Fläskede landfills consisting of a chemical treatment step followed by carbon filtration (More info).</td>
</tr>
<tr>
<td><strong>Treatment of PFAS in leachate with Surface Active Foam Fraction (SAFF) (NSR/Envytech/Avfall Sverige)</strong></td>
<td>Evaluation of foam fractionation in several stages at full scale at a waste facility in Helsingborg (More info).</td>
</tr>
<tr>
<td><strong>Sustainable management of PFAS-contaminated materials (Less-PFAS)</strong> (Luleå University of Technology)</td>
<td>The project's goal is to develop a working treatment methodology for PFAS-contaminated materials. The method must be able to provide a PFAS breakdown in water and soil and includes tests with e.g., with UV and ultrasound treatment (More info).</td>
</tr>
</tbody>
</table>

6.6.2.1 Nanofiltration (NF)

In short-term batch experiments, the treatment of two types of leachates were tested using nanofilters with a cut-off of 270 Da. by short-term batch experiments. In both cases, 80 – 90 % of PFAS measured as PFAS11 was removed (Malovanyy et al., 2021). 97 % water recovery was achieved in the short-term trials, but in the long-term operation, water recovery was estimated to be at most 75 – 80 %. The retentate from nanofiltration was treated with ozone with good oxidation for long-chained PFAS but poor oxidation for the short-chained ones. The success of ozonation in this study may be explained by the fact that ozone reacted with other
substances in the leachate (e.g., iron), resulting in catalytic ozonation. As discussed earlier in the report, ozonation alone is not a recommended technique for the treatment of PFAS-contaminated water. It is because oxidation of PFAS is difficult, and knowledge of possible degradation products from oxidation of PFAS is limited. Recycling of retentate to landfill from membrane treatment is a common method for leachate treatment in other countries such as in Germany and Norway, but not in Sweden. Although recycling of the retentate provides a cost-effective treatment of PFAS, there are risks of leakage of high PFAS concentrations to groundwater and surface water in old landfills that lack bottom sealing. In modern landfills, waste is stored in separate cells, which also makes re-infiltration more difficult.

6.6.2.2 Granular activated carbon filter (GAC)

Treatments of PFAS-contaminated leachate with carbon filters have been tested using at least 9 different landfill leachates (Malovanyy et al., 2021). There are also two full-scale plants using carbon filters at the Tagene and Fläskebo waste disposal sites (both in and near Gothenburg).

Pilot trials showed that PFAS breakthroughs depended more on the DOC (dissolved organic carbon) content of the water than on the PFAS content. An example of this phenomenon is shown in Figure 6.4. The test was conducted with different leachates for PFOS reduction using carbon filters with the same design and carbon type (Malovanyy et al., 2021). The first leachate (A) had a mean DOC content of 13 mg/L, a PFOS content of about 760 ng/L and a ΣPFAS11 content of about 3,300 ng/L (Figure 6.4 top). The second leachate (B) had a much higher DOC content of 92 mg/L, a much lower PFOS content (16 ng/L), but a comparable ΣPFAS11 content (4,400 ng/L). Despite the lower PFOS content in leachate B, the reduction rate was low (already after treatment of 1,000 BV) compared to the reduction of PFOS for leachate A, which had a lower DOC content (until approximately 3,500 BV).
The breakthrough of short PFAS came much earlier than for longer PFAS. Therefore, the filter material may need to be changed already after 200 BV (equivalent to less than one week of operation), if the goal of the treatment is also a good reduction of short PFAS. It is therefore not suitable to remove the shortest PFAS from leachate with activated carbon. Comparative trials with 2 different types of activated carbon also showed that a cheaper reactivated carbon gave a better overall economy compared to fresh high-quality activated carbon (Malovanyy et al., 2021).

6.6.2.3 Ion exchanger (AIX)

Ion exchangers have also been tested in several pilot experiments as described by Malovanyy et al. (2021). In all known documented experiments on contaminated
water (leachate and wastewater) in Sweden, ion exchangers of the type Purolite PFA694E were used. Uppsala Vatten and SLU have also conducted experiments on the treatment of contaminated groundwater with ion exchangers of the type Purolite A-600 (Franke et al., 2019; McCleaf et al., 2017). Trials with ion exchangers from other suppliers are also ongoing in projects by Ragn-Sells and Rambo. In addition, a full-scale facility at the Sofielund waste facility is in the process of design planning. The overall trials suggest that ion exchange resins were effective in removing PFSA and long-chained PFCA. The results also indicated that ion exchange resins were less sensitive to DOC than activated carbon. For example, a reduction of PFOS by more than 80 % could be maintained up to 55,000 BV of the same leachate (leachate B, see also previous section 6.6.2). However, reduction of short PFAS is still problematic where, for example, the reduction of PFBA drops to 0 % after less than 3,500 BV, corresponding to two weeks of operation.

