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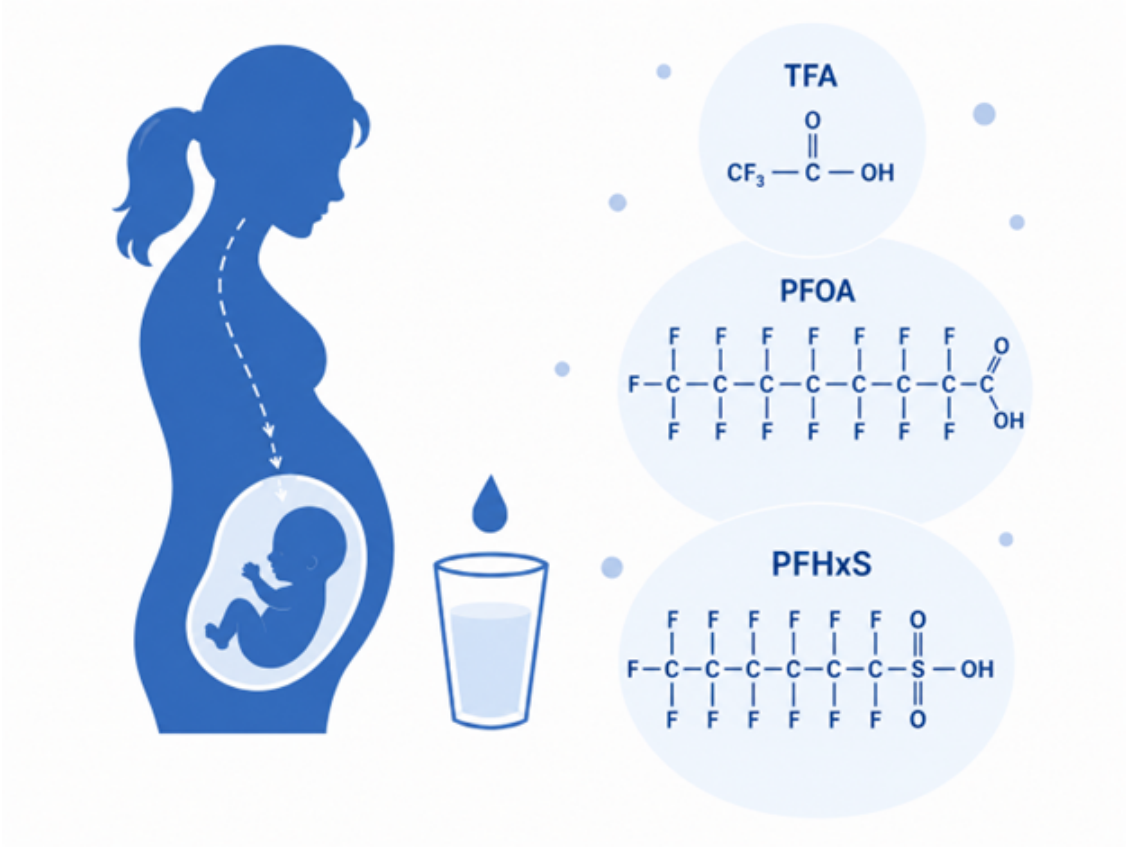

Early-Life PFAS Exposure in Sweden: A Study of Maternal Transfer and Prevention Through Drinking Water Treatment Technologies

Bachelor Thesis in Industrial and Material Science, B.Sc

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Cover: Illustration of maternal and fetal exposure to PFAS compounds via drinking water contamination.

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Abstract

PFAS are persistent and environmentally hazardous substances that occur in drinking water and may pose adverse health effects in humans, particularly in vulnerable populations such as fetuses and children. Exposure can occur through drinking water and be transferred from pregnant women to the fetus via the placenta or from mother to child via breastmilk, making early-life exposure difficult to avoid. Such exposure has been associated with effects on immune function, including reduced vaccine response, as well as impacts on child development. This study aims to investigate the potential health effects of PFAS exposure in fetuses and infants in Sweden, and to evaluate how current water treatment methods need to be developed to meet existing and future regulatory requirements. The study was conducted as a literature review based on scientific articles, complemented by interviews with experts in the field.

The findings indicate that PFAS exposure may affect children's health even at low concentrations, highlighting the importance of reducing exposure. At the same time, the current state of knowledge is characterized by significant uncertainties, particularly regarding short-chain and ultrashort-chain PFAS, where lack of data should not be interpreted as evidence of low risk. Current treatment methods can largely meet existing guideline values but have notable limitations, especially in that they primarily remove rather than degrade PFAS. While conventional techniques are generally effective for long-chain PFAS, they are less effective for short-chain compounds. Emerging technologies such as electrochemical oxidation show potential for destructive PFAS treatment, including degradation of shorter-chain compounds, although challenges related to energy demand, operational costs, and large-scale implementation remain significant.

Overall, the study shows that no optimal treatment method for PFAS currently exists, as the most promising techniques face high cost and large-scale implementation challenges. Future efforts should therefore focus on the development of more cost-effective methods to make advanced treatment technologies, such as electrochemical water treatment, cost-effective for widespread municipal use. Furthermore, the need for PFAS source control is critical, given that PFAS treatment is resource-intensive, and reducing exposure is particularly important for vulnerable populations such as pregnant women and children.

List of Acronyms

PFAS	Per- and polyfluoroalkyl substances
PFOA	Perfluorooctanoic Acid
PFOS	Perfluorooctanesulfonic Acid
PFHxS	Perfluorohexanesulfonic Acid
PFNA	Perfluorononanoic Acid
TFA	Trifluoroacetic Acid
PFPrA	Perfluoropropionic Acid
HSA	Human serum albumin
OAT	Organic Anion Transporter
EFSA	European Food Safety Authority
AFFF	Aqueous Film-Forming Foams
PFBS	Perfluorobutanesulfonic acid
BMDL10	Benchmark Dose Lower Confidence Limit for a 10% Response
TWI	Tolerable Weekly Intake
ECHA	European Chemicals Agency
IX	Ion exchange
OH-	Hydroxide Ion
Cl-	Chloride Ion
GAC	Granular Activated Carbon
UV	Ultraviolet
BDD	Boron-Doped Diamond
EAOP	Electrochemical Advanced Oxidation Process
PFBA	Perfluorobutanoic acid
PFPeA	Perfluoropentanoic Acid
NOM	Natural Organic Matter
DOHaD	Developmental Origins of Health and Disease

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1
Introduction

Per- and polyfluoroalkyl substances (PFAS) are characterized by a carbon chain in which hydrogen atoms are fully or partially replaced by fluorine atoms [1]. This carbon-fluorine bond is one of the strongest bonds in organic chemistry, contributing to the exceptional stability of these substances. As a result, PFAS exhibit properties such as high surface activity, increased chemical resistance, physiological inertness, and high thermal stability [2]. These characteristics have led to their widespread use in industrial and consumer applications, including paints, textiles, paper, food packaging, and firefighting foams. Due to their extensive use and environmental persistence, PFAS are now globally distributed and can be detected in soil, surface and groundwater and biota [3]. PFAS have been used since the 1950s and are now detected in nearly all environmental compartments [2]. Initially, the adverse effects of PFAS were largely unknown, and their advantageous properties enabled the production of cheaper and more efficient consumer and industrial products. However, it is now well established that PFAS are associated with environmental persistence, bioaccumulation, and potential adverse health effects. As a result, increasing efforts are being made to regulate and phase out the use of PFAS.

PFAS are commonly classified into two groups, long-chain and short-chain, based on the number of carbon atoms in their perfluorinated carbon chain [4]. Long-chain PFAS, such as PFOA and PFOS, also known as "legacy PFAS", were largely phased out and substituted by shorter-chain alternatives after their environmental and health risks were recognized, yet remain detectable in the environment due to their extreme persistence [5]. The phase out of these prominent legacy PFAS began in Sweden, following European Union Regulations, in the year 2008 [6]. Limits on these PFAS became more strict over time, with restrictions on PFOA compounds and their salts being adopted from year 2020. Despite these regulations, legacy PFAS persist in soil, water, food chains, and drinking water systems, meaning human exposure to long-chain PFAS has not stopped and remains an ongoing concern.

As the health risks of PFAS become increasingly recognized, the need to protect against exposure has grown. PFAS are of particular concern due to their persistence and potential adverse health effects even at very low concentrations, reflected in drinking water guideline values established in the ng/L range [7]. Pregnant women and their fetuses are especially vulnerable, as fetuses and infants are highly sensitive to chemical exposure due to undergoing rapid developmental phases that are highly affected by nutrition and toxins [8]. PFAS are transmitted from mother to child through the placenta during pregnancy and through breastfeeding after birth [9].

During pregnancy, the fetus develops rapidly and although the placenta provides some protection, PFAS can still cross this barrier. After birth, critical organs and systems such as the brain and immune system continue to develop, making infants highly sensitive to chemical interference. Additional factors, including higher intake of food and fluids relative to body weight and increased contact with contaminated dust or surfaces, further elevate exposure risks [8]. These characteristics make early-life stages partially susceptible to the persistent and bioaccumulative properties of PFAS. As a consequence, children can be exposed even before birth, without any possibility to control or influence their own exposure.

This study presents a literature study that focuses on pregnant women in Sweden, where PFAS contamination of drinking water has highlighted the national relevance of this issue. Although exposure to PFAS can occur through multiple pathways, including food and consumer products, these sources are subject to some extent to individual choice and behavioral modification. In contrast, exposure through drinking water is largely involuntary and difficult for individuals to control. Recent Swedish guidelines regulating PFAS concentrations in drinking water further emphasize the importance of understanding exposure pathways and associated health risks [10]. Drinking water will therefore be considered the primary exposure route in this study, as it represents a significant, well documented, and less controllable source of PFAS exposure in Sweden. The study is limited to the transfer of PFAS from mother to child during pregnancy and breastfeeding, with particular emphasis on direct biological transfer mechanisms via the placenta and breast milk. Exposure pathways occurring later in childhood will not be focused on in order to maintain a clear focus on early-life exposure.

Furthermore, the analysis will concentrate on a selection of well-studied long-chain PFAS, PFOS, PFOA, PFHxS and PFNA due to the availability of robust scientific data. Short-chain PFAS will be briefly discussed to provide comparative context, but will not constitute the main focus. While short-chain PFAS are increasingly used and relevant in current exposure scenarios, long-chain PFAS tend to bind more strongly to proteins than short-chain PFAS [11] and are therefore more suitable for in-depth analysis in this research given their more extensively documented properties. Given the importance of drinking water as a major and largely uncontrollable exposure pathway, effective water treatment methods are essential for reducing PFAS exposure. Several techniques have been developed to remove PFAS from contaminated water, including ion exchange, membrane filtration, carbon filtration, and foam fractionation, with granular activated carbon, a type of carbon filtration, being the most commonly used treatment in Sweden according to Philipp Wanner, Associate Professor in Contaminant Hydrogeology at Gothenburg University (personal communication via e-mail, 12 March 2026). While these methods have shown varying degrees of effectiveness, they also present limitations in terms of cost, efficiency, and applicability to different PFAS compounds. Emerging technologies are therefore being explored as potential future alternatives to improve PFAS removal. By examining the maternal transfer of PFAS and the developmental consequences of early-life exposure in Sweden, this study seeks to clarify potential health risks

and discuss strategies to reduce exposure during pregnancy and early childhood.

2
Theory

The background section will provide an overview of the information relevant to this study. It will introduce PFAS, their chemical properties and the difference between long- and short-chain. Furthermore, exposure, bioaccumulation and transfer within the body with particular focus on early-life are presented. This is followed by a presentation of the main exposure pathway in this study and the health-guidance vaules for PFAS in drinking water. Finally, commonly used and emerging treatment methods for drinking water are presented.

2.1 Overview of PFAS and Their Chemical Properties

PFOA and PFNA are perfluoroalkyl carboxylic acids, menaing they contain a carboxyl functional group ($-COOH$) [12], while PFOS and PFHxS are perfluoroalkane sulfonic acids containing a sulfonic functional group ($-SO_3H$) [13]. Sulfonic acids are relatively stronger and more chemically stable acids, which makes them bind more strongly to protein in the blood and tissues [14]. Chain length also affects bioaccumulative properties. PFNA (C9) has a longer fluorinated carbon chain than PFOA (C8), making it more hydrophobic and bioaccumulative. PFOS tends to bioaccumulate more than PFOA, due to both its chain length and sulfonate group. Despite having a shorter chain length, PFHxS (C6) still bioaccumulates notably because sulfonates bind strongly to proteins.

Carboxylates such as PFOA and PFNA tend to be more soluble in water which generally makes them travel more readily through groundwater than sulfonate PFAS with comparable carbon chain lengths [14]. However, as chain length increases, mobility typically decreases, and the compound is more likely to bind to sediments and organic material, which is seen with PFNA. Within the human body, all four compounds attach to blood proteins rather than accumulating in fatty tissues. PFOS and PFHxS are known to have particularly long biological half-lives. PFOA and PFNA can also remain in the body for several years, but are generally eliminated more quickly, especially compared to PFHxS [15].

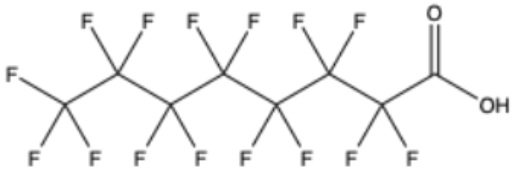

Figure 2.1: Chemical Structure of Perfluorooctanoic Acid (PFOA)

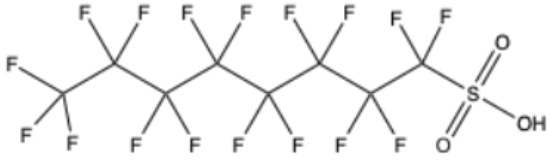

Figure 2.2: Chemical Structure of Perfluorooctanesulfonic Acid (PFOS)

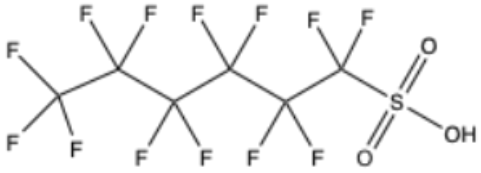

Figure 2.3: Chemical Structure of Perfluorohexane Sulfonic Acid (PFHxS)

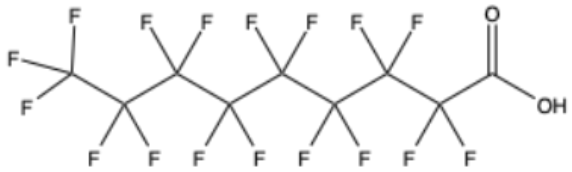

Figure 2.4: Chemical Structure of Perfluorononanoic Acid (PFNA)

2.1.1 Long-Chain and Short-Chain PFAS

PFAS can be broadly categorized as long-chain and short-chain compounds based on the length of their fluorinated carbon backbone, which influences their properties and how they behave in nature [16]. Carboxylic acids with eight or more carbon atoms, such as perfluorooctanoic acid (PFOA), and sulfonic acids with six or more carbon atoms, such as perfluorooctane sulfonic acid (PFOS), are considered long-chain PFAS. Similarly, short-chain PFAS have less than eight and six carbon atoms for carboxylic acids and sulfonic acids respectively. A subgroup of short-chain PFAS compounds, often referred to as ultra-short-chain PFAS, consists of two or three carbon atoms, such as trifluoroacetic acid (TFA) and perfluoropropionic acid (PFPrA) [17]. Generally, short-chain PFAS are more water-soluble and mobile in nature, enabling them to travel more easily in groundwater and drinking water systems than long-chain PFAS [18]. On the other hand, long-chain PFAS tend to bind more strongly to proteins, therefore accumulating at higher concentrations in living organisms and food webs [19]. Long-chain PFAS such as PFOA and PFOS often

show greater bioaccumulation and longer half lives in the body, in comparison to short-chain PFAS which are usually found in the body at much lower concentrations and are typically excreted more rapidly [19, 20].

