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Interim report

Proposal for soil remediation values for Perfluorooctane sulfonic acid (PFOS) and perfluorooctanoic acid (PFOA)

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Study carried out on behalf of OVAM
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SUMMARY

The present report discusses the proposals for soil remediation values (SRV) for Perfluorooctane sulfonic acid (PFOS) and Perfluorooctanoic acid (PFOA). The derivation follows the guidelines set out in Cornelis and Touchant (2016). The calculations of the soil remediation values were carried out with an adapted version of S-Risk 1.3, a model for human exposure and health risk assessment at contaminated sites.
For soil, three toxicology scenarios were retained, namely US-EPA (2016c) (= preferred scenario), Zeilmaker et al. (2016) and EFSA (2018c). For the ecotoxicological reference values, biomagnification was taken into account. Table S 1 and Table S 2 show the resulting soil remediation values proposed for PFOS and PFOA respectively, with the preferred values in green. For the decision on the SRV for landuse type II (agriculture), the study 'Derivation of target values for perfluorinated compounds' is pending, and policy may be adjusted on the basis of the target values and the values for free use of soil.

Table S 1: The proposed SRV for soil (µg/kg dm) for PFOS

Landuse type	II[1]	III	IV	V
Flemish legislation on soil (VLAREBO)	-	-	-	-
Proposal human health based tox US-EPA (2016c)	3.1	204.6	1,949 (drinking water) (IVb)	1,949 (drinking water (Va and b)
Proposal human health based tox Zeilmaker et al. (2018)	0.84	55.05	1,949 (drinking water) (IVb)	1,949 (drinking water (Va and b)
Proposal human health based tox EFSA (2018c)	0.11	6.63	447.2 (IVa)	1,488 (Vb)
Proposal ecotox	3	18	110	9,100
Background value	1.5			

[1] Not final, will be adjusted on the basis of the target values and the values for free use of soil

Table S 2: The proposed SRV for soil (µg/kg dm) for PFOA

	II[1]	III	IV	V
Flemish legislation on soil (VLAREBO)	-	-	-	-
Proposal human health based tox US-EPA (2016c)	4.3	205	643 (drinking water) IV a and IV b	643 (drinking water) Va and Vb
Proposal human health based tox Zeilmaker et al. (2016)	2.7	127	643 (drinking water) IV a and IV b	643 (drinking water) Va and Vb
Proposal human health based tox EFSA (2018c)	0.14	6.2	375 (Va)	443 (Vb)
Proposal ecotox	7	89	1,100	50,000
Background value	1.0			

No background values for Flemish soils were available at the time this study was carried out. On behalf of OVAM, background values were measured in 2020, for which, for PFOS and PFOA, a background value of 1.5 respectively 1.0 µg/kg dm in soil was derived.

The SRV for groundwater has a human health based underpinning, and corresponds to the drinking water standard if this has a toxicological basis (Cornelis & Touchant, 2016). The drinking water standard of 100 ng/l proposed by the EU is mainly based on feasibility and not on toxicology. As such, the SRV for groundwater was calculated for the three toxicology scenarios US-EPA (2016c), Zeilmaker et al. (2016) and EFSA (2018c). The derived SRV for groundwater for PFOS and PFOA are given in Table S 3, with the selected value in green.

Table S 3: The calculated SRV for groundwater (ng/l) for PFOS and PFOA

Toxicological reference value	Value	Unit	SRV groundwater (ng/l)	
			PFOS	PFOA
Set 1 (preference) US-EPA (2016a)				
TDI oral	2.10^{-5}	mg/kg/d	120	120
TCA inhalation	7.10^{-5}	mg/m³		
TDI dermal	2.10^{-5}	mg/kg/d		
Set 2 Zeilmaker *et al.* (2018)				
TDI oral	$6.25.10^{-6}$	mg/kg/d	38	75
TCA inhalation	$21.9.10^{-6}$	mg/m³		
TDI dermal	$6.25.10^{-6}$	mg/kg/d		
Set 3 EFSA (2018b)				
TDI oral	$1.8.10^{-6}$	mg/kg/d	11	4.8
TCA inhalation	$6.3.10^{-6}$	mg/m³		
TDI dermal	$1.8.10^{-6}$	mg/kg/d		

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LIST OF ABBREVIATIONS

BCF	Bioconcentration factor
SRV	Soil remediation value
CL	Clearance
CLP	Classification, Labeling and Packaging – Regulation (EC) Nr. 1272/2008 on classification, labelling and packaging of substances and mixtures
CONTAM	Contaminants in the food chain
d	Day
Da	Diffusion for organic substance in air
DNEL	Derived no effect level
Dpe	Diffusion value (permeation) through polyethylene drinking water pipe
Dpvc	Diffusion value (permeation) through PVC drinking water pipe
dm	Dry matter
Dw	Diffusion for organic substance in water
EFSA	European Food Safety Authority
$SR_{eco\ soil}$	Serious Risk for the soil ecosystem
EU	European Union
GAPS	Global Atmospheric Passive Sampling
HC_{50}	Hazardous concentration for 50% of the soil organisms
HBGV	Health Based Guidance Value
HED	Human Equivalent Dose
HHSV	Human Health Screening Value
IARC	International Agency for Research on Cancer
K_d	Soil/water partition coefficient
K_{oa}	Octanol-air partition coefficient
K_{oc}	Organic carbon-water partition coefficient
K_{ow}	Octanol-water partition coefficient
LB	Lower bound
LC50	Lethal concentration for 50% of organisms
LD50	Lethal dose for 50% of organisms
bw	bodyweight
LOAEL	Lowest Observed Adverse Effect Level
LOD	Limit of Detection
LOQ	Limit of Quantification
MRL	Minimal Risk Level
MTR	Maximum Tolerable Risk
NECC	Nutrient and Energy Cycling Check value
N(L)OEC	No (Lowest)-Effect-Concentration
NOAEL	Lowest Observed Adverse Effect Level
NTP	National Toxicology Program
OC	Organic carbon
PBPK	Physiologically based pharmacokinetic
PFAS	Poly- and perfluoroalkyl substances
PFCA	Perfluorinated carboxylic acids
PFOA	Perfluorooctanoic acid
PFOS	Perfluorooctane sulfonic acid
PFSA	perfluoro-alkyl sulfonic acids
POPs	Persistent Organic Pollutants
RfD	Reference dose

SQG_E	Environmental Soil Quality Guideline
T3	Triiodothyronine
TCA	Tolerable concentration in air
TDI	Tolerable Daily Intake
TSH	Thyroid Stimulating Hormone
TWI	Tolerable Weekly Intake
UB	Upper bound
UDS	Unscheduled DNA-synthesis
UKCOT	United Kingdom Committee on Toxicity of Chemicals in Food, Consumer Products and the Environment
US-EPA	United States of America – Environmental Protection Agency
USA	United States of America

CHAPTER 1. INTRODUCTION

1.1. BACKGROUND

The present report describes the derivation of proposals for soil remediation values (SRV) for:

- Perfluorooctane sulfonic acid (PFOS),
- Perfluorooctanoic acid (PFOA)

The present report discusses in detail the elements of behaviour in soil and physicochemical properties, occurrence in the environment, transfer to plants and animals, toxicology and legal limits. The information is summarised in substance sheets at the end of this report, which serve as a basis for entering the data into the S-Risk model. Via the S-Risk model (www.s-risk.be), proposals for soil remediation values are calculated for the solid phase of the soil. The collected information follows the guidelines set out in Cornelis and Touchant (2016).

1.2. APPROACH

1.2.1. PHYSIOCHEMICAL PROPERTIES

The following sources were consulted: EFSA (2008b), CONCAWE (2016), FSANZ (2017), Moermond *et al.* (2010), Lijzen *et al.* (2018), Pancras (2018), Wintersen (2019) and publications from scientific journals.

1.2.2. OCCURRENCE IN THE ENVIRONMENT

The discussion on the occurrence of PFOS and PFOA in the environment is limited to the environmental compartments and data needed to determine background concentrations and background exposure. This concerns air, drinking water and food.

Background concentrations in indoor air are equal to background concentrations in outdoor air. Possible higher concentrations in indoor air due to indoor sources and the resulting exposure are not included in the background exposure.

For background exposure via food, information for PFOS and PFOA for Belgium is available in Cornelis *et al.* (2009), EFSA (2012), EFSA (2018c) and Klenow *et al.* (2013) .

Concentrations in drinking water are available in various studies (Cornelis *et al.*, 2009; D'Hollander *et al.*, 2009). If drinking water is included in the selected intake study, the background concentration in tap water will be set to zero for deriving the soil remediation value, to avoid double counting.

1.2.3. TOXICOLOGY

The overview of toxicology is primarily based on data from reviews and discussion papers. The chapter on Toxicology provides a description of toxicokinetics and a brief description of the main toxicological effects. The toxicological reference values derived by known bodies are thoroughly evaluated and finally, on the basis of this evaluation, a reference value for deriving the soil remediation value is selected.

1.2.4. CALCULATIONS

The calculations of the proposals for the SRV for the solid phase (soil) have been made with S-Risk version 1.3. Application I was used to calculate the proposals, application II to interpret the results to exposure routes and exposure pathways.
The model concept for the calculation of human health based soil remediation values assumes a homogeneous soil profile, both with regard to soil properties as with regard to contamination. The default land use types with their corresponding scenarios are:

- agricultural land use (type II)
- residential land use (type III)
- recreational land use (day recreation – type IVa and holiday resort – type IVb)
- industrial land use (light industry – type Va and heavy industry – type Vb).

Calculations are made until the soil concentration corresponds to an exposure which results in a risk index equal to 1 for the non-carcinogenic endpoints and/or to an additional lifetime cancer risk equal to $1/10^5$ for the carcinogenic endpoints. The use of pseudo-threshold simulations does not apply to PFOS and PFOA. Calculations are also carried out until the soil concentration corresponds to a concentration in foodstuffs (Type II only), outdoor air, indoor air and drinking water, equal to the limit for these compartments. A legal limit, if available, is always used for foodstuffs. For air and drinking water, the starting point is that a legal limit is used if it exists for Flanders/Belgium. If there is no legal limit, a toxicological limit is used.

The calculation of the SRV proposals for groundwater has been made as described in Cornelis and Touchant (2016). If there is a legal value for drinking water quality, this was included in the proposal for a soil remediation value.

1.2.5. ECOTOXICOLOGICAL REFERENCE VALUES

For the evaluation of ecotoxicological effects, no new primary sources and/or databases were consulted to derive possible new ecotoxicological values. However, it was examined whether substantiated ecotoxicological values have recently been derived by other bodies.

1.2.6. INTEGRATION AND EVALUATION OF THE PROPOSALS FOR VALUES

In deriving a soil remediation value, a number of elements are considered, as described in Cornelis and Touchant (2016). This includes a comparison between human health based and ecotoxicological underpinned soil remediation values, an evaluation of the analytical feasibility, a comparison with target values (are the values analytically distinguishable and higher than the target values), a calculation of guideline values and their feasibility, and finally the impact on current policy.

1.3. READER'S GUIDE

This report discusses the proposals for soil remediation values for PFOS (CHAPTER 2) and PFOA (CHAPTER 3). Each substance has a separate chapter in which the information needed to derive soil remediation values is collected and discussed (paragraph 2.1 to 2.9 for PFOS and paragraph 3.1 to 3.9 for PFOA). The proposals for SRV are derived in paragraphs 2.10 and 3.10 respectively and compared with foreign values in 2.12 and 3.12.
A section on the integration of the different values (human health based proposal, ecotoxicological proposal, target values, guideline values, impact in relation to the current framework) is also provided in paragraphs 2.11 and 3.11. The annexes contain a substance sheet for each substance, see substance sheet PFOS and substance sheet PFOA, which contains the import data into S-Risk for deriving the proposed SRV.

CHAPTER 2. PFOS

2.1. IDENTIFICATION

PFOS	
name English	Perfluorooctane sulfonic acid
name Dutch	Perfluoroctaansulfonzuur
CAS number:	1763-23-1 (potassium salt: 2795-39-3)
EINECS number:	217-179-8
EC index number:	607-624-00-8
formula:	$C_8HF_{17}O_3S$
molecular weight:	500,126 g/mole
conversion:	1 ppm = 20.79 mg/m³; 1 mg/m³ = 0,05 ppm (25°C, 1 atm) (ATSDR draft (2018))

2.2. SOURCES OF PFOS

PFOS does not occur naturally in the environment. Long-chain perfluorinated substances (such as PFOS) are (or have been) used as surfactants in various applications. Historical applications of PFOS include inks, varnishes, waxes, extinguishing foams, coatings, lubricants, water and oil repellents for leather, paper and textiles, detergents and carpet cleaners, shampoos and hand creams, hydraulic fluids for aircraft, insecticides, and surface treatment and cleaning of metals. Precursors (e.g. fluorinated telomeres) can be a source of PFOS, as well as side-chain fluorinated polymers (CONCAWE, 2016).

2.3. BEHAVIOUR IN SOIL AND PHYSIOCHEMICAL PROPERTIES

2.3.1. FATE OF PFOS IN SOIL

The information on behaviour in soil is primarily derived from CONCAWE (2016). PFOS belongs to the group of perfluoroalkyl sulfonic acids (PFSA) within the large group of perfluoroalkyl substances (PFAS); from the viewpoint of behaviour and distribution, PFSA form a homogeneous group. The properties of the group therefore also apply to PFOS, although for certain properties there may be quantitative trends determined by chain length.

→ **Chemical form**

Under typical environmental conditions, most PFAS and their salts occur as solids. The relevant form of PFSA for the environment (soil, groundwater and surface water with a normal pH of 5-9) is the

anion. The formation of anions is accompanied by a decrease in adsorption to soil and sediment as they are usually net negatively charged. The speed of transport through soil or sediment decreases with a longer perfluorinated C chain and with an increasing content of organic carbon (OC) in the soil. PFSAs bind more strongly than perfluorinated carboxylic acids (PFCA, e.g. PFOA) with the same number of C atoms. PFAS (with the exception of telomere alcohols which have a hydroxyl function) are surfactants with a hydrophobic perfluorinated C chain and a hydrophilic functional group (e.g. sulphate or carboxyl). Unlike ordinary surfactants, the hydrophobic perfluorinated C chain of PFAS also has hydrophilic properties, making PFAS coatings resistant not only to water but also to oil and grease. The surface activity of PFAS is stronger than that of similar, ordinary surfactants. On the one hand, PFAS can settle at the interface of different phases, for example groundwater (hydrophilic) and soil air (hydrophobic), and on the other hand micelles can form in solution.

→ **Distribution**

PFSA are widely distributed in the environment due to their high solubility in water, low to moderate sorption to soil and sediment, and resistance to biological and chemical degradation. PFAS have a low vapour pressure, meaning that transport in the vapour phase only plays a minimal role. The Henry coefficients of PFAS are highly varied. The Henry coefficient of PFOS is negligible and shows that PFOS will distribute little from water to the air. The degree of transport of PFAS via water is influenced by the degree of adsorption to sediment or soil during that transport; the higher the adsorption, the more the transport of PFAS via the aqueous phase is retarded. There are two sorption mechanisms that control the degree of adsorption:

- hydrophobic sorption of solid organic particles, and
- sorption on the surface of charged mineral surfaces.

The parameters that measure the sorption of solid organic C particles are the organic carbon partition coefficient (K_{oc}) and the solid/liquid partition coefficient (Kd). The octanol-water partition coefficient (K_{ow}) is not a suitable parameter for adsorption because it is difficult to measure due to the cationic and anionic charge of PFAS (PFAS do not have normal lipophilic behaviour). These PFSA that are strong acids occur almost exclusively as anions; they can adsorb to the charged mineral surfaces present in the soil or sediment, thus influencing the transport of PFAS through water. To demonstrate this possible mechanism, several experiments have been described in the literature, but the degree of adsorption or the impact on transport has not yet been quantified.

→ **Transformation**

The C-F covalent bond is one of the strongest bonds in organic chemistry. PFAS therefore have a high thermal, chemical, photolytic and biological stability. There is no indication that PFSA would undergo biotransformation or photolysis under normal environmental conditions. The half-life for hydrolysis is ≥ 41 years, the half-life for photolysis > 3.7 years (OECD, 2002). Under aerobic conditions with activated sludge, no removal or biotransformation has been measured for PFOS. Some removal of PFOS has been measured under anaerobic conditions, but without formation of metabolites or increase in fluoride. There are no tests demonstrating significant or complete degradation of PFOS under environmental conditions. Due to the strong C-F bond, PFOS is persistent in the environment. Under natural conditions, precursors (alcohol telomeres) can convert to PFSA.

2.3.2. PHYSIOCHEMICAL PROPERTIES

PFOS is an organic solid with a density of 0.52-0.57 g/cm³. PFOS is generally used as salt (K, Na, ammonium) or incorporated in polymers (EFSA, 2008b). The potassium salt of PFOS is also a solid. K-PFOS has a melting point of 54°C and is lighter than water (density ≈ 0.6).

The physicochemical properties of PFOS and PFOS salts are listed in
Table 1 and are discussed below.
PFOS is an acid and only occurs naturally in ionised form (Moermond *et al.*, 2010). For this reason, in the context of this report, a number of physicochemical properties of PFOS salts are relevant for PFOS. Some parameters have been estimated using the EpiSuite modelling platform of the US EPA for comparison with measured values (if available); it should be noted that the estimated values may not accurately reflect actual properties as there are no perfluor structures in the EpiSuite training set (Lijzen *et al.*, 2018).

→ Water solubility (S)

The solubility in pure water is approximately 550 mg/l at 24-25°C for PFOS (OECD, 2002) and 519 mg/l at 20°C (Beach *et al.*, 2006) and 570 mg/l at 24-25°C (OECD, 2002; Deng *et al.*, 2012) for K-PFOS. PFOS dissociates at neutral pH in ions; the complex formation of PFOS anions in non-pure water causes its solubility to decrease significantly with the salinity; e.g. K-PFOS has a solubility of 370 mg/l in fresh water, 25 mg/l in filtered seawater and 12.4 mg/l (22-23°C) in unfiltered seawater (OECD, 2002). For the calculations of the soil remediation values, a solubility of 370 mg/l is assumed, as this is measured in fresh water, which is more realistic than solubility in pure water within the normative framework. The value of 370 mg/l is given in OECD (2002) with reference to a 3M report from 1999, without mention of temperature. The OECD test protocol for solubility (OECD test guideline 105) states that the test should preferably be carried out at 20 ± 0.5°C. For deriving soil remediation values, we assume that 3M has followed the test protocol, and 20°C is taken. Wintersen *et al.* (2019) states 276 mg/l, this is 370 mg/l converted to 10°C for use in CSOIL.

→ Acid dissociation constant (pKa)

PFOS is a strong acid that dissociates in the environment. The pKa is -3.27 (EFSA, 2008b; Moermond *et al.*, 2010), which means that the substance occurs in the environment (pH 6-8) almost exclusively as the anion. The surfactant character of PFOS can inhibit dissociation in water (ATSDR draft, 2018). A pKa of -3.27 is assumed for the calculations of the soil remediation values.

→ Vapour pressure (Vp)

PFOS has a vapour pressure of 6.7 Pa (Pancras *et al.*, 2018); this is sufficiently high to allow some of the PFOS to evaporate (ITRC, 2018). According to Pancras *et al.* (2018) the quantity of PFOS that will volatilise from water to the gaseous phase is practically negligible and volatilisation from water is therefore not considered to be a relevant transport mechanism. (Pancras *et al.*, 2018). The anion of PFOS has a negligible vapour pressure (ITRC, 2018). The vapour pressure of the K-salt is indeed much lower ($3.31.10^{-4}$ Pa at 20°C; OECD (2002)) than that of PFOS. PFOS occurs in the environment almost exclusively as an anion, whereby volatilisation of PFOS is minimal.

For the calculations of the soil remediation values, the vapour pressure of the anion, i.e. $3.31.10^{-4}$ Pa at 20°C is assumed because PFOS occurs in the environment almost exclusively as the anion.

→ Henry coefficient (H)

The Henry coefficient is the ratio between vapour pressure and solubility. The Henry coefficient of PFOS is very low and varies between $<2.10^{-6}$ and $3.09.10^{-4}$ Pa m³/mol (Pancras *et al.*, 2018). The value $3.09.10^{-4}$ Pa m³/mol is calculated for the K-salt in pure water and therefore actually for the anion. The Henry coefficient for K-salt in fresh water is $4.40.10^{-2}$ Pa m³/mol (OECD, 2002). The value calculated for PFOS with HenryWin (EpiSuite) is in-between and amounts to $1.11.10^{-3}$ Pa m³/mol. Since PFOS dissociates in water and primarily occurs as the anion, the Henry coefficient of the K-salt is relevant for PFOS. For the calculations of the soil remediation values, the Henry coefficient is therefore calculated by S-Risk, based on the solubility (in fresh water) and the vapour pressure of the anion.

→ Octanol water partition coefficient (log K_{ow})

The log K_{ow} of surfactants cannot be measured according to the OECD standard test guideline because the substance accumulates in a mixture of octanol and water at the interface instead of in the fluids (Pancras *et al.*, 2018). The hydrophilic side is in the water while the hydrophobic side faces the octanol. The log K_{ow} can be estimated from the solubility in octanol and water, the log K_{ow} of PFOS thus calculated is -1.08 (OECD (2002) in Beach *et al.* (2006)). The log K_{ow} calculated with the COSMOtherm model[2] is several orders of magnitude higher and amounts to 6.43 (Wang *et al.*, 2011). The log K_{ow}, estimated with EpiSuite (K_{ow}Win), is 4.49 and is calculated from the molecular weight, a water solubility of 0.1039 mg/l at 25°C and a correction factor.
For organic substances the log K_{ow} can be used to estimate the bioconcentration. In the case of PFOS, bioconcentration cannot be estimated using conventional algorithms because PFOS does not bioaccumulate in fats but binds to certain proteins (Jones *et al.* (2003) in Beach *et al.* (2006)). The use of water solubility or calculated log K_{ow} values may underestimate the accumulation of PFOS in organisms (Beach et al., 2006). The fact that the log K_{ow} is uncertain also means that it is better to determine the uptake in crops on the basis of empirical data (Lijzen et al., 2018).
In S-Risk the log K_{ow} is used to calculate Kp, K_{oc}, and transfer factors, unless an experimental value is entered. Experimental values are available for these three parameters. S-Risk will therefore not have to deal with log K_{ow}. S-Risk does however require the input of a value for log K_{ow}. The calculated value 4.49 of EpiSuite is entered in S-Risk, because the algorithm of this model is transparent while the algorithm of the commercial COSMOtherm is not freely available. S-Risk does not use this log K_{ow} anywhere in its calculations for PFOS.

→ Organic carbon-water partition coefficient (log K_{oc})

The K_{oc} is a measure for soil adsorption. For organic substances the log K_{oc} can be calculated from the log K_{ow}. However, for surfactants the log K_{ow} is not a good indicator for adsorption to the soil because the log K_{ow} cannot be measured (accurately). The use of water solubility or calculated log K_{ow} values may underestimate the accumulation of PFOS in the soil (Beach *et al.*, 2006).

[2] http://www.cosmologic.de/products/cosmotherm.html

PFOS slightly adsorbs to the soil and is moderately mobile to immobile, but due to its sufficient solubility in water, PFOS can still distribute fairly easily in the environment. The adsorption of PFOS to organic matter is higher than that of PFOA (Pancras *et al.*, 2018). The experimental log K_{oc} values for soil depend on the soil type and vary from 2.4 l/kg for kaolinite to 2.6 for clay loam, 2.8 for clay and 3.1 for sandy loam (Table 1). The log K_{oc}, determined by adsorption measured in six soils with different OC content, was 2.85 (Milinovic *et al.*, 2015). The adsorption of PFOS was measured in sediment at several locations in the Netherlands (Kwadijk *et al.*, 2010). The average log K_{oc} for 19 samples was quite high (3.16±0.28) compared to other published results. The log K_{oc}, calculated with concentrations in soil and water, is 3.7 (Zareitalabad *et al.*, 2013). The log K_{oc} values for freshwater sediment range from 2.4 to 3.8; the K_{oc} for saltwater sediment is higher (4.7).
When measuring the adsorption (log K_{oc}) in three different sediments (Japan), by Ahrens *et al.* (2011), the researchers found that the OC content had a significant influence on the distribution; for the sediment with the lowest OC content, the density of the sediment was the most important factor influencing the distribution. The log Kd also depends on the type of soil (
Table 1). 3M has measured values from 0.99 l/kg for clay loam to 1.26 for clay and 1.55 for sandy loam; the Freundlich coefficients are 14.0, 25.1 and 28.2 respectively (3M-Company, 2000). Log Kd values for the anion are in the same order of magnitude (0.87-1.55). The log Kd values for PFOS, which were measured by Johnson *et al.* (2007) are higher (2.81-8.90) than those of 3M (
Table 1). Johnson *et al.* (2007) calculated the adsorption for four different soils. The adsorption increased for the various materials in the following order (sorption normalised for surface): synthetic goethite (iron oxide mineral) < kaolinite (clay) < high iron sand < standard sand. This upward trend in adsorption was accompanied by a downward trend in surface reactivity (Zareitalabad *et al.*, 2013). Organic carbon appears to play an important role in adsorption while electrostatic attraction may play a role in low-carbon soils (Johnson *et al.*, 2007). The adsorption behaviour of PFOS was measured in six soils with different characteristics, mainly in terms of organic carbon content (Milinovic *et al.*, 2015). The log Kd rose from 1.28 to 2.47 (Kd 19 to 295) l/kg; the corresponding Freundlich coefficients rose from 17 to 349 l/kg. The adsorption was positively correlated with the organic carbon content. Li *et al*. (2018) who carefully analysed this correlation, claim that the correlation was strongly influenced by one sediment with a high OC content; they removed this value, resulting in a weaker correlation. The desorption measured by Milinovic (2015) was less than 13%. According to the authors, the sorption behaviour is primarily determined by the hydrophobia (Milinovic *et al.*, 2015).
The average log Kd of 19 sediments from different locations in the Netherlands is 2.353±0.35 (Kwadijk *et al.*, 2010). Other measured log Kd values for PFOS in river and lake sediment are 0.87 and 7.52 respectively. Log Kd values for adsorption of the anion in sediment range from 0.30 to 1.04.
On behalf of OVAM, Kd values were calculated for soils contaminated with PFAS. To this end, OVAM selected two industrial sites, each with an old and a new fire drill site, the top layer of which was sampled by OVAM (2018). Four samples were subjected to shaking tests and the Kd was calculated as the ratio between total concentration and eluate concentration. It was assumed that the concentrations in solution after the shake test were in balance with the solid phase. The calculated Kd values were respectively 14.8 and 7.0 l/kg for site 1 and 27.1 and 18.1 l/kg for site 2. The median Kd for both sites was 16.4 l/kg. These measurements should be regarded as indicative and cannot be generalised to a general value for Flanders.
In general, anions would not attach more weight to organic carbon than their neutral counterpart (HSDB)[9].
For the calculations of the soil remediation values the log K_{oc} of 2.57 from Higgins and Luthy (2006) is used. This value is one of the lower experimental K_{oc} values in

Table 1 (median 2.85), and thus worst-case (the lower the K_{oc}, the more mobile the substance). EFSA (2008b) and Lijzen *et al.* (2011) also used the log K_{oc} value of Higgins and Luthy (2006) in their evaluations of PFOS.
The Kd is calculated by S-Risk from the K_{oc} with the formula Kd = OC(organic C content) x K_{oc}.

Note: The Dutch expertise centre for PFAS is of the opinion that the relationship between organic carbon and adsorption is less clear for PFAS than for other organic substances, due to the surfactant behaviour of PFAS. It is therefore not straightforward to correct, for PFAS, the intervention value for the soil organic matter content, as is common practice in the soil remediation values system for organic contaminants (Alphenaar *et al.*, 2018). Using published data, Li et al (2018) evaluated the role of organic carbon and other properties in the adsorption of PFAS in soil and sediment. The authors found weak correlations between Kd and only OC, for PFOS the correlation coefficient (R^2) was = 0.05 for 178 samples. For pH alone, the correlation with Kd was also weak, with a R^2 of 0.06 for PFOS over a pH interval of 2.5 to 8.5 (n = 27). Using multiple regression models, it was shown that at least the OC, pH and clay content have a significant effect on the sorption. For PFOS, the R^2 for these three parameters together rose to 0.77.

→ Octanol-air partition coefficient (K_{oa})

The log K_{oa} calculated with the COSMOtherm model is 8.07 g/g (Wang *et al.*, 2011); the log K_{oa} calculated with K_{oa}Win (EpiSuite) is almost 50% lower (4.84 g/g). The calculation formula of EpiSuite uses log K_{ow}. S-Risk also calculates log K_{oa} based on log K_{ow}. However, for the reasons discussed above, using log K_{ow} is avoided as much as possible for the soil remediation values derivation of PFOS, to reduce the uncertainty within the modelling of the soil remediation value. COSMOtherm is a commercial model and the underlying formula for calculating the K_{oa} is not freely available. In S-Risk, K_{oa} is used in the calculation of transfer to plants; as experimental data are available for this purpose, the import of a K_{oa} is not necessary. Entering a K_{oa} value is optional in S-Risk. For the calculations of the soil remediation values it is therefore not necessary to select a log K_{oa}.

→ Permeation through drinking water pipes (Dpe, Dpvc)

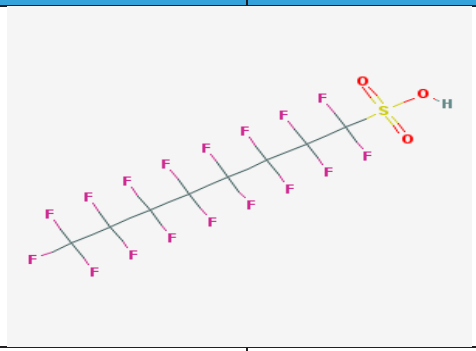
No values were found for permeation (diffusion) of PFOS through polyethylene (Dpe) or PVC (Dpvc) drinking water pipes. In the Netherlands, for the Dpe of PFOA, the default value of 1.10^{-7} m²/d is calculated in the CSOIL model, and this is accounted for as follows: *For the common contaminants, Dpe is in the range 0.10-35.10^{-7} m^2/day (Vonk, 1985). In the absence of data, it is recommended to use the permeation coefficient of a substance with a similar structure (van den Berg, 1997). Failing this, the calculation is made with a default value of 1.10^{-7} m^2d, which is also used in other compounds* (Lijzen *et al.*, 2018). This is the size of the diffusion coefficient for small compounds; the diffusion coefficient decreases with the size of the molecule (Vonk, 1985). Like PFOA, PFOS is a large and long molecule, so the value 1.10^{-7} m²/d will probably be sufficiently conservative for PFOS as well.
By default, the value of Dpvc = Dpe/1000, according to the technical guidance document of S-Risk (Cornelis *et al.*, 2017).
For the calculations of the soil remediation values, it is assumed that the Dpe for PFOS is equal to that of PFOA. The value of 1.10^{-7} m²/d is used; this is the standard value used by Lijzen *et al.* (2018) for calculations for PFOA in CSOIL. For Dpvc, the value 1.10^{-10} m²/d (this is Dpe/1000) is assumed. In its most recent report on risk limits for PFOA, RIVM states a Dpe of 1.10^{-7} m²/d in Table 3.1 (referring to previous RIVM reports) and of $3.15.10^{-10}$ m²/d in Annex 3 (input data for CSOIL); no reference or

calculation method is given for the latter value (Wintersen, 2019). As such, we prefer to follow the Dpe with the justification of Lijzen (2018).

→ **Diffusion for organic substance in air (Da) and water (Dw)**

These values are used to calculate the diffusion when evaporation to outdoor and indoor air occurs. Entry in S-Risk is optional. No values were found for diffusion of PFOS in air or water. Therefore, for the calculations of the soil remediation values, both values are calculated in S-Risk, starting from the molecular weight, as specified in the technical guidance document (Cornelis *et al.*, 2017).

Table 1: Physicochemical properties of PFOS, the values used for the calculations are indicated in bold.

Parameter	Unit	Value	Original reference	Reference
Chemical structure				PubChem[3]
Type		Organic		
Physical state		solid (powder) (K-salt)		EFSA (2008b)
Solubility in water	mg/l	PFOS 550 mg/l (24-25°C) K-PFOS: 519 mg/l (20 ± 0.5°C) in pure water; **370 mg/l** in fresh water	OECD (2002)	EFSA (2008b); Lijzen *et al.* (2011)
		570 mg/l		Deng *et al.* (2012)
		520 – 570 mg/l (20 – 25°C)		Pancras *et al.* (2018)
		12.4 mg/l in sea water (K-salt)	3M-Company (2001)	OECD (2002); Beach *et al.* (2006)
Melting point		54°C (K-salt)	OECD (2002)	EFSA (2008b)

[3] https://pubchem.ncbi.nlm.nih.gov/

Density		0.52-0.57		CONCAWE (2016)
		~0.6 (K-salt)		FSANZ (2016)
Vapour pressure	Pa	**$3.31.10^{-4}$** (K-salt) (20°C)	OECD (2002)	EFSA (2008b); Moermond *et al.* (2010)
		6.7 Pa		Pancras *et al.* (2018)
Henry coefficient[4] (Henry constant)	Pa m³/mol	$<2.10^{-6} – 3.10^{-4}$		Pancras *et al.* (2018)
		$3.09.10^{-4}$ Pa m³/mol ($3,05.10^{-9}$ atm.m³/mol) in pure water (K-salt), calculated value	OECD (2002)	EFSA (2008b)
		$4.40.10^{-2}$ Pa m³/mol in fresh water (K-salt), calculated value		OECD (2002)
		$1.11.10^{-3}$ calculated value		EpiSuite[5] Henrywin
Log K_{ow}[6]	g/g	**Not measurable**		EFSA (2008b)
		-1.08 (calculated from solubility in octanol and water)	OECD (2002)	Beach *et al.* (2006)
		6.43 (estimated value)	Wang *et al.* (2011)	Pancras *et al.* (2018)
		4.49 (estimated value)		EpiSuite K_{ow}Win
Log K_{oc}	dm³/kg	2.6 (clay loam), 2.8 (clay and river sediment) 3.1 (sandy loam)	3M-Company (2000)	Johnson *et al.* (2007); Wang *et al.* (2011); Pancras *et al.* (2018); Zareitalabad *et al.* (2013)

[4] Calculated in S-Risk
[5] https://www.epa.gov/tsca-screening-tools/epi-suitetm-estimation-program-interface
[6] Entered in S-Risk but not used in further calculations

		2.4 (kaolinite), 2.4-2.6 (lake sediment)	Johnson *et al.* (2007)	Zareitalabad *et al.* (2013)
		3.7 (calculated on the basis of measured concentrations in soil and water)	Zareitalabad *et al.* (2013)	
		3.0 (river sediment, sandy, f_{oc} 0.03%) 3.8 (river sediment, muddy, f_{oc} 1.6%) 4.7 (marine sediment, muddy, f_{oc} 1.1%) (normalised for OC content)	Ahrens *et al.* (2011)	Zareitalabad *et al.* (2013)
		2.57 ± 0.13 (anion) (sediment) (log normal average log K_{oc}) (n=4)	Higgins and Luthy (2006)	EFSA (2008b); Lijzen *et al.* (2011)
		2.68 ± 0.09 (anion) (sediment) (regression log K_{oc}) (n=4)	Higgins and Luthy (2006)	
		2.85 (K_{oc} 710) (OC 0.2-39%)	Milinovic *et al.* (2015)	
		3.16 (field data, average of 19 sediment samples)	Kwadijk *et al.* (2010)	
Log K_d[4]	l/kg	0.87-1.55 (Kd 7.41-35.5) (anion)	Beach *et al.* (2006)	EFSA (2008b)
		2.81 (standard sand Ottawa), 5.31 (kaolinite), 7.88 (synthetic goethite), 8.90 (sand with high iron content) 7.52 (lake sediment)	Johnson *et al.* (2007)	Zareitalabad *et al.* (2013)
		1.26 (Kd 18.3) (clay); 0.99 (Kd 9.72) (clay loam); 1.55 (Kd 35.3) (sandy loam); 0.87 (Kd 7.42) (river sediment) Average values	3M-Company (2000)	Zareitalabad *et al.* (2013)
		-1 – 1.99 (Kd 0.1 – 97) (pH 7)		Pancras *et al.* (2018)
		0.30 – 1.04 (sediment) (anion)	de Voogt *et al.* (2006)	EFSA (2008b)
		1.28 (Kd 19) (OC 0.2%)	Milinovic *et al.* (2015)	

		1.51 (Kd 32) (OC 1.6%) 1.58 (Kd 38) (OC 3.9%) 1.88 (Kd 76) (OC 7.7%) 2.04 (Kd 110) (OC 9.4%) 2.47 (Kd 295) (OC 39%)		
		2.35 (1.70-3.04) (field data, sediment, n=19)	Kwadijk *et al.* (2010)	
		1.21 (Kd 16.4) (median for 2 fire drill sites contaminated with PFAS)	This study Indicative value	
$K_{ads}F$ (Freundlich coefficient)	l/kg[7]	25.1 (clay); 14.0 (clay loam); 28.2 (sandy loam); 8.70 (river sediment)	3M-Company (2000)	Beach *et al.* (2006)
		17 (OC 0.2%) 41 (OC 1.6%) 61 (OC 3.9%) 86 (OC 7.7%) 157 (OC 9.4%) 389 (OC 39%)	Milinovic *et al.* (2015)	
Log K_{oa}[8]	g/g	4.837 (calculated)		EpiSuite[5] K_{oa}Win v1.10
		8.07 (calculated)	Wang *et al.* (2011)	
Dissociative		Yes, PFOS is a strong acid and dissociates at neutral pH.		Beach *et al.* (2006)
Acid constant (pKa)		<1.0 (anion PFOSA)	Cheng *et al.* (2009)	HSDB[9]
		-3.27 (calculated value)	Brooke *et al.* (2004)	EFSA (2008b); Moermond *et al.* (2010)
		0.14 (estimated value)		ATSDR draft (2018)

[7] This unit assumes that n=1; the exact unit is $\mu g^{1-1/n}(l)^{1/n}kg^{-1}$

[8] Log Koa is optional in S-Risk, which uses Koa in the calculation of transfer to plants; as experimental data are available for this purpose, a Koa value is not necessary.

[9] https://toxnet.nlm.nih.gov/cgi-bin/sis/search2/r?dbs+hsdb:@term+@rn+@rel+1763-23-1

		-6.0 to -2.6		Pancras *et al.* (2018)
		-3.41 (calculated value)	Wang *et al.* (2011)	
Dpe	m^2/d	No substance-specific data		
		1.10^{-7} (standard value)		Based on Vonk (1985) and Lijzen *et al.* (2011)
Dpvc	m^2/d	**1.10^{-10}** (Dpe/1000)		Cornelis *et al.* (2017)
Da	m^2/d	No data		
Dw	m^2/d	No data		

2.4. OCCURRENCE IN THE ENVIRONMENT

2.4.1. SOIL

PFOS does not occur naturally in the soil. PFOS is persistent (see Beach (2016) for tests of degradability in water and soil).

2.4.2. AIR

→ Outdoor air

There are no data on perfluorinated compounds in the air in Flanders.
In the project BF-Risk (Cornelis *et al.*, 2009) the outdoor air concentration was derived from a number of European studies (Barber *et al.*, 2007; Jahnke *et al.*, 2007a; Dreyer & Ebinghaus, 2009). For outdoor air, the authors used the available measurement data and processed it statistically. This resulted in a P50 of $1.4.10^{-9}$ mg/m³ and a P95 of $4.60.10^{-8}$ mg/m³ for PFOS. Based on European literature, EFSA (2008b) defined two scenarios for low and high exposure, with outdoor air concentrations of $1.00.10^{-9}$ mg/m³ and $1.00.10^{-8}$ mg/m³ respectively, which are of the same order of magnitude as those used in BF-Risk.
In the GAPS (Global Atmospheric Passive Sampling) network, POPs, including PFOS are measured at 21 different places on earth. Rauert *et al.* (2018) compared the concentrations for 2009, 2013 and 2015. Samples were taken in 3 types of areas: background, urban and polar. In Europe, samples were taken in the Czech Republic (background), Norway (polar), Ireland (background) and Paris (urban). In 2015, the background concentrations for PFOS (across all background measuring stations) were $<6.00.10^{-11}$ – $2.30.10^{-8}$ mg/m³, in 2013 they were $<4.00.10^{-11}$ – $7.00.10^{-9}$ mg/m³ and in 2009 they were $<4.00.10^{-11}$ – $3.50.10^{-8}$ mg/m³. The authors also refer to a study in Switzerland where a background concentration of $1.7.10^{-9}$ mg/m³ was calculated (Muller *et al.*, 2012). In 2015, for urban areas the concentrations of PFOS were respectively $4.00.10^{-9}$ – $6.40.10^{-9}$ mg/m³, in 2013 they were $1.70.10^{-9}$ – $3.30.10^{-9}$ mg/m³ and in 2009 they were $1.20.10^{-9}$ – $5.40.10^{-8}$ mg/m³. Muller *et al.* (2012) calculated $2.30.10^{-9}$ mg/m³ for the urban area.
For deriving the soil remediation value for PFOS, we use a concentration of $1.4.10^{-9}$ mg/m³ PFOS in outdoor air (P50 value from Cornelis *et al.* (2009)). This value falls within the ranges for background concentrations calculated in the GAPS network and is of the same order of magnitude as the lowest values for urban areas. In addition, it is similar to the background and urban concentrations calculated by Muller *et al.* (2012).

→ Indoor air

Higher concentrations can be found in indoor air than in outdoor air, due to indoor sources. In the BF-Risk project a concentration of $1.6.10^{-9}$ mg/m³ was assumed for indoor air based on data from

Jahnke *et al.* (2007b). **For deriving soil remediation values, the $1.6.10^{-9}$ mg/m³ background concentration in indoor air was used.**

2.4.3. DRINKING WATER

There are no VMM (Flanders Environment Agency) measurement data for PFOS in drinking water. In the BF-Risk project, 4 samples of tap water from 3 different drinking water companies were analysed. The measured concentrations of PFOS were $2.6.10^{-3}$ - $10.6.10^{-3}$ µg/l with a median of $3.4.10^{-3}$ µg/l and an average concentration of $5.22.10^{-3}$ µg/l, the latter value was used for the intake estimation in the BF-Risk project (Cornelis *et al.*, 2009; D'Hollander *et al.*, 2009).
Costopoulou *et al.* (2015) investigated 11 PFAS in tap water samples from Greece and the Netherlands. For the Dutch samples, PFAS were detected in 49% of the samples.
PFOS above the limit of detection of $0.6\ .10^{-3}$ µg/l was only found in 2 samples (3 and 5.10^{-3} µg/l). Taking both groundwater and surface water into account, an average concentration of 0.2 (lower bound[10]) and $0.8.10^{-3}$ µg/l (upper bound) respectively was recorded for PFOS in Dutch tap water.
EFSA (2012) calculated an average concentration of $3.90.10^{-3}$ µg/l (upper bound results) based on 156 samples of tap water from different European countries over the period 2006-2012. For bottled water (255 samples) the concentration was slightly lower, $1.70.10^{-3}$ µg/l PFOS (upper bound results). The results obtained in BF-Risk for tap water are slightly higher than those obtained by EFSA or for the Netherlands. Due to the limited amount of data (4 samples), it was decided to use the concentration obtained by EFSA, i.e. $3.90.10^{-3}$ µg/l. Depending on the data used for intake via food, this value may or may not be used when calculating the soil remediation value. If drinking water is already included in the diet, it will be equated to zero here in order to avoid double counting.

2.4.4. CONCENTRATIONS IN FOODSTUFFS AND INTAKE VIA FOOD

Concentrations of PFOS in foodstuffs and intake estimates for Flanders/Belgium are discussed in BF-Risk and were calculated by EFSA in 2012 and 2018[11]. In the literature, data are also available for Sweden and the Netherlands, and in the European project PERFOOD the exposure via food was quantified for 4 European countries including Belgium. For the calculation of the soil remediation value, data should ideally be used in which concentrations in food and intake estimates are linked.

BF-Risk – Flanders (2009)

In the project BF-Risk (Cornelis *et al.*, 2009; D'Hollander *et al.*, 2009) samples of foodstuffs of Flemish origin were analysed, with a distinction between samples originating from organic farming and conventional farming. The samples analysed were divided into a number of groups including "vegetables" (potato, carrot, tomato, chicory, onion, lettuce, leek and wheat), "fruit" (apple and strawberry), "meat" (chicken, pork and beef), "dairy" (chicken eggs and raw cow's milk), "fish" (eel,

[10] When values are below the reporting limit in a set of measurement data , it is possible to choose to set the data below the reporting limit to zero (= lower bound approach) or equal to the reporting limit (= upper bound approach) when calculating an average value. A third possibility is to equate it to half of the reporting limit (= medium bound approach). For the measurement data PFOS in drinking water below the LOQ, a 0 value was assumed as concentration in the lower bound approach and a concentration equal to the LOQ ($0.6\ .10^{-3}$ µg/l) was assumed in the upper bound approach.
[11] After finalising this report, EFSA published new data in 2020

cod, rocket, dab, whiting, herring, sprats and flounder) and "drink" (beer and tap water). For each item 6 growers were sought, 3 organic growers and 3 "classic" growers. Three pieces per selected food item were purchased from each grower. Table 2 gives an overview of the results of the analysis for a selection of vegetables, meat, cow's milk and eggs. In fish from the North Sea, $<1.0.10^{-4}$ – $6.0.10^{-4}$ mg/kg of PFOS was detected.

Table 2: Overview of the range (minimum-maximum) of PFOS concentrations and the median in organic and conventional food samples (D'Hollander *et al.*, 2009).

Foodstuff	Organically grown Min-max (median) mg/kg	Conventionally grown Min-max (median) mg/kg
Potato	$<2.00.10^{-5}$ -$1.90.10^{-2}$ ($5.00.10^{-5}$)	$< 2.00.10^{-5}$ ($< 2.00.10^{-5}$)
Carrot	$3.00.10^{-4}$- $5.00.10^{-4}$ ($4.00.10^{-4}$)	$<2.00.10^{-5}$-$2.00.10^{-4}$ ($<2.00.10^{-5}$)
Onion	$<2.00.10^{-5}$-$2.4.10^{-3}$ ($0.4.10^{-3}$)	$<2.00.10^{-5}$ ($<2.00.10^{-5}$)
Tomato	$<2.00.10^{-5}$-$0.7.10^{-3}$ ($0.04.10^{-3}$)	$<2.00.10^{-5}$ ($<2.00.10^{-5}$)
Lettuce	$<2.00.10^{-4}$-$1.00.10^{-2}$	$<2.00.10^{-5}$-$4.00.10^{-4}$ ($<2.00.10^{-5}$)
Beef	$<1.00.10^{-4}$ ($<1.00.10^{-4}$)	$<1.00.10^{-4}$ ($<1.00.10^{-4}$)
Chicken	$<1.00.10^{-4}$ -$2.1.10^{-3}$ ($<1.00.10^{-4}$)	$<1.00.10^{-4}$ - $9.00.10^{-4}$ ($<1.00.10^{-4}$)
Pork	$<1.00.10^{-4}$ -$5.0.10^{-4}$ ($<5.00.10^{-4}$)	$<1.00.10^{-4}$ ($<1.00.10^{-4}$)
Eggs	$<1.40.10^{-3}$-$2.10.10^{-2}$ ($2.60.10^{-3}$)	$<1.20.10^{-3}$-$2.20.10^{-2}$ ($<1.20.10^{-3}$)
Cow's milk	$<6.00.10^{-4}$ ($<6.00.10^{-4}$)	$<6.00.10^{-4}$ ($<6.00.10^{-4}$)

For the calculation of exposure, Cornelis *et al.* (2009) calculated with average concentrations. Figures reported below the limit of quantification have been replaced by the corresponding limit of quantification (= upper bound approach). No account was taken of the influence of preparation and packaging on the levels in foodstuffs consumed. Cornelis *et al.* (2009) included data from foreign studies as foodstuffs on the Belgian (Flemish) market do not only originate from Belgium and the dataset with concentration data measured in the BF-Risk project is too limited. To this end, literature data were looked up from measurements carried out from 2003 and published until mid-May 2009. For the various foodstuffs, with the exception of fish, the measurements are dominated by the data from the BF-Risk project.
For fruit and vegetables, the average levels from the BF-Risk project did not differ significantly from those from other European studies (Spain, UK). Cornelis *et al.* (2009) did however observe that there was significant variation in the data. For vegetables, the authors distinguished between vegetables and potatoes because the latter showed higher concentrations. For meat, the literature data are slightly higher than the Flemish measurements. No data were available for butter and therefore data for dairy were used.

Table 3: Average concentrations of PFOS in a selection of foodstuffs, as used in the intake estimation (mg/kg fresh weight) (Cornelis *et al.*, 2009; Cornelis *et al.*, 2012).

Food group	PFOS
Potato	$6.18.10^{-3}$
Vegetables	$6.02.10^{-4}$
Butter	$2.50.10^{-4}$
Egg	$6.86.10^{-3}$
tap water and bottled water	$5.22.\ 10^{-6}$

Liver	$0.00.10^{0}$
Milk	$2.50.10^{-4}$
Meat	$5.54.10^{-5}$
Poultry meat	$6.33.10^{-4}$
Pork meat	$1.70.10^{-4}$
Seafish	$1.2.10^{-2}$
Fresh water fish	$1.74.10^{-1}$

For the intake via food and drinking water, the results of the Belgian Food Consumption Survey of 2004, carried out by WIV, were used (De Vriese *et al.*, 2006). This was the most recent study into food intake in the Belgian population over 15 years of age.

