The Potential for Phytoremediation of PFAS through Salix

A Case Study of a Firefighting Training Facility in Revinge

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The Potential for Phytoremediation of PFAS through Salix – A Case Study of a Firefighting Training Facility in Revinge

Potentialen för Fytoremediering av PFAS genom Salix – En Fallstudie av en Brandövningsplats i Revinge

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Preface

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Klara Engblom & Maja Mårtensson, Lund 2025

Abstract

PFAS (per- and polyfluoroalkyl substances) are a group of synthetically manufactured chemicals contributing to environmental pollution. The largest point source of PFAS in Sweden is the usage of firefighting foam, and at MSB (Swedish Civil Contingencies Agency) Revinge, firefighting foam has frequently been used. Moreover, PFAS is known to accumulate in plants, and a Salix plant cultivation has been maintained on MSB's property for the past 15 years. This thesis aims to assess the efficiency of the uptake of PFAS in Salix at MSB Revinge, and thus phytoremediation as a remediation method.

The methodology in this thesis consists of a literature review and sampling of plant parts, soil and waters at MSB Revinge. The results showed a decreasing concentration of PFAS in the Salix in the order: leaves> twigs> bark> root> stem. Leaves sampled from Salix in autumn contained higher concentrations than those collected in spring, while leaves collected from the ground, which had fallen previous season, had the lowest concentration amongst the leaf samples. Furthermore, different PFAS accumulated in various parts of the Salix, where the fraction of short-chained PFAS dominated in all parts except the root which contained mainly long-chained PFAS. The input of PFAS to the cultivation through irrigation with contaminated water was calculated to 22-32 g after 15 years. The removal of PFAS through harvest of the Salix cultivation was estimated to 0.0074-0.084 g per year, and if an annual leaf collection had been implemented during the previous 15 years, the potential removal was estimated to 0.25-0.33 g per year. The literature study has identified Salix as a suitable plant choice for phytoremediation of PFAS and sampling results have shown measurable PFAS removal. PFAS accumulates in the leaves and consequently, incorporation of a leaf collection system is essential. Although, calculations in this thesis are based on many assumptions and should therefore be interpreted with caution.

Sammanfattning

PFAS (per- och polyfluorerade alkylsubstanser) är en grupp syntetiskt tillverkade kemikalier som bidrar till förorening av miljön. Den största punktkällan till PFAS i Sverige är användningen av brandsläckningsskum, och vid MSB (Myndigheten för samhällsskydd och beredskap) Revinge har brandsläckningsskum använts flitigt. PFAS kan dessutom ansamlas i växter, och på MSB har en Salixodling funnits de senaste 15 åren. Syftet med detta examensarbete är att bedöma effektiviteten av upptaget av PFAS i Salix på MSB Revinge, och därmed fytoremediering som saneringsmetod.

Metodiken som används i detta examensarbete består av en litteraturstudie och provtagning av växtdelar, jord och vatten på MSB Revinge. Resultaten visade en minskande koncentration av PFAS i Salix i ordningen: löv> kvistar> bark> rot> stam. Löv som samlades in från Salix på hösten innehöll högre koncentrationer än de som samlades in på våren, medan löv som samlats in från marken och som hade fallit föregående säsong hade den lägsta koncentrationen bland bladproverna. Dessutom ackumulerades olika PFAS i olika delar av Salix, där andelen kort-kedjade PFAS dominerade i alla delar utom roten som huvudsakligen innehöll lång-kedjade PFAS. Tillförseln av PFAS till odlingen genom bevattning med förorenat vatten beräknades till 22-32 g efter 15 år. Avlägsnandet av PFAS genom skörd av salixodlingen uppskattades till 0,0074-0,084 g per år, och om en årlig bladinsamling hade implementerats under de föregående 15 åren uppskattades det potentiella avlägsnandet till 0,25-0,33 g per år. Litteraturstudien har identifierat Salix som ett lämpligt val av växt för fytoremediering av PFAS och provtagningsresultaten har visat ett mätbart avlägsnande av PFAS. PFAS ackumuleras i bladen och följaktligen är implementering av ett bladinsamlingssystem avgörande. Beräkningarna i detta examensarbete är dock baserade på många antaganden, och bör därför tolkas med försiktighet.

Table of contents

List of abbreviations

AFFFs	Aqueous film forming foams
BCF	Bioconcentration factor
dw	Dry weight
K_d	Solid/liquid partitioning coefficient
Kow	Octanol-water partitioning coefficient
MSB	Swedish Civil Contingencies Agency
OECD	Organisation for Economic Co-operation and Development
POP	Persistent organic pollutants
SGI	Swedish Geotechnical Institute
SGU	The Geological Survey of Sweden
SOM	Soil organic matter
TOC	Total organic carbon
$UCLM^{95}$	"Upper Confidence Limit of the Mean" with a confidence level of 95%
ww	Wet weight

PFAS	**Per- and polyfluoroalkyl substances**		
PFAA	Perfluoroalkyl acids		
	PFCA	Perfluoroalkyl carboxylic acids	
		TFA	Trifluoroacetic acid
		PFBA	Perfluorobutanoic acid
		PFPeA	Perfluoropentanoic acid
		PFHxA	Perfluorohexanoic acid
		PFHpA	Perfluoroheptanoic acid
		PFOA	Perfluorooctanoic acid
		PFNA	Perfluorononanoic acid
		PFDA	Perfluorodecanoic acid
		PFUdA	Perfluoroundecanoic acid

PFDoA Perfluorododecanoic acid

PFTrDA Perfluorotridecanoic acid

PFTeDA Perfluorotetradecanoic acid

PFHxDA Perfluorohexadecanoic acid

PFODA Perfluorooctadecanoic acid

PFSA Perfluoroalkyl sulfonic acids

PFPrS Perfluoropropanesulfonic acid

PFBS Perfluorobutanesulfonic acid

PFPeS Perfluoropentanesulfonic acid

PFHxS Perfluorohexanesulfonic acid

PFHpS Perfluoroheptanesulfonic acid

PFOS Perfluorooctanesulfonic acid

PFNS Perfluorononanesulfonic acid

PFDS Perfluorodecanesulfonic acid

PFUnDS Perfluoroundecanesulfonic acid

PFDoS Perfluorododecanesulfonic acid

PFTrDS Perfluorotridecanesulfonic acid

PASF Perfluoroalkane sulfonyl fluorides

FTS Fluorotelomer sulfonates

4:2 FTS 4:2 Fluorotelomer sulfonate

6:2 FTS 6:2 Fluorotelomer sulfonate

8:2 FTS 8:2 Fluorotelomer sulfonate

10:2 FTS 10:2 Fluorotelomer sulfonate

FASA Perfluoroalkane sulfonamides

FOSA Perfluorooctane sulfonamides

PFOSA Perfluorooctane sulfonamide

MeFOSA N-Methyl perfluorooctane sulfonamide

EtFOSA N-Ethyl perfluorooctane sulfonamide

Other FASA

PFBSA Perfluorobutane sulfonamide

PFHxSA Perfluorohexane sulfonamide

FASAA Perfluoroalkane sulfonamidoacetic acids

FOSAA Perfluorooctane sulfonamidoacetic acids

PFOSAA Perfluorooctane sulfonamidoacetic acid

MeFOSAA N-Methyl perfluorooctane sulfonamidoacetic acid

EtFOSAA N-Ethyl perfluorooctane sulfonamidoacetic acid

FASE Perfluoroalkane sulfonamidoethanols

FOSE Perfluorooctane sulfonamidoethanol

MeFOSE N-Methyl perfluorooctane sulfonamidoethanol

EtFOSE N-Ethyl perfluorooctane sulfonamidoethanol

PFAS ethers

PFEESA Perfluoro-2-ethoxyethanesulfonic acid

PFMBA Perfluoro-4-methoxybutanoic acid

PFMPA Perfluoro-3-methoxypropionic acid

6:2 Cl-PFAES 9-Chlorohexadecafluoro-3-oxanonane-1-sulfonate

8:2 Cl-PFAES 11-Chloroeicosafluoro-3-oxaundecane-1-sulfonate

DONA Dodecafluoro-3H-4,8-dioxanonanoate

NFDHA Nonafluoro-3,6-dioxaheptanoic acid

Other PFAS

PFECHS Perfluoro-4-ethylcyclohexanesulfonic acid

HPFHpA 7H-Perfluoroheptanoic acid

P37DMOA Perfluoro-3,7-dimethyloctanoic acid

1. Introduction

1.1 The problem with PFAS

Per- and polyfluoroalkyl substances (PFAS) is an umbrella term for a group of more than 10,000 yet identified organic molecules, contributing significantly to environmental pollution (Swedish Environmental Protection Agency 2024). PFAS are entirely synthetic chemicals, meaning they do not occur naturally in the environment. Nevertheless, PFAS have been detected in almost all environments, mainly due to their broad usage and ability to spread long distances, resulting in contamination of for example groundwater, soil and surface waters. The manufacturing and usage of PFAS has been ongoing since the 1950s and the substances can be found in a variety of products, including clothes, paints, make-up and paper packaging (Swedish Chemicals Agency 2024a). The primary point source introducing PFAS in the environment is usage of firefighting foam containing PFAS, also referred to as aqueous film forming foams (AFFFs). In Sweden, AFFFs has been used frequently by rescue services for over 50 years, though current recommendations limit their use to specific situations, such as fires in chemical industries (MSB 2024a).

The challenges regarding PFAS pollution in the environment are a direct consequence of the unique properties of the substances. The molecules consist of carbon chains of varying lengths, where the hydrogen atoms in the carbon chain have been completely or partly replaced by fluorine atoms (Swedish Chemicals Agency 2024a). Due to the strong bond between fluorine and carbon atoms, PFAS are highly persistent and resistant to degradation in the environment. Additionally, some PFAS are known to be bioaccumulative and toxic to humans and other living organisms, although there are reasons to regard all PFAS as toxic (Swedish Chemicals Agency 2024a). Knowledge and research into the health effects on humans due to PFAS exposure is continuously being developed, however it is difficult because of the vast number of PFAS variants existing (EPA 2024a). Current studies have shown that exposure to PFAS can have effects such as increased risk of cancer, developmental delays in children, reduced immune system function and reduced fertility.

1.2 Phytoremediation

Phytoremediation is a technique used to remediate mainly polluted soil and waters by utilising plants to remove, stabilise or degrade contaminants (Kafle et al. 2022). The remediation method can be divided into sub-categories based on the mechanism involved, such as phytoextraction, phytostabilisation and phytovolatilisation. Phytoextraction refers to accumulation of contaminants in the plant matrix and has been identified as potentially the most promising mechanism for phytoremediation of PFAS, due to the chemicals' recalcitrant properties (Nassazzi 2023). Phytoextraction is followed by harvesting of the plant to continuously remove the contaminants which have accumulated in the plant's biomass, followed by appropriate waste disposal (Pivetz 2001). The effectiveness of phytoremediation depends on the plant species selected, as different plants use different mechanisms and

exhibit varying affinities for contaminants, making species choice crucial for the outcome of the remediation (Kafle et al. 2022). Phytoremediation is a natural extraction method, and compared to other existing remediation techniques it is considered cost effective, making it a promising subject to further investigate (Pivetz 2001).

Phytoremediation by plants to remediate soils and waters contaminated with PFAS has been studied previously, though further research is needed to better understand the mechanisms involved in the transport and distribution of PFAS within plants (Nassazzi 2023; He et al. 2023). PFAS has been shown to accumulate in various plant types, including food crops (Felizeter et al. 2012), weeds (He et al. 2023) and woody plants (Gobelius et al. 2017), indicating potential for phytoextraction. The substances within the PFAS group have varying properties, consequently leading to different interactions with plants and their physiology, metabolism and chemical processes (Felizeter et al. 2012). As a result, different PFAS are more likely to accumulate in different parts of the plants, such as in the leaves and roots.

1.3 The PFAS problem at MSB Revinge

MSB Revinge is an organisation that offers education in the form of courses regarding civil preparedness (MSB 2024b). The property is about 45 ha in size and located in Scania, Sweden, near the city of Lund and lake Kranke (Krankesjön), see Figure 1. Among the courses that are offered, several include firefighting practices, during which AFFFs containing PFAS have been used since the early 1980s at various locations at the site (Bennermark & Thelin 2024). This has resulted in PFAS contamination of the area and consequently spreading into groundwater and surrounding surface waters, as confirmed by sampling and environmental controls. Although AFFFs containing PFAS are no longer used at the site, the PFAS contaminations remain.

Figure 1: Map showing the location of MSB Revinge's property, marked in red. ©Lantmäteriet

In 2010, a cultivation of *Salix viminalis*, henceforth referred to as “Salix”, was planted at MSB Revinge as part of a water treatment system, serving as the final treatment step for contaminated water (Nilsson & Eliasson 2010). The treatment step involves irrigation by sprinkling water into the air within the Salix cultivation, with the primary goal of removing nutrients and heavy metals from the water. Previous sampling of plant parts has found detectable concentrations of PFAS in the Salix, however a more detailed investigated is yet to be performed (Bennermark & Thelin 2024).

1.4 Aim and research question

This thesis aims to further investigate the potential for phytoremediation of PFAS through Salix as a remediation method at MSB Revinge's training facility. Furthermore, the objective is to expand the understanding of Salix’s remediation capacity regarding PFAS generally, and thus the knowledge of how to mitigate the occurrence and spread of PFAS in the environment. This study is limited to a case study, based on the firefighting training facility at MSB Revinge, but might still be able to provide support to municipal rescue services, as well as other PFAS contaminated areas, in Sweden. The research questions of this study are as follows:

1. Where does PFAS accumulate in the Salix plant?
2. Do the different types of PFAS accumulate in different zones of the plant, and can this be linked to the properties and structure of the molecule, such as the PFAS-chain length or functional groups?
3. How much PFAS is present in the Salix cultivation, including both plants, soil and groundwater?
4. Does the Salix cultivation overall contribute to a measurable reduction in total PFAS?

By answering these research questions, the aim is to assess the efficiency of the uptake of PFAS in the Salix plants, and thus this technique as a remediation method.

1.5 Scope of the study

This study is limited to one soil, groundwater and surface water sample each. The same applies for the samples for the different plant parts, with only one root, stem, bark and twig sample, respectively. Three leaf samples were collected: one from the shrubs in November, one in April and one was taken from leaves shed in the autumn that was found on the ground in March. The leaf sample from April was taken only a few weeks before the report’s hand-in date and may therefore not be as thoroughly addressed as the rest of the results. Furthermore, no reference sample was taken from an uncontaminated location to track potential PFAS cross-contamination from the fieldwork and sample handling, due to the uncertainties related to finding an uncontaminated spot.

2. Background & Theory

2.1 Per- and polyfluoroalkyl substances

2.1.1 Definitions

Today, there is no universally agreed-upon definition of PFAS. However, OECD (Organisation for Economic Co-operation and Development) has presented a definition that has gained widespread recognition and acceptance (Swedish Chemicals Agency 2024a). According to OECD, PFAS are substances with at least one methyl (-CF3) or methylene (-CF2-) carbon where the hydrogen atoms have been fully exchanged by fluorine (OECD 2021). Specifically, the fully fluorinated methyl or methylene group should not be attached to a hydrogen, chloride, bromide or iodine atom for the molecule to be defined as a PFAS.

PFAS can be classified and divided into different categories, where one way of classification was proposed by Buck et al. (2011), which was also recognised and accepted by OECD (2013). The classification divides PFAS into polymers and non-polymers (Buck et al. 2011). Polymers are large molecules composed of smaller units called monomers, which are repeated in a consistent pattern throughout the entire structure (Holmström et al. 2021). Additionally, PFAS polymers are further divided into three sub-categories, that is fluoropolymers, polymeric perfluoropolyethers (PFPE) and side-chain fluorinated polymers (Buck et al. 2011; OECD 2013). Polymeric PFAS are less studied than non-polymeric, however, they are present in the environment and still gives rise to concern (Lohmann & Letcher 2023). Polymers have been shown to be able to break down into non-polymeric PFAS, contributing to further spread of the non-polymers. While polymeric PFAS can be divided into further, more specific categories, this is not the main focus of this thesis and will thus not be explored in greater detail.

The non-polymeric PFAS can be divided into perfluoroalkyl substances and polyfluoroalkyl substances (Buck et al. 2011). ‘Perfluoroalkyl substance’ refers to molecules where all hydrogens attached to carbons have been replaced with fluorine, except for carbons affixed to a functional group. ‘Polyfluoroalkyl substance’, on the contrary, refers to molecules where not all hydrogen atoms attached to carbon have been replaced by fluorine. OECD (2013) on the other hand divided the non-polymer substances into the categories: perfluoroalkyl acids (PFAA), perfluoroalkane sulfonyl fluoride (PASF) based substances, fluorotelomers and fluorinated ethers. The division of PFAS as described by OECD (2013) is shown in Figure 2.

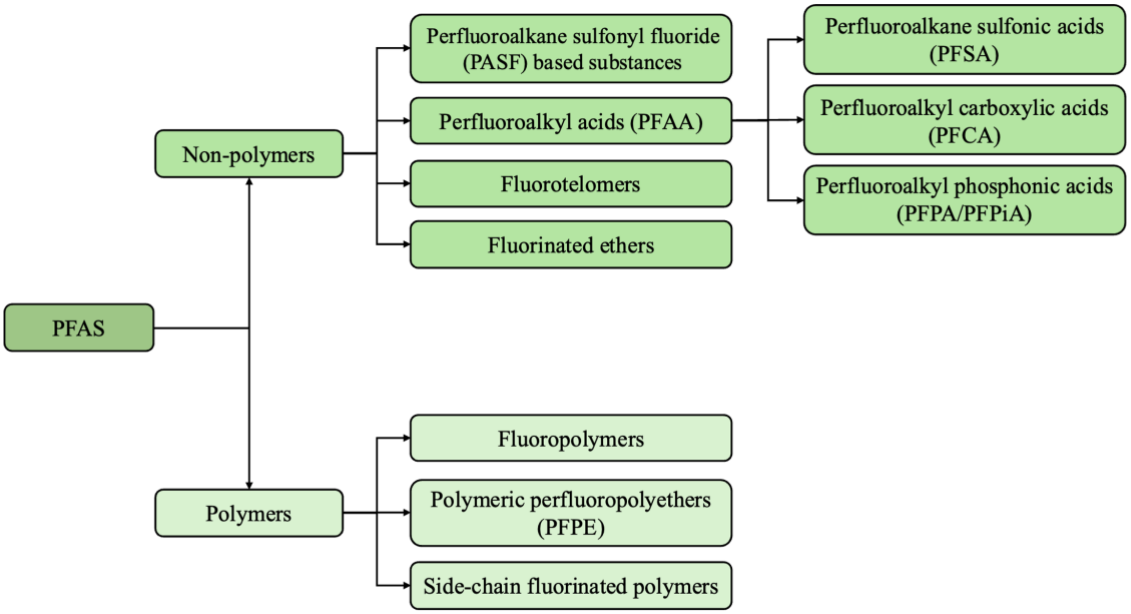
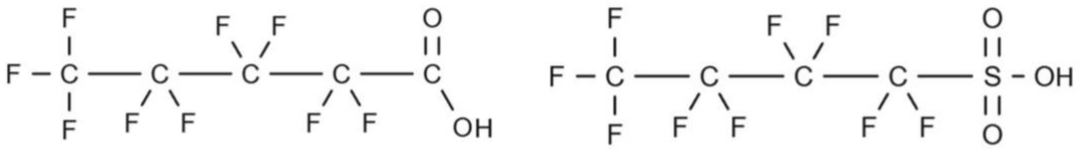


Figure 2: Classification of PFAS into several selected sub-categories. Adapted from Holmström et al. (2021).

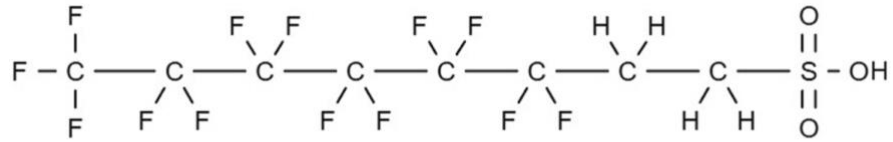
One of the main focuses of this thesis will be on the group PFAA, which can be further divided into three sub-groups, see Figure 2 (Holmström et al. 2021). Commercial laboratories are mostly able to screen PFAS with respect to two of the PFAA sub-groups, namely perfluoroalkyl sulphonic acid (PFSA) and perfluoroalkyl carboxylic acids (PFCA). PFAA are the most extensively studied group of PFAS, where the sub-groups of PFSA and PFCA have been researched for over 20 years. These sub-groups include two of the most well-known and concerning substances: perfluorooctanesulfonic acid (PFOS) and perfluorooctanoic acid (PFOA). While both PFSA and PFCA have fully fluorinated carbon chains, they differ in their functional group, where PFSA have a sulfonic acid group (-SO_2OH), and PFCA have a carboxylic acid group (-COOH), as depicted in Figure 3.



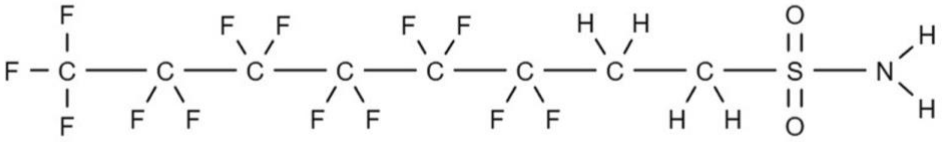
Substance	PFPeA (Perfluoropentanoic acid)	PFBS (Perfluorobutanesulfonic acid)
Group	PFCA (Perfluoroalkyl carboxylic acid)	PFSA (Perfluoroalkane sulfonic acid)

Figure 3: Chemical structure of the two sub-classes of PFAA, with the figure on the left showing an example of a PFCA and figure to the right showing an example of a PFSA.

Other sub-groups that will be mentioned and analysed during this thesis are: FASA (perfluoroalkane sulfonamides), FOSA (perfluorooctane sulfonamide), FOSAA (perfluorooctane sulfonamidoacetic acid), FOSE (perfluorooctane sulfonamidoethanol) and FTS (fluorotelomersulfonic acid). All mentioned groups are precursor PFAS, meaning that they can be degraded into other more stable PFAS (Buck et al. 2011; ITRC 2023). When placing these sub-groups into the divisions depicted in Figure 2, FTS is included in fluorotelomers (Buck et al. 2011), while the remaining sub-groups are all included in the PASF based substances (Buck et al. 2011; OECD 2022). A depiction of molecular structure examples from the FTS and FOSA group are shown in Figure 4.



Substance	6:2 FTS
Group	Fluorotelomers



Substance	PFOSA
Group	FOSA

Figure 4: The molecular structure for 6:2 FTS and PFOSA, showing both the substance name and the corresponding sub-group.

2.1.2 Physical and chemical properties

Many PFAS consist of both a hydrophobic and a hydrophilic part, where the fluorinated carbon chain is hydrophobic, and the attached functional group hydrophilic (Lyu et al. 2022). This chemical structure enables many PFAS to act as surfactants, which means that the molecules place themselves between two boundary layers, such as the interface between liquid and solid media or liquid and gaseous media (Holmström et al. 2021). Because of their surfactant properties, PFAS can rapidly form large films at the interface of boundary layers, which is the reason for the wide usage of PFAS in AFFFs. Both PFCA and PFSA are examples of PFAS containing both hydrophilic and hydrophobic parts which have been frequently used components in AFFFs (Lyu et al. 2022).

PFAS are persistent chemicals and will thus not degrade naturally in the environment into non-PFAS compounds (ITRC 2023). This property is due to the molecules' thermal and chemical stability, which in turn stems from the strong chemical bond between carbon and

fluorine. One of the key reasons for the widespread applications of PFAS is their persistence and ability to withstand harsh conditions, such as low pH and high temperatures, and examples of two of the most thermically stable types of PFAS include PFCA and PFSA (Holmström et al. 2021; ITRC 2023). Although PFAS does not degrade naturally in the environment, larger PFAS molecules can be broken down into new smaller PFAS molecules. The substances for which this occurs are called precursor, e.g. fluorotelomer alcohols can break down into the more stable PFCA, such as PFOA (Buck et al. 2011).