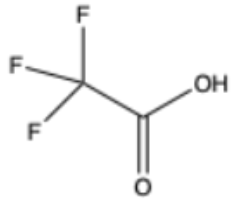

Figure 2.5: Chemical Structure of Trifluoroacetic acid (TFA)

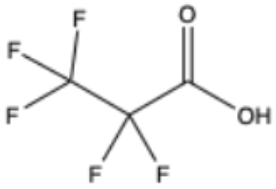

Figure 2.6: Chemical Structure of Perfluoropropionic Acid (PFPrA)

2.2 PFAS Persistence and Early-life Exposure

The structural characteristics of PFAS, including chain length and functional groups, strongly influence their environmental persistence and behavior in the human body [21, 15]. These properties contribute to their tendency to accumulate and remain in biological systems over extended periods [15]. As a result, PFAS exposure is of particular concern during sensitive life stages, where transfer between mother and child can occur.

2.2.1 Persistence and Bioaccumulation

PFAS is often referred to as “a forever chemical”, and the reason behind this is due to their high persistence in the environment, as complete degradation has rarely been observed in studies [15]. This persistence is largely attributed to the presence of high-energy carbon-fluorine bonds [22]. Although some PFAS can undergo partial degradation, they are often converted into other PFAS compounds rather than fully mineralized [15]. This makes it possible for PFAS to bioaccumulate in organisms and has been shown especially in protein-rich organs, tissue and body fluids such as liver, kidneys, lungs, brain, human serum and breast milk [15, 22]. PFAS have also been found to have a high affinity to Human serum albumin (HSA) which is a binding protein [23] that primarily can be found in the plasma fraction of the blood, such as serum or plasma in humans [15]. Continuous exposure may lead to increasing internal concentrations over time and the excretion of PFAS by urine, feces, menstruation and breastfeeding will slow down when PFAS are bound to proteins [15]. It has been found that 1 to 50 molecules of PFAS can be bound to one HSA and this leads to that more than 90% of PFAS are bound in the blood.

Since only the unbound PFAS can be excreted, strong protein binding reduces elimination efficiency [15]. In addition to strong protein binding, other mechanisms further reduce PFAS elimination and contribute to bioaccumulation. Enterohepatic circulation allows PFAS excreted via bile to be reabsorbed back into the bloodstream and liver. PFAS also undergo active reabsorption in the kidneys. Transport proteins can actively reabsorb PFAS from the pre-urine back into the bloodstream, thereby reducing urinary excretion. In humans, organic anion transporters such as OAT4 and possibly OAT1 and OAT3 have been identified as important contributors to the reabsorption process. This contributes to the long biological half-lives observed for several PFAS compounds.

Biological half-life refers to the time required for the concentration of a substance in the human body to decrease by 50 % due to elimination processes [15]. It has been found that structural differences among PFAS affects their half-life. PFAS with eight-carbon chains generally have longer biological half-lives than those with four-carbon chains. Furthermore, sulfonates generally have longer half-lives than carboxylates. The biological half-lives of different PFAS can vary between studies and this variation may reflect differences in study populations and exposure levels. For example, in one study the estimated half-life for PFOS and PFOA in human sera was ranging from 3.8 - 5.4 years [21], while another reported that the serum half-life for PFOS was 3.4 years and for PFOA it was 2.7 years [15]. For PFHxS the half-life in human serum 5.3 years and for PFNA 3.5 years. The variation in biological half-lifes among PFAS compounds can partly be explained by differences in their ability to bind to proteins like HSA.

2.2.2 Placental Transfer

During the pregnancy the placenta will be formed and developed until it is a complete organ, but it is only temporary and will be delivered after birth [24]. Proper placental development is essential for the pregnancy to be successful. It is a vital organ during pregnancy due to its ability to act as the lungs, kidneys, gut and liver of the fetus but also to give a protected environment [25]. It consists of, among other structures, the placental membrane and its function is to exchange gases, nutrients, fetal waste and hormones between the mother and fetus, without mixing the blood [24]. It also consists of the umbilical cord and its function is both to attach the fetus to the placenta and enable fetal blood circulation. One of the most vitals functions of the placenta is the transfer of nutrients that are essential for the fetus from the maternal circulation [22]. Although it acts as a selective barrier and can prevent certain biological macromolecules from crossing, studies have shown that some harmful substances, including PFAS can pass through the placenta barrier. The placenta is considered a potential target organ for PFAS due to its biological and molecular similarities to other target organs [15].

The ratio between maternal and cord serum concentrations is commonly used to estimate transplacental transfer efficiency [15]. In most studies, this ratio is below one and that indicates higher PFAS concentrations in maternal serum compared

to cord serum. It has been seen that the transfer efficiency varies between different PFAS compounds [21]. Table 2.1 presents PFAS concentrations in maternal and cord serum from women in Ronneby, Sweden [26]. Although the population in Ronneby represents an extreme exposure scenario, the study still provides valuable insight into transplacental PFAS transfer. The table shows that the cord serum has a lower PFAS concentration than the maternal serum, while still demonstrating that PFAS can be transferred to the fetus.

Table 2.1: PFAS concentration in maternal and cord serum from a study in Ronneby [26].

PFAS compound	Maternal serum during pregnancy (ng/ml)	Maternal serum at delivery (ng/ml)	Cord serum (ng/ml)
PFOS	18.2	15.8	8.54
PFOA	1.82	1.52	1.25
PFHxS	11.6	10.6	7.24
PFNA	0.42	0.36	0.19

The concentrations presented in table 2.1 are higher than those reported from places without known contamination. For populations with background exposure, one study reported serum PFAS levels of approximately 3 ng/ml for PFOA, 4 ng/ml for PFOS and 0.8 ng/ml for PFHxS [15].

Several studies suggest that carbon chain length, functional groups, isomers and molecular structure influence transplacental transfer efficiency [21, 15]. It has been suggested that there is a U-shaped relationship between fluoroalkyl chain length and transfer efficiency [15]. Meaning that long- and short chain PFAS show a higher transfer efficiency than mid-chain compounds. It has also been seen that branched isomers of PFOS and PFOA have a higher efficiency when crossing the placenta than linear isomers and for PFOS, the closer the branching point is to the sulfonate end, the placenta transfer is increased [15, 21]. Functional groups also appear to play a role [15]. PFAS containing carboxyl groups have been reported to exhibit higher transfer efficiency than sulfonate-containing PFAS. It has been reported that the transfer ratio of PFOA and PFHxS were the highest, approximately 0.67-0.68, while PFOS and PFNA showed a lower ratio of approximately 0.48-0.51 [26]. When summarizing this the ranking of transfer efficiency was PFOA $\rightarrow$ PFHxS $\rightarrow$ PFOS $\rightarrow$ PFNA [21, 26].

The mechanisms underlying placental transfer are not fully understood but are believed to involve both passive diffusion and active transport [15, 23]. Passive diffusion is considered the primary pathway, with the free PFAS in maternal serum being the key determinant of transfer efficiency [23]. When PFAS are bound to HSA, creating an HSA-PFAS binding complex, it is unlikely to pass through the placenta membrane [21]. Since only the unbound fraction of PFAS is assumed to cross the placenta, differences in plasma protein-binding affinity may significantly affect transfer efficiency [21, 15]. Lower binding affinity would increase the avail-

ability of free PFAS for placental passage. In addition to passive diffusion, active transport may contribute to PFAS placental transfer by transporters [23]. Transporter protein such as OAT4 have been suggested being involved in PFAS placental transport [15]. When PFAS interacts with these transporters, it may either facilitate or restrict transfer. But it have been reported that the stronger binding affinities between PFAS and OAT4 the more restricted the potential for transfer from the fetal side to the maternal side [23]. This leads to higher concentrations of PFAS in the placental tissue. The precise molecular interactions between PFAS and placental transporters remain incompletely understood.

2.2.3 Transfer via Breast Milk

In addition to transplacental transfer during pregnancy, infants may be continuously exposed to PFAS postnatally through breastfeeding [21]. The milk contains PFAS because it is transferred from the maternal serum [15]. Although PFAS concentrations in breast milk are typically one to two orders of magnitude lower than those in maternal serum, breastfeeding has been shown to significantly increase PFAS body burden in infants [21]. An estimated daily intake of PFAS in infants from breast milk has been reported to range from 7.1 - 40 ng per kg body weight in Stockholm and 5.2 - 25 ng per kg body weight in Gothenburg during the study [27] .

Studies have demonstrated that breastfeeding is a major contributor to infant PFAS exposure [15, 21]. It has been estimated that breastfeeding might contribute to 94% of PFOS exposure and 83 % of PFOA exposure for six-month-old infants [27]. It has been possible to see a positive correlation between maternal PFOS concentrations and PFOS concentration in breast milk, cord blood and neonatal blood and this further supports the role of lactational transfer as an important exposure pathway [21]. PFOS and PFOA concentrations measured in breast milk have frequently exceeded screening values established for children's drinking water intake and have been reported across different geographical regions. This supports the conclusion that breast milk constitutes a major source of PFAS exposure in infants.

Collected data indicates that breastfeeding duration influences infant PFAS concentrations [27]. Per month of exclusive breastfeeding the serum PFAS concentration can increase with as high as 30 % [28]. It has also been found that, in the first month after the infant was delivered, the prenatal exposure was lower than the exposure from exclusive breastfeeding. Consistent with these findings, infants who were breastfed had a 2-fold higher mean PFAS concentrations than infants that were not breastfed [27]. However, breastfeeding is still recommended as it has important health benefits for both mother and child [28, 29]. Infants may be particularly vulnerable due to their higher intake of food and liquids relative to body weight compared to adults, resulting in a higher exposure dose per kilogram body weight [15].

2.2.4 Vulnerability During Early-life Development

Pregnancy represents a critical window of exposure, as rapid developmental processes occur during this period and the timing of environmental exposures may result in long-term health effects [30]. During pregnancy, both mother and the fetus are especially sensitive to exposure due to dramatic physiological changes. The placenta plays a crucial role in protecting the fetus, which does not have a fully developed detoxification system. PFAS can cross the placental barrier and bioaccumulate in fetal organs, which may contribute to adverse health effects and birth outcomes [30, 22].

During pregnancy, fetal organs are not fully developed which leads to very limited resistance and detoxification capacity [22]. Because of this limited ability to eliminate xenobiotic compounds, prenatal exposure to PFAS may contribute to adverse health effects that can persist throughout childhood. It can also disturb hormonal processes that are required for normal growth and development for the fetus [15].

The fetus may also be indirectly affected if the uterine environment is negatively affected by maternal health conditions [15]. PFAS exposure during pregnancy has been suggested to be associated with pregnancy complications, including preeclampsia and gestational hypertension in the mother [15, 30]. These maternal conditions have been associated with adverse health outcomes in the offspring, including an increased risk of hypertension and cardiac remodeling later in life.

2.3 Exposure Pathways of PFAS

Approximately 100,000 children are born each year in Sweden [31], emphasizing the population-level importance of understanding PFAS exposure during pregnancy. PFAS are released into the environment through the production, use and disposal of PFAS-containing materials [3]. The exposure pathways that contribute to the intake of PFAS include drinking water, diet, the indoor environment and consumer products, with drinking water and diet representing the main exposure routes for the general population. An overview of common everyday PFAS exposure pathways is provided in Appendix B. Most dietary exposure comes from fish, shellfish and other animal products [32]. This is due to the high persistence of PFAS and their ability to accumulate in living organisms by binding to proteins in the blood and liver rather than to adipose tissue. Concentrations can therefore increase along the food chain and dietary habits may contribute to differences in exposure among individuals [33].

PFAS have historically been used in a wide range of consumer products due to their oil- and water-repellent properties [1]. For example, they have been applied in textiles, cosmetics, firefighting foams and food packaging materials [3]. PFAS may migrate from food packaging into food, thereby contributing to indirect oral exposure. The indoor environment also constitutes an important exposure source. Small particles from PFAS-treated materials may be released and accumulate in household dust, potentially leading to oral exposure via hand-to-mouth contact.

This is relevant for young children, who frequently spend time close to the floor and more often place their hands or objects in their mouths. In addition, some PFAS exist in forms that evaporate easily [34]. Exposure may therefore occur either via inhalation or by coming into contact with them later after they have settled in dust. In the case of textiles and cosmetics, dermal exposure through skin contact may also occur and even at low concentrations, it may lead to the gradual accumulation [2, 3]. The skin uptake needs further research to understand the mechanism of dermal absorption [35]. What can be highlighted from the research is that long-chain PFAS could be slowed down by the skin barrier, meanwhile short-chain PFAS can penetrate the skin barrier more easily.

2.3.1 Drinking Water as a Pathway of Great Importance

According to the Swedish Food Agency, individuals in Nordic countries typically consume approximately 1–2 L of drinking water per day, in addition to around 1–1.5 L obtained from food [36]. During pregnancy, water requirements increase and European Food Safety Authority (EFSA) recommends an additional intake of approximately 300 mL per day [37]. Because water is consumed daily and in relatively large volumes, it represents a continuous and significant source of PFAS exposure. Drinking water intake is difficult to avoid and may therefore contribute substantially to the total body burden. For consumer products, PFAS-free alternatives are increasingly available. For example food packaging without fluorinated coatings, PFAS-free textiles and cosmetics labeled as fluorine-free. However, such choices may be difficult to implement consistently without regulations [38].

The significance of drinking water as an exposure pathway varies depending on geographical location and in highly exposed areas, drinking water may account for the majority of total PFAS exposure [3]. In Sweden, PFAS exposure patterns have been strongly influenced by documented contamination events. One of the most well-known cases is the contamination in Ronneby [20, 39], where groundwater used for drinking water was affected by firefighting foams from a nearby military airfield. This resulted in exceptionally high PFAS concentrations and has been extensively studied as an example of drinking water as a dominant exposure pathway. Those who was heavily exposed to PFAS-contaminated drinking water showed higher PFAS concentrations in their bloodstream than the reference population. These findings could not be explained by other lifestyle factors, highlighting drinking water as the dominant exposure pathway in the studied population. Fire-fighting foams, aqueous film-forming foams (AFFF), have historically, in more places than Ronneby, been a major source of PFAS contamination [40], particularly near airports and military sites. Although the use use of AFFF is now being restricted and phased out in the European Union, legacy contamination remains an important source of human exposure, especially through drinking water.