The average intake for the different age groups as calculated in BF-Risk is given in Table 4. The intake for adults is dominated by fish and shellfish, followed by potatoes. In children, the intake of PFOS is dominated by potatoes (48%), followed by similar fractions of fish and fishery products, dairy products, eggs and fruit (around 10% each). The contribution via water (and water-based drinks such as coffee and tea) and beer is insignificant.

Table 4: Intake of PFOS via food by the Flemish (Belgian) population (ng/kg.day). The exposure via food could not be calculated for children aged 0.5 - ≤ 3 years due to a lack of Flemish/Belgian consumption figures (Cornelis *et al.*, 2009).

Age (years)	**1-<3**	**3-<6**	**6-<10**	**10-<15**	**15-<21**	**21-<31**	**31-<41**
Intake (ng/kg.d)		57.2	38.9	27.8	24.0	24.1	20.2

Noorlander et al. (2010) - Netherlands

Noorlander *et al.* (2010) calculated the intake of PFOS via food and drinking water in the Netherlands using the 'total diet method', which is a combination of consumption data, concentration measurements in mixed samples of specific food categories and drinking water, and statistical modelling. Drinking water samples were not measured, the concentrations reported by EFSA (2008b) were used instead, namely $7.00.10^{-3}$ µg/l. Table 5 shows a selection of concentrations measured in food samples in 2009 in the Netherlands.

Table 5: Concentration of PFOS in food groups collected in 2009 in the Netherlands (results > LOD are shown in bold).

Food group	**PFOS in mg/kg**
Oily fish	**$6.10.10^{-5}$**
White fish	**$3.08.10^{-4}$**
Butter	**$3.30.10^{-5}$**
Milk	**$1.00.10^{-5}$**
Eggs	**$2.90.10^{-5}$**
Beef	**$8.20.10^{-5}$**

Vegetables/fruit	$< 4.70.10^{-5}$

The long-term intake was calculated for PFOS, taking into account concentration data from 2010 and data from the Dutch food consumption survey in 1998 (Table 6). The intake of drinking water contributed 33% of the total intake, followed by milk (25%), beef (21%), white fish (9%) and pork (4%). The calculated intakes for the Netherlands are slightly lower than those calculated by EFSA (2012) and Vestergren *et al.* (2012) and a factor of 10-100 lower than those calculated by Cornelis *et al.* (2009).

Table 6: Long-term intake via food for PFOS in the Netherlands (Noorlander *et al.*, 2010; Noorlander *et al.*, 2011) for the P50 percentile (in ng/kg.d). Values under the LOD were approached according to 3 different scenarios.

Age (years)	S1: male	S1: female	S2: male	S2: female	S3: male	S3: female
2	0.687	0.737	0.707	0.747	0.748	0.785
10	0.293	0.314	0.311	0.328	0.341	0.359
40	253	0.273	0.267	0.281	0.286	0.300
Lifelong average	279	0.298	0.291	0.309	0.313	0.329

When the analysis results are ≥LOD but < LOQ, 3 scenarios can be identified:
S1: result = LOD; S2: result = measured concentration; S3: result = LOQ

The intake data from Noorlander *et al.* (2011) are used in Wintersen (2019) to calculate the contribution of the various foodstuffs to the background exposure when calculating risk limits for the application of PFOS-containing soil and sludge for arable and livestock farming. A background exposure of 0.159 ng/kg.d was assumed for livestock and 0.318 ng/kg.d for arable crops. This exposure is deducted from the health-related risk limits.

EFSA (2012 and 2018) - Belgium

EFSA (2012) collected analytical results of PFOS in foodstuffs in 13 European countries over the period 2006-2012 (54,195 analytical results distributed among all PFAS of which 7,523 for PFOS, 2% of these samples came from Belgium). The dataset was characterised by a high proportion of left-skewed data (results < LOD or LOQ), 74% for PFOS (10,889 samples of which 10,191 were eventually retained).
A selection of results is given in Table 7. In comparison with the data used for the intake estimate in Cornelis *et al.* (2009) the average upper bound concentration for potatoes is a factor of 10 lower, as is the case for vegetables and eggs. The concentration in meat with which Cornelis *et al.* (2009) made the intake estimate is in the same order of magnitude as the EFSA upper bound values, as is milk. For fish, the data from Cornelis *et al.* (2009) are approximately 10-100 times higher than those obtained by EFSA.

Table 7: Selection of analysis data for PFOS in foodstuffs according to EFSA (2012).

Food group	Number of samples (including mixed samples)	Proportion of left-skewed distribution of data*	Average Lower bound (mg/kg)**	Average Upper bound (mg/kg)
Root crops	135 (277)	97	$9.50.10^{-6}$	$2.10.10^{-4}$
Bulbous vegetables	8 (68)	88	$2.20.10^{-6}$	$3.95.10^{-5}$
Fruiting vegetables	37 (243)	95	$2.10.10^{-6}$	$6.50.10^{-5}$
Cabbages	23 (111)	96	$1.20.10^{-6}$	$9.90.10^{-5}$
Leafy vegetables	25 (210)	84	$6.00.10^{-7}$	$2.60.10^{-5}$
Leguminous vegetables	4 (13)	100	0	$1.11.10^{-5}$
Stalk vegetables	23 (176)	100	0	$4.66.10^{-5}$
Potatoes and potato products	299 (335)	99.7	$3.60.10^{-6}$	$6.30.10^{-4}$
Beef	232 (1418)	91	$8.60.10^{-6}$	$1.20.10^{-4}$
Poultry	150 (737)	97	$9.70.10^{-6}$	$1.40.10^{-4}$
Offal	1261 (1623)	91	$4.20.10^{-4}$	$1.90.10^{-3}$
Milk	152 (722)	94	$9.00.10^{-7}$	$1.20.10^{-4}$
Butter (animal fat)	13 (55)	92	$8.2.10^{-4}$	$9.5.10^{-4}$
Eggs	95 (581)	85	$3.70.10^{-5}$	$7.0.10^{-4}$
Fish and other	2534 (4395)	63	$1.99.10^{-3}$	$2.4.10^{-3}$
Tap water	114 (156)	91	$9.00.10^{-7}$	$3.90.10^{-6}$
Bottled water	255 (255)	87	$4.00.10^{-7}$	$1.70.10^{-6}$

*Left-skewed data = data < LOQ or LOD

**Lower bound: value 0 assigned to all left-skewed data, upper bound: value of LOQ or LOD assigned to left-skewed data

For the calculation of the intake, EFSA used the data in the Belgian Food Consumption Survey (De Vriese *et al.*, 2006) and for children, from the Flemish toddler study (Huybrechts, 2008). The intakes for Belgium calculated in EFSA (2012) are given in Table 8, drinking water is part of the calculated intakes. In these calculations, EFSA (2012) makes the reservation that a chronic intake estimate via food cannot be accurate when more than 80% of the analysis results are lower than the LOD or LOQ. Moreover, for many food groups, the proportion of left-skewed data is more than 90%. For this reason, the intake estimate below is a rough indication of exposure. If the lower bound method is used, the intake is likely to be underestimated; if the upper bound method is used, it may be significantly overestimated. When the calculated intakes from EFSA are compared with those in Cornelis *et al.* (2009), it can be seen that for most age groups the data calculated in BF-Risk are approximately one order of magnitude higher than those calculated by EFSA (upper bound).

Table 8: Average chronic PFOS intake via food, lower bound (LB) and upper bound (UB) approach (EFSA, 2012).

Age (years)	1-3	3-10	10-18	18-65	65-75	≥75
Intake LB - UB (ng/kg.d)	1.20-9.10	1.20-7.40	0.540-2.50	0.800-2.60.	0.910-2.90	0.840-2.90

For adults (and children), EFSA (2012) mentions fish and seafood (50-80%), fruit and fruit products (8-27%) and meat and meat products (5-8%) as the main food groups contributing to the intake of PFOS (based on lower bound calculations, upper bound data were not published).

At the end of 2018, EFSA published a new intake estimate for chronic exposure to PFOS and PFOA (EFSA, 2018c), based on 21,411 samples for which data for PFOS and PFOA were available (end of 2016). 62% of the samples came from Germany, followed by Norway and France. The samples were reported between 2000 and 2016, but only samples collected after 2007 were included in the calculations. For PFOS, the calculation was made with 10,012 results, for PFOA with 9,828 results. As in 2012, the data were characterised by a large proportion of left-skewed data (results < LOD or LOQ). Overall, according to EFSA, the intake estimate for PFOS is approximately 30% lower in 2018 than in 2012; for Belgium this difference is even greater (see Table 9). The EFSA report is accompanied by annexes in Excel, which present average and P95 intakes for Belgium for different age groups using the lower and upper bound approach, these data are shown in Table 9.

Table 9: Average chronic PFOS intake via food, lower bound (LB) and upper bound (UB) approach for Belgium, obtained by EFSA in 2018 (EFSA, 2018c).

Age (years)	1-3	3-10	10-18	18-65	65-75	≥75
Intake LB - UB (ng/kg.d)	0.59-3.48	0.64-3.03	0.30-1.41	0.51-1.42	0.53-1.41	0.62-1.53

The concentrations used by EFSA for the intake estimation are shown in Table 10, these are also taken from the Excel tables annexed to the EFSA report. Although the intake estimates in 2018 are lower than those in 2017, the concentrations reported by EFSA are higher for most foodstuffs in 2018. No explanation could be found for this in the report.

Table 10: Selection of analysis data for PFOS in foodstuffs according to EFSA (2018c).

Food group	Number of samples	Proportion of left-skewed distribution of data (%)*	Average Lower bound (mg/kg)**	Average Upper bound (mg/kg)
Root vegetables	168	93	$8.0.10^{-6}$	$2.19.10^{-4}$
Bulbous vegetables	77	97	$2.0.10^{-6}$	$2.37.10^{-4}$
Fruiting vegetables	140	95	$2.10.10^{-6}$	$1.38.10^{-4}$
Cabbages	45	95	$2.10.10^{-6}$	$1.13.10^{-4}$
Leafy vegetables	114	95	$2.10.10^{-6}$	$1.10.10^{-4}$
Leguminous vegetables	9	100	0	$3.10.10^{-5}$
Stalk vegetables	92	100	0	$1.88.10^{-4}$
Potatoes	52	100	0	$3.89.10^{-4}$
Beef	150	87	$5.60.10^{-5}$	$1.85.10^{-4}$
Poultry	176	98	$9.00.10^{-6}$	$1.43.10^{-4}$
Offal	1417	88	$6.58.10^{-4}$	$2.117.10^{-3}$
Milk	249	94	$1.00.10^{-6}$	$2.1.10^{-4}$
Eggs	26	92	$2.32.10^{-4}$	$1.122.10^{-3}$
Fish and other	2878	67	$2.244.10^{-3}$	$2.765.10^{-3}$
Tap water	45	63	$1.00.10^{-6}$	$6.00.10^{-6}$
Bottled water	330	90	0	$1.00.10^{-6}$

*Left-skewed data = data < LOQ or LOD
**Lower bound: value 0 assigned to all left-skewed data, upper bound: value of LOQ or LOD assigned to left-skewed data

Vestergren et al. (2012) - Sweden

Vestergren *et al.* (2012) calculated the intakes of various PFAS for the Swedish population in 1999, 2005 and 2010. The authors used a highly sensitive analysis technique, the analysis results for a selection of food samples can be found in Table 11.

Table 11: Estimated (between LOD and LOQ) and measured (> LOQ, indicated in bold) concentrations in mg/kg PFOS in Swedish food samples from 1999, 2005 and 2010 (Vestergren *et al.*, 2012).

Food group	2010	2005	1999
Dairy products	$5.60.10^{-6}$	$4.00.10^{-6}$	< LOD
Meat products	**$2.5.10^{-5}$**	**$8.6.10^{-5}$**	**$1.9.10^{-4}$**
Fats	**$1.3.10^{-5}$**	< LOD	**$1.0.10^{-5}$**
Pastry	**$2.10.10^{-5}$**	**$1.70.10^{-5}$**	**$1.10.10^{-5}$**
Fish	**$1.29.10^{-3}$**	**$7.80.10^{-4}$**	**$1.09.10^{-3}$**
Eggs	**$3.90.10^{-5}$**	**$1.30.10^{-5}$**	**$1.28.10^{-3}$**
Cereal products	$2.20.10^{-6}$	$4.00.10^{-6}$	$3.50.10^{-6}$
Vegetables	**$4.10.10^{-6}$**	**$2.40.10^{-5}$**	**$4.60.10^{-6}$**
Potatoes	**$6.9.10^{-6}$**	**$5.8.10^{-6}$**	$1.9.10^{-6}$

For PFOS, based on consumption data for the Swedish population, average intakes were calculated of 1.44, 0.861 and 1.01 ng/kg.d in 1999, 2005 and 2010, respectively (lower bound scenario, the difference with the upper bound scenario is less than 1%). These intakes were calculated without drinking water. Fish (84.9%) and meat products (7.3%) were the main contributors to the UB scenario in 2010. The contribution of potatoes was 1.2%, vegetables 1.1% and fruit 0.7%. In the lower bound scenario, approximately 80% of the intake comes from the consumption of fish. The intake estimates are similar to those obtained by EFSA (2012) and a factor of 10 lower than those calculated in BF-Risk (Cornelis *et al.*, 2009). The concentration in potatoes is 1000 times lower than that in BF-Risk and 100 times lower than the UB value determined by EFSA (but comparable to the LB value determined by EFSA). The fish concentrations are 10-100 times lower than the values in BF-Risk and comparable to the EFSA values; the meat concentrations are comparable to those in BF-Risk and a factor of 10 lower than those in EFSA (UB; comparable to LB data); the vegetable concentrations are comparable to the LB data in EFSA (and a factor of 10 lower than the UB data) and a factor of 100 lower than those used in BF-Risk.

PERFOOD- Belgium

In the European project PERFOOD (Klenow *et al.*, 2013) the exposure via food was quantified for 4 European countries including Belgium. For this, results from analytical analyses carried out during the project were used (Herzke *et al.*, 2013; Hlouskova *et al.*, 2013) as were consumption data for the individual countries as published by EFSA. The concentrations for meat were similar to those of Cornelis *et al.* (2009), EFSA (2012) and Noorlander *et al.* (2010). As regards vegetables, concentrations were approximately 200 times lower than those used in Cornelis *et al.* (2009) for the intake estimate (average 6.10^{-4} mg/kg versus $3.24.10^{-6}$ mg/kg for vegetables), see also Table 12. For fish, the data are almost a factor of 100 lower than those in Cornelis *et al.* (2009) and a factor of 10 lower than EFSA (2012) UB data and data in Vestergren *et al.* (2012).

Table 12: Concentration data for PFOS in Belgian samples measured in PERFOOD (Herzke *et al.*, 2013; Hlouskova *et al.*, 2013).

Food group	PFOS in mg/kg fresh weight
Vegetables	$3.24.10^{-6}$
Cabbages	$2.58.10^{-5}$
Spinach	$2.35.10^{-5}$
Asparagus	$9.00.10^{-6}$
Broccoli, cauliflower, aubergine, courgette, cucumber, peppers, tomato, beans, peas, chicory, lettuce, carrot, potato, celery, fennel	< LOQ
Beef	$5.7.10^{-5}$
Pork liver	$2.69.10^{-3}$ *
Pork	$2.70.10^{-5}$ *
Lamb	$1.5.10^{-5}$
Butter	$3.28.10^{-5}$
Eggs	$8.09.10^{-5}$ *
Seafood	$5.75.10^{-4}$ *
Fish	$3.01.10^{-4}$ *

* : figures calculated by VITO in the framework of the current soil remediation value project by combining data from figure 2 (sum of PFAS in foodstuffs for Belgium) and figure 3 (PFAS profiles foodstuffs Belgium) published in Hlouskova *et al.* (2013)

Due to the sensitive analysis technique, the authors consider a further interpretation of the results for the lower bound data to be justified. The intake for adults in the lower bound scenario is dominated by fish and seafood (46%) and fruit (33%). For children, fruit (46%) is the most dominant category, followed by meat and offal (26%) and fish and seafood (13%). The authors note that the intake in Belgium is higher than in the other countries studied (the Czech Republic, Italy and Norway). The results from the PERFOOD project and the lower bound results from EFSA (2012) and EFSA (2018c) are of approximately the same order of magnitude, the upper bound results of EFSA are significantly higher. The intake estimate is similar to the one carried out by Noorlander *et al.* (2010) for the Netherlands, 4 times lower than the one for Sweden (Vestergren *et al.*, 2012) and 10-100 times lower than the one calculated in BF-Risk (Cornelis *et al.*, 2009). The authors mention that the results in PERFOOD should not be considered representative for Belgium, as the samples were taken in a short period of time and at a limited number of sampling sites.

Table 13: Average PFOS intake via diet for children and adults in Belgium, calculated according to the lower and upper bound scenario (Klenow *et al.*, 2013).

Age (years)	3-9 average	18-64 average
Intake: LB – UB (ng/kg.d)	0.960-1.11	0.336-0.405

FSANZ (2017) – review and comparison of European studies

A review by FSANZ (2017) discusses recent European studies in which the occurrence of PFOS in foods is examined, among other things. The authors note that, in general, for PFOS, the data from studies published over the period 2012-2016 are well in line with the data published by EFSA (2012), with the exception of the data used for intake estimate by Cornelis *et al.* (2012), which are significantly higher for a number of food groups such as dairy products, eggs, vegetables, potatoes and fish. The intake estimate for PFOS published in Cornelis *et al.* (2012) is also much higher than that in other studies, according to the authors of FSANZ (2017).

Discussion

Cornelis *et al.* (2009) and EFSA (2012) and Klenow *et al.* (2013) published data for the intake estimate of PFOS for the Belgian population (see Figure 1). The highest results were obtained in BF-Risk (Cornelis *et al.* (2009) and then the UB data from EFSA. The lowest intakes were calculated with the LB data of EFSA and in PERFOOD (Klenow *et al.* (2013). The concentrations used for intake estimations are based on own measurements or literature in Cornelis *et al.* (2009) and on own measurement results in Klenow *et al.* (2013). The number of samples analysed in both studies is limited or was collected during a limited period of time, and for this reason the results may not be representative. EFSA has a large amount of samples at its disposal, but it calculates intakes for Belgium with concentrations for foodstuffs from all the European countries that have made data available, so the concentrations do not relate only to foodstuffs present on the Belgian market. Klenow *et al.* (2013) refer to higher concentrations of PFAS in foodstuffs originating from Belgium compared to other European countries, so it is possible that the EFSA underestimates the concentrations used as they do not originate from Belgium alone. EFSA itself considers its UB calculations to overestimate the intake, since 80% of the analysis results are lower than the LOQ or LOD.

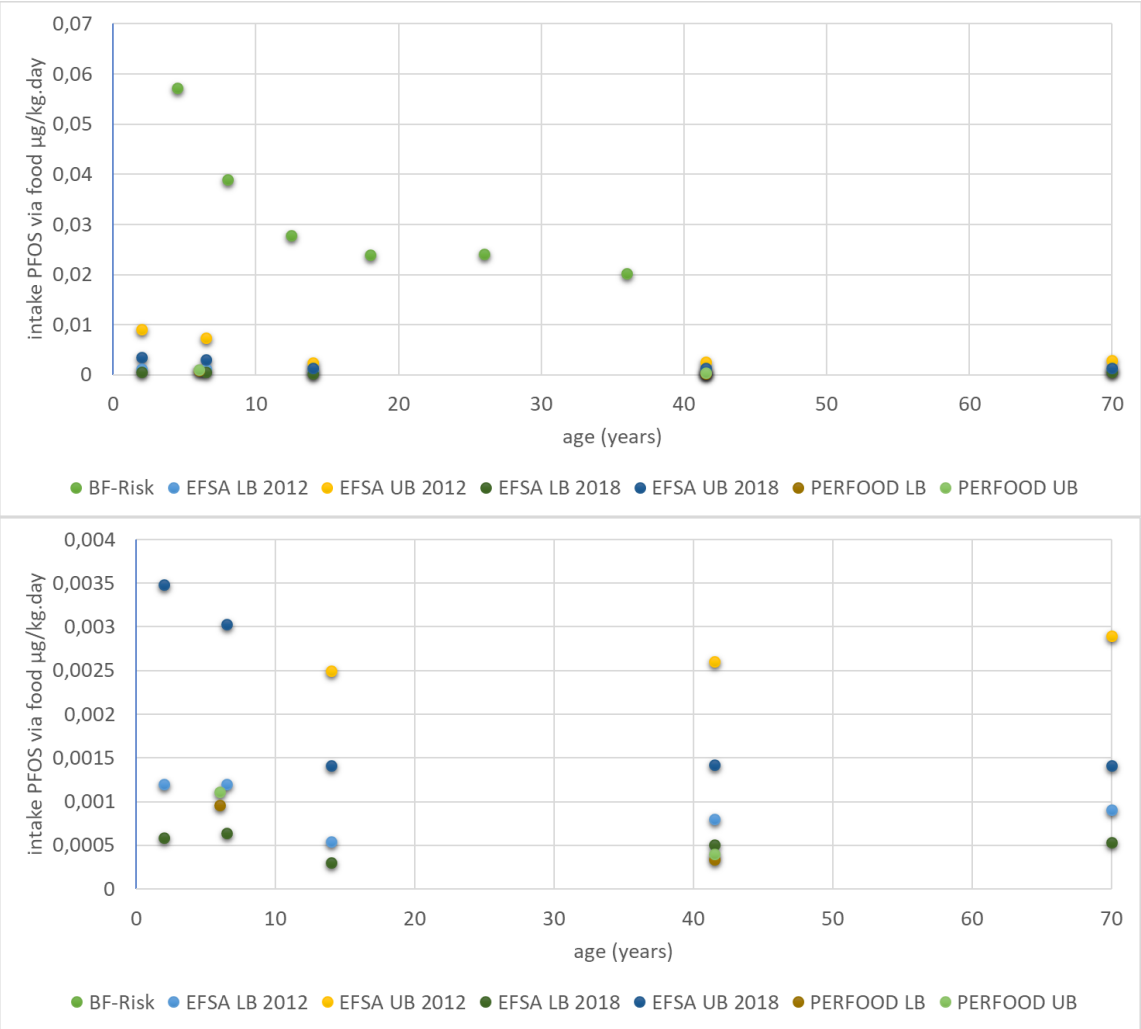


Figure 1: Intake PFOS for Belgium in µg/kg.day for different ages. Comparison of data published in BF-Risk, EFSA (upper bound and lower bound) and PERFOOD (upper bound and lower bound)

Taking into account the various aspects described above, it was decided to initially use the UB intake estimate made by EFSA for the Belgian population as intake estimate for PFOS. Both the data for 2012 and 2018 will be calculated through, and the LB scenario for 2012 will also be calculated. The consumption figures use specific data for Belgium and the concentrations are based on a large amount of samples. According to EFSA, the use of UB intake estimates is a highly conservative approach, whereas the use of LB, on the other hand, may be an underestimate, since undeterminable concentrations are set at zero. This is also shown by a comparison of the intake estimates for the LB and UB scenario of EFSA with recent scientific literature, the LB approach of EFSA approaches the intake estimates based on more sensitive analytical techniques such as in Vestergren *et al.* (2012) and Perfood better than the UB approach. Therefore, both the UB approach (conservative) and the LB approach will be calculated through and tested for feasibility. As long as drinking water is part of the diet used by EFSA to estimate intakes, the concentration in drinking water for deriving soil remediation values will be set at zero to avoid double counting.

Table 14 shows the EFSA 2012 and 2018 (UB and LB) data extrapolated to the age groups present in S-Risk.

Table 14 PFOS intake via food for the Belgian population calculated on the basis of extrapolation of UB approach of EFSA (2012) and EFSA (2018c).

Intake (ng/kg.d)	1-<3	3-<6	6-<10	10-<15	15-<21	21-<31	31 and above
2012 UB	9.10	8.10	6.70	2.85	2.47	2.50	2.77
2012 LB	1.2	1.2	1.08	0.513	0.562	0.634	0.875
2018 UB	3.48	3.25	2.90	1.90	1.47	1.50	1.48
2018 LB	0.590	0.617	0.572	0.368	0.330	0.391	0.514

2.5. TRANSFER TO PLANTS

Due to the amphiphilic nature of PFAS, the formulas of Trapp (2002), Trapp *et al.* (2007) and Trapp and Matthies (1995)) usually applied in S-Risk for non-ionising organic compounds cannot be used to calculate plant uptake of PFAS, and therefore empirical relationships based on bioconcentration factors (BCF) need to be applied. In the literature, BCF-values for PFOS are generally based on soil concentrations, while in S-Risk BCF based on soil pore water are required for organic compounds. For this reason, modifications were made to S-Risk, using a test environment, in order to be able to calculate soil remediation values with BCF_{soil} from the literature (see also 2.10 calculations of the soil remediation value).

For inventorising plant uptake of PFOS we rely, in the first instance, on a review paper of Ghisi *et al.* (2019) where we make sure we verify the numerical values each time against the original papers. Additional sources are searched for and, if available, integrated. Selected BCF are compared to BCF derived in the Netherlands (Wintersen, 2019).
Ghisi *et al.* (2019) bring together all known data related to plant uptake for different PFAS, differentiating between perfluoroalkyl carboxylic acids (PFCAs) and perfluoroalkyl sulfonic acids (PFSAs). Most of the data in this literature review relating to plant uptake for PFAS refer to PFOA and PFOS, C8 representatives for PFCAs and PFSAs respectively. This subset of PFSA-data from Ghisi *et al.* (2019) is the starting point for final selection of BCF values in S-Risk for the derivation of soil remediation values of PFOS. Studies relying on aquacultures for the uptake of PFAS are not taken into consideration for this study. A further literature search/review on data relating to uptake of PFOS by crops did not provide additional data. Due to the water solubility of PFAS, the industrial and urban sludge from water treatment plants used for irrigation are considered important sources for plant uptake. In addition, sludge applications intended to improve soil structure and the use of PFAS as emulsifiers in plant protection products are also important sources of plant uptake.

The literature often distinguishes between the uptake of PFOS in cereal crops and vegetables (bulbous, tuber, fruiting and leaf vegetables). This distinction is based, on the one hand, on the difference between the parts of plants that are suitable for consumption, vegetative parts including fruit in the case of vegetables and seeds in the case of cereal crops. On the other hand, the vegetative parts of cereals are used in fodder crops (straw, chaff, etc.). Ghisi *et al.* (2019) identify 5 publications relating to cereal crops, including maize, oats, wheat and ryegrass (grasses). In S-Risk, BCF derived for cereals and grasses are the basis for the calculation of transfer PFOS via locally grown fodder crops to livestock (biomagnification). A distinction can be made between parts of the plant suitable for human consumption (cereals) and non-edible parts of cereals and grasses used in livestock feed

(straw, chaff). The results are summarised in Table 15. All numerical values in Table 15 are based on measurements taken on spiked soils, with the exception of Blaine *et al.* (2013) and Wen et al. (2014). In the latter studies, the plants were grown on soils to which a mixture of PFAS was added in the form of (organic) sludge. BCF for plant parts suitable for human consumption are often significantly lower than for non-edible parts. Differences in availability of PFOS for different crops are attributed to differences in protein content (Wen et al., 2016) or to morphological differences in e.g. leaf surface and/or root system (Müller et al., 2016).

Table 15: BCF (mg/kg plant dm)/(mg/kg soil dm) for PFOS for cereals and grasses (Selection from Ghisi *et al.* (2019))

Crop	compartment	concentration in soil (mg/kg dm)	BCF (mg/kg plant dm)/(mg/kg soil dm)	reference
maize	straw	0.25	0.132	Stahl *et al.* (2009)
		1	0.104	
	straw	0.25	0.320	Krippner *et al.* (2015)
		1	0.620	
	leaf	38.5	0.800	Navarro *et al.* (2017)
	cobs	0.25	LOD	Stahl *et al.* (2009)
		1	0.003	
	cereal grains	0.25	LOD	Krippner *et al.* (2015)
		1	LOD	
	maize stalks	0.00282	LOD	Blaine *et al.* (2013)
oats	straw	0.25	0.224	Stahl *et al.* (2009)
		1	0.150	
	cereal grains	0.25	0.004	Stahl *et al.* (2009)
		1	0.017	
ryegrass	4 consecutive samples	0.25	0.048	Stahl *et al.* (2009)
		1	0.470	
wheat	straw	0.25	0.200	Stahl *et al.* (2009)
		1	0.270	
	straw	0.0408	0.270	Wen et al., 2014
	cereal grains	0.25	LOD	Stahl *et al.* (2009)
		1	LOD	
	cereal grains	0.0408	0.062	Wen et al., 2014
	Ear of corn	0.0408	0.054	

LOD = Limit Of Detection

The results show that the BCF for crop plants and grasses increase in most cases with higher soil concentrations (Figure 2). There is not enough data to derive a relationship.

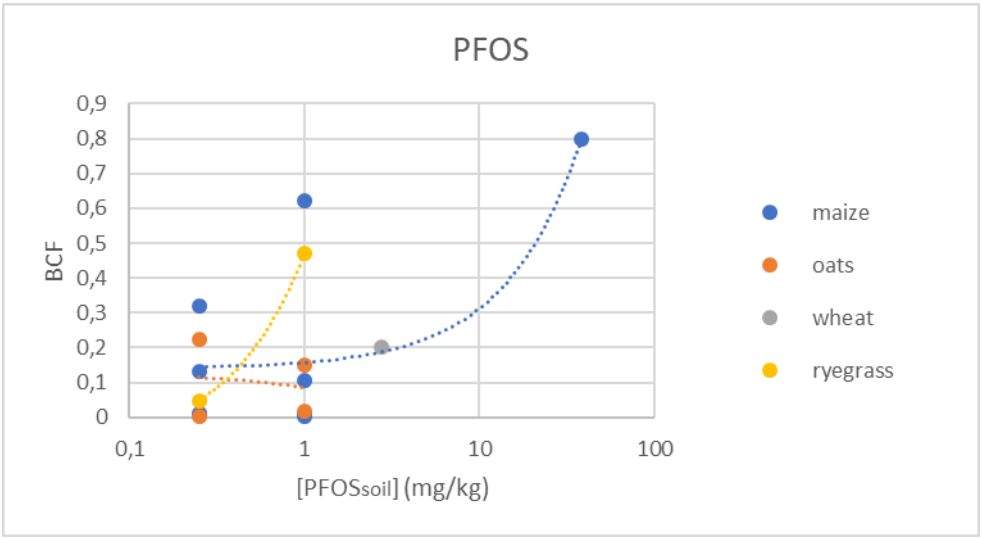


Figure 2: BCF_{PFOS} (mg/kg plant dm)/(mg/kg soil dm) for crop plants and grasses as a function of the concentration of PFOS in soil (mg/kg dm) (Selection from Ghisi *et al.* (2019))

S-Risk takes two BCF values into consideration for forage crops, namely grasses and maize. Preference is given to measurements on soils treated with sludge mixtures (as spiking may lead to overestimation). For Wen et al. (2014), there is a BCF for wheat of 0.06 (mg/kg plant dm)/(mg/kg soil dm) available which meets this condition. For maize, no BCF could be derived in the field study of Blaine *et al.* (2013). Stahl *et al.* (2009) report a BCF_{maize} = 0.003 (mg/kg plant dm)/(mg/kg soil dm) with an added concentration of 1 mg/kg PFOS in the soil. Both values are used for the calculations in S-Risk. The BCF value for ryegrass derived by Stahl *et al.* (2009) with the highest added value to soil deviates strongly from the other BCF values in Table 15 and it was consequently decided to only take the lowest BCF value into consideration for the calculations (see below).

When we extend the analysis for plant uptake of PFOS to vegetables, we find 4 additional studies in the overview publication of Ghisi *et al.* (2019). Additionally, we retrieved one extra publication, i.e., Navarro *et al.* (2017), based on which BCF_{PFOS} can be calculated for spinach and tomato. An overview of the available values is given in Table 16. The BCF derived by Lechner and Knapp (2011) are based on spiked soil. All other studies study soils enriched with PFAS-containing household or industrial sludge.

Table 16: BCF (mg/kg plant dm)/(mg/kg soil dm) for PFOS for vegetables (Selection from Ghisi *et al.* (2019) supplemented with Navarro *et al.* (2017))

Crop		Concentration in soil (mg/kg dm)	BCF (mg/kg plant dm)/(mg/kg soil dm)	reference
carrot	carrot (peeled)	0.01	0.53	Lechner and Knapp (2011)
	carrot (peeled)	0.458	0.43	

	carrot (peeled) Chantenay variety	0.445	0.55	Bizkarguenaga *et al.* (2016)
	carrot (peeled) Nantesa variety	0.335	0.49	
Celery	Celery shoots	0.04966	1.39	Blaine et al., 2014a
	Celery shoots	0.31949	0.05	
Cucumber	pot 1	0.01	-	Lechner and Knapp (2011)
	pot 2	0.556	0.067	
lettuce	leaf	0.04966	1.67	Blaine *et al.* (2013)
	leaf	0.31949	0.32	
	leaf	0.01391	0.10	
lettuce	leaf	0.507	0.15	Bizkarguenaga *et al.* (2016)
peas	fruits	0.04966	0.03	Blaine et al., 2014a
	fruits	0.31949	LOD	
potatoes	peeled	0.015	LOD	Lechner and Knapp (2011)
	peeled	0.317	0.01	
spinach		0.5	3.24	Navarro *et al.* (2017)
		0.23	4.30	
radish	carrot	0.04966	0.70	Blaine et al., 2014a
	carrot	0.31949	0.066	
tomato	fruits	0.04966	-	Blaine *et al.* (2013) Blaine *et al.* (2013) and Blaine et al., 2014a
	fruits	0.31949	-	Blaine *et al.* (2013) and Blaine et al., 2014a
	fruits	0.01391	-	Blaine *et al.* (2013)
		0.47	0.064	Navarro *et al.* (2017)

LOD = Limit Of Detection

A closer look at Table 16 and Table 15 shows that the calculated BCF for vegetables is overall one order of magnitude higher than that for cereals with the exception of potatoes (Figure 3). We can also make a distinction between experiments carried out on soil enriched with PFOS spiked sludge (Lechner and Knapp (2011); Bizkarguenaga *et al.* (2016)) and soil to which PFOS contaminated industrial or household sludge has been added (Blaine *et al.* (2013); 2014a; Navarro *et al.* (2017)). The highest concentrations of PFOS are found in spinach followed by lettuce. In carrots, we register higher concentrations than potatoes (tuber vegetables). No significant differences are found between peeled and unpeeled carrots (Lechner and Knapp (2011)).

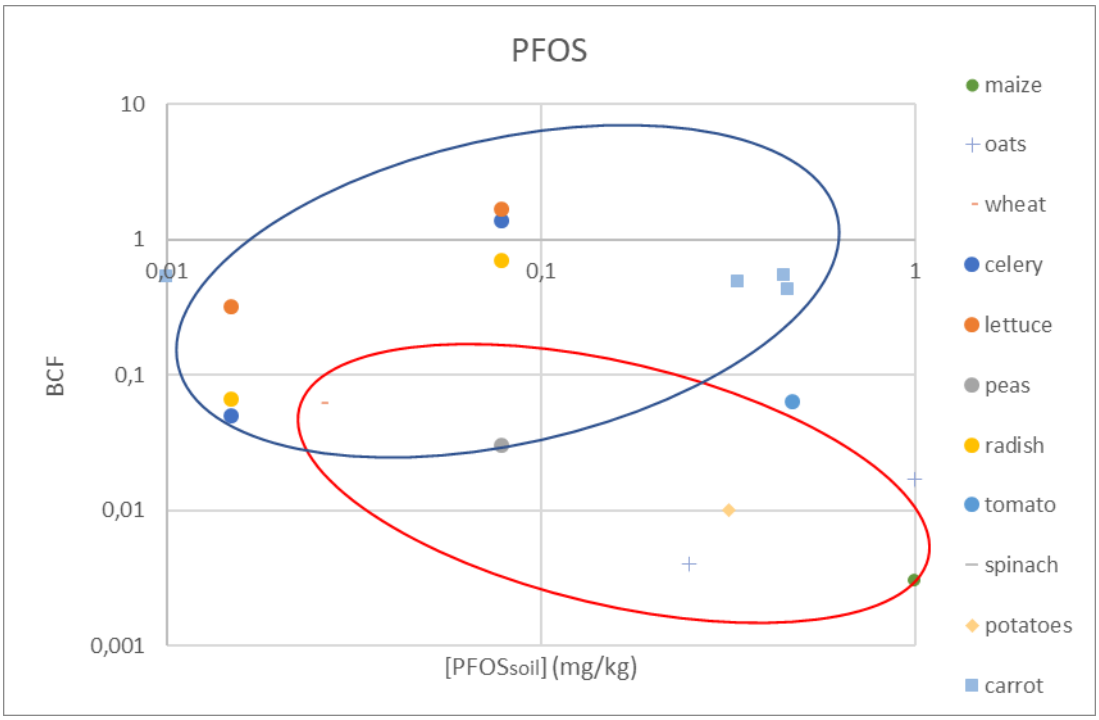


Figure 3: BCF_{PFOS} (mg/kg plant dm)/(mg/kg soil dm) in vegetables (circled blue) and cereal crops (circled red) as a function of the concentration of PFOS in the soil (mg/kg dm).

Blaine *et al.* (2013), 2014 perform measurements on soils enriched with organic sludge, contaminated with a mixture of PFAS. The authors make a distinction between sludge of industrial and household origin. Based on the differences in concentrations of PFOS on the one hand and the different % OC between the two types of enriched soils, it appears that the calculated BCF for vegetables increases with the PFOS concentration in the soil and decreases the higher the % OC. The highest % OC are found in household enriched sludge ((Blaine *et al.*, 2013); 2014) resulting in a lower BCF. We see a comparable result with Navarro *et al.* (2017) where the highest concentrations in spinach are found in soils enriched with industrial sludge.

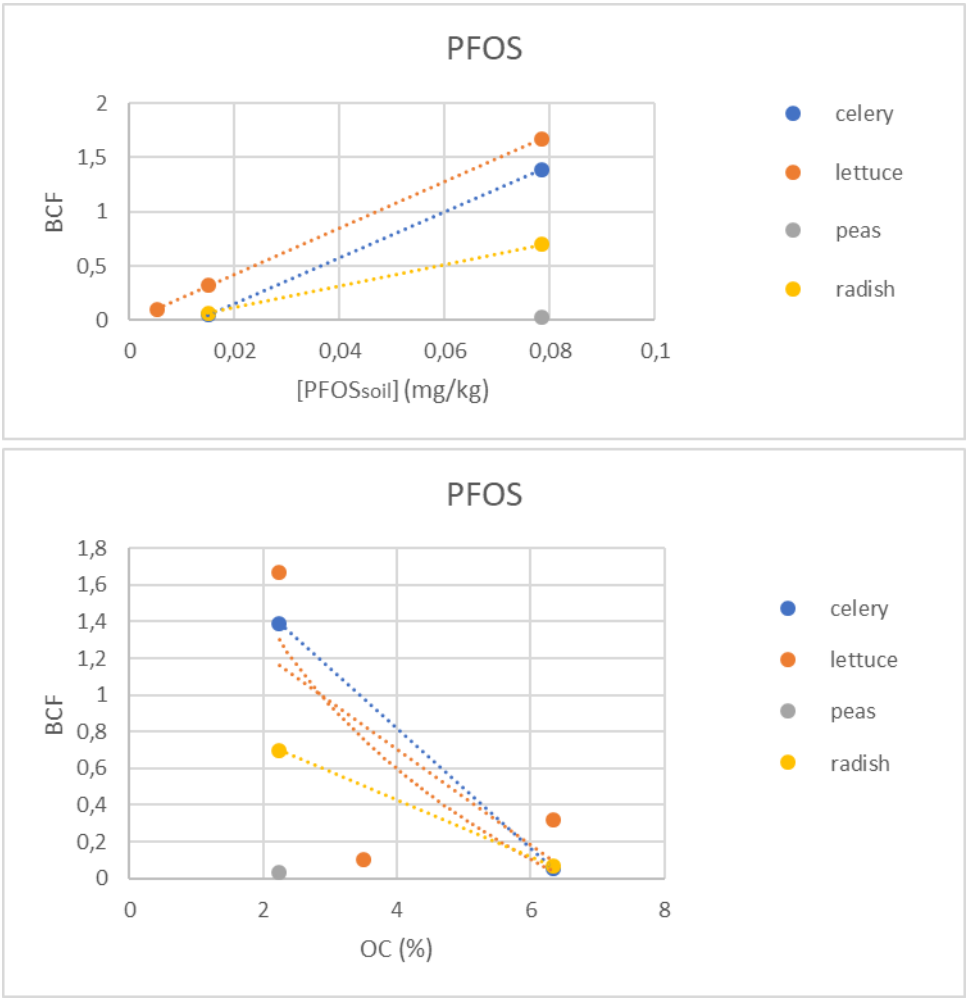


Figure 4: BCF_{PFOS} (mg/kg plant dm)/(mg/kg soil dm) as a function of concentration (mg/kg dm) in the soil and the % OC (based on data from Blaine *et al.* (2013), 2014). The trend lines are only indicative and do not describe a association between the variables in question.

The final selection BCF for cereals, grasses and vegetables is based on Table 16 and Table 15 where the average value was calculated each time more than one value was available (Table 17) for the same crop. For ryegrass, a BCF of 0.26 was originally chosen, which is the average value of Stahl *et al.* (2009). However, this value was subsequently revised because the highest added concentrations in the soil (1 mg/kg dm) applied by Stahl *et al.* (2009) were far above the background values of PFOS in soil (Pancras, 2018) which results in concentrations in the plant far above the values observed for the other grasses and cereals. Due to the large proportion of potatoes in the diet, an additional check was carried out. The concentrations listed by Cornelis et al. (2012) are consistent with the values underpinning the BCF derived by Lechner and Knapp (2011). However, the values published by EFSA are a factor of 10 lower.

Table 17: Selected BCF (mg/kg plant dm)/(mg/kg soil dm) for vegetables and cereals for calculating the soil remediation value (SRV) for PFOS in this study

Crop	PFOS	method of derivation
carrot	0.50	average value Lechner and Knapp (2011) and Bizkarguenaga *et al.* (2016)
Celery	0.72	average values

		Blaine et al., 2014
Cucumber	0.07	single value from Lechner and Knapp (2011)
lettuce	0.56	average value Blaine et al., 2013 and Bizkarguenaga *et al.* (2016)
peas	0.03	single value Blaine et al., 2014
potatoes	0.01	single value Lechner and Knapp (2011)
spinach	3.77	average values Navarro *et al.* (2017)
radish	0.38	average values Blaine et al., 2014
tomato	0.06	single value Navarro *et al.* (2017)
maize (cob)	0.003	single value Stahl *et al.* (2009)
oats (cereal)	0.011	average value Stahl *et al.* (2009)
ryegrass	0.048	lowest value Stahl *et al.* (2009)
wheat (cereal)	0.06	single value Wen et al., 2014

By way of comparison, the most recent RIVM report (Wintersen et al., 2019) refers to the same sources as used by Ghisi *et al.* (2019) with only additional values for potatoes from Schmallenberg, 2008. In order to allow for a comparison between the BCF collected in this study and the BCF values on a fresh weight basis from Wintersen et al. (2019), the latter were converted to a dry weight basis. For this, we used the dry matter contents from the formularium of S-Risk (Cornelis et al (2017)). These converted BCF are shown as a function of the concentration of PFOS in the soil in Figure 5.

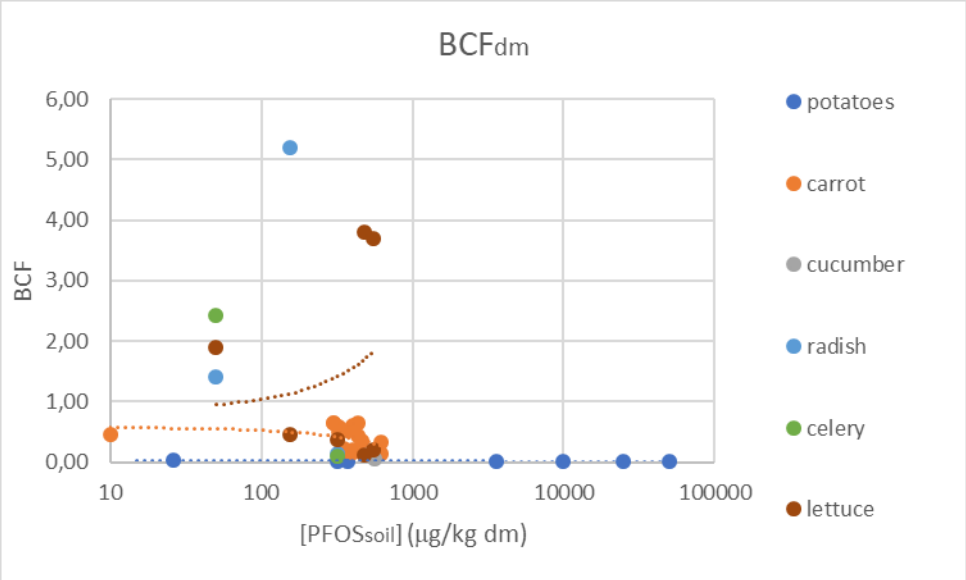


Figure 5: BCF_{PFOS} (mg/kg plant dm)/(mg/kg soil dm) from Wintersen et al. (2019) converted on a dry matter basis as a function of the concentration of PFOS in soil.

A comparison of both datasets for vegetables is made in Figure 6 and Table 18. For peas, spinach and tomatoes no BCF values are derived in Wintersen. For string beans, a single value was derived by

Wintersen et al. (2019) based on Blaine et al. (2014). For crops where values are available in both studies, we find the greatest difference for celery (x1.75 in Wintersen et al. (2019)), tomato ((x2.69 in Wintersen et al. (2019)) and radishes ((x5.85 in Wintersen et al. (2019)).

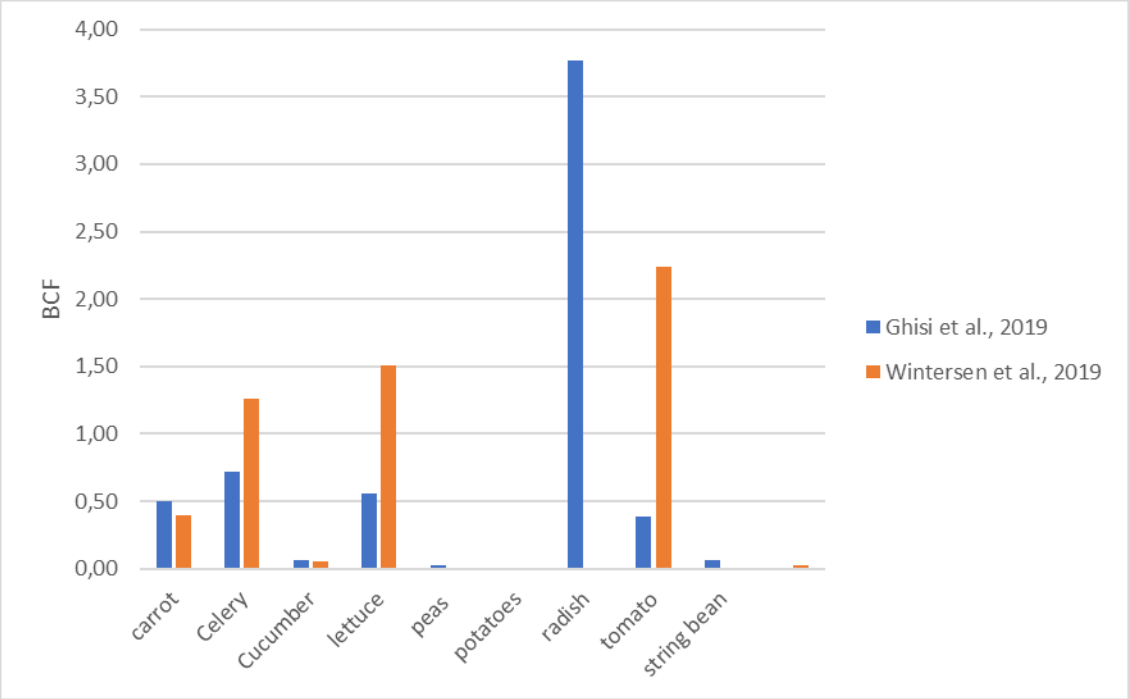


Figure 6: Comparison between BCF_{PFOS} (mg/kg plant dm)/(mg/kg soil dm) derived by Ghisi *et al.* (2019) and the BCF_{PFOS} (mg/kg plant dm)/(mg/kg soil dm) from Wintersen et al. (2019) converted on a dry matter basis.

Table 18: Comparison between BCF_{PFOS} (mg/kg plant dm)/(mg/kg soil dm) derived by Ghisi et al. (2019) and the BCF_{PFOS} (mg/kg plant dm)/(mg/kg soil dm) from Wintersen et al. (2019).

Crop	Ghisi et al., 2019	Wintersen et al., 2019
carrot	0.50	0.39
Celery	0.72	1.26
Cucumber	0.07	0.06
lettuce	0.56	1.51
peas	0.03	
potatoes	0.010	0.009
pumpkin		
spinach	3.77	
radish	0.38	2.24
tomato	0.06	
string bean		0.03
maize	0.003	0.055
oats	0.011	0.008
ryegrass	0.048	0.59

wheat	0.06	0.091

For the calculations in S-Risk, a food basket consisting of commonly consumed vegetables is applied. If no BCF values are found in the literature for one or more vegetables from this basket, equivalence rules as defined in Bierkens et al. (2016) are applied to fill in these missing values. This means that for the crop with missing BCF values from a certain crop group (tuber vegetables, bulbous vegetables, leafy vegetables, etc.) a BCF is applied from a related crop for which a BCF is available from the same crop group. By way of comparison, RIVM uses 3 BCF for their final calculations in C-Soil: a BCF for cereals (BCF = 0.063 (µg/kg plant **dm**)/(µg/kg soil dm)), and further a BCF for "potatoes" (BCF = 0.001 (µg/kg plant **fw**)/(µg/kg soil dm) and "other vegetables" (BCF = 0.017 (µg/kg plant **fw**)/(µg/kg soil dm).

