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Risk assessment of exposure to **PFAS** through food and drinking water in the Netherlands

RIVM report 2023-0011

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Synopsis

Risk assessment of exposure to PFAS through food and drinking water in the Netherlands

RIVM has calculated the quantity of per- and polyfluoroalkyl substances (PFAS) that people ingest through food and drinking water. The study shows that the calculated quantity of PFAS ingested through food and drinking water is above the health-based guidance value. If the amount of PFAS people ingest exceeds this guidance value over a long period of time, it could be harmful to their health.

Results also show that people in the Netherlands ingest more than three times as much PFAS through food as through drinking water. Fish is an important source of PFAS that people can ingest through food as fish can contain high concentrations of PFAS. People also ingest PFAS through tea, coffee, cereal products, milk products, meat, eggs, fruits and vegetables.

The PFAS concentration in drinking water depends on the type of water used as a source. People ingest a lower amount of PFAS through drinking water made from groundwater than through drinking water made from surface water. This is because surface water contains a higher level of PFAS than groundwater. Out of the twenty PFAS studied, people mostly ingest PFUnDA, PFOS and PFDA because these PFAS are found in high concentrations in fish.

The calculation is an update of a previous estimate based on data about PFAS in food from 2009. RIVM used new information about PFAS in food and drinking water from 2021 and 2022 for the updated calculation. It also used information about twenty PFAS, instead of four. Although more PFAS were considered, the ingested quantity is around 40 per cent lower than previously calculated.

PFAS are man-made substances and thus do not occur naturally in the environment. They are found in many different products, such as non-stick coatings, food packaging materials and clothing. PFAS can end up in the air, water and soil both when they are made and when people use PFAS-containing products. From there, they enter our food and drinking water. Most PFAS do not degrade and therefore remain in the environment for a long time.

Eating a varied diet is important to avoid ingesting a large amount of PFAS. This way, people will not eat foods with a high PFAS concentration too often.

Keywords: PFAS, food, drinking water, ingestion, exposure, risk assessment

Publiekssamenvatting

Risicobeoordeling van blootstelling aan PFAS via voedsel en drinkwater in Nederland

Het RIVM heeft berekend hoeveel per- en polyfluoralkylstoffen (PFAS) mensen via voedsel en drinkwater binnenkrijgen. Hieruit blijkt dat de hoeveelheid PFAS die mensen via voedsel en drinkwater kunnen binnenkrijgen boven de zogeheten gezondheidskundige grenswaarde ligt. Als mensen lange tijd meer PFAS binnenkrijgen dan deze gezondheidskundige grenswaarde, zijn schadelijke effecten op de gezondheid mogelijk.

Ook blijkt dat mensen in Nederland via voedsel meer dan drie keer zoveel PFAS binnenkrijgen als via drinkwater. Vis is een belangrijke bron van PFAS via voedsel, omdat er veel van deze stoffen in vis kunnen zitten. Daarnaast krijgen we PFAS binnen via koffie, thee, graanproducten, melkproducten, vlees, eieren, fruit en groenten.

Bij drinkwater hangt de hoeveelheid PFAS af van het soort water waarvan het is gemaakt. Via drinkwater dat van grondwater is gemaakt, krijgen we minder PFAS binnen dan via drinkwater uit oppervlaktewater. Dat komt doordat er in oppervlaktewater meer PFAS zit dan in grondwater. Van de twintig onderzochte typen PFAS krijgen we vooral PFUnDA, PFOS en PFDA binnen, omdat deze typen veel in vis zitten.

De berekening is een update van een eerdere schatting met gegevens over PFAS in voedsel uit 2009. Voor de update is nieuwe informatie over voedsel en drinkwater uit 2021 en 2022 gebruikt. Ook is er informatie over twintig in plaats van vier typen PFAS meegenomen. Hoewel er nu meer PFAS zijn meegenomen, is de hoeveelheid PFAS die mensen binnen kunnen krijgen ongeveer 40 procent lager dan eerder was berekend.