Another example of where PFAS contamination has been associated with AFFF is in Uppsala, Sweden [33]. Drinking water contamination has contributed to elevated PFAS serum levels in both mothers and children. A study on first-time

mothers in Uppsala County, based on serum samples collected between 1996 and 2022, shows that PFAS exposure has decreased over time, resulting in fewer individuals exceeding the serum reference value established by the EFSA. However, 54% of the mothers sampled between 2018 and 2022 still exceeded this level. In addition to contamination hotspots, PFAS exposure is present at background levels in the general Swedish population, indicating widespread background exposure in the population. It is mainly the concentrations of long-chain PFAS, such as PFOS, PFOA and PFHxS, that have decreased over time [41]. Levels of PFNA and certain short-chain PFAS, such as PFBS, have increased instead [42]. These findings are particularly relevant in the context of early-life exposure, as reducing PFAS contamination in drinking water can lower maternal body burden and consequently reduce placental and breast milk transfer to the fetus.

2.4 Health-Based Guidance Values for PFAS in Drinking Water

Mattias Öberg, Associate Professor in Toxicology at Karolinska Institutet (personal communication, 23 March 2026) provided information on how EFSA based its 2020 risk assessment on the sum of four PFAS: PFOA, PFNA, PFHxS and PFOS [43]. The most critical endpoint was reduced antibody response to vaccination in children. EFSA identified a benchmark serum concentration (BMDL10) of 17.5 ng/mL for the sum of these four PFAS in 1-year-old children and from this derived a tolerable weekly intake (TWI) of 4.4 ng/kg body weight per week. EFSA further concluded that parts of the European population exceed this TWI, which is considered a health concern. Using Physiologically Based Pharmacokinetic modelling, EFSA estimated that the serum level in children corresponded to a long-term maternal exposure of 0.63 ng/kg body weight per day.

These health-based recommendations are taken into account in the development of legally binding regulations of municipal drinking water [44]. In Sweden, a legally binding limit of 4 ng/L for the sum of four PFAS (PFOA, PFNA, PFHxS and PFOS) entered into force on 1 January 2026. At the EU level, the revised Drinking Water Directive became applicable in Member States on 12 January 2026 and introduced two PFAS parameters in drinking water: 100 ng/L for the sum of 20 specified PFAS and 500 ng/L for total PFAS. In 2024, the European Commission established harmonised technical guidelines for PFAS monitoring in drinking water, including analytical methods, detection limits, and sampling frequencies across EU Member States [44]. Sweden has added the limitation of 4 ng/L for the sum of four PFAS and this restriction is substantially stricter than the EU parametric value of 100 ng/L for 20 PFAS, even if the values are not directly comparable. Sweden focuses on a smaller group of well-studied PFAS with well-established toxicological effects, while the EU limit covers a broader range of compounds including both long- and short-chain substances. Denmark has an even stricter national limit than Sweden of 2 ng/L for PFAS4 [45]. Other countries have also introduced or plan to introduce stricter national approaches for PFAS in drinking water in addition to the limits

set by the EU [46, 44]. Germany will introduce a limit of 20 ng/L for PFAS4 from 2028. European countries including Sweden, Denmark, Germany, the Netherlands and Norway are also actively involved in broader PFAS regulatory initiatives.

Emerging PFAS such as short-chain and ultrashort-chain compounds, including TFA are less clearly addressed in these regulations [47]. TFA is the smallest compound classified as a PFAS and is widely present in the environment, partly due to its formation as a degradation product of other fluorinated substances and its use in applications such as refrigerants. Unlike long-chain PFAS, TFA does not appear to bioaccumulate in the human body. Current assessments indicate that exposure to TFA through food and drinking water does not pose a health risk at present levels. However, significant knowledge gaps remain and ongoing evaluations by regulatory bodies such as the European Chemicals Agency (ECHA) and the European Food Safety Authority (EFSA) are needed to determine whether future regulatory limits may be required for TFA and other types of PFAS.

2.5 Current Drinking Water Purification Techniques

Drinking water purification in Sweden is not originally designed to clean the water of PFAS. This causes some technical challenges [48]. In Sweden, water producers try to limit PFAS in raw water by for example closing contaminated wells, mixing contaminated water with cleaner raw water or change location of the water source [49]. These measures are often not enough and more advanced methods for purification of the water of PFAS in Sweden is needed. Some of the most proven methods are activated carbon filtration, ion exchange and membrane filtration [50, 51]. Foam fractionation is another purification method that is used to a limited extent in Sweden. [52]. It is common that the methods are combined to clean drinking water of PFAS [53]. For example, activated carbon filtration and membrane filtration can be used together for a better result. In Sweden, the techniques are being studied and how to best optimaze the purification of drinking water. The process of purify drinking water from PFAS is overall expensive and the processes must adapt to the kind of PFAS is in the water.

These conventional treatment methods are primarily based on physical and physicochemical separation processes rather than degradation. Activated carbon filtration, membrane filtration, foam fractionation, and ion exchange differ in their underlying mechanisms but share the common function of transferring PFAS from the water phase to another medium, such as filter materials, concentrates, or foam [48, 50, 51, 52]. Although these approaches can achieve effective removal under suitable conditions, their performance depends on factors such as the length of the PFAS chain, the composition of the water, and the design of the system. Consequently, each method has specific advantages and limitations which are described in the following sections.

According to Philipp Wanner, associate professor in contaminant hydrogeology at Gothenburg University, these methods do not result in the destruction of PFAS, they generate secondary waste streams that require further treatment. Common management strategies include the regeneration of filter materials or high-temperature incineration of media contaminated with PFAS [54]. Thermal treatment is currently one of the primary approaches for achieving near complete degradation, as PFAS can be largely broken down at temperatures around 1000 °C. However, this process requires strict control to ensure efficient destruction and to prevent the formation or release of harmful by-products through flue gases, as incomplete combustion may lead to the formation of smaller PFAS compounds that can be emitted to the atmosphere.

According to Kristin Barkman, Water and Environmental Consultant at Sweco (personal communication, 23 March 2026), Sweden's capacity for treating PFAS contaminated waste remains highly limited. Currently, only one facility in Sweden, located in Kumla, is officially approved to handle such waste. Öberg further explains that the Kumla facility has faced both operational and environmental challenges, including concerns regarding potential leakage and the historical handling of PFAS containing waste. These illustrates some of the practical difficulties associated with managing and treating PFAS contaminated materials in Sweden.

2.5.1 Ion Exchange

Ion exchange (IX) is a treatment method in which PFAS ions in water are exchanged with less harmful ions such as OH- or Cl- for anion exchange resins [51]. The IX method uses synthetic resin beads that contain positively charged functional groups through which contaminated water is passed [55]. Many PFAS compounds, including PFOA and PFOS, are anionic, negatively charged surfactants that are strongly attracted to the positively charged sites on the resin surface. As a result, they are retained on the resin surface primarily through electrostatic interactions. In addition, hydrophobic interactions between the fluorinated carbon chains of PFAS and the resin's polymer backbone further enhance their adsorption. As a result of these combined mechanisms, IX resins are often highly efficient at PFAS removal.

IX demonstrates a strong performance in PFAS separation from water across a wide range of compounds [56]. IX resins have a high sorption capacity and can maintain effective removal over extended periods before breakthrough occurs, making them well suited for continuous treatments systems. The efficiency of the process depends on factors such as PFAS chain length and contact time, with longer contact times generally leading to better removal. This method can effectively capture both long-chain and short-chain PFAS, including those that are more mobile in water.

While highly effective, this method also has several important limitations. Its performance can be significantly influenced by water chemistry, particularly the presence of competing anions such as sulphate and nitrate, as well as natural organic mat-

ter, which can occupy active sites on the resin and reduce PFAS removal efficiency [56]. Over time, the resin becomes saturated and must be replaced or regenerated. Regenerations typically requires concentrated salt solutions, producing a secondary waste stream rich in PFAS that must be properly managed. In addition, the efficiency of the process depends on sufficient contact time. If the contact time is too short, breakthrough can occur more quickly, reducing treatment performance.

2.5.2 Activated Carbon Filtration

Activated carbon filtration is a widely applied method in water treatment to remove PFAS from drinking water and granular activated carbon (GAC) is one of the forms [57]. GAC is currently the most commonly used technology for PFAS removal in drinking water treatment plants in Sweden (Wanner, e-mail). In practice, GAC is typically implemented as fixed-bed filters, often retrofitted into existing treatment processes, which makes the method relatively easy to integrate into established infrastructure. The method is widely used due to its relatively low operational complexity and moderate energy requirements, making it suitable for large-scale drinking water treatment applications. The purification process is primarily based on adsorption, which means that dissolved substances in water bind to the surface of a solid material [57]. GAC is particularly effective because of its high specific surface area and extensive network of pores. This provides many adsorption sites where contaminants, such as PFAS, can bind to the carbon surface. However, the need for periodic replacement or regeneration of the carbon material represents an important operational consideration, particularly in full-scale applications.

When water contaminated with PFAS passes through a filter containing activated carbon, the process occurs in several steps [57]. First, PFAS molecules are transported from the water phase to the surface of the carbon particles through convection and diffusion. As the molecules approach the carbon surface, the water velocity decreases, and a thin layer of nearly stagnant water forms around the particle, often referred to as a boundary layer. When PFAS molecules reach this region, transport instead occurs through diffusion, which is driven by concentration differences. Because PFAS are continuously adsorbed onto the carbon surface, the concentration near the surface becomes lower than that in the surrounding water, creating a concentration gradient that drives PFAS molecules to move spontaneously toward the carbon surface through random molecular motion. The direct energy demand for this step is relatively low, although overall system performance depends on hydraulic design and contact time.

Once the molecules reach the surface of the carbon particles, the next step occurs, where PFAS diffuse into the pore structure of the carbon material, a process often referred to as intraparticle diffusion [57]. As adsorption occurs within the pores, additional molecules continue diffusing deeper into the material. As the molecules move through the pores, they can interact with the pore walls, and if these interactions are sufficiently strong, PFAS may become adsorbed onto the internal carbon surface. In practical applications, multiple GAC filters are sometimes operated in

series to improve removal efficiency and to better control breakthrough, which occurs when the adsorption capacity of the carbon becomes exhausted (Wanner, e-post).

The efficiency of adsorption is, among other things, influenced by the structure of the PFAS molecules [57]. In general, long-chain PFAS are adsorbed more effectively than short-chain PFAS, because longer fluorinated carbon chains have stronger hydrophobic properties and therefore a greater affinity for the carbon surface. Short-chain PFAS are more water-soluble and interact more weakly with activated carbon, making them more difficult to remove using conventional carbon filtration technologies. Under suitable conditions, GAC can achieve removal efficiencies from 92%-100% for many PFAS [58], particularly long-chain compounds such as PFOS and PFOA. However, short-chain PFAS, such as TFA, are removed significantly less efficiently, which represents a key limitation of the method (Wanner, e-mail).

2.5.3 Membrane Filtration

Membrane filtration is an effective method to reduce PFAS concentrations in drinking water [50, 59]. In this process, water is forced through a semi-permeable membrane under pressure. The membrane allows water molecules to pass while retaining dissolved contaminants such as PFAS. As a result, a permeate stream is produced, while rejected contaminants remain in the retentate stream [60, 61].

There are two main membrane technologies used for the removal of PFAS, nanofiltration and reverse osmosis [60]. Both technologies rely on pressure-driven membranes. The main differences are their separation efficiency and pore size, the microscopic openings in the membrane that determine which molecules can pass through the membrane [61]. In reverse osmosis, separation is mainly determined by a solution-diffusion mechanism. Nanofiltration is mainly based on size exclusion. Reverse osmosis membranes have a dense structure, a pore size of about 0.1-1 nm and require high operating pressures, usually between 10 and 100 bar. The method has been shown to achieve very high PFAS rejection, often above 99% under favourable conditions. Nanofiltration has a pore size about 0.5-2 nm and needs less pressure and energy. Generally, the pressure is between 5 and 40 bar. Nanofiltration can achieve high PFAS rejection, although performance varies depending on membrane properties, water chemistry, and the type of PFAS present.

PFAS separation in membrane filtration is governed by several mechanisms, including size exclusion, electrostatic repulsion, and solute–membrane interactions [59]. Size exclusion prevents larger molecules from passing through the membrane, while electrostatic repulsion occurs because both the membrane surface and many PFAS are negatively charged, reducing their transport through the membrane [60, 61]. In addition, physicochemical interactions such as hydrophobic interactions, hydrogen bonding, and dipole interactions can influence PFAS retention.

Membrane performance is further influenced by external factors such as water chemistry and operating conditions [50]. Parameters including pH, ionic strength, nat-

ural organic matter, pressure, temperature, and water flux can affect membrane properties and PFAS rejection [61]. Commercial nanofiltration and reverse osmosis membranes are typically thin-film composite membranes with a selective polyamide layer supported by additional layers for mechanical stability [59].

Despite their high removal efficiency, membrane processes do not destroy PFAS but instead concentrate them in a waste stream that requires further management [61]. Membrane fouling, caused by the accumulation of substances on the membrane surface or within its pores, can reduce both filtration rate and performance. In addition, these systems often require high energy input, particularly in reverse osmosis. Membrane filtration alone is therefore often insufficient for complete PFAS removal and is commonly combined with other treatment methods to improve overall performance.

2.5.4 Foam Fractionation

Foam fractionation is another method that can be used for the removal or enrichment of surface-active substances from water systems [62]. It can be used for removal of proteins from wastewater after food processing, removal of dye from wastewater after textile dyeing and removal of heavy metal ions from industrial wastewater. Studies have also shown that foam fractionation also works to remove PFAS from water systems. It is possible because PFAS have surface-active properties due to their hydrophobic fluorinated chains and hydrophilic functional groups [63]. But the method is not commonly implemented in municipal drinking water treatment in Sweden, as it is primarily suited for highly contaminated water streams rather than as a standard purification step [48, 52].

Foam fractionation is a separation method where air bubbles are introduced into a liquid [62]. This creates numerous air-liquid interfaces and due to thermodynamic interactions, hydrophobic and surface-active compounds, it can absorb into these bubble surfaces. A foam at the surface will start to form when the film surrounding the rising air bubble has a low enough energy to become stable. The stability of the foam is influenced by processes such as drainage of liquid films separating the bubbles, gas diffusion between bubbles and bubble coalescence. The foam is then collected above the foam-liquid interface and collapsed to produce a concentrated liquid stream. The remaining liquid becomes depleted of the surface-active compounds and can be discharged or continue to other processes.