The solubility of PFAS in water varies among the thousands of different molecules within the group (Holmström et al. 2021). For PFAA, the solubility is closely related to the number of carbons in the fluorinated carbon chain. As the fluorinated carbon chain contributes to the hydrophobic part of the molecule, a longer fluorinated chain results in an increased hydrophobicity. Hence, the solubility of PFAA decreases as the number of carbons increases (Holmström et al. 2021; Buck et al. 2011). Moreover, the functional group of the PFAA also influences the solubility (Holmström et al. 2021). When comparing PFCA and PFSA, the carboxyl acid group (-COOH) in PFCA has been shown to result in higher water solubility than the sulfonic acid group (-SO_2OH) in PFSA.

2.1.3 Chain length and functional groups

An important way of categorising PFAS is through differentiation based on carbon chain length. This division is frequently used when referring to the group of PFAA and its sub-groups PFCA and PFSA, in particular (Buck et al. 2011). The widely used and accepted definition is that PFCA with seven or more perfluorinated carbons are “long-chained”, while PFSA are “long-chained” if they have six or more fluorinated carbons (Buck et al. 2011; Holmström et al. 2021; OECD 2013). If the molecules have less than seven and six perfluorinated carbons, respectively, they are classified as “short-chained”. Additionally, remaining PFAA substances are regarded as long-chained if they have seven or more perfluorinated carbons (Buck et al. 2011). PFAS with less than four carbons are defined as “ultrashort-chained” (Liang et al. 2023), and the divisions are depicted in Figure 5. Ultrashort-chained PFAS have not been as widely researched as longer chained compounds, although Barzen-Hanson and Field (2015) managed to demonstrate that these ultrashort-chained molecules exist in some AFFFs, thus indicating spreading of ultrashort-chained PFAS in the environment as well.

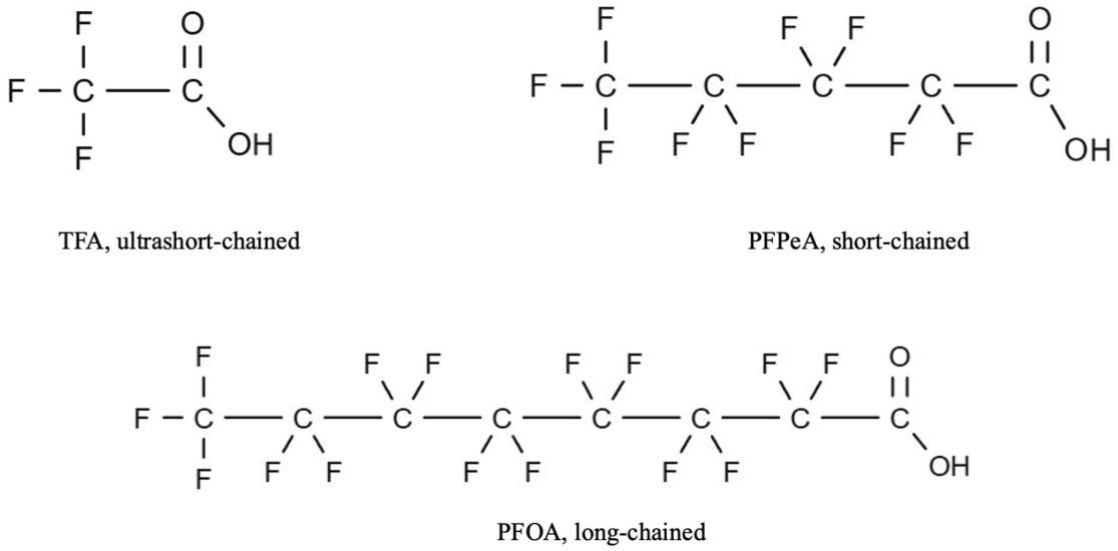


Figure 5: PFCA with different fluorinated carbon chain lengths; ultrashort-chained trifluoroacetic acid (TFA), short-chained perfluoropentanoic acid (PFPeA) and long-chained PFOA.

The attached functional group contributes to the physicochemical properties of PFAS (ITRC 2023). As previously mentioned and illustrated in Figure 3, common functional groups are carboxyl acids and sulfonic acids, however, other possible examples include phosphates and amines. PFAS can also be categorised based on the potential charge of the functional group, with the four categories: anionic, cationic, zwitterionic or non-ionic. Anionic PFAS have one or several acidic functional groups that are prone to release hydrogen ions, while cationic PFAS have one or several basic functional groups that can retain hydrogen ions. Zwitterionic PFAS have functional groups that can both form anions and cations, and non-ionic PFAS do not form ions at all (ITRC 2023). According to Nickerson et al. (2020), most research regarding pollution in the environment through AFFFs have focused on anionic PFAS, such as PFSA and PFCA. However, the same study found high amounts of cationic and zwitterionic PFAS in the soil by a firefighting training site.

2.1.4 Sources of PFAS

There are numerous sources releasing PFAS into the environment, and Gleisner et al. (2019) defined the top sources contributing to PFAS pollution to be usage of AFFFs, landfills, wastewater treatment plants and industries such as paper, plastic and textile factories. Out of these sources, the usage of AFFFs constitutes the biggest source of PFAS to the environment in Sweden (Swedish Chemicals Agency 2024a). AFFFs have historically both been used at fire training facilities as well as during fire rescue missions at scenes of accidents. Some fire training facilities have systems where extinguishing water is collected and treated, however, due to PFAS' persistent properties these treatment processes are seldom efficient in reducing PFAS concentrations (Gleisner et al. 2019).

Landfills also contribute to PFAS pollution through release of leachate, which can contain complex mixtures of PFAS due to the variety of materials present in the landfill (Gleisner et al. 2019). Wastewater treatment plants are another source due to insufficient PFAS removal techniques, resulting in PFAS-release with the treated water. Moreover, PFAS can be accumulated in the residual sludge formed in the treatment plant, which is frequently spread on agricultural lands, hence inducing further contamination of the environment (Lyu et al. 2022).

2.1.5 Transport and fate of PFAS

The transport and spreading of PFAS can be complex to predict and will depend on the varying properties of the numerous molecules within the PFAS group, as well as the geological and hydrogeological setting in which the substances are present (Holmström et al. 2021). PFAS in soil will adhere to the soil particles through adsorption, with the degree of adherence varying depending on the characteristics of the soil and the specific PFAS. An important parameter affecting the adsorption of PFAS onto soil is the organic content, where a higher organic content generally results in more PFAS being adsorbed (Hubert et al. 2023; ITRC 2023).

A way of quantifying the relationship between the concentration of molecules adsorbed to soil and the concentration in water is through the solid/liquid partitioning coefficient K_d, where a high K_d-value indicates more adsorption onto soil (Lyu et al. 2022). Hubert et al. (2023) showed that for long-chained PFAS, specifically PFCA, PFSA and FTS, the K_d-value increased with increasing perfluorinated chain length, supporting that longer chained PFAS are more prone to adsorb to soil than shorter chained PFAS. This can be explained by the hydrophobicity of PFAS increasing with the number of perfluorinated carbons, leading to a higher ratio being adsorbed onto the soil (Hubert et al. 2023; Lyu et al. 2022). Furthermore, in a study conducted by Sörengård et al. (2020), looking into sorption tendencies for 17 different PFAS with varying chain lengths to different inorganic and organic sorbents, it was found that the sorption strength depended on the chain length, where short-chained PFAS tended to exhibit weaker sorption than long-chained. The functional group also influences the K_d-value of PFAS (Holmström et al. 2021; Hubert et al. 2023). For example, the sulfonic acid group of PFSA gives the substances higher K_d-values than the carboxyl acid group present in PFCA, implying that the PFSA will be adsorbed to soil to a higher degree. Sörengård et al. (2020) also found that the sorption strength of PFAS to inorganic and organic sorbents depended on their functional groups, and more specifically the results showed that FOSA displayed the strongest sorption, followed by PFSA, PFCA and lastly FTS. The ionic charge of the functional group of PFAS also affects adsorption, and Xiao et al. (2019) could demonstrate that cationic and zwitterionic PFAS were more prone to adsorb to negatively charged soils than anionic PFAS. PFAS in soil can leach down to the groundwater when the molecules are dissolved in water and not adsorbed to the soil. Consequently, PFAS with low K_d pose a higher risk of contaminating groundwater through leaching.

The Swedish Geotechnical Institute (SGI) developed, by reviewing existing studies, standard values for the partitioning coefficient K_d between soil and water for four PFAS (Pettersson et al. 2024). The values are as follows: 14 L/kg for PFOS, 5 L/kg for PFOA and PFNA and 4 L/kg for PFHxS. These values are representative for Swedish standard soils, with a 1-3 % carbon content and a pH between 5 and 7. K_d is the ratio between the concentration of the contaminant adsorbed to the soil and the concentration in the pore water. However, site-specific K_d for contaminants can also be calculated by dividing the measured concentration in the soil with the concentration observed in the groundwater (Naturvårdsverket 2009). Henceforth, this way of calculating the site-specific K_d will be referred to as K_d*. A drawback of using this method to estimate the partitioning coefficient is that the contaminant concentration in the groundwater can be a result of contamination from a wider area. Moreover, for organic contaminants, K_d can be calculated from the contaminant's K_{oc} value and the organic content in the soil (f_{oc}), according to Equation 1.

$$K_d = K_{oc} * f_{oc} \quad (1)$$

Transport of PFAS in groundwater can mainly be explained by three mechanisms: advection, molecular diffusion and mechanical dispersion (Fetter 2014). Advection refers to the transport of pollutants with the average linear velocity of the groundwater flow. Molecular diffusion is the mechanism where pollutants are transported against the concentration gradient, meaning that the molecules move from areas with high concentrations to lower concentrations, and mechanical dispersion describes the transport and spreading caused by small scale differences in the velocity of the waterflow. Together, these mechanisms explain how PFAS and other pollutants spread through groundwater. Furthermore, the efficiency of these mechanisms is affected by the properties of the groundwater and of the surrounding geological conditions, such as hydraulic conductivity and hydraulic gradient (Fetter 2014). Additionally, groundwater can discharge into surface water, further inducing spreading of PFAS and negative impact on the environment.

Airborne transport is another pathway for PFAS, causing a spreading of PFAS to areas far away from the original release source (Hansson et al. 2016). PFAS can be transported in air with the direction of the wind by existing in a gaseous phase or being sorbed to aerosols (ITRC 2023). Through either wet deposition (precipitation) or dry deposition, airborne PFAS can be deposited onto surfaces such as soil and water. Filipovic et al. (2013) studied the inputs of PFAA into the Baltic Sea, comparing ways of input such as atmospheric deposition, wastewater treatment plant inflow and inflow from rivers. The authors concluded that atmospheric deposition of PFAA is one of the major inputs into the Baltic Sea, giving rise to significant pollution.

The volatility of PFAS varies, where neutral precursor PFAS are one of the more volatile PFAS groups and hence often found in gaseous phase (ITRC 2023). Ahrens et al. (2012) conducted a study where air samples from a suburban area in Toronto were analysed to determine the concentration of gaseous PFAS in the ambient air. The study analysed seven groups of PFAS, including both PFAA precursors and PFAA. They could conclude that 80 %

of the gaseous phase PFAS mass at the sampling area consisted of FTOH (fluorotelomer alcohols), a neutral precursor. PFAA are non-volatile, hence their main airway travel is by sorption to aerosols (Holmström et al. 2021).

Certain PFAS can break down to smaller, more stable PFAS molecules (Holmström et al. 2021). Precursors can be both polymer and non-polymeric PFAS, and degradation can occur both in the environment and inside biological material. Precursor PFAS are commonly broken down into various PFAA, that undergoes no further natural transformation. FTS are examples of precursors that can be broken down into PFCA and have been detected in AFFFs and at sites where AFFFs have been used. Another example is FOSA, which can be degraded into PFOS (Holmström et al. 2021).

2.1.6 Human and environmental concern

Most studies on the toxic effects of PFAS, regarding both human health and environment, have focused on PFAA, specifically PFOS and PFOA (Holmström et al. 2021). Humans are exposed to PFAS through various pathways, including drinking water, consuming certain foods, personal care products and household products such as non-stick pans (EPA 2024a). Determining the health effects of PFAS is challenging due to the vast number of PFAS existing, each with unique molecular structures and properties, and the range of exposure ways. Nevertheless, EPA has reviewed existing research and identified several potential health risks associated with PFAS exposure. Examples of potential effects include reduced fertility in women, increased risk of certain cancers (such as kidney cancer), elevated cholesterol levels, low birth weight in children and reduced immune system function (EPA 2024a). Despite these findings, further research is required on a broader range of PFAS, beyond just PFAA, to fully understand their impact on human health.

Some PFAS are recognised to have toxic effects on the environment, potentially causing significant harm for both animals and plants (Holmström et al. 2021). Studies conducted on mammals and birds exposed to PFAA have demonstrated adverse health effects, including weakened immune system, liver damages, developmental issues in foetuses and hormonal disruptions. Dennis et al. (2020) conducted a study investigating the effects of PFHxS and PFOS exposure through drinking water for the bird species northern bobwhite quail. The study revealed that the female birds during reproduction gained less weight than those not exposed to PFAS, thus indicating development issues for chicks. Moreover, some PFAS are known to bioaccumulate, meaning that these compounds will accumulate up the food web, resulting in high concentrations in top predators (Holmström et al. 2021). Polyfluoroalkyl substances have been found to be less prone to bioaccumulate compared to perfluoroalkyl substances, where for instance, long-chained PFAA have been shown to be bioaccumulating.

2.1.7 Regulations and legislation

There are no regulations and legislations regarding PFAS as a group, however regulations for some specific PFAS exists (Swedish Chemicals Agency 2024a). The Stockholm Convention is a global convention with the goal of restricting persistent organic pollutants (POP) (Swedish Chemicals Agency 2024b). The countries that have signed the Stockholm Convention have agreed on implementing action plans to restrict or remove the continuous release of POPs. PFAS covered by the POP list include for instance PFHxS, PFOA and PFOS.

Sweden has established limit values regarding certain PFAS and PFAS groups in waters, soil and biota. For drinking waters, the limit values are: 4 ng/L for PFAS4 and 100 ng/L for PFAS21 (Livsmedelsverket 2025). These values are based on EU's food safety authority guidelines and are expected to be implemented by the first of January 2026. For inland surface waters the current limit value is 90 ng/L with regards to PFAS11 (Havs- och vattenmyndigheten 2019). Specific limit values exist for PFOS, with the yearly average limit value for PFOS in inland surface waters set at 0.65 ng/L. In biota, such as fish, the limit value is 9.1 µg/kg ww (wet weight). Additionally, SGI has brought forward preliminary limit values regarding PFOS in soil and groundwater, with a value of 45 ng PFOS/L for groundwater (Pettersson et al. 2015). Soil is divided into two categories, sensitive land use (for instance a kindergarten) and less sensitive land use (for instance a gas station). Preliminary limit values for sensitive land use are 3 µg PFOS/kg dw (dry weight), and 20 µg PFOS/kg dw for less sensitive land use.

Implementing stricter guidelines regarding levels of PFAS in water, soil and biota is an important aspect in addressing the PFAS problem. However, PFAS can still be spread from contaminated sites years after the original PFAS-emitting activity (Filipovic et al. 2015), and if all emissions terminated today, the chemicals would still remain in the environment for decades (ECHA n.d.). Thus, remediation is required to manage the issues related to PFAS.

2.2 Remediation of PFAS

PFAS' stability, persistence, physicochemical properties and prevalence in complex matrices makes the task of remediating soil and water from the chemicals challenging (Bolan et al. 2021; Mahinroosta & Senevirathna 2020). In a study by Mahinroosta & Senevirathna (2020), different in-situ and ex-situ remediation techniques for PFAS in soil was reviewed, and the researched techniques were categorised into three sub-groups: 1) sorption and stabilisation, 2) destruction technologies and 3) separation technologies. Sorption and stabilisation methods involves addition of amendments that enhances sorption of PFAS in the soil. Consequently, PFAS is immobilised, and the risk of spreading is lowered. The sorbents decrease the concentration of PFAS in liquid media, thus making the compounds less mobile. Examples of additives to enhance immobilisation are biochar and activated carbon (GAC - *granular activated carbon* or PAC - *powdered activated carbon*) (Sörengård et al. 2019; Sörengård et al. 2021; Sørmo et al. 2024).

The second category, destruction technologies, involves destruction of PFAS (Mahinroosta & Senevirathna 2020). Examples of destruction techniques includes thermal treatment and bioremediation using microorganisms. The category is appealing since it offers the possibility to break down PFAS or transform the compounds into less deleterious products, however, the strong bonds between carbon and fluorine in PFAS limits the options for remediation through destruction technologies. The third category, separation technologies, include remediation methods that mobilise PFAS from the contaminated soil or water to concentrate elsewhere, and examples of techniques are soil washing and foam fractioning. Another separation technology is phytoremediation, where PFAS can be separated from the media and concentrated in the plants' biomass. The process is slow, but implementation and maintenance costs are relatively low, and Bolan et al. (2021) suggests that it is the most economically and ecologically sustainable method for PFAS remediation at present day.

Various PFAS behave differently, e.g. depending on their chain length, and consequently different remediation methods are more suitable for treating different PFAS. Short-chained PFAS generally are more mobile and soluble than long-chained and have also been found to be taken up and accumulate to a higher degree in plants (Jiao et al. 2021). Consequently, phytoremediation might be more effective against short-chained PFAS than long-chained. The fact that a major challenge with PFAS is their occurrence in complex mixtures makes it difficult to single out one suitable treatment method, highlighting the need for a combination of techniques targeting PFAS with different properties (Gleisner et al. 2019). LIFE SOuRCE is an example of a European project with the aim of assessing and demonstrating the potential and possibilities for PFAS remediation techniques for polluted groundwater (LIFE SOuRCE n.d.a). The project uses a combination of the remediation techniques surface foam fractioning, anion exchange resins, electrochemical oxidation and phytoremediation. The goal is to, through the complementary benefits from each technique, be able to remove >99% of long-chained and >95% of short-chained PFAS (LIFE SOuRCE n.d.b).

2.3 Phytoremediation of PFAS

Phytoremediation refers to the utilisation of plants to remediate contaminated media, such as soil, sludge, sediment, water and air, from both organic and inorganic contaminants in-situ (Arthur et al. 2005). Phytoremediation does not only involve a single mechanism, but a range of them, with some examples being accumulation (phytoextraction), degradation (phytodegradation), dissipation (phytovolatilisation), immobilisation (phytostabilisation) and degradation through enhanced microbial activity (phytostimulation) (Arthur et al. 2005; Nassazzi 2023; Pivetz 2001), see Figure 6.

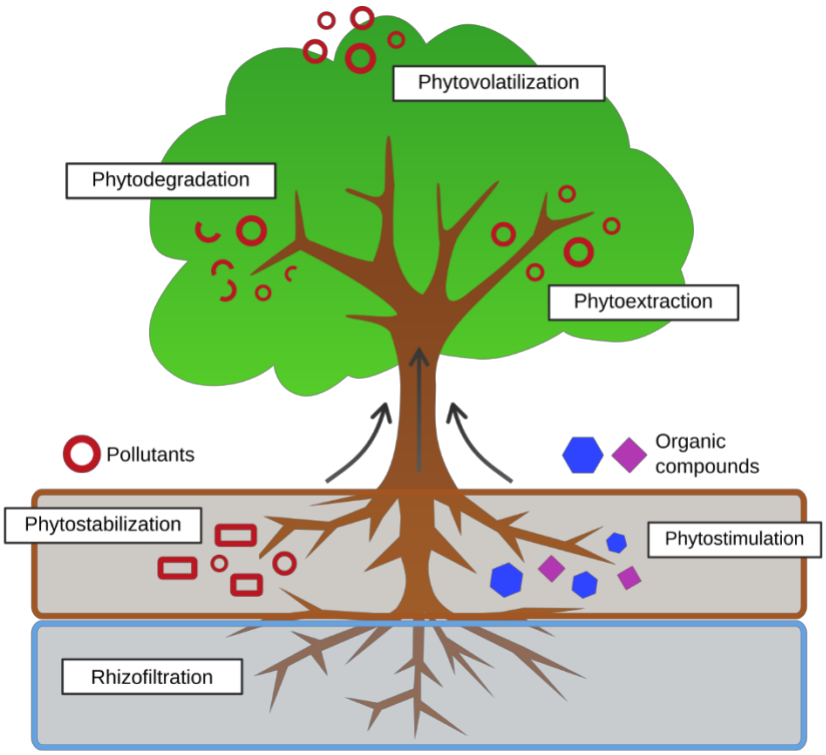


Figure 6: Illustration of different mechanisms for phytoremediation (Townie 2016) CC BY-SA 4.0 (https://creativecommons.org/licenses/by-sa/4.0/deed.en).

The acting mechanisms that the plant uses for remediation varies and depends on e.g. the type of contaminant, the surrounding and plant type (Kafle et al. 2022). For instance, phytodegradation has been shown to be a significant mechanism for phytoremediation of organic pollutants (Arthur et al. 2005). PFAS are a group of organic compounds, however, it is resistant to degradation due to its strong carbon-fluorine bonds, making phytoextraction a more prominent mechanism for phytoremediation of PFAS (Huff et al. 2020; Mahinroosta & Senevirathna 2020; Nassazzi 2023).

An established method of estimating the potential for phytoremediation is through calculating the *bioconcentration factor* (BCF), which is defined as the ratio between the contaminant concentration in a certain plant part to the ambient media (e.g. soil or groundwater), see Equation 2 (Huff et al. 2020; Mishra & Pandey 2019):

$$BCF_{plant} = \frac{C_{plant}}{C_{media}} \quad (2)$$

where C is the concentration in ng/kg ww for plants and solid media, and in ng/L for liquid media (EPA 2024b). When the BCF value > 1, it means that the concentration of the pollutant is higher in the plant tissue than in the media, indicating potential to accumulate the contaminant (Huff et al. 2020). A value > 10 indicates that the plant has potential to hyperaccumulate the contaminant, although there are multiple definitions for the phenomenon.

Another way of estimating a plant's potential for phytoremediation is by calculating the *translocation factor* (TF), which is defined as the relationship between contaminant concentration in the roots to the shoots of a plant (Kafle et al. 2022; Yoon et al. 2006).

2.3.1 Plant uptake and translocation

The main pathway for PFAS acquisition in plants is through soil and water via the roots, and examples of primary sources for PFAS accumulation in plants include irrigation of fields with contaminated water and adding of biosolids (Wang et al. 2020). The uptake can occur both through active and passive transport, meaning it can either be energy-dependent or independent, and the specific mechanisms can vary depending on which PFAS that are being taken up (Wen et al. 2013). For instance, when Zhang et al. (2019) investigated the uptake mechanisms of PFAS in plants, more specifically PFCA (TFA, PFPrA, PFBA, PFHxA and PFOA) and PFOS in wheat, the authors concluded that the uptake mainly was an active process. Also, aquaporins (water channels) and anion channels i.e. proteins enabling transport of water and anions across the cell membrane, were found to participate in the uptake process of TFA and PFPrA, that is the ultrashort-chained PFAS with two and three carbon atoms, respectively. Moreover, in a study on maize, acquisition of PFOS and PFOA was found to follow different uptake pathways (Wen et al. 2013). PFOS was concluded to be taken up through a carrier-mediated passive process, aided by aquaporins and anion channels. In contrast, the PFOA uptake was demonstrated to be an active process.

Apart from root uptake, PFAS also have the potential to be acquired through above-ground plant parts by uptake of aerially transported PFAS, and in a study by Tian et al. (2018) it was suggested that plants can obtain PFAS by foliar uptake. Furthermore, in a recent study on pak choi and radish, it was concluded that uptake of PFAS through leaves is a pathway that should not be dismissed (Xu et al. 2024). However, in a literature study by Wang et al. (2020) on plants' uptake and accumulation of PFAS, it was suggested that only small to negligible amounts of PFAS generally can be attributed to uptake from aerial transport. In other words, there are uncertainties and conflicting views regarding the significance of uptake of aerially transported PFAS in plants, and the varying perspectives highlights the needs for further research.