PFAS zijn stoffen die door de mens zijn gemaakt en komen van nature niet in het milieu voor. Deze stoffen zitten in veel verschillende producten, zoals antiaanbaklagen, verpakkingsmaterialen voor voedsel en in kleding. Bij het proces om PFAS te maken en het gebruik van producten waar ze in zitten, kan PFAS in de lucht, het water en de bodem terecht komen. Vandaaruit komen ze in ons voedsel en drinkwater. De meeste PFAS breken niet af en blijven daardoor lang in het milieu zitten.

Om te voorkomen dat je te veel PFAS binnenkrijgt, is het belangrijk om gevarieerd te eten. Op die manier eet je niet te vaak voedsel met een hoge hoeveelheid PFAS.

Kernwoorden: PFAS, voedsel, drinkwater, inname, blootstelling, risicobeoordeling

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1 Introduction

Per- and polyfluoroalkyl substances (PFAS) is the collective name for a vast group of fluorinated substances. PFAS are man-made substances that do not occur naturally. Due to their useful properties, i.e. water-, grease- and/or dirt-repellent, PFAS can be found in various products, including lubricants, food packaging materials, extinguishing foam, non-stick coatings on pans, clothing, textiles and cosmetics. They are also used in many industrial applications and processes. Due to PFAS emissions during production, and the disposal of PFAS-containing products, PFAS have ended up in the environment. Certain PFAS are extremely persistent, leading to accumulation in the environment, and consequently also in animals and humans after exposure. Food is the major source of PFAS exposure for humans. Contamination of food with PFAS is mainly due to bioaccumulation in the food chain and transfer from contact materials used in food processing and packaging (EFSA, 2020).

Epidemiological studies in humans showed that long-term dietary exposure to PFAS is associated with adverse health effects, such as liver damage, reduced birth weight and a decreased immune response. Of these effects, the effect on the immune system was observed in humans at the lowest levels of exposure to PFAS. Hence, the European Food Safety Authority (EFSA) derived a health-based guidance value for this effect, i.e. a tolerable weekly intake (TWI) of 4.4 nanogram (ng)/kilogram (kg) body weight. This TWI was derived for the sum of four PFAS, namely perfluorooctanoic acid (PFOA), perfluorononanoic acid (PFNA), perfluorohexane sulfonic acid (PFHxS) and perfluorooctane sulfonic acid (PFOS), the so-called EFSA-4. People with an exposure to PFAS exceeding the TWI over a long period of time may have a decreased immune response. Hence, they may be more vulnerable to infections. And potentially, their vaccine response could decrease.

The TWI derived by EFSA is the starting point for risk assessments of PFAS. However, humans can also be exposed to other PFAS besides the EFSA-4 through food and drinking water (EFSA, 2020; Noorlander et al., 2011). Therefore, RIVM decided to use the relative potency factor (RPF) approach when assessing the risks of exposure to PFAS. This approach allows to sum the exposure to two or more PFAS by expressing the toxicological potency of each PFAS relative to the potency of an index PFAS (Zeilmaker et al., 2018; RIVM, 2021).

In 2020, EFSA concluded that the dietary exposure to the EFSA-4 in parts of the European population exceeded the TWI (EFSA, 2020). In 2021, RIVM calculated the exposure to the EFSA-4 through food and drinking water for the Dutch population using the RPF approach, and likewise concluded that the exposure exceeded the TWI (van der Aa et al., 2021). However, RIVM noted that the calculated exposure for the Dutch population may not have been accurate, because it was based on old PFAS concentrations in food products sampled in 2009 and only included the EFSA-4.

In this report, the exposure to PFAS in the Netherlands through food and drinking water was calculated more accurately than in 2021. To this end, a broad range of food products was sampled in 2021 and analysed for a wider range of PFAS than the EFSA-4. Using these PFAS concentrations, together with recent concentrations of PFAS in drinking water and food consumption data from the Dutch food consumption survey of 2012-2016 (van Rossum et al., 2020), the exposure to PFAS in the Netherlands was calculated using the RPF approach (see above). The exposure was compared with the TWI. In addition, these new calculations provide information on which food groups and which individual PFAS contribute most to the exposure.

People may also be exposed to PFAS through other sources than food and drinking water, such as exposure through inhaled air and ingested dust. Exposure through these sources will not be addressed in this report.