Foam fractionation has been shown to be a relatively simple and cost-effective method for PFAS removal [63]. However, several limitations remain. The method is more effective for long-chain PFAS, while the removal of short-chain compounds are limited [64]. To enhance the removal of short-chain PFAS, the addition of co-factors may be applied [63]. This could influence both operational conditions and environmental considerations. In addition, most studies on foam fractionation have been conducted at relatively high PFAS concentrations. The concentrations used are higher than the concentration in drinking water. The method also does not degrade PFAS but instead concentrates them in a foam phase, which requires further

treatment. Furthermore, the performance of the method is influenced by the water composition and operational conditions. This makes it more difficult to achieve stable and reliable performance in full-scale drinking water treatment system. These factors contribute to why foam fractionation is not used as a standard method for municipal drinking water purification.

2.6 Emerging Techniques for Water Treatment

In addition to conventional treatment methods, a range of emerging technologies are being developed to address the limitations of current systems. These approaches aim to improve removal efficiency (especially for short-chain PFAS), reduce energy consumption, and destroy PFAS rather than just concentrate them (Wanner, e-post). Some of the most promising methods are plasma, biobased, photochemical and electrochemical water treatment.

Plasma-based treatment does not refer to a single technology but rather to a group of advanced oxidation-like processes in which different plasma generation techniques are used to produce reactive species capable of degrading PFAS. The process is based on the generation of highly reactive species, such as electrons, radicals, and ions, which can break the strong carbon–fluorine bonds characteristic of PFAS compounds [65, 66, 67]. Through these reactions, PFAS molecules can undergo degradation via mechanisms such as electron transfer, bond cleavage, and defluorination. Biological treatment involves enzymes and microorganisms capable of cleaving the strong carbon-fluorine bond and removing fluorine atoms by oxygenation (oxidation) or gaining electrons (reduction), thus allowing further enzymatic breakdown of the remaining molecular structure [68].

Photochemical degradation of PFAS uses light, usually UV, in combination with photocatalysts to break the strong carbon–fluorine bonds in PFAS [69]. The method works by activating a catalyst, often a metal oxide, with UV or visible light [70]. When the catalyst is irradiated, electrons are excited, leading to the formation of electron–hole pairs that drive subsequent chemical reactions. Dissolved oxygen is reduced by taking electrons to form superoxide radicals, while water is oxidized by the positive holes to produce hydroxyl radicals. These reactive species can degrade organic contaminants, such as PFAS, by attacking their chemical bonds. PFAS are then broken down stepwise, with long-chain compounds first degraded to short-chain intermediates, which are subsequently mineralized into end products such as carbon dioxide, fluoride, and hydrogen fluoride.

Electrochemical treatment focuses on the destructive degradation of these highly persistent compounds through electrically driven reactions. According to Salman Farissi, postdoctoral researcher at Department of Mechanical Engineering at Chalmers University of Technology (personal communication, 31 March 2026), the process involves applying the electric current between submerged electrodes, typically consisting of an anode and a cathode, within contaminated water. To ensure sufficient electrical conductivity, a supporting electrolyte is often added to the water sample.

At the anode, strong oxidative species such as hydroxyl radicals are generated, which react with PFAS molecules and break them down into smaller constituents, including carbon dioxide and fluoride ions. The efficiency of the process is strongly influenced by the choice of electrode material, with advanced materials such as coated titanium or boron-doped diamond offering enhanced reactivity and stability. Due to its ability to transform PFAS at the molecular level, electrochemical treatment is considered a promising approach, although it is still under development and faces challenges related to energy requirements, material costs, and large-scale implementation.

3
Methods

This study is conducted as a literature review. This method is used to collect and summarize existing scientific knowledge on the transfer of PFAS from mother to child, the potential health effects associated with early-life exposure, drinking water as an important exposure route, water purification methods, risk assessment and regulatory limits. A literature based approach was chosen to provide a structured overview of current research and to identify knowledge gaps within the field. Although PFAS exposure occurs through multiple pathways, including diet, consumer products, and indoor environments, drinking water is considered a particularly important exposure route due to its continuous consumption and limited individual control. This study therefore focuses on drinking water as a key pathway, while acknowledging that total PFAS exposure is influenced by multiple sources. The geographical focus is on Sweden, particulary in relation to water treatment methods, drinking water quality and regulatory limits. International studies are included to provide a broader scientific context. PFAS research is continuously developing with new data being generated by researchers and authorities. The conclusions of this study therefore, reflect the current state of knowledge at the time of writing and may change in the future as new knowledge emerges.

The aim of the project is considered achieved if the literature review provides a clear and structured overview of how PFAS are transferred from mother to child and the potential developmental effects of early-life exposure. These are examined in relation to drinking water as an important exposure source, including regulatory limits and water treatment methods in Sweden, based on relevant scientific studies.

3.1 Literature Search and Selection

To collect scientific knowledge, articles were collected through searches in established databases such as PubMed, Google Scholar, Web of Science as well as through databases on Chalmers University of Technology library. A limitation of the study is that not all relevant studies may have been included, as the literature search was not fully systematic. Multiple databases were used to reduce the risk of selection bias. To find suitable articles on the topic, relevant keywords such as "PFAS", "per- and polyfluoroalkyl substances," "exposure", "early life exposure", "placental transfer", "maternal transfer", "breastfeeding", "water treatment", "regulatory limits" and "health effects" were used. Synonyms of keywords were also used to ensure a

thoroughgoing search. Titles and abstracts were initially read first to understand the relevance before full-text articles were reviewed. Studies not relevant to the research aim were excluded. Peer-reviewed articles were prioritized to ensure the quality of the information. The selected studies were reviewed and compared to identify common findings, differences between studies and knowledge gaps. Non-peer-reviewed sources, such as reports and webpages from authorities, universities, and relevant organizations, were included when they provided useful information. Literature published mainly from 2018 onward was prioritized. Publications before the year 2018 were included when they provided foundational information, historical context or established methods and definitions relevant to the study. Authority and regulatory sources were selected based on the most recent available versions. In total, 82 articles were included in the thesis.

3.2 Interviews

In addition to the literature review, relevant experts were contacted to gain contextual understanding and clarify specific aspects of PFAS exposure, transfer mechanisms and water treatment strategies. Five interviews were conducted with researchers and professionals in Sweden working within relevant fields. The selection of academic experts was based on previously reviewed literature and the authors of key studies, as well as recommendations.

The interviews were carried out through either digital meeting platforms or e-mails. The interview sessions on digital meeting platforms were conducted with one or two participants from the project group. The questions were adapted to each interviewee depending on their specific expertise, for example in toxicology, water treatment or transfer from mother to child. The questions were distributed in advance. A compilation of the interview questions can be found in the appendix. Notes and recordings were used to document the interviews and the information was then analyzed to identify what should be included in the study. All participants were informed about the purpose of the study and agreed that their names or companies could be mentioned in the thesis.

3.3 Experimental Demonstration

To complement the literature review, a laboratory visit was conducted with Salman Farissi. During this session, the experimental setup and procedures of a method that removes PFAS from water were presented and explained. The purpose of this observation was to provide a deeper understanding. No primary data were collected to be a part of this study. The observation only served as a complementary learning activity to support the interpretation of the literature.

3.4 Use of Artificial Intelligence

AI has been used as a supportive instrument throughout the project. It has been used to verify whether images are permitted for use in the thesis, to make the front-page image, to assist with LaTeX commands, to transcribe the conducted interviews and to summarize scientific sources at the beginning of the work to gain an initial overview of research papers more efficiently. Its use has been approached with caution. AI has also been used for language corrections and translating text from Swedish to English. All translations were carefully reviewed to ensure accuracy.

4

Results and Discussion

This chapter examines key aspects of PFAS, including treatment technologies, health effects, and regulatory framework. In addition to literature-based analysis, the chapter also includes a laboratory demonstration of electrochemical oxidation, providing practical insight into PFAS treatment processes. First, the demonstration is presented, followed by a discussion of the challenges associated with conventional treatment methods and an overview of emerging alternative technologies. This is followed by an analysis of health effects associated with early-life exposure to PFAS, and the section concludes with a discussion of current guideline values and regulatory challenges.

4.1 Laboratory Demonstration of Electrochemical Oxidation

The authors of this study had the opportunity to observe a laboratory demonstration at the Department of Industrial and Materials Science at Chalmers University of Technology. The session provided insight into the practical implementation of electrochemical oxidation for the treatment of PFAS-contaminated water. This demonstration illustrated the fundamental principle of electrochemical treatment, where contaminated water is exposed to an applied electric current through electrodes. The system consisted of an anode and a cathode placed in a solution. The anode was a titanium plate coated with catalytic metals, while the cathode was made of stainless steel. The coated titanium electrode seen in Figure 4.1 is specifically designed to enhance electrochemical activity compared to uncoated titanium, thereby improving PFAS degradation efficiency.

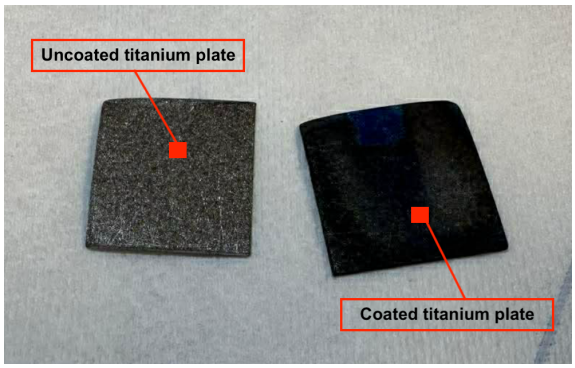

Figure 4.1: Uncoated titanium plate (left) and coated titanium plate (right).

In the experimental setup shown in Figure 4.2, both electrodes were immersed in PFAS-containing water and a low current was applied. To ensure sufficient conductivity of the solution, sodium sulfate was added as a supporting electrolyte. A magnetic stirrer was used to maintain continuous mixing during the operation. Under these conditions, PFAS compounds are transported to the anode surface, where they undergo oxidative degradation. This process can break down PFAS molecules into smaller products, such as carbon dioxide and fluoride ions, rather than transferring them to another phase, as is the case with adsorption-based methods.

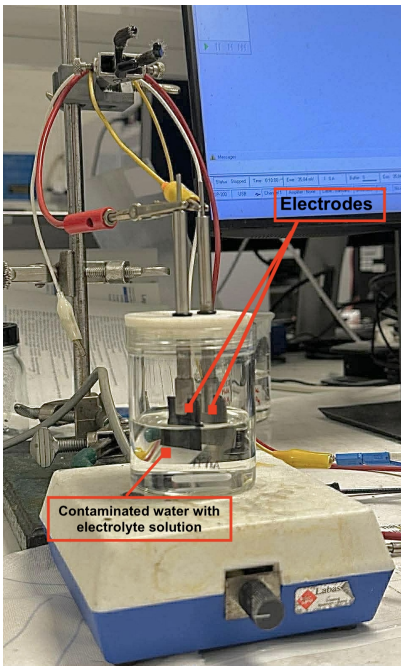

Figure 4.2: Laboratory setup used for electrochemical oxidation of PFAS-contaminated water.

In literature, boron-doped diamond (BDD) electrodes are presented as a more advanced option, which can be seen in Figure 4.3 (left). BDD electrodes offer higher stability, longer operational lifetimes and stronger oxidative capabilities. However, these advantages come at a significantly higher cost, which currently limits their

large-scale application. According to Salman Farissi, the process cost for BDD-electrochemical treatment of long-chain PFAS is around $10 - 15$ KWh $\cdot$ m^{-3} with above 99% removal efficiency. Based on the average electricity price in the SE3 electricity zone in Sweden, which includes both Stockholm and Gothenburg [71], the cost will be $5.5 - 7.7$ SEK $\cdot$ m^{-3}. Furthermore, Salman empathizes that the cost for treatment of short-chain PFAS could be lower, but so will also the removal efficiency (60-70%). To address this limitation, mixed metal oxide electrodes such as catalytic metal oxide coated titanium, shown in Figure 4.3 (right), can bring down the process and production costs that are associated with the BDD-electrochemical process while also showing improved treatment of short-chain PFAS. This is possible due to the low current requirement for the new material to degrade the PFAS, making the electrochemical process affordable overall. The newly synthesized electrode material is reusable more than ten times, making it durable and stable. The large-scale application requires pilot scale studies, which are going on at the lab. This demonstration highlighted the importance of the choice of electrode material.

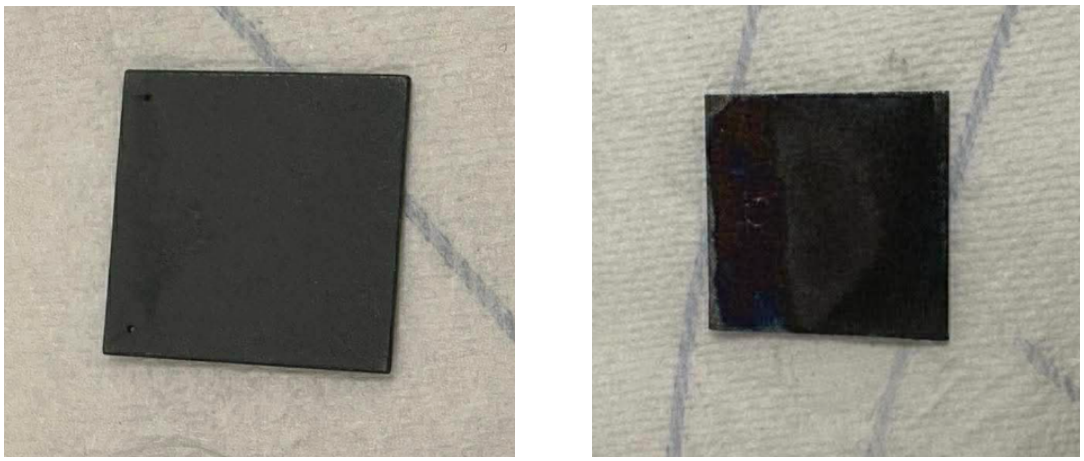

Figure 4.3: Boron-doped diamond (BDD) plate after multiple treatment cycles (left). Coated titanium electrode after multiple treatment cycles (right).

4.2 Limitations for Conventional PFAS Treatments and Emerging Alternatives

Despite the widespread use of conventional treatment methods such as ion exchange, activated carbon filtration and membrane filtration, there are several limitations in the removal of PFAS from drinking water. As drinking water can constitute an important exposure pathway, particularly for women of reproductive age [37], improving treatment efficiency may also contribute to reducing maternal body burdens and subsequent transfer to the fetus and infant. As a result, there is a growing need for alternative treatment strategies that not only remove PFAS from water but also enable their degradation. In response to these limitations, several emerging treatment technologies are currently being investigated (Wanner, e-mail).