PFAS can be obtained from soil or water by the roots through the apoplastic and symplastic pathway, i.e. the extracellular and intracellular transportation for water and nutrients, after which it can be translocated through the xylem up to the shoots (Costello & Lee 2020; Nassazzi 2023; Wang et al. 2020). PFAS can also be translocated to other plant organs through the phloem, which main purpose is to transport organic compounds within the plant (Xu et al. 2022). Both uptake and transport of different PFAS congeners in plants has been found to relate to their physicochemical properties, such as their lipophilicity/hydrophilicity (K_{OW}-value) and size, which in turn e.g. relates to the perfluoroalkyl chain length, functional group and if the chain is branched or linear (Jiao et al. 2021; Qian et al. 2023; Xu et al. 2022). High lipophilicity will limit the solubility of the PFAS in water and thus inhibit the capability to translocate upwards in the plant, and at the same time increase their sorption potential to

soil particles (K_d) and plant roots. Furthermore, studies have hypothesised how the ability for PFAS to pass the Casparian strip, a cellular structure in the plant root, decreases with longer chain length due to lower hydrophilicity and bigger size, inhibiting translocation of long-chained PFAS to the shoots (Felizeter et al. 2012). Accordingly, PFAS with short chains have been found to accumulate to a greater extent in the shoots and plant foliage, while those with longer chains tend to accumulate in the plant roots (Krippner et al. 2014; Qian et al. 2023). While multiple studies have investigated PFAS uptake and transport mechanisms in plants, the overall process remains not fully understood (Nassazzi 2023; Wang et al. 2020).

Uptake and accumulation of PFAS in plants can also depend on the functional group, and in a study by Sharma et al. (2020), where the PFAS accumulation potentials for three Salix species were investigated, it was found that PFSA accumulated less than PFCA for PFAS with the same number of carbon atoms. It was found that PFOA concentrations were higher than PFOS in the leaves, but PFOS concentrations higher than PFOA in the roots. The same pattern was found in a study on PFAS accumulation potential in maize, where PFCA was found in higher concentrations than PFSA in the plant (Krippner et al. 2015). Furthermore, in a literature review conducted by Wang et al. (2020), it was stated that overall, PFCA are more prone to be accumulated in plants than PFSA.

Disregarding the physicochemical properties of individual PFAS, studies have indicated that the total PFAS concentration tends to be highest in leaves (Gobelius et al. 2017). An explanation for the elevated PFAS accumulation in foliage is that most of the water that is transported upward in the plants exits by evapotranspiration through the leaves, resulting in high PFAS concentrations (Stahl et al. 2013).

2.3.2 Conditions and supplements enhancing uptake of PFAS

The plant uptake efficiency does not only depend on the physicochemical properties of the PFAS and the plant physiology, but also on the abiotic and biotic conditions of the surrounding, including e.g. pH, soil organic matter (SOM), salinity, temperature, microbial activity and the concentration of the contaminant in surrounding media (Kafle et al. 2022; Stahl et al. 2009; Wang et al. 2020). The sorption of PFAS to soil has been found to relate to the pH, and Campos-Pereira et al. (2023) observed when investigating six different PFAS in 11 soils that the PFAS sorption was inversely related to the pH for all PFAS examined. The log K_d-value decreased as the pH increased, meaning that the sorption to the soil decreased. Furthermore, the slope steepened with longer perfluorocarbon chain length, indicating that the sorption of long-chained PFAS is more sensitive to changes in pH than short-chained PFAS. Similarly, Nguyen et al. (2020) also found that the K_d-values increased with a decreasing pH, when investigating a range of PFAS. There are also studies that have investigated how pH affects the potential for phytoremediation. For instance, Krippner et al. (2014) looked at PFAS uptake and distribution in maize and its dependency on pH, using nutrient solutions with pH 5, 6 and 7, attempting to imitate the pH in European agricultural soils. The study observed a relationship in plant uptake of PFDA, seeing a significantly higher uptake at pH 5 than pH 7, however, the PFBA and PFHxA showed the opposite

relationship with a significantly higher uptake at pH 7 compared to pH 5. The remaining PFAS studied (PFPeA, PFHpA, PFOA, PFNA, PFBS, PFHxS and PFOS) showed no relationships between uptake rate and pH. Furthermore, in a study on wheat, Zhao et al. (2013) reported a pH dependency regarding PFOS uptake, where the uptake peaked at pH 6, where a pH range of 4-10 was investigated. To conclude, studies have observed relationships between PFAS sorption in soil and pH as well as relationships between plant uptake of PFAS and pH. However, regarding the phytoremediation potential, some results have been inconsistent, and it seems like there is a possibility that different pH conditions favour uptake of different PFAS.

The soil organic matter (SOM) is a key factor affecting PFAS sorption onto soil particles and soils with high total organic carbon (TOC) content have been shown to sorb PFAS approximately 100 times more effectively than soils with low TOC content, as demonstrated by Sørmo et al. (2021) when investigating soils with TOC 34.2 % and 1.6 %. Consequently, increased sorption of PFAS to soil particles reduces the fraction dissolved in water and thus the PFAS available for plant root uptake (Wang et al. 2020). When Zhao et al. (2016) compared PFAA in peat and farmland soil, they found that a higher SOM content led to greater PFAA sequestration, consequently leading to a reduced bioavailability for PFAA. Moreover, in a study on uptake of PFAS through lettuce and carrots, it was found that higher TOC resulted in lower BCFs (Bizkarguenaga et al. 2016).

Salinity can potentially affect the phytoremediation efficiency and a study conducted by Zhao et al. (2013) suggested higher accumulation of PFOS in wheat with increased salinity. Moreover, elevated temperatures can enhance the potential of uptake of contaminants, due to increased transpiration and metabolism (Wang et al. 2020). However, in the cases of both salinity and temperature, the two factors can also impose stress on plants and thereby potentially lead to a decrease in uptake of contaminants instead (Lippmann et al. 2019; Munns & Tester 2008; Stofberg et al. 2015).

It is possible to enhance phytoremediation processes by addition of amendments, such as sewage sludge or microbial supplements (Kafle et al. 2022). Amendments can for instance promote plant growth or stimulate phytoremediation mechanisms. There have been studies that have indicated that the adding of amendments generally are more favourable for the phytoremediation mechanism phytostabilisation rather than phytoextraction, where the latter has been identified as potentially being the main mechanism for remediation of PFAS (Nassazzi 2023). Hence, it is of importance to base the decision of which amendment that should be applied upon which contaminant one wants to remediate, and thus on which phytoremediation mechanisms expected of the plant.

Amendments to increase phytoremediation of heavy metals have been subject to many studies, however, fewer have been conducted with regards to PFAS. Nassazzi et al. (2023) conducted pot experiments where inorganic fertiliser and microbial supplements were added to sunflower, mustard and hemp seedlings, to study the amendments' effects on the plants' potential for phytoextraction. The experiment could not observe any significant effect from inoculation with microorganisms, although, due to microbial additives' potential effect on e.g.

soil fertility, it is believed that this type of supplement can influence the phytoremediation potential of PFAS. In the same study, the adding of fertilisers was found to lower the PFAS concentration in the plants, however, the plants also gained biomass, resulting in a similar total PFAS uptake in the pot with fertiliser as for the control pot without (Ibid.).

In another study, the effect of microbial supplements and phytohormones on willow and poplar was investigated by Nassazzi et al. (2025). Similarly to the previous study, no relationship between inoculation with microorganisms and PFAS concentrations in the plants was observed. For the hormones, no significant pattern was observed for both species and while the application of hormones led to an increase in concentration of PFAS in poplar, no significant change in willow was noticed. Nassazzi (2023) emphasises the need for further studies to better comprehend the impacts of amendments on phytoremediation of PFAS.

Soil alterations may not only be applied to enhance phytoremediation potential, but also to support the overall remediation of PFAS. For instance, since phytoremediation has been proven to mainly be effective towards short-chained PFAS, it theoretically could be beneficial to combine the remediation method with techniques mainly targeting long-chained congeners. Biochar is a carbon-rich material produced through pyrolysis of biomass, with the potential to sorb primarily long-chained PFAS (Liang et al. 2024). Thus, the hybrid solution of using a combination of biochar and phytoremediation could be an interesting approach in scenarios where the PFAS is present in complex mixtures. The combination of phytoremediation and biochar has been shown useful in attempts of remediating heavy metals (Paz-Ferreiro et al. 2014), but has to our knowledge not been subject to many studies regarding PFAS. It is possible that biochar could result in an increase in phytoremediation, by promoting growth through improved soil fertility (Pettersson & Enell 2021). Although, it can also decrease phytoremediation by sorption of PFAS to the biochar, decreasing its bioavailability.

2.3.3 Salix's and other plants' potential for phytoremediation

The *Salicaceae* family, in which the genus *Salix* is included, consists of approximately 300 plant species (Hollsten et al. 2013). Salix grows naturally across all continents, except Australia, and about 30 species can be found in Sweden. The focus in this study is the species *Salix viminalis,* which is one of the more commonly cultivated Salix species in Sweden. Salix is typically grown as a bioenergy crop (short rotation coppice), where the shrubs are harvested and incinerated for energy production. These Salix species are fast-growing plants which can be cut down to stumps and regenerate, making it well suited for both biofuel production and potentially phytoremediation. The regenerative ability allows Salix to accumulate biomass quickly, which is ideal for these types of applications (Hollsten et al. 2013).

Nassazzi et al. (2025) deemed Salix as a suitable choice for phytoremediation of PFAS, due to properties such as fast growth rate, high biomass production and ability to grow in various soil conditions. Further, SGI brought forward a few criteria to consider when choosing a plant

species for phytoremediation (Pettersson & Enell 2021). The criteria include that the plants should take up large amount of water, produce large biomass, be easy to manage, thrive in the climate, be non-invasive, allow regular above-ground harvest and retain its leaves to prevent further spreading of PFAS through deposition of the leaves.

One way to assess a plants potential for phytoremediation is by determining if the plants are accumulating, or even hyperaccumulating, contaminants, as described in section *2.3 Phytoremediation of PFAS* (Huff et al. 2020). Some studies regarding Salix' ability to be hyperaccumulators for certain PFAS have been conducted, however with varying results. Huff et al. (2020) investigated the species *Salix nigra* and found that Salix was a hyperaccumulator (BCF > 10) for five out of the six PFAS investigated. Furthermore, in a study by Nassazzi (2023) the phytoextraction potential for PFAS using *Salix miyabeana* was examined, where Salix was shown to only hyperaccumulate two out of the 10 investigated PFAS.

One important factor in achieving an effective phytoextraction is determining the optimal number of years between harvests. When Salix is cultivated as an energy crop, the recommendation according to Hollsten et al. (2013) is to harvest the Salix approximately every three to five years, or when stems with a diameter of six centimetres at 30 centimetres height can easily be found. Additionally, when the Salix is newly planted the first harvest should be done after approximately four and a half years. In 2023, Skirfors presented preliminary results from a pilot study, as part of the LIFE SOuRCE project. In the study, phytoremediation of PFAS through different plants, amongst others the Salix types *Wilhelm* and *Loden*, were investigated during a period of three years. Skirfors estimated that the uptake of PFAS increased each year, with a PFAS uptake of 22.5 mg/ha after one year, 95.5 mg/ha after two years and 189.3 mg/ha after three years (Skirfors 2023). Thus, indicating that a harvest of Salix after three years is more beneficial regarding phytoextraction than a harvest after one year. However, Skirfors did not mention how the accumulation of PFAS evolved after three years. Furthermore, the presentation highlighted the importance of implementing a leaf collection system during leaf shedding, as the leaves were found to contain high concentrations of PFAS.

Salix cultivations have a lifetime of about 20-25 years and if the plan is to cultivate new Salix, it is appropriate to plant another crop for a few years in between (Hollsten et al. 2013). Although, the recommendations regarding lifetime do not take the phytoremediation potential of PFAS into consideration, but rather only looks at Salix in terms of being an effective energy source. However, since a valuable characteristic of plants suitable for phytoremediation is fast growth and rapid biomass gain, it is a fair assumption that also the potential for phytoremediation decreases when the production of the cultivation decreases.

The phytoremediation potential of PFAS by plants other than Salix has also been studied. Nassazzi (2023) conducted pot experiments comparing the phytoextraction potential of PFAS for five different plants (sunflower, hemp, mustard, willow and poplar). The soil which the plants grew in was spiked with ten PFAS and the accumulation in the plants were measured after 90 days of growth. Nassazzi found that willow and sunflower had the highest

accumulation rate for the ten PFAS measured, and overall, all examined plants showed that the accumulation rate was highest for short-chained PFAS. Further, Huff et al. (2020) investigated the phytoextraction potential of seven woody plant species, among them the Salix species *Salix nigra*, for six different PFAS. The examined Salix showed potential for phytoextraction with the highest BCF of all seven plants, however in terms of the highest mass of PFAS accumulated, the birch species *Betula nigra* performed best. Gobelius et al. (2017) investigated the PFAS accumulation at a firefighting training facility, examining both various tree species and herbaceous plant species, however Salix was not examined. The authors suggested that the best option for phytoremediation of PFAS, based on investigated species, was a combination of coppicing birch and spruce trees together with mowing of an understory of herbaceous plants. In this case ground elder was suggested.

The plants physiological properties will affect the potential for phytoremediation of PFAS. One important factor affecting the phytoremediation is the transpiration rate of plants, where a higher transpiration rate will lead to more short-chained PFAS accumulating in the plant compared to a plant with a lower transpiration rate (Wang et al. 2020). Further, the protein and lipid content of plants has shown to affect the accumulation of PFOS and PFOA in the roots (Wen et al. 2016). A higher protein content in the plant correlated with a higher accumulation of both PFOS and PFOA in the roots, while a higher lipid content on the contrary resulted in a lower accumulation of PFOS and PFOA.

2.4 Site description

The investigation area encompasses a 5,000 m^2 Salix cultivation located within the property *Lund Revinge 1:14* in Revinge, a settlement 20 km east of Lund, see Figure 1. The property is estimated to have a total area of 45 ha, and the training field, within which AFFFs frequently have been used, is estimated to be 20 ha. The closest recipients include Kävlingeån, which borders the property in the north, and Ålabäcken, bordering in the east. Ålabäcken runs from lake Kranke (Krankesjön), located south of the property, and discharges into Kävlingeån. Kävlingeån in turn runs in a westward direction, originating from lake Vomb (Vombsjön) in the east, and ultimately discharges into Öresund in Lomma bay (Lommabukten). In addition to Kävlingeån, agricultural lands are situated to the north of the property, while wooded areas and open fields are found to the south. To the west, the settlement of Revinge, inhabiting approximately 500 residents, is located, and to the east the Swedish Armed Forces' *Södra Skånska Regementet – P7* (Försvarsmakten 2023; Lunds kommun 2024; Thelin 2024).

In the westernmost part of the property, multiple buildings, including for instance classrooms and offices, are located (Thelin 2024), see Figure 7. However, the biggest areal of the property is occupied by the training field, equipped to accommodate a variety of different exercises, such as simulated fire scenarios, smoke diving and traffic accidents (MSB 2012). The Salix cultivation is situated in the easternmost part of the property, adjacent to the training field and near the point where Ålabäcken discharges into Kävlingeån. The cultivation was planted in connection with the implementation of the water treatment system at MSB Revinge, as a final treatment step for extinguishing water (Nilsson & Eliasson 2010).

Initially, the Salix had the main purpose of treating the water with respect to nutrients and heavy metals, not PFAS. Nevertheless, the sprinkler system has since been feeding the cultivation with PFAS, and measurable levels of PFAS have been detected in the Salix during previous sampling. Further details on the treatment system and the Salix cultivation can be found in the sections *2.4.3 Water treatment system* and *2.4.4 The Salix cultivation*.

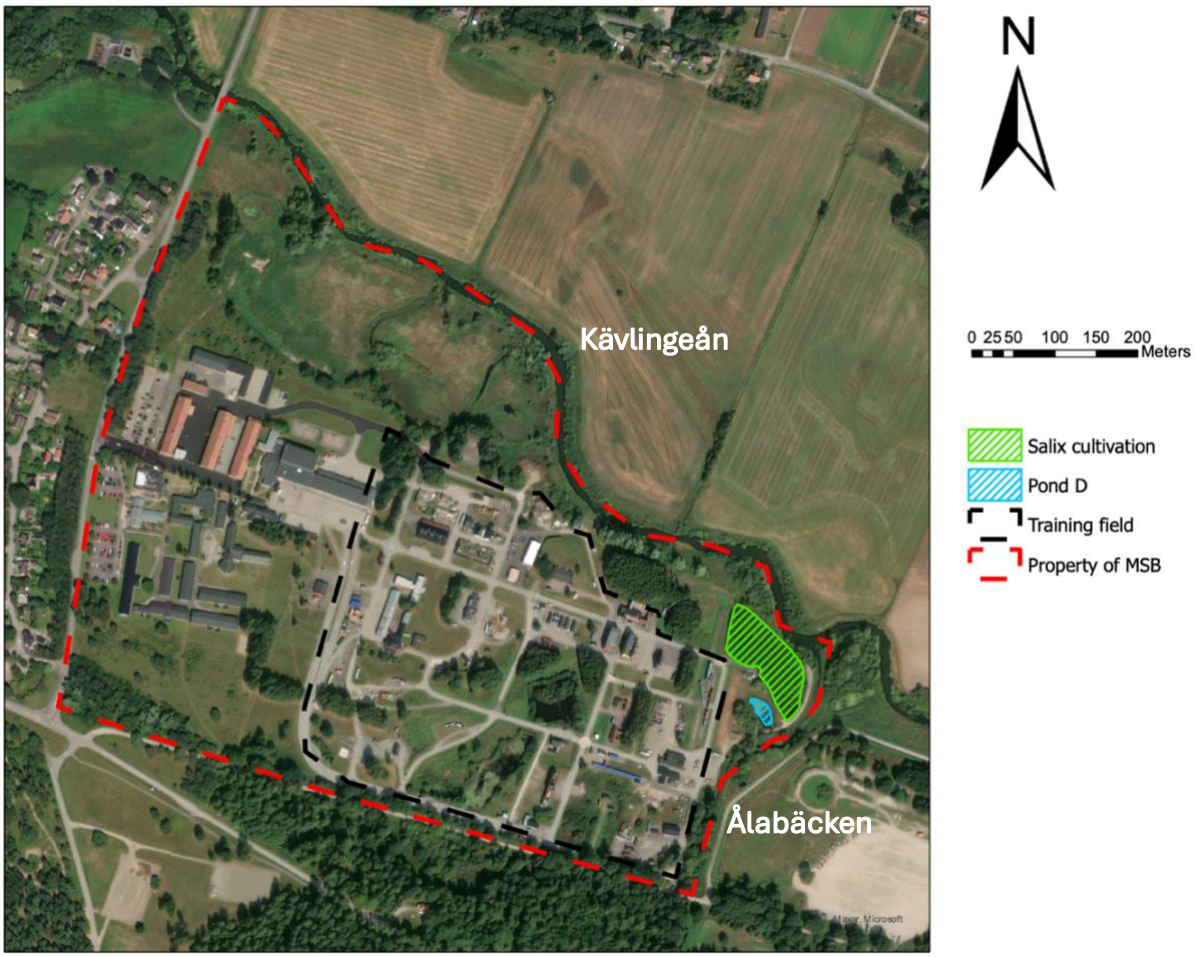


Figure 7: Map of MSB's property (red-dashed line), the training field (black-dashed line), the Salix cultivation (green-striped area) and Pond D (blue-striped area). ©Lantmäteriet

2.4.1 MSB Revinge and past activities

MSB Revinge is an organisation dedicated to enhancing civil protection and preparedness through education and training (MSB 2012; MSB 2024b). They offer a large selection of courses, mainly targeting municipalities, regions, country administrative boards and government agencies, and approximately 4,000 persons are trained at the facility each year. Main practices at the site includes education in how to extinguish fires in case of accidents, and since the early 80's, AFFFs containing varying concentrations of PFAS have been used in fire drills and other exercises simulating accidents (Bennermark & Thelin 2024).

In 2009, the three governmental agencies the Swedish Emergency Preparedness Agency, the Swedish Rescue Agency and the Board of Psychological Defence were disbanded to let the, at the time new agency, MSB take over (Krisinformation 2023). Hence, the ownership of the property Lund Revinge 1:14, which between the years 1986-2009 had belonged to the Swedish Rescue Agency, passed to MSB (Thelin 2024). Before 1986, the civil defence used the land, and based on digital aerial imagery provided by Lantmäteriet, the first buildings were constructed somewhere between 1960 and 1975, prior to which the property consisted of arable land.

2.4.2 Recipients and protected areas

Kävlingeån is classified as having poor ecological status, while Ålabäcken is classified as having moderate status (VISS n.d.a; VISS n.d.b). Neither recipient reaches good chemical status. Ålabäcken does not reach good chemical status with respect to PFOS, as it receives input of PFAS from the training fields in Revingehed, which is operated by the Swedish Armed Forces (VISS n.d.b, Lunds kommun 2023). Due to lack of information, Kävlingeån's chemical status has not been classified with respect to PFOS, but it is assumed to be affected by the input from Revingehed, like Ålabäcken (Bennermark & Thelin 2024; VISS n.d.a).

West and south of the property, the Natura 2000-protected area *Revingefältet* is located (Bennermark & Thelin 2024; Länsstyrelsen Skåne 2018). The area is a hotspot for biodiversity and an important area for plants and animals thriving in dry, sandy grasslands. The fields inhabit red-listed species as well as species listed in EU's Habitate Directive (Nilsson et al. 2021). Moreover, Kävlingeån classifies as a national interest in outdoor recreation (Naturvårdsverket 2025).

2.4.3 Water treatment system

Before the year of 2010, the extinguishing water from exercises at MSB Revinge was either collected and led to a pond (Pond D), see Figure 8, without treatment, or pumped into tanks and transported to Sysav, South Scania Waste (Nilsson & Eliasson 2010). However, in 2010, a local water treatment system was implemented with the aim to reduce the costs associated with transporting and treating the water externally, as well as improving the water quality in Pond D, as previous sampling by Sweco had revealed high concentrations of nitrogen and extremely high concentrations of phosphorus in the pond.

The current water treatment system at MSB Revinge consists of three treatment steps (Nilsson & Eliasson 2010). The first step (1) includes biological treatment through aeration, the second (2) aeration in Pond D and step three (3) involves irrigating the Salix cultivation. The contaminated extinguishing water is first collected and then pumped into two tanks located in close connection with Pond D and the Salix cultivation, see Figure 8. In the tanks the water is aerated in accordance with treatment step (1), after which it is led to a sludge separator which removes suspended solids in the water. The water is then led to Pond D for step (2) where the water is aerated by a pump. Finally, the water is sprinkled over the Salix cultivation, implementing the third and last treatment step (3) (Ibid.).