Ion exchange resin is 5 - 15 times more expensive per m³ than GAC, and to evaluate possibility of cost reduction, regeneration of used ion exchange resin in bench scale with a solution of methanol was tested by Malovanyy et al. (2021) with promising results in the regeneration of sorption capacity. Regeneration of ion exchange resin with an ethanol solution is now also being tested at the Högbytorp waste facility. The project also evaluates how PFAS can be separated from the regeneration solution and whether the resins retain its function after regeneration. However, the suppliers’ general recommendations are single use of ion exchange resins which than should be incinerated. The volumes of used filter material are much less than GAC material, and destruction via high temperature incineration is therefore more manageable.

6.6.2.4 Foam fractionation

Foam fractionation was tested with six different types of leachates from Swedish waste facilities by batch experiments on a bench scale (Malovanyy et al., 2021). Similar result trends from the experiments with different types of leachates were observed. More than 90 % reduction of PFAS with more than six fluorinated carbon atoms was observed whereas less than 40 % was observed for short-chain PFAS (Figure 6.5).
Foam fractionation has been in full operation for more than a year at the Tveta waste facility, where similar results as in Figure 6.5 have also been observed at full scale. The full-scale concentration was 25,000 times (Hinrichsen H., personal communication, 11.03.2021), which creates good probability to send the concentrate for thermal destruction. Operating costs for this technique are much lower than for other techniques and amount to about 0.9 kr/m³, which can be furthered lowered with more optimizations. However, the total cost of treatment is around 7 kr/m³. PFAS concentration in foam was also unintentionally occurred during the biological treatment of leachate. With analysis of foam from respective leachate treatment, high PFAS concentration was found in Västerås with their SBR (sequencing batch reactor), Uppsala with MBBR (Moving Bed Biofilm Reactor), and Helsingborg with aerated pond.

6.6.3 At wastewater treatment plants

During the review for this report (2022), there were no projects which solely investigates PFAS treatment at WWTPs in Sweden. Table 6.3 lists relevant pilot or full-scale projects for the treatment of micropollutants (including PFAS) at Swedish WWTPs. Each project is briefly described below.
The combination of aeration in biological treatments and foam fractionation is an interesting area for future research, however the PFAS-release to the atmosphere also needs to be investigated (see section 3.1.3). It has been observed in several places in Sweden that the concentration of PFAS occurred, when during foaming in biological treatments or pumping of water. For example, the PFAS content in foam from leachate treatments from SBR at Gryta waste facility in Västerås was 200 times higher than in incoming water whereas the PFAS content in foam from an aerated pond at NSR’s Filborna waste facility in Helsingborg was 900 times higher than in incoming water. Extremely high concentrations were observed in foam from leachate treatment in an MBBR (Moving Bed Biofilm Reactor) facility at Hovgården waste facility in Uppsala.

Intense foaming in activated sludge plants is sometimes a problem, especially for plants with a MBR (Membrane Bio Reactor) process. Collecting this foam (in the process of surface removal of excess sludge) and treating it separately could be a way to both reduce the work environment problem of foaming (e.g., manual flushing) and remove PFAS from the water. Samples of foams from the biological treatment in an MBR process have been taken for evaluation from the pilot operated at Hammarby Sjöstadsverk together with Stockholm Vatten och Avfall, and the pilot operated by Syvab at Himmerfjärdsverket. A two-step foam fractionation under a controlled environment has also been tested on a bench scale with the effluent water from the MBR process at Hammarby Sjöstadsverk. PFAS
with six or more full fluorinated carbon atoms were separated to below the detection limit, resulting in an 82% reduction of PFAS11, and a PFAS concentration achieved of 50 – 100 times (internal unpublished data, IVL). However, it is important to note that a small dose of detergent was added to this test, as the foaming was initially insufficient.

Other simple experiments combining biological treatment of water with foam fractionation were described by Malovanyy et al. (2021). Leachate was added to biofilm carriers with established biofilm and aerated for 10 hours to simulate biological treatment in the MBBR process. Foam formed in the process was led to a reactor and collected separately. In these simple experiments, long-chain PFAS were reduced by more than 96%.