4.2.1 Conventional Treatments

The conventional methods primarily rely on adsorption and this leads to that the PFAS are not destroyed but rather collected [68]. As a result, additional handling and disposal of contaminated materials are required and this remains a significant challenge (Wanner, e-mail). There is a risk that the PFAS-containing waste is merely transferred and potentially reintroduced into another environmental compartment instead of being removed and destroyed. In Sweden, this challenge is further emphasized by the limited treatment capacity, as only one approved facility is currently available for PFAS contaminated waste treatment (Barkman, personal communication), creating a bottleneck in safe disposal. Furthermore, reliance on a single treatment facility may increase the vulnerability of the national waste management system and highlights the need for expanded treatment capacity, robust monitoring systems, and improved destruction technologies. In addition, the effectiveness of these methods can vary depending on PFAS properties, such as chain length.

Another limitation for conventional PFAS treatment methods is that the effectiveness varies depending on the chain length. With an increasing use and environmental release of short-chain there is a growing need for treatment methods capable of removing both long- and short-chain compounds. Conventional methods such as GAC and foam fractionation show lower removal efficiency when it comes to short-chain PFAS. In the case of foam fractionation, the addition of co-surfactants is often required to improve the removal efficiency [62]. This leads to higher operational costs and may introduce potential environmental concerns. Whereas ion exchange and membrane filtration perform better [55, 59]. However, these methods are associated with other limitations such as sensitivity to water composition, high energy input and limited lifetime, highlighting that no single conventional method provides a complete solution. The increasing replacement of long-chain PFAS with short-chain alternatives further complicates treatment, as these compounds are generally more mobile and less responsive to conventional removal methods. This creates a mismatch between evolving contamination patterns and the capabilities of existing technologies.

In addition, the performance of both foam fractionation and ion exchange can be significantly influenced by the water composition [63, 56]. The presence of natural organic matter, competing ions such as sulphate and nitrate and other dissolved substances can interfere with treatment processes and this reduces the overall removal performance and introduces variability between different water sources. As a result, methods that perform well under controlled conditions may show lower and less predictable efficiency in real-world applications, highlighting a limitation in their robustness. For ion exchange, these effects can lead to that the resins must be replaced or regenerated, which in turn produces a secondary waste stream rich with PFAS [55]. Similar challenges, that material has a limited lifetime are observed for membrane filtration and GAC [60, 57]. Membranes can become fouled over time, while activated carbon gradually loses adsorption capacity and requires replacement or regeneration. At high contamination levels, this may substantially increase operational costs, energy demand and material consumption. In addition, these processes

generate PFAS containing residuals that require further management. The handling of such secondary waste introduces a risk of PFAS being redistributed into other environmental compartments, rather than being permanently removed. This highlights a fundamental limitation of conventional treatment methods, as they require continuous management and do not provide long-term solutions to PFAS contaminations.

These limitations highlight the need for alternative treatment approaches that not only remove PFAS from water but also enable their effective degradation. As a result, several emerging technologies are currently being investigated as potential solutions to overcome the shortcomings of conventional methods.

4.2.2 Electrochemical Treatment

Electrochemical PFAS treatment shows strong potential, but several key challenges limit its scalability and application in the real world. Most studies are conducted under simplified laboratory conditions, whereas real contaminated wastewater and groundwater typically contain competing ions and organic matter [72]. Low conductivity in water reduces ion transport and increases electrical resistance, and competing species can interfere with PFAS adsorption and oxidation, reducing treatment performance. The electrodes used also impact treatment efficiency. High-performing materials like BDD are effective but expensive, while cheaper alternatives can have lower activity or degrade over time. In addition, electrode fouling and surface degradation during operation can further reduce performance and require maintenance or replacement. Electrode properties strongly influence PFAS adsorption and reaction rates that ideally would be optimized for different water conditions.

Significant technical and economic constraints remain unsolved. Electrochemical treatment often requires substantial energy input, which can lead to high operational costs, particularly at low PFAS concentrations, and applying laboratory set-ups to real world water treatment systems is still challenging [72]. Furthermore, limitations in mass transfer can reduce treatment efficiency in larger-scale systems by slowing down the transport of PFAS molecules through the water to the reactive electrode surfaces where degradation occurs [73]. As a result, the overall reaction rate may be controlled not by the electrode's reactivity, but by how quickly PFAS compounds can reach the surface, an effect that becomes more pronounced in systems with larger volumes, insufficient mixing, or longer transport distances. In addition, there is a lack of comprehensive data on long-term performance and economic feasibility.

Despite these challenges, electrochemical treatment offers a significant advantage over many conventional methods, as it enables the degradation of PFAS rather than merely transferring contaminants to another phase [72]. As a result, considerable efforts have been made to overcome these limitations and bring the technology closer to practical application. This approach, often referred to as electrochemical advanced oxidation processes (EAOP), is now gradually being implemented and

commercialized for real wastewater applications by several companies around the world [74]. One example being the German company PFASuiki that uses EAOP for PFAS destruction [75]. While different companies use different electrode materials, catalysts, and reactor designs, the underlying concept is largely the same: applying electrochemical reactions to oxidize and degrade PFAS compounds in contaminated water [74]. The further development and spread of this can potentially provide a more effective and sustainable solution for PFAS removal. Its development and implementation demonstrate strong potential for addressing the growing public health concerns associated with PFAS contamination, particularly for vulnerable populations such as pregnant women and individuals exposed during early stages of life.

4.2.3 Plasma Water Treatment

Experimental studies have demonstrated that plasma treatment can achieve high degradation efficiencies, up to 90%, under controlled laboratory conditions, particularly for long-chain PFAS [65, 66]. However, these results should be interpreted with caution, as treatment efficiency appears to decline substantially in complex water matrices such as drinking water and wastewater, where competing substances limit the interaction between reactive species and PFAS. This suggest that laboratory performance may not be directly transferable to real world applications.

Despite its promising degradation potential, plasma-based treatment also faces several important limitations. One major challenge is that PFAS are not always fully mineralized, meaning that they are not completely broken down into harmless end products. Instead, transformation products and shorter-chain PFAS may be formed during treatment, which remain persistent and may still pose environmental or health concerns [65, 66]. In the context of prevention strategies, this is particularly relevant, as incomplete destruction may reduce contamination levels without eliminating exposure risks entirely. Furthermore, the process is energy-intensive and currently remains largely limited to laboratory scale applications. Consequently, while plasma-based technologies represent a promising future option, substantial technical development and full-scale validation are required before they can be considered a reliable large-scale solution.

4.2.4 Biological Treatment

Biological treatment offers several advantages as a potential method for PFAS removal [68]. It is considered an environmentally friendly and cost-effective method that can be applied in situ using naturally occurring biological agents. This reduces the need for energy-intensive processes and minimizes disturbance to the surrounding environment. However, despite these advantages, significant limitations remain that restricts its broader applicability. Biological treatment is currently limited to laboratory-scale applications. Experimental studies have been able to show that the removal efficiencies range from approximately 32% to 75%, these results are

depending on the microorganism used and the specific PFAS compound. This variation in the removal efficiency indicates that the performance of the method is not consistent, making it unreliable.

The removal efficiency is also influenced by the duration of the treatment process, however, the relationship is not always linear [68]. This implies that extending the treatment time does not necessarily result in a higher efficiency rate. The slow degradation in biological treatment further reduces its overall effectiveness. This method has primarily been investigated for compounds such as PFOA and PFOS, but complete degradation is rarely achieved due to high chemical stability of PFAS, which may result in the formation of intermediate products. Moreover, it has also been seen that when using biological treatment in water with high concentrations the biotransformation rate is reduced due to the chemical toxicity. This suggest that the method becomes even less effective under conditions where contamination levels are high. Considering that biological treatment already exhibits lower efficiency rate compared to other methods, its performance may be further reduced in such environments. Finally, the available information regarding the biodegredation of PFAS using bacteria and fungi is limited, contributing to the uncertainty surronding the method. Consequently, its effectiveness, feasibility and applicability at larger scales remain uncertain.

4.2.5 Photochemical Treatment

Photochemical methods with photocatalysis have demonstrated high removal efficiencies under controlled conditions [69]. For example, metal oxide–based photocatalysts have achieved a degradation of 98% of PFOA and a degradation of 95,5% of PFOS, highlighting the strong oxidative potential of these systems. However, despite these promising results, complete mineralization is often not achieved. Instead, long-chain PFAS are frequently transformed into short-chain intermediates, such as PFBA and PFPeA, which are more mobile, persistent, and resistant to further degradation. Such changes in PFAS composition reduce overall treatment effectiveness and increase the risk of environmental dispersion, particularly in groundwater systems [69].

Beyond incomplete degradation, several practical and technical constraints limit large-scale application. The efficiency of photochemical processes is highly dependent on water chemistry, as dissolved oxygen, natural organic matter (NOM), and inorganic ions such as nitrates can act as radical scavengers, reducing treatment performance [70]. Consequently, pre-treatment steps, including removal of organic matter or adjustment of chemical conditions, are often required, increasing both complexity and cost. Although advanced systems, such as UV/persulfate and photo-Fenton processes, can enhance the degradation of PFAS and improve mineralization rates, they may also lead to the formation of fluorinated transformation products [69]. These include short-chain perfluoroalkyl acids and other intermediate com-

pounds, some of which may be more persistent, mobile or potentially toxic, particularly when formed through secondary reactions within the water matrix.

Scalability represents an additional major challenge for photochemical treatment. Many photocatalysts require UV light due to their wide bandgaps, meaning that only high-energy radiation can activate the catalyst [70]. This dependence on UV irradiation increases energy demand and limits the feasibility of natural sunlight in large-scale applications. In addition, photocatalysts may gradually lose activity as a result of surface fouling and accumulation of reaction intermediates, which block active sites and alter the catalyst structure. As a result, periodic regeneration is necessary, often involving additional chemical or energy input, further increasing operational costs and system complexity [69]. In general, photochemical methods offer strong oxidative capacity for the degradation of PFAS, but remain largely limited to laboratory and pilot scales due to challenges related to by-product formation, energy demand, and catalyst stability [70].

4.2.6 Comparative Assessment of Emerging Technologies

All emerging PFAS treatment technologies demonstrate strong degradation potential under controlled laboratory conditions, but differ significantly in efficiency, robustness, and scalability. Biological treatment is the most environmentally sustainable and energy-efficient option, but its low degradation rates, variability, and limited maturity make it less suitable for near-term large-scale implementation [68]. Photochemical methods demonstrate very high degradation efficiencies under optimized conditions but are strongly constrained by water matrix sensitivity, light dependency and catalyst deactivation [69]. Plasma-based treatment can achieve rapid and high removal efficiencies, but its high energy consumption and incomplete mineralization issues reduce its near-term applicability [65, 66, 67].

Electrochemical treatment appears to be the most balanced technology, combining relatively high and controllable degradation efficiency with greater robustness in complex water matrices. In addition, it has a clearer pathway toward integration into existing water treatment infrastructure compared to other emerging methods. However, challenges related to energy demand, electrode stability, and mass transfer need to be addressed [73]. Overall, electrochemical treatment represents the most promising approach for future development, while plasma and photochemical methods may be suitable for specific applications, and biobased treatment remains a long-term sustainable option requiring further fundamental research.

4.2.7 Common Issues and Future Research Needs

Despite their differences, all emerging PFAS treatment technologies share several common limitations. Most methods are currently restricted to laboratory or pilot-scale studies, and their performance under real-world conditions remains insufficiently validated [65, 68, 69, 72]. In complex water matrices, competing ions and natural organic matter can significantly reduce treatment efficiency across all tech-

nologies. In addition, many processes suffer from incomplete mineralization, leading to the formation of shorter-chain PFAS or other transformation products that may remain persistent and environmentally mobile [66, 68, 69]. Energy demand is also a key constraint for several methods, particularly electrochemical, plasma, and photochemical systems, which limits economic feasibility at scale [67, 70, 72]. Nevertheless, these technologies have demonstrated the technical ability to degrade PFAS into smaller and potentially benign compounds, unlike conventional methods that primarily transfer PFAS into concentrated waste streams requiring further handling and disposal. Karthik Laxman Kunjali, Co-Founder and CEO of Stockholm Water Technology (personal communication via e-mail, 20 March 2026) further emphasizes that PFAS treatment should be viewed within the broader context of integrated water treatment systems, since contaminated water often contains mixtures of pollutants, including heavy metals, salts, bacteria, and organic compounds. According to Kunjali, future breakthroughs will therefore likely depend on technologies capable of simultaneously address multiple contaminants while maintaining cost-effective operation and minimizing pretreatment requirements. Future research should therefore focus on improving energy efficiency, enhancing treatment robustness, reducing by-product formation, simplifying integration into existing treatment systems, and advancing scale-up from laboratory to full-scale water treatment applications.

Table 4.1: Emerging technologies [65, 66, 67, 68, 69, 70, 71, 72, 73, 76], (Farissi, e-mail)

Method	Biological Treatment	Photochemical Treatment	Plasma Treatment	Electrochemical Treatment
Principle	Uses enzymes and microorganisms to break fluorine–carbon bonds	Uses light-activated catalysts to generate reactive radicals that break carbon–fluorine bonds	Uses plasma to generate highly reactive species (electrons, radicals and ions) that degrade PFAS through bond cleavage and defluorination	Application of oxidation–reduction reactions in water to break down PFAS molecules into smaller, less harmful products.
Removal efficiency	Between 32%–75%	Between 95.5%–98%	Laboratory conditions: up to >90%, reduced in complex water matrices	High in laboratory conditions, reduced in complex water matrices
PFAS Target	PFOA and PFOS	PFOA and PFOS	PFOA and PFOS	Both long- and short-chain PFAS
Advantages	Environmentally friendly, cost-effective and can be applied in situ	Environmentally friendly, highly effective for long-chain PFAS, flexible and tunable, can be combined with other treatments and has potential for in situ application	Capable of degrading PFAS rather than separating them, effective for strong C–F bond cleavage, no need for chemical additives	Destroys PFAS, potential for full mineralization, tunable through electrode/material design
Limitations	Slow and incomplete degradation, low and variable efficiency, long treatment time and formation of intermediates	Energy-intensive, incomplete mineralization, limited effectiveness for short-chain PFAS, sensitive to water matrix, catalyst deactivation and high operational costs	Energy-intensive, incomplete mineralization, formation of transformation products, reduced efficiency in complex water matrices, limited performance for short-chain PFAS	High energy demand, sensitive to water matrix (ions, organic matter), expensive, requires conductive water (electrolyte addition)
Approximate cost (for long-chain)	Under development – cost not yet established	6.68 – 7.2 SEK $\cdot$ m^{-3}	9.2 SEK $\cdot$ m^{-3}	5.5 – 7.7 SEK $\cdot$ m^{-3}
Development stage	Laboratory scale	Laboratory and pilot scale	Laboratory scale	Laboratory to early pilot scale

In addition to the technical challenges associated with PFAS removal, there are difficulties in establishing health-based guideline values and regulatory limits.

