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The potential for binding PFAS using biochar and phytoremediation

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The potential for binding PFAS using biochar and phytoremediation

Potentialen för att binda PFAS med hjälp av biokol och fytosanering

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The potential for Binding PFAS Using Biochar and Phytoremediation

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Master thesis, 30 credits, in *physical geography and ecosystem science*

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Abstract

PFAS is a group of long-lasting chemicals with multiple known toxicological effects which has been found everywhere in nature. One of the possible source points of PFAS is via landfill through leachate leaving the area. It is therefore important to investigate and implementing remediation techniques to reduce the contamination risk. This thesis focuses on two components, first a literature study with the goal of collecting information regarding PFAS and how they are affected by biochar and phytoremediation. The second part focuses on a landfill outside of Höganäs and the usages of *Salix* and *Miscanthus* to decrease the leachate volumes within the area, as well as investigating *Salix* for phytoremediation of PFAS. The leachate uptake for both species was calculated using a Penman – Monteith equation for the evapotranspiration, from which water balances for the landfill could be made, and the PFAS uptake used an equation where that was based on the biomass production. The literature review showed that the combination of biochar and phytoremediation could be useful as these two techniques targets different variations of PFAS which shows a potential for them to target a larger range of PFAS variations if they were to be applied together and should be further investigated. The case study results showed that *Miscanthus* and *Salix* can be useful to reduce the amount of leachate within the landfill, leading to lower release rates from the landfill. Furthermore, based solely on biomass production it does seem that *Salix* can be useful for accumulating PFAS which can then be removed from the landfill.

Key words: Physical geography, ecosystem analysis, PFAS, Per-and polyfluoroalkyl substances, remediation, phytoremediation, biochar

Sammanfattning

PFAS är en grupp av långvariga kemikalier med flera vedertagna toxiska egenskaper som finns överallt i naturen. En möjlig källa är PFAS som släpps ut via lakvatten från deponier. Det är därför viktigt att undersöka och anlägga saneringsmetoder för att minska spridningsrisken. Denna uppsats fokuserar på två komponenter, först en litteraturstudie med fokus på att hämta information om PFAS och hur de påverkas av biokol och fytosanering. Den andra delen fokuserar på en deponi utanför Höganäs och användandet av *Salix* och *Miscanthus* för att minska lakvattenvolymerna inom deponin, såväl som en undersökning av *Salix* för fytosanering av PFAS. Lakvattenupptaget i växterna beräknades med en Penman – Monteith ekvation, som senare användes för att göra vatten-balansberäkningar. Beräkningarna för PFAS upptag inom *Salix* använde en ekvation där PFAS upptaget var baserat på biomassaproduktion. Litteraturstudien visade att kombinationen av biokol och fytosanering skulle kunna vara användbar då dessa två tekniker fokuserar på olika varianter av PFAS vilket indikerar en potential för att de tillsammans skulle kunna komma åt en större variation av PFAS kemikalier om de skulle användas tillsammans och borde undersökas närmare. Fallstudien av deponin utanför Höganäs visade att *Miscanthus* och *Salix* kan vara användbara för att minska mängden lakvatten inom deponin, vilket skulle leda till ett minskat utsläpp av lakvatten från deponin. Vidare visade beräkningarna av PFAS upptaget att *Salix* skulle kunna användas för att ta upp PFAS, som sedan kan tas bort från deponin.

Nyckelord: Naturgeografi, ekosystemanalys, PFAS, Per- och polyfluoralkyl substanser, sanering, fytosanering, biokol

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PFAS Abbreviations

4:2 Fluorotelomer sulfonate	4:2 FTS
6:2 Fluorotelomer sulfonate	6: 2 FTS
7H-Dodecanfluoroheptane acid	HPFHpA
8:2 Fluorotelomer sulfonate	8:2 FTS
N-ethyl perfluorooctane sulfonamide	EtFOSA
N-Ethylheptadecafluorooctane sulfonamidoethanol	EtFOSE
N-ethyl-perfluorooctanesulfonamido acetic acid	EtFOSAA
N-Methylheptadecafluorooctane sulfonamide	MeFOSA
N-Methylheptadecafluorooctane sulfonamidoethanol	MeFOSE
N-methyl-perfluorooctanesulfonamido acetic acid	MeFOSAA
Per- and polyfluoroalkyl substances	PFAS
N-metylperfluoroktansulfonamid-etanol	PFNS
Perfluorinated dodecane acid	PFDoA
Perfluorinatedoctane sulfonamide	PFOSA
Perfluoro(3,7-dimethyloctanoic acid)	P37DMOA
Perfluorobutanesulfonic acid	PFBS
Perfluorobutanoicacid	PFBA
perfluoroalkyl carboxylic acids	PFCA
Perfluorodecane sulfonic acid	PFDS
Perfluorodecanoic acid	PFDA
Per-fluoroalkane sulfonates	PFSA
Perfluorododecane sulfonic acid	PFDoS
Perfluoroheptane sulfonate	PFHpS
Perfluoroheptanoic acid	PFHpA
Perfluorohexadecanoic acid	PFHxDA
Perfluorohexanesulfonic acid	PFHxS
Perfluorohexanoic acid	PFHxA
Perfluorononanoic acid	PFNA
Perfluorooctanesulfonic acid	PFOS
Perfluorooctanesulfonamidoacetic acid	PFOA
Perfluorooctanoic acid	PFPeS
Perfluoropentane sulfonate	PFPeA
Perfluoropentanoic acid	PFTeDA
Perfluorotetradecanoic acid	PFTrDA
Perfluorotridecanoic acid	PFUdA

Non PFAS abbreviation

Activated Carbon	AC
Aqueous Film-Forming Foam	AFFF
Bioconcentration factor	BCF
Cation Exchange Capacity	CEC
European Food Safety Authority	EFSA
Evapotranspiration	ET
Nordvästra Skånes Renhållnings AB	NSR
Organic Carbon	OC
Organic Matter	OM
Organisation for Economic Co-operation and Development	OECD
Kungliga Tekniska Högskolan	KTH
Persistent Organic Pollutant	POP
Soil Organic Matter	SOM
Sveriges Lantbruks Universitet	SLU
Swedish Geotechnical Institute	SGI
Swedish Meteorological and Hydrological Institution	SMHI
Total Organic Carbon	TOC
Örebro Universitet	ORU

1 Introduction

Per- and fluoroalkyl substances, known as PFAS, is a group of chemicals that has been produced since the 1950s. The chemicals are used in a large variety of consumer products such as cooking ware, outdoor gear, and clothes due to them being both water and grease resistant. They have also been closely tied to airports, firefighter training areas, and military areas due to PFAS being common components in aqueous film-forming foams (AFFF), (Banzhaf et al 2017). In recent years, the subject and research regarding PFAS have increased majorly as these substances have been proved to cause major health issues in both humans and animals (Pelch et al 2019; Sunderland et al 2018) and in multiple instances PFAS have been found in both water supplies and food sources in various parts of the ecosystem globally (Domingo & Nadal, 2019) though the risks of long-term exposures to PFAS are still poorly understood and needs to be further studied (Sinclair et al 2020).

The issue of PFAS has risen with findings of PFOS and PFOA (Banzhaf et al 2017), two common substances, in both private and public drinking water sources and it is estimated that roughly 2 million Swedish citizens are affected (Hellström et al 2020). Some data suggests that over 3.6 million inhabitants can be affected (Banzhaf et al 2017). This issue has recently led to the inhabitants in a small Swedish community winning a lawsuit against their municipality for providing PFAS-contaminated drinking water (Sjögren et al 2021). An investigation of the presence of PFAS in Swedish waters found PFAS in 90 % of the samples, indicating that PFAS are present throughout the entirety of Sweden and in different types of waters (Gobelius et al 2018), as well as in waters all over the globe (Giesy 2001; Wild et al 2015).

A landfill can be seen as the last step in the product life cycle chain, and a place where PFAS containing products finally end up (Wei et al 2019). This causes problems as PFAS can spread from the landfills to the surrounding areas affecting both the soil and water profile and increase the contamination concentrations. To prevent this from happening is it important to develop treatments that can lower the risk of PFAS spreading from the landfills (Wei et al 2019). Leachate from landfills usually contains a large variety of pollutants, including multiple types of PFAS which makes the objective of removing PFAS from leachate complex as there is a lack of remediation techniques capable of targeting various groups of PFAS (Modin et al 2018).

Currently there are no national regulations for landfills to do anything about the presence of PFAS in their systems and there are also no limitations dictating how much PFAS the waste is allowed to contain (Håkansson 2021) which could cause large amounts of PFAS contaminated waste to have been disposed of in these areas (Modin et al 2018). It is therefore possible that PFAS could have been leaching from the landfills for a long time, making landfills a point source of PFAS pollution into the environment (Modin et al 2018; Gobelius, 2018). While there are no obligations for landfill owners to do something, there are still some that do see this as a problem for the environment and foresee that such regulations could be implemented in the future. Nordvästra Skånes Renhållnings AB (NSR), a landfill owner for several landfills in the north-western region of Scania in Sweden is one of them. (Sorelius 2021, pers. Comm.).

Two environmentally friendly alternatives for PFAS remediation that are of interest to NSR are biochar, a form of black carbon, and phytoremediation, using plant species ability to take up nutrients from the soils in order to remove or immobilize PFAS. The PFAS that is taken up by the plant can be removed from the site by harvesting over ground biomass and burn it at high temperatures (KemI 2015). What happens to the PFAS that is obtained by biochar is still unclear as most studies has been focusing on shorter timescales and not what the potential long-term effects of PFAS adsorption to biochar could be, which is a serious knowledge gap that should be discussed in any future studies on this topic.

Both biochar and phytoremediation has previously proved suitable for other variations contaminants (Ahmad et al 2014; Meagher, 2000) and these methods are also currently being researched in relation for PFAS remediation and whether they are suitable for this practice or not (Gobelius et al 2017; Kupryianchyk et al 2016).

1.1 Problem

Both phytoremediation and biochar are considered to be two environmentally friendly biological remediation techniques that has been proven useful for other types of pollutants (Shahsavari 2020) and the combination has had some success in terms of combating heavy metal pollution (Paz-Ferreiro 2014). However, they have not been tested together to see if the combination would be useful in terms of PFAS remediation. Therefore, it is of great interest to see if there are any differences in performance when they are being used by themselves and when they are combined with each other. Especially when considering the fact that biochar is normally used as a soil amendment in order to increase soil health and plant growth (Tang et al 2013), which has the potential to increase effects of phytoremediation (Paz-Ferreiro 2014) making both techniques more efficient than if they were to be applied on their own.

Nordvästra Skånes Renhållning AB (NSR) has detected various concentrations of PFAS in the leachate within their landfills. At one of their landfills, Tjörröd, which is situated outside of Höganäs, they have recently planted *Salix* outside of the landfill and plans for a *Miscanthus* plantation on top of the landfill in order to decrease the amount of leachate that would otherwise be needed to be released to a recipient. The reason for planting these two species is that *Salix* can take up large amounts of water (Sharma et al 2020) and are therefore effective in terms of reducing leachate volumes, while *Miscanthus* has shallow root systems and can therefore be planted on top of a landfill that has undergone final covering without the risk of penetrating the protective layers underneath (Sorelius 2020. Pers. Comm). Due to the species relevance to the landfill would it also be interesting to research these two species specifically.

There are a few reasons why the study site was specifically chosen to be Tjörröd. First, Tjörröd is a small landfill with less resources. This means that an application of a cost-effective and self-sufficient cleaning technique is preferable in this area as they require less maintenance than other alternatives (Pilon-Smits, 2005), making phytoremediation and biochar incredibly relevant to this area. Furthermore, the area surrounding Tjörröd landfill is rather sensitive with the Sound (Öresund), together with two nature reserves, and a Natura 2000 area in close proximity to the landfill (Sweco 2020b) making it important to implement techniques to reduce the PFAS concentrations to lower the risk of potential leaching to these areas.

1.2 Aim

The thesis includes a background section regarding PFAS to further the knowledge regarding this topic and why it is of importance. The background section also serves as an introduction to biochar and phytoremediation, and how they could potentially be used for PFAS remediation.

The aim of the thesis was to evaluate the suitability of using the species Salix and Miscanthus for phytoremediation in practice and to investigate how much PFAS these species potentially can remove from a landfill located in the southern part of Sweden. One of NSRs landfills was chosen as a case study: the Tjörröd landfill, located in Höganäs, Scania. By applying a simplified hydrological model and water balance for this study area the following questions were addressed:

1. *Which PFAS have been found at the site, what are their concentrations? Do this differ from the other landfills that are owned by NSR?*
2. *What are the potential evapotranspiration rates for Salix and Miscanthus? How much water could they possibly remove?*
3. *What will be the water balance of the landfill in order to specify the feasibility of water treatment?*
4. *Based on data from previous research, how much PFAS could potentially be removed by Salix?*

2 Background

2.1 What is PFAS?

PFAS is a group of chemicals that became popular in the 1950 due to their water and grease repellent chemical structure. They can be found in a variety of products, such as non-stick pans, clothes, outdoor tools, and many others. As mentioned in the introduction, they are also a main component in AFFF which has become one of the major sources of PFAS in Sweden as AFFF is commonly used in airports and in firefighter training areas (Pelch et al 2019; Sunderland et al 2019; Wang et al 2017). Historically the focus has been on long-chain variations of PFAS, mainly PFOA and PFOS having been put under scrutiny by scientists worldwide (Wang et al 2017). The focus on PFOA, PFOS, and other long-chain variations of PFAS has caused a shift in manufacturing and on the market to short-chain variations, such as PFBS which leads to increasing domination of short-chain PFAS substances in nature (Knutsen et al 2019; Zhang et al 2019). While the usage of PFOS and PFOA have decreased in the last years, there are still concerns that their concentrations in nature will continue to increase due to the desorption of these compounds from the soil and by transformation of their precursor (Bolan et al 2021).

2.2 Definition

PFAS are aliphatic substances including at least one carbon atom that are bound to flour atoms. The defining component of PFAS can therefore be written as C_nF_{2n+1}- (Buck et al 2011). A slightly revised and updated definition to Buck et al (2011) has been introduced in 2018 by the Organisation for Economic Co-operation and Development (OECD) where they have defined the component as -C_nF_{2n}- to include PFAS variations where both ends of the moiety are connected to a headgroup (OECD 2018).

Three main types of PFAS have been identified, fluoropolymers, perfluoroalkyl substances and polyfluoroalkyl substances. Perfluoroalkyl substances consist of both long and short carbon chains with charged functional heads, usually sulfonic or carboxylic. These types are fully fluorinated where all bonds are C-F bonds. Polyfluoroalkyl, on the other hand, are not fully fluorinated. They consist of at least one other type of bind, usually either oxygen or hydrogen. They also contain carbon-hydrogen bonds which are susceptible for degradation (Buck et al 2011; Shahsavari et al 2020). The third type, fluoropolymers, is defined as "carbon-only polymer backbone with fluorine directly attached" (Buck et al 2011).

PFAS can also be defined by the length of their carbon chains, so called tails. Long-chained variations of PFCA have seven or more per-fluorinated carbons while PFSA are required to have six or more fully per-fluorinated carbons to be considered long-chained. The reason for the difference in definition is due to the increased bioaccumulation properties of PFSA, therefore requiring less C atoms to have the same effects as the PFCAs. While there are no other PFAS variations with a clear definition of what can be classified as long-chained variations is it still possible to assume that any perfluoroalkyl substance with more than 7 carbon atoms can be considered to be long-chained (Buck et al 2011).