Table 19: Calculated (bold) and estimated BCFs for PFOS for the different crops in the S-Risk food basket, with indication of the equivalence rules used

Plant	BCF or BCF model
potatoes	
potatoes	**0.01**
root and tuber vegetables	
carrots	**0.50**
salsify	0.44 (= average known root and tuber vegetables)
other root vegetables (such as radish)	**0.38**
bulbous vegetables	
bulbous vegetables (such as onion)	0.44 (= average known root and tuber vegetables)
leek	0.44 (= average known root and tuber vegetables)
fruiting vegetables	
tomato	**0.06**
cucumber	**0.07**
other fruiting vegetables (such as peppers)	0.065 (average known fruiting vegetables)
cabbages	
cabbage	0.44 (= average known root and tuber vegetables)
cauliflower and broccoli	0.44 (= average known root and tuber vegetables)
sprouts	0.44 (= average known root and tuber vegetables)
leafy vegetables	
lettuce	**0.56**
lamb's lettuce	0.56 (= lettuce)
endive	0.62 (average lettuce and celery)
spinach	**3.77**

Plant	BCF or BCF model
chicory	0.62 (average lettuce and celery)
celery	**0.72**
legumes	
beans	0.03 (= peas)
peas	**0.03**
grasses	
grass	**0.048**
cereals	
maize	**0.003**

2.6. TRANSFER TO ANIMAL PRODUCTS

Bioaccumulation of PFAS cannot be simulated on the basis of equilibrium partitioning as is the case for most neutral hydrophobic organic compounds that accumulate primarily in fat tissue. Due to their amphiphilic and anionic character, they are mainly distributed over the serum, liver and kidneys and their toxicokinetics are largely controlled by urinary excretion. The equations derived by Travis and Arms (1988) usied in S-Risk to estimate concentrations in meat and milk based on K_{ow} partition coefficients are not applicable, so we have to start from empirically derived biotransfer factors from case studies. A literature review provides 4 papers with paired measurement data for PFOS in feed and drinking water together with PFOS concentrations in dairy cow tissues/organs (n =3) and/or sheep (n=1) suitable for human consumption, from which biotransfer factors (BTF) can be derived.

Bovines

We find the most detailed study in Vestergren *et al.* (2013). Vestergren *et al.* (2013) derived BTF for dairy cows from agricultural areas without external influence from known PFCAs or PFSAs point sources. The feed consisted of a mixture of silage, maize and barley. Local well water was used as drinking water for the cattle. At the time of the measurements the adult (> 24 months old) dairy cows had had sufficient time to achieve equilibrium between intake and excretion of PFAS. The measurement results for PFOS in feed and drinking water, as well as in the different tissues and animal matrices relevant for the calculations, are summarised in Table 19. In bovines, the highest concentrations of PFOS are found in the liver and blood. The measured values in muscle and milk are 21 ± 19 ng/kg fw and 6.2 ± 1.1 ng/kg fw respectively. Based on a total daily intake of 294.6 ng/d PFOS, the authors calculate a BTF_{muscle} = 0.071 $mg.kg^{-1}$ $fw/mg.d^{-1}$ and BTF_{milk} = 0.021 $mg.kg^{-1}$ $fw/mg.d^{-1}$ (Vestergren, 2013 #8888) . The BTF_{liver}, a measure of the accumulation of PFOS in offal, is 0.442 $mg.kg^{-1}$ $fw/mg.d^{-1}$ (own calculation; Table 21).

van Asselt *et al.* (2013) developed a PBPK model predicting concentrations of PFOS in milk based on PFOS intake from dairy cows (n=6) via naturally contaminated feed (hay and silage grass). Three bovines were slaughtered on day 28. The remaining animals were further monitored after the intake of contaminated feed was stopped. The daily intake of PFOS varied between 3.1 and 5.6 mg/d over the whole 28-day intake period. The model calculations show that the concentration in milk increases further after the intake of contaminated feed is stopped and, depending on the milk production, only reaches equilibrium after 4.5 (12.5 litres of milk/d); 3 (25 l/d) or 2 months (50 l/d). The corresponding concentrations in milk were respectively 240, 120 and 60 μg/l. Based on an intake of 3 mg/d and an

average milk production of 25l/d, we calculate a BTF_{milk} of 0.04 $mg.kg^{-1}$ fw/$mg.d^{-1}$ (30 days after intake of contaminated feed was stopped (Table 21)).

Table 20: Concentrations of PFOS (arithmetic mean and SD) in feed and tissues of dairy cows (Vestergren *et al.*, 2013)

	number of samples	PFOS concentration
intake media		
water	6	0.073 ± 0.014 (ng/l)
ensiled fodder	6	6.3 ± 2.1 (ng/kg)
barley	6	3.9 ± 1.7 (ng/kg)
tissue (bovine)		
liver	5	130 ± 32 (ng/kg)
blood	5	110 ± 19 (ng/kg)
muscle	5	110 ± 19 (ng/kg)
excretion media		
urine	10	3.6 ± 1.5 (ng/l)
faeces	10	<LOD (ng/kg)
milk	6	6.2 ± 1.1 (ng/l)

Kowalczyk *et al.* (2013) carried out a controlled feeding study on 6 dairy cattle divided into two groups, the first group being slaughtered after 29 days of exposure to PFAA contaminated feed, while the second group was put on a non-contaminated control diet for an additional 21 days. The feed consisted of hay and silage from farmland enriched with PFAA by the use of fertilisers and herbicides. The daily intake of PFOS (hay and silage grass) is estimated at 1172.39 μg/d. The concentrations after 29 days in milk, muscle, liver and kidney were respectively 9.06 μg/l, 145 μg/kg fw, 2952 μg/kg fw and 1074 μg/kg fw. After stopping the intake of contaminated feed, these values increased to respectively 33.09 μg/l, 178 μg/kg fw, 3964 μg/kg fw and 1408 μg/kg fw. The BTF for both periods which we calculate on the basis of these concentrations in feed and milk or tissues are shown in Figure 7. The figure shows that an equilibrium has not yet been reached after 29 days when the supply of PFOS-contaminated feed stops. The BTF value after 50 days and the average value that we calculate on the basis of both periods are summarised in Table 21.

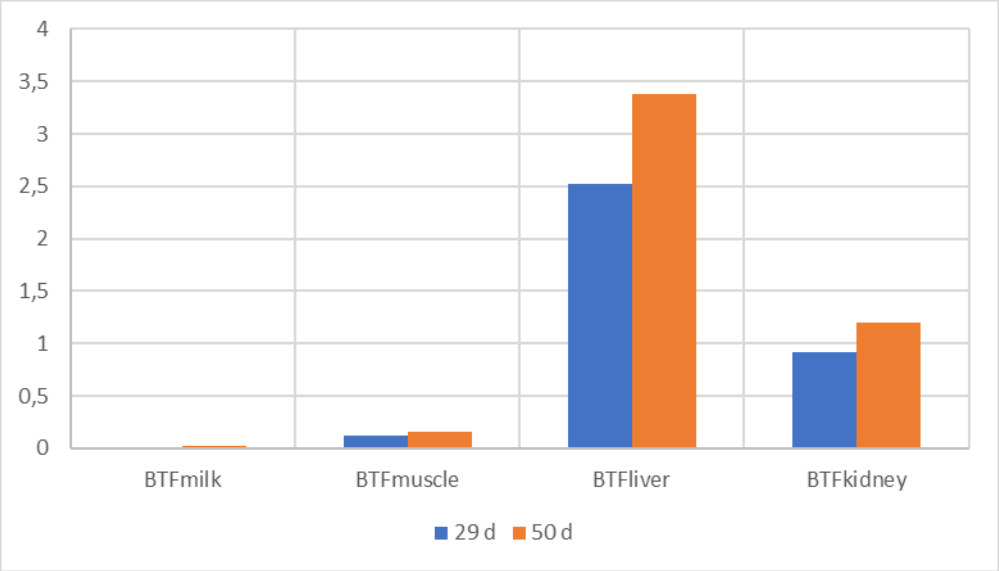


Figure 7: BTF (mg/kg fw)/(mg/d) in milk, muscle, liver and kidney in dairy cattle after 29 and 50 days (feeding with contaminated feed stops after 29 days), based on data from Kowalczyk *et al.* (2013).

Table 21: BTF values ($mg.kg^{-1}$ fw/$mg.d^{-1}$) for bovines and sheep for PFOS derived from available literature data.

	Vestergren et al., 2013 (1)	van Asselt et al., 2013 (1)	Kowalczyk et al 2013 (1) (day 29 (average)	Kowalczyk et al 2012 (2)
BTF_{milk}	0.021	0.040	0.028 (0.018)	0.076
BTF_{muscle}	0.071		0.152 (0.138)	0.387
BTF_{liver}	0.441		3.381 (2.950)	12.920
BTF_{kidney}			1.201 (1.058)	3.153

(1) Bovines; (2) sheep

Sheep

The transfer of PFOS through feed to sheep was studied by Kowalczyk *et al.* (2012). The authors describe a 21-day controlled feed study on sheep (n = 3 including 1 control). The animals were fed PFAA (perfluoroalkylic acid)-contaminated silage (maize). The intake of PFOS was estimated at 58.3 and 90.7 µg/d for sheep 1 and sheep 2. The average concentration in milk during the 21 days was 3.4 and 8.9 mg/l for sheep 1 and 2 respectively. On the basis of these data, we calculate an average BTF_{milk} for both sheep of 0.076 $mg.kg^{-1}$ fw/$mg.d^{-1}$. This value, together with the BTF values for muscle, liver and kidney, was included in Table 21. It was shown in the study by Kowalczyk *et al.* (2012) that there is a significant correlation between PFOS concentrations in blood plasma and milk (r^2 = 0.63). The duration of this study is 21 days, which may underestimate the potential for PFOS as it has been shown in bovines that concentrations in milk continue to increase after the intake of contaminated feed has stopped, as milk is a major elimination route for PFOS.

Chickens

In connection with the transfer of PFOS from feed to chickens and more specifically to eggs, our search resulted in 3 studies, i.e. Yoo et al. (2009), Yeung et al. (2009) and Hanell (2015). Only the latter publication reports paired data in chicken feed and eggs but does not indicate a daily intake which makes it impossible to calculate BTF_{egg}.

Conclusions

Our literature review covered 3 publications that inventorise the transfer of PFOS in feed to bovines, namely Vestergren *et al.* (2013), van Asselt *et al.* (2013)) and Kowalczyk *et al.* (2013). Originally, the calculation was made with a BTF calculated as average log BTF values of Vestergren *et al.* (2013), van Asselt *et al.* (2013) and Kowalczyk *et al.* (2013) (Table 22).

Table 22: Average BTF values (mg.kg^{-1} fw/mg.d^{-1}) for bovines from Vestergren *et al.* (2013), van Asselt *et al.* (2013) and Kowalczyk *et al.* (2013)

	average value
BTF_{milk}	0.023
BTF_{muscle}	0.099
BTF_{liver}	1.135
BTF_{kidney}	1.050

However, this decision was subsequently revised, see also Table 23. The BTF_{milk} derived in van Asselt et al. (2013) is not retained for the final calculation as it is based on PBPK modelling. Furthermore, it was also decided not to use the BTF values from Kowalczyk *et al.* (2013) for the determination of the final BTF values of PFOS because, on the one hand, very high concentrations of PFOS are administered in feed and, on the other hand, no equilibrium concentration was established at the end of the study. Based on this, the BTF values from Vestergren *et al.* (2013) are proposed for the derivation of soil remediation values of PFOS. The missing value for kidney in Vestergren *et al.* (2013) is supplemented with the value for day 50 from Kowalczyk *et al.* (2013). The values for bovines are summarised in Table 23.

For comparison, the most recent RIVM report (Wintersen et al., 2019) also uses the BTF values from Vestergren et al. (2013). The study by Kowalczyk et al. (2013) was taken into consideration but was not retained, as the values derived from it are based on studies in which significantly higher concentrations of PFAS were administered.

Table 23: Average BTF values (mg.kg^{-1} fw/mg.d^{-1}) for bovines from Vestergren *et al.* (2013) and Kowalczyk *et al.* (2013)

	BTF value
BTF_{milk}	0.021
BTF_{muscle}	0.071
BTF_{liver}	0.441
BTF_{kidney}	1.201(1)

(1) Kowalczyk *et al.* (2013)

For sheep we use, for milk and muscle, BTF values of respectively BTF_{milk} of 0.76 mg.kg^{-1} fw/mg.d^{-1} and BTF_{muscle} of 0.387 mg.kg^{-1} fw/mg.d^{-1} (Kowalczyk et al. , 2012). The values for liver and kidney are respectively BTF_{liver} of 12.92 mg.kg^{-1} fw/mg.d^{-1} and BTF_{kidney} of 3.15 mg.kg^{-1} fw/mg.d^{-1}.

2.7. TOXICOLOGY

2.7.1. INTRODUCTION

The overview of the toxicology of PFOS is mainly based on the review reports of CONCAWE (2016); OVAM (2018), EFSA (2008b), EFSA (2018c), ATSDR draft (2018), US-EPA (2016c), FSANZ (2016) and Pancras *et al.* (2018). The toxicokinetics and toxicology of PFOS are first discussed. A summary of the available toxicological reference values is given in section 2.7.4. A proposal for the toxicological reference values to be used for deriving soil remediation values is set out in section 2.7.5.
Based on the physicochemical properties of PFAS, exposure via intake of food and drinking water is highly likely. PFAS is also measured in air and dust, meaning that inhaled air, dust ingestion or dermal contact with dust or aerosols may also be possible routes of exposure.

2.7.2. TOXICOKINETICS

→ Absorption after oral intake

PFOS is easily absorbed after oral intake (PHE, 2009). In an oral rat study by 3M, ^{14}C-PFOS was administered to male rats in an average dose of 4.3 mg/kg bw/d. Within 48 hours, 5% of the radioactivity was found in the faeces and gastrointestinal tract. The researchers concluded that 95% of the radioactivity was absorbed. In addition, 86% of the administered radioactivity was found in the carcasses (after 24 and 48 hours) and traces of radioactivity in the urine (1-2% per day); no radioactivity was measured in the spleen and erythrocytes (Johnson *et al.*, 1979). For deriving soil remediation values, the oral absorption factor is equated by default to 1 (Cornelis *et al.*, 2012)

→ Absorption after inhalation

In 2000, the average concentration of PFOS in serum of 215 workers from the production of fluorinated chemicals (including 3M in Antwerp) was 1400 ng/ml (Olsen *et al.*, 2007b), while the average concentration of 645 blood donors to the Red Cross in the USA in the same year was 34.9 ng/ml (Olsen *et al.*, 2003). A higher concentration among workers than among the population may indicate a significant contribution from inhaled PFOS. Occupational exposure is likely to include the inhalation of aerosols whereby PFOS forms complexes with the airborne particles (ATSDR draft, 2018). Inhalation of soil particles is insignificant (Xiao *et al.*, 2015).
In accordance with Cornelis *et al.* (2012) we assume that absorption by inhalation and by the oral route is the same for both routes, i.e. 95%.

→ Absorption after dermal contact

After administration of one-time low doses of K-PFOS (up to 0.30 mg/kg) and the diethanolamine salt of PFOS (up tp 20 µg/kg) on the clipped, intact skin of rabbits, no elevated fluorine level was measured in the liver after 28 days (Johnson, 1995a; b). This indicates that dermal absorption of PFOS at low doses is not detectable.

Dermal absorption from soil particles is insignificant compared to exposure via oral intake of soil and dust (Cornelis *et al.*, 2012; Xiao *et al.*, 2015). For deriving soil remediation values **the dermal absorption factor from soil and dust is therefore set to 0**.
Dermal absorption from water is driven by the permeability coefficient (Kp, expressed in cm/h). There are no data for the Kp of PFOS. Washburn *et al.* (2005) mentioned a measured average Kp of $9.5.10^{-7}$ cm/h for the anion (ammonium perfluorooctanoate, AFPO) of PFOA; absorption through the skin may therefore be considered low. The calculated Kp of PFOS is 0.0362 cm/h (EpiSuite DermWin). The Kp for organic substances is calculated in Dermwin from the log K_{ow}; for PFOS a log K_{ow} of 6.28 was used for the calculation. The calculated Kp for PFOA is higher (0.144 cm/h). As such, using the measured Kp of APFO for PFOS is probably not an underestimation of the dermal absorption via water.
We propose using the measured **Kp of $9.5.10^{-7}$ cm/h** of APFO for deriving soil remediation values for **dermal absorption of PFOS from water**.

→ **Distribution**

Unlike most other persistent organic pollutants (POPs), PFOS has a low affinity for fats. PFOS binds to proteins located on the surface of cell membranes. In the liver of rats, PFOS binds to the fatty acid-binding protein (L-FABP) which would contribute to the high retention of PFOS in the liver (Luebker *et al.*, 2002). PFOS can accumulate in various organs, especially those with high blood flow (DEPA, 2015).
The highest concentrations are usually found in blood, liver, kidneys, lungs, spleen and bone marrow. Low concentrations occur in the heart, testes, fat, brain and muscles. Among the population, the PFOS concentrations in blood are between sub-ppb and 100 ppb levels (Loganathan & Kwan-Sing Lam, 2011). Median serum concentrations for Belgium, measured between 1998 and 2000, are 10.4 ng/ml for women and 17.6 ng/ml for men (Kannan & al., 2009). The median PFOS serum concentrations were also higher in men (43 ng/ml) than in women (24 ng/ml) in a study conducted on 56 coastal populations in northern Norway (Rylander *et al.*, 2009).
PFOS can migrate through the placenta in both humans and animals, and is found in breast milk (Stahl *et al.*, 2011).

→ **Metabolisation**

PFOS does not undergo any significant metabolisation and therefore accumulates in the body (Stahl *et al.*, 2011).

→ **Clearance**

PFOS is only eliminated very slowly from the human body; the half-life measured in serum in the general public in England is 9 years (PHE, 2009) and among retired workers from the fluorine chemical industry in the USA it is 5.4 years (Olsen *et al.*, 2007a). Half-lifes for animals are 132 and 110 days for male and female Java monkeys (Macaca fascicularis) respectively (Noker & Gorman, 2003), 200 days for Java monkeys (males and females) (Seacat *et al.*, 2002), 4 months for monkeys and 1 to 2 months for rodents (Chang *et al.*, 2012). Clearance seems to vary with the type of organism (e.g. shorter half-life for rodents than for monkeys) and with sex; the reason for this is not clear (CONCAWE, 2016). Clearance in rats is primarily via the kidneys and to a lesser extent via faecal

excretion, while in humans clearance via the kidneys appears to be negligible (EFSA, 2008b). Estimated half-life for humans is about 5 years (EFSA, 2018c).

2.7.3. EFFECTS ON TEST ANIMALS AND HUMANS

→ Acute toxicity

At the global level, criteria have been laid down for classifying substances as hazardous. Within the European Union (EU), these criteria are included in Regulation No 1272/2008 on classification, labelling and packaging (CLP) (EC, 2008). For a number of substances, the hazard classification is laid down in law at EU level, these are the so-called harmonised classifications; they are listed in Annex VI of CLP. PFOS has a harmonised classification; for acute toxicity the classification is: acutely harmful if inhaled or swallowed (Acute Tox. 4). PFOS is mildly irritating to the eyes but not to the skin (OECD, 2002).
The acute LC50, rat for PFOS in air is 5.2 mg/l, after 1 hour exposure (Rusch *et al.*, 1979); acute exposure of rats to 1.9 to 4.6 mg/l PFOS in air led to the following symptoms: signs of emaciation, snotting, discoloured urogenital area, respiratory disturbance and poor general condition (OECD, 2002).
The acute oral LD50, rat is 250 mg/kg bw, the symptoms are hypoactivity, discoloured urogenital zone, decreased tone in the limbs and ataxia (disorder between the motor nervous system and the muscles), swelling of the stomach, and lung congestion (3M-Company, 1999). According to PHE (2009) PFOS appears to cause moderate eye and skin irritation.
There is no known data on acute toxicity in humans following exposure via inhalation, oral ingestion, or dermal or ocular contact (PHE, 2009).

→ Subacute and sub-chronic toxicity

An overview of No (Lowest)-Effect-Concentrations (N(L)OEC) from a number of subacute and (sub)chronic animal studies is given in Table 24.
In a 90 d sub-chronic study, rats were given 0, 30, 100, 300, 1000, or 3000 mg PFOS/kg feed (0, 2, 6, 18, 60, or 200 mg/kg bw/d). In the 100 mg/kg group and above, body weight was lower than that of the control group, and all animals died before the end of the study. In the 300 mg/kg dosed group, all rats showed centrolobular to midzonal hypertrophy of liver cells and focal necrosis of the liver. A whole range of biochemical parameters (including glucose, urea nitrogen, creatinine phosphokinase and other enzymes, haematocrit and haemoglobin) were disrupted in all groups. The lowest dose (2 mg/kg bw/d) is therefore considered to be the LOAEL (Goldenthal *et al.*, 1978). In a 28-day oral study with rats, the LOAEL of 5 mg/kg bw/d was based on a reduction in body weight (Cui *et al.*, 2009).
Seacat *et al.* (2003) fed PFOS to rats at doses of 0, 0.5, 2.0, 5.0, and 20 mg/kg feed (0.0.05, 0.2, 0.4, and 1.5 mg/kg bw/d). The highest dose of 1.5 mg/kg bw/d was considered to be the LOAEL. After 14 weeks, male rats in this dose group showed increased absolute and relative liver weight, lowered cholesterol and increased urea nitrogen in the blood; female rats of the group receiving the highest dose showed increased relative liver weight and increased urea nitrogen in the blood. The dose of 0.4 mg/kg bw/d was considered to be the NOAEL.
In a study with Java monkeys, the animals received 0, 0.03, 0.15 and 0.75 mg K-PFOS/kg bw/d via oral intubation for 183 days (Seacat *et al.*, 2002). In the group with the highest dose, 2 out of 6 male monkeys died as a result of the administration, and the other animals had lower body weight and total cholesterol, increased liver weight and thyroid stimulating hormone (TSH) levels, among other

things, and lowered estradiol levels in the male monkeys. Changes in thyroid hormones and in high-density lipids were measured in the 0.15 mg/bw/d dose group. Contrary to the authors, the EFSA scientific panel is of the opinion that these changes are dose-related; EFSA therefore considers 0.03 mg/kg bw/d to be the NOAEL (EFSA, 2008b).

→ **Chronic studies and carcinogenicity**

Epidemiological studies with workers with work-related exposure to PFOS have not provided convincing evidence of an increased risk of cancer (EFSA, 2008b). Studies on the general public (with no work-related exposure to PFAS) have also not shown a direct correlation between PFOS exposure and cancer (US-EPA (2014) in CONCAWE (2016)).
Thomford (2002) conducted a study in which rats were given 0.5, 2, 5 or 20 mg PFOS/kg feed (0.04, 0.14, 0.36 and 1.42 mg/kg bw/d) over a 104-week period. Liver toxicity, characterised by significant increases in centrolobular hypertrophy, centrolobular eosinophilic granules, pigment and vacuolisation in liver cells, was observed in male and female rats of the groups with the two highest doses (0.36 and 1.42 mg/kg bw/d). Male rats administered 0.14 mg/kg bw/d also showed a significant increase in hepatocellular, centrolobular hypertrophy. Based partly on electron microscopic examination of the liver, the NOAEL for liver effects (not neoplastic) was set at 0.14 mg/kg bw/d. (EFSA, 2008b). Several studies show that PFOS interferes with fatty acid metabolism and lipid and lipoprotein metabolism, but the mechanism leading to liver toxicity in rodents and monkeys is not yet fully understood; one of the possible mechanisms is that PFOS competes with fatty acids, inter alia, to bind to the important fatty acid transport protein of the liver (Luebker *et al.*, 2002) and thus contributes to liver toxicity; this mechanism could also explain the lower levels of cholesterol.
A positive correlation was determined between the exposure and the incidence of hepatocellular adenoma in both males and females of the group with the highest dose (1.42 mg/kg bw/d) (Thomford (2002) in US-EPA (2014)). According to the EFSA panel, the evidence for the induction of thyroid and breast tumours is limited (EFSA, 2008b). In a similar study, hepatocellular adenomas were also observed in male and female animals. A hepatocellular carcinoma was observed only in the group of female rats given a dose of 20 ppm. There were no significant effects on the kidneys or bladder (Butenhoff *et al.*, 2012).
PFOS has a harmonised CLP classification as 'Suspected of causing cancer' (Carc. Category 2) (EC, 2008). US-EPA (2014) concludes that the evidence of carcinogenicity is 'suggestive' but not 'definitive', as the tumour incidence does not indicate a dose-response relationship. The International Agency for Research on Cancer (IARC) has not classified PFOS for carcinogenicity. Stahl *et al.* (2011) conclude that a genotoxic mechanism cannot be presumed, and that it is a question of whether it is not actually a secondary effect promoting tumour formation or an epigenetic[12] effect.

→ **Genotoxic effects**

K-PFOS was negative in the reverse mutation test with *Salmonella typhimurium* (Ames test) and in a mitotic-recombination test with *Saccharomyces cerevisiae* (Litton Bionetics, 1978). PFOS was negative in the *Salmonella-Escherichia coli* reverse mutation test with and without metabolic activation (S9) up to 5000 µg/plate (Mecchi, 1999). It did not induce any chromosome aberrations in cultured human lymphocytes, nor any unscheduled DNA synthesis (UDS) in primary cultured liver

[12] reversible hereditary changes in gene function without changes in the sequence (of base pairs) of DNA (Wikipedia)

cells of the rat up to 4000 µg/ml (Cifone, 1999). PFOS was also negative in the *in vivo* micronucleus test in bone marrow of mice after a single dose of 237.5, 450 and 950 mg/kg bw (Corning Hazleton, 1993).
Precursors including N-ethyl perfluorooctane sulfonamido ethanol (N-EtFOSE), N-ethyl perfluorooctane sulfonamide N-EtFOSA), N-ethyl perfluorooctane sulfonamido ethanol (NMeFOSE), N-methyl perfluorooctane sulfonamide (N-MeFOSA) and potassium N-ethyl-N((hepatodecafluoroctyl)-sulfonyl) glycinate (PFOSAA) were also tested and found negative in various *in vitro* and *in vivo* tests (EFSA, 2008b).
Based on the negative result in a whole range of *in vitro* and *in vivo* short-term studies, genotoxicity does not appear to be a property of PFOS or its salts (EFSA, 2008b).

→ **Development and reproduction toxicity**

Apelberg *et al.* (2007) investigated the association between PFOS concentrations in umbilical cord blood and gestational age, birth weight and height in 293 individual births from November 2004 to March 2005 in Baltimore (USA). PFOS was found in >99% of umbilical cord blood samples, with a median concentration of 5 ng/ml (<0.2-34.8 ng/ml). PFOS was significantly associated with small decreases in body weight and size, but not with birth length or gestational age. The concentrations of PFOS in umbilical cord blood were strongly correlated with those of PFOA. A Danish cohort study suggests that the association may be linked to PFOA rather than PFOS. In a cohort of 1400 women who gave birth between March 1996 and November 2002, the mean PFOS concentration in maternal plasma was 35.3 ng/ml (6.4-106.7 ng/ml) and in umbilical cord blood was 11±4.7 ng/ml (n=50). The plasma concentrations of the mothers showed no consistent association with birth weight at gestational age (Fei *et al.*, 2007).
Rat and mouse studies in which the animals were fed PFOS via a tube during pregnancy indicate that PFOS severely disrupts the postnatal survival of newborn rats and mice, and causes retardation in growth and development, which are associated with hypothyroxinemia (an abnormally low level of thyroxin, a thyroid hormone that regulates cell metabolism and growth, in the blood) in the surviving newborn animals. Rats were given PFOS doses of 1, 2, 3, 5 and 10 mg/kg bw/d during gestation days 2 to 21. Weight gain in the mothers was significant and reduced in a dose-related manner at 2 mg/kg bw and above. The T4 and T3 content in the maternal serum was reduced in all groups. At 3 mg/kg bw/d, 50% of the offspring died; at 5 mg/kg bw/d, 95% died 24 hours after birth. The maternal dose corresponding to the BMDL5[13] for the survival of the offspring on the eighth day after birth is 0.58 mg/kg, the NOAEL for development is 1 mg/kg (Lau *et al.*, 2003).
Luebker *et al.* (2005b) gave K-PFOS to female rats via tube feeding for 6 weeks before mating, during pregnancy and up to day 4 of lactation (total 62-67 days), in doses of 0, 0.4, 0.8, 1.0, 1.2, 1.6 and 2.0 mg/kg bw/d. Significant decreases in the length of pregnancy and in the viability of the offspring were observed from 0.8 mg/kg bw/d. An interval of BMDL5 values of 0.27 to 0.89 mg/kg bw/d was calculated for these effects. Luebker *et al.* (2005a) gave PFOS to male and female rats for 84 days (6 weeks before mating and during pregnancy) at doses of 0, 0.1, 0.4, 1.6 and 3.2 mg/kg bw/d. A late opening of the eyes in F1 young and a transient decrease in body weight of F2 young during lactation was observed at 0.4 mg/kg bw/d (LOAEL). The NOAEL for the late opening of the eyes was 0.1 mg/kg bw/d (Luebker *et al.*, 2005a).
Case *et al.* (2001) administered PFOS to pregnant New Zealand white rabbits via tube feeding at concentrations of 0.1, 1.0, 2.5 and 3.75 mg/kg bw per day from day 6 to day 20 of pregnancy. At the

[13] lower limit of the 95% confidence interval on the benchmark dose for a 5% increase in response above background incidence

two highest doses the birth weight was reduced and the ossification of the offspring was retarded. The NOAEL and LOAEL were 0.1 and 1.0 mg/kg bw/d for maternal toxicity (reduced weight gain) and 1.0 and 2.5 mg/kg bw/d for foetal toxicity, respectively.
A reproduction toxicity study over two generations in rats showed high susceptibility to PFOS (Christian *et al.*, 1999). PFOS was administered by tube at doses of 0, 0.1, 0.4, 1.6, and 3.2 mg/kg bw/d for 42 days before mating, and also in females during pregnancy and lactation. The length of pregnancy was significantly shorter at the highest dose. Lower birth weight and reduced survival was observed in the F1 generation at the highest doses of 1.6 and 3.2 mg/kg bw/d with respective mortality rates of 26% after 4 days and 45% after 1 day (and 100% thereafter). In the F2 generation of the group that had received a dose of 0.4 mg/kg bw/d, the birth weight was reduced (LOAEL). No other signs of toxicity were reported. The NOAEL was 0.1 mg/kg bw/d.
Grasty *et al.* (2003) investigated what the critical period is for prenatal exposure to PFOS by administering K-PFOS salt to pregnant rats at a dose of 25 or 50 mg/kg bw on the following gestation days: 2-5, 6-9, 10-13, 14-17 or 17-20, and at a dose of 25 mg/kg bw on gestation days 19-20. Neonatal mortality occurred in all groups, but the incidence increased as exposure occurred at a later point in the pregnancy, reaching 100% from the 17th to the 20th day. The late period in the pregnancy appears to be a highly vulnerable period.
PFOS has a harmonised CLP classification for reproduction with hazard statements: 'May damage the unborn child' (Rep. 1B) and 'May cause harm to breast-fed children' (EC, 2008).

→ **Neurotoxicity**

Changes in levels of thyroid hormones can affect the development of the brain and consequently affect the behaviour of the offspring. Prenatal exposure to PFOS did not affect learning behaviour and memory, but significant shortcomings were observed in the development patterns of choline acetyltransferase activity (an enzyme sensitive to thyroid hormone status) in rats with a LOAEL of 1 mg/kg bw/d (Lau *et al.*, 2003).
Administration of PFOS to 10 days old mice via gastric intubation at a dose of 0.75 or 11.3 mg/kg bw resulted in poorer performance in behavioural tests when the mice were 2 to 4 months old. There were no visible signs of clinical toxicity. According to the authors, the response was mediated through the cholinergic system (which stimulates the neurotransmitter acetylcholine) (Johansson *et al.*, 2008). In the reproduction toxicity study over two generations by Christian *et al.* (1999) temporary retardations in reflex development were observed in the F1 generation, which may indicate possible neurotoxicity of PFOS (Christian *et al.*, 1999).
Butenhoff *et al.* (2002) investigated functional and morphological changes in the nervous system of rats administered PFOS during pregnancy and lactation. Female rats were given an oral dose of 0, 0.1, 0.3 or 1.0 mg K-PFOS/kg/d on a daily basis throughout pregnancy until the 20th day after giving birth. The offspring were examined for growth, development, motor activity, learning and memory, acoustic shock reflex, various manifestations of behaviour, and brain weight until day 72 after birth. Male offspring of mothers who were given 1.0 mg K-PFOS/kg bw/d showed increased motor activity and reduced habituation on the 17th (but not on the 13th, 21st and 61st) day after delivery. As such, the NOAEL for neurotoxicity is 0.3 mg/kg/d.

→ **Immunotoxicity**

Adult mice were exposed to 0, 0.166, 1.66, 3.3, 16.6, 33 and 166 µg PFOS/kg bw/d daily via oral probe feeding for 28 days. Five days before euthanasia, they were immunised with a suspension of 25% sheep red blood cell (SRBC) via intraperitoneal injection. At the end of the study, lysosome activity

and PFOS concentrations were measured. There were no clinical symptoms of toxicity. Production of SRBC-specific immunoglobulin M was significantly suppressed in both sexes, whereby the males were more sensitive than the females, at doses ≥ 1.66 µg/kg bw/d (male mice) and ≥ 16.6 µg/kg bw/d (female mice). The NOAEL was 0.166 µg/kg bw/d, which is equivalent to a measured serum concentration of 17.8 ng/ml (Peden-Adams *et al.*, 2008).

Table 24: (Sub)acute and (sub)chronic animal studies with PFOS (sources: CONCAWE (2016), EFSA (2008b))

Animal	Administration	Duration of exposure	Parameter	Value	Effects	Reference
Rat	Inhalation	1 hour	LC50	5.2 mg/l	lethality	Rusch *et al.* (1979)
Rat	Inhalation	Acute	-	1.9 tot 4.6 mg/l	emaciation, snotting, discoloured urogenital area, disrupted breathing, poor general condition	OECD (2002)
Rat	Oral	Acute	LD50	250 mg/kg	hypoactivity, discoloured urogenital area, reduced muscle tone, ataxia, swelling of the stomach, lung congestion	3M-Company (1999)
Rat	Tube feeding	28 days	LOAEL	5 mg/kg bw/d	Decrease in body weight	Cui *et al.* (2009)
Rat	Feed	90 days	LOAEL	2 mg/kg bw/d	Liver, biochemical blood parameters, lower body weight, lethality	Goldenthal *et al.* (1978)
Rat	Feed	14 weeks	NOAEL	0.4 mg/kg bw/d	Liver weight, lower cholesterol, increased urea-N	Seacat *et al.* (2003)
			LOAEL	1.5 mg/kg bw/d		
Java monkey	Oral intubation	6 months	NOAEL	0.03 mg/kg bw/d	Contents of thyroid hormones, estradiol and lipids, lower body weight and cholesterol, lethality	Seacat *et al.* (2002)
			LOAEL	0.15 mg/kg bw/d		
Rat	Feed	104 weeks	NOAEL	0.14 mg/kg bw/d	Liver (e.g. hypertrophy, vacuolisation)	Thomford (2002)
			LOAEL	0.36 mg/kg bw/d		
			-	1.42 mg/kg bw/d	hepatocellular adenoma	

Rat	Feed	104 weeks	-	20 ppm in feed	hepatocellular carcinoma	Butenhoff *et al.* (2012)
Rat	Tube feeding	Day 2 to 21 of the pregnancy	BMDL5	0.58 mg/kg bw/d	Viability of the young	Lau *et al.* (2003)
			NOAEL	1 mg/kg bw/d	Development	Lau *et al.* (2003)
			LOAEL	1 mg/kg bw/d	Developmental neurotoxicity due to reduced levels of thyroid hormone in maternal serum	
			LOAEL	2 mg/kg bw/d	Maternal body weight	
Rat, female	Tube feeding	6 weeks before, during and 4 days after pregnancy	BMDL5	0.27 to 0.89 mg/kg bw/d	decrease in the length of pregnancy and the viability of the offspring	Luebker *et al.* (2005b)
Rat	Tube feeding	7 weeks before mating; for females: also during pregnancy and lactation	NOAEL	0.1 mg/kg bw/d	No signs of toxicity	Christian *et al.* (1999)
			-	1.6 and 3.2 mg/kg bw/d	F1 generation: lower birth weight and reduced survival	
			LOAEL	0.4 mg/kg bw/d	F2 generation: reduced birth weight	
Mice, 10 days old	Stomach intubation			0.75 and 11.3 mg/kg bw	poorer performance in behavioural tests (mice 2 to 4 months old)	Johansson *et al.* (2008).
Rat	Tube feeding,	Pregnancy and 20 days thereafter	NOAEL	0.3 mg/kg bw/d	Male offspring: motor activity and habituation	Butenhoff *et al.* (2002)
			LOAEL	1.0 mg/kg bw/d		
Rabbit	Tube feeding	Day 6 to 20 of the pregnancy	NOAEL:	0.1 mg/kg/d (maternal) 1 mg/kg/d (foetal)	Maternal and foetal developmental toxicity	Case *et al.* (2001)
			LOAEL:	1 mg/kg bw/d (maternal) 2.5 mg/kg bw/d (foetal)		

2.7.4. SUMMARY OF THE AVAILABLE TOXICOLOGICAL REFERENCE VALUES

A summary of the toxicological assessment values derived for PFOS by various bodies is given in Table 26.

→ **Oral - non-carcinogenic**

EFSA

In 2008, the CONTAM (Contaminants in the food chain) panel of EFSA derived a TDI for PFOS of **150 ng/kg bw/d** (EFSA, 2008b). This TDI is based on the NOAEL of 0.03 mg/kg bw/d from a sub-chronic study with Java monkeys, in which a decrease in total cholesterol in serum and high-density lipoproteins, an increased content of thyroid stimulant hormone (TSH) and a reduced concentration of triiodothyronine (T3) were all observed, at the next higher dose of 0.15 mg/kg bw/d (Seacat *et al.*, 2002). The NOEAL was associated in females with a plasma concentration of 13.2 µg/ml at the end of the exposure period (day 183). Since the estimated half-life for PFOS in monkeys is 200 days, this internal dose cannot be considered steady-state.

An uncertainty factor of 200 was applied to the NOAEL of 0.03 mg/kg bw/d, composed of 100 for inter- and intraspecies variability, and 2 for uncertainties related to the relatively short duration of the test and the elimination kinetics of the internal dose.

In a scientific opinion from 2018, EFSA published a preliminary[14] oral guideline value for PFOS (EFSA, 2018c). The derivation of the Health Based Guideline Value (HBGV) is based on epidemiological studies which were not yet available in 2008; EFSA identified increases in total cholesterol levels in adults and decreases in response with antibodies in childhood vaccination as critical effects. With regard to the latter effect, following a systematic review of the immunotoxicity associated with exposure to PFOS and PFOA, the NTP (National Toxicology Program) concluded that these substances indeed pose a risk to the human immune system, in particular by suppressing the response with antibodies (NTP, 2016).

For adult serum cholesterol, EFSA used benchmark dose (BMD) modelling for a 5% increase in total cholesterol, for three studies with more than 400 subjects and with results published as quantiles (Steenland *et al.*, 2009; Nelson *et al.*, 2010; Eriksen *et al.*, 2013). An overview of the BMD analysis is given in Table 25.

The three studies give very similar BMDL5 values, i.e. 21-25 ng/ml plasma, which corresponds to an estimated chronic daily intake of 1.7-2.0 (median 1.8) ng/kg bw/day according to a human PBPK model (EFSA, 2018c). For children, the lowest BMDL5 for response with antibodies after vaccination is 10.5 ng/ml. This value is calculated on the basis of an intake by the mother of 1.8 ng/kg bw/day and a period of 6 months with only breastfeeding. For a third critical endpoint, i.e. lower birth weight, the BMDL5 (21 ng/ml) was about the same as for elevated cholesterol. The CONTAM panel considered the three endpoints as critical endpoints and considered that these human studies provided sufficient evidence to derive a health-based guideline value; the reference values for the

[14] Due to the nature of the scientific uncertainties described in this opinion and in the minutes of the expert meeting of 24 September 2018 (EFSA/CONTAM/3503) (https://www.efsa.europa.eu/sites/default/files/news/efsa-contam-3503.pdf), and the possible application of the forthcoming Scientific Committee guidance on combined exposure to multiple chemicals, the conclusions of this assessment will be reviewed in parallel with the finalisation of the EFSA scientific opinion on The risks to human health related to the presence in food of perfluoroalkylated substances other than PFOS and PFOA (EFSA-Q-2017-00549). The indicative timeline for this is December 2019. Until such time, the conclusions and derived tolerable weekly intakes shall be considered provisional.

different endpoints in the studies range from 1.7 to 2.0 ng/kg bw/day; the mean value is 1.84. EFSA rounds off this value to **1.8 ng/kg bw/day (**Table 25**)**; this value is therefore proposed by EFSA as a possible new TDI for PFOS. No additional safety factor was applied, as the BMD modelling is based on large epidemiological studies of the general population, including potentially sensitive groups.

Table 25: Overview of the BMD analysis (EFSA, 2018c)

Human endpoint	BMD5 (ng/ml)	BMDL5 (ng/ml)	Intake via food* (ng/kg bw/d)	Number of persons (cohort)	Type data	Model	Reference
Total cholesterol	27	25	2.0	46,294**	deciles	Log normally cumulative	Steenland *et al.* (2009)
	31	22	1.8	753 (Danish cohort 1996-2002)	octiles	Linear square root	Eriksen *et al.* (2013)
	31	21	1.7	860 (NHANES)[15]	quartiles	Exponential	Nelson *et al.* (2010)
Response to vaccination (children)	11.6	10.5	1.8	413 (Faroese, birth cohort 1997-2002)	deciles	Logarithmic	Grandjean (2012)
Birth weight	36	21	1.9	901 (Norwegian birth cohort)	quartiles	Logarithmic	Whitworth *et al.* (2012)

* estimated value, corresponding to the BMDL5 of a PBPK model (rounded numbers)
** local residents (age ≥ 18 years) who drank water contaminated with PFOS from a perfluoropolymer producing chemical plant in West Virginia for at least 1 year.

Taking into account the long half-life of PFOS, a Tolerable Weekly Intake (TWI) of 13 ng/kg bw per week (7*1.8 = 12.6 and rounded off to 13) was determined.

A new EFSA risk assessment, which was published after the finalisation of this report, includes a tolerable weekly intake (TWI) based on epidemiological data to specifically protect infants. The TWI calculated as the sum of PFOA+PFNA+PFHxS+PFOS (which contribute most to human exposure) is 4.4 ng/kg bw/week. Effects on the immune system were considered the most critical endpoint for the risk assessment. Equal potencies were assumed for the four PFASs (EFSA CONTAM Panel *et al.*, 2020).

The Netherlands

RIVM used the TDI of 150 ng/kg bw/day derived by EFSA in 2008 to derive, for instance, a drinking water limit.

In March 2019, RIVM published a memorandum advising on the new risk limits for a.o. PFOS for soil and groundwater 'for purpose of a temporary framework for the application of soil and dredged material on or in the soil' (RIVM, 2019). The human reference value used to cover the human risk is a toxicological maximum tolerable risk level of 0.00625 µg/kg bw/d; this value was calculated from a health-based guidance value for PFOA of 12.5 ng/kg bw/d (Zeilmaker & Janssen, 2016) and a relative potency factor of 2 (Zeilmaker *et al.*, 2018). RIVM did not use the provisional TDI of EFSA (2018) as RIVM (and several other European scientific institutes) raised substantive objections to the evaluation by EFSA. RIVM recognises that the current evaluation by EFSA may lead to a possible tightening of the human risk limit.

[15] National Health and Nutrition Examination Survey (USA)

Great Britain
In 2006, UKCOT[16] derived a provisional TDI of **300 ng/kg bw/d**. This was based on the NOAEL of 0.03 mg/kg bw/d for reduced serum concentrations of triiodothyronine in Java monkeys after 6 months exposure (Seacat *et al.*, 2002). The uncertainty factor was 100 (for inter- and intraspecies variability). In 2009, UKCOT confirmed the TDI of 300 ng/kg bw/d for PFOS (FSANZ, 2016).

Denmark
In 2015, Denmark derived a TDI of **30 ng/kg bw/d**, based on developmental toxicity and increased liver weight in rats as the most critical endpoints (DEPA, 2015). The basis for the TDI is the $BMDL_{10}$ of 0.033 mg/kg bw/d which US-EPA (2014) (see below) derived from the chronic rat study by Thomford (2002). The uncertainty factor for interspecies differences consists of a substance specific factor of 41 for pharmacokinetic differences based on clearance rate for serum (CL)[17], and a general uncertainty factor of 3 for pharmacodynamic differences. The uncertainty for intraspecies variability is 10.

Germany
In 2016, the German Human Biomonitoring Commission (HBM) set a HBM I value of **5 ng/ml blood plasma** for the general population (Umweltbundesamt, 2016). The HBM I value is a concentration in a body matrix at or below which no adverse health effects are to be expected. For PFOS, the HBM I value is based on epidemiological studies and critical endpoints such as elevated cholesterol levels and disturbed immune response after vaccination. Analogy with results from animal tests increases confidence in the HBM I value of PFOS, according to the authors (Apel *et al.*, 2017).

Sweden
In a study commissioned by the Swedish government, no TDI was derived, but a safe serum concentration was (Derived no effect level (DNEL) (Borg & Håkansson, 2012). The lowest DNEL calculated was that for immunotoxicity, which is **0.12 ng/ml serum**. This value is based on a NOAEL of 17.8 ng/ml serum from the subacute study with mice by Peden-Adams *et al.* (2008) and an uncertainty factor of 150 (6 for extrapolation from subacute to chronic exposure, 2.5 for toxicodynamic differences (no factor for toxicokinetics because serum concentrations of humans and animals are compared), and 10 for intraspecies variability). The DNELs for liver and reproductive toxicity (endpoints relevant for multiple PFAS) were higher (162 and 196 ng/ml serum, respectively).

US-EPA
In 2014, US-EPA derived an RfD of 0.03 µg/kg bw/d; this RfD, like the Danish TDI, is based on developmental toxicity and increased liver weight in rats as the most critical endpoints identified in the study by Thomford (2002) (US-EPA, 2014).
In 2016, US-EPA derived an RfD of **20 ng/kg bw/d** based on reduced birth weight in rats in the two-generation study by Luebker *et al.* (2005b). Using a PBPK model, a human equivalent dose (HED NOAEL) was calculated of 0.00051 mg/kg/d corresponding to a NOAEL of about 30% of a steady-state concentration. An uncertainty factor of 30 was applied to this, consisting of 10 for interspecies variability and 3 for toxicodynamic differences between rat and human (US-EPA, 2016c).

ATSDR
ATSDR published a draft toxicological profile of 13 PFAS (ATSDR draft, 2018). For PFOS, an intermediate oral MRL (minimum risk level) of **2 ng/kg bw/d** is derived. The critical effect for ATSDR was late eye opening and temporary lower body weight of the F2 generation during lactation (Luebker *et al.*, 2005a). An uncertainty factor of 30 was applied to the HED (Human Equivalent Dose)

[16] United Kingdom Committee on Toxicity of Chemicals in Food, Consumer Products and the Environment
[17] CLrat/CLhuman = 0.0033 l/kg/d divided by 0.000081 l/kg/d

NOAEL of 0.000515 mg/kg/d, which is composed as follows: 3 for extrapolation from rat to human with adjustments for dosimetry, and 10 for intraspecies variability. In addition, a modifying factor of 10 was used because immunotoxicity might be a more sensitive endpoint than developmental toxicity (ATSDR draft, 2018).

Australia and New Zealand

In 2017, Food Standards Australia New Zeeland (FSANZ) determined a TDI for PFOS of **20 ng/kg bw/d** (FSANZ, 2017). FSANZ believes that there is insufficient epidemiological data available and therefore bases the TDI on experimental animal studies. This TDI is based on a multi-generation reproductive toxicity study in rats in which a reduced increase in body weight of mothers and offspring was observed (Luebker *et al.*, 2005b). The purity of the product used was 86.9%, prompting the CONTAM panel to question whether the effects observed and their extent are due solely to PFOS (EFSA, 2018c).

The starting point of FSANZ (2017) for the derivation of the TDI was the serum concentrations from which an HED was calculated by a PBPK model. An uncertainty factor of 30 was applied to this, consisting of 10 for interspecies variability and 3 for toxicodynamic differences between species. FSANZ (2017) also calculated HEDs and TDIs from other animal studies; these TDIs remained within one order of magnitude of the selected TDI of 20 ng/kg bw/d and were 100 and 20 ng/kg bw/d respectively for female and male rats (Thomford, 2002; Butenhoff *et al.*, 2012), 40 ng/kg bw/d for female rats (Lau *et al.*, 2003; Thibodeaux *et al.*, 2003), and 100 ng/kg bw/d for female monkeys (Seacat *et al.*, 2002).

→ **Inhalation – non-carcinogenic**

Compared to oral intake data, there is little data available on exposure via inhalation. The importance of this route of exposure is therefore unclear (CONCAWE, 2016).