This study was conducted as part of a three-year PFAS-research programme (2023-2025) to gain insight into the current exposure to PFAS in the Netherlands and the sources that contribute to this exposure, including food and drinking water. This insight will be used to establish which measures could effectively reduce the exposure to PFAS.

2 Sampling and chemical analysis of food and drinking water

2.1 Food

Food products were sampled and analysed for PFAS to calculate the exposure to PFAS. Section 2.1.1 describes the sampling of the food products, and the analysis of PFAS in these food products is described in section 2.1.2.

2.1.1 Sampling

Food products for the analysis of PFAS were selected on the basis of the consumption pattern of the whole Dutch population as reported in the Dutch National Food Consumption Survey (DNFCS) of 2012-2016 (van Rossum et al., 2020). Food products were selected on the basis of their high relative contribution to the consumption of all products belonging to the same food group. Since consumption of small amounts of highly contaminated foods may also contribute to the exposure, some food products were selected, because they were expected to contain high concentrations of PFAS (e.g. pork liver and fish). In total, 54 food products were selected for analysis (see Table 1).

In November and December 2021, the Wageningen Food Safety Research (WFSR) research institute at Wageningen University & Research (WUR) bought the selected food products. Two to fifteen samples per food product were purchased from different supermarkets, specialist shops (such as fishmongers and greengrocers), and local markets. Food products were obtained from four different supermarkets, which covered 74% of the sales via supermarkets in the Netherlands in 2020.[1] In addition, different brands of industrially prepared food products were purchased.

All food product samples were handled separately (i.e. samples were not pooled per food product before analysis). Some food products were pre-processed before analysis, if considered relevant (see Table 1). In addition, some fruits and vegetables were washed (see Table 1). None of the food products were cooked. Drinks, dairy, eggs, flour, sugar, and vegetable fats and oils were only homogenised before analysis, while the other food products were also ground. PFAS-free materials were used for these procedures. Finally, all food products were stored at -20°C before analysis.

Table 1 provides the list of food products included in the study, information on pre-processing, and the number of analysed samples per food product. The food products were listed according to their corresponding main food group (see the grey cells in Table 1) and subgroups (if applicable). The total number of analysed samples was 440. Please note that when referring to food in this report, this also includes bottled natural mineral water.

[1] https://www.supermarkt.team/nielsen-marktaandelen-2020-t-o-v-2019/

Table 1 Overview of food products, listed according to their corresponding main food group (see the grey cells) and subgroups (if applicable), including the pre-processing applied, and the number of analysed samples per food product

Subgroup (if applicable)	Sampled food products	Pre-processing (if relevant)	Number of analysed samples
Vegetables[a]			
Root and tuber vegetables	Beetroots (including vacuum packed)	Peeling	5
	Carrots	Peeling	5
	Potatoes	Peeling	15
Leafy vegetables	Crisp lettuces	Washing	5
	Curly endives	Washing	5
	Lettuces, excluding crisp lettuces	Washing	5
	Spinaches (including frozen)	Washing	10
	Belgian endives	Washing	10
Brassica vegetables	Broccoli	Removing stem	5
	Cauliflowers	Removing stem and outer leaves	5
Bulb vegetables	Onions	Peeling	12
Legumes	French beans	Removing ends and washing	10
	Garden peas (frozen)	N.a.	4
Fruiting vegetables	Cucumbers	N.a.	9
	Sweet peppers	Deseeding and removing stem	11
	Tomatoes and cherry tomatoes	Removing stem and washing	10
Stem vegetables	Leeks	Cutting off the green part and washing	11
Fungi	Mushrooms	Removing soil	10

Subgroup (if applicable)	Sampled food products	Pre-processing (if relevant)	Number of analysed samples
Processed vegetables			
	Beans, canned or jarred	Draining	4
	French fries, pre-baked (including frozen)	N.a.	3
	Peas, canned or jarred	Draining	2
	Sweet corn, canned	Draining	9