2.3 Chemical properties

PFAS consists of carbon-fluorine bonds which are considered to be chemically stable, meaning that these are extremely difficult to break. This causes PFAS to not completely break-down in nature (Shashavari 2020). The PFAS-group is a heterogenic group consisting of more than 3 000 different chemicals with different structures and different characteristics (Modin et al 2018: Ross et al 2018), though they do share some characteristics with each other due to the defining carbon fluorine bond. The different properties of the substances are caused by the molecular structure of the substance and can be caused by structural variations such as different functional headgroups, chain length, branching in the chain length, or mirrored chains (Gleisner et al 2019). Due to their chemical structure, PFAS are both hydrophilic (functional headgroup) and hydrophobic (carbon chain), (Krafft & Riess 2015).

Short chained substances have a higher water solubility, causing them to be more easily leached than other substances with longer chains (Gleisner et al 2019) and are therefore more prone to mobility and movement in water systems (Knutsen et al 2019). PFAS with longer chains are found more in soil and bound on sediments whereas short-chained substances have been detected in plants to a higher degree. The strong C-F bonds found in PFAS and their tendency to withstand degradation causes bioaccumulation in humans, plants, and animals to be commonplace. This accumulation is reported to cause various health concerns in nature and humans (Sima & Jaffe 2021). The bioaccumulation rate can vary between different substances, PFBS is considered to be less prone to bioaccumulation than other PFAS variations as short-chain PFAS generally are less prone to accumulation than those variations with longer chains (Knutsen et al 2019). They are also reported to be less toxic than substances with long chains (Gellrich et al 2012). Though there are some that claim that short-chained variations are as equally dangerous as their long-chained counterparts and that regulations should be implemented as soon as possible (Brendel et al 2018). The toxicity of the compounds is worsened by bioaccumulation and their ability to withstand degradation (Gleisner et al 2019; Pelch et al 2019) due to their chemical structures. The exception for this is that PFAS precursors can degrade to their PFAS variation, and some substances can break down into other PFAS chemicals (Ahrens & Bundschuh 2014).

Two of the most studied PFAS variations, PFOA and PFOS, and their precursors have been defined as Persistent Organic Pollutants according to the 2009 Stockholm convention regarding POPs (Genualdi et al 2010; Wild et al 2015) and it is expected that PFHxS and its salts will be included in the definition in 2022 (KemI 2021). POPs are generally defined as anthropogenic chemicals that are existing in nature all over the world with shared properties of toxicity, bioaccumulation, persistence, and the possibility for long-distance transportation (Wild et al 2015). The main difference between most PFAS substances and POPs is that PFAS has higher surface-active properties caused by their hydrophilic and hydrophobic structure while most POPs are either hydrophilic or hydrophobic in nature making their behaviour more predictable. PFAS, on the other hand, are not only regulated by their hydrophobic/hydrophilic properties but also by electrostatic interactions. This causes issues in understanding sorption and other abilities from looking at only specific variables, such as OC content in a soil and making predictions from that (Pereira et al 2018). Overall, PFAS are soluble in water to a higher degree than other types of persistent chemicals. With this they are also more prone for leaching, especially in soils where the sorption capacity is low, example being sandy soils (Bolan et al 2021).

2.4 Environmental concerns

In nature, PFOS and other PFAS variations have been detected all over the globe including remote areas such as the high arctic and in Antarctica (Giesy & Kannan 2001; Wild et al 2015). The release of PFAS into the environment happens throughout the entire life cycle, from production to disposal either by direct emissions of these substances during their lifecycle or indirectly by transformation of precursors (Ahrens & Bundschuh, 2014). Globally the highest concentrations of PFOS can be found in more populated areas with more industries (Giesy & Kannan 2001). PFOS has historically been the most common substance in the environment and the concentrations increase substantially further up in the food due to their affinity for bioaccumulation. PFOA does not show the same bioaccumulation effects and the concentrations do not seem to differ depending on location in the food chain (Ahrens & Bundschuh, 2014).

In the terrestrial environment there is some data that shows accumulation of PFAS in agricultural crops such as potatoes, wheat, corn, and oats following irrigation of PFAS containing water (McCarthy et a 2017). Another concern within agriculture is for PFAS to enter pastures and fields via application of biosolids and sludge from wastewater contaminated with PFAS which will then be taken up by either grazing animals or plants (Death et al 2021). PFAS has been known to decrease soil quality and impact soil enzyme activity, microbial activity, and cellular structures negatively. It has also been reported that soils containing PFAS have lower bacterial diversity than untainted soils (Abunada et al 2020).

In studies made on animals they found that most of the PFAS accumulate in the offal areas (Zafeiraki et al 2016) with lower concentrations being reported in muscle tissues, which is the preferred food intake for most humans (Death et al 2021). PFAS has shown to affect liver causing liver toxicity, mammary development and immune systems (Lilienthal et al 2017), and shown to cause problems with the thyroid gland and blood fats (Linderoth et al 2016) as well as affect reproduction. PFOS has also been connected to various types of cancer (Gleisner et al 2019). These results are mainly obtained from lab tests and whether these effects are relevant for wild game and livestock is still unclear. Livestock has been reported to be able to handle high amounts of PFAS and show little to no health effects during shorter research periods. However, in the US a farmer experienced cattle death following that the cattle had been drinking water from a polluted water creek. Later it turned out that the water in the creek contained PFOS released from a nearby landfill which was ruled to be the reason for the deaths (Kelly 2016).

It has also been noted that different PFAS substances behave differently depending on animal species. One such example is bioaccumulation behaviour. In cattle, longer PFAS variations, such as PFOS showed a higher bioaccumulation than short-chain PFAS. In chicken eggs the opposite was observed with short-chained PFAS variations showing more bioaccumulation than PFOS (Death et al 2021). Some of the earliest testing of PFAS was done in the 1980s on monkeys where they found PFOA to affect the immune systems. Tests on monkeys were also done for PFOS, but that specific study had to be cancelled due to high mortality rates for the monkeys at all doses. These results were shared with the public in 2000 by the U.S. Environmental Protection Agency (Grandjean 2018).

Within the aquatic systems PFAS is continuously released by both point sources such as manufacturing and nonpoint sources such as atmospheric pollution (Ahrens & Bundschuh, 2014). Studies have also shown that fish and shellfish are one of the main contributors for increased PFAS intake, especially PFOS which has been found in a variety of fish and shellfish. There are also links between increased PFAS intake and a high consumption of these food sources (Christensen et al 2017). In Sweden where they tested the PFOS concentration in fish from lake Vättern and from the Baltic Sea they found that fish from Lake Vättern had higher

PFOS concentrations than the fish caught in the Baltic Sea. This is mainly due to the higher amount of anthropogenic discharge around Lake Vättern, causing the increased levels. The same study did also note that the catchment area also plays a large role when discussing PFAS levels within the fish as they can differ within the same body of water (Berger et al 2009).

2.5 Human concerns

The first observed case for organic fluorine compounds in humans was in 1968 (Taves, 1968). Soon thereafter PFAS was observed in industrial workers and in 1981 PFAS was found in the umbilical cord during a childbirth with the mother being a factory worker. In 1993 PFAS was found to be transported in goat milk, the same was found in humans a while later. By the year 2000 the spreading in nature and persistence of PFAS started to be discovered but it still took some time for the scientific world to research the topic (Grandjean, 2018). There are multiple ways for humans to be exposed to PFAS. Most commonly is the exposure via drinking water, but the chemicals can also be found in various foods, consumer products, and even via dust particles (Wei et al 2019). Concerning dietary sources, the most common source has shown to be through fish and shellfish, though some research suggests that beef is also a cause of major intake of PFAS (Death et al 2021). The health effects of PFAS on humans can differ depending on age, ethnicity, genetic predisposition, gender, and health status (Fenton et al 2021).

Multiple studies have shown several areas of concern regarding intake of PFAS. Clear connections have been found between PFAS and immunology where PFAS has affected vaccines, immunological responses and affected diseases linked to immunosuppression (Fenton et al 2021). Studies in Denmark and Norway have shown that PFAS can alter the ability to fight off infections such as colds with increased symptoms and fever in children (Grandjean 2018). The risk of PFAS affecting vaccines is currently the main critical effect of PFAS, which changed in 2018 when the effects on cholesterol was deemed to be the most concerning issue (EFSA 2020). In children some studies suggest that PFAS can be linked to the neurological development in kids and affects areas such as attention, behaviour, development, and learning. Though this is debated as multiple studies also conclude to not find such a link (Rappazzo et al 2017). For adults, the research is scarcer but there might be suggestions that PFAS can increase neurophysiological functions with improvements in memory and learning (Shrestha et al 2017).

Other areas of concern include connections to thyroid disease, elevated thyroid-stimulating hormones which can cause harmful outcomes during pregnancy and childbirth. Most of the long-chained PFAS are accumulated in the liver in humans which causes toxicity issues for multiple functions connected to the liver. They have also found links between PFAS and various cancer types such as liver cancer and kidney cancer, as well as being linked to reproductive issues in both men and women (Fenton et al 2021).

As previously stated, most of the research is only focused on a few types of PFAS, mainly PFOS and PFOA, and a few others. This is cause for concerns as there are plenty of substances with little to no research regarding their effects on either health or environment (Gleisner et al 2019) even though all known variations of PFAS are deemed to be toxic (Dean et al 2020). Some PFAS, such as PFOA and PFOS have shown to accumulate within humans with half-lives of three to six years, depending on the PFAS variation and gender. The half-life of some PFAS variations depends on the carbon chain length, with shorter PFAS, such as PFBS having a half-life of only one to two months (Lee & Choi, 2017). According to EFSA the most exposed groups are children and toddlers with exposures to PFAS via pregnancy and breastfeeding being one of the leading causes for PFAS in new-borns (EFSA 2020).

2.6 Regulation

Currently there are few regulations and restrictions regarding PFAS. The threshold values for what is deemed as a safe amount varies between the different countries and while most have introduced regulations regarding PFOS and PFOA are regulations regarding other variations still lacking (Gobelius et al 2018). It is only a few countries, including Sweden and Italy who have introduced regulations for other PFAS variations as well. In the US it is currently up to the different states to put regulations regarding PFAS while in Canada, just as Sweden and Italy they have also introduced regulations for more PFAS variations than only for PFOS and PFOA (Pancras 2020). These variations between countries indicate uncertainties regarding their health effects and what amount can be considered safe for ingesting (Gobelius et al 2018).

In 2020 EFSA put new regulations on four of the most common PFAS variations, PFOS, PFOA, PFHxS, and PFNA. The threshold was put to a tolerable weekly intake of 4.4 ng/kg bodyweight (EFSA 2020). Regarding drinking water, the EU introduced new regulations regarding PFAS where they accepted a safe consumption limit to 100 ng/l for 20 specific PFAS variations and a limit of 500 ng/l for the sum of all PFAS variations (EU 2020). In February 2021, the United States Environmental Protection Agency issued a statement regarding PFAS in drinking water where they promised to address the issue of PFAS by collecting data on the issue to finally suggest regulations for PFOA and PFOS under the Safe Drinking Water Act (EPA 2021).

3 Remediation of PFAS

Most of the remediation techniques of PFAS concern either immobilization, mobilization, or degradation with most of the interactions being based on different sorption interactions (Bolan et al 2021).

Looking at remediation of PFAS it is established that various PFAS substances behave differently depending on length of chain. The sorption increases with increased length of the fluorocarbon chain length (Sørmo et al 2020). The soils can also affect the behaviours of PFAS, variables such as pH and the amount of organic matter present in the soil supposedly alters the PFAS dynamics, as well as environmental factors, e.g., precipitation or drought also plays into the dynamics (Bolan et al 2021). Although soil organic matter (SOM) har been identified as an important sorbent of PFAS, mineral surfaces can play an important role as well. For example, ferrihydrite has been shown to be an important sorbent for PFAS with six or more perfluorinated carbons at $pH \leq 5$, especially when the content of organic carbon is low (Pereira et al 2020)

Some studies suggest that short tailed PFAS are more bound to humic and fulvic acids as the net bulk charge of the SOM have a significant effect on these PFAS variations. Long tailed PFAS seemed to be more affected by the pH levels of the SOM as they preferred binding to the humic fraction of SOM, suggesting that cations were not as relevant for this group as it is for the former (Pereira et al 2018). The chain length will also dictate which treatments are suitable for usage and how to apply them. Short-chained PFASs with a high mobility have been detected in plants to a higher degree than long-chained variations. The implication of this is that short-chained variations are more easily taken up by plants than longer variations (Jiao et al 2020). Long-chained variations, including PFOS and PFOA are recorded to react more to immobilisation onto various clay- and carbon-based materials such as activated carbon (AC) and biochar (Bolan et al 2021).

In a situation where the chemical cocktail can be considered complex, including not only variations of both long and short PFAS variations, but also a myriad of other chemical substances as well, the treatment will be more complex. This will include using multiple remediation techniques targeting different groups (Gleisner et al 2019). A suggestion is to

combine biochar and phytoremediation together. The biochar would focus on binding long chained variations of PFAS, and the phytoremediation would be used to adsorb the short-chained PFAS by accumulation within the plants. Adding biochar could also help improve the plant growth by improving soil properties such as increasing the microbial activity and increasing the nutrient availability (Figure 1) (Shahsavari et al 2021).

There are only a few studies which have researched the combination of biochar and phytoremediation which was in relation to heavy metal pollution, not PFAS. They found some evidence that the combination can increase an uptake, but it is still unclear as there is so little research on the subject. There are suggestions made that the combination might be best suited for areas with multiple contaminations where they have different targets (Paz-Ferreiro et al 2014).

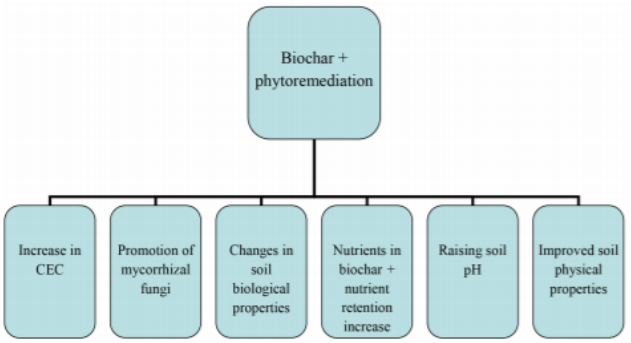


Figure 1 Potential effects for using a combination of biochar and phytoremediation on heavy metal polluted soils. Two notable effects include increased soil pH and improved properties of the soil (Paz-Ferreiro et al 2014)

3.1 Biochar

Biochar is defined as a variation of charcoal with the intention of being used as an amendment to improve soil health, as well as improving the nutrient levels, carbon storage, and general productivity of the soil (Ahmad et al 2014). The procedure of obtaining biochar is similar to the procedure for creating charcoal – feedstock is burned in low or non-oxygen conditions at certain temperatures over a certain amount of time (Verheijen et al 2010). This process is called pyrolysis and depending on type of feedstock and burning temperature the result can vary greatly in chemical and physical properties (Ahmad et al 2014) with the end product being a porous and carbon-rich material (Ni et al 2020). The specific surface area is increased with increasing pyrolysis temperature, which increases the level of carbonisation of the feedstock. Increased temperatures also decrease the amount of amorphous organic matter which increases the absorbing capabilities of organic contaminants (Beesley et al 2011). Generally, increasing temperatures increases the carbon content, pH, aromaticity, surface area, stability, ash content, and pore size of the biochar. On the other hand, the biochar yield, oxygen content, and hydrogen content decreases at the same time (Hassan et al 2020). Looking at feedstock, biochar derived from wood generally has a higher specific surface area than biochar derived from grass. Feedstock also affects properties such as total organic carbon and mineral content of the biochar to a higher degree (Zhao et al 2013).