The only inhalation study available is an acute lethality study in rats (Rusch *et al.*, 1979); no quantitative human data on inhalation are available. Consequently, there are not enough data to derive a reference concentration (RfC) (US-EPA, 2016c). ATSDR also concludes that there is insufficient data to derive a minimum risk level (MRL) for inhalation for acute, intermediate or chronic exposure (ATSDR draft, 2018).

→ **Reference values for carcinogenic effects**

PFOS has a harmonised CLP classification as 'Suspected of causing cancer' (Carc. Category 2) (EC, 2008). US-EPA (2014) concludes that the evidence of carcinogenicity is 'suggestive' but not 'definitive', as the tumour incidence does not indicate a dose-response relationship. In 2016, US-EPA decided that the burden of proof for cancer potential was too limited to carry out a quantitative cancer assessment (US-EPA, 2016c). IARC has not classified PFOS for carcinogenicity.

The consulted sources do not provide reference values for carcinogenic effects.

Table 26: Toxicological criteria for PFOS

Body	Type value	Value	Basis	Critical effect	Study	Factors	Reference
Oral intake (mg/kg.d)							
EFSA	*TDI*	*$1.5.10^{-4}$*	*NOAEL*	*Decrease in total cholesterol and T3 and increase in TSH levels*	*Java monkeys Seacat et al. (2002)*	*200 (100 intra- and interspecies; 2 relatively short test duration and clearance kinetics)*	*EFSA (2008b)*
EFSA	*HBGV - draft*	*$1.8.10^{-6}$*	*BMDL5*	*Increase in total cholesterol and decrease in response with antibodies in vaccination*	*Epidemiological studies: Steenland et al. (2009); Nelson et al. (2010); Eriksen et al. (2013)*		*EFSA (2018c)*
The Netherlands	*TDI*	*$1.5.10^{-4}$*	*TDI taken from EFSA (2008b)*				*Value expected to be revised Pancras et al. (2018)*
	Maximum tolerable human risk level	*6.10^{-6}*	*Health-based guideline value PFOA*			*2 (relative potency factor)*	*RIVM (2019)*
United Kingdom	*TDI*	*3.10^{-4}*	*NOAEL*	*Decrease of the T3 content*	*Java monkeys*	*100 (intra- and interspecies)*	*FSANZ (2016)*
Denmark	*TDI*	*3.10^{-5}*	*BMDL10*	*Developmental toxicity and increased liver weight*	*Rat - Thomford (2002)*	*1230 (interspecies: 41 pharmacokinetic*	*DEPA (2015)*

						and 3; intraspecies 10)	
USA	*RfD*	*2.10^{-5}*	*HED NOAEL*	*Developmental toxicity and increased liver weight*	*Rat - Luebker et al. (2005b)*	*30 (3 toxicodynamic and 10 intraspecies;)*	*US-EPA (2016c)*
ATSDR	*MRL - proposal*	*2.10^{-6}*	*HED NOAEL*	*Young: late opening of eyes and lower body weight*	*Rat - Luebker et al. (2005a)*	*300 (3 interspecies and dosimetry; 10 intraspecies; 10 immunotoxicity potentially more sensitive endpoint)*	*ATSDR draft (2018)*
Australia and New Zealand	*TDI*	*2.10^{-5}*	*HED*	*Reproductive toxicity - reduced increase in body weight of mothers and young*	*Rat - Luebker et al. (2005b)*	*30 (10 interspecies; 3 toxicodynamic)*	*FSANZ (2016)*
Serum concentration ng/ml							
Germany	*HBM I*	*5*	*-*	*Increased cholesterol and disrupted immune response after vaccination*	*Epidemiological studies*	*-*	*Apel et al. (2017)*
Sweden	*DNEL*	*0.12*	*NOAEL*	*Immunotoxicity*	*Mouse - Peden-Adams et al. (2008)*	*150 (6 subacute to chronic; 2.5 toxicodynamic; 10 intraspecies)*	*Borg and Håkansson (2012)*

2.7.5. PROPOSAL FOR TOXICOLOGICAL REFERENCE VALUES TO BE USED

The current TDI of EFSA is based on a study with monkeys. The new, still provisional, TDI is based on epidemiological studies, due to differences in toxicokinetics between animals and humans, the relevance to humans of observed effects in animals and the underlying mechanisms of action. In making this choice, the Netherlands and Denmark posed a number of questions (EFSA, 2018a).
Denmark bases its TDI on the same rat study as the one on which US-EPA based its 2014 RfD, i.e. the study by Thomford (2002). Denmark also looked at human studies for its derivation of the TDI in 2015, but these were found to be unsuitable. At an expert meeting with EFSA in September 2018, Denmark stated that the difference between EFSA and Denmark in the selection of the basis for the TDI was due to different expert opinions on the robustness of the human data, and that Denmark was in the process of carrying out a scientific assessment of the draft Scientific Opinion of EFSA to see whether that difference still exists (EFSA, 2018a). In 2016, US-EPA derived a new RfD, which was based on a different study than that of 2014, meaning that the Danish TDI cannot be selected as a reference value.
At the same expert meeting with EFSA, Germany stated that it could accept the use of total cholesterol as a biomarker for PFOS exposure, albeit with a number of suggestions for clarification in the final Scientific Opinion of EFSA. However, the study by Grandjean (2012) on the reduced formation of antibodies after vaccination would not be used by Germany to derive the TDI, as the inhabitants of the Faroe Islands have a relatively high exposure to a large number of persistent contaminants due to the high consumption of fish and whale meat/blubber, which accumulates in the food chain. As such, other environmental contaminants with high persistence must be considered as disruptive elements. In the Faroese study, only PCBs in children were taken into consideration, and not other contaminants (e.g. heavy metals) or combined exposure as such. Furthermore, it is not clear why a large proportion of children did not see a strong decrease in antibody levels in the follow-up study at the age of 13, and why at this age the trends for tetanus antibodies and PFAS were mostly positive (Grandjean *et al.* (2017); (EFSA stated that this study was published after the deadline for inclusion in the literature list). Germany presents the evidence for reduced formation of antibodies after vaccination caused by PFOS/PFOA as limited to moderate and believes that more (preferably prospective) studies with more statistical strength are needed, with a long lactation period (age 0,(up to 1.5 years) and also the measurement of functional parameters of the immune system and metabolic parameters. Research is also needed to clarify the mechanism of action. Germany also has questions about translating the serum concentration into an external dose, as food concentrations often originate from 'hot spots' and may therefore lead to a significant overestimation of the exposure of people not exposed to food and drinking water from hot spots.
The Netherlands is in favour of making optimal use of epidemiological data in a risk assessment, but believes that the available epidemiological information is not suitable for deriving the HBGV; for example, on the basis of the data from Steenland *et al.* (2009) it is not possible to determine the contribution of each individual PFAS to increased cholesterol levels. EFSA admitted that there may indeed still be some uncertainty as to which substance in the PFAS family causes which effects and that this has been addressed in the 'uncertainties' section of the EFSA opinion. According to EFSA (2018c) the total increase is attributed to both PFOS and PFOA, but if one of the PFAS is fully responsible for the effect on cholesterol (which is the assumption when calculating an individual BMD), then the other PFAS cannot have any impact, according to the Netherlands. A second objection of the Netherlands is that the BMD approach used is not in line with the BMD guideline of EFSA: for example, the confidence interval is not correctly calculated because only the 'best fit' model is taken into account and not the average of all models applied. EFSA agrees that the guideline for BMD from epidemiological studies is less developed and harmonised than for experimental studies.

In a third objection, the Netherlands argues that the lowest quantile is taken as a proxy for background, whereby the background is strongly determined by the number of quantiles. Suppose that the number of quantiles in the study by Steenland *et al.* (2009) was not ten but five, then the total cholesterol content in the lowest quantile would have been higher and, consequently, the estimated BMD too (EFSA, 2018a). To be continued.
In the Netherlands, a maximum tolerable human risk level (MTR) was derived (RIVM, 2019), which is lower than the current TDI of EFSA EFSA (2008a). RIVM advises using this MTR within the temporary framework for the reuse of soil and dredging, but recognises that the opinion may change when EFSA publishes a new and final TDI.
The TDI of the United Kingdom is relatively high and does not correct for the duration of the study on which it is based. As such, this TDI is not selected as a reference value for PFOS.
Germany and Sweden have a reference value for PFOS as a biomarker in serum derived from 5 and 0.12 ng PFOS/l respectively, based on epidemiological data (Germany) or animal tests (Sweden). Both values are significantly lower than the BMDL5 of EFSA (21-25 ng/ml). Neither Germany nor Sweden translate the internal standard into an external standard, which is necessary for soil modelling.
US-EPA and the draft ASTDR base their derivation on a different long-term rat study by Luebker *et al*. for developmental toxicity. The HED NOAEL of both bodies is comparable. The major difference between the two lies in the application of the additional safety factor of 10 by ATSDR for the fact that immunotoxicity may be a more sensitive endpoint than developmental toxicity. Immunotoxicity is recognised as an effect but has not yet been quantified (NTP, 2016). The MRL of ATSDR (2 ng/kg bw/d) is very close to the provisional TDI of EFSA (1.8 ng/kg bw/d). The MRL is still a proposal and therefore cannot be selected as a reference value. The RfD of US-EPA (2016c) is 20 ng/kg bw/d.
In a joint derivation of the TDI, Australia and New Zealand derive a TDI of 20 ng/kg bw/d. They started from the same rat study as US-EPA (2016c) and used the same uncertainty factors to arrive at the same reference value.
Australia and New Zealand believe that the available epidemiological data are not suitable to derive an HBGV for PFOS (FSANZ, 2016). FSANZ evaluated available epidemiological studies on the relationship between PFOS exposure and cholesterol in serum, birth weight and immunotoxicity. The report noted that, overall, the cross-sectional studies come to a fairly consistent conclusion of a positive association between total and LDL cholesterol at low PFOS concentrations in serum, with plateau formation at higher PFOS concentrations. However, a number of limitations were observed, for example that some studies note a correlation between PFOS and PFOA concentrations but do not adjust the results for each individual substance. Similarly, populations with high exposure to PFAS may also have been exposed to other contaminants, but these were not taken into consideration in the studies, and most studies are not adapted for diet or consider the impact of glomular filtration rate (GFR) of the kidneys. As regards birth weight, PFOS concentrations in human studies appear to be lower than those in animal studies, suggesting an effect on birth weight. In general, the studies with numerical data report an association, but in the absence of quantitative data from studies that do not report an effect, there is a risk of selective bias in publications that affect the burden of proof. FSANZ has decided that it is currently not possible to determine whether the association reflects a causal relationship or is the result of a third factor that alters both the PFAS concentration and the birth weight. For example, changes in GFR that occur during pregnancy are likely to affect both birth weight and the clearance rate of PFAS. As regards immunotoxicity, NTP (2016) concluded that PFOS is probably hazardous to the human immune system. This decision was based on a 'high level of evidence' that PFOS suppresses antibody response in animal tests and a 'moderate level of evidence' based on epidemiological studies that higher serum concentrations are associated with suppression of the antibody response. The aim of the NTP report was to identify the hazard, and it does not say at what level of exposure the immune function in humans is affected. FSANZ has also reviewed a number of studies and concludes that there are still too many uncertainties to derive a reliable NOAEL or LOAEL for adverse effects on the immune system. For these reasons, FSANZ based its TDI

on results from animal tests, whereby the study by Luebker *et al.* (2005b) was ultimately selected, because this gave the lowest TDI (FSANZ, 2016).
The MRL of ATSDR (2018) is a proposal and therefore cannot be selected as a reference value.

The RfD of US-EPA (2016c) of **20 ng/kg bw/d** is proposed as a toxicological reference value for the calculation of the soil remediation value based on the following arguments:

- the relevant parties recognise that the current standard of EFSA is too high
- the more stringent EFSA standard is still provisional
- the Dutch MTR is more protective than the current TDI of EFSA, but is likely to be reviewed when EFSA publishes its final (more stringent) TDI
- the MRL of ATSDR is still provisional
- the RfD is based on a long-term study
- the value of the RfD is the same as that of Australia and New Zealand
- the derivations of US-EPA and Australia/New Zealand are recent

Because there is no toxicological reference value for exposure via inhalation, this is calculated from the TDI (20 ng/kg bw/d) with the following parameters: 70 kg body weight, 20 m³/day of breathing volume and 95% inhalation absorption (equivalent to oral absorption). The calculation results in a **tolerable concentration in air (TCA) of 70 ng/m³.**
In order to have an idea of the impact on the soil remediation value, scenarios are also calculated with the MTR of the Netherlands (6.25 ng/kg bw/d) and the TDI proposal of EFSA (1.8 ng/kg bw/d).

2.8. ECOTOXICOLOGY

For the evaluation of ecotoxicological effects, no new primary sources and/or databases were consulted to derive possible new ecotoxicological values. However, it was examined whether substantiated ecotoxicological values have recently been derived by other regulatory bodies. Based on the guidelines for drawing up soil remediation values (Cornelis and Touchant, 2016), the following sources were consulted:

- US EPA: http://www.epa.gov/ecotox/ecossl/index.html
- CCME Canada: http://www.ccme.ca/en/resources/canadian_environmental_quality_guidelines/index.html
- RIVM Netherlands (intervention values, national reference values: http://www.rivm.nl/rvs/Normen/Milieu/Bodeminterventiewaarden, http://www.rwsleefomgeving.nl/onderwerpen/bodem-ondergrond/bbk/instrumenten/nobo
- ECHA database: http://echa.europa.eu/information-on-chemicals

The results of this inventory are summarised in section 2.10.3. Ecotoxicological reference values

2.9. LEGAL LIMITS

2.9.1. OUTDOOR AIR AND INDOOR AIR

PFOS is a low-volatile substance. PFOS does not occur in the WHO Air quality guidelines for Europe (WHO, 2000) or those of other bodies (ANSES, RIVM, Germany, US-Clean air Act, LCI (lowest concentration of interest) for indoor air).

PFOS is not included in the Flemish Indoor Environment Decree (BS, 2018) and also does not appear in the WHO guidelines for indoor air quality (WHO, 2010) or in the list of LCI (lowest concentration or interest) substances for indoor air (EU-LCI, 2016).

The TCA is 70 ng/m³.

2.9.2. DIET AND FOOD

No European legal restrictions (EC, 1998; 2002; 2011).

2.9.3. DRINKING WATER

The drinking water standards of various countries/bodies are set out in Table 27. On 1 February 2018, the European Commission adopted a proposal to revise the Drinking Water Directive 98/83/EC (EC, 1998). This sets the drinking water standard at 0.1 µg/l for individual PFAS (including PFOS) and 0.5 µg/l for PFAS total (EC, 2018).
The Netherlands applies a risk limit value for PFOS in drinking water of 0.53 µg/l; this value has been calculated on the basis of the TDI of EFSA (2008) ($1.5.10^{-4}$ mg/kg/d), the allocation of 10% of the TDI to drinking water, a body weight of 70 kg and a drinking water intake of 2l/day (Moermond *et al.*, 2010). It is expected that the TDI of PFOS for the Netherlands will still be revised as the value is based on the current TDI of EFSA, which is generally considered to be too high.
In Germany, the guideline value (Leitwerte) for drinking water is 0.1 µg/l (UBA, 2017). This value is based on an average of the results of epidemiological studies (UBA, 2016b). The sum parameter for 12 PFAS in drinking water in Denmark is also 0.1 µg/l (DEPA, 2015). In the United Kingdom the drinking water limit is 0.3 µg/l, in Sweden it is 0.09 µg/l (CONCAWE, 2016).
The United States of America has a lower health-based advisory value of 0.07 µg/l for drinking water, which is a sum parameter for PFOS and PFOA (US-EPA, 2016). Australia uses the same value (0.07 µg/l) as the sum parameter for PFOS and PFHxS (Australia, 2016).
In the context of derivation of soil remediation values, the value of **0.1 µg/l** of the European Commission has been selected**.**

Table 27: Drinking water standards of various countries/bodies

Substance	Concentration	Reference value	Country/region	Reference
PFOS/PFHxS	0.07 µg/l	Drinking water quality value	Australia	Australia (2016)
PFOS	0.53 µg/l	Drinking water reference value	The Netherlands	Moermond *et al.* (2010)
PFAS separately and total	0.1 µg/l for individual PFAS and 0.5 µg/l for PFAS total	Proposal in the framework of the revision of Annex 1 of the drinking water Directive 98/93/EC	EU	EC (2018)
PFOA + PFOA	0.07 µg/l	Health Advisory for lifetime exposure	USA	US-EPA (2016a)
PFOS	0.1 µg/l	QCdw with 10% allocation of total oral intake to drinking water		DEPA (2015)
PFOS	0.1 µg/l	Guideline value	Germany	UBA (2017)

2.10. CALCULATION OF THE SOIL REMEDIATION VALUE

The calculations of the soil remediation values were made with a modified S-Risk version 1.3 Application I for the calculations, Application II with modified buffer space (0.75 m) for interpretation of exposure routes and exposure pathways. To avoid the use of the K_{ow}, S-Risk version 1.3 was specifically adapted for PFAS on the VITO test server. Transfer to plants can therefore be calculated with BCF factors based on dry matter concentration in the soil, whereas normally BCF factors for organic substances are expressed on pore water concentrations. Moreover, PFOS was considered a non-dissociative substance in S-Risk in the calculations (PFOS is a dissociative substance), which means that Kd can be calculated directly from the organic carbon content in the soil and K_{oc} without the use of K_{ow}. The user manual of S-Risk states "If a K_{oc}-value is available for a dissociative substance at the correct soil pH, it is also possible to leave the dissociative option button unchecked while filling out the required K_{oc} value. However, the calculations must only be carried out for the applicable pH range".

2.10.1. GROUNDWATER

The soil remediation value for groundwater has a human health based underpinning, and corresponds to the drinking water standard if this has a toxicological basis (Cornelis & Touchant, 2016). The drinking water limit of 0.1 µg/l included in the revision of the European Drinking Water Directive is a general limit primarily based on feasibility and does not have a direct substance-specific toxicological link, and moreover it is also a proposal. Therefore, the soil remediation value for groundwater is calculated using the following formula (Cornelis and Touchant, 2016):

$$SRV = \frac{TDIoral\ x\ 1000\ x\ RF\ x\ BW}{(Q + Qeq)}$$

Whereby
SRV soil remediation value (µg/l)
TDI tolerable daily intake (mg/kg/d)
RF reduction factor (standard 0.2)
BW body weight (60 kg)
Q drinking water consumption (2l)
Qeq drinking water equivalent for inhalatory and dermal exposure (l)

EFSA carried out a comprehensive study of chronic exposure to PFOS via food, setting upper and lower limits for minimum, average and maximum intakes (EFSA, 2018b). In the lower limits of average exposure, the highest contribution for drinking water (up to 10%) was found in infants and young children; in adults the contribution was up to 3%. In a Swedish study measuring the relative contribution of different pathways to total exposure, the contribution of PFOS via drinking water ranged from 0.57 to 0.68% (Haug *et al.*, 2011). A reduction factor of 10% for the derivation of a drinking water standard would therefore be sufficient. Germany, for example, allocates 10% of the TDI to drinking water upon derivation of a drinking water guideline value (UBA,2016). However, since a soil remediation value is not a drinking water standard, at the request of OVAM, we retain the standard value of 20% of WHO for the SRV.
US-EPA applies a Relative Source Contribution (RSC) of 20% for PFOS in its derivation of a health-based drinking water advisory value (US-EPA, 2016a). The derivation of a drinking water standard for the Netherlands has been made in accordance with the WHO guideline of 1993 (WHO, 1993) whereby a maximum of 10% of the TDI of PFOS may consist of a contribution via drinking water (Moermond *et al.*, 2010; Wintersen *et al.*, 2016). In 2011, WHO increased its default value from 10% to 20% (WHO, 2011)(Cornelis & Geerts, 2016), and 20% is currently used as the standard.

Dermal absorption coefficient of PFOS: a measured value is not available and a calculated value is not very reliable because the calculation uses the log K_{ow}. Dermal absorption of PFOS at low doses is very low (Johnson, 1995a; b). As such, the dermal drinking water equivalent is set to 0.
The air-water partition coefficient is $<2.10^{-6}$ (3M Company 3M-Company (2003) in EFSA (2008a)), which is lower than the minimum value of 5.9×10^{-4}. Consequently, inhalatory exposure via drinking water should not be included in the derivation of the guideline value.
The SRV values for groundwater for the three sets of toxicological reference values are set out in Table 28. The reduction factor used is 20%.

Table 28: Reference values for groundwater

Toxicological reference value	Value	Unit	SRV groundwater
Set 1 (preference)US-EPA (2016a)			
TDI oral	2.10^{-5}	mg/kg/d	120 ng/l
TCA inhalation	7.10^{-5}	mg/m³	
TDI dermal	2.10^{-5}	mg/kg/d	
Set 2 Zeilmaker *et al.* (2018)			
TDI oral	$6.25.10^{-6}$	mg/kg/d	38 ng/l

TCA inhalation	$21.9.10^{-6}$	mg/m³	
TDI dermal	$6.25.10^{-6}$	mg/kg/d	
Set 3 EFSA (2018b)			
TDI oral	$1.8.10^{-6}$	mg/kg/d	11 ng/l
TCA inhalation	$6.3.10^{-6}$	mg/m³	
TDI dermal	$1.8.10^{-6}$	mg/kg/d	

The quality criterion for groundwater in Denmark is 0.1 µg/l; this is a sum parameter for 12 PFAS. In the German Länder of Bavaria and Baden Würtenberg, the guideline value for PFOS in groundwater is 0.23 µg/l (CONCAWE, 2016; UBA, 2016a).

2.10.2. SOIL

The calculations were made for 3 different sets of toxicological reference values, as described in the substance sheets at the end of this report. The first scenario makes use of the RfD in US-EPA (2016c) (= preferred scenario), the second scenario is based on the MTR of the Netherlands as described in Zeilmaker et al. (2018) and the third scenario is calculated with the proposed TDI-value of EFSA (2018c). First a BCF for grass of 0.26 (mg/kg plant dm)/(mg/kg soil dm) (average value Stahl *et al.* (2009)) and BTF values as shown in Table 22 were used, these values were subsequently revised as described in Table 21 and Table 23.
In the first instance, calculations were carried out using the UB food consumption and concentration data of EFSA (2018c). However, initial calculations gave negative background exposure for the landuse type agriculture through food consumption. This indicates that the exposure via locally grown foods in an agricultural setting exceeds the general background exposure via dietary intake of consumtion foods, possibly because the estimated intake via locally grown vegetables is overestimated by the available BCF. This may also be due to what was mentioned in section 2.4.4, that in EFSA (2018c) the exposure data were lower than in the earlier analysis from 2012, where the concentration data were higher for many foods. Also for the calculations based on the UB data of EFSA (2012) it appeared that for tox scenario 2 (Zeilmaker et al. (2018)) and 3 (EFSA (2018c)) the oral toxicological reference value (TDI oral = 6.25 and 1.8 ng/kg/d) for a number of landuse types is already fully filled in by the background intake via food 2.77-9.10 ng/kg/d). On the basis of previous observations, the final calculations for deriving the SRV were carried out on the basis of the lower bound (LB) intake figures from EFSA (2012). The calculations were made for the three different scenarios or three different sets of toxicological limit values.

→ Proposal calculated with LB data from EFSA (2012), BCF for grass of 0.26 (mg/kg plant dm)/(mg/kg soil dm) and BTF values as shown in Table 22

The calculations were made with LB intakes and concentrations from EFSA (2012), see Table 29. This is not a conservative assumption, but probably gives a much more realistic picture. The results (Table 29) suggest that the impact on landuse type II (most critical standard proposal) for tox scenario 1 is only limited, the proposed soil remediation value increases by about 15% to 2.4 µg/kg dm. Contrary to the UB approach, proposals can now also be calculated for scenarios tox 2 and tox 3.

Table 29: Proposed human health based soil remediation values for PFOS (µg/kg dry matter) with LB data from EFSA (2012).

	II	III	IV	V
S-Risk tox 1 US-EPA (2016c)				
	2.399 (threshold)	**204.6** (threshold)	IVa **14,030** (threshold)	Va **36,080** (threshold)
			IVb **15,520** (threshold)	Vb **27,770** (threshold)
Adjustment	-	-	IVb **<u>1,949</u>** (drinking water)	Va and Vb **<u>1,949</u>** (drinking water)
S-Risk tox 2 Zeilmaker et al. (2018)	**0.6349** (threshold)	**55.05** (threshold)	IVa **3,768** (threshold)	Va **10,240** (threshold)
			IVb **4,168** (threshold)	Vb **7,885** (threshold)
Adjustment	-	-	IVb **<u>1,949</u>** (drinking water)	Va and Vb **<u>1,949</u>** (drinking water)
S-Risk tox 3 EFSA (2018c)	**0,0833** (threshold)	**6.628** (threshold)	IVa **447.2** (threshold)	Va **1,880** (threshold)
			IVb **494.6** (threshold)	Vb **1,448** (threshold)

bold: values proposed as soil remediation values based on tox values
<u>bold underlined</u>: values proposed as soil remediation values based on adjustment based on binding legal reference values
-: the concentration indices are not critical, no adjustment is needed
Threshold: the non-carcinogenic endpoint is the most critical, the proposed soil remediation value corresponds to the value at which there is no longer a risk for children 1 - 6 years (with the exception of industry where no children are present)

Expertisecentrum PFAS (2018) indicates for soil an upper limit (intervention value level) of 6,600 µg/kg dm (not protective for groundwater when used as drinking water, in this case a safe upper limit is 100 µg/kg dm) and lower limit of 0.1 µg/kg dm, for residence with garden 11 µg/kg dm (leaching from soil to drinking water) and residence with vegetable garden was not determined.

Lijzen *et al.* (2011) indicate a reporting limit of 0,1 µg/kg dm in soil. On the basis of a literature study, the authors indicate 6.5 µg/kg dm as the highest reported value for a sample originating from an area in Germany with increased exposure to PFOS. Other reported concentrations in this area were

mostly below the limit of detection of 3 µg/kg. The authors assert that in a relatively unaffected area, the concentration of PFOS will be below the current limit of detection.
The expertise centre PFAS (Pancras & van Bentum, 2018) also collected data on the presence of PFOS and PFOA in the topsoil (up to about 0.5 m minus ground level) in the Netherlands. By using data from Southern Holland, Utrecht and Northern Brabant for PFOS, the aim was to rule out the influence of potential risk locations or known PFOS sources. The report states the percentile values for PFOS as shown in Table 30.

Table 30: Calculated percentile values for diffuse load of PFOS in the topsoil (µg/kg dm) (Pancras & van Bentum, 2018).

Percentiles	**PFOS**
25 percentile	0.25
Median; 50 percentile	0.46
75 percentile	0.93
90 percentile	1.6
95 percentile	1.9

The proposals for soil remediation values set out in tox1 are all higher than the reporting limit of 0.1 µg/kg dm indicated by Lijzen *et al.* (2011). The value for agriculture is just above the P95 background value of 1.9 µg/kg dm as determined for the Netherlands (Pancras & van Bentum, 2018), the background value for Flanders will presumably be similar.
For tox2 the proposals are also above the reporting limit, but the calculated value for agriculture is between the P50 (0.46 µg/kg dm) and P75 (0.93 µg/kg dm) background value for the Netherlands (Pancras & van Bentum, 2018). For the most strict tox scenario 3 based on EFSA (2018c) the same conclusion can be drawn, here, the calculated value for agriculture is below the P25 percentile value (0.25 µg/kg dm) background value for the Netherlands for diffuse loading of PFOS in topsoil.

→ Proposal calculated with LB data from EFSA (2012), BCF for grass of 0.048 (mg/kg plant dm)/(mg/kg soil dm) and BTF values as shown in Table 23

In a subsequent scenario, calculations were made for the LB EFSA (2012) intake data and concentrations in combination with a BCF for grass adjusted to 0.048 (mg/kg plant dm)/(mg/kg soil dm) and BTF values as shown in Table 23. These calculations only have an impact on the landuse type agriculture, as only animal products from own cultivation are consumed here. Compared to previous calculations, this results in an increase of the proposals for human health based soil remediation values of about 30% for the agricultural scenario.

Table 31: Proposed human health based soil remediation values for PFOS (µg/kg dry matter) with LB data from EFSA (2012).

	II	III	IV	V
S-Risk tox 1 US-EPA (2016c)				
	3.109 (threshold)	**204.6** (threshold)	IVa **14,030** (threshold)	Va **36,080** (threshold)
			IVb **15,520** (threshold)	Vb **27,770** (threshold)
Adjustment	-	-	IVb **<u>1,949</u>** (drinking water)	Va and Vb **<u>1,949</u>** (drinking water)
S-Risk tox 2 Zeilmaker et al. (2018)	**0,8439** (threshold)	**55.05** (threshold)	IVa **3,768** (threshold)	Va **10,240** (threshold)
			IVb **4,168** (threshold)	Vb **7,885** (threshold)
Adjustment	-	-	IVb **<u>1,949</u>** (drinking water)	Va and Vb **<u>1,949</u>**
S-Risk tox 3 EFSA (2018c)	**0,1107** (threshold)	**6.628** (threshold)	IVa **447.2** (threshold)	Va **1,880** (threshold)
			IVb **494.6** (threshold)	Vb **1,448** (threshold)

bold: values for the proposed as soil remediation values based on tox values
<u>bold underlined</u>: values proposed as soil remediation values based on adjustment adjustment based on binding legal reference values
-: the concentration indices are not critical, no adjustment is needed
Threshold: the non-carcinogenic endpoint is the most critical, the proposed soil remediation value corresponds to the value at which there is no longer a risk for children 1 - 6 years (with the exception of industry where no children are present)

The proposed human health based soil remediation values for PFOS (µg/kg dry matter) with LB data from EFSA (2012) are shown in Figure 8. Only tox scenario 1 results in a proposal for landuse type II (agriculture) above the 95th percentile value of 1.9 µg/kg dm for diffuse load of PFOS in the topsoil in the Netherlands (Pancras & van Bentum, 2018). The other landuse types are all for the 3 tox scenarios above the 95th percentile value.

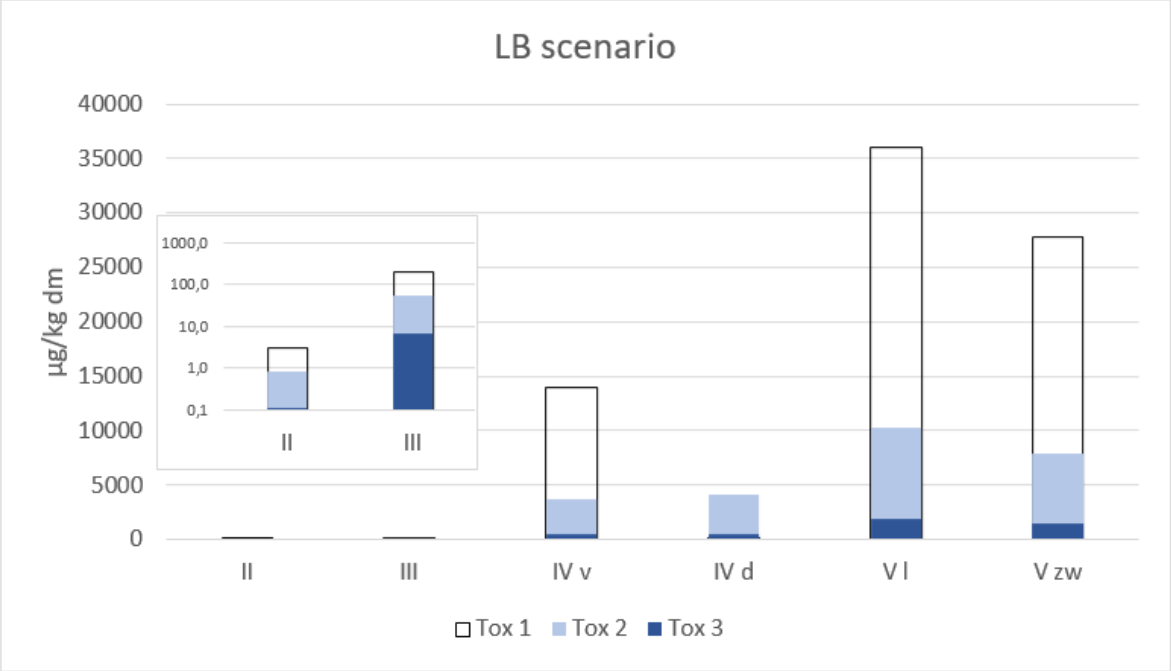


Figure 8: Comparison of the SRV (µg/kg dm) calculated on the basis of three sets of toxicity reference values dmusing LB intake and concentrations in food (EFSA 2012) for the different landuse types.

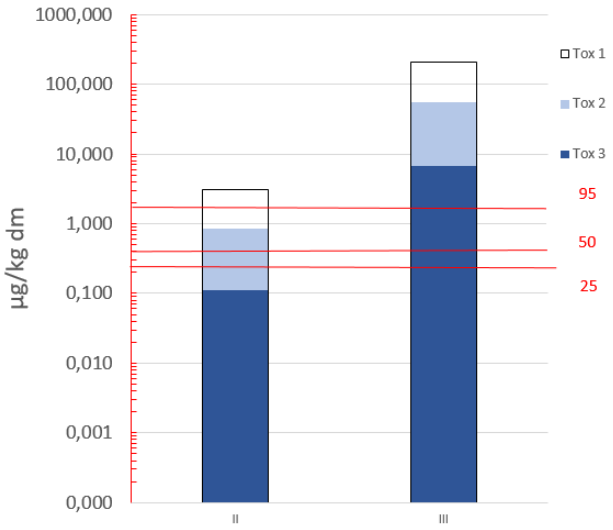


Figure 9: Comparison of the SRV on the basis of 3 tox scenarios with respect to the median, 95th percentile and 25th percentile value of the diffuse loading of PFOS in topsoils in the Netherlands (Pancras, 2018). dm

In Figure 9 the calculated SRV for the landuse types II and III are compared for each of the three sets of tox values, with the median, the 95th and 25th percentile values of the concentrations of PFOS in topsoils of Southern Holland. The SRV for agricultural soils derived for tox2 and tox3 are below the 95 percentile value of background PFOS from Pancras & van Bentum (2018). Current scientific knowledge and available calculation methodologies do not allow the derivation of feasible SRV for

agricultural areas in combination with toxicity data according to Zeilmaker et al (2018) and EFSA (2018c).
Based on the three toxicity scenarios, for residence with vegetable gardens, a SRV with a sufficiently large margin above the 95 percentile value of the PFOS background values in the Netherlands can be derived.

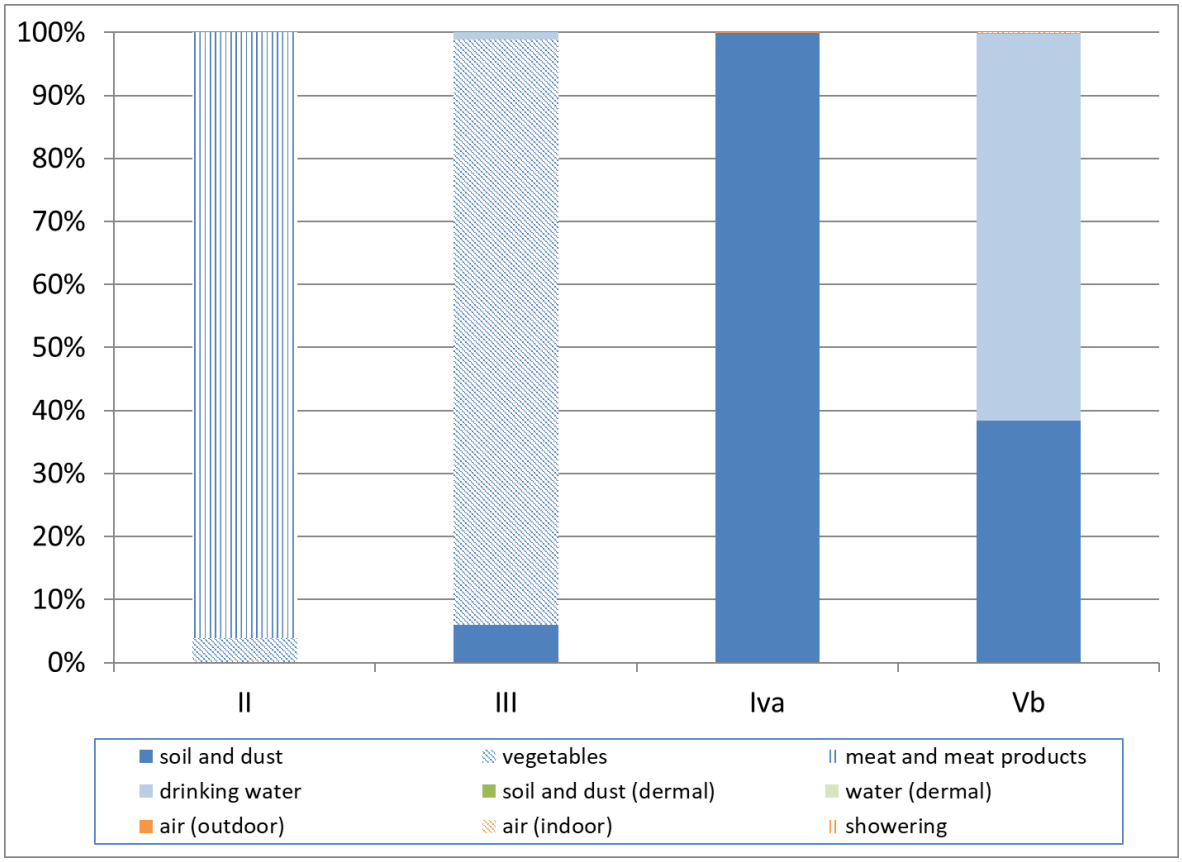


Figure 10: Contribution of exposure pathways to overall risk for PFOS in tox scenario 1 (US-EPA (2016c)) and with background dietary exposure based on the LB approach of EFSA (2012) (blue: oral, green: dermal, orange: inhalation). The contribution has been calculated for a soil concentration equal to proposed soil remediation values and - with the exception of landuse type Vb - for children (1-6 years).

The contribution of the exposure pathways to the risk for the different landuse types is shown in Figure 10. This figure has been calculated on the basis of the toxicologically based soil remediation value as calculated for each landuse type, and as such it does not take into account adjustments . We see that for all landuse types the oral exposure route (blue) is dominant. In type II, the determining factor is consumption of local meat and animal products (96%) with only a limited contribution from vegetables. Local meat and animal products consist of 91% milk for children, 49% for adults, followed by beef (31%) and butter (19%). Offal contributes less than 3% for adults. The concentration calculated in milk at a concentration in soil equal to the proposed soil remediation value for tox1 (3.11 µg/kg dm) is 0.62 µg/kg fw, more or less a factor of 5 higher than the UB concentration as determined by EFSA (2012) (0.12 µg/kg fw). If we repeat the calculations at a soil concentration of 0.46 µg/kg dm (median value for diffuse loading of PFOS in the topsoil in the Netherlands. (Pancras & van Bentum, 2018)), this results in a milk concentration of 0.09 µg/kg fw – higher than the average lower bound value of 9.10^{-4} µg/kg fw reported by EFSA (2012), but slightly

lower than the upper bound value reported by EFSA (2012), 0.12 µg/kg fw. The concentration of milk is presumably still overestimated. By way of illustration, when this last calculation is repeated with the lowest BTF value found in the literature (0.018 $mg.kg^{-1}$ fw/$mg.d^{-1}$ Kowalczyk et al 2013), this results in a milk concentration of 0.08 µg/kg fw which is still higher than the LB value of EFSA (2012). For landuse type III, the consumption of vegetables weighs the most (92%), with a limited contribution from ingestion of soil and dust. Type IVa is dominated by soil and dust ingestion and type Vb is determined by soil and dust ingestion (38%) and drinking water (61%).

→ Comparison with risk limits calculated in the Netherlands for the application of PFOS-containing soil and dredging for arable and livestock farming.

Wintersen *et al.* (2019) calculated risk limits for the application of PFOS-containing soil and sludge for arable and livestock farming in the Netherlands, using the tox values used in scenario 2 (tox2) of the current report. For PFOS this results in risk limits of 109 µg/kg for arable crops farming, 7.6 µg/kg for livestock farming and 92 µg/kg for residences with vegetable gardens. For residences with vegetable garden this value is about 1.7 times higher than the value calculated in the current report with the EFSA 2012 LB data (55 µg/kg dm). The Dutch risk limits for arable and livestock farming are more than 100 times higher than the propsed soil remediation value in this report (0.844 µg/kg) based on the tox2 scenario.

Table 32 provides an overview of a number of parameters that differ in the calculation method used by Wintersen et al. (2019) and this report. For the arable and livestock farming scenario, Wintersen et al. take into account the background exposure via food, by subtracting the background exposure from the MTR[18] human ($6.25.\ 10^{-3}$ µg/kg bw/ day)[19]:

- in the case of livestock farming: baseline background exposure (fish, eggs, drinking water, etc.) + background exposure to arable products, $0.159.10^{-3}$ µg/kg bw/day;
- in the case of arable farming: baseline background exposure (fish, eggs, drinking water, etc.) + background exposure to livestock products, $0.318.10^{-3}$ µg/kg bw/day.

For the derivation of soil remediation values with S-Risk, an age-related background exposure through food is taken into account, ranging from 1.2 ng/kg bw/day (1-3 years) to 0.875 ng/kg bw/day (31 years and older). The S-Risk model subtracts from this background exposure via food - depending on the landuse type (described in annex IV of the S-Risk Technical Guidance) - a part that is replaced by food from own cultivation. For the most sensitive group (children), the background exposure applied in the Netherlands is a factor of 10 lower than that applied in Flanders; moreover, it is 20-30 times lower than the MTRhuman. The transfer to milk is based on Vestergren (2013), and is the same for Flanders and the Netherlands. In the Netherlands, an intake of 330 g of milk and dairy products per day is assumed, in Flanders it ranges from 395 g/d (1-3 years) to 181 g/d (34-41 years). Since milk is the most important exposure route in the agricultural scenario for Flanders (see above), and both transfer to milk and consumption quantities differ only slightly between the Netherlands and Flanders, the reason for the difference in proposed soil remediation values / risk limits for agriculture and livestock farming may lie more in the background exposure via food that is higher for Flanders.

For vegetable gardens, the background exposure is not used in the Netherlands for the risk limit calculations, but it is in Flanders. The order of magnitude of the calculated risk limits for the Netherlands is 1.7 times higher than the soil remediation values proposed for Flanders, probably mainly due to the different approaches in the inclusion or exclusion of background exposure via food.

[18] Maximum tolerable risk

[19] Personal communication Frank Swartjes, October 2019

Table 32: Differences in approach for calculating risk limits for PFOS in the Netherlands and soil remediation values (for tox scenario 2) in Flanders.

Parameter	Wintersen *et al.* (2019)	Current study
Calculated soil remediation value / risk limit	Arable farming: 109 µg/kg dm Livestock farming: 7.6 µg/kg dm Vegetable gardens: 92 µg/kg dm	Agriculture: 0.844 µg/kg Residence with vegetable garden: 55 µg/kg dm
Toxicological reference values (oral)	$6.25.10^{-3}$ µg/kg.d	
Background exposure food	Taken into account and deducted from the health-related limit value Arable farming: 0.318 ng/kg bw/day Livestock farming: 0.159 ng/kg bw/day Vegetable gardens: none Based on data from Noorlander *et al.* (2010); Noorlander *et al.* (2011)	Taken into account for all landuse types. For agriculture and vegetable gardens a part is being replaced by own cultivation. EFSA 2012 LB: 1.2 ng/kg bw/day (1-3 years) to 0.875 ng/kg bw/day (31 years and above)
Consumption of vegetables from vegetable garden	100% vegetable garden vegetables from own cultivation (standard data CSOIL)	Fraction of age-related vegetable consumption (tables 11 and 13 annex IV of the S-Risk Technical guidance)[20]

2.10.3. ECOTOXICOLOGICAL REFERENCE VALUES

Based on the guidelines for drafting soil remediation values (Cornelis and Touchant, 2016), the database data of the following 4 international bodies were consulted: US-EPA, ECHA, CCME and RIVM[21]. For PFOS, ecotoxicological limit values were only derived by RIVM (2019). These are summarised in Table 33. In contrast to Flanders, in the Netherlands, biomagnification (accumulation to higher trophic levels) is taken into account for the determination of the ecotoxicological limit value and therefore a distinction is made in the table between direct ecotoxicity through soil contact and biomagnification.

[20] https://www.s-risk.be/sites/s-risk.be/files/SRisk%20model%20equations%20-%20Annex%20IV.pdf consulted September 2019

[21] US EPA: http://www.epa.gov/ecotox/ecossl/index.html
CCME Canada: http://www.ccme.ca/en/resources/canadian_environmental_quality_guidelines/index.html
RIVM Netherlands: http://www.rivm.nl/rvs/Normen/Milieu/Bodeminterventiewaarden, http://www.rwsleefomgeving.nl/onderwerpen/bodem-ondergrond/bbk/instrumenten/nobo
ECHA database: http://echa.europa.eu/information-on-chemicals

Table 33: Overview of ecotoxicologically based reference values (direct soil contact and biomagnification in brackets) derived for PFOS by RIVM (explanation in the text) in µg/kg dm.

Reference value	Agriculture areas	Residence with vegetable garden	Industrial areas	Reference
Ecological Risk direct (biomagnification)	**16**[c] (3)	**380**[a] (18)	**9,100** [b] (110)	RIVM, 2019

*a) medium protection level (*Geometric mean of HC5 and HC50*); b) moderate protection level (SR_{eco} Serious-Risk soil corresponding to the HC_{50} level) ; c) high protection level (MTR_{eco} Maximum Permissible Risk soil corresponding to the HC_5 level).*

The direct ecological risks are tested against two risk limits in the Netherlands: the Serious Risk to the soil ecosystem. ($SR_{eco\ soil}$) is the concentration at which harmful effects of the substance on the soil ecosystem are likely to occur and corresponds to the HC_{50} protection level[22] and the Maximum Permissible Risk for the ecosystem (MTR_{eco}) which corresponds to the HC_5 protection level[23]. Below this level, no negative effects on the soil ecosystem are expected. Where the $SR_{eco\ soil}$ applies to industrial areas, the MTR_{eco} is applied to agricultural and nature reserve areas. For the soil function class 'Residence with vegetable garden', a middle level is defined as the geometric mean of both.

To derive the limit values for PFOS we use Verbruggen et al. (2020; in prep.) (Verbruggen & al., 2020). In this RIVM letter report the previously published data from Bodar et al. (2011) are supplemented with recent literature data on PFOS soil toxicity, biomagnification and bioaccumulation data. To derive the direct ecological risks $SR_{eco,direct}$ and $MTR_{eco,direct}$ the methodology described in Vlaardingen and Verbruggen (2007) was used (van Vlaardingen & Verbruggen, 2007). To derive the ecotoxicological risk limits for indirect toxicity, the renewed methodology as described by Verbruggen (2014) was applied, taking into account the energy requirement and food intake of worm-eating birds and mammals, as well as their predators (Verbruggen & al., 2014). The evaluation of the supplemented dataset for direct ecotoxicity of PFOS provides a $SR_{eco,direct}$ of 9.100 µg/kg dm and a $MTR_{eco,direct}$ of 16 µg/kg dm. The middle level for direct toxicity used in the Netherlands for deriving maximum values in soil management is 380 µg/kg dm. If biomagnification is taken into account, Verbruggen et al. (2020; in prep.) derive the following ecotoxicological risk limits: $SR_{eco,indirect}$ = 110 µg/kg dm, $MTR_{eco,indirect}$ = 3 µg/kg dm and a middle level = 18 µg/kg dm. These values are adopted as a preliminary proposal for ecotoxicological standards for PFOS in Flanders (Table 34). Due to the persistent nature of PFAS as a substance group as a whole, it is proposed that, exceptionally and in contrast to normal practice in Flanders, biomagnification should be taken into consideration for the PFOS proposals for the SRV_{eco} for the landuse types 'Agriculture' and 'Residence with vegetable garden' and 'Recreational areas'. **The proposal for the SRV_{eco} for these landuse types is then respectively 3, 18 and 110 µg/kg dm (in bold in Table 34).**

Table 34: Proposal for ecotoxicological values for PFOS in Flanders (µg/kg dm); values based on direct toxicity are shown in brackets.

Reference value	Agriculture areas (type II)	Residence with vegetable garden (type III)	Recreational areas (type IV)	Industrial areas (type V)
SRV_{eco}	**3** (16)	**18** (380)	**110** (9,100)	**9,100**

[22] The Hazardous Concentration for 50% of the soil organisms (HC_{50})
[23] The Hazardous Concentration for 5% of the soil organisms (HC_5)

2.10.4. TARGET VALUES

No target values for Flemish soils were available at the time this study was carried out. On behalf of OVAM, background values were measured in 2020, for which, for PFOS, a background value of 1.5 µg/kg dm in soil was derived, more information can be found in Touchant *et al.* (2020). The Netherlands applies a temporary background value of 0.9 µg/kg dm in soil. (Wintersen *et al.*, 2019)[24].