4.3 Varying Guideline Values and Regulatory Limits

A key challenge in PFAS risk management is the difference between health-based guidance values and regulatory limits in drinking water. EFSA has established a tolerable weekly intake of 4.4 ng/kg body weight per week [43]. However, the legally binding limits in the EU drinking water directive (100 ng/L for PFAS20) are much higher than what would be expected based only on EFSA´s health-based assessment. This gap reflects the complexity of translating scientific evidence into policy. Regulatory limits are not based exclusively on toxicological data but also take into account technical feasibility, analytical limitations and economic considerations. As a result, there may be a gap between what is considered biologically safe and what is achievable in water treatment systems.

National regulations further illustrate this variation. Denmark has implemented an even stricter limit of 2 ng/L for the same compounds as Swedens limitations [45]. Both Swedens and Denmark's limits are substantially lower than the regulations in EU and are more aligned with EFSA´s health-based recommendations, indicating a more precautionary approach. The stricter Danish limit raises questions about whether the Swedish limit is sufficiently protective, particularly for vulnerable populations such as pregnant women and infants. Öberg underlines that guideline values for PFAS in drinking water are derived by linking observed effects in human blood to estimated daily intake levels, followed by back-calculation to acceptable concentrations in drinking water based on assumed consumption rates. He notes that national limit values not only reflect on toxicological evidence but also policy decisions regarding how much of total PFAS exposure should be attributed to drinking water. While he means that current limits are considered protective to a large extent, he points out that they are not fully protective for all population groups. For example, highly exposed individuals, particularly children, may still exceed recommended intake levels even when drinking water meets existing standards. This may increase the risk of adverse developmental and immunological effects.

Although current assessments by the Swedish Food Agency indicate that TFA does not bioaccumulate in the human body and is not considered to pose health risks [47], Öberg emphasizes that this does not necessarily imply that TFA poses no risk, particularly under continuous exposure conditions. He means that continuous exposure to TFA, may still result in relatively stable concentrations in the body. Furthermore, TFA is highly persistent, mobile in the environment and originates from multiple precursor substances, which makes long-term exposure difficult to control [17]. Ultrashort-chain PFAS such as TFA are also characterized by high water solubility and environmental mobility. This raises concerns that current risk assessments may underestimate the potential health implications of such compounds.

An additional challenge identified by Öberg concerns the limitations of current risk assessment frameworks. He emphasizes that regulatory approaches remain largely based on individual substances, even though human exposure occurs as complex mixtures of many different PFAS, a challenge increasingly recognized in the scientific literature [17, 43]. There are also significant knowledge gaps regarding new and short-chain PFAS, as well as degradation products, which are not consistently included in current regulations. Öberg further stresses that the ongoing shift from long-chain to short-chain PFAS may not necessarily reduce health risks, but rather replace well-studied compounds with less understood alternatives. Although short-chain PFAS are generally less bioaccumulative and have shorter biological half-lives than long-chain PFAS, studies have shown that half-lives tend to increase with fluorinated chain length. In exposed individuals, estimated half-lives ranged from approximately 44 days for short-chain PFBS to nearly 3 years for PFHxS and PFOS after exposure to contaminated drinking water had ended [77]. Despite remaining in the human body for a shorter time, short-chain PFAS are more mobile in the environment and can persist in water systems [17]. This raises concerns that the current transition may repeat earlier patterns, where substances initially considered less harmful later prove to pose significant risks.

Taken together, these factors suggest that current regulatory frameworks may underestimate total PFAS exposure and indicate the need for adaptive regulatory frameworks as new evidence becomes available. This further supports the argument that PFAS regulation must address not only individual compounds, but also the broader chemical group and its dynamic evolution over time. As discussed above, current water purification methods have important limitations, and PFAS remains in the Swedish municipal drinking water. This is particularly relevant for pregnant women, fetuses and infants, as contiunues exposure may contribute to maternal body burden and subsequent transfer during pregnancy and breastfeeding. This underlines the importance of understanding the potential effects of PFAS exposure.

4.4 Effects of Early-Life PFAS Exposure

Annelise Blomberg, Associate Reserecher in Environmental Exposure and Epidemiologi at Lunds University (personal communication, 23 March 2026) provided valuable insights into the health impacts of PFAS on fetuses and children and the associated environmental exposure pathways. While several health outcomes have been consistently linked to PFAS exposure, the relationship between exposure and effect remains complex and, in many cases, difficult to interpret with certainty. The most well-established associations include reduced birth weight and impaired immune function, particularly reflected in decreased antibody responses to children's vaccinations [78]. Öberg emphasizes that there are indications of health effects even at relatively low exposure levels, although it remains difficult to define what constitutes "very low" exposure, since nearly all individuals already carry measurable PFAS levels in their blood. According to Öberg, the strongest evidence concerns the four PFAS included in EFSA's assessment (PFOA, PFNA, PFHxS and PFOS), where effects on the immune system, particularly reduced antibody response to vac-

cination in children, have been observed at comparatively low serum concentrations. He also notes that associations with increased blood lipid levels and reduced birth weight have been reported, although uncertainties remain regarding causality and effect thresholds. PFAS accumulation in placental tissue has also been associated with pregnancy complications, suggesting that the placenta itself may represent a target organ for PFAS toxicity [79], and emerging evidence also suggest that prenatal PFAS exposure may increase susceptibility to disease later in life [80]. This illustrates that even low-level exposure may be biologically relevant, while the exact dose–response relationships are still not fully understood. Blomberg explains that these associations are primarily based on epidemiological studies, which identify correlations rather than direct causation. This distinction is crucial, as children are simultaneously exposed to multiple environmental contaminants, making it challenging to isolate PFAS as the sole driver of observed health effects.

This difficulty is further emphasized by the nature of PFAS exposure itself. Individuals are rarely exposed to a single compound in isolation but rather to complex mixtures of different PFAS, often alongside other pollutants [78]. Additionally, exposure patterns tend to persist over time. For example, children with high prenatal exposure are also likely to experience continued postnatal exposure through breastfeeding and shared environmental sources, for example in areas with contaminated drinking water, resulting in correlated exposure patterns across early developmental stages [28, 27, 19]. This makes it methodologically challenging to determine whether specific developmental stages, such as in utero exposure versus early childhood, are more critical than others. Although frameworks such as the Developmental Origins of Health and Disease (DOHaD) support the idea that early-life exposures have long-term consequences, human data remains limited due to ethical constraints and the long timeframes required to observe chronic outcomes. However, a recent systematic review and meta-analysis reported associations between prenatal PFAS exposure and adverse neurodevelopmental outcomes in children [81]. Recent cohort studies have also reported associations between prenatal PFAS exposure, altered thyroid hormone function, and reduced intelligence quotient (IQ) in children [82].

The distinction between long-chain and short-chain PFAS introduces another layer of complexity to both scientific understanding and risk assessment (Blomberg, personal communication). Long-chain PFAS, such as PFOS and PFOA, are well studied due to their persistence in the human body, with half-lives spanning several years. Their tendency to bioaccumulate makes them easier to detect in human serum and has allowed researchers to establish clearer associations with health outcomes. In contrast, short-chain PFAS are more rapidly excreted and therefore less likely to accumulate to detectable levels in the body. While this has sometimes been interpreted as indicating lower risk, Blomberg and Öberg highlight that this assumption is not scientifically supported. The apparent lack of evidence for short-chain PFAS toxicity is largely a consequence of analytical and methodological limitations rather than proof of safety.

One of the major challenges in PFAS research today is the difficulty in detecting

and quantifying short-chain and emerging PFAS compounds. Unlike legacy PFAS, which can be measured using targeted analytical methods, many newer compounds require non-targeted approaches that are less precise and more uncertain (Blomberg, personal communication).

The issue of PFAS lies not only in the documented effects of known compounds but also in these substantial uncertainties. The persistence of PFAS in the environment, combined with their widespread use and resistance to degradation [78], means that exposure is ongoing and difficult to mitigate once contamination has occurred. Overall, the current state of knowledge suggests that PFAS exposure poses a meaningful risk to children's health, particularly during early developmental stages. However, significant uncertainties remain regarding the severity of these risks, the relative importance of different exposure pathways, and the effects of newer PFAS compounds. Importantly, the absence of conclusive evidence for many emerging PFAS should not be interpreted as evidence of safety. This combination of established harm and unanswered questions makes PFAS a particularly challenging and urgent issue in environmental health research today.

4.5 Societal and Ethical Considerations

Early-life exposure to PFAS represents a significant societal and ethical concern because it affects the most vulnerable populations, fetuses and infants, who cannot control their exposure. Since PFAS can be transferred from mother to child during pregnancy and breastfeeding, exposure occurs involuntarily and is shaped by environmental conditions [8, 78]. This shifts responsibility from individuals to society, including regulatory authorities and companies that use PFAS in their products. The potential for long-term health effects, even at low exposure levels, makes PFAS exposure a broader societal concern rather than solely an individual issue, particularly due to the potential future healthcare burden. Since many PFAS compounds remain insufficiently studied, particularly newer short-chain alternatives [2, 17], regulatory decisions often need to be made before complete toxicological evidence is available. This poses an ethical dilemma for regulatory authorities, as waiting for complete toxicological certainty may allow continued exposure to highly persistent substances with potentially harmful long-term effects. Therefore, precautionary approaches that prioritize early risk reduction may be important, especially when exposure affects vulnerable groups such as pregnant women and infants. In addition, PFAS contamination is often unevenly distributed, with some communities facing higher exposure due to historical pollution or less risk-aware producers [78, 49]. This raises issues of environmental justice and unequal protection.

The varying regulatory limits for PFAS in drinking water between different countries reflect differences in scientific interpretation, risk awareness, policy priorities, and regulatory feasibility [1, 78]. These limits should not be understood as absolute thresholds of safety, but rather as negotiated standards under uncertainty, where levels below the set limits may still pose risks, particularly for sensitive groups. As a result, differences in regulations can lead to unequal levels of protection across

populations and may undermine public trust when regulatory authorities define and communicate "safe" exposure levels inconsistently. From a societal perspective, this highlights the need for transparent risk communication to the general public, continuous investment in effective water treatment technologies, and thorough evaluation of chemicals before they are introduced on a larger scale. Ultimately, reducing PFAS exposure should be a shared societal responsibility involving regulatory authorities, industry, and water utilities. Since PFAS are highly persistent and difficult to remove once released into the environment, preventive measures aimed at limiting emissions and contamination may be more effective than relying solely on remediation and risk management after exposure has already occurred. Due to the potential for exposure to continue across generations, current regulatory decisions may influence not only present populations but also future generations.

5
Conclusion

This literature review suggests that current treatment methods can meet existing guideline values for PFAS in drinking water, but that these methods have significant limitations, particularly for short-chain and ultrashort-chain PFAS. As more research is conducted on these PFAS groups, it is likely that stricter restrictions will be introduced in the future, which may be challenging to meet with current treatment technologies. Furthermore, the analysis indicates a possible gap between what is biologically safe and what is technically and economically feasible. The fact that some countries, such as Sweden and particularly Denmark, have introduced stricter national drinking water limits than those established under the EU Drinking Water Directive underlines the uncertainty surrounding what constitutes sufficiently protective exposure levels. Vulnerable groups, including fetuses and children, may still be at risk of exceeding recommended exposure levels even when drinking water complies with existing regulatory limits. Of particular concern is that early-life exposure to PFAS has been associated with immunological and developmental effects, with reduced antibody responses to childhood vaccinations being among the most consistently reported findings. At the same time, current understanding is characterized by significant uncertainties. The lack of data, especially for short-chain and ultrashort-chain PFAS, should not be interpreted as evidence of low risk, but rather as a limitation of current analytical and epidemiological methods.

In addition, current risk assessment and regulation remain mainly substance-specific, despite human exposure occurring as complex mixtures. This may lead to an underestimation of the overall risk. The ongoing shift from long-chain to short-chain PFAS also poses a risk for so called "regrettable substitution", where less studied compounds replace well-studied hazardous substances without sufficient evidence of safety. Among the evaluated techniques, electrochemical oxidation appears particularly promising, as it has the potential to fully degrade PFAS, thereby minimizing the generation of waste streams that require further treatment. EAOP has already been commercialized in several countries, further supporting its potential as a promising technology that could be implemented more widely to reduce long-term environmental and public health risks through drinking water, one of the largest sources of PFAS exposure that consumers and vulnerable populations cannot avoid through personal choice. However, this must be viewed in the context that no optimal treatment method currently exists, since all available technologies have inherent limitations related to energy consumption, operational costs, scalability, and integration into existing water treatment systems. Furthermore, future breakthroughs will likely depend on treatment technologies capable of simultaneously addressing

multiple contaminants while maintaining cost-effective operation in complex water matrices. Overall, the study points to a need for both the development of more efficient and economically feasible treatment solutions, alongside stricter and more comprehensive regulatory approaches that consider PFAS as a group rather than as individual compounds. In the long term, increased emphasis on source control may be necessary to reduce environmental and human exposure, as complete degradation of PFAS is likely to be resource-intensive and associated with high economic and energy costs.

Bibliography

[1] Swedish Chemicals Agency. PFAS [Internet]. 2025. [cited 2026-02-17]. Available from: https://www.kemi.se/en/chemical-substances-and-materials/pfas#h-WhatarePFAS

[2] Brendel S, Fetter É, Staude C, Vierke L, Biegel-Engler A. Short-chain perfluoroalkyl acids: environmental concerns and a regulatory strategy under REACH. *Environmental Science Europe*. 2018;30(1):9. Available from: https://link.springer.com/article/10.1186/s12302-018-0134-4

[3] De Silva A, Armitage J, Bruton T, Dassuncao C, Heigher-Bernays W, Hu X, et al. PFAS Exposure Pathways for Humans and Wildlife: A Synthesis of Current Knowledge and Key Gaps in Understanding. *Environmental Toxicology and Chemistry*. 2021;40:3. Available from: https://doi.org/10.1002/etc.4935

[4] Everfilt Admin. Short-Chain vs. Long-Chain PFAS: What Sets Them Apart? [Internet]. 2025. [cited 2026-02-17]. Available from: https://www.everfilt.com/post/short-chain-vs-long-chain-pfas-what-sets-them-apart

[5] Li F, Duan J, Tian S, Ji H, Zhu Y, Wei Z, et al. Short-chain per- and polyfluoroalkyl substances in aquatic systems: Occurrence, impacts and treatment. *Chemical Engineering Journal*. 2020;380:122506. Available from: https://www.sciencedirect.com/science/article/abs/pii/S1385894719319096

[6] Macheka-Tendenguwo LR, Olowoyo JO, Mugivhisa LL, Abafe OA. Per- and polyfluoroalkyl substances in human breast milk and current analytical methods. *Environmental Science and Pollution Research.* 2018;25:36064–36086. Available from: https://doi.org/10.1007/s11356-018-3483-z

[7] United States Environmental Protection Agency (EPA). Per-and Polyfluoroalkyl Substances (PFAS) [Internet]. 2026. [cited 2026-05-11]. Available from: https://www.epa.gov/sdwa/and-polyfluoroalkyl-substances-pfas?