Understanding exactly how feedstock and temperature affects the properties of biochar can enhance the possibility of developing designed biochar (Zhao et al 2013) that can be used for specific purposes (Novak et al 2009). In terms of remediation, biochar and other types of black carbon are recalcitrant with the ability to influence hydrophobic organic pollutants - affecting their mobility and bioavailability in soils (Ogbonnaya & Semple 2013). Wood-derived biochar are more suitable for remediation of organic contaminants due to increased hydrophobic interactions whereas grass and manure biochar works best for ionic contaminants as they have

an increased variety of functional groups (Hassan et al 2020). Wood-derived biochar formed by high pyrolysis temperature can be suitable for targeting organic contaminants as the aromatic structures found in biochar are beneficial in hydrophobic interactions with organic compounds. The high temperature also causes the pores to increase in size which heightens the sorption capabilities of the biochar. Furthermore, if the organic compounds are charged, they can be adsorbed by electrostatic interactions (Hassan et al 2020; Omo-Okoro et al 2018). The added benefits of increased carbon sequestration, reduced greenhouse gas emissions, and improved general soil quality are all arguments of biochar potentially being an environmentally friendly alternative for remediation of POPs (Ni et al 2020).

According to a report regarding on-site wastewater treatment where they tested how biochar would affect the presence of various PFAS groups they reached the conclusion that biochar could immobilise PFAS via adsorption. The results showed that long-chained PFAS such as PFOA and PFOS were more easily absorbed due to the long carbon chains. Short-chained variations, on the other hand, experienced a much lower removal efficiency and has previously been found to be replaced by long-chained PFAS variations in anion exchanges (Dalahmeh 2019). Feedstock and pyrolysis temperature affects the physiochemical properties of the biochar and influences the immobilization of PFAS. The biochar needs to have a high specific surface area which is achieved by high pyrolysis temperatures (Sörengård et al 2020) preferably in combination with wood being used as feedstock (Hassan et al 2020). This type of biochar also has a high carbon content which furthers the sorption capabilities of various contaminants (Tang et al 2013). The high specific surface area in combination with the large amount of various functional groups associated with biochar is theorized to have the ability to absorb organic pollutants together with the native OM in the soil (Guo et al 2017; Ni et al 2020).

Biochar has been found to be effective at removing/immobilizing PFAS with more than 5 carbon atom chains in wastewater (Dalahmeh 2019). In aqueous environments biochar is a viable option for removing PFAS compounds, with a higher effectiveness for removal of long-chained variations such as PFOS and PFOA in contrast to short-chained variations. Between PFOS and PFOA the sorption of PFOS was found to be greater than the sorption of PFOA. Sorption of PFAS onto biochar also has a linear relationship to pH where the sorption increases with decreasing pH (Zhang et al 2019). Looking at biochar remediation of PFAS in soil the effects has not been found to be as effective although there are still differences between different types of soils (Sörengård et al 2020). In soils with a lower amount of total organic carbon (TOC) is the effect of biochar greater with a high effectivity in reducing PFAS in leachate. In soils with a high presence of TOC the biochar did not have the same effect. This is probably due to the high amount of OM in the TOC rich soils which can cause clogging of biochar pores and increased sorption capability of the soil itself. High TOC soil has been reported to be roughly around 100 times stronger than low TOC soil in terms of PFAS adsorption. This means that the high TOC soil would require stronger sorbents to achieve the same results as the low TOC soils (Sørmo et al 2020).

In a project by Askeland et al (2020) the effects of pine derived biochar which had been pyrolyzed at 750 ˚C was tested on two different types of soils: sandy clay loam and loamy sand with differences in OM and TOC. The soils were treated with a PFAS solution containing PFOA, PFOS, PFHxS, and PFHxA and they tested the substances both individually and in combination with each other. The sorption of PFAS was found to be the fastest in the beginning, with most PFAS being adsorbed in the first hour, followed by decreasing sorption-rates, which is consistent with Li et al (2019). They also found that the biochar seems to be less effective in high carbon content soils due to increased competition for sorption sites, though they do not conclude that intra-PFAS competition is to be a big consideration. Apart from studying sorption, they also studied the desorption of PFAS from the biochar. Overall, it seems that

biochar had the most effect on the sandy clay loam with increasing adsorption and at the same time a higher desorption rate than in loamy sand. Though biochar did lower the desorbed fractions for all PFAS for both soils (Askeland et al 2020).

Looking at the effects of pyrolysis temperature on PFAS remediation a few studies has been conducted One study looked at biochar derived from biosolids that was pyrolyzed at 500 – 600℃. The results from that study were that they found the biochar to be able to adsorb >80 % of long-chained PFAS, with a possibility of 90 % adsorption if the biochar was created using a pyrolysis-combustion integrated process (a technique combining combustion and pyrolysis to, in this case create biochar) while it did not perform as favourable with short-chain PFAS, where only 19-27 % of the PFAS were removed. The biochar used in this study was also found to be highly stable with an expectancy to persist over millennia in soil (Kundu et al 2021). In another study Hassan et al (2020) tested adsorption of PFOS on corn straw derived biochar at various pyrolysis temperatures between 250 and 700℃. They found that higher temperatures increased the adsorption capacity, with the highest being achieved at 700 ˚C. This behaviour was caused by electrostatic interactions and the increased hydrophobicity of the biochar. It was also noted that the pH was an important factor for the adsorption of PFOS (Guo et al 2017) which is also suggested by Pereira et al (2018), where decreasing pH caused an increase in adsorption capacity. Though some studies suggest that organic compounds are not affected by pH at all (Hassan et al 2020).

3.2 Phytoremediation

Phytoremediation is defined as “the use of green plants to remove pollutants from the environment or to render them harmless” (Cunningham & Berti, 1993; Salt et al 1998). The key for phytoremediation is naturally occurring processes where plants and their associated microbes degrade and sequester organic and inorganic materials. The term itself is an umbrella term for various mechanisms that can remove pollutants, including phytofiltration, rhizofiltration, phytoextraction, phytoimmobilization, and phytodegradation (Arthur et al 2005; Pilon-Smits, 2005) and can therefore be used in multiple ways (Figure 2) (Pilon-Smits, 2005). Most organic pollutants can be remediated through various processes – adsorption, translocation, transportation, hyperaccumulation, transformation, and mineralization. For toxic pollutants, the main use of phytoremediation is the hyperaccumulation where the pollutants can be harvested above ground (Meagher, 2000).

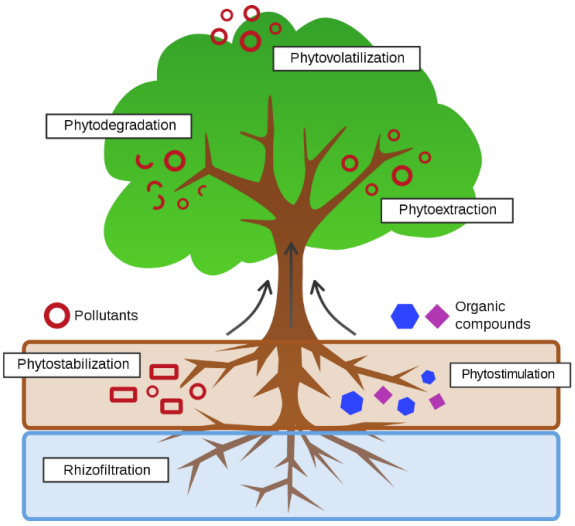


Figure 2 Different pathways and mechanism regarding uptake of various pollutants that are included in the term phytoremediation (Townie 2016) CC BY-SA 4.0

This type of environmental clean-up has increased in popularity during the last few decades as it is considered to be an extremely cost effective, green alternative for in-situ application that is driven by the sun (Ali et al 2013). Using vegetation for removing pollution also adds the benefits of reduced unnecessary human or animal contact with the affected site (Arthur et al 2005). Though there are many advantages of using phytoremediation, its drawbacks cannot be denied. As phytoremediation is an in-situ technique where the plants grow where the pollution occurs, the properties of the soil, toxicity level, and local climate affect the potential growth of the plant. Lastly, phytoremediation is also a slower alternative to other methods where it can take years to remove the pollution from the soil, it is also affected by the bioavailability of the pollutant, if the bioavailability is low then phytoremediation is not an applicable method as it only affects bioavailable pollutants (Pilon-Smits, 2005).

Regarding PFAS there are less reports specifically focusing on phytoremediation, but more focus has been on reported findings of PFAS in plants and vegetation (Huff et al 2020). There have been significant findings of PFAS accumulation in several types of agricultural crops, including cereals, fruits, and vegetables. The accumulation is dependent on plant species and on the substance (Ghisi et al 2019). There are some recent findings that suggest that there are not only differences between species regarding the uptake and toxicological effects of PFAS but that variations are also occurring within the same species (Costello & Lee, 2020). Within a plant PFAS could end up in different parts. In the roots they have mainly found long-chained variations of PFAS and in the foliage mainly short-chained variations accumulated (Jiao et al 2020).

Parameters such as protein content, composition, root surface areas, and transpiration potential, together with various climate factors – humidity, irradiance, and temperature amongst others are said to affect the uptake of PFAS (Ghisi et al 2019). Generally, it is the physiochemical properties of a plant that affect the uptake possibilities (Jiao et al 2020). According to the current research, the key metric to assess plants available for phytoremediation is the bioconcentration factor (BCF) (Bolan et al 2021; Huff et al 2020). BCF is defined as the contaminant concentration in plant tissue in relation to the contaminant concentration in the surrounding soil, water, or air (Sima & Jaffe, 2021).

There are multiple plants that have been identified for possible PFAS remediation, *Juncus effusus*, a common wetland plant naturally distributed in temperate and sub-tropical areas throughout Eurasia and North America has previously been known for remediation of metals

and various organic pollutants. The plant was tested for selected variations of PFAS, and the result showed that shorter PFAS variations could translocate most PFAS further up the plant and accumulate them in the vegetative parts of the plants. The longer variations showed a higher accumulation in the roots and a limited upward movement within the plant (Zhang et al 2019). Other species that have been noted to take up PFAS are *Betula pendula* and *Picea abies* in a study looking at PFAS uptake outside of the Arlanda airport in Stockholm (Gobelius et al 2017). The effects that the carbon chain-length has on the PFAS uptake in plants as noted by Zhang et al (2019) is supported elsewhere. In another experiment by Huff et al (2020) they researched the potential for PFAS remediation in different plants and found that the highest above-ground uptake was for PFPeA, a five-carbon chain PFAS and the lowest for PFOS, with an eight-chained carbon tail. It was also noted that herbaceous plants obtained high concentrations of tested PFAS compounds and in woody plants the highest accumulation rates were found in the foliage for the tested species, with some accumulation in the wooden parts (Huff et al 2020).

One of the most studied plant species for phytoremediation is *Salix.* Their characteristics of increased biomass production, increased transpiration, rapid nutrient uptake, and a documented ability for removing both organic compounds and other toxins such as heavy metals make them an interesting option for phytoremediation (Mleczek et al 2010; Pulford & Watson, 2003; Weih & Nordh, 2002). In a study where they tested three types of *Salix* (*Eleagnos, Purpurea,* and *Triandra*) against a combination of various types of PFAS, including PFOS and PFOA they concluded that all three variations of *Salix* could take up and accumulate PFAS with *S. Purpurea* being the most effective (Sharma et al 2020). The highest accumulation of short-chained variations happened in the foliage and long-chained variations were accumulated in the roots. The study did not take any accumulation in the wooden parts into consideration (Sharma et al 2020).

4 Study site and methods

4.1 Study area

For this thesis, the specific study area was chosen to be the Tjörröd landfill outside of Höganäs (Figure 3). The landfill is owned by NSR and situated in the north-western parts of Scania. The entire area is roughly 26 ha and includes three ponds for water treatments, a recycling station, a recently cultivated *Salix* area of 2 ha, and other types of vegetation (Sweco, 2020c). To the south of the landfill there is a small creek called Margretebergsbäcken that ends in the Sound (Öresund). Along the shore, close to the landfills, two Natura2000 areas are situated, and two nature reserves close by, one roughly 1.3 km north of the landfill and the other 1 km to the south. Due to the sensitive nature surrounding the landfill, NSR decided to propose emission conditions if they see the need to release leachate and water from the landfill into the creek, for example if there is a risk for the ponds to become flooded (Sweco, 2020a).

The landfill itself started in the early 1950s and closed in 2001 with the goal of the entire landfill being completely covered up by the end of 2028 and partly covered by the end of this year. It is estimated that around 1 000 000 m^3 of waste has been deposited here throughout the years (Sweco, 2020b). At the recycling station within the landfill are they currently allowed to take care of 80 000 ton non-hazardous and 1 000 ton hazardous waste (Sweco, 2020c).

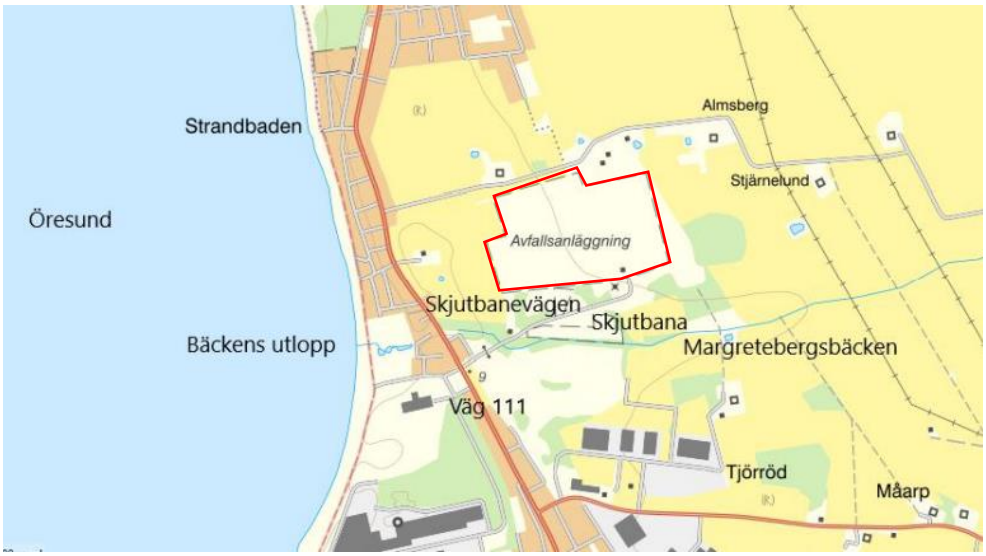


Figure 3 Map over Tjörröd landfill which is situated close to Höganäs (Sweco 2020b). The landfill is marked with red lines. Avfallsanläggning – landfill, Skjutbana – shooting range, Bäckens utlopp

Currently the landfill is undergoing final covering in two different steps (Figure 4). The first step, phase 1, is supposed to be completely finished during 2025 and the second step, phase 2 is to be finished in 2028 (Sorelius 2021. Pers. Comm).