2.11. INTEGRATION AND EVALUATION

2.11.1. SOIL

The calculations for deriving the soil remediation values for PFOS were carried out in an adapted version of the S-Risk model version 1.3 (for the time being only available on an internal VIT0 test server) taking into account the amphiphilic character of PFOS, substances for which the log K_{ow} cannot be measured according to the OECD standard test guideline. In order to avoid the use of the log K_{ow}, the transfer to plants was initially calculated on the basis of BCF factors relative to the solid phase of the soil, in contrast to the usual method where BCF factors for organic substances in S-Risk are expressed on the basis of pore water concentrations. In addition, PFOS was considered as a non-dissociative in S-Risk during the derivation of the soil remediation values (PFOS is a dissociative substance) so that the sorption of soil particles (K_d) can be calculated directly from the organic carbon content in the soil and the K_{oc} without the intervention of K_{ow}.
Various scenarios with combinations of parameter values were calculated and tested for feasibility. The following parametric values were used to derive the proposed soil remediation values for PFOS:

- Toxicology:
  - The RfD of US-EPA (2016c) of **20 ng/kg bw/d** and the derived tolerable concentration in the air (TCA) of **70 ng/m³** (preferred scenario);
  - The RfD of Zeilmaker et al. (2016) of **6.25 ng/kg bw/d** and the derived tolerable concentration in the air (TCA) of **21.9 ng/m³**;
  - The RfD of EFSA (2018c) of **1.8 ng/kg bw/d** and the derived tolerable concentration in the air (TCA) of **6.3 ng/m³**;

- Background exposure: The lower bound intake and concentration data of EFSA (2012);
- Plant uptake: The BCF_{PFOS} values derived by Ghisi et al. (2019) after comparing the original data with the approach followed by Wintersen et al. (2019) applying a complete diet;
- Animal transfer: average BTF values derived from Vestergren et al. (2013) and Kowalczyk et al. (2013);

All parameter values used for the final human health based soil remediation values are summarised in the substance sheet at the back.

[24] https://www.rivm.nl/sites/default/files/2019-11/254%202019%20MV%20Bijlage%201%20Notitie%20TAW%20PFAS%20in%20grond.pdf

For the evaluation of ecotoxicological effects, no new primary sources and/or databases were consulted to derive possible new ecotoxicological values, but it was examined whether ecotoxicological values have recently been derived from other bodies. For PFOS the values derived by RIVM (2019) were used, whereby biomagnification was taken into account.

During this study, insufficient measurement data were available to derive reliable target values. On behalf of OVAM, background values were measured in 2020, for which, for PFOS, a background value of 1.5 µg/kg dm in soil was derived (Touchant *et al.*, 2020).

A comparison of the proposed human health based and ecotoxicological soil remediation values is given in Table 35 with in green the preferred value based on the preferred toxicology scenario. At present there are no soil remediation values for PFOS in the Flemish legislation on soil (VLAREBO) meaning that a comparison is not possible. If the US EPA scenario is used for the human health based SRV, the values in green are used as reference value (provisional SRV).
A first comparison with Dutch background values (Pancras, 2018) shows that, with current scientific knowledge and available calculation methods, no feasible SRV for agricultural areas can be derived (landuse type II). The decision concerning the SRV for landuse type II (agriculture) (and thus also for landuse type I - nature) is awaiting the study 'Derivation of target values for perfluorinated compounds' commissioned by OVAM. The soil remediation values for landuse type I (nature) and landuse type II (agriculture) can be adjusted on the basis of the target values and the values for free use of soil.

Table 35: The proposed SRV for soil (µg/kg dm) for PFOS with the preferred value in green. The soil remediation values for landuse type II may be further adjusted to a feasible value on the basis of the target values and the values for free use of soil.

	II[25]	III	IV	V
Flemish legislation on soil (VLAREBO)	-	-	-	-
Proposal human health based tox US-EPA (2016c)	3.1	204.6	1,949 (drinking water) (IVb)	1,949 (drinking water (Va and b)
Proposal human health based tox Zeilmaker et al. (2018)	0.84	55.05	1,949 (drinking water) (IVb)	1,949 (drinking water (Va and b)
Proposal human health based tox EFSA (2018c)	0.11	6.63	447.2 (IVa)	1,488 (Vb)
Proposal ecotox	3	18	110	9,100
Background value	1.5			

2.11.2. GROUNDWATER

The soil remediation value for groundwater has a human health based underpinning, and corresponds to the drinking water standard if this has a toxicological basis (Cornelis & Touchant, 2016). The drinking water standard of 100 ng/l proposed by the EU is a general limit (not specific for PFOS) which is mainly based on feasibility and not only on toxicology.
As such, the soil remediation value for groundwater was also calculated, with the standard formula (paragraph 2.10.1) for the three toxicological reference values with which the soil remediation value was calculated. The corresponding calculated values for groundwater are:

- 120 ng/l, based on the RfC of US-EPA (2016)
- 38 ng/l, based on the maximum tolerable human health based risk level (Zeilmaker *et al.,*2018)
- 11 ng/l, based on the TDI proposal of EFSA (2018).

120 ng/l, based on the RfC of US-EPA (2016) is the preferred value based on the preferred toxicology scenario. This is also the most closely related to the groundwater criterion put forward at EU level, i.e. 100 ng/l.

2.11.3. GUIDELINE VALUES

Guideline values are not yet available at the time of publication of this report and will be published in a separate document.

[25] Not final, will be adjusted on the basis of the target values and the values for free use of soil

2.12. COMPARISON WITH FOREIGN SOIL REMEDIATION VALUES

In 2016, RIVM derived generic risk limits for non-agricultural soil functions that allow local authorities to develop a site-specific approach to PFOS contamination (Wintersen *et al.*, 2016). The derived generic intervention and target values for soil and groundwater are set out in Table 36. The values are derived according to the applicable method, but they are not national soil remediation values as such. The lower limit for soil (0.1 µg/kg dm) is the reporting limit, and is based on background concentrations in relatively unaffected areas. The upper limit (6600 µg/kg dm) is the lowest value of the human maximum tolerable risk (MTR) and the Serious Risk level (SR) for the environment. The lower limit for groundwater is the generic target value, derived from the Negligible Risk to the environment (NR_{eco}); the upper limit for groundwater is the lower of the following values: MTR_{human}, $_{groundwater}$, MTR_{DW} (safe value for drinking water for consumption) and $SR_{eco,\ groundwater}$ (Wintersen *et al.*, 2016). The generic value for residence with garden (11 µg/kg dm) takes into account the leaching of PFOS to groundwater used as drinking water. The generic value for 'other green spaces, buildings, infrastructure and industry' (8 µg/kg dm) is a $SR_{eco,\ BM}$ that takes into account biomagnification (BM); this calculation is based on the assumption that areas with this function are large enough to serve as habitats for birds and mammals, whereby biomagnification to higher organisms can play a role. This is not assumed in the case of 'residence with a garden (Wintersen *et al.*, 2016).

In 2019, RIVM derived the following national risk limits for PFOS for a temporary framework for the application of soil and dredge spoil on or in soils: 3 µg/kg dm for agriculture, 18 µg/kg dm for residential and 110 µg/kg dm for industry (RIVM, 2019). These are not real soil remediation values, but values that are used within the PFAS temporary action framework. For these three soil function classes it appears that biomagnification (ecology) determines the lowest risk limit value; this is because PFOS is mobile and accumulates in higher organisms (RIVM, 2019). The risk limit values based on the human health based maximum tolerable risk level (6.25 ng/kg bw/d) of Zeilmaker (2018) are higher (Table 36).
The proposed ecological SRV for agriculture (3 µg/kg dm) (Table 35) is the same as the Dutch risk limit value for agriculture and comparable to the human health based SRV for tox scenario 1 (3.1 µg/kg dm). The human health based SRV proposals for agriculture for tox scenario 2 and 3 (0.84 and 0.11 µg/kg dm) (Table 35) are respectively 3.5 and 27 times lower than the Dutch risk limit for agriculture (3 µg/kg dm).
The proposed ecological SRV for residences (18 µg/kg dm) is the same as the Dutch risk limit value for residences. The human health based SRV proposals for residences for tox scenario 1, 2 and 3 (204.6, 55.05 and 6.63 µg/kg dm) (Table 35) are respectively 11 and 3 times higher, and 3 times lower than the Dutch risk limit for residences (18 µg/kg dm).
The proposed ecological SRV for recreation (110 µg/kg dm) is the same as the Dutch risk limit value for industry. The human health based SRV proposals for recreation for tox scenario 1, 2 and 3 (1949, 1949 and 447.2 µg/kg dm) (Table 35) are respectively 17, 17 and 4 times higher than the Dutch risk limit for industry (110 µg/kg dm).
The proposed human health based SRV for industry (1949 µg/kg dm, adjusted for drinking water) is 10 times lower than the Dutch human health based risk limit (18800 µg/kg dm), 18 times higher than the Dutch risk limit taking into account biomagnification (110 µg/kg dm) and 5 times lower than the Dutch ecological risk limit taking into account direct toxicity only (9100 µg/kg dm).
The Netherlands based the choice of an ecotoxicologically-underpinned intervention value over a health-related underpinned intervention value on an evaluation with the reference value of Zeilmaker (2016) and not with the proposed TDI from EFSA (2018). It is therefore only useful to compare the human health based SRV for scenario 2 with the Dutch human health based risk limit. The human health based SRV is lower than the Dutch risk limit $_{human}$ for all landuse types (with recreation and industry adjusted for drinking water).

The proposal for a guideline value in Norway is 0.1 µg/kg dm; this value is based on a study with earthworms (Stubberud, 2006) and also applies as a sum parameter for PFAS. Denmark has a quality criterion of 390 µg/kg soil as sum parameter (DEPA, 2015), but this value is based on the old RfD of US-EPA (0.03 µg/kg/d) and therefore in fact obsolete.
The human health based screening value is 1260 µg/kg dm in the USA and 2100 µg/kg dm in Canada; the screening value which takes groundwater protection into account is much lower (0.378 µg/kg dm in the USA). Australia has derived the following health-based screening values as a sum parameter for PFOS and PFHxS: 9 µg/kg dm for residence with garden, 2000 µg/kg dm for residence with minimum risk of soil contact, 1000 µg/kg dm for public areas and 20000 µg/kg dm for industry and trade.

Table 36 Foreign reference values for soil and groundwater[a].

Soil				
Agriculture/nature	Risk limit value$_{eco}$ for the temporary framework for action	3.0 µg/kg dm	The Netherlands	RIVM (2019) and update (RIVM-actualisatie, 2019)
Residence	Risk limit value$_{eco}$ for the temporary framework for action	18 µg/kg dm		
Industry	Risk limit value$_{eco}$ for the temporary framework for action	110 µg/kg dm		
Agriculture/nature	Risk limit value$_{human}$	19000 µg/kg dm		
Residence with vegetable garden	Risk limit value$_{human}$	92 µg/kg dm		
Residence with garden	Risk limit value$_{human}$	1200 µg/kg dm		
Industry/recreation[d]	Risk limit value$_{human}$	18800 µg/kg dm		
Upper limit (intervention value)	MTR$_{human-soil}$	6600 µg/kg dm	The Netherlands	Wintersen *et al.* (2016)
Lower limit (target value)	Reporting limit	0.1 µg/kg dm		
Residence with garden	MTR$_{residence-garden}$	11 µg/kg dm		
Residence with garden	Health based screening value (PFOS+PFHxS) (HHSV)	9 µg/kg dm	Australia	Australië (2018)
Residence with minimal risk of soil contact	Health based screening value (PFOS+PFHxS) (HHSV)	2000 µg/kg dm		
Public area	Health based screening value (PFOS+PFHxS) (HHSV)	1000 µg/kg dm		

Other green spaces, buildings, infrastructure, industry	$SR_{eco,biomagnification}$	8[c] µg/kg dm	Location-specific Netherlands	Wintersen *et al.* (2016)
Industry - commerce	Health based screening value (PFOS+PFHxS) (HHSV)	20000 µg/kg dm	Australia (2018)	Australië (2018)
Quality criterion	QC_{soil}	390 µg/kg dm	Denmark	DEPA (2015)
Proposal of guideline value		0.1 µg/kg dm	Norway	DEPA (2015)
Protection of groundwater	RSL (Regional Screening Level)	0.378 µg/kg dm	United States of America (2017)	Australië (2018)
Screening value human	RSL	1260 µg/kg dm		
Screening value human	SSV (Soil Screening Value)	2100 µg/kg dm	Canada (2017)	
Groundwater				
Upper limit (intervention value)	MTR_{DW}	4.7 µg/l	The Netherlands	Wintersen 2016
Lower limit (target value)	NR_{eco} = 1/10 MTR_{eco}	$0.23.10^{-3}$ µg/l		
Human risk limit residence with garden (Csoil)		310 µg/l	The Netherlands	Alphenaar *et al.* (2018)
Human risk limit residence with vegetable garden (Csoil)		Not determined	The Netherlands	

[a] The values for the Netherlands are generic risk limits, not national soil remediation values; [b] Application of correction to standard soil is recommended; [c] RIVM has concluded that the data on which this value has been determined may not be complete. A new inventory of the available data is necessary to determine whether this value of 8 µg/kg is correct (Alphenaar et al., 2018). [d] No crop consumption, limited soil contact

CHAPTER 3. PFOA

3.1. IDENTIFICATION

PFOA	
name English	Perfluorooctanoic acid
name Dutch	Perfluoroctaanzuur
CAS number:	335-67-1
EINECS number:	206-397-9
EC index number:	607-704-00-2
formula: C8HF15O2	C8HF15O2
molecular weight:	414.07 g/mole
conversion:	1 ppm=17.21 mg/m³; 1 mg/m³=0.06 ppm (calculated based on molecular weight) (ATSDR draft (2018))

3.2. SOURCES OF PFOA

PFOA does not occur naturally in the environment. Perfluorinated substances with a long C chain are (or have been) used in various applications, such as textiles, making carpets and leather water- and dirt-resistant, surfactants, extinguishing foams and grease-freeing paper (EFSA, 2008c). Specific applications of PFOA are as an adjuvant in Teflon (non-stick coating) for frying pans, and in herbicides and insecticides. Precursors (e.g. fluorinated telomeres) can be a source of PFOA, as well as side-chain fluorinated polymers. (CONCAWE, 2016).

3.3. BEHAVIOUR IN SOIL AND PHYSIOCHEMICAL PROPERTIES

3.3.1. FATE OF PFOA IN SOIL

The information on behaviour in soil is primarily derived from CONCAWE (2016). PFOA belongs to the group of perfluoroalkyl carboxyl acids (PFCA) within the large group of perfluoroalkyl substances (PFAS); from the viewpoint of behaviour and distribution, PFCA form a homogeneous group. The properties of the group therefore also apply to PFOA, although for certain properties there may be quantitative trends determined by chain length.

→ **Chemical form**

Under typical environmental conditions, most PFAS and their salts occur as solids. The relevant form of PFCA for the environment (soil, groundwater and surface water with a normal pH of 5-9) is the anion. The formation of anions is accompanied by a decrease in adsorption to soil and sediment as

they are usually net negatively charged. The speed of transport through soil or sediment decreases with a longer perfluorinated C chain and with an increasing content of organic carbon (OC) in the soil. PFSA (such as PFOA) bind more strongly than PFCA with the same number of C-atoms. PFAS (with the exception of telomere alcohols which have a hydroxyl function) are surfactants with a hydrophobic perfluorinated C chain and a hydrophilic functional group (e.g. sulphate or carboxyl). Unlike ordinary surfactants, the hydrophobic perfluorinated C chain of PFAS also has hydrophilic properties, making PFAS coatings resistant not only to water but also to oil and grease. The surface activity of PFAS is stronger than that of similar, ordinary surfactants. On the one hand, PFAS can settle at the interface of different phases, for example groundwater (hydrophilic) and soil air (hydrophobic), and on the other hand micelles can form in solution.

→ Distribution

PFCA are widely distributed in the environment due to their high solubility in water, low to moderate sorption to soil and sediment, and resistance to biological and chemical degradation. PFAS have a low vapour pressure, meaning that transport in the vapour phase only plays a minimal role. The Henry coefficients of PFAS are highly varied. The Henry coefficient of PFOA is comparable to that of benzene and xylenes. Evaporation from water is therefore not considered to be a significant process. The degree of transport of PFAS via water is influenced by the degree of adsorption to sediment or soil during that transport; the higher the sorption, the more the transport of PFAS via the aqueous phase is retarded. There are two sorption mechanisms that control the degree of adsorption:

- hydrophobic sorption of solid organic particles, and
- sorption on the surface of charged mineral surfaces.

The parameters that measure the sorption of solid organic C particles are the organic carbon partition coefficient (K_{oc}) and the solid/liquid partition coefficient (Kd). The octanol-water partition coefficient (K_{ow}) is not a suitable parameter for adsorption because it is difficult to measure due to the cationic and anionic charge of PFAS (PFAS do not have normal lipophilic behaviour). These PFCA that are strong acids occur almost exclusively as anions; they can adsorb to the charged mineral surfaces present in the soil or sediment, thus influencing the transport of PFAS through water. To demonstrate this possible mechanism, several experiments have been described in the literature, but the degree of adsorption or the impact on transport has not yet been quantified.

→ Transformation

The C-F covalent bond is one of the strongest bonds in organic chemistry. PFAS therefore have a high thermal, chemical, photolytic and biological stability. There is no indication that PFCA would undergo biotransformation or photolysis under normal environmental conditions. Under aerobic conditions with activated sludge, no removal or biotransformation was measured for PFOA. Some removal of PFOA has been measured under anaerobic conditions, but without formation of metabolites or increase in fluoride. There are no tests demonstrating significant or complete degradation of PFOA under environmental conditions. Due to the strong C-F bond, PFOA is persistent in the environment. Under natural conditions, precursors (alcohol telomeres) can convert to PFCA. For example, biotransformation of the 8:2 telomere alcohol (8:2 FTOH[26]) to PFOA would proceed via an oxidation of the alcohol to an acid, and then a complete biodegradation (oxidation) of the non-fluorinated part.

[26] Consists of 8 fully fluorinated C atoms, an ethyl group and an alcohol function

3.3.2. PHYSIOCHEMICAL PROPERTIES

PFOA is a powder with a melting point, depending on the source, ranging from 37 to 60°C and a density of 1.8 g/cm³. The physicochemical properties of PFOA are listed in Table 37. Some parameters have been estimated using the EpiSuite modelling platform of the US EPA for comparison with measured values (if available); it should be noted that the estimated values may not accurately reflect actual properties as there are no perfluor structures in the EpiSuite training set (Lijzen *et al.*, 2018).

→ Water solubility (S)

PFOA has good solubility in water. The measured solubility ranges from 2.3 to 9.5 g/l. Due to its good solubility in water, PFOA can distribute fairly easily through the water into the environment. PFOS and PFOA have the same number of C-atoms, but PFOS has more C-F units than PFOA which has a carboxylic group; consequently, PFOA is less hydrophobic than PFOS (Milinovic *et al.*, 2015).
The anion also has a high solubility (Prevedouros *et al.*, 2006).
A solubility of $9.5.10^3$ mg/l (25°C) is assumed for the calculations of the soil remediation values. This is the value stated in the REACH Annex XV Restriction report (ECHA, 2014), and which is also used in the derivation of risk limits by the Netherlands (Lijzen *et al.*, 2018; Wintersen *et al.*, 2019) and standards by Australia and New-Zealand (FSANZ, 2016). Wintersen *et al.* (2019) states $7.09.10^3$ mg/l, this is $9.5.10^3$ mg/l (25°C) converted to 10°C for use in CSOIL.

→ Acid dissociation constant

PFOA dissociates and is available as an anion in environmentally relevant pH (3-8) (EFSA (2008b); ATSDR draft (2018)). PFOA has a pKa of -0.2 to 3.8. This means that in a neutral soil, PFOA is mainly present as an anion. At pH 7, only 3 to 6 in 10^5 molecules are PFOA, the rest is the anion; at pH 1, approximately 6% is PFOA (Prevedouros *et al.*, 2006). For the calculations of the soil remediation values a pKa of 2.8 is assumed, given that several authorities use this value in their recent evaluation documents (US-EPA, 2016d; Lijzen *et al.*, 2018).

→ Vapour pressure (Vp)

The measured vapour pressure of PFOA ranges from 4 to 1300 Pa (CONCAWE, 2016). In its most recent evaluation of PFOA, US-EPA indicates an experimental vapour pressure of 70 Pa at 25°C (US-EPA, 2016d). ECHA mentions a vapour pressure of only 4.2 Pa at 25°C, this value is calculated by extrapolation from a vapour pressure at 59.3°C. (ECHA, 2014). The volatilisation of PFOA is negligible under environmentally relevant conditions (pH > 2.5) (Johansson *et al.*, 2017). The anion presumably has a negligible vapour pressure (Prevedouros *et al.*, 2006). The measured vapour pressure of the ammonium salt (0.0081 Pa) is indeed lower than that of the acid.
For the calculations of the soil remediation values, a vapour pressure of 70 Pa (25°C) is assumed, as it is a measured value, at a realistic ambient temperature which, moreover, was selected by the Netherlands and US-EPA in their recent evaluations of PFOA. In S-Risk the vapour pressure is converted to a vapour pressure at 10°C. Furthermore, especially for PFOA, in S-Risk the vapour pressure is corrected for pKa (Vp_z), using the following formula used by Lijzen *et al.* (2018) in CSOIL: $Vp_z = VP_{10°C}/10^{(pH-pKa)}$. Lijzen 2018 obtains a vapour pressure of $1.7x10^{-2}$ Pa.

→ **Henry coefficient (H)**

The Henry coefficient is the ratio between vapour pressure and solubility. The Henry coefficient of PFOA ranges from 0.04 to 0.09 Pa m^3/mol (CONCAWE, 2016). The Henry coefficient of the protonated form (at very low pH) is 0.362 Pa m^3/mol). PFOA dissociates in water and the anions have a strong tendency towards the aqueous phase, which makes the substances less volatile than calculated on the basis of their physical and chemical properties (Pancras *et al.*, 2018). For the calculations of the soil remediation values, the Henry coefficient is calculated by S-Risk, based on the solubility and the corrected vapour pressure (Vp_z).

→ **Octanol water partition coefficient (log K_{ow})**

The log K_{ow} cannot be measured because the substance forms several layers in a mixture of octanol and water. FSANZ (2016) indicates an estimated value of 6.3. The log K_{ow} calculated with the COSMOtherm model is one order of magnitude lower and amounts to 5.3 (Wang *et al.*, 2011). The log K_{ow} estimated with EpiSuite is even lower and is 4.81. The log K_{ow} of the ammonium salt is much lower (0.7).
In S-Risk the log K_{ow} is used to calculate Kp, K_{oc}, and transfer factors, unless an experimental value is entered. Experimental values are available for these three parameters. S-Risk will therefore not have to deal with log K_{ow}. S-Risk does however require the input of a value for log K_{ow}. The calculated value 4,81 of EpiSuite is entered in S-Risk, because the algorithm of this model is transparent while the algorithm of the commercial COSMOtherm is not freely available. S-Risk does not use this log K_{ow} anywhere in its calculations for PFOA.

→ **Organic carbon-water partition coefficient (log K_{oc})**

Monitoring studies suggest that PFOA is mobile in soil and can seep into groundwater (Prevedouros et al., 2006)(Davis *et al.*, 2007).
For organic substances the log K_{oc} can be calculated from the log K_{ow}. However, for surfactants the log K_{ow} is not a good indicator for sorption to the soil because the log K_{ow} cannot be measured (accurately).
PFOA slightly adsorbs to the soil with log K_{oc} values of 1.15 to 2.96 (K_{oc} 14 to 912) (Prevedouros *et al.*, 2006; ECHA, 2014; CONCAWE, 2016), but less strong than PFOS; PFOA leaches significantly better than PFOS (Alphenaar *et al.*, 2018).
The adsorption of PFOA was measured in sediment at several locations in the Netherlands (Kwadijk *et al.*, 2010). The average log K_{oc} for 19 samples was 2.63±0.34 (Table 37). The average value for the anion in 2 samples, measured by Higgins and Luthy (2006), is slightly lower (2.06). At lower pH and with increasing Ca in the soil, the absorption of PFOA is stronger (Higgins & Luthy, 2006). The log K_{oc}, determined by adsorption measured in six soils with different OC content, was 2 (K_{oc} 96) (Milinovic *et al.*, 2015). Desorption was 24 to 58%, which is higher than the desorption of PFOS (<13%) (Milinovic *et al.*, 2015).
The log K_{oc} measured by Ahrens *et al.* (2010) in sediment from the bay of Tokyo is 1.9±0.1. Log K_{oc} values for sandy and oily river sediment from Japan are higher, with values of 4.5 and 2.5 respectively. The OC content had a significant impact on adsorption. (Ahrens *et al.*, 2011). de Voogt *et al.* (2006) have measured Kd values in sediments from the Rhine, in laboratory experiments and in the field. The log Kd intervals were -0.22 to 0.53 and -0.40 to 0.30 respectively.
The average log Kd of 19 sediments from different locations in the Netherlands is 1.83±0.40 (Kwadijk *et al.*, 2010). The adsorption behaviour of PFOA was measured in six soils with different characteristics, mainly in terms of organic carbon content (0.2 to 39%) (Milinovic *et al.*, 2015). The

log Kd rose from 0.34 to 1.58 (Kd 2.2 to 38) l/kg; the corresponding Freundlich coefficients rose from 4 to 40 l/kg. There was a positive correlation between adsorption and organic carbon content. Li *et al.* (2018) who carefully analysed this correlation, claim that the correlation was strongly influenced by one sediment with a high OC content; they removed this value, resulting in a weaker correlation. According to the authors, the sorption behaviour is primarily determined by the hydrophobia Milinovic *et al.* (2015) The lower limit of the log Kd interval for the ammonium salt is significantly lower (log Kd from -0.39 to 0.94) (Dupont, 2003).
On behalf of OVAM, Kd values were calculated for soils contaminated with PFAS. To this end, OVAM selected two industrial sites, each with an old and a new fire drill site, the top layer of which was sampled by (OVAM, 2018). Four samples were subjected to shaking tests and the Kd was calculated as the ratio between total concentration and eluate concentration. It was assumed that the concentrations in solution after the shake test were in balance with the solid phase. The Kd values were respectively 12.5 and 1.7 l/kg for site 1 and 9.5 and 10.6 l/kg for site 2. The median Kd for both sites was 10.0 l/kg.

The adsorption of PFOA is moderate and lower than that of PFOS. Stronger adsorption takes place at lower pH; an increase in calcium in the soil also reinforces the adsorption (Higgins & Luthy, 2006). For the calculations of the soil remediation values the log K_{oc} of 2.06 from Higgins and Luthy (2006) is used. This value is one of the lower experimental K_{oc} values in Table 37 (median 2.28), and thus worst-case (the lower the K_{oc}, the more mobile the substance). EFSA (2008b), the Netherlands (Lijzen *et al.*, 2018) and US-EPA (2016d) also selected the log K_{oc} value of Higgins and Luthy (2006) in their evaluations of PFOA.
The Kd is calculated by S-Risk from the K_{oc} with the formula Kd = OC(organic C content) x K_{oc}.

Note: The Dutch expertise centre for PFAS is of the opinion that the relationship between organic carbon and adsorption is less clear for PFAS than for other organic substances, due to the surfactant behaviour of PFAS. It is therefore not straightforward to correct, for PFAS, the intervention value for the soil organic matter content, as is common practice in the soil remediation values system for organic contaminants (Alphenaar et al., 2018). Using published data, Li *et al.* (2018) evaluated the role of organic carbon and other soil properties in the adsorption of PFAS in soil and sediment. The authors found weak correlations between Kd and only OC (R^2 = 0.05-0.07). For pH alone, the correlation with Kd was also weak, with a R^2 of 0.07 for PFOA over a pH interval of 4.5 to 10 (n = 36). Using multiple regression models, it was shown that at least the OC, pH and clay content have a significant effect on the sorption. For PFOA, the R^2 for these three parameters together rose to 0.45.

→ Octanol-air partition coefficient (K_{oa})

The log K_{oa} calculated with the COSMOtherm model is 7.23 (Wang *et al.*, 2011); the log K_{oa} calculated with K_{oa}Win (EpiSuite) is 3 orders of magnitude lower (4.24). The formula used by K_{oa}Win is K_{oa} = K_{ow}/Kaw. S-Risk also calculates log K_{oa} based on log K_{ow}. However, for the reasons discussed above, using log K_{ow} is avoided as much as possible for deriving the SRV, to reduce the uncertainty within the modelling of the soil remediation value. COSMOtherm is a commercial model and the underlying formula for calculating the K_{oa} is not freely available. In S-Risk, K_{oa} is used in the calculation of transfer to plants; as experimental data are available for this purpose, the import of a K_{oa} is not necessary. Entering a K_{oa} value is optional in S-Risk. For the calculations of the soil remediation values it is therefore not necessary to select a log K_{oa}.

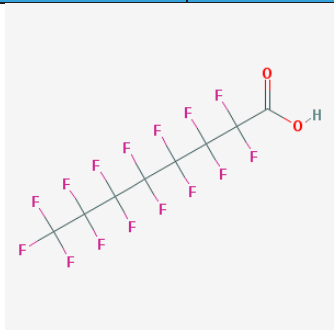
→ **Permeation coefficient through drinking water pipes (Dpe, Dpvc)**

No values were found for diffusion of PFOA through polyethylene or PVC drinking water pipes. In the Netherlands, for Dpe, the default value of 1.10^{-7} m²/d is calculated in CSOIL, and this choice is accounted for as follows: *For the common contaminants, Dpe is in the range 0.10-35 * 10^{-7} m^2/day (Vonk, 1985). In the absence of data, it is recommended to use the permeation coefficient of a substance with a similar structure (van den Berg, 1997). Failing this, the calculation is made with a default value of 1*10^{-7} m^2d, which is also used in other compounds* (Lijzen *et al.*, 2018). According to the authors, diffusion is likely to be low compared to that of the most common contaminants (Vonk, 1985) because PFOA is a large and elongated molecule and therefore takes a lot of energy to move through the polymer structures. However, PFOA has moderate solubility and it cannot be ruled out that PFOA has a considerable affinity with the apolar environment of the polyethylene. This effect is probably smaller than the high resistance to movement (Lijzen *et al.*, 2018). In its most recent report on risk limits for PFOA, RIVM states a Dpe of 1.10^{-7} m²/d in Table 3.1 (referring to previous RIVM reports) and of $3.15.10^{-10}$ m²/d in Annex 3 (input data for CSOIL); no reference or calculation method is given for the latter value (Wintersen *et al.*, 2019). As such, we prefer to follow the Dpe with the justification of Lijzen *et al.* (2018).
By default, the value of Dpvc = Dpe/1000, according to the technical guidance document of S-Risk (Cornelis *et al.*, 2017).
For the calculations of the soil remediation values, the value for Dpe is assumed to be 1.10^{-7} m²/d.; this is the standard value used by Lijzen *et al.* (2018) for calculations in CSOIL. For Dpvc, the value 1.10^{-10} m²/d (this is Dpe/1000) is used.

→ **Diffusion for organic substance in air (Da) and water (Dw)**

These values are used to calculate the diffusion when evaporation to outdoor and indoor air takes place. Entry in S-Risk is optional. No values were found for diffusion of PFOA in air or water. Therefore, for the calculations of the soil remediation values, both values are calculated in S-Risk, starting from the molecular weight, as specified in the technical guidance document (Cornelis *et al.*, 2017).

Table 37: Physicochemical properties of PFOA, the values used for the calculations in S-Risk are indicated in bold.

Parameter	Unit	Value	Original reference	Reference
Chemical structure				PubChem[3]
Type		Organic		
Physical state		solid (powder)		EFSA (2008b)
Solubility in water	mg/l	3400 mg/l; 4100 mg/l at 22°C	US-EPA (2005)	EFSA (2008b)
		9.5.10³ mg/l (25°C)	ECHA (2014)	Lijzen *et al.* (2018); FSANZ (2016)
		9.5.10³ mg/l (25°C) 4.14.10³ mg/l (22°C)	ECHA (2014)	Lijzen *et al.* (2018)
pKa		3400 mg/l;		Deng *et al.* (2012)
		3400 – 9500 mg/l at 20-25°C		CONCAWE (2016); Pancras *et al.* (2018)
		2290 (at 24°C) 3300 4340 (at 24.1°C)		HSDB
Melting point		54.3°C		FSANZ (2016)

		45-50°C	US-EPA (2005)	EFSA (2008b)
		37-60°C		CONCAWE (2016)
Density	g/cm³	1.8 at 20°C	Kroschwitz and Howe-Grant (1994)	CONCAWE (2016); ATSDR draft (2018) draft
Vapour pressure (Vp)	Pa	100 Pa (20°C) 4.2 Pa (25°C) (0.0081 Pa at 25°C ammonium salt)	US-EPA (2005)	EFSA (2008b)
		4-1300 Pa		CONCAWE (2016); Pancras *et al.* (2018)
		70 (at 25°C)	EpiSuite (exp); US-EPA (2016d; 2016b)	HSDB; Lijzen *et al.* (2018)
		128 (at 59.3°C) 4.2 (extrapolated to 25°C) 2.3 (extrapolated to 20°C)	ECHA (2014); US-EPA (2016d; 2016b)	ATSDR draft (2018); Lijzen *et al.* (2018)
Henry coefficient (Henry constant)[27]	Pa m³/mol	0.04-0.09		Pancras *et al.* (2018)
		Cannot be estimated[28]		EFSA (2008b)
		0.362 (protonated form) (very low pH)	Kwan (2001)	ATSDR draft (2018)
Log K_{ow}[29]	g/g	0.7 (ammonium salt)	3M-Company (1979)	EFSA (2008b)
		Not measurable	US-EPA (2016d; 2016b)	EFSA (2008b); ATSDR draft (2018); Lijzen *et al.* (2018)

[27] Calculated in S-Risk

[28] *The vapour pressure of the pure solid is sufficient to sustain mg/kg concentrations of vapour in the atmosphere, but in practice this is unlikely as PFOA will dissociate in aqueous media thereby reducing its vapour pressure above aqueous solutions. For this reason the Henry's Law constant cannot be estimated from the vapour pressure and solubility.*

[29] Entered in S-Risk but not used in further calculations

		6.30 (estimate) in octanol-water mixture		FSANZ (2016); ILO-ICSC[30]
		5.30 (calculated)	Wang *et al.* (2011)	CONCAWE (2016); Pancras *et al.* (2018)
		4.81(calculated)	EpiSuite	
Log K_{oc}	l/kg	**2.06** (anion) (sediment) (log normal average log K_{oc}) (n=2)	Higgins and Luthy (2006); US-EPA (2016d; 2016b)	EFSA (2008b); Lijzen *et al.* (2018)
		2.11 (anion) (sediment) (regression log K_{oc}) (n=4)	Higgins and Luthy (2006)	
		1.15 – 2.96	ECHA (2014)	Lijzen *et al.* (2018)
		1.31 – 2.35		CONCAWE (2016); Pancras *et al.* (2018)
		2.63 (field data, sediment, n=19)	Kwadijk *et al.* (2010)	
		1.9 (field data, sediment)	Ahrens *et al.* (2010)	Ahrens *et al.* (2011)
		4.5 (river sediment, sandy, f_{oc} 0.03%) 2.5 (river sediment, muddy, f_{oc} 1.6%)	Ahrens *et al.* (2011)	Zareitalabad *et al.* (2013)
		2 (K_{oc} 96) (OC 0.2-39%)	Milinovic *et al.* (2015)	
		1.2-2.4 (K_{oc} 17-230)	Prevedouros *et al.* (2006)	ATSDR draft (2018)
Log K_d	dm^3/kg	-0.2-0.53 (sediment)	de Voogt *et al.* (2006)	EFSA (2008b)[31]
		-0.4-0.30 (field data, sediment)	de Voogt *et al.* (2006)	
		1.83 (1.19-2.85) (field data, sediment, n=19)	Kwadijk *et al.* (2010)	
		-0.39-0.94 (ammonium salt) (OECD protocol 106)	Dupont (2003)	EFSA (2008b)
		≤ 0.53 (Kd 0 – 3.4 (pH 7)		CONCAWE (2016); Pancras *et al.* (2018)
		0.34 (Kd 2.2) (OC 0.2%) 0.41 (Kd 2.6) (OC 1.6%) 0.43 (Kd 2.7) (OC 3.9%)	Milinovic *et al.* (2015)	

[30] http://www.ilo.org/dyn/icsc/showcard.display?p_version=2&p_card_id=1613

[31] EFSA (2018) indicates 0,55 as the highest value of the interval

		0.59 (Kd 3.9) (OC 7.7%) 0.85 (Kd 7.1) (OC 9.4%) 1.58 (Kd 38) (OC 39%)		
$K_{ads}F$ (Freundlich coefficient)	l/kg[32]	4 (OC 0.2%) 2 (OC 1.6%) 2 (OC 3.9%) 2 (OC 7.7%) 7 (OC 9.4%) 40 (OC 39%)	Milinovic *et al.* (2015)	
Log K_{oa}[33]	g/g	4.24 (calculated)		EpiSuite K_{oa}Win v1.10[5]
		7.23 (calculated)	Wang *et al.* (2011)	
Dissociative		Yes		EFSA (2008b)
Acid dissociation constant (pKa)		**2.80**	Moody and Field (2000)	FSANZ (2016); HSDB; Lijzen *et al.* (2018)
		3.80	Burns *et al.* (2008)	FSANZ (2016)
		2 – 3	Prevedouros *et al.* (2006)	EFSA (2008b)
		1.30	Kutsuna and Hori (2008)	HSDB
		-0.16 to 3.8		CONCAWE (2016); Pancras *et al.* (2018);
		-0.2 to 3.8	ECHA (2014)	Lijzen *et al.* (2018)
		0.5 (estimated)		FSANZ (2016)
Dpe	m^2/d	**1.10^{-7}**	Vonk (1985); van den Berg (1997)	Lijzen *et al.* (2018)
Dpvc	m^2/d	**1.10^{-10}** (Dpe/1000)		Cornelis *et al.* (2017)
Da	m^2/d	No data		
Dw	m^2/d	No data		

[32] This unit assumes that n=1; the exact unit is $\mu g^{1-1/n}(l)^{1/n}kg^{-1}$

[33] Log Koa is optional in S-Risk, which uses Koa in the calculation of transfer to plants; as experimental data are available for this purpose, a Koa value is not necessary.

3.4. OCCURRENCE IN THE ENVIRONMENT

3.4.1. SOIL

PFOA does not occur naturally in the soil.

3.4.2. AIR

→ Outdoor air

There are no data on perfluorinated compounds in the air in Flanders.
In the project BF-Risk (Cornelis *et al.*, 2009) the outdoor air concentration was derived from a number of European studies (Barber *et al.*, 2007; Jahnke *et al.*, 2007a; Dreyer & Ebinghaus, 2009). For outdoor air, the authors used the available measurement data and processed it statistically. This resulted in a P50 of $8.9.10^{-9}$ mg/m³ and a P95 of $5.52.10^{-7}$ mg/m³ for PFOA. Based on European literature, EFSA (2008b) defined two scenarios for low and high exposure, with outdoor air concentrations of $3.00.10^{-9}$ mg/m³ and $3.00.10^{-7}$ mg/m³ respectively, which are of the same order of magnitude as those used in BF-Risk.
In the GAPS (Global Atmospheric Passive Sampling) network, POPs, including PFOA are measured at 21 different places on earth. Rauert *et al.* (2018) compared the concentrations for 2009, 2013 and 2015. Samples were taken in 3 types of areas: background, urban and polar. In Europe, samples were taken in the Czech Republic (background), Norway (polar), Ireland (background) and Paris (urban). In 2015, the background concentrations for PFOA (across all background measuring stations) were $<6.00.10^{-10}$ – $7.20.10^{-9}$ mg/m³, in 2013 they were $<9.00.10^{-10}$ – $6.20.10^{-9}$ mg/m³ and in 2009 they were $<9.00.10^{-10}$ – $1.20.10^{-8}$ mg/m³. The authors also refer to a study in Switzerland where a background concentration of $1.7.10^{-9}$ mg/m³ was calculated (Muller *et al.*, 2012). In 2015, for urban areas the concentrations of PFOA were respectively $<9.00.10^{-10}$ – $4.00.10^{-8}$ mg/m³, in 2013 they were $2.50.10^{-10}$ – $9.90.10^{-9}$ mg/m³ and in 2009 they were $<9.00.10^{-10}$ – $3.20.10^{-9}$ mg/m³. Muller *et al.* (2012) calculated $7.70.10^{-9}$ mg/m³ for the urban area.
For deriving the soil remediation value for PFOA , we use a concentration of $8.9.10^{-9}$ mg/m³ PFOA in outdoor air (P50 value from Cornelis *et al.* (2009)). This value is of the same order of magnitude as the background concentrations and concentrations in urban areas determined in the GAPS network. In addition, it is similar to the urban concentrations calculated by Muller *et al.* (2012).

→ Indoor air

Higher concentrations can be found in indoor air than in outdoor air, due to indoor sources. In the BF-Risk project a concentration of $2.6.10^{-9}$ mg/m³ was assumed for indoor air based on data from Jahnke *et al.* (2007b). **For deriving soil remediation values, the background concentration in indoor air is equal to that in outdoor air**.

3.4.3. DRINKING WATER

There are no VMM (Flanders Environment Agency) measurement data for PFOA in drinking water. In the BF-Risk project, 4 samples of tap water from 3 different drinking water companies were

analysed. The measured concentrations of PFOA were $1.1.10^{-3}$ - $4.7.10^{-3}$ µg/l with a median of $1.1.10^{-3}$ µg/l and an average concentration of $2.04.10^{-3}$ µg/l, the latter value was used for the intake estimation in the BF-Risk project (Cornelis *et al.*, 2009; D'Hollander *et al.*, 2009). Costopoulou *et al.* (2015) investigated 11 PFAS in tap water samples from Greece and the Netherlands. For the Dutch samples, PFAS were detected in 49% of the samples. When the tap water came from groundwater, the results were smaller than the LOQ. The measured concentrations of PFOA in Dutch drinking water were 1.9 – $11.1.10^{-3}$ µg/l. Taking both groundwater and surface water into account, an average concentration of 1.7 (lower bound[34, 10]) and $1.9.10^{-3}$ µg/l (upper bound[35]) respectively was recorded in Dutch tap water.
EFSA (2012) calculated an average concentration of respectively $2.30.10^{-3}$ µg/l and $5,30.10^{-3}$ µg/l (upper bound results) based on 152 samples of tap water from different European countries over the period 2006-2012. For bottled water (254 samples) the concentration was slightly lower, $0.30.10^{-3}$ µg/l (lower bound) and $1,60.10^{-3}$ µg/l PFOA (upper bound results) respectively.
Although the results determined in BF-Risk come from a limited amount of samples, the results are similar to those determined by EFSA and Costopoulou *et al.* (2015). For this reason, a concentration of $2.04.10^{-3}$ µg/l PFOA is assumed for tap water (average value 4 Flemish tap water samples determined in BF-Risk). Depending on the data used for intake via food, this value may or may not be used when calculating the soil remediation value. If drinking water is already included in the diet, it will be equated to zero here in order to avoid double counting.

3.4.4. CONCENTRATIONS IN FOODSTUFFS AND INTAKE VIA FOOD

Concentrations of PFOA in foodstuffs and intake estimates for Flanders/Belgium are discussed in BF-Risk and were calculated by EFSA in 2012 and 2018[36]. In the literature, data are also available for Sweden and the Netherlands, and in the European project PERFOOD the exposure via food was quantified for 4 European countries including Belgium. For the calculation of the soil remediation value, data should ideally be used in which concentrations in food and intake estimates are linked.

BF-Risk – Flanders

In the project BF-Risk (Cornelis *et al.*, 2009; D'Hollander *et al.*, 2009) samples of foodstuffs of Flemish origin were analysed, with a distinction between samples originating from organic farming and conventional farming. The samples analysed were divided into a number of groups including "vegetables" (potato, carrot, tomato, chicory, onion, lettuce, leek and wheat), "fruit" (apple and strawberry), "meat" (chicken, pork and beef), "dairy" (chicken eggs and raw cow's milk), "fish" (eel, cod, rocket, dab, whiting, herring, sprats and flounder) and "drink" (beer and tap water). For each item 6 growers were sought, 3 organic growers and 3 "classic" growers. Three pieces per selected food item were purchased from each grower. Table 2 gives an overview of the results of the analysis for a number of vegetables, meat, cow's milk and eggs. In fish from the North Sea, $6.0.10^{-4}$ mg/kg of PFOA was detected.

Table 38: Overview of the range (minimum-maximum) of PFOA concentrations and the median in organic and conventional food samples (D'Hollander *et al.*, 2009).

Foodstuff	Organically grown	Conventionally grown

[34] For the results below the LOQ, a 0 value was assumed as concentration
[35] A concentration equal to the LOQ ($0.6 .10^{-3}$ µg/l) was assumed for results below the LOQ.
[36] After finalising this report, EFSA published new data in 2020

	Min-max (median) mg/kg	Min-max (median) mg/kg
Potato	<2.00.10^{-4} -2.00.10^{-3} (3.00.10^{-4})	< 6.00.10^{-4} – 9.00.10^{-4}- (<6.00.10^{-4})
Carrot	2.00.10^{-4}-5.00.10^{-4} (3.00.10^{-4})	<6.00.10^{-4} (<6.00.10^{-4})
Onion	<1.00.10^{-4}-1.30.10^{-3} (3.00.10^{-4})	<6.00.10^{-4} -2.3.10^{-3} (1.30.10^{-3})
Tomato	<1.00.10^{-4}-6.00.10^{-4} (3.00.10^{-4})	<6.00.10^{-4} (<6.00.10^{-4})
Lettuce	<1.00.10^{-4}-9.00.10^{-4} (<1.00.10^{-4})	<6.00.10^{-4} -8.00.10^{-4} (<6.00.10^{-4})
Beef	<1.00.10^{-4} (<1.00.10^{-4})	<1.00.10^{-4}- 3.3.10^{-3} (1.70.10^{-3})
Chicken	<1.00.10^{-4} (<1.00.10^{-4})	<1.00.10^{-4} (<1.00.10^{-4})
Pork	<1.00.10^{-4} (<1.00.10^{-4})	<1.00.10^{-4} (<1.00.10^{-4})
Eggs	<1.00.10^{-4}-7.00.10^{-4} (3.00.10^{-4})	<6.00.10^{-4} -5.0.10^{-3} (<6.00.10^{-4})
Cow's milk	<3.00.10^{-4} (<3.00.10^{-4})	<3.00.10^{-4}- 3.00.10^{-4} (<3.00.10^{-4})

For the calculation of exposure, Cornelis *et al.* (2009) included data from foreign studies as foodstuffs on the Belgian (Flemish) market do not only originate from Belgium and the dataset with concentration data measured in the BF-Risk project is too limited. To this end, literature data were looked up from measurements carried out from 2003 and published until mid-May 2009. For fruit and vegetables, the levels from the BF-Risk project did not differ significantly from those from other European studies (Spain, UK). Cornelis *et al.* (2009) did however observe that there was significant variation in the data. For vegetables, the authors distinguished between vegetables and potatoes, see Table 3. No data were available for butter and therefore the data for dairy were used.

Table 39: Average concentrations of PFOA in a selection of foodstuffs as used in the intake estimate by Cornelis *et al.* (2009) (mg/kg fresh weight).

Food group	PFOA
Potato	6.74.10^{-4}
Vegetables	6.47.10^{-4}
Butter	1.23.10^{-4}
Egg	8.63.10^{-4}
tap water and bottled water	2.04. 10^{-6}
Liver	0.00.10^{-3}
Milk	1.23.10^{-4}
Meat	5.20.10^{-4}
Poultry	5.49.10^{-5}
Seafish	5.86.10^{-4}
Fresh water fish	7.82.10^{-4}

For the intake via food and drinking water, the results of the Belgian Food Consumption Survey of 2004, carried out by WIV, were used (De Vriese *et al.*, 2006). This was the most recent study into food intake in the Belgian population over 15 years of age.

The average intake for the different age groups as calculated in BF-Risk is given in Table 4. The intake is dominated by potatoes, fruit, vegetables and fish and shellfish.

Table 40: Intake of PFOA via food by the Flemish (Belgian) population (ng/kg.day) (Cornelis *et al.*, 2009)

Age	1-<3	3-<6	6-<10	10-<15	15-<21	21-<31	31-<41

(years)							
Intake (ng/kg.d)		20.1	12.1	8.09	6.32	6.18	5.64

Noorlander *et al.* (2010) - *Netherlands*

Noorlander *et al.* (2010) calculated the intake of PFOA via food and drinking water in the Netherlands using the 'total diet method', which is a combination of consumption data, concentration measurements in mixed samples of specific food categories and drinking water, and statistical modelling. Drinking water samples were not measured, the concentrations reported by EFSA (2008b) were used instead, namely $9.00.10^{-3}$ µg/l. Table 41 shows a selection of concentrations measured in food samples in 2009 in the Netherlands.

Table 41: Concentration of PFOA in food groups collected in 2009 in the Netherlands (results > LOD are shown in bold).

Food group	PFOS in mg/kg
Oily fish	**$8.00.10^{-6}$**
White fish	**$2.30.10^{-5}$**
Butter	**$1.60.10^{-5}$**
Milk	**$1.00.10^{-6}$**
Eggs	$<3.20.10^{-5}$
Beef	$<5.0.10^{-6}$
Vegetables/fruit	**$5.00.10^{-6}$**

The long-term intake was calculated for PFOA, taking into account concentration data from 2010 and data from the Dutch food consumption survey in 1998 (Table 41). The intake of drinking water contributed 55% of the total intake, followed by fruit and vegetables (19%), flour (15%) and pork (6%). The calculated intakes for the Netherlands are slightly lower than those calculated by EFSA (2012) for the UB scenario (comparable for LB scenario) and Cornelis *et al.* (2009) and comparable with those calculated by Vestergren *et al.* (2012).

Table 42: Long-term intake via food for PFOA in the Netherlands (Noorlander *et al.*, 2010; Noorlander *et al.*, 2011) for the P50 percentile (in ng/kg.d). Values under the LOD were approached according to 3 different scenarios.