In conclusion, PFAS could potentially be separated from wastewater by foam fractionation. However, it is unclear whether this process could be combined with aeration in biological treatments. Method optimizations are required. Further investigations are required to determine if PFAS could be transferred to the air phase via volatile PFAS and aerosols. Collection of foam formed in the activated sludge process seemed to be inefficient for effective PFAS treatment. However, it could be interesting to analyze foams from the biological steps of a MBBR process, due to the higher air intensity and lower sludge content compared to the activated sludge process.

6.6.4 Sludge, soil, and water resources

Several projects are currently underway in other areas to address PFAS-contaminated water resources (Table 6.4). These projects are not directly connected work carried out by the water sector as presented in previous chapters.

| New and existing water treatment techniques for removing PFAS substances in waterworks (Svenskt Vatten) |
| An extensive literature review of both current and new treatment methods as well as an assessment of their treatment efficiency, costs, and energy requirements (More info). |
| Microbial degradation of PFAS in remediation of contaminated soil and groundwater (UU, SLU) |
| Research project to expand knowledge about microbial degradation of PFAS. The project will identify PFAS-degrading bacteria, map optimal conditions for microbial degradation in soil and groundwater and investigate which substances can be completely degraded. The knowledge must also be applied in a demonstration trial. The project will last for three years between 2022 and 2025 (More info: Fritjof Fagerlund). |
| Degradation of PFAS in-situ by sonolysis (ultrasound) (LTU) |
| The sonolysis technique is to be tested in the field for the treatment of PFAS-contaminated groundwater. The study is carried out in collaboration between researchers at Luleå University of Technology and Clarkson University in the USA, entrepreneur and user, respectively. Laboratory |
6.6.5 Suitable sludge treatment techniques

Apart from sludge incineration, which is currently not widely applied in Sweden, there are no appropriate alternative treatment methods to reduce high PFAS content in sludge. Sludge incineration is a thermal process and is discussed in detail in section 6.5.1. Another thermal process is pyrolysis, but the process parameters required for the destruction of PFAS are still unclear. Mechanochemical destruction (grinding), which is used for the treatment of contaminated soils, may also be relevant for sludge treatment, however, further research is required. A downside of using thermal and mechanochemical treatments removing PFAS in sludge is that some desirable resources within the sludge (e.g., nutrients, mulch, and trace elements), would be destroyed and lost for recycling.

Giovanoulis et al., (2019), investigated whether open and/or covered storage and composting of mesophilic or thermophilic digested sludge had an effect on the reduction of PFAS content within sludge. Experiments with storage showed no reduction of PFAS contents in the sludge during storage (Table 6.4, project: Storage of sewage sludge under different conditions). However, the PFAS content increased almost ten times during 12 months of storage in non-composted sludge, which could have resulted from the conversion of PFAS precursors. Results from TOP and EOF showed that a significant proportion (91 – 97 %) of previously unidentified PFAS were present in the sludge in both experiments. A compost treatment of digested sludge resulted in some reduction of PFAS levels.

Ebrahimi et al. (2021) have shown that both the treatment process (e.g., pH, salinity, additives) and sludge stabilization methods (e.g., digestion, aeration, and composting) could alter the leaching potential of PFAS, thus affecting whether...
PFAS in sludge could be released into the surrounding environment. However, more research is needed in this area.

### 6.6.6 Treatment techniques applicable for temporary discharges

Temporary construction works, such as major road projects, often need to deal with temporary release of construction runoff and/or county water. In these cases, the operator obtains a permit from the Swedish regulatory authority for the release into the stormwater network or to the recipient water. If PFAS are suspected to be present in the county water or if the recipient water already has a high PFAS content, the regulatory authority may pose special terms. Assessments are usually made on a case-by-case basis. Some water authorities, such as Stockholm Vatten och Avfall, have developed guidelines for PFAS-containing county wastewater, which define requirements on what tests and analysis are to be conducted, as well as possible treatment of PFAS-contaminated county wastewater.

The techniques that can be used for the treatment or destruction of PFAS-contaminated matrices depend entirely on the type and volume of streams produced. In general, the same techniques outlined in section 6.4 can be applied.