Once phase 1 has been completely covered up there is a proposal for planting *Miscanthus* on top of the landfill. *Miscanthus* is a species that has already been cultivated on landfills in England (Sorelius 2021. Pers. Comm).

The reason for choosing *Miscanthus* is that the grass has shallow roots which, most likely, will not penetrate the protective layers on top of the landfill, breaking the coverage (Lewandowski et al 2000). The reasons for planting *Salix* and *Miscanthus* are to reduce the leachate flow in the landfill and therefore prevent that any leachate needs to be released to a recipient, as well as any positive effects on remediation that can be achieved by these plants (Sorelius 2021. Pers. comm). The plantations of *Salix* and the plans for planting *Miscanthus* on top of the landfills makes the species of interest for investigating potential phytoremediation.

According to Sweco (2020b), NSR should be able to irrigate *Salix* and *Miscanthus* for 200 days per year. For *Salix*, the irrigation rate is 5 mm per day, leading to a total of 20 000 m^3 water to enter the 2 hectares large vegetation area per year, and for *Miscanthus* the rate is 3 mm over an area of 3 ha with the total volume being 18 000 m^3 in a year. The maintenance also includes harvesting the vegetation to remove any contaminants that might have been accumulated. *Miscanthus* will be harvested every year while for *Salix* they are usually harvested after 4 - 5 years (Dimitriou & Aronsson, 2005).

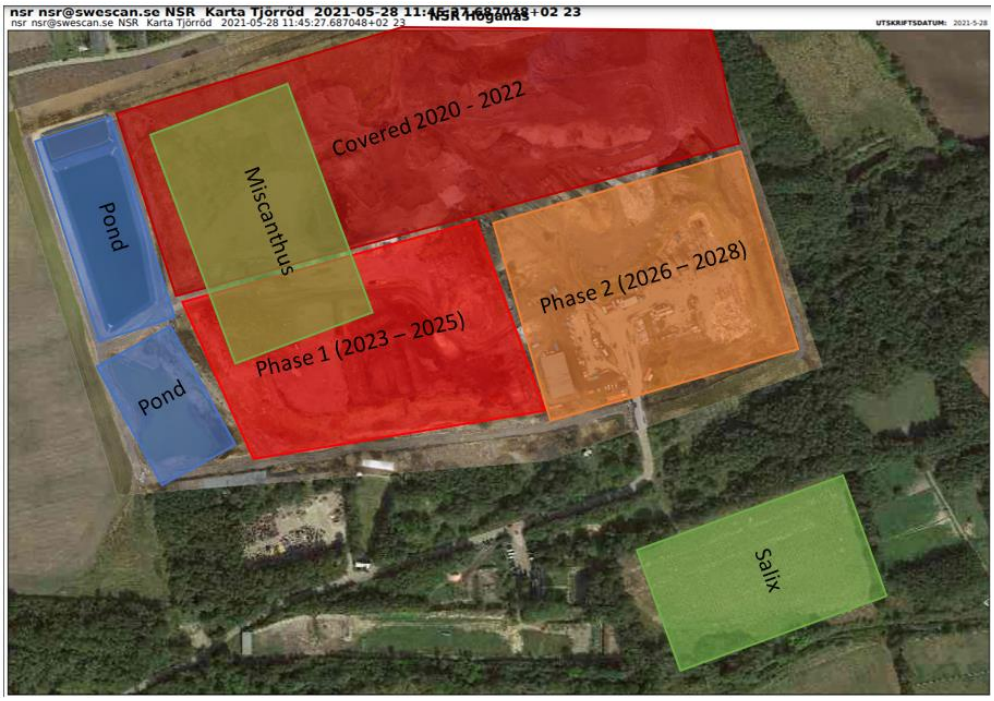


Figure 4 Image showcasing landfill and the areas of the two different phases, as well as where the ponds and the Salix vegetation are situated. Miscanthus is shown on top of phase 1 but it is still unclear as to where exactly it will be planted. The Salix is planted outside of the landfill.

The mean precipitation at the area between 2015 – 2020 (Figure 5) was 630 mm per year. During this period, there were two relatively wet years, 2015 and 2017, where 749 mm and 770 mm precipitation were received. In this period there was also one year with a significant drought where only 430 mm fell in 2018.

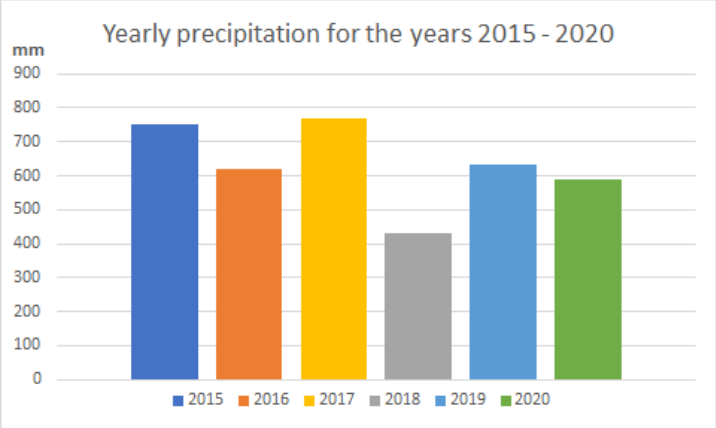


Figure 5 Annual precipitation during the years 2015 and 2020 (SMHI 2021). The precipitation roughly ranges between 400 mm in 2018 and above 750 mm during 2017. The data was obtained from the weather station in Helsingborg.

Temperature (Figure 6) shows clearly the year 2018 as an exceptionally hot year with a yearly mean temperature of 13 ˚C where the mean temperature for all the years is 10 ˚C and the coldest years being 2015 and 2016 around a yearly mean of 9 ˚C. The increased warmth in 2018 correlates with being a drier year as shown above.

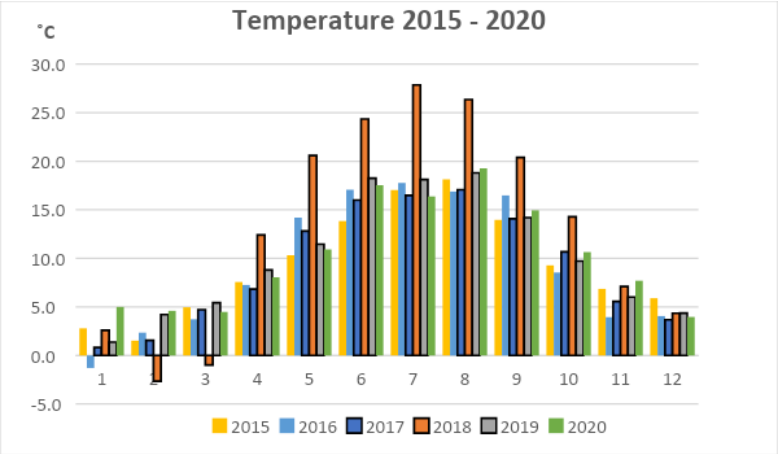


Figure 6 Average temperatures for the years 2015 to 2020. For most years, the temperatures are situated around 15 – 20 ˚C during the summer. 2018 is an exception to this as the summer temperatures reached above 25 ˚C during both July and August, as well as June being close to have average temperatures of 25 ˚C. In both September and October 2018 still experiences somewhat elevated temperatures in comparison to the other years. (SMHI 2021).

4.1.1 Hydrological flowcharts for Tjörröd landfill

To show the water balance of the landfill a flowchart was made, based on data and previous designs provided by Sweco and NSR (Figure 7). There are two main water sources within the landfill - leachate, and stormwater. Stormwater is runoff water that has been formed by precipitation, and leachate is water that has been leached from the waste disposal. Volumes for both have been provided by Sweco as calculated averages for both phase 1 and phase 2 and were based on an average annual precipitation of 710 mm that was obtained from SMHI (Sweco 2020a).

The treated leachate will be led onto *Salix* and *Miscanthus* as irrigation with the goal of the leachate to be taken care of by the plants. If there is too much water in the vegetative areas then the idea is gathering up the water and then leading it back to the treatment ponds again, thus creating a closed system for the leachate. If there is too much water in the system then the water will be released to one of two recipients, the nearby creek if the leachate considered to be clean enough, or to a sewage treatment plant if the leachate is too contaminated for release to the creek. There is some uncertainty which is connected to the stormwater, as it is still unclear if the stormwater will enter the same system as the leachate, or if it will be released directly to the creek.

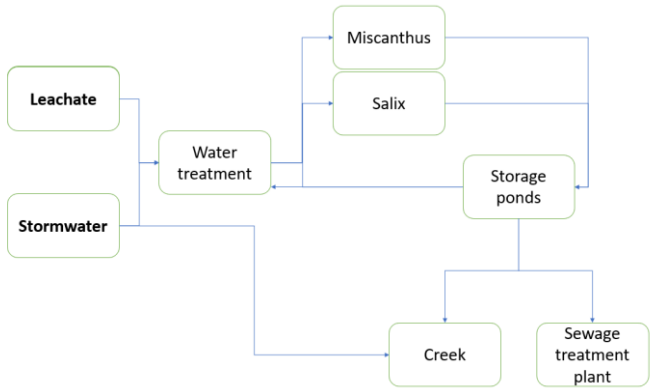


Figure 7 Flow chart for a proposed water movement within the landfill. After the leachate has been treated the water will be led onto the vegetation as irrigation, any remaining water is to be led back to the water treatment. There is an opportunity to release excess water to the nearby creek or a sewage treatment plant during periods If there is too much water within the landfill. There are still uncertainties regarding the stormwater as it will either be directly released to the nearby creek or go

into the water treatment. In this thesis it was assumed that the stormwater entered the water treatment together with the leachate.

Looking closer at the vegetation area the water flow can be modelled where the incoming water flows is by irrigation and precipitation and the outgoing flow is through evapotranspiration (Figure 8). The irrigation is the proposed irrigation rates for *Salix* and *Miscanthus* by Sweco (2020b) which are based on average water uptakes for both *Salix* and *Miscanthus*. The water uptake, which is also affected by temperature and precipitation, is assumed to remain constant for different years. This means that the irrigation rates do not take either precipitation or temperatures into consideration, even though both parameters influence the possible water uptake for both plants. This means that in years with high temperatures and little precipitation the water uptake should be higher than for wet and cold years.

Furthermore, as *Salix* is planted outside of the landfill there is a risk of incoming water leaching to lower soil layers, thus spreading PFAS from the plantation. For *Miscanthus,* this is not a risk of happening as the species is planted on top of the landfill, which leads to runoff from *Miscanthus* ending up entering the system again, with no risk for spreading PFAS outside of the landfill.

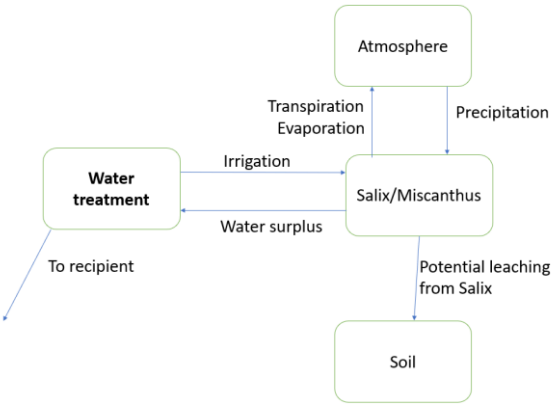


Figure 8 Water flow through the vegetation system. Incoming water are by irrigation and precipitation and water leave via evapotranspiration (transpiration and evaporation) or to the water treatment as any water surplus is returned to the landfill. Once returned to the landfill the water can be released to a recipient. Specifically for Salix there is potential for leaching as it is planted outside of the landfill with no protective layer underneath.

The main flow within the vegetation flowchart (Figure 8) is the evapotranspiration. The evapotranspiration (Figure 9) is the combined concepts of soil and interception evaporation and transpiration. The evaporation can either happen directly from a bare soil or water body, or it can also refer to water that evaporates from the surface of the vegetation, which can also be called interception. The transpiration is simply the water that is taken up by the plants and then released into the atmosphere from the leaves (Yimam, 2015).

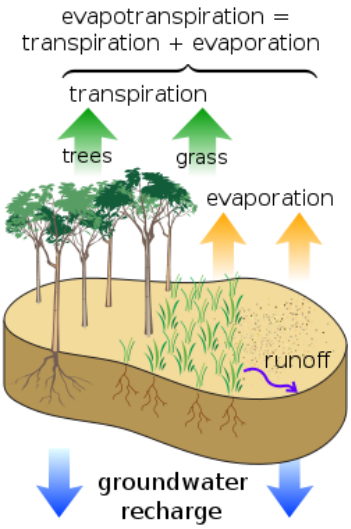


Figure 9 The evapotranspiration consists of two parts - transpiration and evaporation. The evaporation is either directly from the ground as soil evaporation, or evaporation from the surface of vegetation, also known as interception. Transpiration refers to the water taken up by the plants and then released into the atmosphere (Knuckles) CC BY 3.0.

4.2 Concentration of PFAS in Tjörröd compared to other landfills

PFAS has been found in landfill leachate all throughout Sweden (Modin et al 2018). Comparing the concentration values of PFAS between Tjörröd and three of NSRs landfills (Table 1) there is a distinct difference regarding the detection of PFAS between the landfills that were tested in 2014 compared with the landfills tested in 2020. Some PFAS variations were not analysed in 2014 but has been included for analysation in 2020. While certain PFAS variations were not analysed back in 2014 does that not mean that they do not exist within the leachate. There are probably also PFAS variations from the tests in 2020 that were not analysed and therefore not found. Furthermore, some of the PFAS variations within the landfills had so low concentrations that they were below the minimum analysis value, and it is unclear exactly what their concentrations are due to their low concentrations.

The main difference between NSRs landfills is the vastly different total PFAS concentration found in Filborna, almost 4 000 ng/l, compared to the other three where Nyvång had the highest recorded total PFAS concentration with over 500 ng/l. It is also interesting to note that there is one PFAS variation that were observed only in Tjörröd, PFPeA, even though Törröd and Filborna were tested the same year. Equally interesting is the concentration of 6:2 FTS that was observed in Filborna in this test, 2100 ng/L. This is the highest concentration of a single compound within these landfills. The next highest concentration, 580 ng/L was also found in Filborna, which is far below the concentration of 6:2 FTS.

Table 1 concentrations in leachate at four of NSRs landfills, Sventorp, Nyvång, Filborna, and Tjörröd (Sorelius 2021). Tjörröd, Sventorp, and Nyvång all have rather low PFAS concentrations, around 300 to 500 ng/l of PFAS. Filborna has in comparison a rather high concentration of PFAS in their leachate which almost reaches 4 000 ng/l. The highest single concentration found was for 6:2 FTS in Filborna with a concentration of 2 100 ng/l. NA =not analysed.

Unit ng/L				
Substance	**Sventorp 2014**	**Nyvång 2014**	**Filborna 2020**	**Tjörröd 2020**
PFBS	<25	<25	130	15
PFHxS	34.9	64	98	19
PFHxA	38.5	163	580	58
PFHpA	27.4	62.2	230	24
PFOA	74.7	133	270	57
PFNA	<16.7	<16.7	21	<1
PFDA	<16.7	<16.7	<10	<1
6:2 FTS	NA	NA	2 100	1.4
PFOS	57.9	55	140	20
PFBA	NA	NA	310	47
PFPeA	NA	NA	NA	57
Total	**291.8**	**535.6**	**3 889**	**300.4**

The division between long and short PFAS variations (Figure 10) within NSRs landfills varies between landfills. In Tjörröd and Nyvång it is the short chain variations that are the dominant PFAS variations. For Filborna and Sventorp the composition is the opposite with the longer variations having the highest presence. Filborna, the largest landfill has the highest presence of long-chained variations of any of the landfills.