Age (years)	S1: male	S1: female	S2: male	S2: female	S3: male	S3: female
2	0.385	0.470	0.424	0.452	0.691	0.736
10	0.181	0.192	0.203	0.216	0.328	0.355
40	0.222	0.236	0.237	0.252	0.314	0.338
Lifelong average	0.120	0.226	0.229	0.244	0.324	0.346

When the analysis results are ≥LOD but < LOQ, 3 scenarios can be identified:
S1: result = LOD; S2: result = measured concentration; S3: result = LOQ

The intake data from Noorlander *et al.* (2011) are used in Wintersen *et al.* (2019) to calculate the contribution of the various foodstuffs to the background exposure when calculating risk limits for the application of PFOA-containing soil and sludge for arable and livestock farming.

EFSA (2012 and 2018) - Belgium

EFSA (2012) collected analytical results of PFOA in foodstuffs in 13 European countries over the period 2006-2012 (54,195 analytical results distributed among all PFAS of which 7,536 for PFOA, 2% of these samples came from Belgium). A selection of results is given in Table 43. The dataset was characterised by a high proportion of left-skewed data (results > LOD or LOQ), 91% for PFOA (10,522 samples of which 9,828 were eventually retained).
When the concentrations (upper bound) are compared with the samples for conventional cultivation measured in BF-Risk (Table 38), we can see that for onions, tomatoes, lettuce and beef, the EFSA results are up to 10 times lower than those measured in D'Hollander *et al.* (2009). For potatoes, carrot, milk and eggs, the results are in the same order of magnitude. A comparison with the data used for the intake estimation in BF-Risk (Table 39) shows that the mean concentrations for potatoes, a number of vegetables, eggs, milk and meat are in the same order of magnitude.

Table 43: Selection of concentration data for PFOA in foodstuffs according to EFSA (2012).

Food group	Number of samples (including mixed samples)	Proportion of left-skewed distribution of data	Average Lower bound (mg/kg)	Average Upper bound (mg/kg)
Root crops	134 (276)	97	$3.40.10^{-6}$	$2.00.10^{-4}$
Bulbous vegetables	8 (68)	88	$2.20.10^{-6}$	$4.00.10^{-5}$
Fruiting vegetables	37 (243)	81	$4.50.10^{-6}$	$6.70.10^{-5}$
Cabbages	23 (111)	87	$1.90.10^{-6}$	$1.10.10^{-4}$
Leafy vegetables	25 (210)	64	$6.20.10^{-6}$	$3.90.10^{-5}$
Leguminous vegetables	4 (13)	25	$2.50.10^{-5}$	$2.80.10^{-5}$
Stalk vegetables	23 (176)	78	$3.00.10^{-6}$	$8.20.10^{-5}$
Potatoes and potato products	299 (335)	99.7	$9.00.10^{-7}$	$6.40.10^{-4}$

Beef	232 (1418)	95	$6.10.10^{-6}$	$1.30.10^{-4}$
Poultry	150 (735)	99	$2.40.10^{-6}$	$1.40.10^{-4}$
Offal	1265 (1655)	98	$3.40.10^{-5}$	$1.40.10^{-3}$
Milk	152 (722)	100	0	$1.20.10^{-4}$
Butter (animal fat)	12 (54)	92	$1.7.10^{-6}$	$1.3.10^{-4}$
Eggs	99 (585)	87	$8.80.10^{-5}$	$7.6.10^{-4}$
Fish and other	2542 (4403)	95	$8.20.10^{-5}$	$6.9.10^{-4}$
Tap water	110 (152)	92	$2.30.10^{-6}$	$5.30.10^{-6}$
Bottled water	254 (254)	82	$3.00.10^{-7}$	$1.60.10^{-6}$

Left-skewed data = data < LOQ or LOD
Lower bound: value 0 assigned to all left-skewed data, upper bound: value of LOQ or LOD assigned to left-skewed data

For the calculation of the intake, EFSA used the data in the Belgian Food Consumption Survey (BVCP 2004, Devriese *et al.* (2006)) and for children, from the Flemish toddler study (Huybrechts, 2008). The intakes for Belgium calculated in EFSA (2012) are given in Table 44, drinking water is part of the calculated intakes. In these calculations, EFSA (2012) makes the reservation that a chronic intake estimate via food cannot be accurate when more than 80% of the analysis results are lower than the LOD or LOQ. For this reason, the intake estimate below is a rough indication of exposure. If the lower bound method is used, the intake is likely to be underestimated; if the upper bound method is used, it may be significantly overestimated. When the calculated intakes from EFSA are compared with those in Cornelis *et al.* (2009), it can be seen that for most age groups the data calculated in BF-Risk are of the same order of magnitude, and slightly higher than those calculated by EFSA (upper bound).

Table 44: Average chronic PFOA intake via food for the Belgian population, lower bound (LB) and upper bound (UB) approach (EFSA, 2012).

Age (years)	1-3	3-10	10-18	18-65	65-75	≥75
Intake LB - UB (ng/kg.d)	0.220-11.0	0.180-8.80	0.09-2.8	0.110-2.6	0.91-2.50	0.120-2.70

EFSA (2012) states fish, fruit, eggs, drinking water, meat and vegetables (in descending order) as the main food groups contributing to the intake of PFOA for adults. For children, depending on their age, meat (3-10 years old) or fruit (1-3 years old) are the main sources.

At the end of 2018, EFSA published a new intake estimate for chronic exposure to PFOS and PFOA EFSA (2018c), based on 21,411 samples for which data for PFOS and PFOA were available (end of 2016). 62% of the samples came from Germany, followed by Norway and France. The samples were reported between 2000 and 2016, but only samples collected after 2007 were included in the calculations. For PFOS, the calculation was made with 10,012 results, for PFOA with 9,828 results. As in 2012, the data were characterised by a large proportion of left-skewed data (results < LOD or LOQ). The intake estimate for PFOA, according to EFSA, is about 30% lower in 2018 than in 2012 when using the UB approach, the LB approach results in a four times higher intake for 2018. Milk and drinking water contribute most to PFOA exposure. The EFSA report is accompanied by annexes in Excel, which present average and P95 intakes for Belgium for different age groups using the lower and upper bound approach, these data are shown in Table 45.

Table 45: Average chronic PFOA intake via food, lower bound (LB) and upper bound (UB) approach for Belgium, obtained by EFSA in 2018 (EFSA, 2018c).

Age (years)	1-3	3-10	10-18	18-65	65-75	≥75
Intake LB - UB (ng/kg.d)	0.34-4.02	0.34-3.62	0.33-1.67	0.27-1.44	0.21-1.39	0.21-1.35

The concentrations used by EFSA for the intake estimation are shown in Table 46, these are also taken from the Excel tables annexed to the EFSA report. Although the intake estimates in 2018 are lower than those in 2017, the concentrations reported by EFSA are higher for most foodstuffs in 2018. No explanation could be found for this in the report.

Table 46: Selection of concentration data for PFOA in foodstuffs according to EFSA (2018c).

Food group	Number of samples	Proportion of left-skewed distribution of data (%)*	Average Lower bound (mg/kg)**	Average Upper bound (mg/kg)
Root vegetables	163	90	$1.6.10^{-5}$	$2.39.10^{-4}$
Bulbous vegetables	75	98	$1.0.10^{-6}$	$2.48.10^{-4}$
Fruiting vegetables	122	88	$4.0.10^{-6}$	$1.74.10^{-4}$
Cabbages	35	82	$2.0.10^{-6}$	$1.52.10^{-4}$
Leafy vegetables	103	81	$7.0.10^{-6}$	$1.41.10^{-4}$
Leguminous vegetables	5	29	$2.3.10^{-5}$	$6.2.10^{-5}$
Stalk vegetables	85	89	$2.0.10^{-6}$	$2.30.10^{-4}$
Potatoes	50	100	0	$4.06.10^{-4}$
Beef	146	96	$5.40.10^{-5}$	$1.86.10^{-4}$
Poultry	172	98	$3.00.10^{-6}$	$1.45.10^{-4}$
Offal	1415	97	$4.50.10^{-5}$	$1.390.10^{-3}$
Milk	241	98	$6.30.10^{-5}$	$2.66.10^{-4}$
Eggs	166	92	$1.10.10^{-4}$	$3.69.10^{-4}$
Fish and other	2855	94	$2.24.10^{-4}$	$8.84.10^{-4}$
Tap water	46	61	$3.00.10^{-6}$	$9.00.10^{-6}$
Bottled water	330	86	0	$1.00.10^{-6}$

*Left-skewed data = data < LOQ or LOD
**Lower bound: value 0 assigned to all left-skewed data, upper bound: value of LOQ or LOD assigned to left-skewed data

Vestergren et al. (2012) - Sweden

Vestergren *et al.* (2012) calculated the intakes of various PFAS for the Swedish population in 1999, 2005 and 2010. The authors used a highly sensitive analysis technique, the analysis results for a selection of food samples can be found in Table 47.

Table 47: Estimated (between LOD and LOQ) and measured (> LOQ, indicated in bold) concentrations in mg/kg PFOA in Swedish food samples from 1999, 2005 and 2010 (Vestergren *et al.*, 2012).

Food group	2010	2005	1999
Dairy products	**$2.90.10^{-5}$**	**$1.60.10^{-5}$**	< LOD
Meat products	**$1.20.10^{-5}$**	**$1.60.10^{-5}$**	**$2.40.10^{-5}$**
Fats	< LOD	**$1.50.10^{-5}$**	**$1.00.10^{-5}$**
Pastry	**$1.80.10^{-5}$**	**$3.60.10^{-5}$**	**$4.70.10^{-5}$**
Fish	**$5.00.10^{-5}$**	**$3.20.10^{-5}$**	**$1.10.10^{-4}$**
Eggs	**$3.90.10^{-5}$**	**$7.20.10^{-6}$**	**$3.10.10^{-5}$**
Cereal products	**$6.20.10^{-5}$**	**$1.10.10^{-5}$**	**$1.20.10^{-5}$**
Vegetables	**$2.20.10^{-5}$**	**$5.20.10^{-5}$**	**$3.50.10^{-5}$**
Potatoes	**$5.70.10^{-5}$**	**$1.20.10^{-5}$**	**$4.9.10^{-6}$**

For PFOA, based on consumption data for the Swedish population, average intakes were calculated of 0.348, 0.495 and 0.692 ng/kg.d in 1999, 2005 and 2010, respectively (lower bound scenario, the difference with the upper bound scenario is less than 1%). These intakes were calculated without drinking water. The intake in the UB scenario is dominated by the food groups cereals (38%), dairy products (32%), vegetables (11%) and fruit (9.6%). Fish only contributes 6.2% to the intake (UB scenario). These results are comparable to the lower bound scenario obtained by EFSA (2012) and for 2010 are a factor of 4-10 lower than the upper bound scenario calculated by EFSA and the data in Cornelis *et al.* (2009). The concentrations in potatoes are a factor of 10 lower than those used in BF-Risk for the intake estimate and the UB data of EFSA (but almost a factor of 100 higher than the LB data of EFSA). As regards vegetables, the concentrations are similar or slightly lower than those in BF-Risk and EFSA (UB). For fish, the concentrations are about 10 times lower than the concentrations in BF-Risk and in the UB data of EFSA (but comparable to the LB data); for meat the concentrations are a factor of 2 lower than the UB data of EFSA and the concentrations used in BF-Risk.

PERFOOD- Belgium

In the European project PERFOOD (Klenow *et al.*, 2013) the exposure via food was quantified for 4 European countries including Belgium. For this, results from analytical analyses carried out during the project were used (Herzke *et al.*, 2013; Hlouskova *et al.*, 2013) as were consumption data for the individual countries as published by EFSA.
The concentrations for vegetables were approximately 60 times lower than those used in Cornelis *et al.* (2009) for the intake estimate (average 6.50^{-4} mg/kg versus $1.03.10^{-5}$ mg/kg for vegetables), see also Table 48.

Table 48: Concentration data for PFOA in Belgian samples measured in PERFOOD (Herzke *et al.*, 2013; Hlouskova *et al.*, 2013).

Food group	PFOA in mg/kg
Vegetables	$1.03.10^{-5}$
Broccoli	$1.02.10^{-5}$
Beans	$1.20.10^{-5}$
Peas	$1.05.10^{-5}$
Lettuce	$8.0.10^{-6}$
Spinach	$1.12.10^{-4}$
Asparagus	$1.60.10^{-5}$
Celery	$1.48.10^{-5}$
Cabbage, cauliflower, aubergine, courgette, cucumber, peppers, tomato, beans, peas, chicory, lettuce, carrot, potato, fennel	< LOQ
Pork	$7.20.10^{-6}$ *
Eggs	$4.90.10^{-6}$ *

* : figures calculated by VITO by combining data from figure 2 (sum of PFAS in foodstuffs for Belgium) and figure 3 (PFOA profiles foodstuffs Belgium) published in Hlouskova *et al.* (2013)

Due to the sensitive analysis technique, the authors consider a further interpretation of the results for the lower bound data to be justified. The intake for adults in the lower bound scenario is dominated by alcoholic drinks (52%) and fruit (26%). For children, fruit (74%) is the most dominant category. The results from the PERFOOD project and the lower bound results from EFSA (2012) are of approximately the same order of magnitude, the upper bound results of EFSA (2012) are approximately 10 times higher. The intake estimate is similar to the one carried out by Noorlander *et al.* (2010) and the one for Sweden (Vestergren *et al.*, 2012) and 30-60 times lower than the one calculated in BF-Risk (Cornelis *et al.*, 2009). The authors mention that the results in PERFOOD should not be considered representative for Belgium, as the samples were taken in a short period of time and at a limited number of sampling sites.

Table 49: Average PFOA intake via diet for children and adults in Belgium, calculated according to the lower and upper bound scenario (Klenow *et al.*, 2013).

Age (years)	3-9 average	18-64 average
Intake: LB – UB (ng/kg.d)	0.280-0.389	0.186-0.231

Discussion

Both Cornelis *et al.* (2009) and EFSA (2012) EFSA (2018c) and Klenow *et al.* (2013) published data for the intake estimate of PFOA for the Belgian population (see Figure 11). The highest results were obtained in BF-Risk (Cornelis *et al.* (2009) and the UB data from EFSA. The lowest intakes were calculated with the LB data of EFSA and in PERFOOD (Klenow *et al.* (2013). The concentrations used for intake estimations are based on own measurements or literature in Cornelis *et al.* (2009) and on own analysis results in Klenow *et al.* (2013). The number of samples analysed in both studies is limited or was collected during a limited period of time, and for this reason the results may not be representative. EFSA has a large amount of samples at its disposal, but it calculates intakes for Belgium with concentrations for foodstuffs from all the European countries that have made data available, so the concentrations do not relate only to foodstuffs present on the Belgian market.

Klenow *et al.* (2013) refer to higher concentrations of PFAS in foodstuffs originating from Belgium compared to other European countries, so it is possible that the concentrations used by EFSA are an underestimate. EFSA itself considers its UB calculations to overestimate the intake, since 80% of the analysis results are lower than the LOQ or LOD.

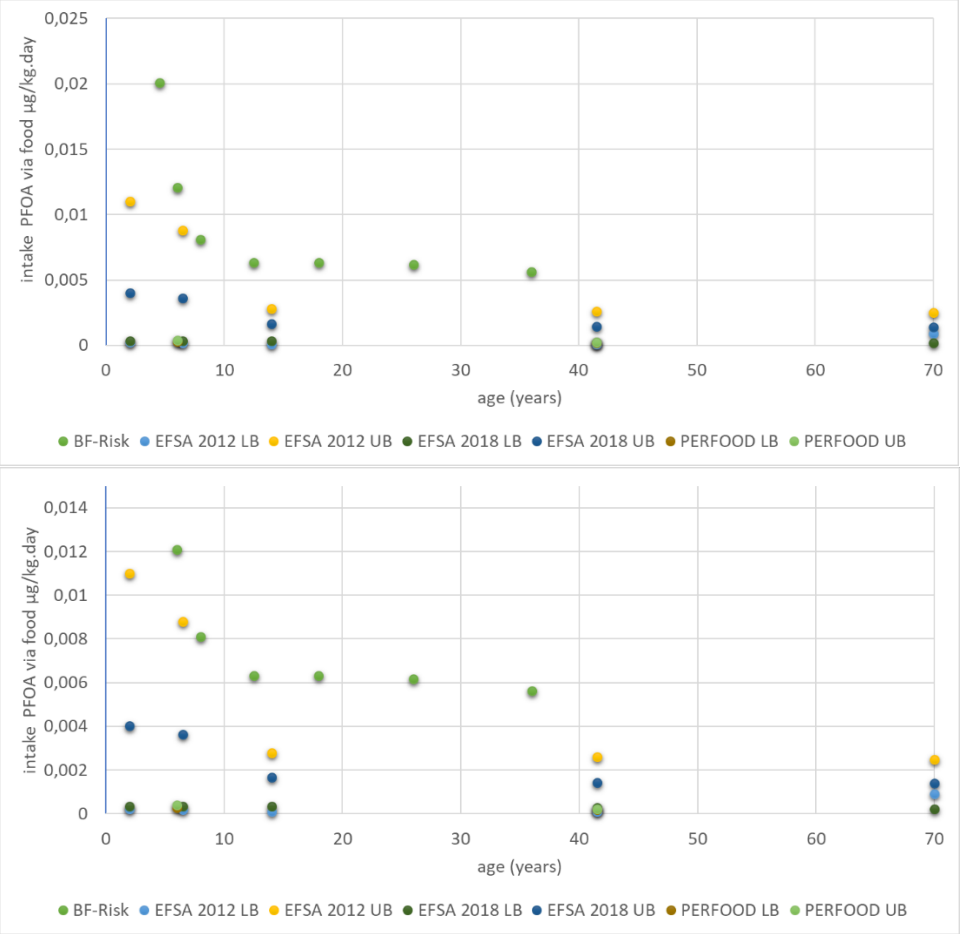


Figure 11: Intake PFOA for Belgium in µg/kg.day for different ages. Comparison of data published in BF-Risk, EFSA (upper bound and lower bound) and PERFOOD (upper bound and lower bound)

Taking into account the various aspects described above, it was decided to initially use the UB intake estimate made by EFSA for the Belgian population as intake estimate for PFOA. The consumption figures use specific data for Belgium and the concentrations are based on a large amount of samples. Using UB intake estimates is a very conservative approach, according to EFSA. This is also shown by a comparison of the intake estimates for the LB and UB scenario of EFSA with recent scientific literature, the LB approach of EFSA approaches the intake estimates based on more sensitive analytical techniques such as in Vestergren *et al. (2012)* and Perfood better than the UB approach. For deriving soil remediation values, both the LB and UB data are calculated through. As long as drinking water is part of the diet used by EFSA to estimate intakes, the concentration in drinking water for deriving soil remediation values will be set at zero to avoid double counting.

Table 50 shows the EFSA data extrapolated to the age groups present in S-Risk.

Table 50 PFOA intake via food for the Belgian population calculated on the basis of extrapolation of LB and UB approach of EFSA (2012) and (2018).

Intake (ng/kg.d)	1-<3	3-<6	6-<10	10-<15	15-<21	21-<31	31 and above
2012 UB	11.0	9.75	8.00	4.00	2.77	2.72	2.63
2012 LB	0.220	0.198	0.162	0.108	0.0924	0.0980	0.111
2018 UB	4.02	3.80	2.90	1.55	1.66	1.59	1.81
2018 LB	0.340	0.340	0.338	0.332	0.321	0.303	0.210

3.5. TRANSFER TO PLANTS

Like PFOS, PFOA is a strong acid with a long organic tail and does not behave like a standard organic compound. Due to the amphiphilic nature of PFAS, the formulas of Trapp (2002), Trapp *et al.* (2007) and Trapp and Matthies (1995)) usually applied in S-Risk for non-ionising organic compounds cannot be used to calculate plant uptake of PFAS, and therefore empirical relationships based on bioconcentration factors (BCF) need to be applied. In the literature, BCF-values for PFOA are generally based on soil concentrations, while in S-Risk BCF based on soil pore water are required for organic compounds. For this reason, modifications were made to S-Risk, using a test environment, in order to ba able to calculate soil remediation valuess with BCF_{soil} from the literature (see also 3.10 calculations of the soil remediation value).

For the plant uptake of PFOA we rely in the first instance on two important sources: i) a review paper by Ghisi *et al.* (2019) and the study by Lijzen *et al.* (2018). As far as possible, the numerical values from these overview publications are verified against the original data. Additional information is consulted and, if available, integrated.

Ghisi *et al.* (2019) bring together all known data related to plant uptake for different PFAS, differentiating between perfluoroalkyl carboxylic acids (PFCAs) and perfluoroalkyl sulfonic acids (PFSAs). Most of the data in this literature review relating to plant uptake for PFAS refer to PFOA and PFOS, C8 representatives for PFCAs and PFSAs respectively. This subset of PFSA-data from Ghisi *et al.* (2019) is the starting point for final selection of BCF values in S-Risk for the derivation of soil remediation values of PFOA. Studies relying on aquacultures for the uptake of PFAS are not taken into consideration for this study. A further literature search/review on data relating to uptake of PFOA by crops did not provide additional data. Due to the water solubility of PFAS, the industrial and urban sludge from water treatment plants used for irrigation are considered important sources for plant uptake. In addition, sludge applications intended to improve soil structure and the use of PFAS as emulsifiers in plant protection products are also important sources of plant uptake.

The literature often distinguishes between the uptake of PFOA in cereal crops and vegetables. This distinction is based, on the one hand, on the difference between the parts of plants that are suitable for consumption, vegetative parts including fruit in the case of vegetables and seeds in the case of cereal crops. On the other hand, the vegetative parts of cereals are used in fodder crops (straw, chaff, etc.). Ghisi *et al.* (2019) identify 3 publications relating to cereal crops, including maize, oats, wheat and ryegrass (grasses). BCF derived for cereals and grasses are the basis for the calculation of transfer

PFOA via locally grown fodder crops to livestock (biomagnification). A distinction can be made between parts of the plant suitable for human consumption (cereals) and non-edible parts of cereals and grasses used in livestock feed (straw, chaff). The results are summarised in Table 51. All numerical values in the table are based on measurements taken on spiked soils, with the exception of Wen et al. (2014). In the latter studies, the plants were grown on soils to which a mixture of PFAS was added in the form of (organic) sludge. The BCF for parts suitable for human consumption are often significantly lower than for non-edible parts. Differences in availability of PFOA for different crops are attributed to differences in protein content (Wen et al., 2016) or to morphological differences in e.g. leaf surface and/or root system (Müller et al., 2016).

Table 51: BCF (mg/kg plant dm)/(mg/kg soil dm) for PFOA for cereals and grasses (Selection from Ghisi *et al.* (2019))

Crop	compart-ment	concentration in soil (mg/kg dm)	BCF (mg/kg plant dm)/(mg/kg soil dm)	reference
maize	straw	0.25	0.272	Stahl *et al.* (2009)
		1	0.126	
maize	straw	0.25	0.560	Krippner *et al.* (2015)
		1	0.650	
maize	cobs	0.25	0.008	Stahl *et al.* (2009)
		1	0.004	
maize	cereal grains	0.25	LOD	Krippner *et al.* (2015)
		1	0.002	
oats	straw	0.25	0.880	Stahl *et al.* (2009)
		1	0.690	
	cereal grains	0.25	0.048	Stahl *et al.* (2009)
		1	0.054	
ryegrass	4 consecutive samples	0.25	0.128	Stahl *et al.* (2009)
		1	7.520	
wheat	straw	0.25	3.200	Stahl *et al.* (2009)
		1	1.900	
wheat	straw	0.0261	0.847	Wen et al., 2014
wheat	cereal grains	0.25	0.096	Stahl *et al.* (2009)
		1	0.009	
wheat	cereal grains	0.0261	0.111	Wen et al., 2014
wheat	Ear of corn	0.0261	0.160	

An increase in BCF_{PFOA} with higher soil concentrations (as recorded for PFOS) is not observed with the exception of ryegrass. Higher mobility and leaching from the soil of PFAO compared to PFOS offers a possible explanation. Preference is given to measurements of Wen et al. (2014) on soils treated with sludge mixtures (as spiking may lead to overestimation).

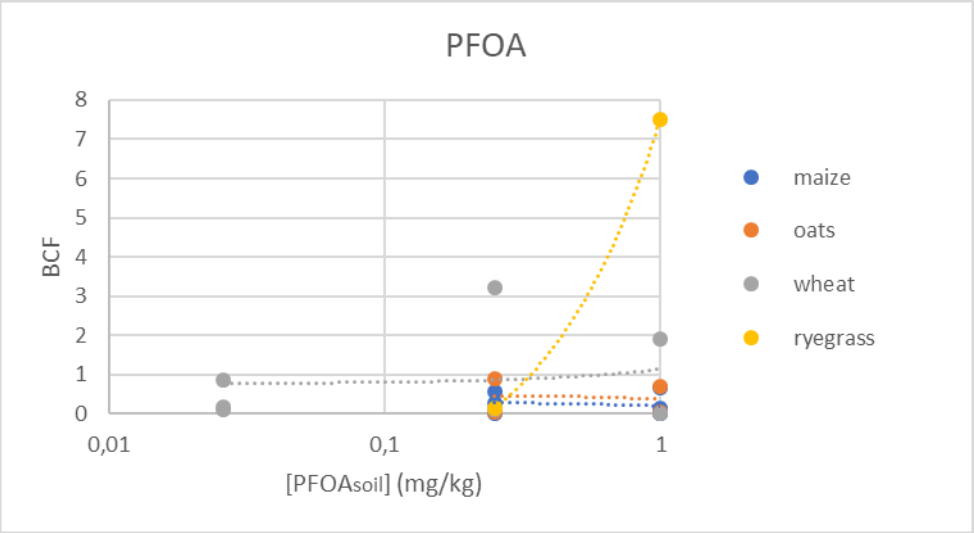


Figure 12: BCF_{PFOA} (mg/kg plant dm)/(mg/kg soil dm) for cereals and grasses as a function of the concentration of PFOA in soil (mg/kg dm) (Selection from Ghisi *et al.* (2019))

When we extend the analysis for plant uptake to vegetables, we find 4 additional studies in the review paper of Ghisi *et al.* (2019). Additionally, we retrieved one extra publication, i.e., Navarro *et al.* (2017), based on which BCF_{PFOA} can be calculated for spinach and tomato. The BCF derived by Lechner and Knapp (2011) are based on spiked soil. All other studies study soils enriched with PFAS-containing household or industrial sludge.

Table 52: BCF for PFOA for vegetables (Selection from Ghisi *et al.* (2019) supplemented with Navarro et al., 2017)

Crop		Concentration in soil (mg/kg)	BCF (mg/kg plant dm)/(mg/kg soil dm)	reference
Carrot	carrot (peeled)	0.681	0.49	Lechner and Knapp (2011)
	carrot (peeled)	0.676	0.49	
	carrot (peeled) Chantenay variety	0.528	0.28	Bizkarguenaga *et al.* (2016)
	carrot (peeled) Nantesa variety	0.485	0.3	
Celery	Celery shoots	0.07852	0.71	Blaine et al., 2014a
	Celery shoots	0.01491	0.13	
cucumber	pot 1	0.406	0.79	Lechner and Knapp (2011)
	pot 2	0.805	0.85	
lettuce	leaf	0.07852	2.52	Blaine et al., 2013
	leaf	0.01491	1.34	
	leaf	0.00517	LOD	
lettuce	leaf	0.56	1.85	Bizkarguenaga *et al.* (2016)
peas	fruits	0.07852	0.03	Blaine et al., 2014a
	fruits	0.01491	LOD	
spinach		2.73	0.87	Navarro *et al.* (2017)
			LOD	
potatoes	peeled	0.276	0.065	Lechner and Knapp (2011)
	peeled	0.795	0.045	
radish	carrot	0.07852	0.85	Blaine et al., 2014a
	carrot	0.01491	0.54	
tomato	fruits	0.07852	0.11	Blaine et al. (2013) and Blaine et al., 2014a
	fruits	0.01491	LOD	Blaine et al. (2013) and Blaine et al., 2014a
	fruits	0.00517	LOD	Blaine et al., 2013
		0.12	1.5	Navarro *et al.* (2017)

A closer look at Table 51 and Table 52 shows that the calculated BCF for vegetables is overall one to two orders of magnitude higher than that for cereals (Figure 13). We can also make a distinction between experiments carried out on soil enriched with PFOA spiked sludge (Lechner and Knapp (2011); Bizkarguenaga *et al.* (2016)) and soil to which PFOA contaminated industrial or household

sludge has been added (Blaine et al., 2013; 2014a; Navarro *et al.* (2017)). The highest concentrations of PFOA are found in lettuce, followed by tomato and spinach.

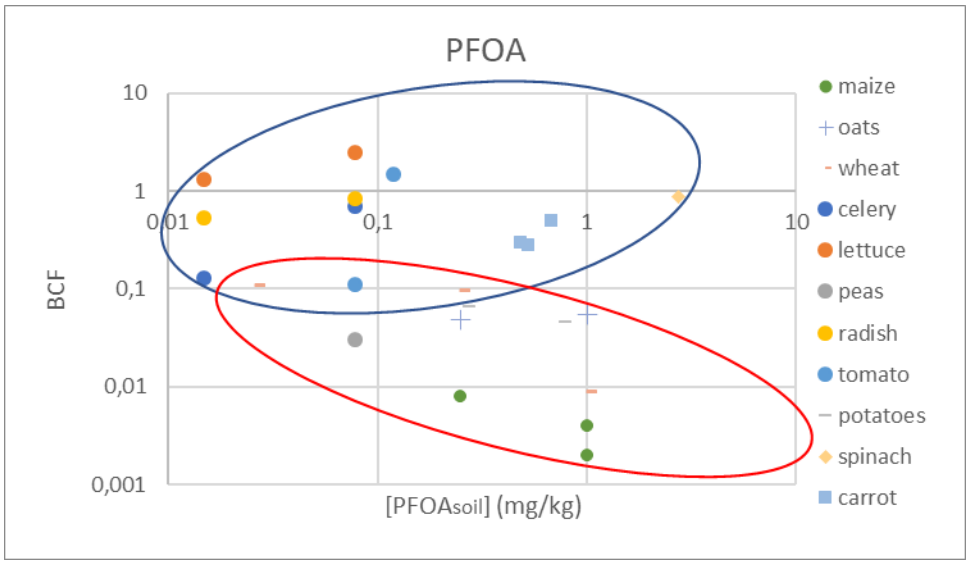


Figure 13: BCF_{PFOA} (mg/kg plant dm)/(mg/kg soil dm) in vegetables (circled blue) and cereal crops (circled red) as a function of the concentration of PFOA in the soil.

Blaine et al. (2013), 2014 perform measurements on soils enriched with sludge, contaminated with PFAS. The authors make a distinction between sludge of industrial and household origin. Based on the differences in concentrations of PFOA in the soil on the one hand and the different % OC between the two types of enriched soils, it appears that the calculated BCF for vegetables increases with the PFOA concentration in the soil and decreases the higher the % OC. The highest % OC are found in household enriched sludge (Blaine et al., 2013); 2014) resulting in a lower BCF.

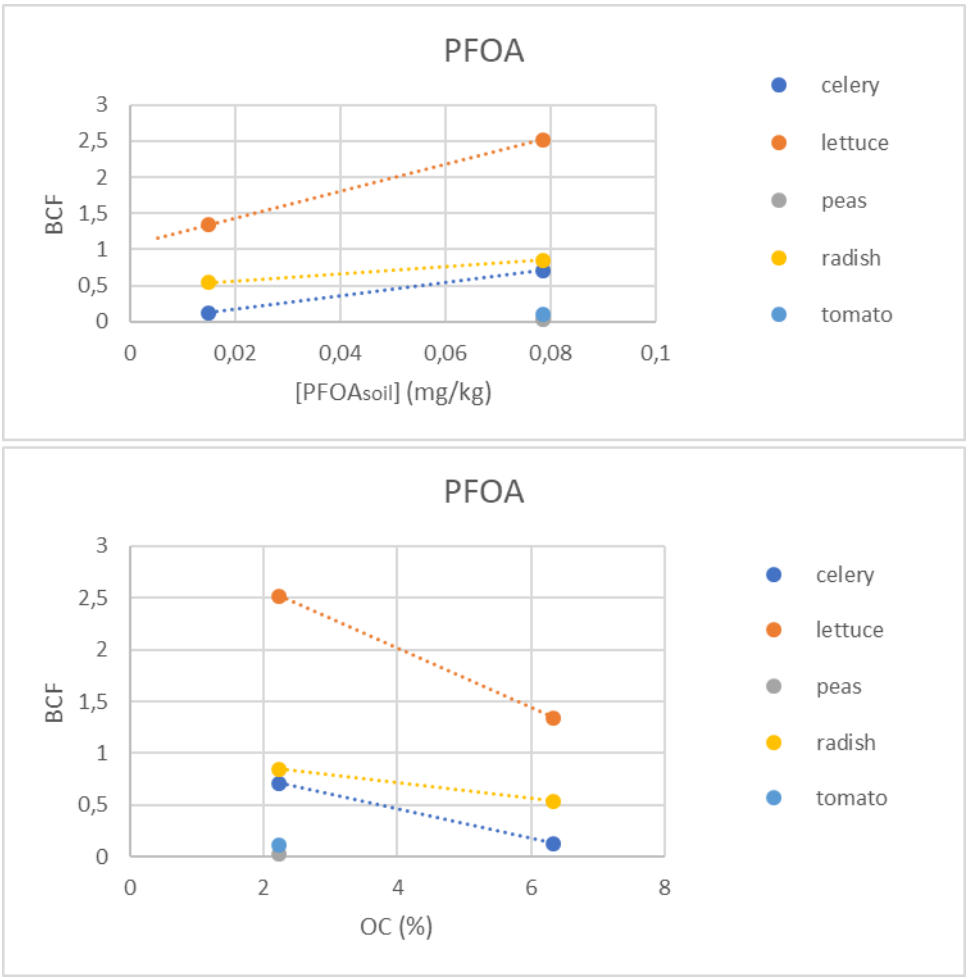


Figure 14: BCF_{PFOA} (mg/kg plant dm)/(mg/kg soil dm) as a function of the concentration of PFOA (mg/kg dm) in the soil and the % OC (based on data from Blaine et al. (2013), 2014)

We identified a second important study by Lijzen *et al.* (2018) on PFOA plant uptake. They rely on the same sources as Ghisi *et al.* (2019) with additional values for potatoes from Müller (2008) and pumpkins from Mohammadi *et al.* (2015). The values for potato are consistent with those reported by Lechner and Knapp (2011)) when converted to fresh weight basis. Pumpkins are not included in the food basket used by S-Risk. Lijzen *et al.* (2018) do not take any cereal crops into consideration. This is remedied in the most recent RIVM report by Wintersen et al. (2019). These authors retain the same BCF values derived in Lijzen et al. (2018) for vegetable garden crops. RIVM uses 3 BCF for their final calculations in C-Soil: a BCF for cereals (BCF = 0.063 (µg/kg plant **dm**)/(µg/kg soil dm)), and further a BCF for "potatoes" (BCF = 0.012 (µg/kg plant **fw**)/(µg/kg soil dm) and "other vegetables" (BCF = 0.012 (µg/kg plant **fw**)/(µg/kg soil dm)[37].

In order to allow for an additional comparison between the BCF for garden crops collected in this study and the BCF that were selected by Lijzen *et al.* (2018), the BCF values on a fresh weight basis from Lijzen *et al.* (2018) were converted to a dry weight basis. For this, we used the dry matter contents from the formularium of S-Risk (Cornelis *et al.* (2017)). These converted BCF are shown as a function of the concentration of PFOS in the soil in Figure 15.

[37] Note that the BCF for cereals is expressed on a dry matter basis where the BCF for vegetable crops is calculated on a fresh weight basis.

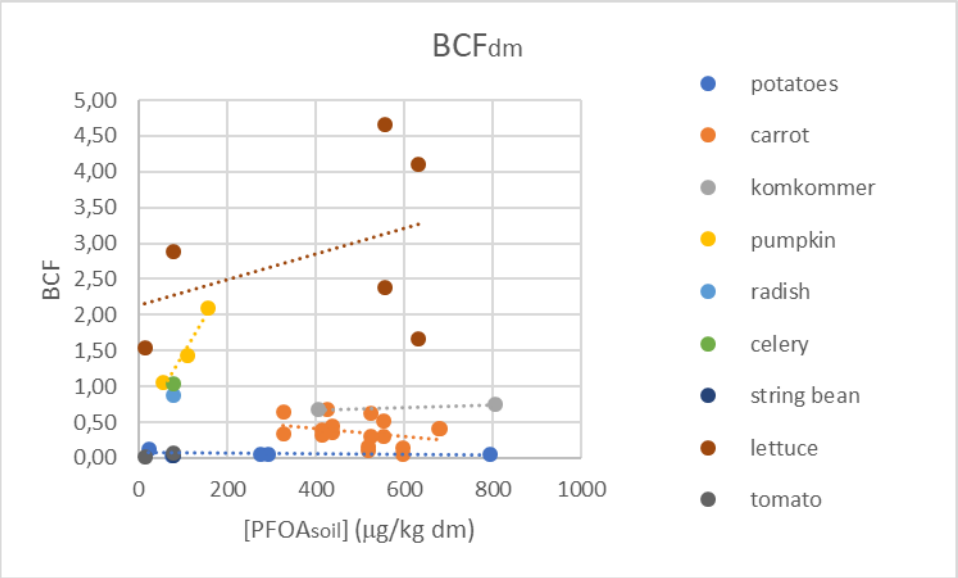


Figure 15: BCF_{PFOA} (mg/kg plant dm)/(mg/kg soil dm) from Lijzen *et al.* (2018) converted on a dry matter basis as a function of PFOA concentrations in soil

A comparison of both datasets is made in Figure 16. Despite the fact that the BCF values on a fresh weight basis from Lijzen *et al.* (2018) are converted to dry matter basis to allow for comparison, where we did not use the original dry matter levels in C-Soil, both datasets are comparable. The largest discrepancy (mg/kg plant dm)/(mg/kg soil dm) is found for celery, lettuce and tomato. For celery, Lijzen *et al.* (2018) rely on the BCF from Blaine *et al.* (2013) 2014) contaminated with industrial bio-sludge with an organic carbon content of 6.34%. The lower BCF values based on soil enriched with household bio-sludge (% OC = 2.24) were not retained. For lettuce, a subset of the data from Blaine *et al.* (2013) by Lijzen *et al.* (2018) is not taken into account, due to the very low levels of PFOA in the soil. For tomato, the BCF derived by Navarro *et al.* (2017) is not included in the study by Lijzen *et al.* (2018). This also applies to spinach, for which no value is indicated in Lijzen *et al.* (2018). For string beans, a BCF value is only available at Lijzen et al. (2018).

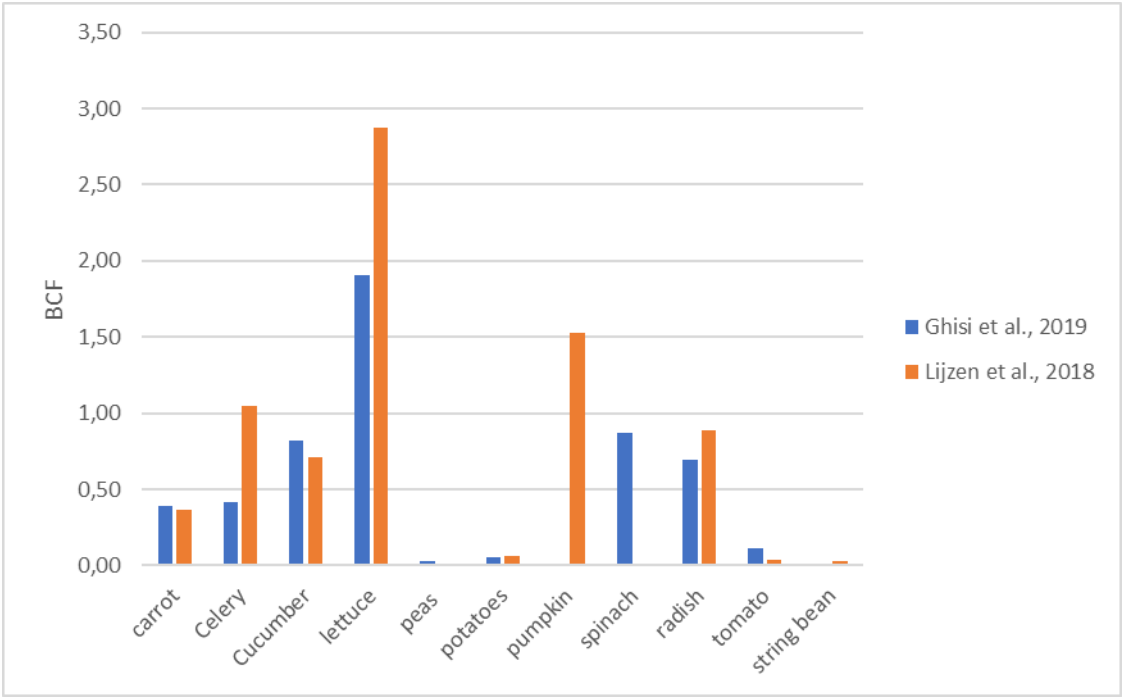


Figure 16: Comparison between BCF_{PFOA} (mg/kg plant dm)/(mg/kg soil dm) derived by Ghisi *et al.* (2019) and the BCF_{PFOA} (mg/kg plant dm)/(mg/kg soil dm) from Lijzen *et al.* (2018) converted on a dry matter basis. For peas and spinach only one value was derived in this study. A value for pumpkin is only available at Lijzen et al. (2018)

For the final selection we rely on Ghisi *et al.* (2019) supplemented by data for cereals and grasses from Table 51 and supplemented with the data from Navarro *et al.* (2017). For ryegrass, only the BCF calculated at 0.25 mg/kg dm in the soil is retained (Stahl *et al.* (2009)).

Table 53: Selected BCF_{PFOA} (mg/kg plant dm)/(mg/kg soil dm) for vegetables and cereals for calculating the soil remediation value for PFOA in this study

Crop	PFOS	method of derivation
carrot	0.39	average value Lechner and Knapp (2011) and Bizkarguenaga et al. (2016)
Celery	0.42	average values Blaine et al., 2014a
Cucumber	0.82	average values from Lechner et al., 2011
lettuce	1.90	average value Blaine et al., 2013 and Bizkarguenaga et al. (2016)
peas	0.03	single value Blaine et al., 2014
potatoes	0.06	average values Lechner et al., 2011
spinach	0.87	single value Navarro et al. (2017)
radish	0.70	average values Blaine et al., 2014a
tomato	0.81	single value Blaine et al., 2013 and Blaine et al., 2014
maize (cob)	0.005	single value Stahl et al. (2009)
oats (cereal)	0.051	average value Stahl et al. (2009)
ryegrass	0.128	average value Stahl et al. (2009)
wheat (cereal)	0.072	single value Wen et al., 2014

For the calculations in S-Risk, a food basket has been defined consisting of vegetables that are commonly consumed. If no BCF values are found in the literature for one or more vegetables from this food basket, equivalence rules as defined in Bierkens et al. (2016) are used. This means that for the crop with missing BCF values from a certain crop group (tuber vegetables, bulbous vegetables, leafy vegetables, etc.) a BCF is applied from a related crop for which a BCF is available from the same crop group (Table 54).

Table 54: Calculated (bold) and estimated BCFs for PFOA for the different crops in the S-Risk diet, with indication of the equivalence rules used

Plant	BCF or BCF model
potatoes	
potatoes	**0.06**
root and tuber vegetables	
carrots	**0.39**
salsify	0.55 (average value of known root and tuber vegetables)
other root vegetables (such as radish)	**0.70**
bulbous vegetables	0.55 (= average known root and tuber vegetables)
bulbous vegetables (such as onion)	0.55 (= average known root and tuber vegetables)
leek	0.55 (= average known root and tuber vegetables)
fruiting vegetables	
tomato	**0.81**
cucumber	**0.82**
other fruiting vegetables (such as peppers)	0.81 (=tomato)
Cabbages	
Cabbage	0.55 (= average known root and tuber vegetables)
cauliflower and broccoli	0.55 (= average known root and tuber vegetables)
sprouts	0.55 (= average known root and tuber vegetables)
leafy vegetables	
Lettuce	**1.90**
lamb's lettuce	1.90 (=lettuce)
endive	1.06 (= average of all known leafy vegetables)
spinach	**0.87**
chicory	1.06 (= average of all known leafy vegetables)
celery	**0.42**
legumes	
beans	0.03 (= peas)
peas	**0.03**
Grasses	
Grass	**0.128**
Cereals	
Maize	**0.005**

3.6. TRANSFER TO ANIMAL PRODUCTS

Bioaccumulation of PFAS cannot be simulated on the basis of equilibrium partitioning as is the case for most neutral hydrophobic organic compounds that accumulate primarily in fat tissue. Due to their amphiphilic and anionic character, they are mainly distributed over the serum, liver and kidneys and their toxicokinetics are largely controlled by urinary excretion. The equations derived by Travis and Arms (1988) using S-Risk to estimate concentrations in meat and milk based on K_{ow} partition distribution coefficients are not applicable, so we have to start from empirically derived biotransfer factors from case studies. A literature review provides 4 papers with paired measurement data for PFOA in feed and drinking water together with PFOS concentrations in bovine tissues/organs (n =3) and/or sheep (n=1) suitable for human consumption, from which biotransfer factors (BTF) can be derived.

Bovines

We find the most detailed study in Vestergren *et al.* (2013). Vestergren *et al.* (2013) derived BTF for dairy cows from agricultural areas without external influence from known PFCAs or PFSAs point sources. The feed consisted of a mixture of silage, maize and barley. Well water from local origin was used as drinking water. At the time of the measurements the adult (> 24 months old) dairy cows had had sufficient time to achieve equilibrium between intake and excretion of PFAS. The measurement results for PFOA in feed and drinking water, as well as in the different tissues and animal matrices relevant for the calculations, are summarised in Table 55. In bovines, the highest concentrations of PFOA are found in the liver and blood. The measured values in muscle and milk are 7.0 ± 4.3 ng/kg fw and 6.7 ± 1.8 ng/kg fw respectively. Based on a total daily intake of 294.6 ng/d PFOA, the authors calculate a BTF_{muscle} = 0.012 $mg.kg^{-1}$ fw/$mg.d^{-1}$ and BTF_{milk} = 0.011 $mg.kg^{-1}$ fw/$mg.d^{-1}$ (Table 56). The BTF_{liver}, a measure of the accumulation of PFOA in offal, is 0.015 $mg.kg^{-1}$ fw/$mg.d^{-1}$ (own calculation; Table 56).

Table 55: Concentrations of PFOA (arithmetic mean and SD) in feed and tissues of dairy cows (Vestergren et al., 2013)

	number of samples	PFOA
intake media		
water	6	0.23 ± 0.03 (ng/l)
ensiled fodder	6	13.0 ± 4.4 (ng/kg)
barley	6	8.3 ± 2.8 (ng/kg)
tissue (bovine)		
liver	5	9.0 ± 2.0 (ng/kg)
blood	5	16.0 ± 4.1 (ng/kg)
muscle	5	7.0 ± 4.3 (ng/kg)
excretion media		
urine	10	11.0 ± 0.6 (ng/l)
faeces	10	<LOD (ng/kg)
milk	6	6.7 ± 1.8 (ng/l)

Kowalczyk *et al.* (2013) carried out a controlled feeding study on 6 dairy cattle divided into two groups, the first group being slaughtered after 29 days of exposure to PFAA contaminated feed, while the second group was put on a non-contaminated control diet for an additional 21 days. The feed consisted of hay and silage from farmland enriched with PFAA by the use of fertilisers and herbicides. The daily intake of PFOA (hay and silage grass) is estimated at 4472.04 µg/d. The concentrations after 29 days in milk, muscle, liver and kidney were respectively 0.07 µg/l, 0.6 µg/kg fw, 10.1 µg/kg fw and 8.7 µg/kg fw. After stopping the intake of contaminated feed, these values fell drastically. For milk and muscle, the measured values fall below the limit of detection, while the concentrations in the liver and kidney decline to respectively 0.8 µg/l and 0.4 µg/kg fw. The BTF for both periods which we calculate on the basis of these concentrations in feed and milk or tissues are shown in Figure 17. The decrease in concentrations in animal products after the intake of contaminated feed was stopped indicates that the equilibrium of PFOA in cattle was already reached after 29 days and/or that PFOA practically does not accumulate in the samples examined, as such we only take into account the values obtained after 29 days, see Table 56.

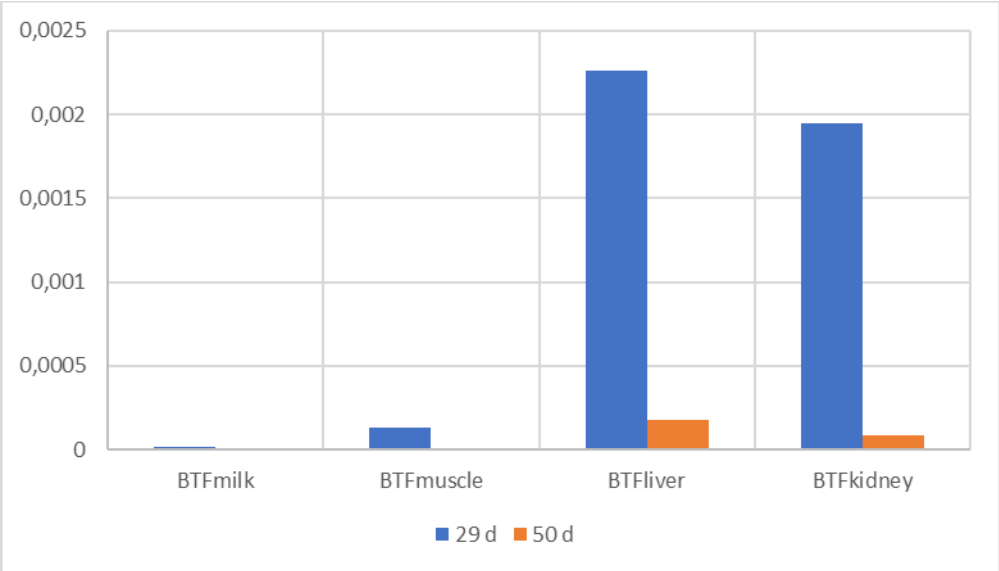


Figure 17: BTF in milk, muscle, liver and kidney in dairy cattle after 29 and 50 days (feeding with contaminated feed stops after 29 days), based on data from Kowalczyk *et al.* (2013).