### 6.7 Combination of removal of PFAS and other micropollutants such as pharmaceutical residues

Many of the techniques that can be used to remove PFAS from wastewater are also commonly used as advanced treatment techniques for the removal of pharmaceutical residues from wastewater (Baresel et al., 2017a, b; Swedish Environmental Protection Agency 2017). Applying these techniques at WWTPs will result in the simultaneous treatment of PFAS and other micropollutants such as phenols. However, the treatment efficiency of these techniques can be significantly different for different micropollutants. This is particularly evident for adsorptive treatment methods where different micropollutants compete for the same adsorption areas. Depending on which pollutants need to be removed from water with the chosen technique, the design and operation need to be optimized. As discussed earlier, ozonation alone cannot successfully oxidize PFAS.

As an example of such combined treatments of pharmaceutical residues and PFAS in GAC filters, breakthrough curves from column experiments at Getteröverket
(Varberg) and Främby WWTP (Falun) is presented in Figure 6.6. The designs of a GAC filter are basically the same, regardless of which micropollutants are to be removed and to what concentrations or reduction rates. With different frequencies of changing the filter material, in these cases activated carbon, the degree of reduction for the different target substances was controlled to achieve specific treatment objectives and requirements. The figure shows the reduction efficiency for diclofenac, the pharmaceutical with an established basis for assessment in surface water bodies. This controls when the activated carbon needs to be changed to avoid breakthroughs.

As Figure 6.6 shows, PFAS (in this case exemplified by PFOS, PFHxA, PFAS4 and PFAS11) had a lower absorption to the carbon compared to diclofenac, which means PFAS have a faster breakthrough than diclofenac. Also, long-chained PFAS adsorb better than short-chain ones. PFHxA has been selected as a representative substance for short PFAS and demonstrates the breakthrough in the Figure 6.6. (The other, even shorter PFAS molecules, are separated with similar or worse efficiency.) As shown in Figure 6.6, the breakthrough of PFHxA occurs earlier than PFOS and ΣPFAS4.
Figure 6.6 Combined treatment of pharmaceutical residues and PFAS in GAC filters (based on Baresel et al., 2021 and Hedén et al. 2021).

Note that in praxis, it is advisable to have two filter in series in a so-called “lead-layer” configuration to better exploit the sorption capacity. When designing two-stage GAC filters, it is most resource-efficient to change the carbon in the first filter only when its full capacity is reached, thus preventing further reduction of the target substance (diclofenac in this case). The carbon adsorption capacity for a specific substance corresponds to the area under the breakthrough curve. From Figure 6.6 it can be concluded that the capacity for the reduction of PFOS is about half of what it is for diclofenac. The capacity for the treatment of PFHxA is about 1/4 that for diclofenac. This means that if a carbon filter was operated to have adequate reduction of diclofenac, an average of about 50% of PFOS would be removed. If the carbon filter was operated to achieve adequate PFOS removal, the carbon consumption would have to be doubled in the two examples. Assuming a carbon consumption for sufficient diclofenac reduction is 15 g/m³ (corresponding
here to about 30,000 BV before replacement), then a doubling of the carbon dose is required, thus an additional cost of 0.3 – 0.5 SEK/m³.

It is difficult to say how much ΣPFAS11 could be reduced before exchanging the carbon in a plant adapted for adequate treatment of pharmaceutical residues. It would depend very much on the distribution between short and long PFAS in the specific water tested. As shown by the examples from Främby WWTP and Getteröverket, the ΣPFAS11 reduction could be estimated to be about 25 – 40 % for different waters (Figure 6.6). However, the reduction could have varied depending on the distribution of PFAS and on the treatment for targeted pharmaceuticals. If the activated carbon filter material would be changed often, it theoretically would allow short-chained PFAS to be adequately purified. However, this would require a larger plant (probably three or more filters in series) and come with challenges (e.g., the transport of fresh and used carbon filters which results in very high costs). Further optimizations for such process developments are required. As previously discussed, other treatment techniques which work well for the reduction of pharmaceutical residues (e.g., ozonation), may lead to an increase in PFAS concentrations (e.g., PFAS precursors could be converted to PFAA (see section 6.5)). This is important as there is currently insufficient knowledge of this aspect when simultaneous reduction of pharmaceutical residues and PFAS is preferred.

Synergy effects from technique combinations have yet to be investigated. It has been reported that a combination of ozonation and activated carbon could be beneficial for pharmaceutical and PFAS treatments (Baresel et al., 2015b; Kaiser et al., 2021). Ozonation could result in parent compounds of pharmaceuticals adsorbing better to activated carbon. But there is also a risk that ozonation creates polar transformation products of unknown toxicity that do not adsorb to carbon to any great extent (Betsholtz et al., 2022).