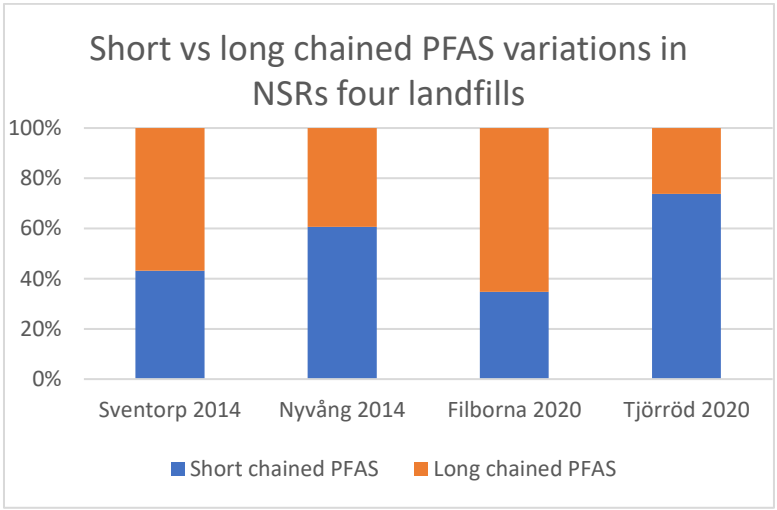


Figure 10 Comparison between concentrations of short-chained and long-chained variations (carbon chain division happens at 6 to 7 C) using the data from table 1. Tjörröd and Nyvång shows similar results with a higher presence of shorter PFAS variations. In Filborna and in Svemtorp the situation is reversed with long chained variations dominating.

4.3 Materials and methods

The modelling-based part of the thesis was conducted by constructing overviews of the water balance of the landfill, as shown above as hydrological flow charts, as well as calculating the possible evapotranspiration, water balance and potential uptake of PFAS. The water balance and evapotranspiration were calculated for six consecutive years, 2015 – 2020, in order to track variations in water flow between the different years.

Calculations for evapotranspiration could be made for both *Salix* and *Miscanthus* while the potential PFAS uptake could only be calculated for *Salix* due to lack of data for *Miscanthus* in relation to PFAS uptake. In addition, no calculations for adding biochar as a substrate in the soil were made also due to lack of available data and due to time constraints as there was deemed to not be enough time for making such calculations in regards to biochar.

4.3.1 Calculations

In this thesis it was decided to calculate the evapotranspiration for *Salix* and *Miscanthus*, as well as using a PFAS uptake equation with the basis in biomass production. The evapotranspiration is used because there is a connection between the evapotranspiration and the uptake of organic contaminants (Gobelius, 2017) and certain phytoremediation mechanisms connected to the translocation of contaminants and volatilization are more effective when the evapotranspiration is high (Pilon-Smits, 2005). For calculating the actual PFAS uptake finding relevant equations was more difficult as the research is still relatively new with plenty of uncertainties regarding which parameters are the most important and there are therefore difficult to identify equations suitable for the task. The equation that was used based the uptake on biomass production (Larsson 2020) which is a reasonable parameter as most plant species that are promising for phytoremediation are plant species that are fast growing, have a high plant uptake, as well as have a high biomass production, amongst other things (Pilon-Smits, 2005).

Evapotranspiration

The phytoremediation of PFAS is dependent on how much water the specific plants could take up (Arthur et al 2005). To get this information the transpiration rates and evaporation rates were calculated for both *Salix* and *Miscanthus.* The evapotranspiration was based on the Penman-Monteith equation *(eq 1)* as proposed by Allen et al (1998). This equation is a so called combination equation based on two parts: first an energy flux calculation where the net radiation and soil heat flux is calculated for available energy, the second part relates to meteorological conditions for the potential evapotranspiration at that time expressed with temperature, windspeed and vapour pressure (Allen et al 1998). Closer descriptions of the calculations are provided in appendix 1.

$$ET_0 = \frac{0.408\Delta(R_n - G) + \gamma \frac{900}{T + 273} U_2(e_s - e_a)}{\Delta + \gamma(1 + 0.34U_2)} \qquad \textit{Eq. 1}$$

ET_0 = reference evapotranspiration [mm/day]
R_n = net radiation at crop surface [MJ/m^2/day]
G = soil heat flux density [MJ/m^2/day]
T = mean daily air temperature at 2 m height [˚C]
U_2 = wind speed at 2 m height [m/s]
e_s = saturation vapour pressure [kPa]
e_a = actual vapour pressure [kPa]
e_s - e_a = saturation vapour pressure deficit [kPa]
Δ = slope vapour pressure [kPa/ ˚C]
γ = psychrometric constant [kPa/ ˚C]

The results from the Penman-Monteith equation gives the potential evapotranspiration rate for a reference crop (ET_0). The species-specific evapotranspiration is calculated using crop coefficients (Kc) values following the equation (Allen et al 1998)

$$ET_{Salix,Miscanthus} = Kc_{Salix,Miscanthus} \cdot ET_0 \qquad \textit{Eq. 2}$$

The Kc value (appendix 1) is a monthly value that changes throughout the year. The species-specific evapotranspiration could then be parted into the specific flows – evaporation, transpiration, and interception. For *Salix*, a ratio of 67%, 22% and 11% was used for transpiration, evaporation, and interception (Persson, 1992). No such data could be found for *Miscanthus*, instead, data estimated for *Panicum Virgatum* was used since this species has been deemed to have similar properties as *Miscanthus*. The ratio used here was 72%, 7% and 21% for transpiration, evaporation, and interception (Yimam et al 2015).

PFAS uptake in Salix

PFAS can be taken up in different ways, through the root systems or by foliar uptake (Jiao et al 2020). In this thesis the interest is mainly in uptake through the root system, and it has been proven that different PFAS variations follow different paths (Jiao et al 2020). Though, most studies that have been reviewed has focused on which plants are suitable for phytoremediation and has not delved further into understanding the mechanisms for the uptake.

The calculations for a theoretical removal of PFAS by *Salix*, by uptake via roots, could be made based on equations by Larsson (2020) where she researched the potential PFAS removal looking at different landfills in Sweden. The equation is as following and is further explained in appendix 3.

$$X = U \cdot B \cdot 10^{-6} \qquad \textit{Eq. 3}$$

X is the potential removal of PFAS based on biomass and average PFAS uptake. U is the average uptake in plants for dry biomass [µg/kgDW], and B is the biomass production for *Salix* in dry weight [kgDW/m^2/year].

For this calculation, a medium and maximum estimation of biomass production was used to calculate B, in order to get an expected medium and maximum for potential PFAS removal based on the biomass. The biomass production was based on *Salix* that had been growing for four years as that is usually when they are beginning to be harvested (Dimitriou & Aronsson, 2005). After four years the biomass production range was expected to be 1.75 – 3.5 kgDW/m^2/ (Salixenergi).

The PFAS uptake, U, was based on PFAS data from a test regarding PFAS uptake by *Salix* in Filborna. For one growing period the *Salix* had been irrigated with leachate from the landfill containing PFAS. After an event where foam had formed on a nearby leachate treatment pond, parts of the *Salix* plantation was covered with this foam as the wind dragged the foam from the treatment pond to the plantation. The foam contained high amounts of PFAS. Therefore, it is uncertain how much PFAS the *Salix*-plants were exposed for. However, NSR decided to take samples from different parts from the trees, (foliage, wood, and roots), as well as soil samples to see how much PFAS had been taken up at in relation to the soil concentrations (appendix 3). Some of the tree samples were washed before analysis to get rid of the foam and some samples

were analysed without washing for comparison. In the calculations made here results for washed samples were used to eliminate uptake from the foam as much as possible.

4.3.2 Water balance

Finally, based on the hydrological flowcharts and species-specific evapotranspiration rates, two water balance calculations could be made, the calculations are provided in appendix 2. The first water balance calculation looks at the water movement through *Salix* and *Miscanthus* based on the vegetation area, Swecos irrigation rates (Table 2), and how the leachate and stormwater volumes change based on yearly precipitation. The second calculation is based on the evapotranspiration for both species and is used to investigate if the irrigation rates, as proposed by Sweco, are reasonable or if any adjustments for the irrigation rates needs to be made.

4.4 Data

Table 2 In data for water balance calculations. The area and precipitation for Salix and Miscanthus has been used to calculate species-specific water balance which uses the evapotranspiration to calculate how much irrigation is possible. The irrigation, precipitation, leachate, and stormwater have all been used as parameters within the landfill water balance calculation.

Parameters	area (ha)	irrigation (m3)	Precipitation		Leachate (m3)	Stormwater (m3)
Salix	2	20 000	2015 – 2020	*Phase 1*	56 031	38 280
Miscanthus	3	18 000	2015 – 2020	*Phase 2*	15 620	28 471

The data used for the evapotranspiration and other calculations are provided in table 3 and comes from SMHI weather station in Helsingborg (2021), the ICOS research station in Hyltemossa (Heliasz 2021), Sweco, NSR, and data from previous research.

Table 3 Table of the data used in the calculations and the data sources. The climate data was obtained from the weather station in Helsinborg, while the radiation data was obtained from the ICOS research station in Hyltemossa. PFAS data was obtained from NSR, and the data for previous PFAS uptake in Salix comes from a field trial in Filborna landfill. Previous waterbalance calculations were obtained from Sweco

Data	**Source**
Temperature (Daily maximum & minimum)	SMHI, Helsingborg (2015 – 2020)
Precipitation (monthly data)	SMHI, Helsingborg (2015 – 2020)
Wind speed (Daily mean)	SMHI, Helsingborg (2015 – 2020)
Net radiation	ICOS, Hyltemossa (2015 – 2020)
PFAS in landfills	NSR (2014, 2020)
PFAS uptake in Salix	NSR (2020)
Water balance calculations	Sweco (2020)

Atmospheric data were provided by SMHI (temperature, precipitation, and wind speed) and from the ICOS station in Hyltemossa (net radiation). Data regarding PFAS concentrations at the landfills and any landfill specific data for Höganäs were obtained from NSR and technical reports from Sweco (2020).

Calculations done by Sweco on behalf for NSR shows that for a partly covered up landfill the leachate can be assumed to be 56 031 m^3 and the total inflow of water including stormwater being 71 651 m^3. In 2028 the landfill is expected to be completely covered up leading to lower water flows through the area. The leachate is expected to be around 38 280 m^3 and the total water flow to be 59 296 m^3.

For calculating potential PFAS uptake in *Salix* previous obtained data was used. For *Miscanthus* no such data was available making the calculations for *Miscanthus* currently impossible.

5 Results

5.1 How much PFAS could potentially be removed

At Tjörröd it is expected that around 6 g PFAS per year will be added to the *Salix* area by irrigation (based on a constant irrigation of 20 000 m3 leachate per year and the concentrations of PFAS in the leachate (Table 1), see Appendix 3). Assuming that the *Salix* plants at Tjörröd could take up the same amount of PFAS yearly as the *Salix* plants did after one growth season in the experiment at Filborna (appendix 3), and that harvest could be done every fourth year, the potential removal could reach up to about 39 – 77% (calculated as the sum of PFAS 11). This calculation is based on an estimated medium and maximum biomass production at the site (see 4.2.1 Calculations). PFBA and 6:2 FTS shows the highest uptake potential whereas PFBS and PFOS have lower potentials.

Table 4 Calculation of potential PFAS removal at Tjörröd landfill after 4 years based on medium and maximum biomass production. Two short-chain and long-chain variations were chosen, as well as the total PFAS concentration at the landfill. Highest uptakes are reported for 6:2 FTS (1.8 – 3.6 g) and PFBA (2.3 – 4.5 g), while PFBS (0.014 – 0.03) and PFOS (0.03 – 0.04) both have low uptakes. This means that there is both a high and low potential uptake for both the long and short chained PFAS variations. The total PFAS uptake after for years can potentially reach 19 g, 77 % of the total PFAS concentration.

PFAS	*Incoming mass flow of PFAS* *[g/year]*	*Potential PFAS removal through harvest of Salix leaves and stems after 4 years, based on medium or maximum biomass production* *[g]*		*Potential PFAS removal* *[%]*
		Medium	Maximum	
PFBA (Short)	0.94	2.3	4.5	61 – 121
PFBS (Short)	0.3	0.014	0.03	1 – 2
6:2 FTS (Long)	0.024	1.8	3.6	1588 – 3177
PFOS (Long)	0.4	0.03	0.06	2 – 4
PFAS 11	6	9	19	39 – 77

The separation of PFAS in the above ground biomass (Table 5) shows that there is generally a higher uptake within the foliage than in the wood. The two exceptions for this are PFBS where there was no uptake at all, and for PFOS where the maximum uptake was calculated to only 0.04 g after four years of *Salix* growth.

Table 5 PFAS Calculated uptake in wood (after four years, assuming accumulation over the years) and in foliage (at the fourth year, assuming equal concentration as in Salix at the Filborna Experiment after one growth season) in dry weight (d.w.). The foliage could take up larger amounts of PFAS than wood, the exceptions are PFOS and PBS where the wood had higher accumulations than foliage. No data could be calculated for the roots which will affect the total amount of PFAS that can be accumulated.

PFAS uptake after 4 years based on medium or maximum biomass production	Wood (g) (medium – max)	Foliage (g) (medium – max)
PFBA	0.06 - 0.13	2.22 - 4.43
PFBS	0.01 - 0.02	0
6:2 FTS	1.63 - 3.25	0.15 - 3.56
PFOS	0.02 - 0.04	0.01 - 0.02
PFAS 11	2.23 - 4.46	7.05 - 14.1

5.2 Potential evapotranspiration and transpiration rates for Salix and Miscanthus

The differences between the species-specific evapotranspiration and reference evapotranspiration (Figure 11) for the years 2015 - 2020 shows that *Salix* has a much higher capacity for evapotranspiration than *Miscanthus* for the period 2015 - 2020.

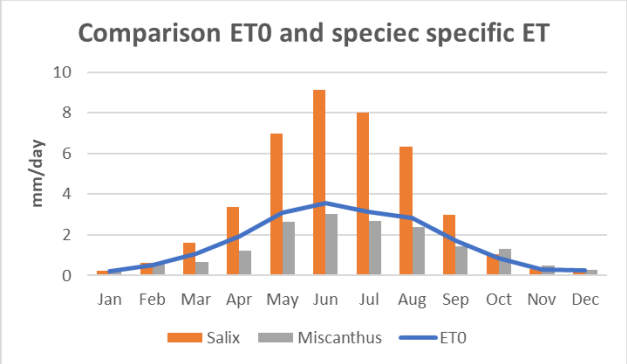


Figure 11 Evapotranspiration for both Salix, Miscanthus, and a reference crop (ET0). ET0 shows the evapotranspiration for a reference crop in the area around Tjörröd. During the summer, the ET0 reaches almost 4 mm/day. In comparison, Salix has a summer evapotranspiration of around 9 mm/day for the same conditions. On the other hand, Miscanthus only reaches up to 3 mm/day.