Table 56: BTF values ($mg.kg^{-1}$ fw/$mg.d^{-1}$) for bovines and sheep for PFOA derived from available literature data.

	Vestergren et al., 2013 (1)	Kowalczyk et al 2013 (1)	Kowalczyk et al 2012 (2)
BTF_{milk}	0.011	$1.6x10^{-5}$	0.014
BTF_{muscle}	0.012	$13x10^{-5}$	0.007
BTF_{liver}	0.015	$2.26x10^{-3}$	0.079
BTF_{kidney}		$1.94x10^{-3}$	0.145

(1) Bovines; (2) sheep

Sheep

Kowalczyk *et al.* (2012) describe a 21-day controlled feed study on sheep (n = 3 including 1 control). The animals were fed PFAA-contaminated silage (maize). The intake of PFOA was estimated at 21.29 and 33.10 μg/d for sheep 1 and sheep 2 respectively. The average concentration in milk during the 21 days was 0.2 and 0.7 mg/l for sheep 1 and 2 respectively. On the basis of these data, we calculate an average BTF_{milk} for both sheep of -0.014 $mg.kg^{-1}$ $fw/mg.d^{-1}$. This value, together with the BTF values for muscle, liver and kidney, has been included in Table 56 (for these matrices, only data are available for 1 sheep, there is therefore significant uncertainty). It was shown in the study by Kowalczyk *et al.* (2012) that there is a significant correlation between PFOA concentrations in blood plasma and milk (r^2 = 0.95). The study once again shows the strong affinity for PFOA for plasma proteins, whereby their transfer to milk can be considered negligible. This is a pilot study in which two animals are tested, where ultimately one animal will have significant values available for all organs.

Chickens

In connection with the transfer of PFOA from feed to chickens and more specifically to eggs, our search resulted in 3 studies, i.e. Yoo et al. (2009), Yeung et al. (2009) and Hanell (2015). Only the latter publication reports paired data in chicken feed and eggs for different PFAS, but does not indicate any usable data for PFOA, which makes it impossible to calculate BTF_{egg}.

Conclusion

In contrast to PFOS, the BTF values from Kowalczyk *et al.* (2013) for PFOA are retained in order to determine the final BTF values together with the data from Vestergren *et al.* (2013). The reason for this is that the dates indicate that an equilibrium was reached on day 29. Although the concentrations of PFOA in the feed are much higher than the background values, they are representative for contaminated sites and therefore relevant as a worst case approach. Based on the publications by Vestergren *et al.* (2013) and Kowalczyk *et al.* (2013) on day 29, average values are calculated for the BTF_{milk} of (0.006 ± 0.008) $mg.kg^{-1}$ $fw/mg.d^{-1}$. The average value for the BTF_{muscle} and BTF_{liver} from Vestergren *et al.* (2013) and Kowalczyk *et al.* (2013) are respectively 0.006 ± 0.008) $mg.kg^{-1}$ $fw/mg.d^{-1}$ and (0.009 ± 0.009) $mg.kg^{-1}$ $fw/mg.d^{-1}$. For kidneys, we only find in Kowalczyk *et al.* (2013) a BTF value of -0.002 $mg.kg^{-1}$ $fw/mg.d^{-1}$. All these values for bovines are summarised in Table 57.

For comparison, the most recent RIVM report (Wintersen et al., 2019) also uses the BTF values from Vestergren et al. (2013). The study by Kowalczyk et al. (2013) was taken into consideration but was not retained, as the values derived from it are based on studies in which significantly higher concentrations of PFAS were administered.

Table 57: Average (± SD) BTF values (mg.kg^{-1} fw/mg.d^{-1}) for bovines from Vestergren, 2013 and Kowalczyk *et al.* (2013)

	average value	SD
BTF_{milk}	0.006	0.008
BTF_{muscle}	0.006	0.008
BTF_{liver}	0.009	0.009
BTF_{kidney}	0.002	

For sheep we use for milk and muscle BTF values of respectively BTF_{milk} 0.014 mg.kg^{-1} fw/mg.d^{-1} and BTF_{muscle} 0.007 mg.kg^{-1} fw/mg.d^{-1} (Kowalczyk et al. , 2012) lThe values for liver and kidney are respectively BTF_{liver} 0.079 mg.kg^{-1} fw/mg.d^{-1} and BTF_{kidney} 0.145 mg.kg^{-1} fw/mg.d^{-1}

3.7. TOXICOLOGY

3.7.1. INTRODUCTION

The overview of the toxicology of PFOA is mainly based on the review reports of CONCAWE (2016); OVAM (2018), EFSA (2008b); ECHA (2015), EFSA (2018c), ATSDR draft (2018), US-EPA (2016d), FSANZ (2016), Lijzen *et al.* (2018) DEPA (2015) and Pancras *et al.* (2018). The toxicokinetics and toxicology of PFOA are first discussed. A summary of the available toxicological reference values is given in section 3.7.4. A proposal for the toxicological reference values to be used for deriving soil remediation values is set out in section 3.7.5.
Based on the physicochemical properties of PFAS, exposure via intake of food and drinking water is highly likely. PFAS is also measured in air and dust, meaning that inhaled air, dust ingestion or dermal contact with dust or aerosols may also be possible routes of exposure.

3.7.2. TOXICOKINETICS

→ Absorption after oral intake

PFOA is easily absorbed after oral intake (PHE, 2009). After administration of a single oral dose to rats, absorption was 93% after 24 hours. Peak concentrations in the blood were reached at 1-2 hours after administration. In male rats, PFOA was mainly found in the liver and plasma, while in females, PFOA was mainly found in the plasma and kidneys (van den Heuvel *et al.*, 1991). More than 95% of an oral dose of ammonium PFOA was absorbed by rats after a single dose of tube feeding, ranging from 0.1 to 25 mg/kg. The highest plasma concentration in male rats was reached after about 10 hours and the half-life for clearance from the plasma was about 170 hours in these animals (Kemper, 2003).
After intravenous administration of PFOA to male rats of 0.041 and 16.56 mg/kg bw, proportionally more of the low dose was found in the liver (52%) and of the higher dose more in serum, other tissues and the carcass, and less in the liver (27%) (Kudo *et al.*, 2007).
For deriving soil remediation values, the oral absorption factor is equated by default to 1 (Cornelis *et al.*, 2012).

→ **Absorption after inhalation**

Human studies specifically for PFOA are not available. In rats, 1-25 mg ammonium PFOA PFOA was found in the plasma 30 minutes after the onset of exposure via aerosol (nose only). Plasma concentrations continued to increase during the 6-hour exposure; the highest level was reached after 9 hours (3 hours after administration was stopped) in male rats and after 7 hours (1 hour after stopping) in female rats (Hinderliter *et al.*, 2006). This corresponds to a half-life for absorption of about 1.3 hours. The fact that the highest concentration occurs faster in females appears to be related to the faster clearance of absorbed PFOA compared to rats (see below: clearance) (ATSDR draft, 2018). Exposure of rats to dust (nose only) of ammonium PFOA induced significant increases in absolute and relative hepatic weight, which is an indirect indication of absorption of the substance by inhalation (Kinney *et al.*, 1989).
Inhalation of soil particles is insignificant (Xiao *et al.*, 2015).
In accordance with Cornelis *et al.* (2012) we assume that absorption by inhalation and by the oral route is the same for both routes, i.e. 95%.

→ **Absorption after dermal contact**

In vitro dermal penetration of PFOA was investigated with isolated epidermis from humans, rats and mice. This showed that the skin of rats and mice is more permeable than that of humans (Fasano *et al.*, 2005; Franko *et al.*, 2012). Following *in vitro* administration of an aqueous solution of the ammonium salt of PFOA to rat and human skin, approximately 0.048% of the dose was absorbed through human skin and 1.44% through rat skin, in 40 hours.
Approximately 24% of a dermal dose of PFOA (0.5 mg in 1% acetone) was absorbed *in vitro* by human skin and 45% of the dose was retained in the skin. (Franko *et al.*, 2012); the authors note, however, that the acetone and glycerol used in the pre-treatment of the skin may have facilitated absorption. The permeability of the acid appears to be higher than that of the anion.
Dermal absorption from soil particles is insignificant compared to exposure via oral intake of soil and dust (Cornelis *et al.*, 2012; Xiao *et al.*, 2015). For deriving soil remediation values **the dermal absorption factor from soil and dust is therefore set to 0**.
Dermal absorption from water is driven by the permeability coefficient (Kp, expressed in cm/h), indicating a measured mean Kp of $9.5.10^{-7}$ cm/h of PFOA; absorption through the skin may therefore be considered low.
The average *in vitro* measured Kp of the anion of PFOA (ammonium perfluorooctanoate, AFPO) is $9.49.10^{-7}$ cm/h for the human and $3.25.10^{-5}$ cm/h for the rat humans (Fasano *et al.*, 2005; Franko *et al.*, 2012). The calculated Kp is 0.114 cm/h (EpiSuite DermWin). The Kp for organic substances is calculated in Dermwin from the log K_{ow}; for PFOA a log K_{ow} of 6,30 was used for the calculation.
We propose using the measured **Kp of $9.49.10^{-7}$ cm/h** for dermal **absorption via water** in the soil remediation value calculation

→ **Distribution**

After absorption, PFOA primarily ends up in the liver and the serum (PHE, 2009). In a study with workers from the perfluoroalkyl industry, the serum:plasma ratio was 1:1, regardless of the concentration(Ehresman *et al.*, 2007). Distribution from plasma to tissues is mainly to the liver. Liver,

blood and kidneys accounted for 22, 22 and 2% of an oral dose of 1 mg/kg in male rats, and 6, 7 and 3% in female rats (faster excretion in females) (Kemper, 2003).
A Japanese cohort study investigated the influence of age and sex on PFOA levels in blood and urinary excretion. In the sub-cohort of 20-50 year olds, blood concentrations were higher in males than in females, while in the age group > 50 years the average concentrations in males and females were not different (Harada *et al.*, 2004). The interpretation of these data is limited due to the small size of the cohort. Renal clearance was negligible in both sexes, much lower than in rats and monkeys.
Unlike most other persistent organic pollutants (POPs), PFOS has a low affinity for fats. ^{14}C-PFOA, administered as a single oral dose was rapidly absorbed. After 24 hours, the absorption of radioactivity was 93%. Peak levels in blood were reached 1 to 2 hours after administration. Analysis of 14C in the tissues showed that the liver and plasma of male rats and the liver, kidneys and plasma of female rats were the primary tissues for distribution (van den Heuvel *et al.*, 1991). Han *et al.* (2003) estimate that more than 90% of PFOA appears to bind to serum albumin in the blood of both rats and humans. An important factor that plays a role in the distribution is therefore the binding to proteins, including the binding to the fatty acid binding protein (L-FABP) in the liver of rats (Luebker *et al.*, 2002).
Hinderliter *et al.* (2005) have demonstrated that after oral administration of the ammonium salt of PFOA to rats, PFOA passed from the mother to the foetus via the placenta and to the offspring via lactation. The concentrations in the foetal plasma were half the steady-state concentrations in the maternal plasma, while steady-state concentrations in milk were about one-tenth lower than those in the maternal plasma. The information on PFOA transfer through the human placenta is limited. Fei *et al.* (2007) compared PFOA concentrations in maternal blood from week 4-14 and later during pregnancy with umbilical cord blood. The ratio fell from 1.83 to 1,46.
In a development study with mice, serum concentrations of PFOA were measured in female mice at weaning on postnatal day 22. A 4x higher dose was found in females without pups (10400 ng/ml) than in those with pups, indicating that there was an extensive transfer to the mother's milk (Abbott *et al.*, 2007).

→ **Metabolisation**

PFOA does not undergo any significant metabolisation and therefore accumulates in the body (Stahl *et al.*, 2011).

→ **Clearance**

PFOS is only eliminated very slowly from the human body; the half-life measured in serum in the general public in England is 4 years (PHE, 2009), in the USA it is 2.9 to 8.5 years (Seals *et al.*, 2011) and 2.3 years (Bartell *et al.*, 2010), and among retired workers from the fluorine chemical industry in the USA it is 2.3 to 3.8 years (Olsen *et al.*, 2007a). Clearance via the kidneys is almost negligible in humans (PHE, 2009). Half-life periods for animals are 33 and 21 days for male and female Java monkeys (Macaca fascicularis) respectively (Butenhoff *et al.*, 2004) and 5.63 and 0.08 days respectively for male and female rats (Ohmori *et al.*, 2003). Clearance appears to vary with the type of organism and sex (CONCAWE, 2016). In female rats, 91% of the dose was cleared via the urine within 24 hours of administration. During the same period, males cleared only 6% of PFOA via urine; faecal clearance is the most important clearance route for male rats. This sex-related difference in clearance via urine resulted in a half-life for total clearance of 15 days for males and less than 1 day for females (van den Heuvel *et al.*, 1991). This sex-related difference in clearance is due to an active excretion mechanism for organic acids in the female rat (Hanhijarvi *et al.*, 1982), while testosterone appears to suppress clearance via the kidneys in the rat (van den Heuvel *et al.*, 1992). The faster

clearance of PFOA by female rats is not due to the formation of a PFOA metabolite (van den Heuvel *et al.*, 1992).
In humans, there appeared to be no significant difference in clearance between men and women. Active transport plays an important role in clearance (EFSA, 2008b)

3.7.3. EFFECTS ON TEST ANIMALS AND HUMANS

An overview of the studies discussed below is given in Table 58.

→ Acute toxicity

PFOA is classified as hazardous according to the CLP criteria (EC, 2008); PFOA is harmful following acute exposure via inhalation or oral ingestion (referred to as Acute Tox. 4), and causes serious eye damage (Eye Dam. 1).
The LC50, rat after inhalation for 4 hours of the ammonium salt of PFOA is 980 mg/m³. This concentration caused an increase in size of the liver and corneal turbidity. These effects diminished with time after exposure. Repeated subacute administration for 10 days (6h/d, 5d/week at 0, 1, 8 or 84 mg/m³) suppressed the increase in body weight at 84 mg/m³ and increased the weight of the liver at 8 mg/m³. The NOEL was 1 mg/m³; at this concentration the average blood concentration was 13 ppm on the tenth day (Kennedy *et al.*, 1986). The oral LD50 fluctuates around 500 mg/kg bw/d. The dermal LD50 for rabbits is higher than 2000 mg/kg bw/d and the substance is mildly irritant to the skin (EFSA, 2008b).

→ Subacute and (sub)chronic toxicity

Subacute and sub-chronic studies are available for rodents and monkeys.
The weight of the liver and peroxisomal beta-oxidation[38] in the liver appear to be the most pronounced targets when laboratory animals are exposed to PFOA. In an oral 90-d study, male rats were given 0.6, 1.7, 5.6, 18 and 64 mg PFOA/kg bw per day via diet and female rats 0.7, 2.3, 7.7, 22.4 and 76 mg PFOA/kg bw per day. The absolute and relative weight of the liver was increased in the males with the two highest doses and in the females with the highest dose. Increases in absolute liver weight and hepatocellular necrosis were observed at 1.7 mg/kg bw/d in the males. On the basis of liver effects, the NOAEL was 0.6 mg/kg bw/d for males and 22 mg/kg bw/d for females (Goldenthal *et al.*, 1978).
In another oral 90-d study, male rats were given 0, 0.06, 0.64, 1.94 and 6.4 mg PFOA/kg bw per day via diet. The animals in the group with the highest dose showed a lower body weight. The dose group of 0.64 mg/kg bw/d and above showed increased palmitoyl CoA oxidase activity in the liver, which is a marker for peroxisome proliferation, and increased relative liver weight. Histopathological changes in the liver were hepatocellular hypertrophy and necrosis of liver cells (Perkins *et al.*, 2004).
In a sub-chronic study with Java monkeys, the animals were given a daily dose of 0, 3, 10 or 30 mg/kg PFOA for 6 months. All groups showed a dose-dependent increase in liver weight associated with mitochondrial proliferation. There was no histopathological evidence of liver damage in dose groups 10 and 30 mg/kg bw/day. Two animals died before the end of the study: one in the 3mg/kg group and one in the 30 mg/kg group (Butenhoff *et al.*, 2002).

[38] Peroxisomes are small organelles that break down fatty acids to acetyl CoA (= beta-oxidation).

→ **Reproduction and development**

A lot of information is available on the effect of exposure to PFOA on reproduction and development, and other effects, both in experimental animals and humans. An overview of No (Lowest)-Effect-Concentrations (N(L)OEC) from several (sub)chronic animal studies is given in Table 58.
The most sensitive endpoint (birth weight) identified in developmental and reproductive toxicity studies comes from a study in mice by Abbott *et al.* (2007). In this study, dams were exposed to the ammonium salt of PFOA in doses of 0.1, 0.3, 0.6, 1, 3, 5, 10 and 20 mg/kg bw/d for days 1-17 of pregnancy. A decreasing neonatal survival rate was observed at doses ≥0.6 mg/kg bw/d resulting in a NOAEL of 0.3 mg/kg bw/d (EFSA, 2018c).
Lau *et al.* (2006) observed an increase in the incidence of the loss of the complete litter (and some additional neonatal mortality) from a dose of 5 mg/kg/d administered throughout pregnancy. Birthweight was only affected at a dose ≥20 mg/kg/d, but a lower growth rate of the young of 25-30% during postnatal days 13-23 was observed at a dose of 3 mg/kg/d or higher, resulting in a NOAEL of 1 mg/kg/d and a calculated benchmark dose lower limit (BMDL5) of 0.86 mg/kg/d (for reduced growth of the young). However, the weight did normalise as they reached adulthood. Based on Figure 3 in the paper by Lau, ECHA (2015) estimates that the serum concentration of PFOA was roughly 20000 ng/ml in the dams exposed to 1 mg/kg/d on the 18th day of pregnancy. The NOAEL is therefore approximately 20000 ng/ml (BMDL5 0;86 mg/kg/d according to Borg and Håkansson (2012)).
PFOA has a harmonised CLP classification as 'May damage to the unborn child' (Rep. 1B) and 'May cause harm to breast-fed children' (EC, 2008).

→ **Carcinogenicity**

PFOA has a harmonised CLP classification as 'Suspected of causing cancer' (Carc. Category 2) (EC, 2008). US-EPA (2016d) concludes that there is suggestive evidence for carcinogenicity of PFOA based on epidemiological studies showing an association between PFOA in serum and renal and testicular tumours in highly exposed members of the general population. The International Agency for Research on Cancer (IARC) has classified PFOA as a potential human carcinogen (Group 2B) based on the limited evidence that human exposure to PFOA is associated with testicular and renal cancer and on the limited evidence in animal studies (IARC, 2017)(Abbott, 2007).

→ **Genotoxic effects**

The CONTAM panel found no evidence of a direct genotoxic mechanism of action *in vitro* or *in vivo*. This seems to indicate an indirect (non-genotoxic) mechanism for carcinogenicity.

→ **Neurotoxicity**

Like PFOS, PFOA exhibits developmental neurotoxicity in rodents and causes widespread effects on the expression of genes encoding proteins relevant to signal transmission in the brain. Male offspring appear to be more sensitive than female offspring. The most common impact on behaviour is increased spontaneous movement activity, in contrast to PFOS where a decrease is observed. Onishchenko *et al.* (2011) conducted a comparative study between PFOS and PFOA at an oral dose of 0.3 mg/kg bw/d with mice. Pregnant mice were exposed through diet throughout pregnancy. Only male offspring showed altered movement activity, increased after exposure to PFOA, decreased after exposure to PFOS. Sobolewski *et al.* (2014) exposed mice to 0.1 mg PFOA/kg bw/d, alone or in

combination with mixtures of endocrine disruptors, from the 7th day of pregnancy until 21 days after childbirth. The offspring were tested for cognitive functions and movement on the 60th day after birth. The males exposed to PFOA alone showed increased horizontal movement, lower resting time and reduced habituation to repeat testing. In a recognition test of new objects (which tests memory), both male and female mice exposed to PFOA showed a lower exploration during the initial phase; the males generally had a lower exploration time (EFSA, 2018c).

→ **Immunotoxicity**

PFOA has an effect on the immune system. In a chronic dietary study, male rats and mice were exposed for 29 days daily to doses of 0, 0,3, 1, 10, and 30 mg NH^{4+}PFOA/kg bw/d. In addition, animals in the group with the highest dose were given intravenous administration of SRBCs[39] or pure water for 5 days from day 23 or 24 to measure the immunosuppressive potential of PFOA (Loveless *et al.*, 2008). In mice, 10 and 30 mg PFOA/kg bw/d caused a decrease in body weight, a tripling of liver weight, a 2.3-fold increase in serum corticosterone, a moderate reduction in triglycerides, an increase in neutrophils and monocytes in the blood and a decrease in lymphocytes in peripheral blood (only at 30 mg/kg). Immunological effects include reduced IgM response to SRBCs, lower spleen and thyroid weight and cell count. In the rats, the dose of 10 mg/kg or higher caused a decrease in body weight and an increase in relative liver weight and moderate hepatocellular hypertrophy, with indications of hepatocellular necrosis. Unlike the mice, no SRBC-IgM response was reported for the rats.
In other mouse studies, a dietary study by Vetvicka and Vetvickova (2013) and a drinking water study by Son et al. (2009), an impact on the immune system was also observed EFSA (2018c). The CONTAM panel of EFSA therefore concludes that there is evidence that exposure to PFOA has an effect on the immune system *in vivo*. These effects were manifested in the cellular composition of bone marrow, spleen and thyroid and in functional effects including reduced antibody response to T-cell dependent antigens and increased specific IgE antibody response and inflammatory response. These data seem to indicate a deregulation of the immune system. Effects were mostly observed at doses that also induced general toxic effects such as on food intake and body weight. However, effects on the immune system seemed to last longer than effects on body weight and liver, when treatment was stopped. Based on the 28-day study by Loveless *et al.* (2008) the NOAEL for systemic toxicity was 0.3 mg/kg bw/d, based on hepatocellular necrosis at 1 mg/kg bw/d. The NOAEL for immunotoxicity was 1mg/kg bw/d, based on suppression of the anti-SRBC IgM response (EFSA, 2018c).

[39] sheep red blood cells

Table 58: (Sub)acute and (sub)chronic animal studies with PFOA (sources:EFSA (2018c) CONCAWE (2016), EFSA (2008b), US-EPA (2016d)))

Animal	Administration	Duration of exposure	Parameter	Value	Effects	Reference
Rat	Inhalation	4 hours	LC50	980 mg/m³ (NH4-salt)	Lethality	Kennedy *et al.* (1986)
Rat	Inhalation	10 days	NOEL	1 mg/m³ (13 mg/l in blood)	Increased liver weight at 8 mg/m³ and lower weight gain at 84 mg/m³	Kennedy *et al.* (1986)
Rabbit	Dermal	acute	LD50	>2000 mg/kg bw/d	Lethality	Glaza (1995)
Rat	Oral	acute	LD50	680/430 (male/female) mg/kg bw/d	Lethality	Dean and Jessup (1978) reviewed by Griffith and Long (1980)
Rat	Food	90 days	NOAEL	0.6/22 mg/kg bw/d (male/female)	Increased liver weight, hepatocellular necrosis	Goldenthal *et al.* (1978)
Rat	Food	90 days	-	0, 0.06, 0.64, 1.94 and 6.4 mg/kg bw/d	Liver effects from 0.64 mg/kg bw/d; lower body weight at highest dose	Perkins *et al.* (2004)
Monkeys	Food	6 months	-	0, 3, 10 and 30 mg/kg bw/d	Dose-dependent increase in liver weight	Butenhoff *et al.* (2002)
Mouse	Food	day 1 to 17 of the pregnancy	NOAEL	0.3 mg/kg bw/d (NH4-salt)	Decreasing neonatal survival rate at ≥0.6 mg/kg bw/d	Abbott *et al.* (2007)
Mouse	Food	Full pregnancy	NOAEL	1 mg/kg bw/d (20 mg/l in blood)	Decreased growth rate of the young at 3 mg/kg/d; increase loss of full litter at 5 mg/kg/d; Lower birth weight at 20 mg/kg/d	Lau *et al.* (2006); ECHA (2015)
Mouse	Food	Full pregnancy	-	0.3 mg/kg bw/d	Increased movement activity (males)	Onishchenko *et al.* (2011)

Mouse	Food	from day 7 of pregnancy until 21 days after birth	-	0.1 mg/kg bw/d	Effect on cognitive functions and movement	Sobolewski *et al.* (2014)
Rat	Tube feeding	29 days	NOAEL	0.3 mg/kg bw/d	Hepatocellular necrosis	(Loveless *et al.*, 2008); EFSA (2018c)
Mouse	Tube feeding	29 days	NOAEL	1 mg/kg bw/d	Immunotoxicity	
Mouse	Tube feeding	29 days	-	0, 0.3, 1, 10, and 30 mg/kg bw/d (NH4-salt)	Decrease in body weight and increase in liver weight, from 10 mg/kg bw/d; Immunological effects (only tested at 30 mg/kg bw/d)	

Summary: the developing foetus appears to be highly sensitive to exposure to PFOA. Effects associated with PFOA are (see also 3.7.4):

- Lower birth weight and falling neonatal survival rate
- Liver effects (weight, increase in enzymes and necrosis)
- Deregulation of the immune system
- Neurotoxicity: increased spontaneous movement activity
- High cholesterol
- Lower vaccination response
- Disruption of thyroid function
- Testicular cancer and kidney cancer

3.7.4. SUMMARY OF THE AVAILABLE TOXICOLOGICAL REFERENCE VALUES

A summary of the toxicological assessment values derived for PFOA by various bodies is given in Table 60.

→ **Oral - non-carcinogenic**

EFSA

The TDI of EFSA (2008b) is based on the 95% lower limit of the confidence interval of the benchmark dose for a 10% increase in absolute hepatic weight (BMDL10) from multiple studies in rats and mice. The BMDL10 values ranged from 0.3 to 0.7 mg/kg/d. The lowest BMDL10 (0.3 mg/kg/d) was obtained in studies in male rats after 7 weeks of exposure (Palazzolo, 1993; Perkins *et al.*, 2004). An uncertainty factor of 200 (100 for inter- and intraspecies variability and 2 for uncertainties about the kinetics of the internal PFOA dose as the kinetics of PFOA in both rat and human are still unclear) was applied to the BMDL10 in order to arrive at a TDI of **1500 ng/kg/d**.

In a scientific opinion from 2018, EFSA published a preliminary[14] oral health-based guideline value for PFOA (EFSA, 2018c). The derivation of this guideline value is based on epidemiological studies that were not yet available in 2008. The increase in cholesterol levels in the serum was considered a critical effect by CONTAM. Two major studies investigating the association between PFOA levels and cholesterol in serum (Steenland *et al.*, 2009; Eriksen *et al.*, 2013) showed a very similar BMDL5 for 5% increase in cholesterol (9.2 and 9.4 ng/ml), corresponding to an estimated chronic daily intake of 0.8 ng/kg bw/d, according to a human PBPK model. An overview of the BMD analysis and the associated modelled daily dietary intake is given in Table 59. For the BMD modelling study by Steenland *et al.* (2009), with the lowest decile as reference, a 5% increase could not be modelled because the dose response curve flattened out at high PFOA concentrations in serum. The median PFOA concentration at the lowest decile (5.5 ng/ml) was higher than in other cohorts and in biomonitoring studies. Therefore, a "low" (L) concentration of 1 ng/ml, which is half of the median of PFOA in serum in Europe (1.9 ng/ml), was chosen. CONTAM is aware that this is an extrapolation outside the aggregated data of the study by Steenland *et al.* (2009) but it is within the interval of individual data of the lowest decile in the same study. This method increases the uncertainty about the BMDL5. The study by Eriksen *et al.* (2013) does not have this shortcoming, but has a smaller cohort than the study by Steenland *et al.* (2009). CONTAM therefore decided to use both studies to derive the HBGV. CONTAM has not corrected for simultaneous exposure to PFOS (and possibly other

PFAS) but acknowledges that this could potentially lead to slightly higher BMDL5 levels and associated daily intake estimates.
For the association between PFOA and alanine transferase (Health Canada)in serum, the modelling was performed for an increase in the absolute risk of ALT above its reference value (> 45 IU[40]/L for males and > 34 IU/l for females), with the results of the study by (Dallas *et al.*, 1989). An absolute increase of 5% ALT in serum did not occur even in the highest decile and could therefore not be modelled, an increase of 3% (from 9% to 12%) could. The association between PFOA and birth weight could also be modelled, with a benchmark response of 5% corresponding to a reduction of roughly 170 g in birth weight.

Table 59: Overview of the BMD analysis of PFOA (EFSA, 2018c)

Human endpoint	BMDL5 (ng/ml)	Intake via food[d] (ng/kg bw/d)	Number of persons (cohort)	Type data	Model	Reference
Total cholesterol	9.4[a)]	0.8	46,294[b)]	deciles	Log normally cumulative	Steenland *et al.* (2009)
	9.2	0.8	753 (Danish cohort 1996-2002)	octiles	Linear (square root)	Eriksen *et al.* (2013)
Alanine transferase[(c)]	21	2.0	47,092	deciles	Logistical	Gallo *et al.* (2012)
Birth weight	10.6	1.0	1,400 (Danish birth cohort 1996-2002)	deciles	Linear	Fei *et al.* (2007)
	4.0	0.4	901 (Norwegian birth cohort)	quartiles	Exponential	Whitworth *et al.* (2012)

(a) modelled with extrapolation of a reference value of 1 ng/ml PFOA in serum; an increase of 5% in the response observed in the lowest quantile could not be modelled
(b) local residents (age ≥ 18 years) who drank water contaminated with PFOA from a chemical plant in West Virginia for at least 1 year.
(c) BMDL3; alanine transferase is a marker for hepatocellular damage
(d) estimated value, corresponding to the BMDL5 of a PBPK model (rounded numbers)

The five studies in Table 59 give BMDL5 values of 4.0 to 21 ng/ml plasma, corresponding to a lifetime daily intake of 0.4 to 2.0 (median 0.8) ng/kg bw/day according to a human PBPK model. The CONTAM panel considered that these human studies provided sufficient evidence to derive a health-based guideline value; the value of **0.8 ng/kg bw/day** is therefore proposed by EFSA as a possible new TDI for PFOA.
No additional uncertainty factor was applied, as the BMD modelling was based on large epidemiological studies of the general population, including sensitive sub-groups.
Taking into account the long half-life of PFOA, a Tolerable Weekly Intake (TWI) of 6 ng/kg bw per week.
Due to the fact that both toxicity and the underlying mechanism of action are not sufficiently known and may be different but also overlap, the CONTAM panel decided not to derive group-HBGV for PFOS and PFOA (EFSA, 2018c).
A new EFSA risk assessment, which was published after the finalisation of this report, includes a tolerable weekly intake (TWI) based on epidemiological data to specifically protect infants. The TWI calculated as the sum of PFOA+PFNA+PFHxS+PFOS (which contribute most to human exposure) is 4.4 ng/kg bw/week. Effects on the immune system were considered the most critical endpoint for the risk assessment. Equal potencies were assumed for the four PFASs (EFSA CONTAM Panel *et al.*, 2020).

[40] International Units

ECHA

In 2015, the Risk Assessment Committee (RAC) established a Derived No Effect Level (DNEL) for the general population, within the framework of a restriction proposal for PFOA and its salts, of **800 ng/ml serum** (ECHA, 2015). The basis for the DNEL was the NOAEL of approximately 20000 ng/ml serum from the reproduction study with mice by Lau *et al.* (2006). The uncertainty factor is 250 (10 for intra-species differences and 2.5 for intra-species differences). A factor for kinetic differences is not necessary because the starting point is a serum concentration. ECHA also evaluated the evidence for humans of the data on cholesterolemia and immunotoxicity, but concluded that the data were not robust enough or the damage was unclear or there were uncertainties in the dose response. At an expert meeting on the new TDI of EFSA, ECHA stated that the restriction on PFOA, its salts and related substances is based on identification as persistent, bioaccumulative and toxic (PBT); the sole purpose of the human health assessment was to give the restriction a broader basis and not to derive a definitive reference value (EFSA, 2018a).

The Netherlands

In the Netherlands, the TDI is determined at **12.5 ng/kg bw/d** (Zeilmaker et al., 2016). While ECHA uses effects on reproduction to derive a reference value, the Netherlands bases its TDI on the prevention of liver effects (hypertrophy) as the most critical endpoint in animal studies. Perkins *et al.* (2004) exposed male rats to 1, 10, 30 and 100 ppm PFOA in food (0.06, 0.64, 1.94 and 6.50 mg/kg bw/d) for 13 weeks. Effects on liver weight and liver cell hypertrophy were found at doses ≥ 10 ppm (0.64 mg/kg bw/d); the LOAEL for elevated liver weight is 0.64 mg/kg bw/d, the NOAEL is 0.06 mg/kg bw/d. The serum concentration corresponding to the NOAEL was 7.1 ng/ml. The NOAEL in the rat corresponds to 1.0 µg/kg bw/d in humans (how Zeilmaker obtains this figure is not indicated; is allometric scaling taken into account[41]?). Furthermore, the following uncertainty factors are applied:

- 1 for toxicodynamics within the interspecies variability, due to the higher sensitivity of rats compared to humans, for liver effects when exposed to PFOA.
- 10 for intraspecies variability
- 8 as an adjustment for chronic exposure; (this factor is based on an empirically derived distribution proposed by IPCS (2014)[42] and has a range of 95%, i.e. there is 95% confidence that this factor takes sufficient account of possible sub-chronic differences.

This results in a TDI of 12.5 ng/kg bw/d, corresponding to an acceptable level of 89 ng/ml in serum (Zeilmaker & Janssen, 2016).

Great Britain

In 2006, UKCOT[16] derived a provisional TDI of 3000 ng/kg bw/d. This was based on liver effects in a number of studies with rats and mice. The TDI was derived from a dose of 0.3 mg/kg bw/d and an uncertainty factor of 100 (for inter- and intraspecies variability). In 2009, UKCOT revised its TDI following the publication by EFSA (2008b) of a TDI of 1500 ng/kg bw/d. The difference in both TDIs is due to the use by EFSA of an additional uncertainty factor of 2 for uncertainties related to the kinetics of the internal dose. UKCOT concluded that an additional factor for interspecies differences in toxicokinetics was justified and adjusted its TDI to **1500 ng/kg bw/d**. However, the TDI remains provisional as it will be revised when new information becomes available (FSANZ, 2016).

Denmark

[41] Allometric scaling extrapolates doses according to an overall assumption that equitoxic doses (when expressed in mg/kg bw/day) scale with body weight to the power of 0.75. (REACH guidance R.8 https://www.echa.europa.eu/documents/10162/13632/information_requirements_r8_en.pdf/e153243a-03f0-44c5-8808-88af66223258

[42] http://www.who.int/ipcs/methods/harmonization/uncertainty_in_hazard_characterization.pdf?ua=1

In 2015, Denmark derived a TDI of **100 ng/kg bw/d**, based on developmental toxicity and increased liver weight in rats as the most critical endpoints (DEPA, 2015). Denmark continues to build on the BMDL10 values derived by EFSA (2008) and US-EPA (2014) from the Pallazo study to derive its TDI. Both bodies use other studies in their more recent derivations. The basis of Denmark's TDI is therefore in fact obsolete.

Germany
The HBM I value of PFOA is set at **2 ng/ml blood plasma**, for the general population (Umweltbundesamt, 2016). For PFOA, the HBM I value is based on epidemiological studies and critical endpoints such as immunological effects, birth weight and development (puberty). Analogy with results from animal tests increases confidence in the HBM I value of PFOS, according to the authors (Apel *et al.*, 2017).

Sweden
In a study commissioned by the Swedish Environment Agency, no TDI was derived, but a safe serum concentration (Derived no effect level (DNEL)) for the general population and for different endpoints (liver toxicity, reproductive toxicity and other effects) was derived (Borg & Håkansson, 2012). The lowest DNEL was for 'other effects', i.e. development of mammary glands and increased body weight in mice (LOAEL 0.01 mg/kg bw/d or 15 µg/ml serum; White *et al.* (2007); White *et al.* (2009); White *et al.* (2011)), and is **2.0 ng/ml serum**. This value is based on a POD of 150 ng/ml serum and an uncertainty factor of 75 (3 for extrapolation from LOAEL to NOAEL, 2.5 for interspecies variability (no factor for toxicokinetics because serum concentrations of humans and animals are compared), and 10 for intraspecies variability). The DNELs for liver and reproductive toxicity (endpoints relevant for multiple PFAS) were higher (142 and 628 ng/ml serum, respectively).

US-EPA
The RfD of US-EPA is **20 ng/kg bw/d** (US-EPA, 2016d). This value is based on effects observed in a development study with mice (Lau *et al.*, 2006). The basis for the RfD was the LOAEL of 1 mg/kg/d; this LOAEL corresponds to a calculated average serum concentration of 38.0 mg/l for which it was determined to represent 56% of the steady-state concentration. The critical effects associated with the LOAEL are reduced ossification of proximal phalanges[43] of front and hind legs of male and female pups and accelerated (4 days earlier than controls) adolescence in male pups from dams exposed to PFOA via tube feeding on day 1-17 of pregnancy and sacrificed at weaning (Lau *et al.*, 2006). From the calculated serum concentration (38.0 mg/l) a human equivalent dose (HED) of 5.3 µg/kg/d was derived. An uncertainty factor of 300 (10 for intraspecies variability, 3 for toxicodynamic differences between animals and humans and 10 for the use of a LOAEL) was applied to the HED LOAEL to obtain an RfD of 0.02 µg/kg/d. No correction was made for lifetime exposure, as the average serum concentrations associated with the development studies are more protective than those of the long-term systemic toxicity studies (US-EPA, 2016d). The RfD of 20 ng/kg bw/d appears to correspond to a serum concentration of 142 ng/ml (Zeilmaker & Janssen, 2016).

ATSDR
In its provisional toxicological profile of 13 PFAS, ATSDR derived an intermediate oral MRL (minimum risk level) for PFOA of **3 ng/kg bw/d** (ATSDR draft, 2018). The critical effects were altered activity at 5-8 weeks of age and changes in skeleton at 13 and 17 months of age in offspring of mice fed PFOA from the 1st to the 21st day of pregnancy (Onishchenko et al. 2011). The MRL is based on a HED LOAEL of 0.000821 mg/kg/d and a total uncertainty factor of 300 (10 for LOAEL →NOAEL, 3 for rat extrapolation to humans with adjustments for dosimetry, and 10 for intraspecies variability).

[43] Finger or toe joints lying between the metacarpals and middle phalanges (Wikipedia)

Australia and New Zealand
The health based guideline value (HBGV) is derived from the most sensitive endpoint from the mouse study by Lau *et al.* (2006), namely foetotoxicity. The HED_{NOAEL} is 4.9 µg/kg bw//d. The uncertainty factor is 300 (10 for inter-species and 3 for intra-species variability). The HBGV is **160 ng/kg bw/d**. (FSANZ, 2016)

→ **Inhalation, non-carcinogenic**

Compared to oral intake data, there is little data available on exposure via inhalation. The importance of this route of exposure is therefore unclear (CONCAWE, 2016). There are not enough data to derive a reference value (US-EPA, 2016d; ATSDR draft, 2018).

Table 60: Toxicological criteria for PFOA

Body	Type value	Value	Basis	Critical effect	Study	Factors	Reference
Oral intake (mg/kg.d)							
EFSA	*TDI*	*$1.5.10^{-3}$*	*NOAEL*	*Liver effects*	*Rat, mouse (Palazzolo, 1993; Perkins et al., 2004)*	*200 (100 for inter- and intraspecies variability and 2 for kinetics)*	*EFSA (2008b)*
EFSA	*HBGV - draft*	*$0.8.10^{-6}$*	*BMDL5*	*Increase in cholesterol levels in serum*	*Epidemiological studies: Steenland et al. (2009); Nelson et al. (2010); Eriksen et al. (2013)*	*-*	*EFSA (2018c)*
The Netherlands	*MTR*	*$1.25.10^{-5}$*	*NOAEL*	*Liver effects (hypertrophy)*	*Rat*	*Converted to human equivalent dose, and 1 (toxicodynamics) and 10 (intraspecies)*	*Zeilmaker and Janssen (2016); RIVM (2019)*
United Kingdom	*TDI*	*3.10^{-3}*	*NOAEL*	*Liver effects*	*Rats and mice*	*100 (intra- and interspecies)*	*FSANZ (2016)*
Denmark	*TDI*	*1.10^{-4}*	*BMDL10*	*developmental toxicity and increased liver weight*	*Rat -(Palazzolo, 1993)*		*DEPA (2015)*
US-EPA	*RfD*	*2.10^{-5}*	*HED LOAEL*	*development toxicity*	*Mice (Lau et al., 2006).*	*300 (10 for intraspecies, 3 for toxicodynamics, 10 for LOAEL)*	*(US-EPA, 2016d).*

ATSDR	*MRL - proposal*	*3.10^{-6}*	*HED LOAEL*	*development toxicity*	*Mice – Onishchenko et al. (2011)*	*300 (10 for LOAEL, 3 for interspecies and dosimetry, 10 for intraspecies*	*ATSDR draft (2018)*
Australia and New Zealand	*TDI*	*$1,6.10^{-4}$*	*HED NOAEL*	*foetotoxicity.*	*Mice (Lau et al., 2006).*	*30 (10 interspecies; 3 intraspecies)*	*FSANZ (2016)*
Serum concentration ng/ml							
ECHA-RAC	*DNEL (not intended to be reference value)*	*800*	*LOAEL*	*reproduction*	*Mouse (Lau et al., 2006).*	*250 (10 intraspecies; 2.5 interspecies)*	*(EFSA, 2018a)*
Germany	*HBM I*	*2*	*-*	*immunological effects, birth weight, development*	*Epidemiological studies*	*-*	*Apel et al. (2017) (Umweltbundesamt, 2016)*
Sweden	*DNEL*	*2*	*LOAEL*	*development of the mammary glands and body weight*	*Mice - White et al., 2007, 2009, 2011*	*75 (3 for LOAEL, 2.5 for interspecies, 10 for intraspecies)*	*Borg and Håkansson (2012)*

→ **Carcinogenic**

US-EPA (2016) derived a human equivalent dose of 0.58 mg/kg bw/d and an slope factor of 0.07 $(mg/kg/d)^{-1}$. The basis for this factor are the dose-response data from Leydig cell tumours in rats (Butenhoff, 2012). The slope factor corresponds to a dose of $1.43.10^{-4}$ mg/kg bw/d for an additional cancer risk of $1/10^5$. US-EPA states that the lifetime health advisory (2.10^{-5} mg/kg bw/d) based on non-cancer effects is protective for the cancer endpoint (US-EPA, 2016b)
A unit risk for inhalation or a slope factor for dermal contact are not available.

3.7.5. PROPOSAL FOR TOXICOLOGICAL REFERENCE VALUES TO BE USED

It is generally accepted that the current TDI of EFSA is too high. The new TDI of EFSA is still provisional and therefore cannot be selected as a reference value. Moreover, the Netherlands, Germany and Denmark have raised substantive objections to the evaluation by EFSA (EFSA, 2018c).
The TDI of **12.5 ng/kg bw/d** of the Netherlands is in line with the RfD of 20 ng/kg bw/d; however, the Netherlands will review its TDI when EFSA publishes its final TDI (RIVM, 2019).
ATSDR, US-EPA 2016 and ECHA/RAC do not use information from epidemiological studies to derive a health reference value, which RIVM considers to be a drawback (Zeilmaker, 2016).
The MRL of ATSDR (2018) is a proposal and therefore cannot be selected as a reference value.

EFSA has applied an additional uncertainty factor of 2 for differences in toxicokinetics. To avoid this, US-EPA used PBPK modelling to derive an HED, which served as a starting point for the derivation of an RfD. A PBPK model uses, as it were, substance-specific evaluation factors and using a HED NOAEL may therefore be considered more reliable than using an experimental NOAEL and standard uncertainty factors.

The RfD of US-EPA (20 ng/kg bw/d) is selected as a reference value for deriving soil remediation values.

Because there is no toxicological reference value for exposure via inhalation, this is calculated from the TDI (20 ng/kg bw/d) with the following parameters: 70 kg body weight, 20 m^3/day of breathing volume and 95% inhalation absorption (equivalent to oral absorption). The calculation results in a tolerable concentration in air (TCA) of **70 ng/m^3**.

In order to have an idea of the impact on the soil remediation value, scenarios are also calculated with the MTR of the Netherlands (12.50 ng/kg bw/d) and the TDI proposal of EFSA (0.8 ng/kg bw/d).

3.8. ECOTOXICOLOGY

For the evaluation of ecotoxicological effects , no new primary sources and/or databases were consulted to derive possible new ecotoxicological values. However, it was examined whether substantiated ecotoxicological values have recently been derived by other regulatory bodies. Based on the guidelines for drawing up soil remediation values (Cornelis and Touchant, 2016), the following sources were consulted:

- US EPA: http://www.epa.gov/ecotox/ecossl/index.html

- CCME Canada: http://www.ccme.ca/en/resources/canadian_environmental_quality_guidelines/index.html
- RIVM Netherlands (intervention values, national reference values: http://www.rivm.nl/rvs/Normen/Milieu/Bodeminterventiewaarden, http://www.rwsleefomgeving.nl/onderwerpen/bodem-ondergrond/bbk/instrumenten/nobo
- ECHA database: http://echa.europa.eu/information-on-chemicals

The results of this inventory are summarised in section 3.10.3. Ecotoxicological reference values

3.9. LEGAL LIMITS

3.9.1. OUTDOOR AIR AND INDOOR AIR

PFOA is a low-volatile substance. PFOA does not occur in the WHO Air quality guidelines for Europe (WHO, 2000) or those of other bodies (ANSES, RIVM, Germany, US-Clean air Act, LCI (lowest concentration of interest) for indoor air).
APFOA is not included in the Flemish Indoor Environment Decree (BOG2018) and also does not appear in the WHO guidelines for indoor air quality (WHO, 2010) or in the list of LCI (lowest concentration or interest) substances for indoor air (EU-LCI, 2016).

The TCA is 70 ng/m³.

3.9.2. DIET AND FOOD

There are restrictions on the use of ammonium perfluorooctanoate (Case No 3825-26-1) in plastics intended to come into contact with foodstuffs (EU, 2011)).
PFOA is not included in the European list of undesirable substances in animal feed (EC, 2002). There is no European standard for PFOA in water intended for human consumption (EC, 1998).

3.9.3. DRINKING WATER

On 1 February, the **European Commission** adopted a **proposal** to revise the Drinking Water Directive 98/83/EC (EC, 1998). This sets the drinking water standard at **0.1 µg/l** for individual PFAS (including PFOA) and 0.5 µg/l for PFAS total (EC, 2018).
The Netherlands applies an indicative guideline value for drinking water of 87.5 ng/l for lifelong exposure (Zeilmaker & Janssen, 2016; van der Aa & al., 2017). This value is calculated on the basis of a TDI of 12.5 ng/kg/day, a contribution of drinking water to the TDI of 20%, a body weight of 70 kg and drinking water consumption of 2l/day. This indicative guideline value is based on the TDI used by the Netherlands pending a definitive new TDI from EFSA.
In Germany, the guideline value (Leitwerte) for drinking water is 0.1 µg/l (UBA, 2017). This value is based on an average of the results of epidemiological studies (UBA, 2016a). The United States of America has a lower health-based advisory value of 0.07 µg/l for drinking water, which is a sum parameter for PFOS and PFOA (US-EPA, 2016). By way of comparison, US-EPA has calculated a drinking water limit from the slope factor for testicular tumours. US-EPA takes into account an additional cancer risk of $1/10^6$, 80 kg body weight and drinking water consumption of 2.5 litres. The result is 0.5 µg/l and exceeds the US-EPA standard (0.07 µg/l) based on non-carcinogenic effects.
Australia applies a limit of 0.56 µg/l for PFOA (Australia, 2016).

For deriving soil remediation values , the value of **0.1 µg/l** is selected as it is an accepted value within the revision of the European Drinking Water Directive.

Table 61: Drinking water standards of various countries/bodies

Substance	Value	Details	Country/region	Reference
PFOA	0.56 µg/l	Drinking water quality value	Australia	Australia (2016)
PFOA	0.0875 µg/l	Drinking water reference value	The Netherlands	Alphenaar *et al.* (2018)
PFAS separately and total	0.1 µg/l for individual PFAS and 0.5 µg/l for PFAS total	Proposal of limit values for drinking water	EU	EC (2018)
PFOA + PFOA	0.07 µg/l (70 ppb)	Health Advisory for lifetime exposure	USA	US-EPA (2016b)
PFOA	0.1 µg/l	Guideline value	Germany	UBA (2017)

3.10. CALCULATION OF THE SOIL REMEDIATION VALUE

The calculations of the soil remediation values were made with a modified S-Risk version 1.3 Application I for the calculations, Application II with modified buffer space (0.75 m) for interpretation of exposure routes and exposure pathways. To avoid the use of the K_{ow}, S-Risk version 1.3 was specifically adapted for PFAS on the VITO test server. Transfer to plants can therefore be calculated with BCF factors based on dry matter concentration in the soil, whereas normally BCF factors for organic substances are expressed on pore water concentrations. Moreover, PFOA was considered a non-dissociative substance in S-Risk in the calculations (PFOA is a dissociative substance), which means that K_d can be calculated directly from the organic carbon content in the soil and K_{oc} without the use of K_{ow}. The user manual of S-Risk states "If a K_{oc}-value is available for a dissociative substance at the correct soil pH, it is also possible to leave the dissociative option-button unchecked while filling out the required K_{oc} value. However, the calculations must only be carried out for the applicable pH range".