A combination of activated carbon and ion exchange resins has also shown promising synergistic effects in initial trials at the Kungsängsverket WWTP in Uppsala in Sweden. Further long-term pilot-scale tests are planned (Table 6.3). An unintended synergistic effect has also been observed for MBR-GAC in Syvab plant, combining a Membrane Bio Reactor with a carbon filter (GAC). Initial lab-scale tests showed that the membrane material (PVDF) appears to have a high adsorption capacity for PFOS (IVL, unpublished data). The high removal efficiency for PFOS already in the MBR step, but at the same time steadily decreasing, could possibly support this theory. The MBR process has so far shown an average treatment capacity of more than 80 % for PFOS since commissioning in September
2020. Moreover, the surface extraction installed for foaming sludge may contribute to additional PFOS removal in the MBR process. Foaming is common in MBR processes and can, to some extent, be similar to foam fractionation. Therefore, via a controlled skimming of the foam, PFOS could also be removed from the process and treated appropriately.

7 Guidance

This guidance is written for municipal water actors in Sweden. Environmental and health problems associated with PFAS have greatly increased in recent years, and as such new legislation and risk reductions are underway. Hence, all of Sweden's wastewater operators need to be aware of PFAS-associated problems at their plants.

The level of efforts that WWTPs should take varies between different stakeholders. This guidance therefore presents an overview assessment of how WWTPs can prioritize their work depending on its size and industries connected to the plants. Factors such as whether the operation is in the process of applying for a permit, or whether the operation has been mandated to reduce its PFAS emissions are also crucial.

All WWTPs with a load greater than 2,000 p.e. should carry out an inventory of what knowledge exist. Mapping of PFAS is recommended to all WWTPs larger than 10,000 p.e. or WWTPs of size of 2,000 – 10,000 p.e. that currently have reconstruction plans or permit application renewal or connected industry that has high indication of PFAS risk. For smaller WWTPs, two sampling events per year may initially be sufficient, while for the larger WWTPs with a size of 10,000 – 100,000 p.e., more frequent sampling events per year are appropriate (e.g., more than four). For larger WWTPs (more than 100,000 p.e.), and specifically if they are to impose a significant environmental impact with PFAS, monthly sampling may be appropriate. For treatment plants where technical measures are required, TOP/EOF analyses are also recommended. Based on the results, the impact on the recipient water could then be calculated. For all WWTPs where sampling indicated a significant environmental impact from the effluent wastewater, the project team recommends initiation of upstream work to trace the sources of PFAS. For the largest treatment plants, this work could also be included in the existing upstream work. To proceed with measures at the WWTPs, the assessment of environmental impact can show that this is justified, i.e., that the WWTP contributes a significant
environmental impact regarding PFAS, and that the investments are justified to burden the Water Sector.

Outlines of recommendation for aiding operators in addressing PFAS problems are illustrated in Table 7.1, in a step process.