Looking into the monthly species-specific evapotranspiration for the same years shows that for *Salix* (Figure 12) the monthly average during these years reaches up to a maximum ET of 8 mm/day. Looking at the specific years 2018 shows the highest ET with over 10 mm per day. The ET peaks somewhere around June for most years, 2015 and 2018 are exceptions to this as the peaks during both those years occurs later in the season, between July and August. 2017 has the lowest ET peak above 6 - 7 mm/day.

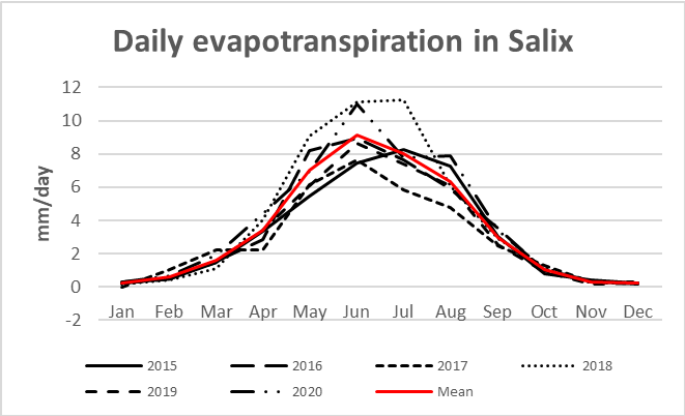


Figure 12 The evapotranspiration for Salix over a year simulated for the years 2015 – 2020. Highest points were reached in June and July for the years 2018 and 2020 when the daily evapotranspiration reached over 10 mm/day. The mean value ranges from 0 mm/day during the winter months, to 8 or 9 mm/day in the summer.

The same patterns can be seen for the daily evapotranspiration in *Miscanthus* (Figure 13) with 2018 experiencing the most overall evapotranspiration with a peak in July. Here it seems that 2015 has an even later peak in August only to rapidly fall in September. 2020 peaks in June just slightly below the rates in 2018, only to have a smaller peak again in August. The mean evapotranspiration reaches a maximum 3 mm/day during June.

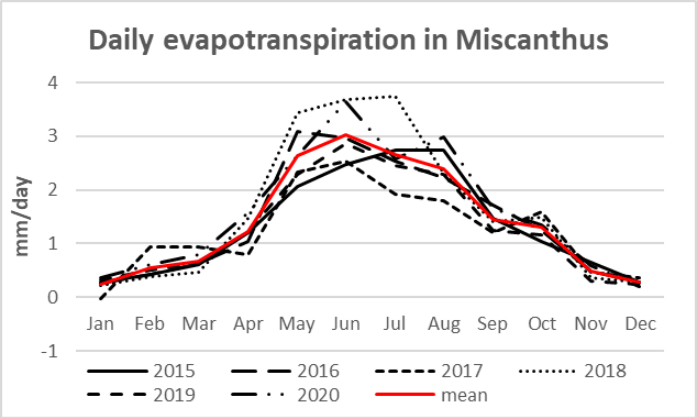
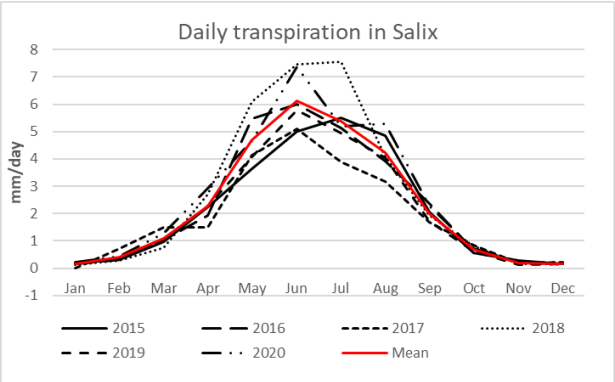


Figure 13 Daily evapotranspiration rates for Miscanthus over a year simulated for the years 2015 – 2020. 2018 and 2020 reaches an evapotranspiration rate around 3.5 mm/day during June and July, which are the highest evapotranspiration rates for Miscanthus that occurred during this period. The mean reaches around 3 mm/day during the summer and is situated close to 0 mm/day in winters.

Looking specifically at the changes in transpiration rates (Figure 14), it is clear that *Salix* has a higher transpiration of water than *Miscanthus* with the means reaching a water uptake of 6 mm/day and 2 mm/day respectively.



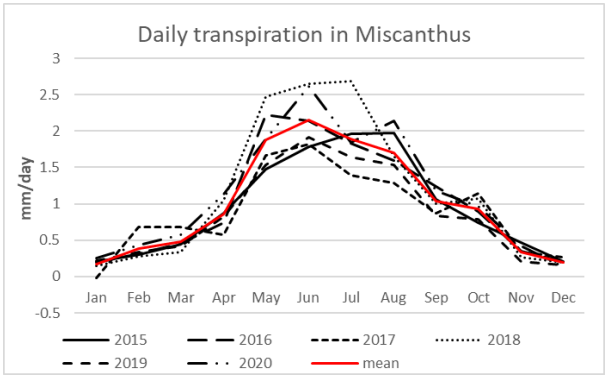


Figure 14 Transpiration rates for Salix and Miscanthus between the years 2015 – 2020. The transpiration rates for Salix, reaches up to 6 mm/day on average during summer but can reach up to 7 mm/day, Miscanthus has a transpiration rate that during the years 2015 – 2020 on average reached 2 mm/day in June, but in 2018 and 2020 exceeded the average with 0.5 mm/day, reaching up to 2.5 mm/day.

5.3 Prospective volume rates for the future

Looking at the actual water removal by evapotranspiration in the vegetation for both *Salix* and *Miscanthus* (Table 6) it is clear that the highest water removal happens via transpiration. The irrigation, which is based on the difference between the incoming precipitation to the vegetative area and the evapotranspiration, indicates the volume of water that could be added to the vegetation as irrigation. For *Salix*, the evapotranspiration is higher than the incoming precipitation which means that irrigation is possible. Though the rate of irrigation will differ between the years. For *Miscanthus*, the evapotranspiration is generally lower than the precipitation, meaning that *Miscanthus* does not really need any extra water in form of irrigation. The exception for this is 2018 where the evapotranspiration was higher than the precipitation.

Table 6 Water balance focusing on the possible irrigation rate for Salix and miscanthus. The irrigation is based on the evapotranspiration rates for both species, as well as the yearly incoming precipitation. Due to the differences in precipitation is it shown that a steady irrigation is not possible. For Salix, the irrigation rate was 20 000 m³/year which can only be achieved during 2018 which was a warm year with low precipitation rates. Miscanthus which has a proposed irrigation of 18 000 m³ has no need for irrigation for most years, as shown by the irrigation rate being 0. However, 2018 did require some irrigation, as shown below.

Salix (m^3)		**2015**	**2016**	**2017**	**2018**	**2019**	**2020**
Vegetation model	*Precipitation*	*15 000*	*12 400*	*15 400*	*8 600*	*12 600*	*11 800*
	Interception	5 200	5 500	4 600	6 400	5 00	6 000
	net Prec	9 800	6 900	10 800	2 200	7 600	11 700
	Transpiration	15 800	16 700	13 900	19 600	15 400	18 400
	Soil Evap	2 600	2 700	2 300	3 200	2 500	3 000
	ET	23 600	25 000	20 800	29 300	23 00	27 600
	Irrigation	8 600	12 600	5 400	20 700	10 400	15 600
Miscanthus (m^3)							
Vegetation model	*Precipitation*	22 500	18 600	23 100	12 900	18 900	17 700
	Interception	3 000	3 300	2 800	3 700	3 000	3 000
	net Prec	19 500	15 400	20 300	9 200	16 000	14 700
	Transpiration	10 300	11 200	9 800	12 700	10 200	10 600
	Soil Evap	1 000	1 100	950	1 200	1 000	1 000
	ET	14 300	15 500	13 600	17 600	14 200	3 000
Remaining water	*Irrigation*	0	0	0	4 700	0	0

The tables for water balance calculations (Table 7 and table 8) are based on the flowchart created for Tjörröd landfill (Figure 7). The landfill water is the total water amount that can emerge in the landfill and includes both leachate and stormwater. The precipitation only includes *Salix* as *Salix* is situated outside of the landfill and therefore is not the incoming precipitation accounted for within the stormwater. For *Miscanthus*, the precipitation is calculated within the stormwater, which is based on the runoff for the entire landfill. The evapotranspiration rates and irrigation rates are based on what was obtained in table 6. Lastly, the water to storage indicates the water that enters the storage, which includes the leachate that was not used for irrigation, stormwater, as well as any runoff obtained from *Miscanthus*. The water to recipient refers to the amount of water that will be needed to be released to a recipient and is based on the water to storage volumes with the assumption that every year the ponds can store the maximum volume of 26 000 m^3.

The water balance calculations for phase 1 (Table 7) shows that within the landfill the largest waterflows happened in 2017 where it almost reached 80 000 m^3, and the lowest flows was as little as 43 500 m^3 water during 2018. For Sweco there would not need to be a high release to a recipient at all. However, when looking at the other years where the precipitation within the vegetation area has also been considered then the need for a recipient is much larger. The only exception is 2018 when no water will be needed to be released to a recipient.

Table 7 Water balance for phase 1 made for both Sweco and the years 2015 to 2020. The Landfill water indicates that the total water volume fluctuates between 40 000 m^3 to 75 000 m^3 during this period. Included in the total water flow is the incoming precipitation for Salix, which is not included in the landfill water calculation but still an addition of water to the system. The different irrigation rates that have been obtained for Sweco and the years 2015 – 2020 show that Swecos irrigation rate of 38 000 m^3 per year is rather high. This rate is not reached at all during the period 2015 – 2020, indicating that Sweco has overestimated the irrigation potential. This leads to Sweco having lower water volumes that will be needed to be stored and in turn released to a recipient, than what is expected for the different years. The exception is 2018, which was warm and dry leading to low water volumes in the landfill and no need for releasing any water.

Phase 1 (m^3)		**Sweco**	**2015**	**2016**	**2017**	**2018**	**2019**	**2020**
Landfill water	Leachate	48 600	48 800	40 400	50 200	28 100	41 000	38 500
	Stormwater	23 100	13 800	11 400	14 100	7 900	11 600	10 900
	Total	**71 700**	**75 600**	**62 700**	**77 700**	**43 500**	**63 600**	**59 600**
Precipitation	Salix	14 200	9 800	6 900	10 800	2 200	7 600	11 700
Evapotranspiration	Salix	20 000	23 600	25 000	20 800	29 300	23 000	27 400
	Miscanthus	18 000	14 200	15 500	13 500	17 600	14 100	14 600
	Total	**38 000**	**37 800**	**40 500**	**34 300**	**47 000**	**37 100**	**42 100**
Irrigation	Salix	20 000	8 600	12 600	5 400	20 700	10 400	15 600
	Miscanthus	18 000	0	0	0	4 700	0	0
	Total	**38 000**	**8 600**	**12 600**	**5 400**	**25 400**	**10 400**	**15 600**
Water to storage		33 700	76 800	57 000	83 100	20 300	60 800	55 700
Water to recipient		7 700	50 800	31 000	57 100	-5 700	34 800	29 700

Looking at phase 2 (Table 8) the total water volumes in the landfill for the same period would be between 36 000 m3 and 64 000 m3. As phase 2 corresponds to the landfill being completely covered up this means that most of the decreased volumes is due to decreased leachate rates and some decrease in stormwater. As the evapotranspiration rates will remain the same for this phase will also the irrigation rate remain the same. There are however slightly lower rates for water that is needed to be released to a recipient.

Table 8 Water balance for phase 2, landfill is completely covered up. The leachate rates are much lower than during phase 1, while the stormwater has only slightly decreased or even increased instead. Here, using the numbers provided by Sweco, it would be expected that there would be no need to release any water at all. However, looking at the calculations for 2015 – 2020 this is only the case for 2018. During this year, the landfill water volume is low, and the evapotranspiration rates high. This leads to no water having to be released during this year. For the other years, there will still be water that will be needed to be released to a recipient, though these rates would be much lower than the rates that are shown in phase 1 when the landfill is not completely covered up yet.

Phase 2 (m^3)		**Sweco**	**2015**	**2016**	**2017**	**2018**	**2019**	**2020**
Water to treatment	Leachate	38 200	40 400	33 500	41 500	23 200	34 000	31 800
	Stormwater	21 100	22 200	18 400	22 800	12 800	18 700	17 500
	Total	**59 300**	**36 100**	**51 500**	**63 800**	**35 800**	**52 300**	**49 000**
Precipitation	Salix	14 200	9 800	6 900	10 800	2 200	7 600	11 700
Evapotranspiration	Salix	20 000	23 600	25 000	20 800	29 300	23 000	27 400
	Miscanthus	18 000	15 000	14 300	13 600	17 600	14 200	14 700
	Total	**38 000**	**37 800**	**40 500**	**34 300**	**47 000**	**37 100**	**42 100**
Irrigation	Salix	20 000	8 600	12 600	5 400	20 700	10 400	15 600
	Miscanthus	18 000	0	0	0	4 700	0	0
	Total	**38 000**	**8 600**	**12 600**	**5 400**	**25 400**	**10 400**	**15 600**
Water to storage		21 300	63 800	46 200	69 700	12 800	49 900	45 400
Water to recipient		-4 700	37 800	20 200	43 700	-13 200	23 900	19 400

The evapotranspiration calculations for *Miscanthus* shows that there is no need for further irrigation which greatly reduces the irrigation volume. This means that there are higher volumes of water that is needed to be stored during the years when *Miscanthus* does not need to be irrigated. *Salix* should also for most years have lower irrigation rates than what is proposed. With these lower irrigation rates then there is an increased need for release to a recipient.

6 Discussion

6.1 Potential of reducing PFAS in leachates from Swedish landfills

According to research there are usually a few short-chained PFAS variations that are common in landfills, both in Sweden and elsewhere (Gobelius et al 2018: Knutsen et al 2019). The fact that the shorter variations are dominating within the leachate at Tjörröd is not good as the short-chained variations are much more mobile than their long-chain counterparts which increases the risk of PFAS spreading from the landfill. Combine high mobility PFAS with no regulations regarding PFAS concentrations in the landfills and the results could lead to even higher levels of PFAS contamination in the environment.

The research regarding using phytoremediation and biochar for PFAS remediation is still relatively new with most of the studies having been published during the last decades. Furthermore, most of the studies researching the remediation of PFAS using these two alternatives are either performed in laboratory situations or in other environmental climates making it somewhat difficult to theorize about their practical application. Theoretically, phytoremediation would be better suited for short chained PFAS variations as they accumulate within the plant and biochar for long variations as these can bind to the surface of the biochar. This could indicate that the combination could be useful in situations where the PFAS contamination is complex as it within landfills. The combination has previously been used and tested with regards for heavy metal remediation rather successfully where it was noted that the

combination should be used in areas where they could target different contaminants (Paz-Ferreiro 2014).