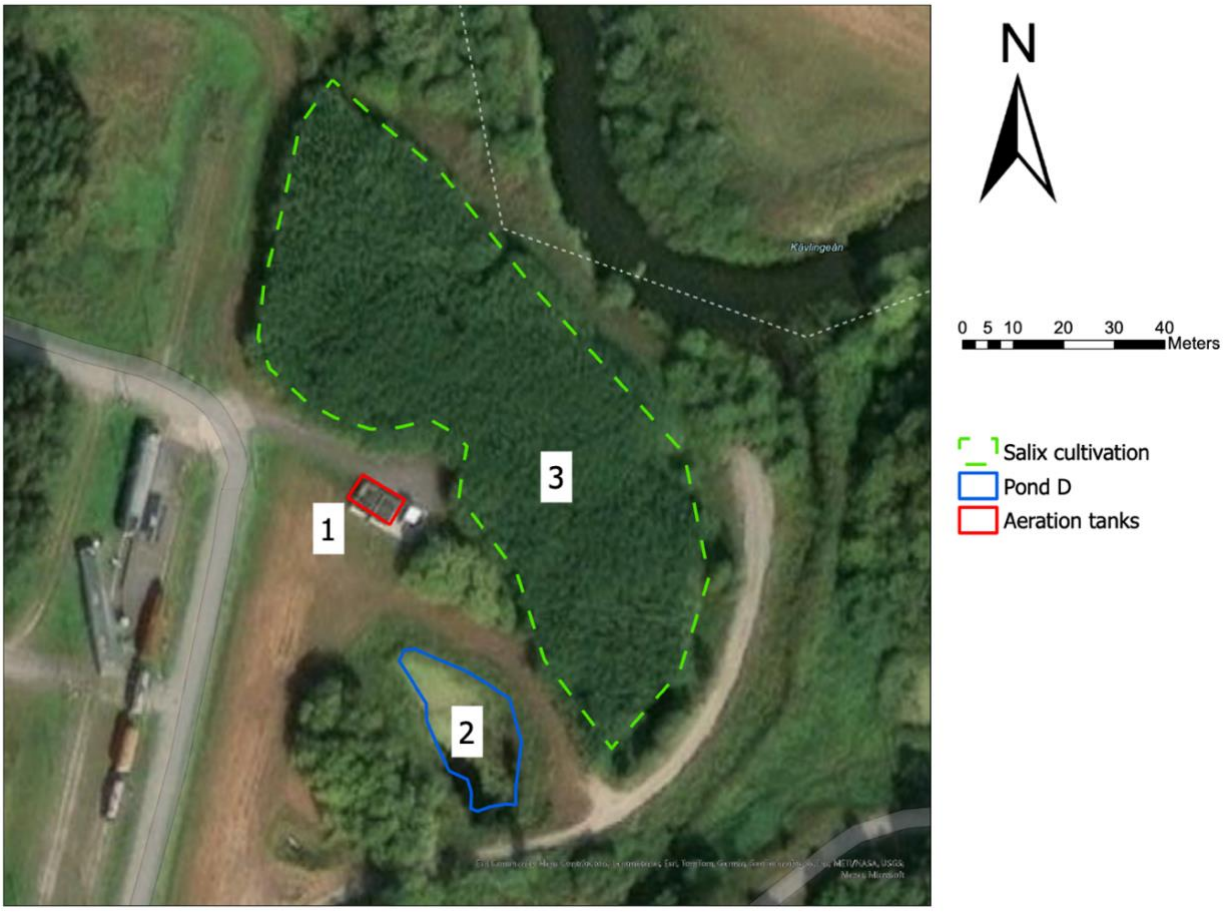


Figure 8: Overview of the three treatment steps for extinguishing water. ©Lantmäteriet

2.4.4 The Salix cultivation

When first planted, the Salix cultivation was approximately 4,000 m^2 large, dimensioned after the volume extinguishing water that arises yearly (Nilsson & Eliasson 2010). However, due to Salix's viability and low maintenance the cultivation is currently closer to a size of 5,000 m^2, according to satellite and drone imagery, see Figure 8 - Figure 10. The Salix was planted to treat the water by removing nutrients and heavy metals from the extinguishing water pumped from Pond D and can after harvest serve as an energy crop for biofuel. Additional values from the sprinkler system are that the soil in the Salix cultivation serves as a multifilter, providing both mechanical filtering as well as treating the water through chemical and biological processes (Ibid.).

Figure 9: Image of the Salix cultivation captured via drone in February 2025 by Johan Friberg, MSB. In the lower right part of the picture Pond D can be seen and in the lower middle part the two aeration tanks are located.

Figure 10: Image of the Salix cultivation captured via drone in February 2025 by Johan Friberg, MSB. Kävlingeån can be seen meandering behind the Salix.

In total, 20 sprinklers with a height of approximately 1.5 meters are used to distribute the water pumped from Pond D to the Salix[1]. Every year the cultivation is provided with a water volume of 1,000 - 1,500 m^3 from the pond (Bennermark & Thelin 2024). However, the irrigation only takes place around 130 days per year, during the warmer seasons, and during the colder periods Pond D acts as storage (Nilsson & Eliasson 2010). The pond is dimensioned to store approximately 600-700 m^3.

[1] Magnus Nilsson, Field Manager, MSB. Site visit 30 January 2025.

The Salix is harvested every 3-4 years, most recently in May 2022, i.e. three years ago[2]. The harvested shrubs had a total weight of 26,950 kg and were transported to Sysav Malmö, see Figure 11.

Figure 11: The most recent Salix harvest in May 2022[3].

2.4.5 Floodings

MSB Revinge experiences periodic flooding during heavy precipitation events, leading to the widening of Kävlingeån (Bennermark & Thelin 2024; Thelin 2024). These heavy events occur every 4-5 years, but smaller flooding events also occur more regularly. During floodings, the soil becomes saturated and contaminated groundwater stands in direct contact with the surface water. The Salix cultivation is flooded during these heavy rainfalls[4].

2.4.6 Geology and hydrogeological conditions

According to the soil depth map provided by SGU, the soil depth on the property ranges from 30 to 50 meters (SGU 2025a). The training field and building-covered areas occupy the largest portion of the property and SGU's soil type map indicates that these areas have a base layer composed of fill material, underlain by glaciofluvial sediments, as shown by the green-grey striped areas in Figure 12 (SGU 2025b). Along the northern border of the property, adjacent to Kävlingeån, floodplain deposits mainly consisting of clay and silt are found (pink area). This soil layer extends along Kävlingeån and is also found beneath the Salix cultivation. South of the property, the soil layers mainly consist of glaciofluvial sediments and 500 meters to the north of sandy and clayey till. The bedrock within the area is sedimentary, and the bedrock in the western side of the property is constituted by mudstone, claystone and siltstone (blue colour), while the eastern part, where the Salix is located, is constituted by marlstone (green colour), see Figure 13 (SGU 2025c).

[2] Dennis Göransson, Environmental Officer, MSB. E-mail 31 January 2025.
[3] Dennis Göransson, Environmental Officer, MSB. E-mail 31 January 2025.
[4] Magnus Nilsson, Field Manager, MSB. Telephone conversation 29 April 2025.

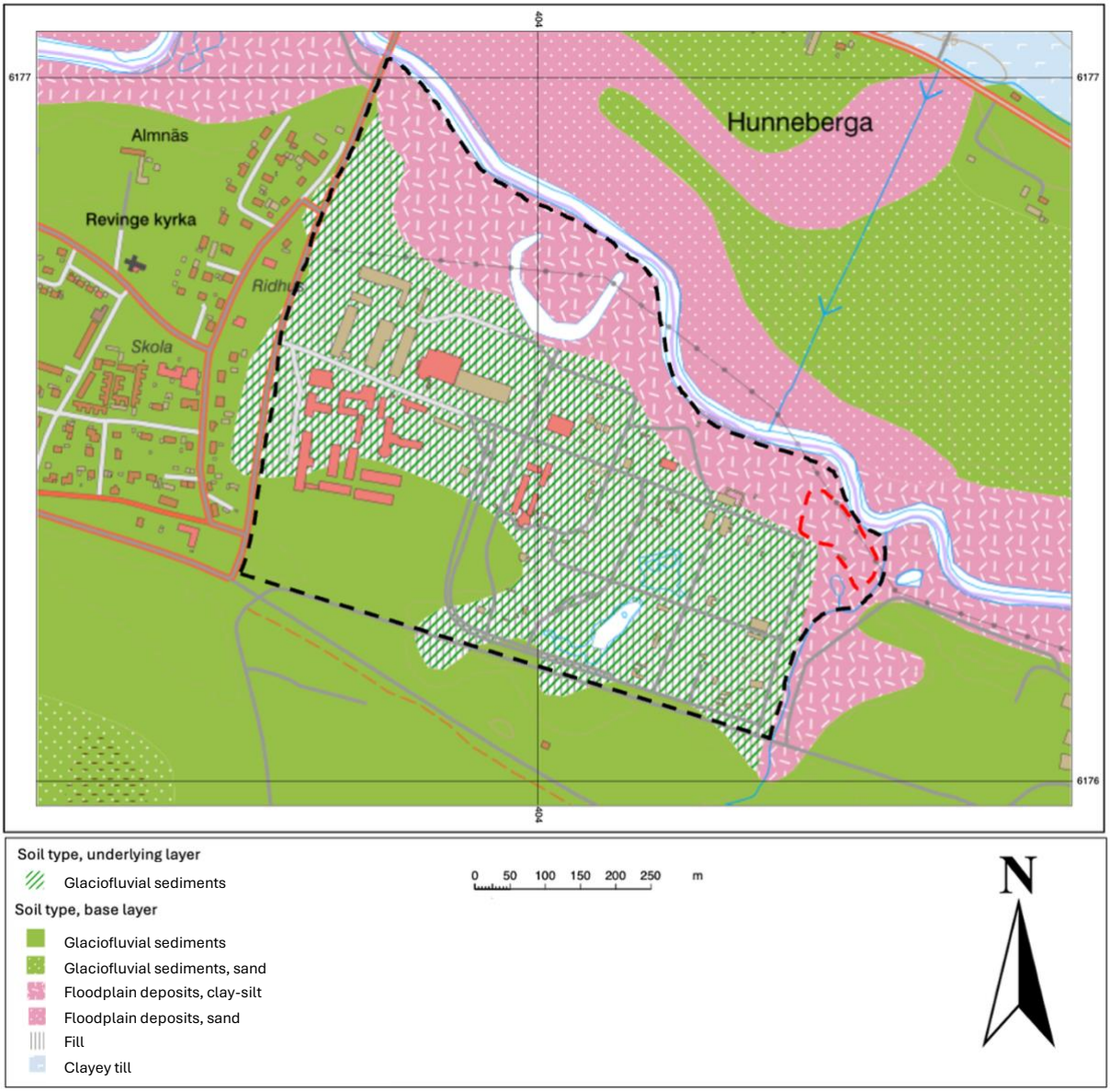


Figure 12: Map over the soil types according to SGU's soil type map (1:25 000-1: 100 000) (SGU 2025b). The property is marked with a black-dashed line and the Salix cultivation with a red-dashed line. Coordinates: Sweref99TM. © SGU

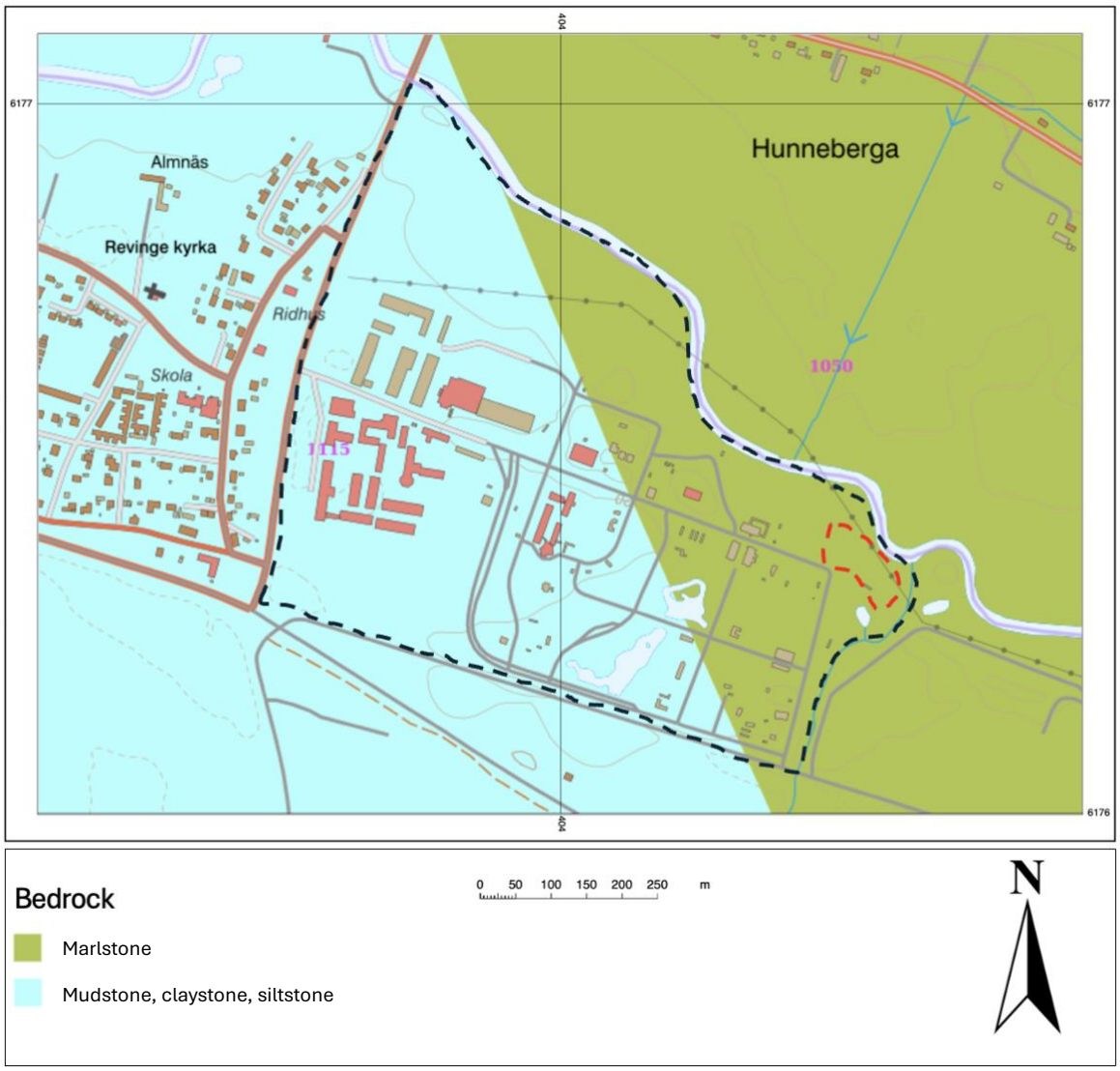


Figure 13: Map over the bedrocks according to SGU's bedrock map (SGU 2025c). The property is marked with a black-dashed line and the Salix cultivation with a red-dashed line. Coordinates: Sweref99TM. © SGU

The property is situated in the Vomb Trough (Vombsänkan), which is a lowland area located within the Sorgenfrei-Tornquist zone (Dahlqvist et al. 2021). The elevation within the property is approximately 20 m.a.s.l (Thelin 2024).

SGU's groundwater reservoir map shows that the uppermost aquifer classifies as a porous aquifer with a withdrawal potential ranging from 5 to 125 l/s within the property (SGU 2025d), see Figure 14. Several wells (both for pumping and observation) are found less than 1 km southwest from the property (Engleson 2017; Engvall 2015). Three aquifers have been identified at the location, where the uppermost aquifer is an unconfined porous aquifer in the soil type layer, which is consistent with SGU's groundwater reservoir map. Below this, two deeper aquifers in the sedimentary bedrock are identified as porous and fractured aquifers. Since the upper aquifer is unconfined (i.e. open to the atmosphere) it is more vulnerable to contamination from above (Fetter 2014). Furthermore, there is a well present within the property that is used for filling up extinguishing tanks with water as well as drainage

pumping (Bennermark & Thelin 2024; SGU 2025e). There are also multiple wells located within a 1 km radius of the area, which according to SGU's map over wells are individual water sources used for the household, secondary residences or small-scale agricultures, the closest one located less than 500 meters to the west.

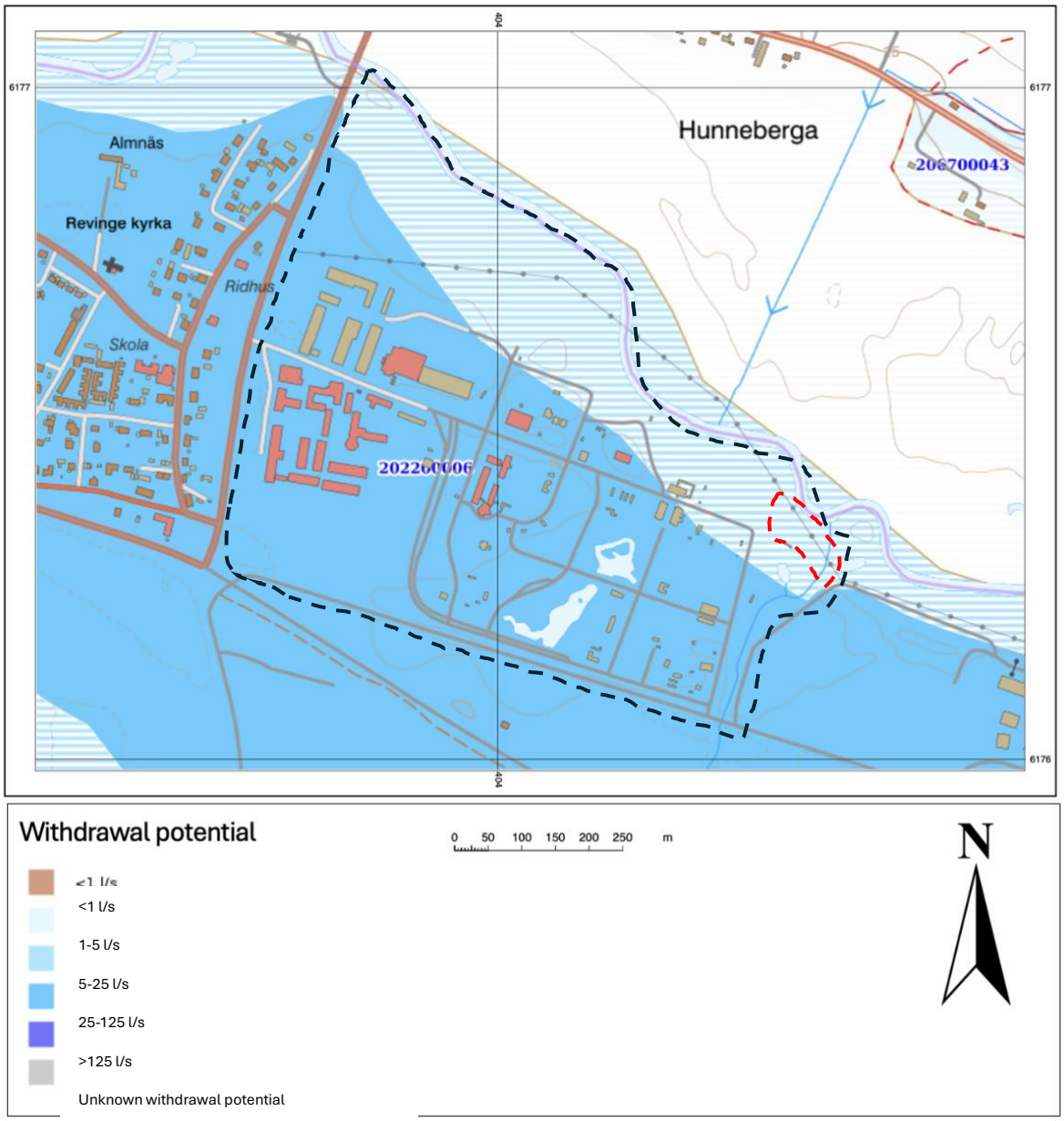


Figure 14: Map over the withdrawal potential in soil type layer (uppermost aquifer) according to SGU's groundwater reservoir map (SGU 2025d). The property is marked with a black-dashed line and the Salix cultivation with a red-dashed line. Coordinates: Sweref99TM. © SGU

There are multiple observations wells located within the property, two of which are situated in close connection to the Salix cultivation, see Figure 15. The groundwater level at the property ranges from 0.5 to 3 meters below the ground surface, with an average of approximately 1.5 meters (Bennermark & Thelin, 2024). The groundwater flow is directed primarily towards the two nearest recipients, Kävlingeån and Ålabäcken, moving mainly northwards.

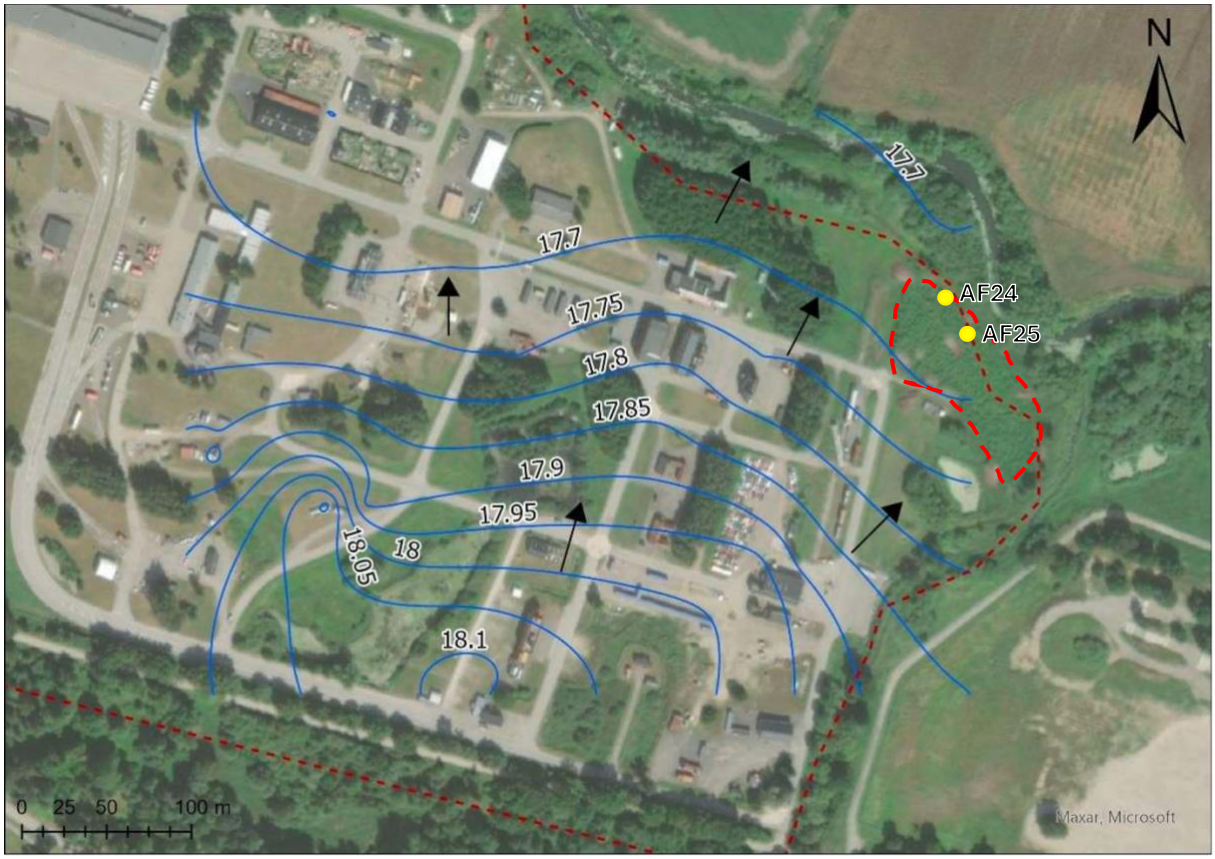


Figure 15: The values show the groundwater levels in the soil for each isoline (blue) given in meters above sea level (Bennermark & Thelin 2024). The isolines are interpolated from groundwater measurements conducted in 2024. The Salix cultivation is marked with a bright-red dashed line and two of the observation wells are situated within the cultivation, marked with yellow dots (approximate location).

2.4.7 Results from previous studies

Since 2005, sampling and monitoring of PFAS have been conducted at Revinge's training field by both Sweco, WSP and AFRY (Bennermark & Thelin 2024). The sampling has included groundwater, surface waters, sediment from both dams and adjacent streams, wastewater from tanks, and soil. At least one sampling has been performed in the Salix cultivation, and the results have indicated uptake of PFAS in the plants. The investigation was conducted by WSP in May 2022, with samples collected from the soil 0.0-0.3 meters below ground surface, as well as from the leaves, bark, and stem of the Salix (Dukic 2023). These samples were analysed for PFOS and PFOA by Eurofins Environment Testing Sweden AB and the results are presented in Table 1.

Table 1: Results from WSP's sampling of the Salix cultivation from May 2022, given in µg/kg dw (Dukic 2023).