Table 7.1 Recommendations for aiding WWTPs for PFAS treatment

<table>
<thead>
<tr>
<th>Step 1: Knowledge inventory – what do we already know?</th>
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</thead>
<tbody>
<tr>
<td>Compile existing knowledge about the PFAS load at the treatment plant by:</td>
</tr>
<tr>
<td>• Documenting and summarizing historical sampling with associated sampling procedure.</td>
</tr>
<tr>
<td>• Calculating (if possible) incoming load, degree of reduction across the sewage treatment plant, as well as outgoing load and impact on the recipient by dividing outgoing levels by dilution rate at the time of sampling. For coastal works, concentrations in the recipient can be estimated with dilution maps.</td>
</tr>
<tr>
<td>Compile identified PFAS sources for the WWTP as well as investigate businesses connected and list potentially polluting businesses. In this inventory, focus on connected landfills, manufacturing industries, scrap and recycling facilities, businesses that work with the surface treatment of e.g., paper, textiles or other materials, airports, washing facilities and levels in drinking water within the business area. Also document known fire drill locations and potentially contaminated areas within the operational area.</td>
</tr>
<tr>
<td>Compile knowledge of recipient impact:</td>
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<tr>
<td>• Is the recipient impact and the treatment plant's part of the impact mapped through sampling?</td>
</tr>
<tr>
<td>• Are other potential sources and pathways mapped, for example through upstream sampling?</td>
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<tr>
<td>• Are there sensitive and protected areas in the recipient, for example downstream raw water sources or water withdrawals?</td>
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<table>
<thead>
<tr>
<th>Step 2: Knowledge completion – what do we need to know more about?</th>
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<tbody>
<tr>
<td>Map PFAS levels across the treatment plant and in the recipient:</td>
</tr>
<tr>
<td>• Take several samplings throughout the year to capture variations.</td>
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<tr>
<td>• Preferably take weekly samples with a focus on the water phase. Only IN and OUT are needed as a starting point, but individual process steps could be included. The number of samplings is adapted to the aim of investigation.</td>
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<tr>
<td>• PFAS11 should be analysed, but TOP/EOF analyses may also be relevant. For the sewage treatment plants that deposit the sludge by spreading sludge on arable land, analyses of PFAS24 and EOF/TOF in the sludge should be carried out.</td>
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<td>• Make sure to get analyses with reasonable reporting limits.</td>
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<tr>
<td>• Follow up the results actively to detect deviations.</td>
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<td>• If necessary, take a sample upstream of the discharge point in the recipient to gain knowledge of the background load.</td>
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<tr>
<td>• Assess future changes in dilution in the recipient due to, among other things, increased inflow, and climate change.</td>
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<tr>
<th>Step 3: Assessment of environmental impact</th>
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<tbody>
<tr>
<td>The sewage treatment plant's load and treatment efficiency:</td>
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<tr>
<td>• Identify dominant point sources and the magnitude of their contribution.</td>
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<tr>
<td>• Identify any ambiguities that require further investigation (for example abnormally high PFAS removal).</td>
</tr>
<tr>
<td>Impact assessment in receiving recipient or other downstream areas:</td>
</tr>
<tr>
<td>• Quantify the sewage treatment plant’s contribution to the PFAS load in the recipient.</td>
</tr>
<tr>
<td>• Initially assess the recipient’s status based on analyses upstream and downstream of the treatment plant and comparison of results with the limit value for, for example, PFOS.</td>
</tr>
<tr>
<td>• Calculate the treatment plant’s contribution by comparing downstream concentrations with output concentrations divided by the dilution factor. The dilution factor can be obtained from S-HYPE. NOTE: sync the sampling time with the current flow at the treatment plant and in the recipient</td>
</tr>
</tbody>
</table>
**Step 4: Strategy work**

▷ Characterize and, if necessary, map different PFAS sources:
- Point sources such as leachate, industrial drains, and fire drills. Sampling at various points in the pipeline can identify any major PFAS contributions. Targeted analyses of specific PFAS can help identify the sources.
- Diffuse contributions, for example via leakage or runoff of ground water from contaminated areas.
- Drinking water. If PFAS are present in drinking water, this may be a reason for higher PFAS levels in wastewater.

▷ Describe planned changes that may affect PFAS emissions in the future:
- Planned expansion or process change.
- Planned expansion with advanced treatment for pharmaceutical residues.
- Planned upstream measures affecting the PFAS load.

▷ Describe overarching policy decisions that may affect the PFAS strategy (for example, goals to reuse water or zero emissions).

▷ Assess suitable future target levels for PFAS in outgoing wastewater with the starting point that limit value/assessment grounds for relevant PFAS are complied with in the recipient. Keep in mind that ongoing revisions can mean significantly changed limit values and target values, which should be reflected in target levels.

**Step 5: Exploration of action options at the sewage treatment plant**

▷ Think long-term and include the system perspective both related to the business (for example regarding process integration or reuse of purified wastewater) but also of the regional/local conditions and the results of the knowledge inventory carried out for diffusion routes and sources.

▷ Work proactively to consider stricter limit values and treatment requirements which may be set in the future.

▷ Investigate whether there is enough knowledge within the organization to investigate different options without the risk of suboptimization.

▷ Consider where in the system a purge or clean-up should be implemented to have the greatest effect.

▷ Think through which treatment or destruction techniques are relevant for implementation from a larger perspective upstream work (e.g., upstream work)

▷ Think about how to not only clean PFAS from, for example, water, but also remove it from the cycle. Consider appropriate handling all the way out. If, for example, a PFAS-rich residual stream is produced, it needs to be handled appropriately.

**Step 6: Possible implementation of measures at treatment plants**

▷ Is there a need for practical tests with different techniques to obtain a decision basis (including design, treatment effect, investment and operating costs, cost, and operating economics)?

▷ How should you think if you are investigating the need for, building or already have a step for pharmaceutical removal? Will the planned removal of pharmaceutical residues provide some reduction of PFAS? Can another technique provide similar separation rate and cost for pharmaceutical residues but higher reduction of PFAS?