Various plant species has been noted to accumulate PFAS to varying degrees. Exactly which parameters are important for identifying suitable plant species are still unclear though one of the most common ways of determining suitability seems to be by calculating a BCF factor where a higher BCF factor means that the plants are better for the task. In Sweden, specifically there is one major study looking at the accumulation of PFAS within various plants which was performed by Gobelius et al (2017). They concluded that phytoremediation could be useful for PFAS removal but that not all species had high accumulations, and that there should preferably be a mixture of a few species as they are likely to accumulate PFAS chemicals differently and thus increasing the chance of targeting more substances. A few species that they, and others, have indicated to have a possibility are Silver Birch (*Betula Pendula*), Norway spruce (*Piceas abies*) (Gobelius et al 2017), and Red fescue (*Festuca rubra)* (Huff et al 2020), and the green parts of various cereals (Gishi et al 2019), which all are common species in Sweden and can thrive in a cold climate. It should be noted that the suitability will probably also depend on latitude as one species will most likely preform differently depending on if the landfill site is in Scania or somewhere in the more northern areas.

6.2 Suitability of using *Salix* and *Miscanthus* in Tjörröd landfill for phytoremediation in practice

6.2.1 Concentrations of PFAS found in leachates

The waste within the landfill contains PFAS. The waste leaches causing the leachate that forms within the landfill to contain PFAS as well. The PFAS contamination within the leachate will vary between different landfills (Gobelius et al 2018: Modin et al 2018), as is shown by table 1. The reason for Filborna having a larger PFAS contamination within their leachate in comparison to the other NSR landfills is most likely due to Filborna being much larger than the other three and handles higher volumes of waste. Comparing the concentrations for Filborna and Tjörröd it is clear that the concentrations are much higher for Filborna than Tjörröd. This will have implications for the calculations of the PFAS uptake in Tjörröd as those calculations are based on a test regarding PFAS uptake in Filborna, which will most likely give extremely overestimated results regarding the potential PFAS removal for Tjörröd, as higher PFAS concentrations in the soil (or soil-water) will lead to a higher uptake (Gobelius et al 2017).

Table 2, which looks at the PFAS chain length in the different landfills shows that depending on landfill the percentages between long-chain and short-chain PFAS are different between each other. In Tjörröd and Nyvång there is a higher presence of shorter variations. However, Filborna and Sventorp are exceptions to this as they have a larger amount of long chained PFAS variations than short chained.

Comparisons between the different landfills is rather difficult as some of them were tested for PFAS back in 2012 when they only looked at 9 PFAS variations. The other landfills were tested in 2020 with detection for 11 PFAS variations (Eurofins 2017). This means that there is a high possibility that more PFAS variations are present within the leachate for the landfills that were tested back in 2012, they just did not look for more variations which could have led to them missing to detect some substances.

6.2.2 How much PFAS could potentially be removed

The amount of PFAS that could potentially be removed is based on the results from the Filborna experiment with Salix and an estimated medium and maximum biomass production, with harvest every fourth year. The calculations indicated that 9 - 19 g of sum PFAS-11 could be removed from Tjörröd landfill, which translates to 39 to 77% of the total sum PFAS-11 added

by an irrigation of 20 000 m^3 per year for four years (Table 4). The large span is because of uncertainty within biomass production and age of the trees as older trees take up more water than younger trees, leading to higher uptake rates. If the soil would be healthy then it could be expected for Salix to have a large biomass production of roughly 35 ton/hectare in Scania, but since the Salix will be planted near a landfill and be irrigated with contaminated water it is not unreasonable to expect a lower biomass production, though it is difficult to say how much the growth will be affected by the conditions prevailing within the site.

As can be seen in table 4 and 5 the accumulation within the plants differs between different substances. PFOS, which is a long chain variation, has an incredibly small uptake within Salix with a maximum of 4% of the total addition of PFOS (Table 4), most of which is taken up within the wood (Table 5), and lower amounts being expected in foliage. This is not surprising as PFOS does not accumulate very well within foliage (Sharma et al 2020). PFBA is a short variation with an incredibly high uptake (Table 4), most of which accumulates within the foliage (Table 5). The reason for PFBA being taken up more than 100 % is most likely due to the assumed uptake concentrations, which were obtained from Filborna, are too high for Tjörröd landfill where the concentrations are much lower. Table 1 shows that the concentrations in Filborna were 10 times higher than the concentrations in Tjörröd, 3 900 ng/l against 300 ng/l. This leads to over assumptions of how much PFAS Salix can take up when the concentrations are lower. Both PFOS and PFBA are showing behaviours that are as expected, long chained variations having low accumulation in above ground vegetation and short-chain variations having a high accumulation, especially within the foliage.

On the other hand, 6:2 FTS, a long variation, and PFBS both show opposite behaviour, with 6:2 FTS having higher uptakes in above ground biomass than PFBS (Table 4). Looking at the expected uptake for wood and foliage in Tjörröd (Table 5), 6:2 FTS has a higher medium accumulation within wood than within foliage which is also shown for PFOS and is explained by long-chained variations being less mobile and cannot travel as far. For PFBS there is no uptake within foliage at all and only a small uptake within the wooden parts of the plant. This behaviour can be explained by the higher incoming mass flow of 6:2 FTS compared to PFBS in the Filborna experiment, which is the data that the calculations were based on. 6:2 FTS has an extremely high concentration, over 2 000 ng/L whereas PFBS is barely present within the leachate (Table 1). As mentioned above, the uptake is also depending on the concentrations of the contaminant that the plant roots are exposed to. This makes the investigation of the PFAS concentrations within the leachate, or actually the soil-water to which the roots are exposed, extremely important as well as the chemical mixture as this also influences whether a remediation method is suitable or not for usage.

The calculations for PFAS removal were done for four consecutive years as that is the minimum growing period for *Salix* before harvest and to simulate the effects of litterfall during autumn. As the leaves will fall down every year, they are bound to add PFAS back into the soil, which will cause the PFAS concentrations to increase every year with the continuation of leachate irrigation. There is also a risk of PFAS contamination outside of the *Salix* plantation if the leaves are transported by wind to the nearby area.

The potential PFAS uptake includes sources of errors. The calculations were only based on the biomass production which is a simplification of the uptake as it most likely also influenced by other parameters, such as the transpiration of water within the plants. Furthermore, the uptake has only been calculated for the above ground biomass and ignoring the roots, even though PFAS has been shown to be accumulated by the roots, which was proven by Sharma et al (2020). The accumulation of long PFAS below ground causes the long-chained variations to remain in the soil and are therefore still susceptible to leaching. This is not an issue for

Miscanthus as this plantation has a protective layer under the soil, but for *Salix* it is a problem as there is not a protective layer surrounding the vegetation, which can lead to percolation of the PFAS that has not been bound to the roots if too much water is added to the vegetation.

Another issue is that the PFAS uptake for this calculation was based on values obtained from Filborna landfill and not from Tjörröd. Table 1 shows that the concentrations and composition of PFAS differs from each other which causes the result for the potential PFAS uptake to be different from what actually could be obtained. The data from the Filborna test also included uncertainties as the *Salix* in Filborna was not only subjected to PFAS by irrigation, but also to additional PFAS in the form of leachate foam which had covered the part of the plantation. This causes issues with not only knowing the incoming mass flow of PFAS to the area, but it is also making it difficult to evaluate if the PFAS was taken up through the soil or through the foam.

All in all, this results in that the values shown in table 4 and 5 are greatly overestimated for Tjörröd landfill and the uptake will most likely be much lower than what is shown in this study as the PFAS concentrations also affects how much PFAS the vegetation can take up (Gobelius 2017). It is therefore important to test the phytoremediation of PFAS at the actual site of concern to achieve more reliable results.

6.2.3 Potential evapotranspiration and transpiration rates for Salix and Miscanthus

It was decided to calculate the evapotranspiration potential of Salix and *Miscanthus* to evaluate if the vegetation areas are capable of handling the water flow and keep the outflow to a recipient as low as possible. The equation used was the Penman-Monteith as it is one of the most used equations for evapotranspiration calculations (Zotarelli, 2010).

Comparing the evapotranspiration rates of the plants, (Figure 12 & Figure 13), *Salix* should be more effective than *Miscanthus* for PFAS adsorption due to a higher water uptake, over 10 mm/day in favourable conditions, which is comparable to other studies regarding evapotranspiration of *Salix* (Frédette, 2019). *Miscanthus* was found to have an evapotranspiration of 3.5 mm/day during the same conditions, which is lower than what has previously been reported (Holder et al 2018) though the study was conducted in Wales which would lead to their results being slightly higher than what can expected for the rates in Sweden. On the other hand, *Miscanthus* has a higher transpiration percentage than *Salix,* 0.72 against 0.67 which should mean that *Miscanthus* is more effective for transpiration than *Salix*. Even if *Miscanthus* is more effective in terms of transpiration is the plant still not capable to handle all of the proposed water volume and without *Salix* more water would be needed to be let out to a recipient.

What should be mentioned is that though *Miscanthus* can accumulate less PFAS within one year than *Salix,* based on the evapotranspiration, will also all of the accumulated PFAS be removed from the site as *Miscanthus* will be harvested yearly. *Salix* is only harvested every 4 to 5 years, meaning that during the winter months when there is no transpiration there could be an extra input of PFAS to the soil in the form of fallen leaves which will degrade on the site over winter, releasing more PFAS to the soil, or the leaves could be transported outside of the vegetation area by the wind, thus spreading PFAS to surrounding areas instead. To lower the risk of PFAS being spread from fallen leaves there should be some maintenance for *Salix* every autumn where the leaves are gathered together with *Miscanthus* and disposed of immediately, effectively removing the PFAS from the area. The disposal would most effectively be burning the biomass at high temperatures to completely get rid of the PFAS compounds (KemI 2015).

It should be noted that while the ET partitioning for *Salix* was taken from literature there was no such data to be found for *Miscanthus.* The transpiration rate was instead obtained from

Panicum Virgatum, a similar plant to *Miscanthus*, meaning that the transpiration rate is only similar to the actual transpiration rate for *Miscanthus* and not the actual transpiration rate.

6.2.4 Prospective volume rates for the future

Before the landfill has completely undergone a final covering (Table 7) is leachate the main water that is being produced at the landfill. For the second phase (Table 8) the leachate has been reduced to a large degree whereas the stormwater rates almost stay the same or even increases. In the future it is to be expected that the total water volume and in particular the leachate volume will decrease once the landfill has been completely covered up.

Comparing the hydrological model produced within this thesis and with Sweco water balance, Swecos water balance assume that the irrigation rates that they propose would not require to be adjusted for different years. This is not the case, as suggested by the results in table 7 and table 8. The irrigation rates are based on the evapotranspiration rates, provided in table 6 which shows the actual evapotranspiration rates during the period 2015 – 2020. The irrigation, which is based on the precipitation and evapotranspiration shows great variance between different years.

For *Salix*, the proposed irrigation values vary between 8 000 m^3 and 20 000 m^3, meaning that additional irrigation is possible but to a lesser degree than what is proposed by Sweco for most parts. *Miscanthus* does not seem to need any additional irrigation at all (Table 6) as the plants will continuously have available water in the form of precipitation and therefore is the irrigation 0 for most years. Although the irrigation is 0 for *Miscanthus* does that not mean that irrigation could not be added. Since *Miscanthus* is planted on top of the covered-up landfill is there little to no concern for any leaching of PFAS from this vegetation to the surrounding nature. All the extra water will only flow back down the landfill and enter the leachate system again, causing no potential for eventual PFAS leaching. This means that even though *Miscanthus* does not need an extra water volume of 18 000 m^3 a year is it still completely safe to irrigate the vegetation with that amount, which will lead to lower amounts of leachate that will be needed to be released to a recipient.

While overirrigation is not a problem for *Miscanthus* is it not the same case for *Salix*. Even though the water is also gathered up in the *Salix* vegetation and returned to the landfill is there still a concern for potential overirrigation. Overirrigating *Salix* could lead to PFAS leaching to lower soil layers as there is no protective layer underneath the vegetation, stopping such leaching from happening. With this in mind is the assumed volumes of 20 000 m^3 most likely an overestimation of how much water that could be used for irrigation, with exceptions for dry and warm years, which then the vegetation could be irrigated with the proposed irrigation volume.

The main source that has caused the overestimation of irrigation is the assumption of steady irrigation. Sweco assumed steady irrigation rates of 3 mm/day and 5 mm/day for all 200 irrigation days. While *Salix* could reach over 5 mm/day (Figure 14) during the summer months so is there still lower transpiration rates both earlier and later during the growing season. For *Miscanthus* they most likely they overestimated the transpiration which they assumed to be over 3 mm/day based on a study preformed in Wales (Holder et al 2018). The study found that *Miscanthus* had an evapotranspiration rate of roughly 5 to 10 mm/day in Wales. Sweco adjusted these rates for Swedish conditions, but it seems that their rate of 3 mm/day is still too high. The calculations in this research, using the same Kc as Holder et al (2018) found that the highest transpiration rates for *Miscanthus* in Scania reached 2.5 mm/day (Figure 14). To adjust this issue NSR could investigate if they could increase the area of the *Miscanthus* vegetation in order to increase the removal of leachate.

Another potential issue, which was difficult to deter, was the precipitation. When they calculated the irrigation rates, they only considered the area of the vegetation, the potential daily transpiration rate, and for how many days they could irrigate the vegetation. It did not say anywhere if they considered additional water in the form of precipitation which will enter the vegetation area. As shown in table 7 and in table 8, when adjusting for the additional water in form of precipitation the irrigation rates decrease, leading to more water having to be led back to the landfill and possibly even released to a recipient during years with heavy precipitation.

These calculations have been purely theoretical in nature. If there actually will be a PFAS uptake remains to be seen as most of the previous research into phytoremediation has used solutions with rather high PFAS concentrations, while at the specific landfill the concentration is rather small with a further dilution being possible by precipitation. The process of phytoremediation is also rather complex, as suggested by the literature. Though the only parameters that were researched in this thesis was connected to chain-length, biomass production, and evapotranspiration are there most certainly other parameters which will affect the phytoremediation as well, not to mention how adding biochar could affect the system.

6.3 Uncertainties

There are multiple uncertainties with the calculations preformed within this study. The first and most important uncertainty is related to the PFAS uptake calculations, which is still a relatively new research area with limited datasets and reliable calculations. The calculations only used one dataset obtained from Filborna landfill, making the results obtained rather uncertain as they cannot be compared with other datasets and validated as reasonable results. Furthermore, the equation obtained from Larsson (2020) only adjusted for biomass production, when in fact it is most likely that there are other parameters that affect the uptake as well which makes the equation itself difficult to assess in terms of producing reliable results. Due to the presence of these uncertainties from the beginning was it decided to not make an uncertainty analysis as there is too little data to analyse to be able to draw any conclusions regarding the uncertainty analysis. To properly assess the reliability of the method there needs to be more data and a further investigation into the mechanisms of PFAS uptake within plants

Furthermore, this thesis only investigated four PFAS variations, and the total sum of PFAS-11 and how well they can be accumulated within the plants. This is an issue as there are thousands of different PFAS compounds, but the focus is only on a few of them. So, even if the results can say something for the compounds that were tested within this thesis, would these results not be enough to draw any conclusions regarding other PFAS variations that were not tested.

This study is severely limited by the inability of collecting research data due to the time constrain as setting up and finishing this type of project would require a lot of work and time. Therefore, it was decided to design this project as a prototype using previous data in order to create a starting point for furthering the investigation into this topic. The equations used in this project were investigated and tested throughout the working process and as such the focus came to be on calculating PFAS uptake by phytoremediation. In the future, developing such calculations for biochar or other remediation techniques as well, are heavily encouraged. It would also be useful to research how the remediation techniques change during longer timescales, over a year or two to further develop these calculations as well.