	PFOS	PFOA	Sum PFAS
Soil	8.5	0.94	9.4
Leaves	0.12	<0.091	0.17
Bark	0.069	<0.063	0.10
Stem	<0.050	<0.050	<0.050

In 2024, a PFAS risk assessment was conducted by AFRY with the aim of estimating and mapping the contamination situation at MSB Revinge (Bennermark & Thelin 2024). The study estimated PFAS7 concentrations both based on arithmetic means from the different samples as well as from $UCLM^{95}$ ("Upper Confidence Limit of the Mean" with a confidence level of 95 %), where the latter is calculated in order to ensure that the contamination degree is not underestimated (Norrman et al. 2009). For the groundwater, the concentration of PFAS7 was calculated to be 4290 ng/L (mean) and 9000 ng/L ($UCLM^{95}$), which in terms of total load of PFAS dissolved in groundwater was calculated to 0.58 kg and 1.2 kg, respectively (Bennermark & Thelin 2024). The yearly mass transfer of PFAS7 from the groundwater to Kävlingeån is estimated to be in the interval between 66 g/year (mean) and 140 g/year ($UCLM^{95}$). SGI's preliminary guideline values for groundwater for protecting surface waters is 230 ng/L for PFOS, a value 97 % of the groundwater samples exceeded with regards to PFAS7 (Bennermark & Thelin 2024; Pettersson et al. 2015). Considering the input of PFAS7 to Kävlingeån in relation to its water flux, the load is high.

There are two observation wells installed within, or in close connection, to the Salix cultivation, and sampling has concluded that there are elevated levels of PFAS7 in the area (480-900 ng/L) (Bennermark & Thelin 2024). Although, there is some uncertainty as to whether these levels can be attributed to the irrigation of the Salix. The levels are considered elevated, however, they are of the same order of magnitude as the levels in three other observation wells installed at the same distance from Kävlingeån. In May 2024, the groundwater levels in the pipes were measured to 0.99 and 1.3 meters below the ground surface. See Figure 15 for placement of observation wells.

Using arithmetic mean and $UCLM^{95}$, Bennermark and Thelin (2024) also assessed the concentration of PFAS7 in soil based on samplings conducted by AFRY in 2024. PFAS was assumed not to be present in soil below a depth deeper than 1 m, because of PFAS' low K_d-values. Regardless, PFAS have been detected in occasional sampling spots where peat was present. The concentration of PFAS in soil was calculated to be 20.3 µg/kg dw (mean) and 56 µg/kg dw ($UCLM^{95}$). SGI's guideline value for PFOS in soil to protect surface water is 27 µg/kg dw, a value 19 % of the samples exceed with regards to PFAS7 (Bennermark & Thelin 2024; Pettersson et al. 2015). The total load PFAS7 present in the area in the soil was calculated to 5 kg and 15 kg for arithmetic mean concentration and $UCLM^{95}$, respectively.

Surface water and sediment samples have also been collected from Pond D. The most recent samplings were conducted in 2022, for which a PFAS-concentration (PFOS + PFOA) of 129 ng/L and <3.0 µg/kg dw was reported for the surface water and sediment (Bennermark & Thelin 2024; Dukic 2023). Compiled results from sampling by Sweco and WSP in Pond D are found in Appendix A.

3. Method

3.1 Field work

3.1.1 Sampling plan and strategy

A large-scale sampling was carried out on the 6th and 7th of March 2025, during two sunny days with an approximate temperature of 13 °C. The outlines of the cultivation were measured using a GPS to acquire a more precise estimation of the area. No reference site was used for the sampling due to the challenges related to finding an uncontaminated location with similar conditions where *Salix viminalis* could be found.

The sampling performed in March included one groundwater, one surface water, 30 soil and 135 plant samples, comprising of 15 root, 30 bark, 30 stem, 30 twig and 30 leaf (from the ground) samples. After collection, the soil and plant samples were combined separately into composite samples, resulting in one soil and five plant samples, each corresponding to one of the five different plant parts collected. The sub-samples for soil and plant parts were distributed randomly across the Salix cultivation area, with the sample numbers selected to ensure adequate coverage and provide a representative average of PFAS concentrations in the shrubs. The water samples were sent to the lab for analysis the same day as collection, while the soil and plant material were stored at 8 °C over the weekend and sent following Monday. Soil samples were stored in diffusion-tight plastic bags and plant samples in 400 ml plastic containers specific for PFAS analysis.

A supplementary sampling was performed on April 15th to collect fresh leaves from the Salix, as it was too early in the year for leaves in March. This leaf sample also consisted of 30 sub-samples, forming one composite sample, and was sent for analysis the same day. All samples collected were sent to Eurofins Environment Testing Sweden AB for analysis. During transportation to the lab, all samples were contained in insulated packages with ice packs to maintain cool temperatures during transit.

Before the large-scale sampling in March, a “test-sample” was sent for analysis to the lab to assess whether the PFAS levels in the trees exceeded the reporting limits. The test-sample consisted of leaves collected from the shrubs in November 2024 by MSB, which were stored in plastic bags in a shed before being sent to the lab on February 10th, 2025.

3.1.2 Groundwater and surface water sampling

One groundwater sample from observation well AF25, see Figure 15, as well as one surface water sample from Pond D, were taken on the 6th of March 2025. The groundwater level was first measured using a water level gauge, after which a peristaltic pump was used to pump up the water. A multiparameter meter (YSI) was used to measure the water’s temperature (°C), dissolved oxygen (%), specific conductance (µS/cm) and oxygen reduction potential (mV). The water was collected when said parameters had stabilised, after approximately 15 minutes.

The surface water was sampled using a water collector with a telescopic arm. Both water samples were collected in two 100 ml plastic bottles each.

3.1.3 Soil sampling

One composite sample of soil, consisting of 30 sub-samples, was taken on the 6th of March 2025. A stainless-steel shovel was used to dig 30 holes in the ground to a depth of approximately 0 – 0.3 m. The samples were collected using a smaller shovel, sampling throughout the circumference of the hole, and along the depth with 30 small digs. Diffusion-tight plastic bags measuring 300 x 180 mm were used to collect each sub-sample and sealed using cable ties. On the 10th of March, after storage in the fridge over the weekend, all sub-samples were mixed to one composite sample. A smaller shovel was used to ensure that approximately the same volume of soil from each sub-sample was added to the composite sample, which was collected in a diffusion-tight plastic bag and sealed with a cable tie.

3.1.4 Salix sampling

Leaves, twigs, bark, stem and roots, were sampled on the 7th of March 2025. Similar sampling vessels, i.e. 400 ml plastic containers specific for PFAS analysis, were used for all plant samples. In some cases, a diffusion-tight bag was initially used for convenience during the collection but was afterwards transferred to 400 ml containers. All plant samples were collected as composite samples, consisting of 30 sub-samples, except for the roots, which were constituted of only 15 sub-samples, because of their limited availability.

Leaves were sampled three times, in November, in March and in April. The November leaves were withered leaves that had not yet fallen and were collected from three locations in the Salix cultivation. The leaf samples in March and April were both composite samples, containing 30 sub-samples each, however, leaves in March were collected from the ground.

The diameter of a twig was defined as maximum 0.5 cm and the twigs were collected at different heights along the Salix stems. Due to the height of the Salix being approximately 4 meters, a stainless-steel saw was used to cut down the shrubs to reach the twigs at the top. The bark was stripped from the Salix stems using a stainless-steel knife. To collect the stems of Salix, the saw was used to make two cuts a few centimetres apart, to receive a small sample of the stem. The bark of the stem samples was removed using a knife. The root samples were collected using a shovel to dig up the soil near a Salix, hence exposing the roots. The roots were then separated from the plant using a knife. Further, the roots were cleansed using water to remove soil and other debris and shaken to remove excess water.

3.1.5 Assessing shrub proportions

To estimate the load of PFAS on one shrub, the biomass proportions were assessed in field. One shrub was dug up using an excavator and the different plant parts were separated using a

saw and a secateur and weighed individually. To estimate the bark weight, a whole stem was first weighed with its bark and after the bark had been removed using a knife, to estimate the stem-to-bark weight ratio. The weighing of the plant parts could both give an indication of the proportions of a Salix shrub, but also on what the total weight of a Salix shrub could be. However, due to accessibility reasons, the shrub chosen for the assessment was in the outskirts of the cultivation. These shrubs were generally larger than the ones found further into the cultivation, introducing a risk of overestimating the biomass.

3.2 Laboratory Analysis

3.2.1 Chemicals analysed

The soil, surface water and groundwater samples were screened for 35 different PFAS. The PFAS was categorised according to their chemical structure as follows: short-chained PFCA (PFBA, PFPeA, PFHxA, PFHpA), long-chained PFCA (PFOA, PFNA, PFDA, PFUdA, PFDoA, PFTrDA, PFTeDA, PFHxDA), short-chained PFSA (PFBS, PFPeS), long-chained PFSA (PFHxS, PFHpS, PFOS, PFNS, PFDS, PFUnDS, PFDoS, PFTrDS), FOSA (PFOSA, MeFOSA, EtFOSA), FOSAA (FOSAA, MeFOSAA, EtFOSAA,), FOSE (MeFOSE, EtFOSE), fluorotelomer sulfonates (4:2 FTS, 6:2 FTS, 8:2 FTS) and other PFAS (HPFHpA, P37DMOA).

All plant samples were screened for 47 different PFAS as opposed to the soil and water, due to differences in the laboratory's analysis packages. All above mentioned substances were included in the screening of plant material, except for P37DMOA. Additional PFAS screened were: long-chained PFCA (PFODA), ultrashort-chained PFSA (PFPrS), fluorotelomer sulfonates (10:2 FTS), PFAS ethers (6:2 Cl-PFAES, 8:2 Cl-PFAES, DONA, PFMPA, PFMBA, NFDHA, PFEESA), FASA, other than FOSA, (PFHxSA, PFBSA) and other PFAS (PFECHS).

3.2.2 Sample preparation and laboratory analysis

All samples were prepared and analysed by Eurofins Environment Testing Sweden AB. The preparation of the plant samples included drying at 35 – 37 °C and grinding, and the groundwater had to be centrifuged before analysis due to high turbidity. The concentration of PFAS in all samples was assessed using the method liquid chromatography tandem mass spectrometry (LC-MS-MS), according to the standards *DIN 38414-14 mod. Anal. Chem.2005,77,6353 mod.* for soil, *DIN38407-42, UNEP Chemicals Branch 2015 mod.* for water and *J. Anal. Chem. 77 (2005) 6353 mod.* for cellulose-rich matrices (Eurofins n.d.). The measuring uncertainties for the analyses were ± 36 %, ± 31 % and ± 37 % for soil, water and cellulose-rich matrices, respectively.

For the soil, the TOC and pH was also analysed by the lab. The TOC was determined using gravimetry (*SS-EN 12880:2000 mod.* and *SS-EN 12879:2000*) and the pH through electrometry (*SS-EN ISO 10390:2022*) (Eurofins n.d.).

3.3 Quality assurance

Measures were taken to reduce the risk of contamination during the sample collection and handling. Equipment containing fluorinated compounds were avoided, and all equipment were washed with tap water and wiped with paper between each collection to prevent cross-contamination. Additionally, plastic gloves were worn at all times and changed between every sample. If any sample came into contact with skin, clothing, or was dropped on the ground, it was discarded.

3.4 Calculations

3.4.1 Total tree burden

The total tree burden of one shrub, as well as the whole cultivation, was calculated using Equation 3, adapted after Gobelius et al. (2017).

$$\sum PFAS_{shrub\ tissue}(\mu g) = C_{PFAS\ in\ tissue}\ (\mu g\ kg^{-1}\ ww) * \ weight\ fraction\ (\%) * biomass\ (kg) \quad (3)$$

Where C is the concentration of PFAS in the tissue of a plant part and the weight fraction is the ratio of the plant part to the whole shrub, calculated as described in *3.1.5 Assessing shrub proportions.* The biomass is either the weight of one shrub or of the whole cultivation.

3.4.2 Mass balance

The aim of the mass balance calculations was to estimate how much PFAS had been transferred from Pond D to the cultivation since 2010 as well as how much PFAS the Salix has remediated during this time. The input from Pond D was estimated by calculating the annual load PFAS sprinkled over the cultivation, based on the results from the analysed surface water. The removal of PFAS was calculated by combining the calculated total tree burden and estimation of the total biomass harvested over the years. The potential PFAS that could have been removed refers to a hypothetical scenario where the fallen leaves from the Salix was picked up every season.

The total PFAS content in the soil within the Salix cultivation was estimated by calculating the total mass of soil above the groundwater surface and using the analysis result from sampling. However, it is not certain that all input of PFAS to the soil can be attributed to the sprinkling of water from Pond D since PFAS can have spread in other ways, such as through surface runoff. The load on groundwater was calculated by combining the measured PFAS concentration with the estimated volume of groundwater beneath the Salix cultivation. Although, it is challenging to know to which extent the groundwater concentrations can be attributed to the PFAS added from the sprinkler system versus PFAS originating from other point sources withing the training field.

In this study, “PFAS uptake/accumulation” refers to the PFAS content in the Salix shrubs and “PFAS removal” to the PFAS that is transferred away from the system, either through harvests or leaf collection.

3.4.3 Soil/groundwater partitioning

For each PFAS detected in both soil and groundwater, a soil/groundwater partitioning coefficient K_d* was calculated. K_d* was calculated by taking the concentration of a specific PFAS in soil (µg/kg dw) divided by the concentration in groundwater (µg/L). This method is in accordance with the description of area specific determination of K_d* provided Naturvårdsverket (2009), see *2.1.5 Transport and fate of PFAS* for definition.

3.4.4 Bioconcentration factor

BCFs were calculated for all plant tissues and for all PFAS with a concentration over the lab’s reporting limit, to assess the shrubs’ ability to accumulate and remediate PFAS. The BCFs were calculated for both soil and groundwater, since the roots are estimated to be in contact with groundwater, at least during some periods of the year. The BCFs were calculated according to Equation 2.

4. Results

4.1 Field observations

The soil horizon was homogenous throughout the 0.3-meter-deep pits as well as across the cultivation. The soil was a dark brown organic soil, containing a lot of plant material and worms. At the time of sampling, the groundwater level was measured to 1.57 meters below ground level, in observation well AF25.

In March, coordinates along the outskirts of the Salix cultivation were measured and in ArcGIS the area was calculated to 5,000 m^2, as suspected. Moreover, at the site it was noticed that an old pile of wood had been left during the last harvest, which also can be observed in Figure 9. Consequently, the weight of the shrubs in the 2022 harvest was rounded up to 30,000 kg, as opposed to the weight transported to Sysav (26,950 kg).

When the leaves from the November shrubs were collected in February, they were dry with a reported water content of 12 % from the lab. Since calculations to convert dry weight concentrations to wet weight concentrations were a prerequisite for the tree burden, mass balance and BCF calculations, the water content of the leaves was instead assumed to be equal to the leaves sampled by WSP in May 2022, i.e. 70 % (Dukic 2023). Furthermore, in March, leaves had not yet grown out and instead leaves from last season were sampled from the ground, which had started decomposing but with most of their structure intact. However, the shrubs had catkins, which were not sampled separately but to some extent represented as they were attached to some of the collected twigs. In April, the leaves were smaller as they had recently started growing out, and the sprinkler system had not yet started for the season. Regarding the shrub that was dug up, it was observed that most of the root mass was in the top decimetres, but the roots stretched up to a length of 0.75 meters, approximately. However, it was apparent that some roots had been torn of, indicating that some likely were even longer.

4.2 Analysis results

4.2.1 Groundwater and surface water

In the groundwater, 9 out of the 35 PFAS analysed for were detected and the total concentration was found to be 1600 ng/L. For the surface water in Pond D, 21 of the 35 analysed PFAS were found and the total PFAS concentration was 1400 ng/L. The PFAS detected over the lab's reporting limit, and their concentrations in the waters, are compiled in Table 2. Complete results from the analysis, including reporting limits, are found in Appendix B and Appendix C.

Table 2: The concentrations of PFAS found in the groundwater and surface water including their sub-group and detected concentrations. All concentrations are reported in ng/L.

Substance	Group	Groundwater (ng/L)	Surface water (ng/L)
PFBA	PFCA short	140	60
PFPeA	PFCA short	640	190
PFHxA	PFCA short	530	180
PFHpA	PFCA short	200	84
PFOA	PFCA long	20	49
PFNA	PFCA long	-	12
PFDA	PFCA long	-	5.6
PFUdA	PFCA long	-	0.92
PFDoA	PFCA long	-	0.43
PFBS	PFSA short	32	14
PFPeS	PFSA short	14	10
PFHxS	PFSA long	38	96
PFHpS	PFSA long	-	4.5
PFOS	PFSA long	-	480
PFNS	PFSA long	-	0.35
PFOSA	FOSA	-	7
FOSAA	FOSAA	-	0.42
MeFOSAA	FOSAA	-	1.1
EtFOSAA	FOSAA	-	0.32
6:2 FTS	FTS	19	210
8:2 FTS	FTS	-	34
Sum PFAS		1600	1400

The PFAS content in the groundwater was dominated by PFBA (140 ng/L), PFPeA (640 ng/L), PFHxA (530 ng/L) and PFHpA (200 ng/L), i.e. the short-chained PFCA, which constituted 92 % of the PFAS. For the surface water, PFOS (480 ng/L) accounted for the largest fraction of the PFAS concentration and the group long-chained PFSA constituted 40 % of the PFAS. The second most frequent PFAS in Pond D was 6:2 FTS (210 ng/L), followed by the short-chained PFCA which as a group made up for 36 % of the PFAS content. The compositions of PFAS are displayed in the graphs in Figure 16 and the compositions of the PFAS groups are found in Figure 17.

During the collection of the groundwater sample, the multiparameter meter read a temperature of 9.7 °C, dissolved oxygen of 48 %, specific conductance of 42 µS/cm and oxygen reduction potential of 370 mV.

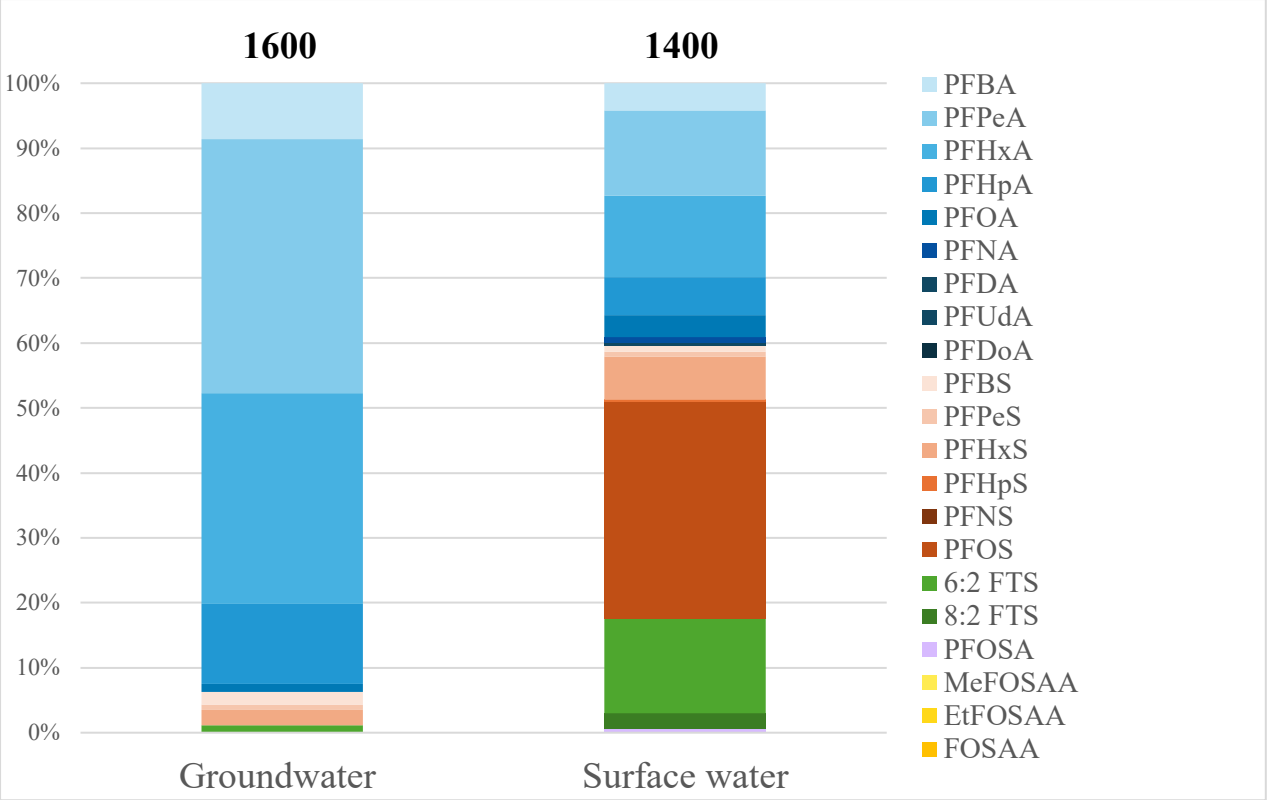


Figure 16: PFAS composition in groundwater and surface water in Pond D. PFCA have blue colours, PFSA red, FOSA purple, FOSAA yellow and FTS green. The total PFAS concentrations (ng/L) are found at the top of the bars.

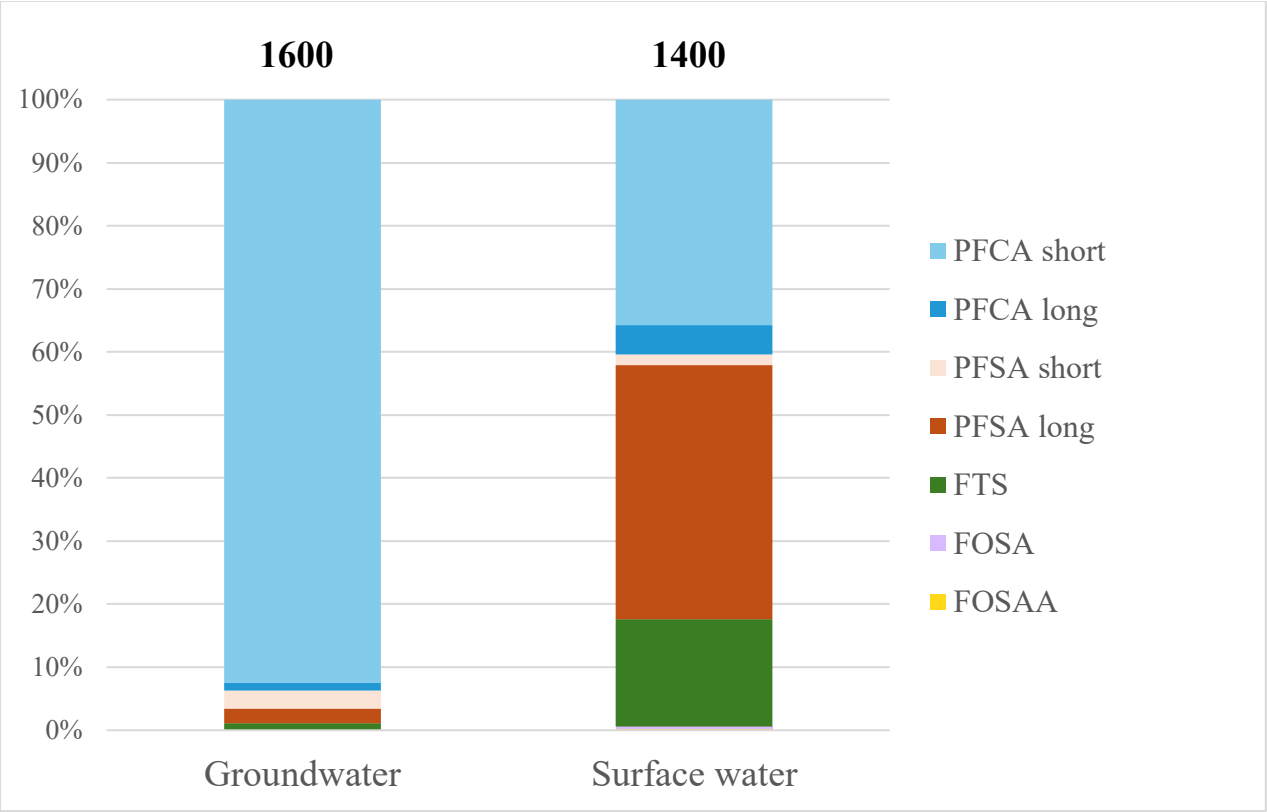


Figure 17: Composition of PFAS groups in groundwater and in the surface water from Pond D.