▷ Which analysis should be done in test pilots (for example TOP and at least PFAS to for mass balances)?

▷ What reporting limits are required for adequate technique evaluation.
8 Conclusions and recommendations

This chapter begins with a discussion of the conclusions that the project group wish to highlight. Then the recommendations that important to be communicate are presented.

8.1 Discussion and conclusion

8.1.1 PFAS substances are everywhere

We can conclude that PFAS and PFAS precursors are everywhere and even if there was a total ban on PFAS, PFAS would still be around for a long time. For such a ban to have an effect, it must be done globally. Unfortunately, the banning of a single PFAS today only leads to its substitution by another substance, which unfortunately may be equally toxic or persistent (so-called false substitution).

8.1.2 Measures for wastewater treatment plants

From the mass flow scenario presented in this report (see Figure 3.1 and Figure 5.1) effective removal of PFAS from society and environment can only be achieved if the introduction of PFAS via products is stopped. PFAS handling globally has also to be phased out to reduce atmospheric imports of PFAS to Sweden. Focusing solely on WWTPs as a potential removal opportunity for PFAS in the cycle, through the implementation of additional treatment steps or the destruction of sludge, will therefore would not provide sufficient improvement in the environment or to raw water resources.

The PFOS flow through Swedish wastewater treatment plants is estimated to be about 22 kg/year, which represents approximately 5 % of the PFOS input. As imports and leakage from contaminated areas are many times higher, this flow appears to be less of a priority. Nevertheless, resource-efficient treatment steps for PFAS at WWTPs may provide synergy with the removal of other micropollutants to protect sensitive recipients. As discussed in this report, approximately 50 % of PFOS and other PFAS, which otherwise would be discharged, could be removed as a by-product by installing activated carbon filters for the purpose of removal of pharmaceutical residues. Moreover, the majority of PFAS could be destroyed and removed from the cycle when these used filter materials are treated with process of thermal regeneration or destruction.
8.1.3 Treatment of sewage sludge

Measures for a change in sludge management appear to be less effective in tackling the PFAS problem from a wider perspective. For example, atmospheric deposition of PFOS on arable land is in the same range as PFOS dispersal via sludge fertilization. However, if a general ban on PFAS is not implemented, even sewage sludge with low PFAS concentrations could become a challenge, as persistent PFAS would accumulate in the environment over time. To reduce PFAS dispersion via sludge fertilization, effective upstream management, (e.g., landfill and/or industrial wastewater treatment), should be prioritized. For WWTPs with very high PFAS removal, the causes should be investigated, in the case of high PFAS content transferred to sludge with efficient removal from water phase, alternative methods for sludge management such as incineration should also be considered.

8.1.4 Focus of PFAS management

In principle PFAS management should focus on PFAS contaminants as a group, but also on individual PFAS substances that have been shown to have high toxic effects on humans and other organisms. With extremely long half-lives, some up to 30 years, PFAS leaves a large trace behind. Only direct action may therefore prove effective over a longer period. It is valuable if the technological solution can remove several PFAS simultaneously, and that a holistic approach is considered. For example, focusing solely on PFOS removal is not resource efficient nor cost-effective and would not solve the overall problem of PFAS pollution. In addition to PFOS, other PFAS with established tolerable intakes (e.g., PFOA, PFHxS, PFNA) should be monitored and reduced, if abnormally high levels are present, or if effluents discharge into sensitive recipients. A comprehensive approach to micropollutants, that also includes PFOS, has a good chance of providing an environmental boost and benefit socio-economically.

In some cases, the focus may also be on a single PFAS or a selection of several PFAS. This may be the case with (I.) specific emissions, (II.) the presence of specific PFAS, or (III.) if some specific PFAS may represent indicator pollutants for other PFAS. The latter may be useful when evaluating a particular treatment technique. Should a global phase-out of PFAS take place, WWTPs become one of the few places where PFAS flows can be continuously monitored, measurements for removal can be put in place.
8.1.5 Technology options for PFAS measures

In reviewing possible technologies for PFAS remediation, the project team has observed that there are currently no promising techniques at WWTPs for high-level removal of most PFAS (e.g., PFOS and short-chained PFAS). However, long-chain PFAS, which are proven to have toxic effect on humans, could be partially removed if advanced treatments for other micropollutants such as based on sorption to activated carbon were introduced (i.e., pharmaceutical residues). Removal of PFAS with foam fractionation with relatively low energy consumption is promising, however the technique requires further developments and adaptations in order to be used with higher waterflow. Investments are required for technique developments, upstream work, and coordination with other activities on a national scale.