[8] Swedish Chemicals Agency. Children are especially sensitive [Internet]. 2021. [cited 2026-02-19]. Available from: https://www.kemi.se/en/chemicals-in-our-everyday-lives/chemicals-in-the-everyday-lives-of-children/children-are-especially-sensitive

[9] Szilagyi J, Avula V, Fry R. Perfluoroalkyl Substances (PFAS) and Their Effects on the Placenta, Pregnancy and Child Development: A Potential Mechanistic Role for Placental Peroxisome Proliferator-Activated Receptors (PPARs). *Current Environmental Health Reports*. 2020;7(3):222–230. Available from: https://pubmed.ncbi.nlm.nih.gov/32812200/

[10] Livsmedelsverket. PFAS - Per- och polyfluorerade alkylsubstanser [Internet]. 2026. [cited 2026-02-23]. Available from: https://www.livsmedelsverket.s

e/livsmedel-och-innehall/oonskade-amnen/miljogifter/PFAS-poly-och-perfluorerade-alkylsubstanser/

[11] Birchfield A, Musayev F, Castillo A, Zorn G, Fuglestad B. Broad PFAS Binding with Fatty Acid Binding Protein 4 Is Enabled by Variable Binding Modes. *JACS Au*. 2025;5(6):2469–2474. Available from: https://pubs.acs.org/doi/full/10.1021/jacsau.5c00504

[12] Karolinska institutet. Carboxylic Acids [Internet]. d.u. [cited 2026-05-07]. Available from: https://mesh.kib.ki.se/term/D002264/carboxylic-acids

[13] Britannica. Sulfonic acid [Internet]. 2026. [cited 2026-05-07]. Available from: https://www.britannica.com/science/sulfonic-acid

[14] Brase RA, Mullin EJ, Spink DC. Legacy and Emerging Per- and Polyfluoroalkyl Substances: Analytical Techniques, Environmental Fate, and Health Effects. *International Journal of Molecular Sciences*. 2021;22(3):995. Available from: https://doi.org/10.3390/ijms22030995

[15] Ebel M. *Per- and polyfluoroalkyl substances (PFAS) in early life: exposure and health effects.* [Internet] Lund: Lund University; 2023. Available from: https://lup.lub.lu.se/search/files/166707509/Matilda_Ebel_kappa_e-spik.pdf

[16] Lu Y, Guan R, Zhu N, Hao J, Peng H, He A, et al. A critical review on the bioaccumulation, transportation, and elimination of per- and polyfluoroalkyl substances in human beings. *Critical Reviews in Environmental Science and Technology*. 2023;54(2):95–116. Available from: https://www.tandfonline.com/doi/full/10.1080/10643389.2023.2223118

[17] Ateia M, Maroli A, Tharayil N, Karanfil T. The overlooked short- and ultrashort-chain poly- and perfluorinated substances: A review. *Chemosphere*. 2019;220:866–882. Available from: https://www.sciencedirect.com/science/article/pii/S0045653518325281

[18] Danish EPA. *Short-chain Polyfluoroalkyl Substances (PFAS): A literature review of information on human health effects and environmental fate and effect aspects of short-chain PFAS.* [Internet]. Copenhagen: Danish Environmental Protection Agency; 2015. Available from: https://cswab.org/wp-content/uploads/2018/05/Short-chain-PFAS-literature-review-Danish-EPA-2015.pdf

[19] Mamsen LS, Björvang RD, Mucs D, Vinnars MT, Papadogiannakis N, Lindh CH, et al. Concentrations of perfluoroalkyl substances (PFASs) in human embryonic and fetal organs from first, second, and third trimester pregnancies. *Environment International*. 2019;124:482–492. Available from: https://doi.org/10.1016/j.envint.2019.01.010

[20] Lunds Universitet. *Ronneby PFAS Research Program: sammanfattning av forskning 2014.* [Internet] Lund; Lund University; 2014. Available from: https://lucris.lub.lu.se/ws/portalfiles/portal/161273411/Ronneby_PFAS_Research_Program_sammanfattning_av_forskning_2014.pdf

[21] Wu Y, Bao J, Liu Y, Wang X, Qu W. A Review on Per- and Polyfluoroalkyl Substances in Pregnant Women: Maternal Exposure, Placental Transfer, and Relevant Model Simulation. *Toxics*. 2023;11(5):430. Available from: https://doi.org/10.3390/toxics11050430

[22] Ma D, Lu Y, Liang Y, Ruan T, Li J, Zhao C, et al. A Critical Review on Transplacental Transfer of Per- and Polyfluoroalkyl Substances: Prenantal Exposure Levels, Characteristics, and Mechanisms. *Environmental Science & Technology* 2022;56(10):6014–6026. Available from: https://doi.org/10.1021/acs.est.1c01057

[23] Xu Y, Sui X, Li J, Zhang L, Wang P, Liu Y, et al. Early-life exposure to per- and polyfluoroalkyl substances: Analysis of levels, health risk and binding abilities to transport proteins. *Eco-Environment & Health.* 2024;3(3):308-316. Available from: https://doi.org/10.1016/j.eehl.2024.04.007

[24] Herrick EJ, Bordoni B. Embryology, Placenta. *In: StatPearls* [Internet]. Treasure Island (FL). 2023. Available from: https://www.ncbi.nlm.nih.gov/books/NBK551634/

[25] Burton GJ, Jauniaux E. What is the placenta?. *American Journal of Obstetrics and Gynecology* 2015;213(4):S6.e1-S6.e8. Available from: https://pubmed.ncbi.nlm.nih.gov/26428504/

[26] Norén E, Blomberg AJ, Lindh C, Pineda D, Jakobsson K, Nielsen C. Transplacental transfer efficiency of perfluoroalkyl substances (PFAS) after long-term exposure to highly contaminated drinking water: a study in the Ronneby Mother-Child Cohort. *Journal of Exposure Science & Environmental Epidemiology.* 2025;35:445–453. Available from: https://www.nature.com/articles/s41370-025-00758-2

[27] Nyberg E, Awad R, Bignert A, Ek C, Sallsten G, Benskin JP. Inter-individual, inter-city, and temporal trends of per- and polyfluoroalkyl substances in human milk from Swedish mothers between 1972 and 2016. *Environmental Science: Processes Impacts.* 2018;20:1136-1147. Available from: https://doi.org/10.1039/C8EM00174J

[28] Blomberg AJ, Norén E, Haug LS, Lindh C, Sabaredzovic A, Pineda D, et al. Estimated Transfer of Perfluoroalkyl Substances (PFAS) from Maternal Serum to Breast Milk in Women Highly Exposed from Contaminated Drinking Water: A study in the Ronneby Mother-Child Cohort. *Environmental Health Perspectives* 2023;131(1):017005. Available from: https://pmc.ncbi.nlm.nih.gov/articles/PMC9869870/

[29] World Health Organization. Breastfeeding [Internet]. Geneva: World Health Organization; [cited 2026-05-04]. Available from: https://www.who.int/health-topics/breastfeeding#tab=tab_1

[30] Haimbaugh A, Meyer DN, Connell ML, Blount-Pacheco J, Tolofari D, Gonzalez G, et al. Enviromental Exposure to Per- and Polyfluoroalkyl Substances (PFAs) and Reproductive Outcomes in the General Population: A systematic Review of Epidemiological Studies. *International Journal of Environmental Research and Public Health.* 2024;21(12):1615 Available from: https://doi.org/10.3390/ijerph21121615

[31] Statistiska centralbyrån (SCB). Födda i Sverige [Internet]. 2026. [cited 2026-03-03]. Available from: https://www.scb.se/hitta-statistik/sverige-i-siffror/manniskorna-i-sverige/fodda-i-sverige/

[32] Zhao L, Teng M, Zhao X, Li Y, Sun J, Zhao W, et al. Insight into the binding model of per- and polyfluoroalkyl substances to proteins and mem-

branes. *Environment International*. 2023:175:107951. Available from: https://doi.org/10.1016/j.envint.2023.107951

[33] Gyllenhammar I, Benskin JP, Plassmann M, Kruså M, McCleaf P, Hedvall Kallerman P, et al. PFAS in first-time mothers from Sweden: temporal trends and the impact from fish/seafood consumption and drinking water exposure. *Environment International*. 2025;202:109671. Available from: https://doi.org/10.1016/j.envint.2025.109671

[34] Sveriges geologiska undersökning (SGU). Per- och polyfluorerade alkylsubstanser (PFAS) [Internet]. 2024. [cited 2026-03-05]. Available from: https://www.sgu.se/anvandarstod-for-geologiska-fragor/bedomningsgrunder-for-grundvatten/grundvattnets-kvalitet--organiska-amnesgrupper/pfas/
X

[35] Espartero LJL, Ishaq Z, Bradley S, Moore M, Yamada M, Wang X, et al. Dermal permeation of perfluoroalkyl substances in human skin: An in-vitro study.*Chemosphere*. 2025;378:144408. Available from: https://doi.org/10.1016/j.chemosphere.2025.144408

[36] Livsmedelsverket. Vatten [Internet]. 2025. [cited 2026 Mar 23]. Available from: https://www.livsmedelsverket.se/livsmedel-och-innehall/naringsamne/vatten/

[37] European Food Safety Authority. Scientific Opinion on Dietary Reference Values for water. *EFSA Journal*. 2010;8(3):1459. Available from: https://efsa.onlinelibrary.wiley.com/doi/epdf/10.2903/j.efsa.2010.1459

[38] Gluge J, Scheringer M, Cousins IT, DeWitt JC, Goldenman G, Herzke D, et al. An overview of the uses of per- and polyfluoroalkyl substances (PFAS). *Environmental Science: Processes & Impacts*. 2020;22:2345–2373. Available from: https://doi.org/10.1039/D0EM00291G

[39] Xu Y, Nielsen C, Li Y, Hammarstrand S, Andersson EM, Li H, et al. Serum perfluoroalkyl substances in residents following long-term drinking water contamination from firefighting foam in Ronneby, Sweden. *Environmental International*. 2021;147:106333. Available from: https://doi.org/10.1016/j.envint.2020.106333

[40] Anderson RH, Long GC, Porter RC, Anderson JK. Occurrence of select perfluoroalkyl substances at U.S. Air Force aqueous film-forming foam release sites other than fire-training areas: Field-validation of critical fate and transport properties. *Chemosphere*. 2016;150:678–685. Available from: https://doi.org/10.1016/j.chemosphere.2016.01.014

[41] Shu H, Lindh CH, Wikström S, Bornehag CG. Temporal trends and predictors of perfluoroalkyl substances serum levels in Swedish pregnant women in the SELMA study. *PLoS ONE*. 2018;13(12):e0209255. Available from: https://doi.org/10.1371/journal.pone.0209255

[42] Plassman M, Benskin JP, Halldin Ankarberg E, Gyllenhammar I. *Levels of ultra-short chain perfluoroalkyl substances (PFAS) in urine samples from first-time mothers in Uppsala, Sweden*. Uppsala: Naturvårdsverket; 2022. Available from: https://urn.kb.se/resolve?urn=urn:nbn:se:naturvardsverket:diva-10368

[43] EFSA CONTAM Panel. Risk to human health related to the presence of perfluoroalkyl substances in food. *EFSA Journal*. 2020;18(9):6223. Available from: https://doi.org/10.2903/j.efsa.2020.6223

[44] European Parliament and council of the European Union. Directive (EU) 2020/2184 of the European Parliament and of the Council of 16 December 2020 on the quality of water intended for human consumption. *Official Journal of the European Union*. 2020. Available from: https://eur-lex.europa.eu/eli/dir/2020/2184/oj

[45] Grung M, Hjermann DØ, Rundberget T, Bæk K, Thomsen C, Knutsen HK, et al. Low levels of per- and polyfluoroalkyl substances (PFAS) detected in drinking water in Norway, but elevated concentrations found near known sources. *Science of The Total Environment*. 2024;947:174550. Available from: https://doi.org/10.1016/j.scitotenv.2024.174550

[46] Brennan, N.M.; Evans, A.T.; Fritz, M.K., Peak, S.A., von Holst, H.E. Trends in the Regulation of Per- and Polyfluoroalkyl Substances (PFAS): A Scoping Review. Int. J. Environ. Res. Public Health 2021, 18, 10900. Available from https://doi.org/10.3390/ijerph182010900

[47] Livsmedelsverket. PFAS – poly- och perfluorerade alkylsubstanser [Internet]. 2026. [cited 2026-03-10]. Available from: https://www.livsmedelsverket.se/livsmedel-och-innehall/oonskade-amnen/miljogifter/pfas-poly-och-perfluorerade-alkylsubstanser

[48] Svenskt Vatten. PFAS [Internet]. 2026. [cited2026-03-03]. Available from: https://www.svensktvatten.se/vara-sakomraden/klimat-och-hallbarhet/pfas/

[49] Banzhaf S, Filipovic M, Lewis J, Sparrenbom CJ, Barthel R. A review of contamination of surface-, ground-, and drinking water in Sweden by perfluoroalkyl and polyfluoroalkyl substances (PFASs). *Ambio.* 2016;46(3):335–346. Available from:https://doi.org/10.1007/s13280-016-0848-8.

[50] Rahman MF, Peldszus S, Anderson WB. Behaviour and fate of perfluoroalkyl and polyfluoroalkyl substances (PFASs) in drinking water treatment: A review. *Water Research*. 2014;50:318-340. Available from: https://doi.org/10.1016/j.watres.2013.10.045

[51] Appleman TD, Higgins CP, Quiñones O, Vanderford BJ, Kolstad C, Zeigler-Holady JC, et al. Treatment of poly- and perfluoroalkyl substances in U.S. full-scale water treatment systems. *Water Research.* 2014;51:246-255. Available from: https://doi.org/10.1016/j.watres.2013.10.067.

[52] McCleaf P, Stefansson W, Ahrens L. Drinking water nanofiltration with concentrate foam fractionation — A novel approach for removal of per- and polyfluoroalkyl substances (PFAS). *Water Research.* 2023;232:119688. Available from: https://doi.org/10.1016/j.watres.2023.119688.

[53] IVL Svenska Miljöinstitutet. PFAS-rening av vatten [Internet]. 2025. [cited 2026-03-03]. Available from: https://www.ivl.se/vart-erbjudande/vara-tjanster/pfas-rening-av-vatten.html

[54] Winchell L, Ross J, Wells M, Fonoll X, Norton JR J, Bell K. Per-and polyfluoroalkyl substances thermal destruction at water resource recovery facilities: A

state of the science review *Water Environment Research.* 2021;93(6):826-843. Available from: https://doi.org/10.1002/wer.1483.

[55] Dixit F, Dutta R, Barbeau B, Berube P, Mohseni M. PFAS removal by ion exchange resins: A review. *Chemosphere.* 2021;272:129777. Available from: https://doi.org/10.1016/j.chemosphere.2021.129777.

[56] Murray CC, Marshall RE, Liu CJ, Vatankhah H, Bellona CL. PFAS treatment with granular activated carbon and ion exchange resin: Comparing chain length, empty bed contact time, and cost. *Journal of Water Process Engineering.* 2021;44:102342. Available from: https://doi.org/10.1016/j.jwpe.2021.102342.