4.2.2 Soil

13 of the 35 analysed PFAS were found above the reporting limit for the soil sample, with a total PFAS concentration of 25 µg/kg dw. Concentrations of these substances can be seen in Table 3 and complete result from the soil analysis can be found in Appendix D. Furthermore, the soil pH was found to be 6.4 and the TOC 4.0 %.

Table 3: Substances found over the reporting limit for the soil sample and corresponding concentrations. All concentrations are reported in µg/kg dw. No value indicates that the PFAS did not exceed the reporting limit.

Substance (abbreviation)	Group	Soil (µg/kg dw)
PFBA	PFCA short	1.7
PFPeA	PFCA short	5.1
PFHxA	PFCA short	2.3
PFHpA	PFCA short	1.8
PFOA	PFCA long	1.1
PFNA	PFCA long	1.1
PFDA	PFCA long	0.18
PFUdA	PFCA long	0.12
PFHxS	PFSA long	1.1
PFHpS	PFSA long	0.058
PFOS	PFSA long	10
6:2 FTS	FTS	0.033
PFOSA	FOSA	0.33
Sum PFAS		25

The highest substance concentration found in the soil sample was PFOS, at 40 % of the PFAS, followed by PFPeA at 20 %. The composition of the detected substances in the soil sample are depicted in Figure 18. Further, when categorising the substances into PFAS groups, long-chained PFSA constituted the major fraction of the detected PFAS, with 45 % of the total concentration. Additionally, short-chained PFCA followed closely at 44 % of the PFAS. The PFAS group composition of the soil sample is displayed in Figure 19. No short-chained PFSA were detected above the reporting limit.

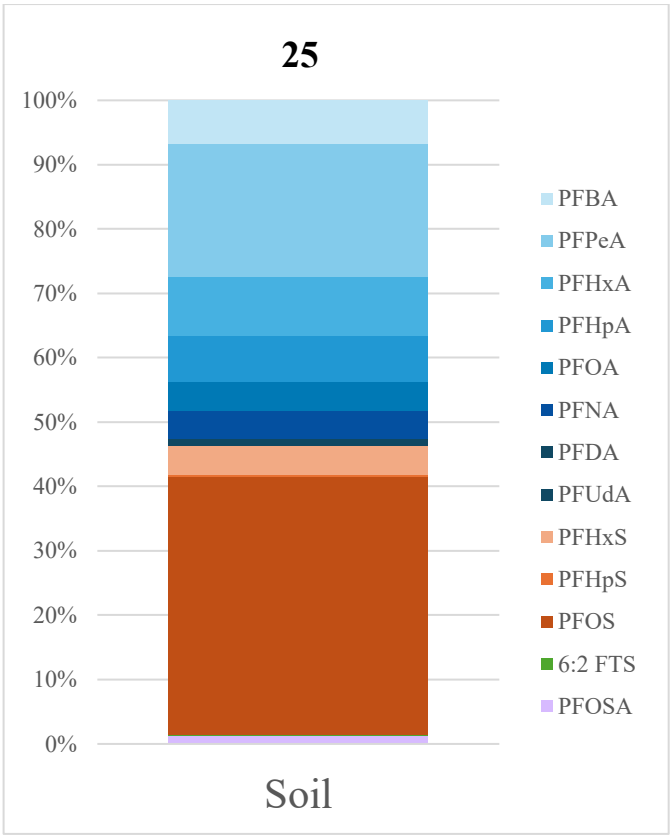


Figure 18: PFAS composition in soil. PFCA has blue colours, PFSA red, FTS green and FOSA purple. The total PFAS concentration (µg/kg dw) is found at the top of the bar.

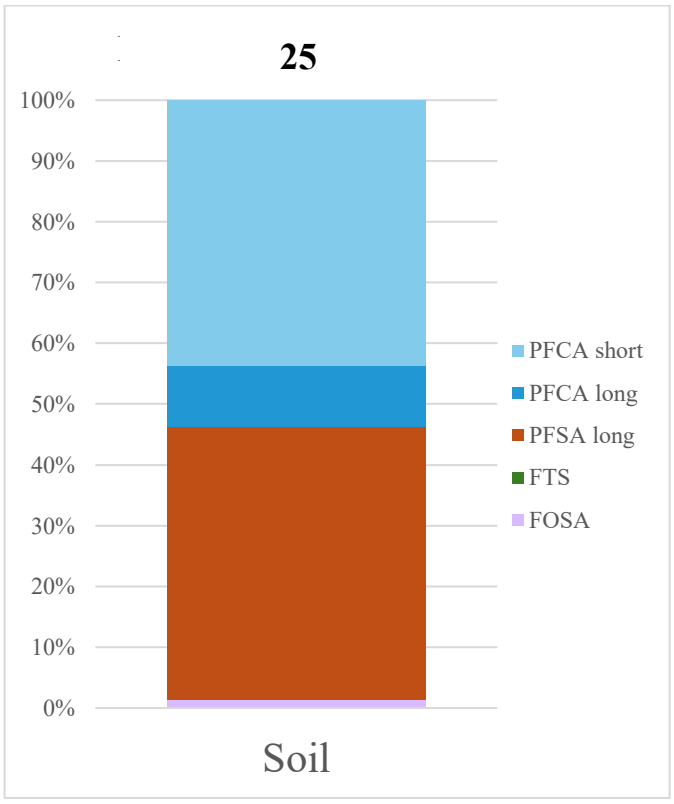


Figure 19: Composition of PFAS groups in the soil.

4.2.3 Salix

For the Salix samples (leaves November, leaves March, leaves April, twigs, bark, stem and root) 15 out of 47 analysed PFAS were found present in at least one of the samples. The concentrations of the detected PFAS can be seen in Table 4 and the complete result including reporting limits is found in Appendix E and Appendix F. The total PFAS concentration was highest for the November leaves with 730 µg/kg dw, followed by the April leaves at 63 µg/kg dw and March leaves (from the ground) at 22 µg/kg dw. Further, twigs and bark had a concentration of 3.5 µg/kg dw each, root 1.1 µg/kg dw and for the stem no PFAS were detected above the reporting limits.

Table 4: The detected PFAS in each part of the Salix with corresponding concentration (µg/kg dw). No value indicates that the PFAS did not exceed the reporting limit.

Substance	Group	Leaves Nov 24	Leaves Mar 25	Leaves Apr 25	Twigs	Bark	Stem	Root
PFBA	PFCA short	160	-	11	-	-	-	-
PFPeA	PFCA short	380	2.7	32	1.7	1.5	-	-
PFHxA	PFCA short	43	1.6	13	1	1	-	-
PFHpA	PFCA short	14	0.35	1.5	0.2	0.35	-	0.16
PFOA	PFCA long	2.9	0.31	0.25	-	-	-	-
PFNA	PFCA long	1.1	0.26	-	-	-	-	-
PFDA	PFCA long	-	0.19	-	-	-	-	-
PFBS	PFSA short	1	-	0.21	-	-	-	-
PFPeS	PFSA short	3.5	-	-	-	-	-	-
PFHxS	PFSA long	15	0.34	0.42	0.13	0.1	-	0.18
PFOS	PFSA long	9.3	12	1.5	0.3	0.25	-	0.75
6:2 FTS	FTS	96	2.1	2.0	0.18	0.27	-	-
8:2 FTS	FTS	0.31	0.97	0.96	-	-	-	-
PFOSA	FOSA	-	0.37	-	-	-	-	-
PFHxSA	FASA	0.11	0.33	-	-	-	-	-
Sum PFAS		730	22	63	3.5	3.5	-	1.1

The substances constituting the biggest fraction in the November leaves were PFPeA at 52 %, followed by PFBA at 22 % of the total concentration. The November leaves mainly consisted of the group short-chained PFCA with 82 % of the PFAS. The April leaves had similar PFAS composition as the November leaves with 51 % PFPeA and 92 % short-chained PFCA. PFOS had the highest concentration in the March leaves at 56 % of the PFAS, and the group long-chained PFSA constituted 57 %. The largest fractions of PFAS in the twigs was PFPeA at 48 % and PFHxA at 28 %, and in total, the short-chained PFCA made up 83 %. Similarly to twigs, the biggest fractions of PFAS in the bark was PFPeA at 43 % followed by PFHxA at 29 %. Additionally, the major PFAS group in bark was short-chained PFCA with 82 % of the PFAS. Only three PFAS were found in the root, with PFOS constituting the major fraction at 69 %. Long-chained PFSA dominated the PFAS in the roots with 85 %. The composition of

the detected PFAS and the composition of the PFAS groups for each Salix sample are displayed in Figure 20 and Figure 21, respectively.

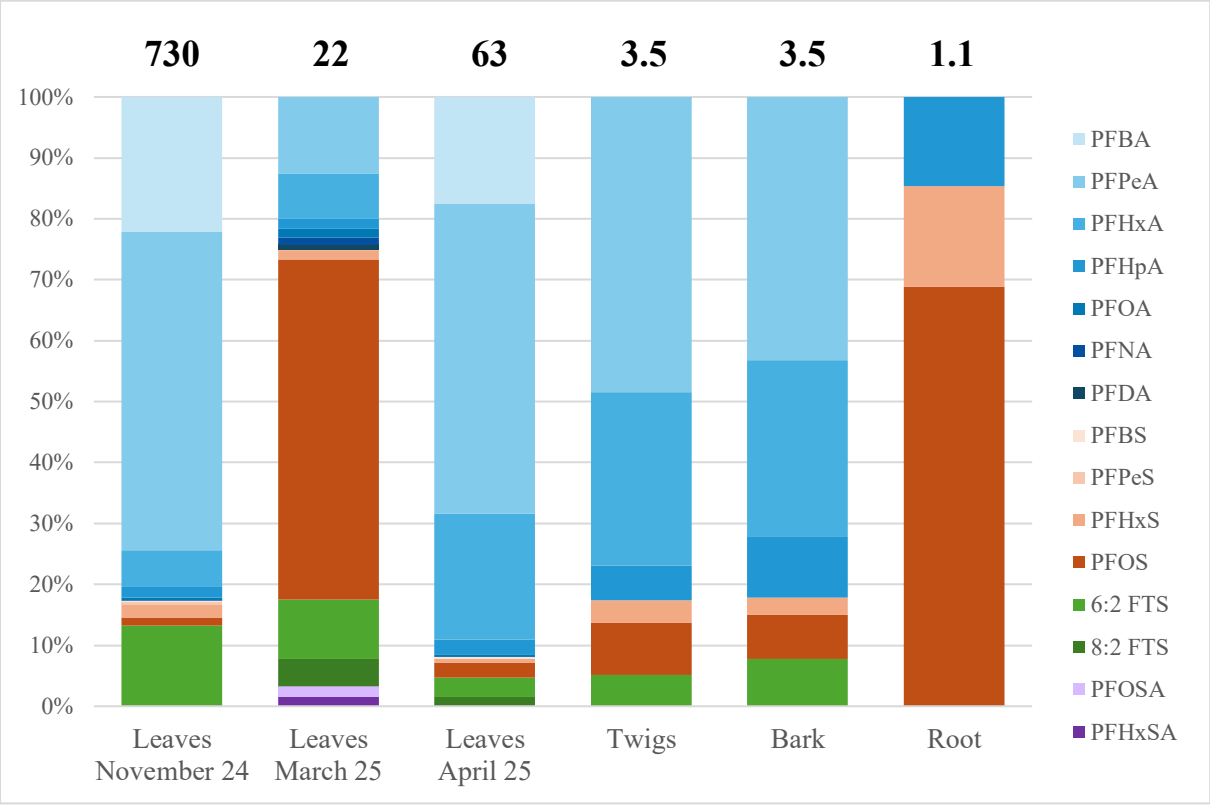


Figure 20: PFAS composition in the analysed parts of the Salix. PFCA have blue colours, PFSA red, FTS green and FOSA purple. The total PFAS concentrations (µg/kg dw) are found at the top of the bars.

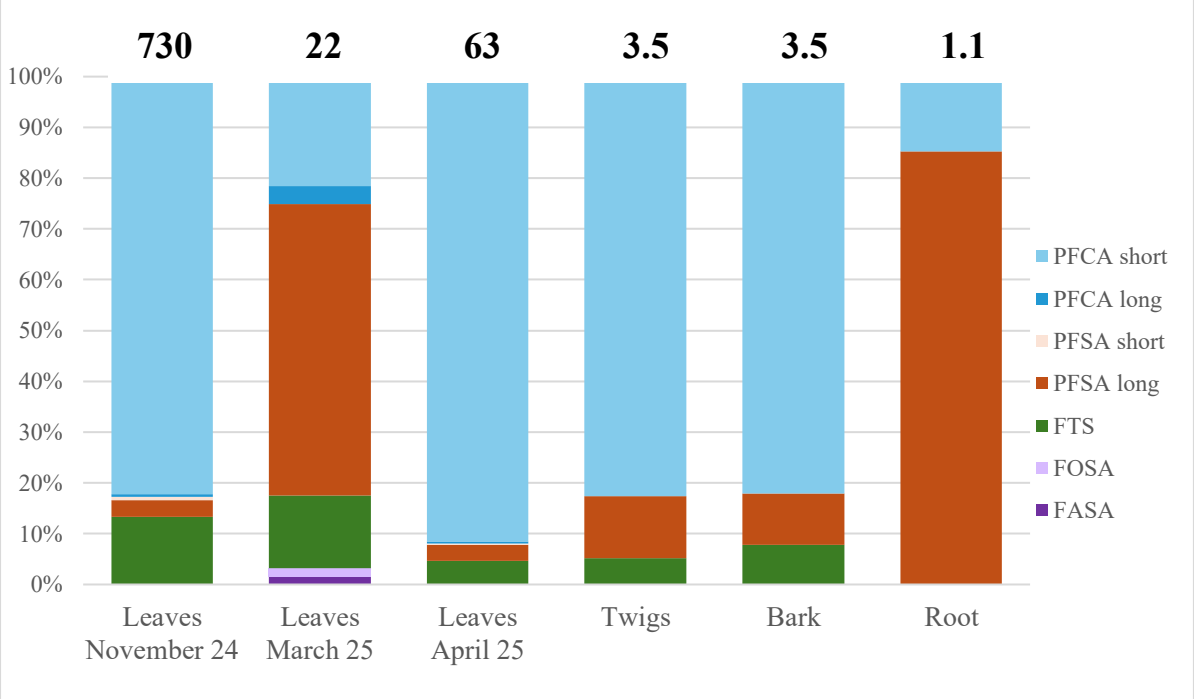


Figure 21: PFAS group composition for the analysed parts of the Salix.

4.3 Total tree burden

The results from the weighing of the Salix shrub are displayed in Table 5, including both the weight of each plant part as well as the calculated weight proportions. As the shrubs were bare, the weight proportion of the leaves were assumed to be 5.0 % excluding roots, as proposed by Skirfors (2023), where 5.0 % is a rough estimate used in the Swedish Salix industry[5]. To calculate the PFAS load on the leaves, the proportions were translated to include roots, see Table 5.

By converting the results in Table 4 from dry weight to estimated wet weight, the total PFAS load on each plant part, as well as the whole shrub, were calculated using Equation 3. The total PFAS load on a shrub in November was estimated to be 250 µg and in April 26 µg, assuming the PFAS concentrations in the plant parts, other than the leaves, were the same.

Table 5: The estimated allometric relationship in the whole Salix shrub and the calculated total PFAS burden.

Plant part	Weight (kg)	Proportion (%)	Total PFAS (µg)
Root	14	38	4.6
Stem w/o bark	17	48	-
Bark	2.3	6.4	3.9
Twigs	1.5	4.3	2.5
Leaves (Nov)	1.1	3.1	240
Leaves (Apr)	1.1	3.1	15
Total (Apr/Nov)	36	100	26/250

The total above-ground weight for the whole cultivation was assumed to be the same as for the last Salix harvest, i.e. 30,000 kg. The total weight, including roots, was calculated to be 48,000 kg, based on the proportions in Table 5. Assuming the estimated allometric relationship for the weighed shrub is representative for the whole Salix cultivation, and a weight of 48,000 kg, Equation 3 was used to calculate the total PFAS burden on the cultivation. The total PFAS burden was calculated to be 0.036-0.34 g, where the lower value represents April concentrations and the higher November concentrations.

4.4 Mass balance

The yearly input of PFAS to the cultivation was estimated using the total PFAS concentration found in Pond D in March 2025, i.e. 1400 ng/L. Assuming the concentration is representative for the 15 years of irrigation of the Salix, the total input of PFAS to the cultivation is 22-32 g PFAS, given an annual sprinkled water volume of 1,000-1,500 m^3. However, this is a gross simplification and should be interpreted with care, since the PFAS concentrations fluctuates in the pond. The concentrations have most likely been higher during periods when AFFFs

[5] Oscar Liljeström, PhD student at the Department of Aquatic Sciences and Assessment, SLU. E-mail 4 April 2025.

with PFAS was actively used at MSB. Results and their uncertainties will be further addressed in the discussion.

The Salix cultivation was assumed to have been harvested every fourth year, meaning it has been harvested three times in total. 30,000 kg was harvested in May 2022, however, no data was available regarding the first two harvests. Therefore, the weights were assumed equal all three times. The assumptions regarding harvest frequency and biomasses were made in consultation with MSB. To calculate the removed amount of PFAS through the harvests, the proportions in Table 5 were translated to the above-ground biomass relationships, see Table 6 for weights, proportions and PFAS load on one Salix shrub, excluding its roots.

Table 6: The estimated allometric relationship in the above-ground part of a Salix shrub and the calculated total PFAS burden.

Plant part	Weight (kg)	Proportion (%)	Total PFAS (µg)
Stem w/o bark	17	78	-
Bark	2.3	10	3.9
Twigs	1.5	6.9	2.5
Leaves (Nov)	1.1	5.0	240
Leaves (Apr)	1.1	5.0	15
Total (Apr/Nov)	22	100	22/250

By inserting the new proportions from Table 6, the concentrations from Table 4 and the harvested weight into Equation 3, the PFAS removed was calculated to be 0.030-0.34 g/harvest and 0.089-1.0 g in total over the years, excluding the current load on the shrubs. If a system for leaf collection was implemented the years between harvests, an additional 3.9 g could have been removed during the 15 years, assuming November concentrations in the leaves. This would have resulted in a total removal of 4.0-4.9 g PFAS from the system over the past 15 years. The yearly removal of PFAS during a four-year period, where leaves are collected three years and the Salix is harvested one year, thus becomes 0.25-0.33 g. However, the calculations do not consider how the concentrations of PFAS in the tissue will be affected by a change in the returned PFAS from the fallen leaves to the soil, if the leaves are collected. Furthermore, it was assumed that the same weight of leaves falls every year, that is the same weight of leaves as during the last harvest. The consequences of these assumptions will be explored in more detail in *5.3 The potential of Salix.*

For the soil, the total PFAS load was calculated using the measured area (5,000 m^2) and the average depth of the soil in the unsaturated zone (1.3 m), based on the measured groundwater level depth in this study and the measured depths in the two pipes within the Salix cultivation from May 2024, see *2.4.7 Results from previous studies.* The PFAS concentration was assumed equal throughout the depth. To determine the weight of the soil in the unsaturated zone, the general value for dry soil density 1.5 kg/dm^3 set by Naturvårdsverket (2009) was used to calculate a wet soil density of 1.4 kg/dm^3 (71 % dry weight, 29 % water content). The total PFAS load in the soil in the unsaturated zone was calculated to be 150 g. The calculated

load in soil is higher than the calculated input from Pond D, an aspect that will be further discussed in *5.1 PFAS in water and soil.*

The total load on the groundwater beneath the Salix cultivation was calculated assuming that the water conducting soil layer has a thickness of 2 meters (Bennermark & Thelin 2024) and that the pore volume is 35 %[6]. With an area of 5,000 m^2, and the assumption that 1600 ng/L is a representative value for the whole cultivation surface, the load on the groundwater was calculated to be 5.7 g. See Table 7 for a summary of the mass balance calculations.

Table 7: Summary of the PFAS loads on different medias.

Media	PFAS load (g)	Comment
Sprinkled water	22-32	Over the past 15 years
Salix	0.089-1.0	Removed past three harvests
Soil	150	Current load in soil
Groundwater	5.7	Current load on groundwater

Considering the calculated total input from the pond and the removed PFAS from the past harvests, the cultivation has removed 0.27-4.7 % of the total input.

4.5 Soil/groundwater partitioning

Seven PFAS were detected above the reporting limit in both soil and groundwater and the calculated K_d* are displayed in Table 8. PFBS and PFPeS were found in the groundwater and not in the soil and consequently, no partitioning could be calculated for these substances. Similarly, PFNA, PFDA, PFUdA, PFHpS, PFOS and PFOSA were present in the soil but not groundwater and thus no K_d* were calculated. The long-chained PFOA and PFHxS had the highest K_d* at 39 L/kg and 20 L/kg, respectively, while the short-chained substances had lower K_d* ranging from 3.1 L/kg to 8.6 L/kg. 6:2 FTS had the lowest K_d* at 1.2 L/kg.

Table 8: Partitioning between the soil and groundwater concentrations for PFAS. Wet weight concentrations were used for the soil. Values are given in L/kg.

Substance	Group	K_d*
PFBA	PFCA short	8.6
PFPeA	PFCA short	5.6
PFHxA	PFCA short	3.1
PFHpA	PFCA short	6.4
PFOA	PFCA long	39
PFHxS	PFSA long	20
6:2 FTS	FTS	1.2

[6] Joakim Robygd, University Lecturer, Engineering Geology, LTH. E-mail 25 April 2025.

4.6 Bioconcentration factors

March leaves were not included in the BCF_{soil} or $BCF_{groundwater}$ calculations as they were sampled from the ground which most likely affected the PFAS concentration. The BCF_{soil} could be calculated when PFAS was present in both the Salix and in the soil, which was the case for nine substances: PFBA, PFPeA, PFHxA, PFHpA, PFOA, PFNA, PFHxS, PFOS and 6:2 FTS. The BCF_{soil} was highest for the November leaves for all nine PFAS, with the highest value for 6:2 FTS at 1200. The November leaves had an BCF_{soil} >1 for all but two substances, PFNA and PFOS. The leaves collected in April had BCF_{soil} >1 for four substances, PFBA, PFPeA, PFHxA and 6:2 FTS. Twigs and bark only showed a BCF_{soil} >1 for one substance, 6:2 FTS. The root did not display BCF_{soil} >1 for any substance. The total PFAS BCF_{soil}, disregarding the type of PFAS, exceeded 1 only for November leaves, with a BCF_{soil} of 13. The results are depicted in Figure 22 and exact values can be found in Appendix G.

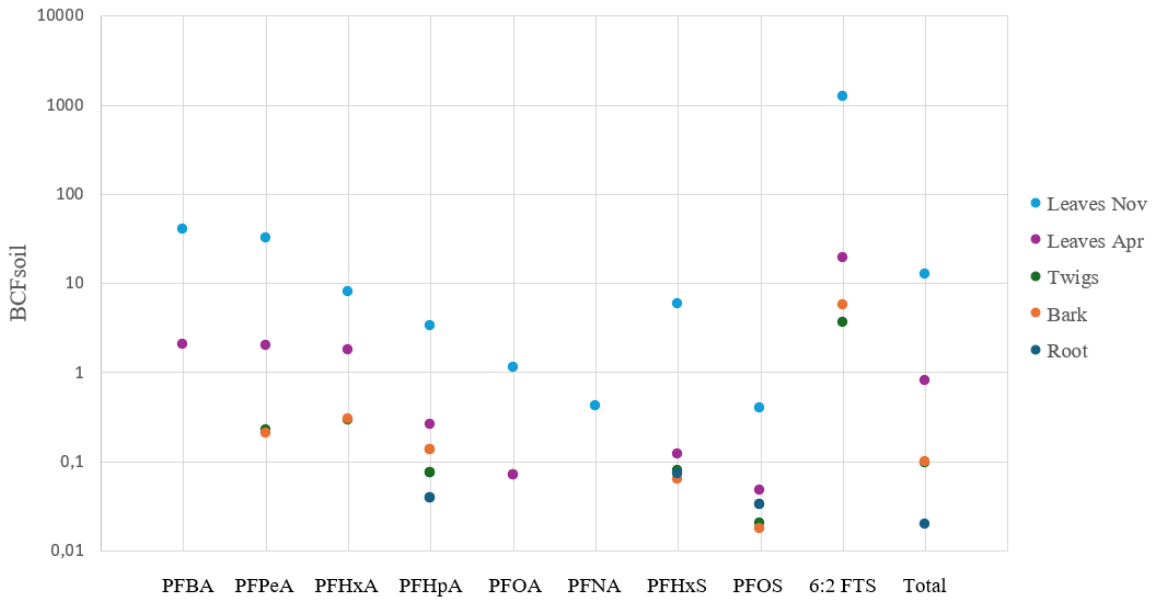


Figure 22: Bioconcentration factors for nine PFAS in the soil for Salix root, bark, twigs and November and April leaves.