8.1.6 Prioritization depending on contamination levels

The levels of PFAS that are present in heavily contaminated sites such as fire training sites and in other environments such as "normal" surface- and groundwater and in sewage, could be described as highly variable. In these circumstances, users of PFAS who have caused or contributed to the contamination should have a responsibility to take the necessary measures for removal. It is most resource-efficient and effective to first address the heavily contaminated sites to prevent the leakage and spread of PFAS to the environment. On the other hand, complementary measures are also needed for PFAS streams with lower concentrations in order to achieve very low limit values currently in place such in surface water or in the pipeline for drinking water. These measures may be needed for example at WWTPs if their contribution is a significant part of the PFAS contamination in the recipient water, and when remedies at other sources are more difficult.

8.2 Future recommendations on PFAS

Based on the above discussion and conclusions, the project team has developed some recommendations for what we consider to be important for the future work on PFAS for Swedish wastewater treatment plants and water industry. The recommendations are grouped into three areas as below.
8.2.1 Establishment and effective dissemination of knowledge

- A broad and organized mapping of PFAS for treatment plants and recipients should be carried out at Swedish WWTPs. For assessment, plants with the following conditions should primarily be included: plants larger than 100,000 PE; those with a low initial dilution in the receiving water (<10 times); and those connected to industries that may contribute PFAS-contaminated wastewater. For coordinated and thorough data collection, the mapping should be repeated periodically, and the work can be carried out under the auspices of an existing PFAS network.

- Sampling of sewage transport pipelines to identify "concentrated" PFAS streams for a better understanding of e.g., how the PFAS load into wastewater treatment plants is affected by leakage from contaminated land via stormwater and infiltration.

- Improved environmental monitoring and data management - and access for PFAS is needed in general, to enable continuous knowledge building.

- Improved inventory of PFAS use and flows in Sweden, and improved documents for upstream work with information on the type of industries that can potentially pollute and dispersion pathways.

- Improved access to existing databases related to PFAS with streamlined reporting and data integration with VISS (Vatten Information System Sverige, Water Information System Sweden) or other appropriate systems used for environmental monitoring.

- More research/knowledge on PFAS emissions via process air at WWTPs is needed, especially with treatment processes such as MBR and MBBR with higher aeration rate. Health and safety issues must also be addressed.

- More research/knowledge on PFAS emissions and leaching potential from sludge in different treatment methods (partly included in ongoing SVU project by Anna Kärrman, Örebro University, SVU project 21-107).

8.2.2 PFAS quantification and analyses

- Only PFAS analyses with a reporting limit lower than the current limit in inland and coastal waters should be procured for assessment of recipient status and impact.

- Improved analytical confidence. For example, collaboration efforts to compare analysis among analytical laboratories on yearly basis (accredited and non-accredited) using both environmental and prepared samples, e.g., this could be under the guidance of Svenskt Vatten as part of other micropollutants.
Broad-spectrum analyses such as EOF or TOF (for sludge only) should complement specific PFAS analyses such as PFAS11, when assessing flow balances or evaluating treatment and destruction techniques.

8.2.3 Development of treatment and destruction technologies

- Wastewater treatment plants with a high reduction of PFAS should be investigated. What techniques/mechanisms have been done to lead to such high reduction rates, and how (if possible) PFAS end up in other fractions to a greater extent (including air)? Both targeted and broad analyses should be used here.
- Projects that explore PFAS treatment technologies which provide efficient destruction without risk of by-products, and with low resource consumption should be supported.
- Synergy effects in the treatment of PFAS and other micropollutants such as pharmaceutical residues need to be investigated. This can be used to determine whether technical installations for PFAS treatment at Swedish wastewater treatment plants are reasonable.
- Treatment techniques that are not primarily intended for PFAS treatment, but benefit PFAS separation, should also be further investigated. Such techniques may include the MBR process, as foaming may be more substantial in this process and potential adsorption of PFAS to the ultrafilter membranes may be utilized. PFAS removal by foam fractionation in other process configurations and different ways of collecting foam should also be evaluated.

Finally, yet importantly, efforts in regular knowledge transfer to WWTPs on new developments in the PFAS field is recommended, when for example new guidelines will come to effect, or when new techniques or new projects information becomes available.
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This report has been reviewed and approved in accordance with IVL’s audit and approval management system.