7 Conclusion

PFAS has shown in this thesis to be a concerning issue due to their ability to remain in the nature for a long time and having severe toxic effects, with difficulties for environmental clean-up if there are multiple variations of PFAS within the same area. The current research regarding combining phytoremediation and biochar to help target different PFAS with different chain-lengths is small. Although some tests have been made in regards for heavy metal clean-ups. By looking at the literature there is some evidence suggesting that this idea should be further tested to see how they would work together and if the idea of phytoremediation targeting short-chained PFAS and biochar immobilizing long-chained variations holds up.

Looking specifically at the situation in Höganäs and their plantations of Miscanthus and Salix it seems that they both will reduce the amount of leachate in the area via evapotranspiration which can reach up to 10 mm/day for Salix and 3 mm/day for Miscanthus in favourable conditions. The leachate will be used as irrigation for the species, thus lowering the amount of leachate and PFAS that will be released to a recipient. The irrigation rates were previously proposed to be 20 000 m^3/year for Salix and 18 000 m^3/year for Miscanthus. However, the actual irrigation rates for both species will be lower than the proposed rates when taking incoming precipitation into account as well. *Salix* can be irrigated with volumes varying between 8 600 m^3 to 20 000 m^3 between different years. *Miscanthus* receives enough water from the precipitation to not need any additional water but due *Miscanthus* being planted on top of the landfill without any risk of spreading PFAS can they still be irrigated with the proposed amount of 18 000 m^3/year. It should be noted that while *Miscanthus* can handle the proposed amount of irrigation is the situation different for *Salix*. Here they should be more careful of how much water they can irrigate based on precipitation as there is a risk of percolating of leachate to lower soil layers as there is no protection underneath the plantation.

While the vegetation can take up some of the leachate is it still shown within the water balance that there will be a need to release leachate from the landfill. This means that it is important to implement techniques to remove as much PFAS before the leachate is released to a recipient. Salix can be a viable option for further testing as the PFAS uptake showed promising results, with a PFAS uptake ranging from 9 to 19 g after 4 years, which is equal to 39 – 77% of the total concentrations, with most PFAS being accumulated within the foliage. Though these results are most likely overestimations as they are based on previous PFAS uptake data obtained from Filborna landfill. It is therefore important to base any PFAS uptake calculations from data obtained from the actual site in order to gain more reliable results. Any PFAS uptake within Miscanthus is still unclear as there are no previous testing for the species. Considering that Miscanthus have a large transpiration rate, even if the total evapotranspiration rate is rather low, indicates that the plant might be able to absorb large amounts of water and therefore could be an interesting plant to look further into.

Biochar has not been tested in this thesis but the conclusion from the literature is still that NSR should look into the added effects of biochar, or at least some other type of black carbon to target the longer PFAS variations found at Höganäs.

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Personal comments

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Appendix 1

Explanation and calculation of the Penman – Monteith equation *eq. 1* which was used for the calculation of the reference-evapotranspiration (Allen 1998).

$$ET_0 = \frac{0.408\Delta(R_n - G) + \gamma \frac{900}{T + 273} U_2(e_s - e_a)}{\Delta + \gamma(1 + 0.34U_2)}$$

ET_0 = reference evapotranspiration [mm/day]
R_n = net radiation at crop surface [MJ/m^2/day]
G = soil heat flux density [MJ/m^2/day]
T = mean daily air temperature at 2 m height [˚C]
U_2 = wind speed at 2 m height [m/s]
e_s = saturation vapour pressure [kPa]
e_a = actual vapour pressure [kPa]
e_s - e_a = saturation vapour pressure deficit [kPa]
Δ = slope vapour pressure [kPa/ ˚C]
γ = psychrometric constant [kPa/ ˚C]

Net radiation

The net radiation was obtained as NetRad in the unit W/m2 as 30 min averages for an entire year- The first step was to calculate the entire net radiation during 30 min intervals by using the equation.

$$R_n = \text{NetRad} \cdot 30 \cdot 60 \qquad \textit{Eq. 4}$$

The current unit for the data is now J/30min/day. Following this calculation, the net radiation needed to be transformed into daily data which was done by adding all net radiation values during a single day to get the total net radiation for one day (J/m^2/day) which was then transformed into MJ/m^2/day by conversion of Joule into Megajoule.

Soil heat flux

The soil heat flux is based on the net radiation and was assumed for this calculation to be 10 percent of the total net radiation (Santanello Jr & Friedl, 2003).

$$G = R_n \cdot 0.1 \qquad \textit{Eq. 5}$$

Temperature

The temperature was obtained from daily minimum and maximum temperature data from SHMI. These where then averaged to obtain a daily mean temperature. The data was collected at 2 meters height so no adjustment for height needed to be made.

Wind speed at 2 m:

The windspeed was obtained as hourly m/s data. This data was then subsequently transformed into a daily mean by averaging all the datapoints for one day. The data was collected at 10 m above ground and therefore an adjustment needed to be made in order for the data to be the wind speed at 2 m. This was done by following the equation from (Zotarelli et al 2010).

$$U_2 = U_h \frac{4.78}{\ln(67.8h - 5.42)} \quad \textit{Eq. 6}$$

H = height of measurement above ground surface

U_h = wind speed at height h [m^{-1}]

U_2 = wind speed at 2 m

Saturation vapour pressure

The saturation vapour pressure is based on temperature. The saturation pressure is calculated as a daily average based on minimum and maximum saturation pressure.

$$e_s = \frac{E_s(T_{min}) - E_s(T_{max})}{2} \quad \textit{Eq. 7}$$

The minimum and maximum vapour pressure uses the daily minimum and maximum temperature and calculated as

$$E_s(T_{min}, T_{max}) = 0.6108 \exp\left[\frac{17.27T}{T + 237.3}\right] \quad \textit{Eq. 8}$$

where T is the temperature.

Actual vapour pressure

The actual vapour pressure is based on the daily minimum air temperature by the equation

$$e_a = 0.611 \exp\left[\frac{17.27T_{min}}{T_{min} + 237.3}\right] \quad \textit{Eq. 9}$$

Saturation vapour deficit

The saturation vapour deficit is the difference between the saturation vapour pressure and the actual vapour pressure.

$$e_s - e_a \quad \textit{Eq. 10}$$

Slope saturation curve

The slope saturation curve is based on the temperature

$$\Delta = \frac{4098\left[0.6108exp\left(\frac{17.27\mathrm{T}}{\mathrm{T}+273.3}\right)\right]}{(\mathrm{T}+273.3)^2} \quad \textit{Eq. 11}$$

Psychrometric constant

The psychrometric constant is the is based on the atmospheric pressure and the latent heat.

$$\gamma = 0.00163\frac{\mathrm{P}}{\lambda} \quad \textit{Eq. 12}$$

The atmospheric pressure changes depending on the elevation above sea level and can be calculated as where z is elevation above sea level.

$$\mathrm{P} = 101.3\left(\frac{293-0\text{-}0065\mathrm{z}}{293}\right)^{5.26} \quad \textit{Eq. 13}$$

The latent heat is a constant and was assumed for this calculation to be 2.45

Species-specific ET

The species-specific evapotranspiration was obtained by *Eq. 2* where crop coefficient (Kc) was used.

The *Salix* Kc values are based on (Curneen & Gill 2016) and *Miscanthus* (Holder et al 2018).

Table 1 Monthly Kc values for both Salix and Miscanthus

	Salix	**Miscanthus**
Jan	1	1.12
Feb	1.25	1.12
Mar	1.5	0.63
Apr	1.75	0.63
May	2.25	0.85
Jun	2.56	0.85
Jul	2.56	0.85
Aug	2.25	0.85
Sep	1.75	0.85
Oct	1.25	1.57
Nov	1	1.57
Dec	1	1.12

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Appendix 2

Water balance calculations for Tjörröd landfill.

Evapotranspiration balance

The evapotranspiration balance is based on the total evapotranspiration calculated for *Salix* and *Miscanthus*, and is based in the Penman - Monteith equation, for the years 2015 – 2020. They were transformed into species-specific evapotranspiration rates by using Kc values. The different rates were calculated using partitioning percentages and were calculated as following

$$I = ET \cdot I_{salix,miscanthus} \quad \textit{Eq. 14}$$

$$E = ET \cdot E_{salix,miscanthus} \quad \textit{Eq. 15}$$

$$T = ET \cdot T_{salix,miscanthus} \quad \textit{Eq. 16}$$

I is the interception rate and refers to the precipitation that is intercepted by the foliage and not reaching the ground. E is the evaporation rate from the soil, and T is the transpiration rate.

The next step is to calculate the irrigation rates, which is the difference between evapotranspiration and precipitation. Negative results means that there is no need for additional irrigation as there are enough water through.

$$\text{Irrigation} = ET - P \quad \textit{Eq. 17}$$

Where ET is the evapotranspiration and P is the yearly total precipitation that is received by the plantations in m^3.

P is the total yearly volume of precipitation that reaches the area in m^3. As the precipitation was given in mm a transformation into meters was needed to be made and summed together to represent the total precipitation for that given year. This value was then multiplied by the plantation area, in this case either 2 000 m^2 (2 ha) or 3 000 m^2 (3 ha) (Sweco 2020b).

Water balance

The water balance calculation was made to see how the water flows within the landfill.

First step was to identify the different water flows, which is the irrigation of $W_{landfill}$ (landfill leachate and stormwater), incoming water via P (precipitation), and the outgoing flow through ET (evapotranspiration).

$W_{landfill}$ is the total amount of water created on the landfill and as there are still uncertainties regarding the handling of the stormwater was it decided to include the stormwater in the calculations for safety.

$$W_{landfill} = \text{Leachate} + \text{Stormwater} \quad \textit{Eq. 18}$$

Incoming water via precipitation is only adjusted for *Salix* as the incoming precipitation for

Miscanthus is covered within the water volume for the landfill. The evapotranspiration is obtained by the previous Penman – monteith equation and evapotranspiration balance.

To see how much landfill water that would remain after the irrigation the following calculation was made.

$$W_{storage} = W_{landfill} + P(net)_{Salix} - Irrigation \quad \textit{Eq. 19}$$

$W_{storage}$ is how much water that is left for storage. The irrigation is the total irrigation that is added to the vegetation and is adjusted for the different years. For Sweco, the proposed irrigation rates were used, 20 000 m^3 for *Salix* and 18 000 m^3 for *Miscanthus* (Sweco 2020b). The irrigation rates for the years 2015 – 2020 were based on the irrigation obtained from the evapotranspiration balance provided above. $P(net)_{Salix}$ is the incoming net precipitation for the *Salix* area, which is not included in the calculation for the water within the landfill as the *Salix* is planted outside of the landfill. The net precipitation is the amount of precipitation that reaches the ground and is therefore seen as an addition of water.

From here, the calculation to see how landfill water that should be released to a recipient was calculated as

$$W_{out} = W_{storage} - storage\ ponds \quad \textit{Eq. 20}$$

W_{out} is how much water that will be needed to be released to the recipient. Storage ponds is the total size of the storage ponds at Tjörröd, 26 000 m^3 (Sweco 2020b).

References

Sweco. 2020b. Bilaga A Teknisk beskrivning. Sweco. Ordered by NSR

Appendix 3

The equation for potential PFAS removal was obtained from Larsson (2020) and is based on biomass production and previous PFAS uptake within *Salix.*

Calculation of potential PFAS removal

The equation for calculating the potential PFAS removal looks as following

$$X = U \cdot B \cdot 10^{-6} \qquad \textit{Eq. 21}$$

X is the potential PFAS removal

U is the average uptake in plants for dry biomass [µg/kgDW] and is based on the PFAS uptake in wood and foliage obtained from Filborna landfill.

B is the biomass production for *Salix* in dry weight [kgDW/m^2/year]

As the *Salix* will be standing for at least four years before harvest it was decided to do calculations based on four years.

Biomass production

The biomass production was calculated as following

$$B = kgDW/m^2/year \cdot A \qquad \textit{Eq. 22}$$

A is the *Salix* vegetative area, 2 000 m^2

The biomass production of *Salix* after four years was assumed to be 1.75 to 3.5 kg/DW/m^2/. To get the yearly biomass for *Salix*

$$B = \frac{kgDW/m^2}{4} \cdot A \qquad \textit{Eq. 23}$$

Per year, the biomass production in Tjörröd could be expected to range between 8750 – 17500 kgDW/year. This is given as the total plant biomass per year, but to calculate the differences between foliage and wood their specific biomass was needed to be obtained.

The biomass percentage of foliage was assumed to be 5% and the wood 95% (Rosenquist 2021. Pers.comm.).

$$B_{foliage,wood} = B_{min,max} \cdot \%_{\text{foliage,wood}} \quad \textit{Eq. 24}$$

The biomass for wood was calculated to be 8313 – 16625 kgDW/year, and the foliage 438 – 875 kgDW/year.

The PFAS uptake in wooden biomass after four years is pretty straight forward as seen below.

$$X_{4\ years\ wood} = B_{wood} \cdot U_{wood} \cdot 4 \quad \textit{Eq. 25}$$

The foliage, on the other hand is not as easy. The foliage was only calculated for *Salix* fourth growth year as there is litterfall every year.

$$X_{4\ years\ foliage} = B_{Total\ biomass} \cdot \%_{foliage} \cdot U_{foliage} \cdot 2000 \quad \textit{Eq. 26}$$

The total PFAS uptake was finally obtained by summarising the uptake in wood and foliage.

Removal percentage

To evaluate how large the PFAS removal was in percentage the following equation was used

$$\text{x\%} = \frac{\text{x}}{\text{Q}_{\text{m,in}}} \cdot 100 \quad \textit{Eq. 27}$$

X% is the potential removal in % of incoming mass

$Q_{m,in}$ is the incoming mass flow of PFAS [g/year] and is calculated as

$$Q_{m,in} = PFAS\ concentration\ (ng/l) \cdot irrigation\ volume \quad \textit{Eq. 28}$$

The PFAS concentration is the concentration for the Tjörröd landfill, (e.g. 300 ng/l in the year 2020 for Sum PFAS-11).

The total irrigation volume for *Salix* is assumed to be 20 000 m^3 based on Swecos calculations, which is then converted into 20 000 000 L.

PFAS data from Filborna test

The data used for the calculation of PFAS uptake within *Salix* obtained from the Filborna landfill outside of Helsingborg in 2020.

Table 1 Data for PFAS uptake within Salix from Helsingborg for one growing period where the Salix had been irrigated with leachate from the landfill containing PFAS. After an event where foam had formed on a nearby leachate treatment pond, parts of the Salix plantation was covered with this foam as the wind dragged the foam from the treatment pond to the plantation. The foam contained high amounts of PFAS. Therefore, it is uncertain how much PFAS the Salix-plants were exposed to.

Substance	ng dw foliage	ng dw wood
6:2 FTS	86.67	48.94
PFOS	6.25	0.59
PFHxS	5.54	0.51
PFBS	1.39	0.35
PFOA	4.88	0.59
PFHxA	304.17	2.75
PFPeA	2354.17	11.45
PFBA	1266.67	1.89
SUMMA	4029.72	67.07

References

Larsson, E. 2020. Översilningsytors potential att rena lakvatten från PFAS. Masters thesis. Uppsala, Sweden: Ultuna, Sveriges Lantbruks Universitet.