[57] Mian Md, Zhu J, Jiang X, Deng S. Recent advances in activated carbon driven PFAS removal: structure-adsorption relationship and new adsorption mechanisms . *Frontiers of Environmental Science and Engineering.* 2025;19:78. Available from:https://doi.org/10.1007/s11783-025-1998-3

[58] Belkouteb N, Franke V, McCleaf P, Köhler S, Ahrens L. Removal of per- and polyfluoroalkyl substances (PFASs) in full-scale drinking water treatment plant: Long-term performance of granular activated carbon (GAC) and influence of flow-rate. *Water Research.* 2020;182:115913. Available from:https://doi.org/10.1016/j.watres.2020.115913

[59] Liu C, Zhao X, Faria AF, Deliz Quiñones KY, Zhang C, He Q, et al. Evaluating the efficiency of nanofiltration and reverse osmosis membrane processes for PFAS removal from water: A critical review. *Separation and Purification Technology.* 2022;302:122161 Available from: https://doi.org/10.1016/j.seppur.2022.122161

[60] Tow EW, Ersan MS, Ersan MS, Kum S, Lee T, Speth TF, et al. Managing and treating per- and polyfluoroalkyl substances (PFAS) in membrane concentrates. *AWWA Water Science.* 2021;3(5):e1233. Available from: https://doi.org/10.1002/aws2.1233

[61] Mastropietro TF, Bruno R, Pardo E, Armentano D. Reverse osmosis and nanofiltration membranes for highly efficient PFASs removal: overview, challenges and future perspectives. *Dalton Transactions.* 2021;50:5398–5410. Available from: https://doi.org/10.1039/D1DT00360G

[62] Buckley T, Xu X, Rudolph V, Firouzi M, Shukla P. Review of foam fractionation as water treatment technology. *Separation Science Technology* 2022;57(6):929-958. Available from:https://doi.org/10.1080/01496395.2021.1946698

[63] We ACE, Zamyadi A, Stickland AD, Clarke BO, Freguia S. A review of foam fractionation for the removal of per- and polyfluoroalkyl substances (PFAS) from aqueous matrices. *Journal of Hazardous Materials.* 2024;465:133182. https://doi.org/10.1016/j.jhazmat.2023.133182

[64] Klevan C, Allen OV, Mukai K, Gomes A, Xia S, Caines S, et al. Removal of long- and short-chain PFAS from groundwater by foam fractionation. *Environmental Science: Water Research & Technology.* 2025;11:2295–2307. Available from: https://pubs.rsc.org/en/content/articlehtml/2025/ew/d5ew00440c

[65] Topolovec B, Jovanovic O, Puac N, Skoro N, Cuervo Lumbaque E, Petrovic M. Plasma water treatment for PFAS: Study of degradation of perfluorinated substances and their byproducts by using cold atmospheric pressure plasma jet

Journal of Environmental Chemical Engineering. 2024;12(3):112979. Available from: https://doi.org/10.1016/j.jece.2024.112979

[66] Tshangana CS, Nhlengethwa ST, Glass S, Denison S, Kuvarega AT, Nkambule TTI, et al. Technology status to treat PFAS-contaminated water and limiting factors for their effective full-scale application. *npj Clean Water*. 2025;8:41. Available from: https://doi.org/10.1038/s41545-025-00457-3

[67] Buck RC, Franklin J, Berger U, Conder JM, Cousins IT, de Voogt P, et al. Perfluoroalkyl and polyfluoroalkyl substances in the environment: terminology, classification, and origins. *Integrated Environmental Assessment and Management*. 2011;7(4):513–541. Available from: https://doi.org/10.1002/ieam.258

[68] Shahsavari E, Rouch D, Khudur LS, Thomas Duncan, Aburto-Medina A, Ball AS. Challenges and Current Status of the Biological Treatment of PFAS-Contaminated Soils. *Frontiers in Bioengineering and Biotechnology*. 2020;8:602040. Available from: https://www.frontiersin.org/journals/bioengineering-and-biotechnology/articles/10.3389/fbioe.2020.602040/full

[69] Asharuddin S, Othman N, Al-Gheethi A, Noman A, Alsubhi L, Madhi A, et al. Recent Advancement and Understanding on the "Forever Chemicals" , PFAS in Drinking Water. *Water, Air, & Soil Pollution* 2025;237:66. Available from: https://doi.org/10.1007/s11270-025-08636-1

[70] Tabatabaei M, Cho D, Fahad S, Jeong D, Hwang J. Photocatalytic innovations in PFAS removal: Emerging trends and advances. *Science of The Total Environment* 2025;980:179567. Available from: https://doi.org/10.1016/j.scitotenv.2025.179567

[71] Elbruk. Spotpriser för elområde SE3-Stockholm. [Internet]. 2026. [cited 2026-05-11]. Available from: https://www.elbruk.se/timpriser-se3-stockholm

[72] Radjenovic J, Duinslaeger N, Avval SS, Chaplin BP. Facing the Challenge of Poly- and Perfluoroalkyl Substances in Water: Is Electrochemical Oxidation the Answer? *Environmental Science & Technology* 2020;54(23):14815–14829. Available from: https://doi.org/10.1021/acs.est.0c06212

[73] Li J, Li D, Liu C, Shi W, Yu J, Chen M, et al. A review of the flow-through electrocatalytic water treatment process. *Separation and Purification Technology* 2025;365:132545. Available from: https://doi.org/10.1016/j.seppur.2025.132545

[74] Xu X, Li Y, Vo PHN, Shukla P, Ge L, Zhao CX. Electrochemical advanced oxidation of per- and polyfluoroalkyl substances (PFASs): Development, challenges and perspectives. *Chemical Engineering Journal*. 2024;500:157222. Available from: https://www.sciencedirect.com/science/article/pii/S1385894724087138?via%3Dihub

[75] PFASuiki GmbH. Electrochemical PFAS Destruction - For a Cleaner World [Internet]. 2026. [Cited 2026-05-13] Available from: https://www.pfasuiki.com/

[76] Liu C, McKay G, Jiang D, Tenorio R, Cath T, Amador C, et al. Pilot-scale field demonstration of a hybrid nanofiltration and UV-sulfite treatment train for groundwater contaminated by per- and polyfluoroalkyl substances (PFASs).

Water Research. 2021;205:117677. Available from: https://doi.org/10.1016/j.watres.2021.117677

[77] Xu Y, Fletcher T, Pineda D, Lindh CH, Nilsson C, Glynn A, et al. Serum Half-Lives for Short- and Long-Chain Perfluoroalkyl Acids after Ceasing Exposure from Drinking Water Contaminated by Firefighting Foam. *Environmental Health Perspectives*. 2020;128(7):077004. Available from: https://doi.org/10.1289/EHP6785

[78] Sunderland EM, Hu XC, Dassuncao C, Tokranov AK, Wagner CC, Allen JG. A review of the pathways of human exposure to poly- and perfluoroalkyl substances (PFASs) and present understanding of health effects. *Journal of Exposure Science & Environmental Epidemiology*. 2019;29:131–147. Available from: https://doi.org/10.1038/s41370-018-0094-1

[79] Groisman L, Berman T, Quinn A, Pariente G, Rorman e, Karakis I, et al. Levels of PFAS concentrations in the placenta and pregnancy complications. *Ecotoxicology and Environmental Safety*. 2023:262:115165. Available from: https://www.sciencedirect.com/science/article/pii/S0147651323006693

[80] Örebro University. PFAS kan öka risken för sjukdomar hos ofödda barn [Internet]. 2024. Available from: https://www.oru.se/nyheter/nyhetsarkiv/nyhetsarkiv-2024/pfas-kan-oka-risken-for-sjukdomar-hos-ofodda-barn/

[81] Liu D, Yan S, Liu Y, Chen Q, Ren S. Associations of prenatal exposure to perfluorinated and polyfluoroalkyl substances with childhood neurodevelopment: A systematic review and meta-analysis. *Ecotoxicology and Environmental Safety*. 2024:271:115939. Available from: https://www.sciencedirect.com/science/article/pii/S0147651324000149

[82] Zhang B, Wang Z, Zhang J, Dai Y, Ding J, Guo J, et al. Prenatal exposure to per- and polyfluoroalkyl substances, fetal thyroid function, and intelligence quotient at 7 years of age: Findings from the Sheyang Mini Birth Cohort Study. *Environment International*. 2024;187:108720. Available from: https://doi.org/10.1016/j.envint.2024.108720

A

Appendix 1

A.1 Interview Questions

This appendix presents the interviews conducted as part of this study. The purpose of these interviews was to provide expert insights into PFAS-related challenges. The interviewees represent a range of relevant professional fields. The interview questions are presented below.

A.1.1 Philipp Wanner - Associate Professor in Contaminant Hydrogeology at Gothenburg University

- Which technologies are currently used in Sweden to remove PFAS from drinking water and how are they implemented in practice at water treatment plants?
- How effective are the commonly used PFAS removal technologies and what are their limitations?
- How does the PFAS composition, for example short-chain vs long-chain PFAS, influence the efficiency of different treatment methods?
- How does the composition of the raw water, such as the presence of natural organic matter, competing ions or pH, affect the efficiency of PFAS removal?
- Why are PFAS particularly difficult to remove from drinking water compared to other organic contaminants?
- What happens to the PFAS that are removed during the treatment process and are there risks that the contamination problem is simply transferred elsewhere?
- Which emerging or developing technologies for PFAS removal are currently being researched and how promising do you consider them compared to existing methods?
- Looking ahed, what are the biggest challenges and research gaps in PFAS removal from drinking water and do you think current technologies will be sufficient to meet future regulations?

A.1.2 Mattias Öberg - Associate Professor in Toxicology at Karolinska Institutet

- Are there signs of effects even at very low exposure levels?
- How are guideline values for PFAS in drinking water established? Are current limits sufficiently protective?

- How significant is contaminated drinking water as an exposure source for pregnant women and their fetuses compared to other exposure pathway?
- What are the major knowledge gaps regarding PFAS and health risks?
- What consequences might the shift to short-chain PFAS have from a toxicological perspective?
- What is needed to improve PFAS risk assessments in the future?

A.1.3 Annelise Blomberg - Associate Resercher in Environmental Exposure and Epidemiologi at Lunds University

- Based on your research, how important is drinking water as a source of PFAS exposure compared with diet or consumer products?
- Which types of PFAS are most commonly detected in drinking water compared with those detected in human blood or serum? Are legacy PFAS generally found at lower concentrations today compared with short-chain PFAS that are more widely used?
- How well do PFAS concentrations measured in blood or serum reflect actual exposure from sources such as drinking water?
- Which health effects n children currently have the strongest evidence linking them to early-life PFAS exposure?
- Are there particularly sensitive periods during pregnancy or early childhood when PFAS exposure may have a greater impact?
- Do short-chain PFAS pose similar risks to fetal or child development as long-chain PFAS?
- What are the biggest knowledge gaps regarding PFAS exposure and children's health?

A.1.4 Karthik Laxman Kunjali - Co-Founder and CEO of Stockholm Water Technology

- Could you briefly explain your technology and how it works in practice?
- Do you currently work with PFAS in your solutions and if so, in what way?
- How does the composition of the raw water, for example organic matter, affect how your technology performs or is applied?
- Do you see differences in how different types of PFAS, such as long- and short-chain compounds, can be removed with your technology? Are there particular challenges when it comes to short-chain PFAS?
- What would you say is the main limitation of current PFAS treatment? Is it the technology, the cost or regulation?
- How are the contaminants that become concentrated during your process handled after treatment?
- Which types of water sources do you see as being most affected by PFAS?
- How do you see your role evolving in the future of PFAS and water treatment solutions?

A.1.5 Kristin Barkman - Water and Environmental Consultant at Sweco

- How does the composition of the raw water (e.g. organic matter) influence the choice of treatment technology?
- How do you manage the PFAS concentrate that may remain after treatment?
- What role does Sweco typically have in PFAS projects? Is it mainly advisory work, design of the treatment systems, or management of the entire process?
- Are there any promising research projects or pilot studies that you are currently following?
- What is the main limitation of current PFAS treatment: technology, cost, or legislation?
- Are different treatment technologies selected based on the type of PFAS involved? IS the choice based on whether the PFAS are long-chain or short-chain? What are the limitations when it comes to removing short-chain PFAS?
- Which types of water sources (groundwater, surface water, private wells) are most vulnerable?

A.2 Table With Common Sources of PFAS

This appendix represents common sources of PFAS exposure for pregnant women and children. The classification is based on a synthesis of literature on PFAS exposure pathways in everyday life and should be interpreted as indicative rather than complete, as exposure may vary depending on local contamination and individual behaviour.

Table A.1: Examples of common sources of PFAS exposure for mom and baby [3, 33, 32, 47, 43, 38, 32, 78]

Category	**Examples**	**Relevance for PFAS exposure**
Drinking water	Tap water, groundwater, surface water	Continuous daily intake makes drinking water a major exposure pathway, particularly in contaminated areas.
Food (fish and seafood)	Fish, shellfish, freshwater fish	PFAS can bioaccumulate in aquatic food chains, leading to higher concentrations in fish and seafood, especially from contaminated waters.
Infant food	Baby food, infant formula	Important exposure pathway for infants, who have higher intake per body weight and increased susceptibility to PFAS exposure.
Processed foods	Ready meals, fast food, packaged snacks	May contribute to PFAS exposure through both contaminated ingredients and contact with PFAS-containing packaging material.
Food packaging	Fast food wrappers, pizza boxes, microwave popcorn bags, grease-resistant paper and cardboard	PFAS are used for grease resistance and may migrate from packaging materials into food, contributing to dietary exposure.
Animal products	Meat, eggs, dairy products, liver	PFAS bind to proteins and may be present in animal-derived foods, contributing to dietary exposure.
Indoor environment	Household dust, treated carpets, furniture	PFAS-containing materials can release particles that accumulate in dust and may be ingested.
Beverages (other than water)	Coffee, tea, soft drinks, juice and other beverages	May contribute to PFAS exposure through water used in production or contact with packaging materials.
Cookware	Non-stick cookware, e.g. PTFE-coated pans	PFAS-related substances have historically been used in coatings and may contribute to exposure, particularly from older or damaged products.
Textiles	Waterproof clothing, stain-resistant fabrics, carpets, upholstery	PFAS are used for water- and stain-repellent properties and may contribute to indirect exposure via dust and skin contact.
Consumer products	Cosmetics, cleaning products, personal care products	PFAS are used for performance properties such as durability and water repellency and may lead to dermal or indirect exposure.
Firefighting foams	Aqueous film-forming foams (AFFF)	Major historical source of environmental contamination, particularly near airports and military sites.
Plant-based foods	Vegetables, fruits, root crops	PFAS can be taken up from contaminated soil and irrigation water, contributing to dietary exposure, particularly in polluted areas.
Grains and cereals	Wheat, rice, bread, cereals	Generally contain lower PFAS concentrations, but may contribute to background exposure due to high consumption rates.

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