The $BCF_{groundwater}$ were calculated for nine PFAS. November leaves had the highest $BCF_{groundwater}$ for all PFAS, with all $BCF_{groundwater}$ >1 with the highest value for 6:2 FTS at 1500. April leaves had $BCF_{groundwater}$ >1 for all PFAS except PFPeS, which was not detected in the leaves. Bark and twigs showed $BCF_{groundwater}$ >1 for PFPeA, PFHxS and 6:2 FTS, however the root $BCF_{groundwater}$ did only surpass 1 for PFHxS. The total PFAS $BCF_{groundwater}$, disregarding the type of PFAS, exceeds 1 for all plant parts but the root. Results are displayed in Figure 23, exact results can be found in Appendix G.

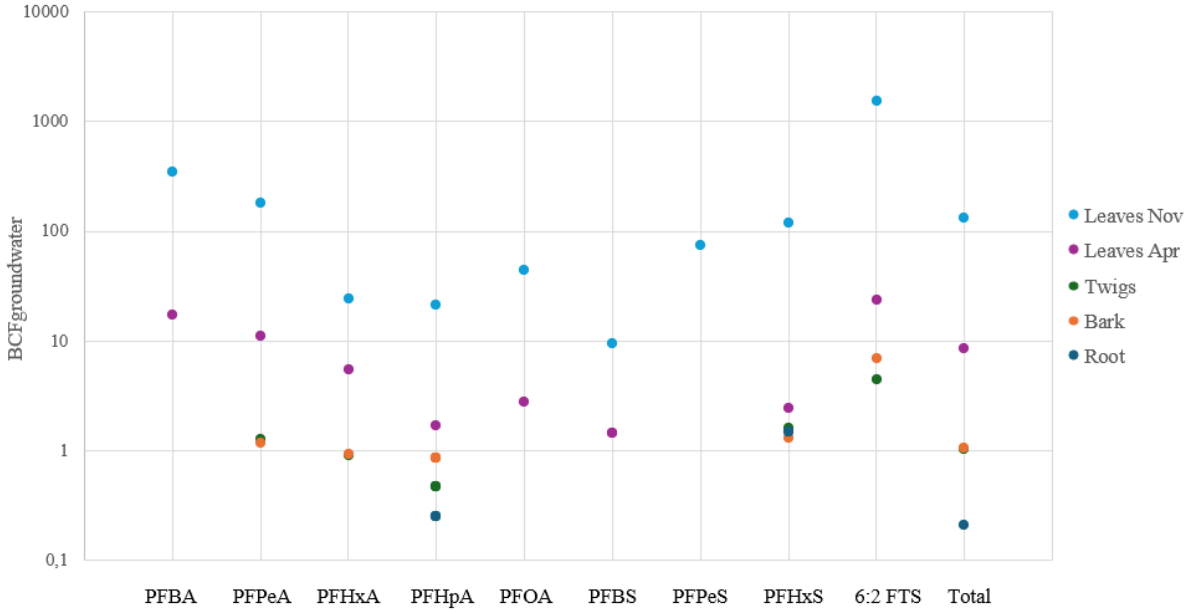


Figure 23: Bioconcentration factor for nine PFAS in groundwater for Salix root, bark, twigs and November and April leaves.

5. Discussion

5.1 PFAS in water and soil

Groundwater in the northern outskirts of the Salix cultivation and surface water from Pond D had similar total PFAS concentrations with 1600 ng/L and 1400 ng/L, respectively. However, the composition of the waters differed, where the groundwater showed a more homogeneous PFAS group composition compared to the water in the pond, as displayed in Figure 17. The groundwater consisted of short-chained PFAS (both PFCA and PFSA) to 95 %, while the water in Pond D consisted of long-chained PFAS to 45 % and short-chained to 37 %. For the soil sample, 55 % was made up by long-chained PFAS as showed in Figure 19. Groundwater constituting of mostly short-chained PFAS and soil mostly long-chained PFAS correlates to previous mentioned theory in *2.1.2 Physical and chemical properties* and *2.1.5 Transport and fate of PFAS,* that short-chain PFAS are more water soluble than long-chained, which are more prone to be sorbed to soil. The soil sample also consisted of short-chained PFAS to 44 %, which however is reasonable considering the PFAS profile in Pond D.

One interesting finding is that the water in Pond D contained relatively high concentrations of FTS, while both the groundwater and soil samples contained considerably lower concentrations. This can be explained by plant uptake, as can be seen in Figure 20 and Figure 21, but FTS are also precursors. Thus, it is possible that the FTS has been degraded to more stable PFAS. Furthermore, biodegradation of PFAA have been studied on laboratory scale, and findings propose that certain microbes have the potential to degrade PFOS and PFOA to shorter PFAA (Smorada et al. 2024). Although, to date, no studies have investigated the potential of PFAA biodegradation in nature. Future findings within this field can potentially give interesting insight in future studies at MSB, as PFOS is one of the main contaminants in Pond D and the soil but has been detected in much lower concentrations in groundwater and plants.

An interesting finding from the mass balance calculation is that, based on our assumptions, the soil has a higher PFAS content (150 g) than the calculated total input from the sprinklers (22-32 g). In all likelihood, this deviation can mainly be attributed to the simplification that the concentration in the pond in March 2025 is representative for the last 15 years. However, concentrations have probably been higher some previous years, especially during times when AFFFs containing PFAS was still used. Another factor that could explain the deviation is that the soil could have other inputs of PFAS, for instance from flooding events when PFAS can be spread through surface runoff. Furthermore, in this study only the top 0.3 meters of the soil was sampled, and the concentration was assumed equal down to 1.3 meter below ground level.

The groundwater sample was collected in the northern part of the Salix cultivation in observation well AF25, see Figure 15. Due to the location, the PFAS concentration of the groundwater sample might not be completely representative for the whole cultivation. Sampling of groundwater in the centre of the cultivation might have given a more accurate view of the PFAS concentration in the groundwater below the cultivation. Furthermore, more

samples or samples from multiple spots within the cultivation would result in a more accurate picture. Although, as seen in Figure 15 the groundwater flows in a mainly northern direction, indicating a suitable location of the groundwater pipe, however, it also means that the groundwater results most likely is affected by inputs from other PFAS point sources within the property.

Two PFAS had both K_d* values calculated in Table 8, as well as K_d values determined by SGI, that is PFOA and PFHxS. For both substances, the site-specific K_d* was higher than the K_d, at 39 L/kg and 20 L/kg compared to 5 L/kg and 4 L/kg. One explanation to the K_d* being higher than SGI's values could be that the properties of the soil at MSB Revinge are not entirely corresponding to the properties used when determining the SGI values. At the sampling site, the TOC was 4 %, while SGI's values were representative for soils with TOCs of 1-3 %. As mentioned previously, a higher organic content in the soil results in more PFAS being sorbed, see Equation 1. This is a possible explanation, as the expected amount of PFAS in the soil would be higher with an increased organic content and thereby give a higher K_d*. The pH at the site was in the represented interval 5-7, at 6.4. Furthermore, the method of calculating K_d* used in this study is not the accurate way of calculating K_d, which could also explain the big contrast between the values. The SGI values of K_d are the partitioning between the pore water and the soil and was determined experimentally by leaching tests (Pettersson et al. 2024). The K_d* was however determined by taking the concentration of a soil sample from the unsaturated zone divided by the groundwater concentration. Here, no distinction was made between the pore water and the soil, and thus, the soil sample concentration included both the concentration of the soil and of the pore water. Additionally, the concentration in soil in the saturated zone is most likely lower than the concentration in the unsaturated zone. Using the soil concentration in the saturated zone would thus lead to a lower K_d*. Also, the pore water probably contains higher concentrations than the groundwater, which if it was used in the calculations instead also would lead to a lower K_d*. This lowers the comparability of K_d* and K_d and could also potentially explain the great difference between the values.

The K_d* values in Table 8 shows that the long-chained PFAA, PFOA and PFHxS, had the highest K_d* values and short-chained PFAA as well as 6:2 FTS had lower values. This is in accordance with previously mentioned theory in *2.1.5 Transport and fate of PFAS*, that long-chained PFAA are more prone to sorb onto soil than short-chained. Among the four short-chained PFAA no pattern could be distinguished. That is, no correlation between increasing fluorinated carbon chain length and increasing K_d* could be seen.

5.2 PFAS in Salix

The highest PFAS concentration in the plants was found in the November foliage (730 µg/kg dw) of the Salix, followed by the April leaves (63 µg/kg dw) and the March leaves (22 µg/kg dw), where the latter had laid on the ground since autumn. That the highest concentrations were found in foliage aligns with the theory that PFAS accumulates in the leaves as water is transported upwards in the plants, and when the water exits the leaves through

evapotranspiration the PFAS accumulates. This is also likely the explanation to why the concentration was found to be higher in the November leaves than in the April leaves. More water has been translocated to the leaves in autumn and thus more PFAS has been able to accumulate. Furthermore, a strong seasonal variation of PFAS concentrations was found in a study conducted on white willow, black poplar and black alder in Germany, where the autumn leaves in 15 out of 18 cases contained higher concentrations than summer leaves (Würth et al. 2023).

The composition of PFAS in the November and April leaves were dominated by short-chained PFCA, with 82 % and 92 %, supporting mentioned theory concerning the correlation between translocation and PFAS chain length. The second largest portion of PFAS in the November and April leaves were constituted by FTS, mostly 6:2 FTS. The leaf samples collected from the ground in March differs from the two other leaf samples as the largest PFAS fraction was constituted by long-chained PFSA, at 57 %. Short-chained PFCA and PFSA made up merely 22 % and 0 % of the total concentration, respectively. The concentration in the November leaves showed a concentration approximately 33 times higher than the concentration in the March leaves, which can be compared to the results in the study conducted by Gobelius (2016). The concentration of PFAS in leaves from birches close to Arlanda had on average 27 times higher PFAS concentrations in June, when the leaves were collected from the trees, as opposed to in March when the leaves were collected from the ground. Gobelius explained that the leaves picked from the ground were half decayed, which is believed to be the case in this study too. The much lower PFAS concentration in the leaves on the ground is believed to depend on return of PFAS to the soil, and one explanation to why the leaves on the ground show a higher fraction of long-chained PFAS can be that short-chained PFAS are more hydrophilic and mobile, and thereby probably more easily leached out of the leaves. Less hydrophilic molecules are probably not leached out as easily from the leaves, as indicated by the November leaves and March leaves which most likely are from the same growth season, having similar total PFOS concentrations.

The PFAS concentrations in the twigs and the bark were both approximately 3.5 µg/kg dw, however, if reported with more precision, the twigs show a slightly higher concentration. Also, looking at Figure 20 and Figure 21, one can see that the PFAS compositions in both plant parts are similar. The PFAS were mainly constituted by short-chained PFCA, followed by long-chained PFSA and 6:2 FTS. Interestingly, the concentrations in the sample from the Salix stems (without bark) was lower than the reporting limits for all 47 different PFAS analysed for, sparking the question of how much of the PFAS content can be attributed to uptake and how much that depends on adherence of PFAS to surfaces due to water sprinkling. In this study, the samples were not washed before being sent to the lab, and thus, it is not certain to what degree the PFAS content in the Salix is caused by root uptake. When Gobelius et al. (2017) investigated plants close to Arlanda, a location without an irrigation system, the plant samples were washed as part of the sample preparation before drying. There, the results showed that core samples without bark had higher concentrations than the core samples with bark for both birch and spruce at all sites, which are results differing from those in this study. However, an argument against the impact of adherence is that at MSB, the sprinklers had not

been active for several months at the time of sampling, arguing that much adhered PFAS had already been naturally washed off by rain. Furthermore, the PFAS profiles for bark, twigs and leaves would most likely also look more similar to the profile in Pond D if adherence was the main source for PFAS.

To determine the proportion of PFAS which can be attributed to uptake through roots or adherence of sprinkled water is an example of a future aspect that would be interesting to explore in a potential succeeding study, as knowledge about source and reason behind the PFAS concentration can help making more accurate recommendations regarding maintenance and management of the Salix cultivation to optimise PFAS remediation. For instance, if adherence is the predominant source to the PFAS concentrations in the Salix, then the cultivation maybe should be maintained to have a high surface area to volume ratio in the shrubs, and it might then be an idea to harvest more frequently. However, even if the difference in the concentrations between bark and stem gives an indication of the role of adherence, the leaves are the part of the Salix with most potential to accumulate PFAS and thus it is probably a more suitable recommendation to maintain the cultivation to optimise the volume of leaves and implement a system for collection of leaves at the end of the seasons.

The roots displayed the lowest concentration, apart from the stem, with 1.1 µg/kg dw, and as can be seen in Figure 20 and Figure 21, the PFAS is mainly constituted by long-chained PFSA, aligning with the theory that mainly long-chained PFAS is present in the roots. The roots are not included in the calculations of how much PFAS is removed each harvest, and they only contribute to PFAS removal when the cultivation is replanted, which is approximately every 20th-25th year.

To summarise, in the November and April leaves, as well as in the bark and twigs, short-chained PFCA was the predominant PFAS group, followed by either FTS or long-chained PFSA. For the March leaves and the roots, long-chained PFSA constituted the largest portions, followed by short-chained PFCA. As the leaves in November displayed considerably higher concentrations than the rest of the plant parts, the calculations are sensitive to the assumptions made with regards to the leaves. These potential sources of errors will be further expanded upon in *5.3 The potential of Salix*, as will the discussion about how the different types of PFAS accumulate in different zones of the plant, and their link to the properties and structure of the molecule, such as chain-length and functional group.

5.3 The potential of Salix

The load on one shrub was calculated to be 26-250 µg and the load on the whole cultivation 0.036-0.34 g. The removal of PFAS per harvest was calculated to be 0.030-0.34 g/harvest and 0.089-1.0 g in total over the 15 years. Lastly, in a scenario where leaves would have been picked up after each season, 4.0-4.9 g PFAS could have been removed in total the past 15 years. The ranges were based on measured April and November concentrations in the leaves, and of course during the periods when the trees are bare, the load on the shrubs themselves will be lower. The calculations were based on the assumptions that the results from the

samples both are representative for the whole cultivation and for all the years the Salix was harvested. Although, the concentration in the Salix changes over time, and in Skirfors's presentation (2023), it was suggested that the concentration (mg/ton) in Salix decreased for each year over a three-year period. However the uptake of PFAS per area (mg/ha) increased each year, see *2.3.3 Salix's and other plants' potential for phytoremediation.* Conclusively, since it has been three years since the last harvest, and harvests are assumed to have occurred every fourth year, it means that the PFAS removal might have been overestimated, as a concentration for younger Salix was used while still using the biomass (30,000 kg) for older Salix.

Another assumption made during the calculations was that every PFAS not detected over the reporting limit was assumed to have a concentration of zero, leading to a potential underestimation of the concentrations if they were in fact above zero. Additionally, it is a simplification to assume that the concentrations in the shrubs each harvest were the same, since it is likely that the concentration in the soil have increased over the years due to the irrigation from Pond D. When the concentration in the surrounding media increases, it is fair to assume that the concentrations in the plants also increase. Furthermore, when calculating the potential removal of PFAS during the past 15 years, with a hypothetically implemented leaf collection system, it was assumed that the weight of the leaves always were the same as for a fourth-year Salix cultivation, hence overestimating the weight of the returned leaves to the ground for the years when the shrubs were smaller, and underestimating for when they were bigger. Also, if the leaves were collected, the PFAS would not return to the system, which too can lead to an overestimation of the PFAS that could have been removed the past 15 years. The calculations have also not taken into consideration that during years when harvest have occurred in spring, leaves will have grown out and will fall in autumn as well. Moreover, the mass balance estimations have not taken into account that the size of the root system have changed over the years, a factor which probably influence the uptake of PFAS and thus the load on the shrubs. To summarise, it is a complex system where a lot of simplifications have been made and rebound effects overlooked, to make estimations on PFAS load and PFAS removal feasible.

During a four-year period, 0.0074-0.084 g of PFAS has been removed yearly. If a leaf collection system was implemented the years between harvests, the removal would be 0.25-0.33 g yearly. Meanwhile, the yearly input through irrigation from Pond D is 1.4-2.2 g, based on the sampling in March. Over a 50 year-period, with a collection system, 12.5-16.5 g could be removed in total, and it would approximately take 3,000-4,000 years to remediate 1 kg PFAS. Compared to the estimated load of PFAS7 in soil (5-15 kg) and groundwater (0.58-1.2 kg) the remediation may seem underwhelming, however, in relation to what is fed from Pond D each year this way of remediation has some potential, especially with regards to the relatively low costs and maintenance.

An important aspect of the findings from the study is that the November leaves held a higher PFAS concentration that the leaves from April. This coincides with what was expected, since more water has had time to transport to the leaves and through evapotranspiration the PFAS is let to accumulate. The last harvest conducted took place in May 2022, and considering that

the leaves are relatively new during spring, this was probably not the optimal time of year to harvest with regards to PFAS remediation. Also, according to Hollsten et al. (2013), the best time to harvest Salix is after the leaves have fallen and the average temperature is below 4 °C, however, these recommendations were not proposed with phytoremediation optimisation as the main goal. Since the autumn leaves contain the highest concentrations, it may therefore be a good idea to harvest the shrubs in autumn in combination with picking up the leaves. Although, as previously mentioned, in the calculation of potential PFAS removal with spring harvests, the leaves that grow out, fall and are picked up during harvest years were excluded, leading to an underestimation of potential PFAS removed in case of spring harvests. Thus, the difference between potential PFAS removal with spring and autumn harvests is probably smaller.

As the leaves, especially those from November, contained significantly higher PFAS concentrations than the other plant parts, the calculations made in this study have had high sensitivity towards the assumptions made with regards to the leaves. Some assumptions include the water content in the November leaves as well as the allometric proportions and thereby weight of the leaves. The leaves were assumed to constitute 5.0 % of the above-ground biomass, as this is a rough estimate used in the Salix industry. The sensitivity of the results to the leaf-assumptions can be demonstrated with an example. For instance, if the leaves instead of 5.0 %, would be assumed to take up 7.0 % of the weight of the shrubs, the PFAS removal per harvest would be 0.037-0.46, instead of 0.030-0.34 g, the total removed PFAS over the last 15 years would be 0.11-1.4 g, instead of 0.089-1.0 g and the total removed PFAS if a leaf collection system was implemented would be 5.5-6.8 g, instead of 4.0-4.9 g over the past 15 years. Thus, values adopted in the calculations that affect the PFAS load on leaves can have a large effect on the results. To understand more about the dependencies on assumed values, a more thorough and systematic sensitivity analysis would have to be conducted to know for which parameters high accuracy are most important.

As mentioned in *2.3.3 Salix's and other plants' potential for phytoremediation,* SGI brought forward a few parameters to consider when choosing species for phytoremediation. Salix thrives in Swedish climate, produces biomass quickly, is non-invasive to Sweden, and can be regularly harvested and regenerated. However, Salix sheds its leaves which does not coincide with one criterion, hence a leaf collection system is recommended. Further, Pettersson & Enell (2021) mentioned that the number of studies on phytoremediation of PFAS is limited, and more research regarding different plants' potential is needed. Gobelius et al. (2017) proposed a system for phytoremediation which could potentially remove 1.4 g PFAS /(year*ha), where the system included birches, spruce and an understory of herbaceous plants and yearly removal of leaves. The calculated potential removal through Salix by harvest and leaf collection at MSB Revinge is 0.51-0.66 g /(year*ha), which is less than the findings from Gobelius et al (2017). However, a direct comparison is complicated by the differing conditions at the sites. Gobelius et al. (2017) investigated three locations, where two out of three had considerably higher groundwater concentrations, and one out of three had considerably higher soil concentrations than found in the Salix cultivation at MSB Revinge. Plants will accumulate more PFAS when more PFAS is present in surrounding media, which

could pose as an explanation to the lower amount PFAS accumulated in the Salix at MSB Revinge. Further, Pettersson & Enell (2021) mentioned that previous findings suggest that Salix has a good potential to accumulate PFAS. Conclusively, assessed by current knowledge, Salix is a suitable choice for phytoremediation of PFAS.

5.4 Bioconcentration factors

Overall, the November leaves had the highest BCF for all PFAS for both soil and groundwater, see Figure 22 and Figure 23. According to the definition of BCF, November leaves hyperaccumulate all PFAS from groundwater, except for PFBS which is only accumulated. The calculated BCF_{soil} shows that the November leaves accumulate four and hyperaccumulate three out of the total nine PFAS. The November leaves showed especially high values for 6:2 FTS for both the soil and groundwater. This coincides with previous findings by Gobelius (2016), who determined BCF for various plants at Arlanda where 6:2 FTS had the highest accumulation in most samples. The other plant parts also displayed ability to accumulate some of the PFAS, however not to the same degree as the November leaves.

A pattern can be seen for the accumulation of PFCA in the November leaves in Figure 22, where the BCF_{soil} decreases with increasing perfluorinated chain length, from PFBA to PFNA. This coincides with previous mentioned theory in *2.3.1 Plant uptake and translocation,* and can be attributed to the increasing hydrophobicity with increasing perfluorinated carbon chain length, which inhibits the ability of PFAS to translocate along with water and nutrients upwards in the plant. For $BCF_{groundwater}$ (Figure 23), a similar trend can be observed for PFCA, although it is less distinct. Furthermore, the April leaves partly follow the same pattern as the November leaves for both BCF_{soil} and $BCF_{groundwater}$.

Comparing the BCF for PFCA and PFSA in the November leaves with the same number of carbons shows that for BCF_{soil}, PFHxA > PFHxS and PFOA > PFOS, and for $BCF_{groundwater}$ PFBA > PFBS, PFPeA > PFPeS and PFHxA < PFHxS. To summarise, in four out of the five cases, PFCA accumulates more than PFSA for PFAS with equal number of carbon atoms, agreeing with theory. For April leaves, a similar trend can be observed. Although, an equal number of carbon atoms is not the same as an equal number of perfluorinated carbons, where for instance PFOS has one more perfluorinated carbon than PFOA.

When calculating the $BCF_{groundwater}$, it was assumed that the roots were in contact with the groundwater, however, this is hard to know. In March, the groundwater level was measured to 1.57 meters below ground level and the roots estimated to be 0.75 meters. Although, previous samplings have shown a higher groundwater table, and during periods with heavy or frequent rainfalls, or even floodings, it is probably even higher. Moreover, when the shrub was dug up, the roots seemed to have been broken off, indicating that they continued further, and considering that the cultivation is 15 years old, an evolved root system is expected. It is a possibility that the roots are in contact with the groundwater only during some periods of the year, and it could be that these periods are a peak in PFAS uptake, especially for short-

chained PFCA. Furthermore, BCF is only calculated for the media which the plants are in direct contact with and thus, the PFAS adsorbed from the sprinkled water were not taken into consideration.