3.10.1. GROUNDWATER

EFSA carried out a comprehensive study of chronic exposure to PFOA via food, setting upper and lower limits for minimum, average and maximum intakes (EFSA, 2018c). In the lower limits of average exposure, the highest contribution for drinking water was found in a Swedish study measuring the relative contribution of different pathways to total exposure; the contribution of PFOA via drinking water ranged from 9.1 to 11% (Haug *et al.*, 2011). As the contribution of PFOA is larger than PFOS, the contribution of drinking water to the TDI should not be lower than that of PFOS; therefore, the WHO standard value of 0.2 is taken as the contribution. The SRV for groundwater is calculated using the standard formula (see 3.10.1).
The mean, measured dermal absorption coefficient (Kp) of PFOA = 5.8 x 10-5 cm/h (Franko *et al.*, 2012). This value is lower than the minimum Kp for the contribution to drinking water (0.022 cm/h), therefore dermal exposure via drinking water should not be considered in the derivation of the soil remediation value.

The basis for determining the inhalatory drinking water equivalent is the Henry coefficient. This cannot be estimated from the vapour pressure and the solubility because the vapour pressure above an aqueous solution is reduced, as PFOA dissociates in water (EFSA, 2008c). For this reason, the inhalatory drinking water equivalent cannot be calculated, but it is probably low; for the purposes of deriving soil remediation values , it is set at zero.

Table 62: Reference values for groundwater

Toxicological reference value	Value	Unit	SRV groundwater (ng/l)
Set 1 (preference) (US-EPA, 2016d).			
TDI oral	2.10^{-5}	mg/kg/d	120
TCA inhalation	7.10^{-5}	[mg/m³]	
TDI dermal	2.10^{-5}	mg/kg/d	
Set 2 RIVM (2019)			
TDI oral	$12.5.10^{-6}$	mg/kg/d	75
TCA inhalation	$43.8.10^{-6}$	[mg/m³]	
TDI dermal	$12.5.10^{-6}$	mg/kg/d	
Set 3 EFSA (2018c)			
TDI oral	$0.8.10^{-6}$	mg/kg/d	4.8
TCA inhalation	$2.8.10^{-6}$	[mg/m³]	
TDI dermal	$0.8.10^{-6}$	mg/kg/d	

3.10.2. SOIL

The calculations were made for 3 different sets of toxicological reference values, as described in the substance sheets at the end of this report. The first scenario makes use of the RfD in (US-EPA, 2016d) (= preferred scenario), the second scenario is based on the MTR of the Netherlands as described in RIVM (2019) and the third scenario is calculated with the proposed TDI-value of EFSA (2018c).
As was the case for PFOS, the calculations were carried out in the first instance using the UB food consumption and concentration data of EFSA (2018c). However, the calculations gave negative background exposure for the landuse type agriculture through food consumption. This indicates that the exposure via locally grown foods in an agricultural setting exceeds the general background exposure via dietary intake of consumtion foods, possibly because the estimated intake via locally grown vegetables is overestimated by the available BCF. Also for the calculations based on the UB data from EFSA (2012) it appeared that for scenario 3 the oral toxicological reference value (TDI oral = 0.8 ng/kg/d) is already fully filled in by the background intake via food (2.63 ng/kg/d).
On the basis of previous observations, the final calculations for deriving the SRV were carried out on the basis of the lower bound (LB) intake figures from EFSA (2012). The calculations were made for the three different scenarios or three different sets of toxicological limit values. The results are summarised in Table 63 and compared graphically in Figure 18.

Table 63: Proposed human toxicological soil remediation values for PFOA (µg/kg dry matter) when lower bound intake figures from EFSA (2012) are used. The calculations were made for 3 sets of toxicological reference values, see also the substance sheets at the end of this report and chapter 3.7 Toxicology.

	II	III	IV	V
S-Risk tox 1 (US-EPA, 2016d)				
	4.3 (threshold)	**205** (threshold)	IV a	V a

			12,610 (threshold)	14,080 (threshold)
			IV b 14,760 (threshold)	V b **12,770** (threshold)
Adjustment			IV a and IV b **643** (drinking water)	V a and V b **643** (drinking water)
S-Risk tox 2 RIVM (2019)				
	2.7 (threshold)	**127** (threshold)	IV a **7,831** (threshold)	V a 8,769 (threshold)
			IV b 9,166 (threshold)	V b **7,954** (threshold)
Adjustment	-	-	IV a and IV b **643** (drinking water)	V a and V b **643** (drinking water)
S-Risk tox 3 EFSA (2018c)				
	0.14	**6.2**	IV a **375** (threshold) IV b 439 (threshold)	V a 489 (threshold) V b **443** (threshold)
Adjustment	-	-	-	-

bold: values proposed as soil remediation values based on tox values
bold underlined: values proposed as soil remediation values based on adjustment based on binding legal reference values
-: the concentration indices are not critical, no adjustment is needed
Threshold: the non-carcinogenic endpoint is the most critical, the proposed soil remediation value corresponds to the value at which there is no longer a risk for children 1 - 6 years (with the exception of industry where no children are present)

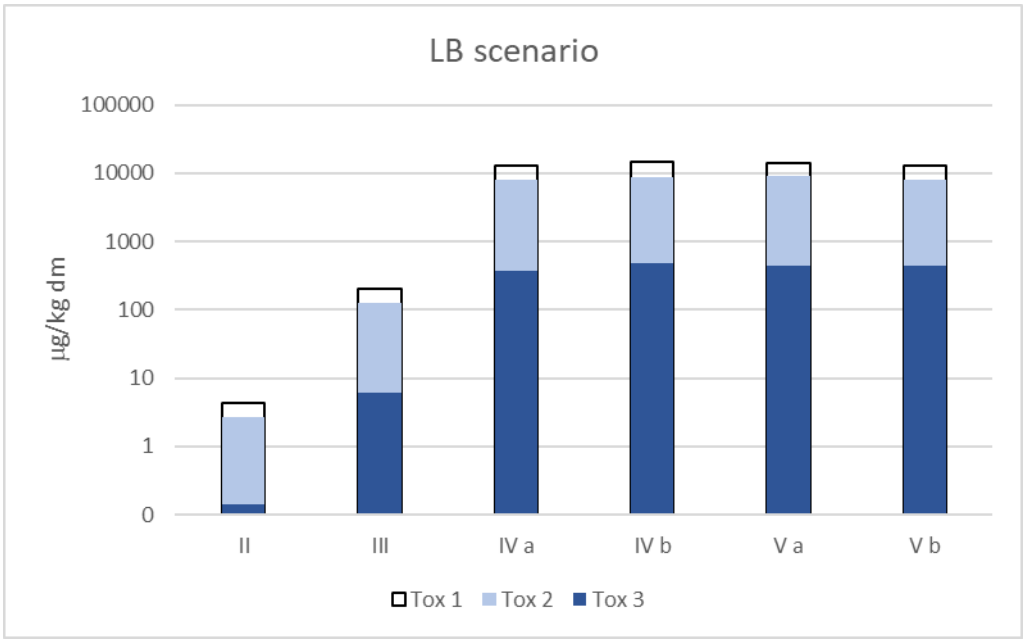


Figure 18: Comparison of the SRV (µg/kg dm) calculated on the basis of three sets of toxicity reference values under scenario 2 (LB scenario) for the different landuse types

The contribution of the exposure pathways to the risk for the different landuse types is shown in Figure 19.

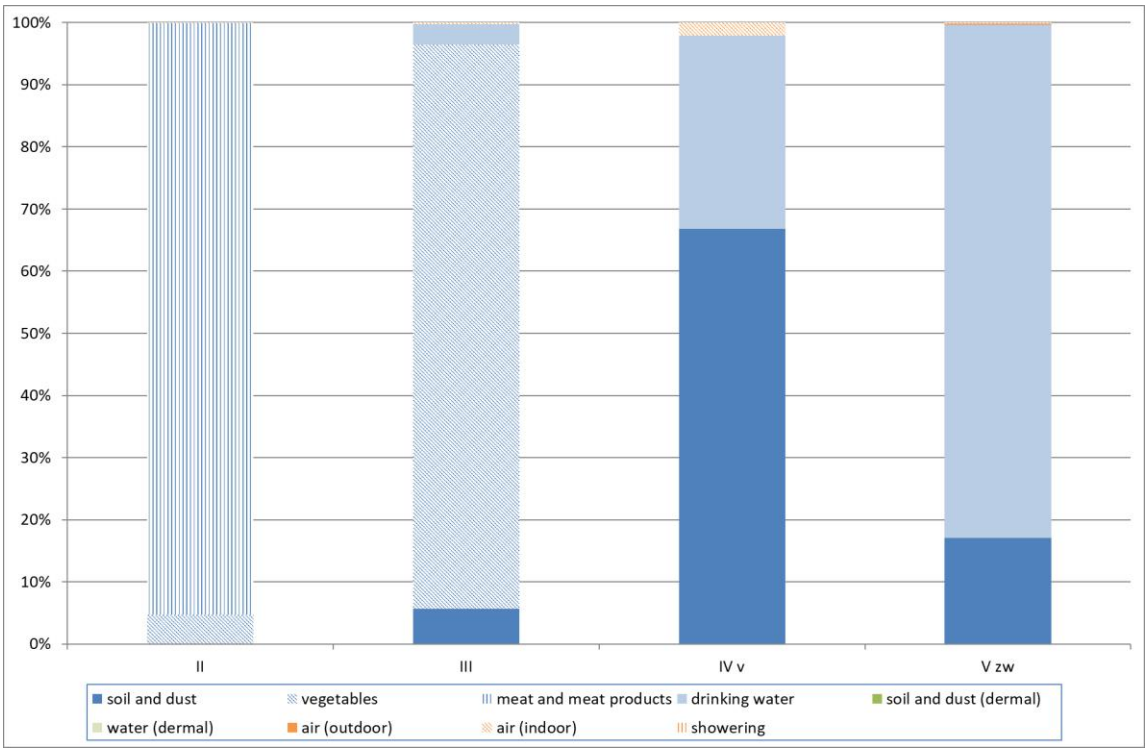


Figure 19: Contribution of exposure pathways to overall risk for PFOA in tox scenario 1 (US-EPA (2016c)) and with background dietary exposure based on the LB approach of EFSA (2012) (blue: oral, green: dermal, orange: inhalation). The contribution has been calculated for a soil concentration

equal to proposed soil remediation values and - with the exception of landuse type Vb - for children (1-6 years).

This figure has been calculated on the basis of the toxicologically based soil remediation value as calculated for each landuse type, and as such it does not take into account adjustments. We see that for all landuse types the oral exposure route is dominant. In type II, the determining factor is consumption of local meat and animal products (95.2%) with only a limited contribution from vegetables (4.6%). Local meat and animal products consist of 95% milk for children, 51% for adults, followed by beef (11%). Offal contributes less than 0.3% for adults. The concentration calculated in milk at a concentration in soil equal to the proposed soil remediation value for Tox1 (4.32 µg/kg dm) is 0.69 µg/kg fw, more or less a factor of 6 higher than the UB concentration as determined by EFSA (2012), i.e. 0.12 µg/kg fw. At 2.68 µg/kg dm in the soil (SRV based on Tox2) the concentration of PFOA in milk is 0.43 µg/kg fw (factor 4). At 0.14 µg/kg dm in the soil (SRV based on Tox 3) we calculate a PFOA concentration in milk of 0.22 µg/kg fw, still about a factor of 2 higher than the UB from EFSA (2012). If we take into account the left-skewed data distribution, it appears that the concentration in 94% of milk samples was lower than the LOD or LOQ, and as a result of the assumption under the UB scenario, were equated with the LOD or LOQ by EFSA. However, the LB concentration determined by EFSA (2012) is 0 mg/kg fw. This shows that the current calculations most probably overestimate the concentration in animal products, and milk in particular.

For type III, the consumption of vegetables weighs the most (91.1%), with a limited contribution from ingestion of soil and dust (5.7%). Type IV is dominated by soil and dust ingestion (68%), drinking water contributes about 32% and type V is largely determined by drinking water (83%).

To ascertain whether we obtain realistic SRV for the various scenarios, a comparison is made with measured background values for PFOA in the Netherlands (as insufficient measurements are available for Flanders). In the Netherlands the expertise centre PFAS (Pancras & van Bentum, 2018) collected data on the presence of PFOS and PFOA in the topsoil (up to about 0.5 m minus ground level) in the Netherlands. By using data from Northern Holland, Utrecht and Northern Brabant for PFOA, the aim was to rule out the influence of potential risk locations or known PFOA sources. The report states the percentile values for PFOA as shown in Table 64.

Table 64: Calculated percentile values for diffuse load of PFOA in the topsoil (Pancras & van Bentum, 2018).

Percentiles	PFOA
25 percentile (µg/kg dm)	0.40
Median; 50 percentile (µg/kg dm)	0.73
75 percentile (µg/kg dm)	1.10
90 percentile (µg/kg dm)	2.00
95 percentile (µg/kg dm)	4.60

In Figure 20 the calculated SRV for the landuse types II and III are compared for each of the three sets of tox values, with the median, the 90th and 95th percentile values of the concentrations of PFOA in topsoils of Northern Holland. The SRV for agricultural soils derived for all three sets of toxicity values are always below the 95 percentile value of background PFOA from (Pancras & van Bentum, 2018). If and when PFOA background levels in Flemish topsoil are proven to be at the same level of those in the Netherlands, the above findings would indicate that, with current scientific knowledge and available calculation methods, no feasible SRV for agricultural areas can be derived in Flanders.

Based on the toxicity criteria from (US-EPA, 2016d) (tox 1) and RIVM (2019) (tox 2) we can derive, for residence with vegetable gardens, a SRV with a sufficiently large margin above the 95 percentile value of the PFOA background values in the Netherlands.

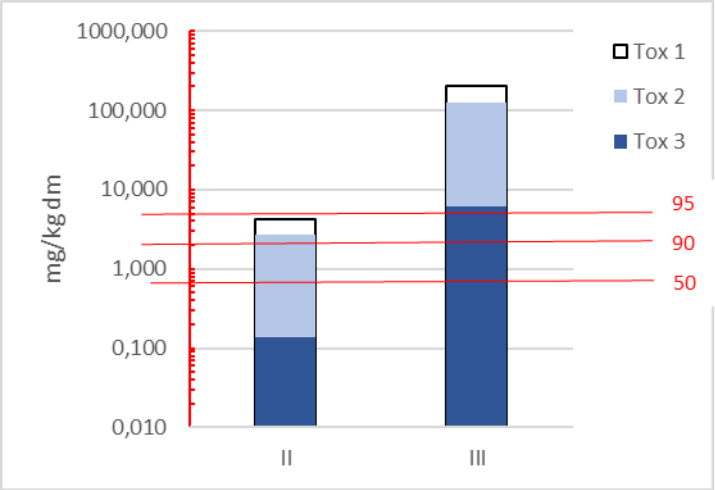


Figure 20: Comparison of the SRV on the basis of 3 tox scenarios with respect to the median, 90th percentile and 95th percentile value of the diffuse loading of PFOA in topsoils in the Netherlands (Pancras, 2018)

3.10.3. ECOTOXICOLOGICAL REFERENCE VALUES

Based on the guidelines for drafting soil remediation values (Cornelis and Touchant, 2016), the database data of the following 4 international bodies were consulted: US-EPA, ECHA, CCME and RIVM[44]. For PFOA, ecotoxicological limit values were only derived by RIVM (2019). These are shown in Table 65. In contrast to Flanders, in the Netherlands, biomagnification (accumulation to higher trophic levels) is taken into account for the determination of the ecotoxicological limit value and therefore a distinction is made in the table between direct ecotoxicity through soil contact and biomagnification.

[44] US EPA: http://www.epa.gov/ecotox/ecossl/index.html
CCME Canada: http://www.ccme.ca/en/resources/canadian_environmental_quality_guidelines/index.html
RIVM Netherlands: http://www.rivm.nl/rvs/Normen/Milieu/Bodeminterventiewaarden, http://www.rwsleefomgeving.nl/onderwerpen/bodem-ondergrond/bbk/instrumenten/nobo
ECHA database: http://echa.europa.eu/information-on-chemicals

Table 65: Overview of ecotoxicologically based reference values (direct soil contact and biomagnification in brackets) derived for PFOA in the Netherlands (RIVM, 2019) (µg/kg dm).

Reference value	Nature and agriculture areas	Residential and park areas (*)	Commercial and industrial areas	Reference
Ecological Risk direct (biomagnification)	**500**[a)] (7[c)])	**5,000**[a)] (89)	**50,000** [b)] (1,100)	RIVM, 2019

*a) medium protection level (*Geometric mean of HC5 and HC50*); b) moderate protection level (SR_{eco} Serious-Risk soil corresponding to the HC_{50} level) ; c) high protection level (MTR_{eco} Maximum Permissible Risk soil corresponding to the HC_5 level); (*) park areas are not a separate landuse type in the Netherlands*

The direct ecological risks are tested against two risk limits in the Netherlands: the Serious Risk to the soil ecosystem. ($SR_{eco\ soil}$) is the concentration at which harmful effects of the substance on the soil ecosystem are likely to occur and corresponds to the HC_{50} protection level[45] and the Maximum Permissible Risk for the ecosystem (MTR_{eco}) which corresponds to the HC_5 protection level[46]. Below this level, no negative effects on the soil ecosystem are expected. Where the $SR_{eco\ soil}$ applies in industrial areas, the MTR_{eco} is applied to agricultural and nature reserve areas. For the soil function class 'Residence with vegetable garden', a middle level is defined as the geometric mean of both. RIVM (2019) bases its derivation of the PFOA limit values on the previously published data from Lijzen et al., 2018. Based on the original data for direct ecotoxicity of PFOA, this gives a $SR_{eco,direct}$ of 50,000 µg/kg dm and a $MTR_{eco,direct}$ of 500 µg/kg dm. The middle level for direct toxicity is 5,000 µg/kg dm. If biomagnification is taken into account, RIVM (2019) derives the following ecotoxicological risk limits: $SR_{eco,indirect}$ = 1,100 µg/kg dm, $MTR_{eco,indirect}$ = 7 µg/kg dm and a middle level = 89 µg/kg dm. These values are adopted as a preliminary proposal for ecotoxicological standards for PFOS in Flanders (Table 66). Due to the persistent nature of PFAS as a substance group as a whole, it is proposed that, exceptionally and in contrast to normal practice in Flanders, biomagnification should be taken into consideration for the PFOA standard proposals for the SRV_{eco} for the landuse types 'Agriculture' and 'Residence with vegetable garden' and 'Recreational areas'. **The proposal for the SRV_{eco} for these landuse types is then respectively 7, 89 and 1,100 µg/kg dm (in bold in Table 66).**

Table 66: Proposal for ecotoxicological values for PFOA in Flanders (µg/kg dm); values based on direct toxicity are shown in brackets.

Reference value	Agriculture areas (type II)	Residence with vegetable garden (type III)	Recreational areas (type IV)	Industrial areas (type V)
SRV_{eco}	**7** (500)	**89** (5,000)	**1,100** (50,000)	50,000

3.10.4. TARGET VALUES

No target values for Flemish soils were available at the time this study was carried out. On behalf of OVAM, background values were measured in 2020, for which, for PFOA, a background value of 1.0

[45] The Hazardous Concentration for 50% of the soil organisms (HC_{50})
[46] The Hazardous Concentration for 5% of the soil organisms (HC_5)

µg/kg dm in soil was derived, more information can be found in Touchant *et al.* (2020). The Netherlands applies a temporary background value of 0.8 µg/kg dm in soil (Wintersen *et al.*, 2019)[24].

3.11. INTEGRATION AND EVALUATION

3.11.1. SOIL

The calculations for deriving the soil remediation values for PFOA were carried out in an adapted version of the S-Risk model version 1.3 (for the time being only available on an internal VIT0 test server) taking into account the amphiphilic character of PFOA, substances for which the log K_{ow} cannot be measured according to the OECD standard test guideline. In order to avoid the use of the log K_{ow}, the transfer to plants was initially calculated on the basis of BCF factors relative to the solid phase of the soil, in contrast to the usual method where BCF factors for organic substances in S-Risk are expressed on the basis of pore water concentrations. In addition, PFOA was considered as a non-dissociative in S-Risk during the derivation of the soil remediation values (PFOA is a dissociative substance) so that the sorption of soil particles (K_d) can be calculated directly from the organic carbon content in the soil and the K_{oc} without the intervention of K_{ow}.
Various scenarios with combinations of parameter values were calculated and tested for feasibility. The following parametric values were used to derive the proposed soil remediation values for PFOA:

- Toxicology:
  - The RfD of US-EPA (2016c) of **20 ng/kg bw/d** and the derived tolerable concentration in the air (TCA) of **70 ng/m³** (preferred scenario);
  - The RfD of Zeilmaker et al. (2016) of **12.25 ng/kg bw/d** and the derived tolerable concentration in the air (TCA) of **43.8 ng/m³**;
  - The RfD of EFSA (2018c) of **0.8 ng/kg bw/d** and the derived tolerable concentration in the air (TCA) of **2.8 ng/m³**;
- Background exposure: The lower bound intake and concentration data of EFSA (2012);
- Plant uptake: The BCF_{PFOS} values derived by Ghisi et al. (2019) after comparing the original data with the approach followed by Wintersen et al. (2019) applying a complete diet;
- Animal transfer: average BTF values derived from Vestergren et al. (2013) and Kowalczyk et al. (2013);

All parameter values used for the final human toxicological soil remediation values are summarised in the substance sheet at the back.

For the evaluation of ecotoxicological effects, no new primary sources and/or databases were consulted to derive possible new ecotoxicological values, but it was examined whether ecotoxicological values have recently been derived from other bodies. For PFOA the values derived by RIVM (2019) were used, whereby biomagnification was taken into account.

During this study, insufficient measurement data were available to derive reliable target values.
On behalf of OVAM, background values were measured in 2020, for which, for PFOA, a background value of 1.0 µg/kg dm in soil was derived (Touchant *et al.*, 2020).

A comparison of the proposed human and ecotoxicological soil remediation values is given in Table 67 with in green the preferred value based on the preferred toxicology scenario. At present there

are no soil remediation values for PFOA in the Flemish legislation on soil (VLAREBO) meaning that a comparison is not possible. If the US EPA scenario is used for the human health based SRV, the values in green are used as reference value (provisional SRV). A first comparison with Dutch background values (Pancras, 2018) shows that, with current scientific knowledge and available calculation methods, no feasible SRV for agricultural areas can be derived (landuse type II). The decision concerning the SRV for landuse type II (agriculture) (and thus also for landuse type I - nature) is awaiting the study 'Derivation of target values for perfluorinated compounds' commissioned by OVAM. The soil remediation values for landuse type I (nature) and landuse type II (agriculture) can be adjusted on the basis of the target values and the values for free use of soil.

Table 67: The proposed SRV for soil (µg/kg dm) for PFOA with the preferred value in green. The soil remediation values for landuse type I (nature) and landuse type II (agriculture) may be further adjusted to a feasible value on the basis of the target values and the values for free use of soil.

	II[47]	III	IV	V
Flemish legislation on soil (VLAREBO)	-	-	-	-
Proposal human health based tox US-EPA (2016c)	4.3	205	643 (drinking water) IV a and IV b	643 (drinking water) Va and Vb
Proposal human health based tox Zeilmaker et al. (2016)	2.7	127	643 (drinking water) IV a and IV b	643 (drinking water) Va and Vb
Proposal human health based tox EFSA (2018c)	0.14	6.2	375 (Va)	443 (Vb)
Proposal ecotox	7	89	1,100	50,000
Background value	1.0			

3.11.2. GROUNDWATER

The soil remediation value for groundwater has a human health based underpinning, and corresponds to the drinking water standard if this has a toxicological basis (Cornelis & Touchant, 2016). The drinking water standard of 100 ng/l proposed by the EU is a general limit (not specific for PFOA) which is mainly based on feasibility and not only on toxicology.
As such, the SRV for groundwater was also calculated, with the standard formula (paragraph 2.10.1) for the three toxicological reference values with which the soil remediation value was derived. The corresponding calculated values for groundwater are:

- 120 ng/l, based on the RfC of US-EPA (2016)
- 75 ng/l, based on the maximum tolerable human health based risk level (Zeilmaker *et al.,*2016)
- 4.8 ng/l, based on the TDI proposal of EFSA (2018).

[47] Not final, will be adjusted on the basis of the target values and the values for free use of soil

120 ng/l, based on the RfC of US-EPA (2016) is the preferred value based on the preferred toxicology scenario. This is also the most closely related to the groundwater criterion put forward at EU level, i.e. 100 ng/l.

3.11.3. GUIDELINE VALUES

Guideline values are not yet available at the time of publication of this report and will be published in a separate document.

3.12. COMPARISON WITH FOREIGN SOIL REMEDIATION VALUES

In 2018, RIVM derived generic risk limits for non-agricultural soil functions that allow local authorities to develop a site-specific approach to PFOA contamination (Lijzen *et al.*, 2018). The derived generic intervention and target values for soil and groundwater are set out in Table 68. The values are derived according to the applicable method, but they are not national standards as such. The lower limit for soil (0.1 µg/kg dm) is the reporting limit, and is based on background concentrations in relatively unloaded areas. The upper limit (900 µg/kg dm) is the lowest value of the human maximum tolerable risk (MTR) and the Serious Risk level (SR) for the environment. The lower limit for groundwater is the generic target value, derived from the Negligible Risk to the environment (NR_{eco}); the upper limit for groundwater is the lower of the following values: $MTR_{human,\ groundwater}$, MTR_{DW} (safe value for drinking water for consumption) and $SR_{eco,\ groundwater}$ (Wintersen *et al.*, 2016).
The generic intervention value for residence with garden (900 µg/kg dm) takes into account the leaching of PFOA to groundwater used as drinking water. The generic value for 'other green spaces, buildings, infrastructure and industry' (1137 µg/kg dm) is a $SR_{eco,\ BM}$ that takes into account biomagnification (BM); this calculation is based on the assumption that areas with this function are large enough to serve as habitats for birds and mammals, whereby biomagnification to higher organisms can play a role. This is not assumed in the case of 'residence with a garden (Wintersen *et al.*, 2016).

In 2019, RIVM derived the following national risk limits for PFOA for a temporary framework for the application of soil and dredge spoil on or in soils: 7 µg/kg dm for agriculture, 89 µg/kg dm for residential and 1100 µg/kg dm for industry (RIVM, 2019). These are not real soil remediation values, but values that are used within the PFAS temporary action framework. For these three soil function classes it appears that biomagnification (ecology) determines the lowest risk limit value; this is because PFOS is mobile and accumulates in higher organisms (RIVM, 2019). The risk limit values based on the human health based maximum tolerable risk level (12.5 ng/kg bw/d) of Zeilmaker (2016) are higher (Table 68).
The proposed human health based SRV for agriculture (4.3 µg/kg dm) (Table 35) is slightly more than half of the Dutch risk limit $value_{eco}$ for agriculture (7 µg/kg dm).
The proposed ecological SRV for residences (89 µg/kg dm) is the same as the Dutch risk limit value for residences. The human health based SRV proposals for residences for tox scenario 1, 2 and 3 (205, 127 and 6.2 µg/kg dm) (Table 35) are respectively 2 and 1.5 times higher, and 20 times lower than the Dutch risk limit for residences (89 µg/kg dm).
The proposed human health based SRV for recreation and industry (643 µg/kg dm, adjusted for drinking water) (Table 35) is slightly more than half of the Dutch risk limit value for industry taking

into account biomagnification (1100 µg/kg dm). The Netherlands based the choice of an ecotoxicologically-underpinned intervention value over a health-related underpinned intervention value on an evaluation with the reference value of Zeilmaker (2016) and not with the proposed TDI from EFSA (2018). It is therefore only useful to compare the human health based SRV for scenario 2 with the Dutch human toxicological risk limit. The human health based SRV is lower than the Dutch risk $limit_{human}$ for landuse type agriculture, recreation (adjusted for drinking water) and industry (adjusted for drinking water), and higher for landuse type residence with vegetable garden (127 vs. 86 µg/kg dm).

Denmark has a quality criterion of 0.39 mg/kg soil as sum parameter (DEPA, 2015), but this value is based on the old RfD of US-EPA (0.03 µg/kg/d) and therefore in fact obsolete.
Australia has derived health-based guideline values for different landuse types: 100 µg/kg dm for residences with gardens, 20000 µg/kg dm for residences with minimum risk of soil contact and 10000 µg/kg dm for public areas (Australië, 2018).
The human health based screening value is 1260 µg/kg dm in the USA and 850 µg/kg dm in Canada; the screening value which takes groundwater protection into account is much lower (0.172 µg/kg dm in the USA).

Table 68 Foreign soil remediation values for soil and groundwater[a].

Soil				
Agriculture/nature	Risk limit $value_{eco}$ for the temporary framework for action	7.0 µg/kg dm	The Netherlands	Table 5.2 in RIVM (2019)[48]
Residence	Risk limit $value_{eco}$ for the temporary framework for action	89 µg/kg dm		
Industry	Risk limit $value_{eco}$ for the temporary framework for action	1100 µg/kg dm		
Agriculture/nature	Risk limit $value_{human}$	37000 µg/kg dm	The Netherlands	Table 3.2 in RIVM (2019)
Residence with vegetable garden	Risk limit $value_{human}$	86 µg/kg dm	The Netherlands	
Residence with garden	Risk limit $value_{human}$	1100 µg/kg dm	The Netherlands	
Industry/recreation[c]	Risk limit $value_{human}$	36500 µg/kg dm	The Netherlands	

48 https://www.rivm.nl/sites/default/files/2019-04/Notitie%20mogelijkheden%20onderbouwing%20landelijke%20normen%20PFAS_0.pdf

Upper limit (ad hoc intervention value)	$MTR_{human-soil}$	900 µg/kg dm	The Netherlands	Lijzen *et al.* (2018)
Lower limit (target value)	Reporting limit	0.1 µg/kg dm	The Netherlands	
Residence with garden	$MTR_{residence-garden}$	900 µg/kg dm	The Netherlands	
Residence with garden	Health based guideline value (HHSV)	100 µg/kg dm	Australia	Australië (2018)
Residence with minimal risk of soil contact	Health based screening value (HHSV)	20000 µg/kg dm		
Public area	Health based screening value (HHSV)	10000 µg/kg dm		
Other green spaces, buildings, infrastructure, industry	$SR_{eco,biomagnification}$	1137 µg/kg dm	The Netherlands	Lijzen *et al.* (2018)
Industry - commerce	Health based screening value (HHSV)	50000 µg/kg dm	Australia (2018)	Australië (2018)
Quality criterion	QC_{soil}	390 µg/kg dm	Denmark	DEPA 2015
Protection of groundwater	RSL	0.172 µg/kg dm	USA (2017)	Australië (2018)
Screening value human	RSL	1260 µg/kg dm		
Screening value human	SSV	850 µg/kg dm	Canada (2017)	
Groundwater				
Upper limit (intervention value)	MTR_{DW}	0.39 µg/l	The Netherlands	Lijzen *et al.* (2018)
Lower limit (target value)	-	Not determined	The Netherlands	
Residence with garden (Csoil)	Human risk limit	130 µg/l	The Netherlands	Alphenaar *et al.* (2018)
Residence with vegetable garden (Csoil)	Human risk limit	12 µg/l	The Netherlands	

[a] The values for the Netherlands are generic risk limits, not national soil remediation values
[b] Application of correction to standard soil is recommended
[c] No vegetable consumption, limited soil contact

ANNEX A: PFOS SUBSTANCE SHEET

Parameter	Unit	Value	Source
Name		Perfluorooctane sulfonic acid	
CAS number EC number		1763-23-1 217-179-8	
Type		organic	
Dissociative		no[1]	
Acid constant (pKa)		-3.27	Brooke *et al.* (2004)
Molar mass	g/mol	500,126	
Water solubility	mg/l	370 (K-salt)[2]	OECD (2002)
Vapour pressure	Pa	$3.31.10^{-4}$ (K-salt) (20°C)	OECD (2002)
Henry coefficient	Pa m^3/mol	-	Calculated in S-Risk
Log K_{ow}[49] K_{ow}	g/g	4.49 (calculated value)[3] 30902,95	EpiSuite
Log K_{oc} K_{oc}	dm^3/kg	2.57 (anion) 371.54	Higgins and Luthy (2006)
Log K_{oa}	g/g	-[4]	optional in S-Risk
BCF	(mg/kg dm)/(mg/m^3)	see table below	
Dpe	m^2/d	1.10^{-7} (standard value)	Based on Vonk (1985) and Lijzen *et al.* (2011)
Dpvc	m^2/d	1.10^{-10} (Dpe/1000)	Cornelis *et al.* (2017)
Diffusion for organic substance in air (Da)	m^2/d	-	Calculated in S-Risk
Diffusion for organic substance in water (Dw)	m^2/d	-	Calculated in S-Risk
Kp	[cm/h]	$9.5.10^{-7}$ (AFPO)	Washburn *et al.* (2005)
FA	-	1	Cornelis *et al.* (2017)
ABS dermal soil/dust	-	0	Xiao *et al.* (2015)
BTF beef	d/kg	0.071	Vestergren *et al.* (2013)

[49] Entered in S-Risk but not used in further calculations

Parameter	Unit	Value	Source
BTF sheepmeat	d/kg	0.387	Kowalczyk et al 2012
BTF liver	d/kg	0.441	Vestergren *et al.* (2013)
BTF kidney	d/kg	1.201	(1) Kowalczyk *et al.* (2013)
BTF milk	d/kg	0.021	Vestergren *et al.* (2013)
BTF soil – egg	d/kg		
BTF food - egg	d/kg		
Carcinogenicity		Carc. 2	EG (2008)
Systemic effects threshold [5]			
TDI oral	mg/kg.d	2.10^{-5}	Preferred scenario US-EPA (2016)
TCA inhalatory	[mg/m³]	7.10^{-5}	calculated from TDI oral
TDI dermal	mg/kg.d	2.10^{-5}	= TDI oral
smoothing - ages		child, adolescent, adult	
Limit in air	mg/m³	-	
Limit in drinking water	mg/m³	0.1	EC (2018)
Crop standard	mg/kg fw		
Meat standard			
Beef	mg/kg fw		
Sheepmeat	mg/kg fw		
Liver	mg/kg fw		
Kidney	mg/kg fw		
Milk	mg/kg fw		
Butter	mg/kg fw		
Egg	mg/kg fw		
Dietary background all age groups including children	mg/kg day	$1.2.10^{-6}$ (1 - < 3 y)	Extrapolation based on EFSA (2012) Lower bound
		$1.2.10^{-6}$ (3 - < 6 y)	
		$1.08.10^{-6}$ (6 - < 10 y)	
		$0.513.10^{-6}$ (10 - < 15 y)	
		$0.562.10^{-6}$ (15 - < 21 y)	
		$0.634.10^{-6}$ (21 - < 31 y)	
		$0.875.10^{-6}$ (≥ 31 y)	
Background potato	mg/kg fw	$3.60.10^{-6}$	EFSA (2012) LB
Background root vegetables	mg/kg fw	$9.50.10^{-6}$	EFSA (2012) LB

Parameter	Unit	Value	Source
Background bulbous vegetables (onion, etc.)	mg/kg fw	$2.20.10^{-6}$	EFSA (2012) LB
Background fruiting vegetables	mg/kg fw	$2.10.10^{-6}$	EFSA (2012) LB
Background cabbage	mg/kg fw	$1.20.10^{-6}$	EFSA (2012) LB
Background leafy vegetables	mg/kg fw	6.10^{-7}	EFSA (2012) LB
Background legumes	mg/kg fw	0	EFSA (2012) LB
Background beef	mg/kg fw	$8.60.10^{-6}$	EFSA (2012) LB
Background offal	mg/kg fw	$4.20.10^{-4}$	EFSA (2012) LB
Background milk	mg/kg fw	$9.00.10^{-7}$	EFSA (2012) LB
Background butter	mg/kg fw	$8.2.10^{-4}$	EFSA (2012) LB (Assimilated to *animal fat*)
Background eggs	mg/kg fw	$3.7.10^{-5}$	EFSA (2012) LB
Background outdoor air	mg/m³	$1.4.10^{-9}$	P50 value from Cornelis *et al.* (2009)
Background indoor air	mg/m³	$1.6.10^{-9}$	Jahnke *et al.* (2007b) in Cornelis *et al.* (2009)
Background drinking water	mg/m³	0	Assimilated to zero since it is included in the intake estimation of EFSA (2012)

(1) in S-Risk 'no' is entered because the Kd of dissociative substances is calculated from log K_{ow}, which we want to avoid; for non-dissociative substances the Kd is calculated from the K_{oc}

(2) The value of 370 mg/l is given in OECD (2002) with reference to a 3M report from 1999, without mention of temperature. The OECD test protocol for solubility (OECD test guideline 105) states that the test should preferably be carried out at 20 ± 0.5°C. As such, 20°C is used in S-Risk.

(3) Log K_{ow} is mandatory in S-Risk, and is used to calculate Kp, K_{oc}, and transfer factors, unless an experimental value is entered. Experimental values are available for these three parameters.

(4) Log K_{oa} is optional in S-Risk, which uses K_{oa} in the calculation of transfer to plants; as experimental data are available for this purpose, a K_{oa} value is not necessary.

[5] Due to the ongoing discussions on the new proposed TDI of EFSA, scenarios with 3 different sets of toxicological reference values will be calculated. The three sets are in the table below.

Toxicological reference value	Value	Unit	Reference
Set 1 (preference)			
TDI oral	2.10^{-5}	mg/kg/d	US-EPA (2016c)
TCA inhalation	7.10^{-5}	mg/m³	calculated from TDI oral
TDI dermal	2.10^{-5}	mg/kg/d	= TDI oral
Set 2			
TDI oral	$6.25.10^{-6}$	mg/kg/d	Zeilmaker et al. (2018)
TCA inhalation	$21.9.10^{-6}$	mg/m³	calculated from TDI oral
TDI dermal	$6.25.10^{-6}$	mg/kg/d	= TDI oral
Set 3			
TDI oral	$1.8.10^{-6}$	mg/kg/d	EFSA (2018c)
TCA inhalation	$6.3.10^{-6}$	mg/m³	calculated from TDI oral
TDI dermal	$1.8.10^{-6}$	mg/kg/d	= TDI oral

The RfD of US-EPA (2016c) of **20 ng/kg bw/d** is proposed as a toxicological reference value for the calculation of the soil remediation value based on the following arguments:

- experts recognise that the current standard of EFSA is too high
- the more stringent EFSA standard is still provisional
- the Dutch MTR is more protective than the current TDI of EFSA, but is likely to be reviewed when EFSA publishes its final (more stringent) TDI
- the MRL of ATSDR is still provisional
- the RfD is based on a long-term study
- the value of the RfD is the same as that of Australia and New Zealand

the derivations of US-EPA and Australia/New Zealand are recent

Plant	BCF or BCF model
potatoes	
potatoes	**0.01**
root and tuber vegetables	
carrots	**0.50**
salsify	0.44 (= average known root and tuber vegetables)
other root vegetables (such as radish)	**0.38**
bulbous vegetables	
bulbous vegetables (such as onion)	0.44 (= average known root and tuber vegetables)
leek	0.44 (= average known root and tuber vegetables)
fruiting vegetables	
tomato	**0.06**
cucumber	**0.07**
other fruiting vegetables (such as peppers)	0.065 (average known fruiting vegetables)
cabbages	
cabbage	0.44 (= average known root and tuber vegetables)
cauliflower and broccoli	0.44 (= average known root and tuber vegetables)
sprouts	0.44 (= average known root and tuber vegetables)
leafy vegetables	
lettuce	**0.56**
lamb's lettuce	0.56 (= lettuce)
endive	0.62 (average lettuce and celery)
spinach	**3.77**
chicory	0.62 (average lettuce and celery)
celery	**0.72**
legumes	
beans	0.03 (= peas)
peas	**0.03**
grasses	
grass	**0.048**
cereals	
maize	**0.003**

ANNEX B: PFOA SUBSTANCE SHEET

Parameter	Unit	Value	Source
Name		Perfluorooctanoic acid	
CAS number EC number		335-67-1 206-397-9	
Type		organic	
Dissociative		no[(1)]	
Acid constant (pKa)		2.8	Moody and Field (2000)
Molar mass	g/mol	414,07	
Water solubility	mg/l	$9.5.10^3$ (25°C)	ECHA (2014)
Vapour pressure	Pa	$1.7.10^{-2}$ (10°C)	Lijzen *et al.* (2018)
Henry coefficient	Pa m³/mol	-	Calculated in S-Risk
Log K_{ow}[50] K_{ow}	g/g	4.81 (calculated value)[(2)] 64565,42	EpiSuite
Log K_{oc} K_{oc}	dm³/kg	2.06 114.82	Higgins and Luthy (2006)
Log K_{oa}	g/g	-[(3)]	optional in S-Risk
BCF	(mg/kg dm)/(mg/m³)	See table below	
Dpe	m²/d	1.10^{-7} (standard value)	Vonk (1985); Lijzen *et al.* (2018)
Dpvc	m²/d	1.10^{-10} (Dpe/1000)	Cornelis *et al.* (2017)
Diffusion for organic substance in air (Da)	m²/d	-	Calculated in S-Risk
Diffusion for organic substance in water (Dw)	m²/d	-	Calculated in S-Risk
Kp	[cm/h]	$9.49.10^{-7}$	Fasano *et al.* (2005)
FA	-	1	Cornelis *et al.* (2017)
ABS dermal soil/dust	-	0	Xiao *et al.* (2015)
BTF beef	d/kg	$5,999.10^{-3}$	Vestergren, 2013 and Kowalczyk *et al.* (2013)
BTF sheepmeat	d/kg	$6,950.10^{-3}$	Vestergren, 2013 and Kowalczyk *et al.* (2013)
BTF liver	d/kg	$8,756.10^{-3}$	Vestergren, 2013 and Kowalczyk *et al.* (2013)

[50] Entered in S-Risk but not used in further calculations

Parameter	Unit	Value	Source
BTF kidney	d/kg	$1{,}945.10^{-3}$	Vestergren, 2013 and Kowalczyk *et al.* (2013)
BTF milk	d/kg	$5{,}686.10^{-3}$	Vestergren, 2013 and Kowalczyk *et al.* (2013)
BTF soil – egg	d/kg		
BTF food - egg	d/kg		
Carcinogenicity		Carc. 2	EC (2008)
Systemic effects threshold [4]			
TDI oral	mg/kg.d	2.10^{-5}	US-EPA (2016)
TCA inhalatory	mg/m³	7.10^{-5}	calculated from TDI oral
TDI dermal	mg/kg.d	2.10^{-5}	= TDI oral
smoothing - ages		child, adolescent, adult	
Systemic effects without threshold			
Slope factor oral	$(mg/kg/d)^{-1}$	0.07[5]	US-EPA (2016d)
Unit risk	$(mg/m^3)^{-1}$	-	
Slope factor dermal	$(mg/kg/d)^{-1}$	-	
Smoothing duration		lifelong	
Limit in air	mg/m³	-	
Limit in drinking water	mg/m³	0.1	EC (2018)
Crop standard	mg/kg fw		
Meat standard			
Beef	mg/kg fw		
Sheepmeat	mg/kg fw		
Liver	mg/kg fw		
Kidney	mg/kg fw		
Milk	mg/kg fw		
Butter	mg/kg fw		
Egg	mg/kg fw		
Dietary background all age groups including children	mg/kg day	$2.20.10^{-7}$ (1 - < 3 y)	Extrapolation based on EFSA (2012) Lower bound
		$1.98.10^{-7}$ (3 - < 6 y)	
		$1.62.10^{-7}$ (6 - < 10 y)	
		$1.08.10^{-7}$ (10 - < 15 y)	
		$0.924.10^{-7}$ (15 - < 21 y)	
		$0.98.10^{-7}$ (21 - < 31 y)	
		$1.11.10^{-7}$ (≥ 31 y)	
Background potato	mg/kg fw	$9.00.10^{-7}$	EFSA (2012) LB

Parameter	Unit	Value	Source
Background root vegetables	mg/kg fw	$3.4.10^{-6}$	EFSA (2012) LB
Background bulbous vegetables (onion, etc.)	mg/kg fw	$2.2.10^{-6}$	EFSA (2012) LB
Background fruiting vegetables	mg/kg fw	$4.5.10^{-6}$	EFSA (2012) LB
Background cabbage	mg/kg fw	$1.9.10^{-6}$	EFSA (2012) LB
Background leafy vegetables	mg/kg fw	$6.2.10^{-6}$	EFSA (2012) LB
Background legumes	mg/kg fw	$2.5.10^{-5}$	EFSA (2012) LB
Background beef	mg/kg fw	$6.1.10^{-6}$	EFSA (2012) LB
Background offal	mg/kg fw	$3.4.10^{-5}$	EFSA (2012) LB
Background milk	mg/kg fw	0	EFSA (2012) LB
Background butter	mg/kg fw	$1.7.10^{-6}$	EFSA (2012) LB assimilated to *animal fat*
Background eggs	mg/kg fw	$8.8.10^{-5}$	EFSA (2012) LB
Background outdoor air	mg/m^3	$8.90.10^{-9}$	Cornelis *et al.* (2009)
Background indoor air	mg/m^3	$8.90.10^{-9}$	Assimilated to outdoor air
Background drinking water	mg/m^3	0	Assimilated to 0 since it is included in the intake estimation of EFSA (2012)

(1) in S-Risk 'no' is entered because the Kd of dissociative substances is calculated from log K_{ow}, which we want to avoid; for non-dissociative substances the Kd is calculated from the K_{oc}

(2) Log K_{ow} is mandatory in S-Risk, and is used to calculate Kp, K_{oc}, and transfer factors, unless an experimental value is entered. Experimental values are available for these three parameters.

(3) Log K_{oa} is optional in S-Risk, which uses K_{oa} in the calculation of transfer to plants; as experimental data are available for this purpose, a K_{oa} value is not necessary.

(4) Due to the ongoing discussions on the new proposed TDI of EFSA, scenarios with 3 different sets of toxicological reference values will be calculated. The three sets are in the table below.

(5) The slope factor corresponds with a dose of $1.43.10^{-4}$ mg/kg bw/d or 143 ng/kg bw/d for an additional cancer risk of $1/10^5$. This value is higher than the toxicological reference value (20 ng/kg bw/d) used for the derivation of the soil remediation values. Hence a soil remediation value based on carcinogenic effects was not derived as it could be expected to be higher than for non-carcinogenic effects. This statement is in line with US-EPA who confirmed that the lifetime health advisory (2.10^{-5} mg/kg bw/d) based on non-cancer effects is protective for the cancer endpoint (US-EPA, 2016b).

Toxicological reference value	Value	Unit	Reference
Set 1 (preference)			
TDI oral	2.10^{-5}	mg/kg/d	US-EPA (2016c)

TCA inhalation	7.10^{-5}	mg/m³	calculated from TDI oral
TDI dermal	2.10^{-5}	mg/kg/d	= TDI oral
Set 2			
TDI oral	$12.5.10^{-6}$	mg/kg/d	Zeilmaker et al. (2016)
TCA inhalation	$43.8.10^{-6}$	mg/m³	calculated from TDI oral
TDI dermal	$12.5.10^{-6}$	mg/kg/d	= TDI oral
Set 3			
TDI oral	$0.8.10^{-6}$	mg/kg/d	EFSA (2018c)
TCA inhalation	$2.8.10^{-6}$	mg/m³	calculated from TDI oral
TDI dermal	$0.8.10^{-6}$	mg/kg/d	= TDI oral

The RfD of US-EPA (2016c) of **20 ng/kg bw/d** is proposed as a toxicological reference value for the calculation of the soil remediation value based on the following arguments:

- experts recognise that the current standard of EFSA is too high
- the more stringent EFSA standard is still provisional
- the Dutch MTR is more protective than the current TDI of EFSA, but is likely to be reviewed when EFSA publishes its final (more stringent) TDI
- the MRL of ATSDR is still provisional
- the RfD is based on a long-term study
- the value of the RfD is the same as that of Australia and New Zealand
  the derivations of US-EPA and Australia/New Zealand are recent

BCF values PFOA

Plant	BCF or BCF model
potatoes	
potatoes	**0.06**
root and tuber vegetables	
carrots	**0.39**
salsify	0.55 (average value of known root and tuber vegetables)
other root vegetables (such as radish)	**0.70**
bulbous vegetables	0.55 (= average known root and tuber vegetables)
bulbous vegetables (such as onion)	0.55 (= average known root and tuber vegetables)
leek	0.55 (= average known root and tuber vegetables)
fruiting vegetables	
tomato	**0.81**
cucumber	**0.82**
other fruiting vegetables (such as peppers)	0.81 (=tomato)
Cabbages	
Cabbage	0.55 (= average known root and tuber vegetables)
cauliflower and broccoli	0.55 (= average known root and tuber vegetables)
sprouts	0.55 (= average known root and tuber vegetables)
leafy vegetables	
Lettuce	**1.90**
lamb's lettuce	1.90 (=sla)
endive	1.06 (= average of all known leafy vegetables)
spinach	**0.87**
chicory	1.06 (= average of all known leafy vegetables)
celery	**0.42**
legumes	
beans	0.03 (= peas)
peas	**0.03**
Grasses	
Grass	**0.128**
Cereals	
Maize	**0.005**

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