5.5 Incineration of harvested Salix

After previous harvest of the Salix at MSB Revinge, the biomass was sent to Sysav as park and garden waste and incinerated at the waste facility. The temperature reaches approximately 900-1000 ˚C in the incineration at Sysav (Sysav 2023), which cannot guarantee irreversible destruction of PFAS. Studies regarding the temperature needed for destruction of PFAS have been conducted, however, not with coherent results. For instance, when Strandberg et al. (2021) reviewed studies on the destruction of PFAS, temperatures needed ranged from 750 to 1000 ˚C, while Holmström et al. (2012) recommends an incineration temperature of at least 1100 ˚C for a potential destruction of PFAS. Further, Strandberg et al. (2021) investigated the PFAS content in incineration residues from 27 incineration plants in Sweden, where PFAS were found present in all but five of the investigated facilities. Because of the likely insufficient incineration temperatures at Sysav, together with the knowledge that many incineration facilities in Sweden have PFAS in their residues, it is probable that the PFAS in the Salix is not irreversibly destroyed when incinerated at Sysav. However, the PFAS in incineration residues is out of the scope of this study and will not be further discussed.

During previous harvest, the Salix was categorised as non-hazardous waste. If the Salix would be categorised as hazardous waste, it would be incinerated at a temperature of 1200 ˚C (Sysav n.d). Incineration at 1200 ˚C indicates a higher probability of the PFAS being irreversibly destroyed. However, classification of the Salix as hazardous waste would result in a bigger economic burden for MSB. There are PFAS limit values for waste, where the treatment method should ensure destruction of the PFAS (Naturvårdsverket 2024). The PFAS covered are PFOA, PFOS and PFHxS, as well as their related derivates and salts, and the limit values ranges from 1 mg/kg to 50 mg/kg. The highest concentration in the Salix were in the November leaves at 0.73 mg/kg dw, which does not exceed the lowest limit value even when considering the total PFAS concentration. However, as only one sample of the November leaves was taken in this study together with a measuring insecurity of ± 37 % from the lab analysis, there is a possibility that the concentration in the leaves could surpass the lowest limit values. Therefore, it should be considered to possibly handle the autumn leaves separately as hazardous waste, to achieve destruction of PFAS.

5.6 Future studies

Suggestions for future studies includes comparing washed and un-washed samples with each other, to get a better understanding of how much of the PFAS content depend on uptake and how much comes from adhered PFAS from the sprinkled water. If this was further investigated, then there would be more information to base the recommendations for

maintenance of the Salix cultivation on, see *5.2 PFAS in Salix*. When Gobelius et al. (2017) studied phytoremediation, samples were washed with Milli-Q water, and in a study by Nassazzi et al. (2022), an even more thorough method for washing is explained. Alternatively, some Salix shrubs could be sheltered from the sprinklers, ruling out PFAS adherence. One advice could be to implement a more soil-directed irrigation system, to prevent airborne spreading of PFAS down-wind, especially if future studies would find root uptake as the main source for PFAS in the Salix.

Another suggestion is to sample Salix more regularly over the years and throughout the seasons. This way, an improved understanding can be acquired regarding how the PFAS concentration varies in the Salix and possibly get an indication on whether the phytoremediation potential decreases with age.

Since it cannot be guaranteed that PFAS is destroyed completely at 1000 °C, it is also a suggestion to more thoroughly investigate what the most optimal waste disposal of the harvested Salix is. Within the scope of LIFE SOuRCE, incineration will be investigated, and the final report is planned to be published in 2025 (Skirfors 2023).

Apart from remediation from the Salix, another suggestion could be to investigate the potential of implementing phytoremediation through native plants in Pond D. For example, Solna municipality has introduced phytoremediation through *Carex* in the stream Igelbäcken, which has shown noticeable improvements, especially with regards to the concentrations of short-chained PFAS (Stockholms stad 2023). In Igelbäcken, the Carex is reaped each year and upkeeping efforts are performed 2-3 times per season. Since PFOS is the dominating PFAS in the pond, an interesting future study would be to investigate if there are any aquatic plants efficient in accumulating PFOS.

6. Conclusions and recommendations

In this master's thesis, a comprehensive literature study, along with sampling of plant parts, soil, groundwater and surface water have been conducted, with the aim of investigating the potential for phytoremediation of PFAS through Salix at MSB Revinge's training facility. Hopefully, the findings can help improving the knowledge of Salix's capacity with regards to remediating PFAS, and can serve as support to municipal rescue services, as well as other PFAS contaminated areas, in Sweden.

The literature study concludes that *Salix viminalis* is a suitable choice for phytoremediation, as it takes up large amount of water, produce large biomass, are easy to manage, thrive in the climate, are non-invasive and allow regular above-ground harvest. However, they do not retain their leaves, and a leaf collection system is therefore highly recommended. Other plant species, such as birch and sunflower, have also been demonstrated to have potential for phytoremediation of PFAS.

The sampling at MSB Revinge shows that the PFAS accumulating potentials, based on the dry weight concentrations, in the Salix plant tissue decrease in following order: leaves > twigs > bark > root > stem. Furthermore, the leaf samples contained PFAS concentrations in the order: November leaves > April leaves > March leaves (collected from the ground), with the November leaves having PFAS concentrations >10 times higher than the April and March leaves, and >100 times higher than the other examined plant parts. Different plant parts were shown to accumulate different types of PFAS, where the leaves still attached to the shrubs, bark and twigs mainly contained short-chained PFAS, whereas the roots had a higher fraction of long-chained PFAS. Moreover, for the November and April leaves, the BCF_{soil} decreased with increasing PFAS chain-length. For $BCF_{groundwater}$ the trend was not as clear, but overall, the same pattern could be observed for PFCA. In general, the BCFs for November and April leaves were higher for PFCA than PFSA, when comparing PFAS with the same number of carbon atoms.

The PFAS load on the groundwater and soil below the Salix cultivation was calculated to be 5.7 g and 150 g, respectively. The load on the whole Salix cultivation including roots was estimated to 0.036-0.34 g, where the range is based on April and November leaf concentrations. The input of PFAS through irrigation from the pond was calculated to 22-32 g after 15 years. Moreover, the Salix cultivation contributes to a measurable reduction of PFAS, with an estimated 0.034-0.30 g removal each harvest, and a 0.0074-0.084 g removal per year. If previous harvests had been supplemented with annual leaf collections, the potential PFAS removal is estimated to be 0.25-0.33 g per year.

The primary recommendation to MSB is to implement a system for leaf collection. It is of importance that the leaves are collected before starting to decay since PFAS return to the soil when the leaves decompose, as supported by the results presented in this thesis. Furthermore, the Salix have a lifetime of about 20-25 years. Since the findings from the literature review suggests that Salix is a suitable choice for phytoremediation, replanting Salix seems like a reasonable course of action. However, it can be advisable to have another crop at the site a

couple of years before replanting Salix. Recommendations for future studies includes regular sampling of the Salix over the years and throughout the seasons, a more detailed assessment regarding uptake of PFAS through roots versus adherence from sprinkled water, an investigation of what the most optimal waste management is and research on the potential for implementing phytoremediation in the pond. Lastly, in 2025, LIFE SOuRCE is publishing their report on remediation techniques for PFAS, including phytoremediation. A report recommended to read to stay up to date with the latest findings regarding solving the PFAS problem.

7. References

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Appendix

A. Previous sampling in Pond D by Sweco and WSP

The previous sampling in Pond D conducted by Sweco and WSP from the years 2009-2022 are compiled in Table A. The table shows the fluctuating concentrations of PFOS and PFOA in the pond.

Table A: Previous measurements of PFOS and PFOA in Pond D conducted by Sweco and WSP

Date	PFOS ng/L	PFOA ng/L	Total ng/L	Conducted by
2009-04-07	8600	370	9000	Sweco
2009-10-07	13 000	570	14000	Sweco
2010-04-21	710	210	920	Sweco
2011-04-26	1100	60	1200	Sweco
2011-10-31	2300	190	2500	Sweco
2015-06-01	370	31	400	Sweco
2016-06-13	280	53	330	Sweco
2017-06-01	200	36	240	WSP
2018-06-12	130	46	180	WSP
2019-04-29	149	56	210	WSP
2019-09-19	200	63	260	WSP
2020-04-29*	230	65	290	WSP
2020-09-17*	250	80	330	WSP
2021-04-28*	120	42	160	WSP
2021-10-21*	200	55	260	WSP
2022-05-25	60	16	76	WSP
2022-09-26	96	33	130	WSP

**Estimated from diagram*

B. Results groundwater

The results from the groundwater sampling are compiled in Table B. The sampling methodology is described in *3.1.2 Groundwater and surface water sampling* and the sample was analysed by Eurofins Environment Testing Sweden AB.

Table B: Analysis results for groundwater. Sampling of groundwater was conducted the 6th of March 2025.

Substance (abbrev.)	Group	Groundwater (ng/L)
PFBA	Short PFCA	140
PFPeA	Short PFCA	640
PFHxA	Short PFCA	530
PFHpA	Short PFCA	200
PFOA	Long PFCA	20
PFNA	Long PFCA	<0.30
PFDA	Long PFCA	<0.30
PFUdA	Long PFCA	<0.30
PFDoA	Long PFCA	<0.30
PFTrDA	Long PFCA	<1.0
PFTeDA	Long PFCA	<1.0
PFHxDA	Long PFCA	<0.30
PFBS	Short PFSA	32
PFPeS	Short PFSA	14
PFHxS	Long PFSA	38
PFHpS	Long PFSA	<0.30
PFOS	Long PFSA	<2.0
PFNS	Long PFSA	<0.30
PFDS	Long PFSA	<0.30
PFUnDS	Long PFSA	<0.30
PFDoS	Long PFSA	<1.0
PFTrDS	Long PFSA	<0.30
4:2 FTS	FTS	<0.30
6:2 FTS	FTS	19
8:2 FTS	FTS	<0.30
PFOSA	FOSA	<0.30
MeFOSA	FOSA	<1.0
EtFOSA	FOSA	<1.0
FOSAA	FOSAA	<0.30
MeFOSAA	FOSAA	<0.30
EtFOSAA	FOSAA	<0.30
MeFOSE	FOSE	<1.0
EtFOSE	FOSE	<1.0
HPFHpA	Other PFAS	<0.30
P37DMOA	Other PFAS	<2.0

Sum PFAS4 (EU EFSA)	58
Sum PFAS SLV 11	1600
Sum PFAS	1600
Sum PFAS20 ((EU) 2020/2184)	1600
Sum PFAS21 (LIVSFS 2022:12)	1600

C. Results surface water

The results from the surface water (Pond D) sampling are compiled in Table C. The sampling methodology is described in *3.1.2 Groundwater and surface water sampling* and the sample was analysed by Eurofins Environment Testing Sweden AB.

Table C: Analysis results for surface water in Pond D. Sampling of surface water was conducted the 6th of March 2025.

Substance (abbrev.)	Group	Surface water (ng/L)
PFBA	Short PFCA	60
PFPeA	Short PFCA	190
PFHxA	Short PFCA	180
PFHpA	Short PFCA	84
PFOA	Long PFCA	49
PFNA	Long PFCA	12
PFDA	Long PFCA	5.6
PFUdA	Long PFCA	0.92
PFDoA	Long PFCA	0.43
PFTeDA	Long PFCA	<1.0
PFHxDA	Long PFCA	<0.30
PFTrDA	Long PFCA	<1.0
PFBS	Short PFSA	14
PFPeS	Short PFSA	10
PFHxS	Long PFSA	96
PFHpS	Long PFSA	4.5
PFOS	Long PFSA	480
PFNS	Long PFSA	0.35
PFDS	Long PFSA	<0.30
PFUnDS	Long PFSA	<0.30
PFDoS	Long PFSA	<1.0
PFTrDS	Long PFSA	<0.30
4:2 FTS	FTS	<0.30
6:2 FTS	FTS	210
8:2 FTS	FTS	34
PFOSA	FOSA	7
MeFOSA	FOSA	<1.0
EtFOSA	FOSA	<1.0
FOSAA	FOSAA	0.42
MeFOSAA	FOSAA	1.1
EtFOSAA	FOSAA	0.32
MeFOSE	FOSE	<1.0
EtFOSE	FOSE	<1.0
HPFHpA	Other PFAS	<0.30

P37DMOA	Other PFAS	<2.0
Sum PFAS4 (EU EFSA)		640
Sum PFAS SLV 11		1400
Sum PFAS		1400
Sum PFAS20 ((EU) 2020/2184)		1200
Sum PFAS21 (LIVSFS 2022:12)		1400

D. Results soil

The results from the soil sampling are compiled in Table D, with the sampling methodology presented in *3.1.3 Soil sampling*. The sample was analysed by Eurofins Environment Testing Sweden AB.

Table D: Analysis results for soil. Sampling of soil was conducted the 6th of March 2025.

Substance (abbrev.)	Group	Soil (µg/kg dw)
PFBA	Short PFCA	1.7
PFPeA	Short PFCA	5.1
PFHxA	Short PFCA	2.3
PFHpA	Short PFCA	1.8
PFOA	Long PFCA	1.1
PFNA	Long PFCA	1.1
PFDA	Long PFCA	0.18
PFUdA	Long PFCA	0.12
PFDoA	Long PFCA	<0.10
PFTeDA	Long PFCA	<0.030
PFHxDA	Long PFCA	<0.030
PFTrDA	Long PFCA	<0.10
PFBS	Short PFSA	<0.030
PFPeS	Short PFSA	<0.10
PFHxS	Long PFSA	1.1
PFHpS	Long PFSA	0.058
PFOS	Long PFSA	10
PFNS	Long PFSA	<0.20
PFDS	Long PFSA	<0.030
PFUnDS	Long PFSA	<1.0
PFDoS	Long PFSA	<1.0
PFTrDS	Long PFSA	<1.0
4:2 FTS	FTS	<0.030
6:2 FTS	FTS	0.033
8:2 FTS	FTS	<0.10
PFOSA	FOSA	0.33
MeFOSA	FOSA	<0.030
EtFOSA	FOSA	<0.20
FOSAA	FOSAA	<0.10
MeFOSAA	FOSAA	<0.030
EtFOSAA	FOSAA	<0.10
MeFOSE	FOSE	<0.030
EtFOSE	FOSE	<0.10
HPFHpA	Other PFAS	<0.10
P37DMOA	Other PFAS	<0.50

Dry weight	70.6 %
Sum PFAS incl. ½ LOQ	27
Sum PFAS 4 incl. ½ LOQ	13
Sum PFAS excl. LOQ	25
Sum PFAS 4 excl. LOQ	13

E. Results Salix leaves

A complication of concentrations of PFAS in the sampled leaves from November, April and March (from the ground) is shown in Table E, with methodology described in *3.1.4 Salix sampling.* Samples were analysed by Eurofins Environment Testing Sweden AB.

Table E: Analysis results for Salix leaves. Sampling of leaves was conducted in November 2024, 7th of March 2025 and 15th of April 2025.

Substance (abbrev.)	Group	Leaves Nov. (µg/kg dw)	Leaves Mar. (µg/kg dw)	Leaves Apr. (µg/kg dw)
PFBA	Short PFCA	160	<1.0	11
PFPeA	Short PFCA	380	2.7	32
PFHxA	Short PFCA	43	1.6	13
PFHpA	Short PFCA	14	0.35	1.5
PFOA	Long PFCA	2.9	0.31	0.25
PFNA	Long PFCA	1.1	0.26	<0.1
PFDA	Long PFCA	<0.10	0.19	<0.10
PFUdA	Long PFCA	<0.10	<0.10	<0.10
PFDoA	Long PFCA	<0.10	<0.10	<0.10
PFTrDA	Long PFCA	<0.10	<0.10	<0.10
PFTeDA	Long PFCA	<0.10	<0.10	<0.10
PFHxDA	Long PFCA	<0.30	<0.30	<0.30
PFODA	Long PFCA	<0.30	<0.30	<0.30
PFPrS	Ultra short PFSA	<0.30	<0.30	<0.30
PFBS	Short PFSA	1	<0.10	0.21
PFPeS	Short PFSA	3.5	<0.10	<0.10
PFHxS	Long PFSA	15	0.34	0.42
PFHpS	Long PFSA	<0.30	<0.30	<0.30
PFOS	Long PFSA	9.3	12	1.5
PFNS	Long PFSA	<0.30	<0.30	<0.30
PFDS	Long PFSA	<1.0	<1.0	<1.0
PFUnDS	Long PFSA	<1.0	<1.0	<1.0
PFDoS	Long PFSA	<1.0	<1.0	<1.0
PFTrDS	Long PFSA	<1.0	<1.0	<1.0
4:2 FTS	FTS	<0.30	<0.30	<0.30
6:2 FTS	FTS	96	2.1	2.0
8:2 FTS	FTS	0.31	0.97	0.96
10:2 FTS	FTS	<0.30	<0.30	<0.30
PFOSA	FOSA	<0.10	0.37	<0.10
MeFOSA	FOSA	<1.0	<1.0	<1.0
EtFOSA	FOSA	<1.0	<1.0	<1.0
FOSAA	FOSAA	<0.30	<0.30	<0.30
MeFOSAA	FOSAA	<0.10	<0.10	<0.10

EtFOSAA	FOSAA	<0.10	<0.10	<0.10
EtFOSE	FOSE	<3.0	<3.0	<3.0
MeFOSE	FOSE	<3.0	<3.0	<3.0
PFBSA	FASA (not FOSA)	<0.30	<0.30	<0.30
PFHxSA	FASA (not FOSA)	0.11	0.33	<0.10
PFEESA	PFAS ether	<0.30	<0.30	<0.30
PFMBA	PFAS ether	<0.30	<0.30	<0.30
PFMPA	PFAS ether	<0.30	<0.30	<0.30
6:2 Cl-PFAES	PFAS ether	<0.30	<0.30	<0.30
8:2 Cl-PFAES	PFAS ether	<0.30	<0.30	<0.30
DONA	PFAS ether	<0.10	<0.10	<0.10
NFDHA	PFAS ether	<0.30	<0.30	<0.30
PFECHS	Other PFAS	<0.30	<0.30	<0.30
HPFHpA	Other PFAS	<0.30	<0.30	<0.30
Water content		12.1 %	49.5 %	77.8 %
Sum PFAS TOT exkl. LOQ		730	22	63
Sum PFAS22 exkl. LOQ		730	20	62
Sum PFAS4 exkl. LOQ		28	13	2.2

F. Results Salix twigs, stem, bark, root

A complication of concentrations of PFAS in the sampled plant parts twigs, bark, stem and root at is shown in Table F, with methodology described in *3.1.4 Salix sampling.* Samples were analysed by Eurofins Environment Testing Sweden AB.

Table F: Analysis results for twigs, bark, stem and root. Sampling of plant parts was conducted the 7th of March 2025.

Substance (abbrev.)	Group	Twigs (µg/kg dw)	Bark (µg/kg dw)	Stem (µg/kg dw)	Root (µg/kg dw)
PFBA	Short PFCA	<1.0	<1.0	<1.0	<1.0
PFPeA	Short PFCA	1.7	1.5	<1.0	<1.0
PFHxA	Short PFCA	1.0	1.0	<1.0	<1.0
PFHpA	Short PFCA	0.20	0.35	<0.10	0.16
PFOA	Long PFCA	<0.10	<0.10	<0.10	<0.10
PFNA	Long PFCA	<0.10	<0.10	<0.10	<0.10
PFDA	Long PFCA	<0.10	<0.10	<0.10	<0.10
PFUdA	Long PFCA	<0.10	<0.10	<0.10	<0.10
PFDoA	Long PFCA	<0.10	<0.10	<0.10	<0.10
PFTrDA	Long PFCA	<0.10	<0.10	<0.10	<0.10
PFTeDA	Long PFCA	<0.10	<0.10	<0.10	<0.10
PFHxDA	Long PFCA	<0.30	<0.30	<0.30	<0.30
PFODA	Long PFCA	<0.30	<0.30	<0.30	<0.30
PFPrS	Ultrashort PFSA	<0.30	<0.30	<0.30	<0.30
PFBS	Short PFSA	<0.10	<0.10	<0.10	<0.10
PFPeS	Short PFSA	<0.10	<0.10	<0.10	<0.10
PFHxS	Long PFSA	0.13	0.10	<0.10	0.18
PFHpS	Long PFSA	<0.30	<0.30	<0.30	<0.30
PFOS	Long PFSA	0.30	0.25	<0.10	0.75
PFNS	Long PFSA	<0.30	<0.30	<0.30	<0.30
PFDS	Long PFSA	<1.0	<1.0	<1.0	<1.0
PFUnDS	Long PFSA	<1.0	<1.0	<1.0	<1.0
PFDoS	Long PFSA	<1.0	<1.0	<1.0	<1.0
PFTrDS	Long PFSA	<1.0	<1.0	<1.0	<1.0
4:2 FTS	FTS	<0.30	<0.30	<0.30	<0.30
6:2 FTS	FTS	0.18	0.27	<0.10	<0.10
8:2 FTS	FTS	<0.30	<0.30	<0.30	<0.30
10:2 FTS	FTS	<0.30	<0.30	<0.30	<0.30
PFOSA	FOSA	<0.10	<0.10	<0.10	<0.10
MeFOSA	FOSA	<1.0	<1.0	<1.0	<1.0
EtFOSA	FOSA	<1.0	<1.0	<1.0	<1.0
FOSAA	FOSAA	<0.30	<0.30	<0.30	<0.30
MeFOSAA	FOSAA	<0.10	<0.10	<0.10	<0.10

EtFOSAA	FOSAA	<0.10	<0.10	<0.10	<0.10
EtFOSE	FOSE	<3.0	<3.0	<3.0	<3.0
MeFOSE	FOSE	<3.0	<3.0	<3.0	<3.0
PFBSA	FASA (not FOSA)	<0.30	<0.30	<0.30	<0.30
PFHxSA	FASA (not FOSA)	<0.10	<0.10	<0.10	<0.10
PFEESA	PFAS ether	<0.30	<0.30	<0.30	<0.30
PFMBA	PFAS ether	<0.30	<0.30	<0.30	<0.30
PFMPA	PFAS ether	<0.30	<0.30	<0.30	<0.30
6:2 Cl-PFAES	PFAS ether	<0.30	<0.30	<0.30	<0.30
8:2 Cl-PFAES	PFAS ether	<0.30	<0.30	<0.30	<0.30
DONA	PFAS ether	<0.10	<0.10	<0.10	<0.10
NFDHA	PFAS ether	<0.30	<0.30	<0.30	<0.30
PFECHS	Other PFAS	<0.30	<0.30	<0.30	<0.30
HPFHpA	Other PFAS	<0.30	<0.30	<0.30	<0.30
Water content		52.7 %	50.8 %	56.0 %	68.9 %
Sum PFAS TOT exkl. LOQ		3.5	3.5	ND	1.1
Sum PFAS22 exkl. LOQ		3.5	3.5	ND	1.1
Sum PFAS4 exkl. LOQ		0.43	0.35	ND	0.93

G. Results BCF

The calculated BCF_{soil} and $BCF_{groundwater}$ are shown in Table G1 and Table G2, and methodology is described in *3.4.4 Bioconcentration factor.*

Table G1: Calculated BCF_{soil} for nine PFAS as well as total BCF for each plant part.

Substance	Root	Bark	Twigs	Leaves November	Leaves April
PFBA	-	-	-	40	2.0
PFPeA	-	0.20	0.22	32	2.0
PFHxA	-	0.30	0.29	8.0	1.8
PFHpA	0.039	0.14	0.074	3.3	0.26
PFOA	-	-	-	1.1	0.071
PFNA	-	-	-	0.43	-
PFHxS	0.072	0.063	0.079	5.8	0.12
PFOS	0.033	0.017	0.020	0.40	-
6:2 FTS	-	5.7	3.7	1200	19
Total	0.020	0.10	0.10	13	0.80

Table G2: Calculated $BCF_{groundwater}$ for nine as well as total BCF for each plant part.

Substance	Root	Bark	Twigs	Leaves November	Leaves April
PFBA	-	-	-	340	17
PFPeA	-	1.2	1.3	180	11
PFHxA	-	0.93	0.89	24	5.4
PFHpA	0.25	0.86	0.47	21	1.7
PFOA	-	-	-	44	2.8
PFBS	-	-	-	9.4	1.5
PFPeS	-	-	-	75	-
PFHxS	1.5	1.3	1.6	120	2.5
6:2 FTS	-	7.0	4.5	1500	23
Total	0.21	1.0	1.0	130	8.5