Subgroup (if applicable)	Sampled food products	Pre-processing (if relevant)	Number of analysed samples
Fruits (and nuts)[a]			
Berries and small fruits	Grapes	Removing stem and washing	10
	Strawberries (including frozen)	Removing crown and washing	10
Pome fruits	Apples	Peeling and removing stem and core	5
	Pears	Peeling and removing stem and core	5
Miscellaneous fruits with inedible peel	Bananas	Peeling	10
Citrus fruits	Mandarins	Peeling	5
	Oranges	Peeling	5

Subgroup (if applicable)	Sampled food products	Pre-processing (if relevant)	Number of analysed samples
Cereals and cereal products			
	Bread, wheat and whole grains	N.a.	11
	Rice grains, polished	N.a.	10
	Wheat flour	N.a.	11

Subgroup (if applicable)	Sampled food products	Pre-processing (if relevant)	Number of analysed samples
Vegetable fats and oils			
	Margarine, blended	N.a.	6
	Margarine, traditional	N.a.	5
	Olive oil, classic and extra virgin	N.a.	11
	Peanut butter and peanut sauce	N.a.	9
	Sunflower oil	N.a.	10

Subgroup (if applicable)	Sampled food products	Pre-processing (if relevant)	Number of analysed samples
Fish and fish products[a]			
	Cod (including frozen)	N.a.	10
	Fish fingers, pollack (including frozen)	N.a.	10

Subgroup (if applicable)	Sampled food products	Pre-processing (if relevant)	Number of analysed samples
Fish and fish products[a]			
	Pangasius and tilapia (including frozen)	N.a.	8
	Salmon (including canned and smoked)	Draining (when canned)	11
	Tuna (including canned and frozen)	Draining (when canned)	9

Subgroup (if applicable)	Sampled food products	Pre-processing (if relevant)	Number of analysed samples
Meat and meat products[a]			
	Beef	N.a.	2
	Chicken	N.a.	13
	Minced meat, beef with/without pork	N.a.	12
	Pâté, pork liver	N.a.	5
	Pork	N.a.	8
	Pork liver-type sausages	N.a.	5

Subgroup (if applicable)	Sampled food products	Pre-processing (if relevant)	Number of analysed samples
Drinks			
	Coffee	Prepared by pouring[b]	12
	Natural mineral water, bottled	N.a.	7
	Tea	Prepared by steeping for two minutes	13

Subgroup (if applicable)	Sampled food products	Pre-processing (if relevant)	Number of analysed samples
Dairy			
	Milk, cow, semi-skimmed	N.a.	8

Subgroup (if applicable)	Sampled food products	Pre-processing (if relevant)	Number of analysed samples
Eggs			
	Eggs, chicken	Removing shell	9

Subgroup (if applicable)	Sampled food products	Pre-processing (if relevant)	Number of analysed samples
Sugar			
	Sugar	N.a.	10
Total			**440**

N.a.: not applicable
[a] Food products in this food group were sampled fresh, unless stated otherwise.
[b] Coffee was prepared by pouring boiled water (from the boiling water tap) into a filter containing ground coffee.

2.1.2 Chemical analysis of PFAS

The food products were analysed by WFSR for 17 PFAS that were included in the analytical method (see Table 2).[2]

Table 2 Overview of analysed PFAS in food products

PFAS	PFAS abbreviation
Sulfonic acids	
Perfluorobutane sulfonate	PFBS
Perfluorohexane sulfonate	PFHxS
Perfluoroheptane sulfonate	PFHpS
Perfluorooctane sulfonic acid	PFOS
Perfluorodecane sulfonate	PFDS

PFAS	PFAS abbreviation
Carboxylic acids	
Perfluorobutanoic acid	PFBA
Perfluoropentanoic acid	PFPeA
Perfluorohexanoic acid	PFHxA
Perfluoroheptanoic acid	PFHpA
Perfluorooctanoic acid	PFOA
Perfluorononanoic acid	PFNA
Perfluorodecanoic acid	PFDA
Perfluoroundecanoic acid	PFUnDA
Perfluorododecanoic acid	PFDoDA
Perfluorotridecanoic acid	PFTrDA
Perfluorotetradecanoic acid	PFTeDA

PFAS	PFAS abbreviation
Ether carboxylic acids	
Hexafluoropropylene oxide dimer acid	HFPO-DA (GenX)

PFAS: per- and polyfluoroalkyl substances

PFAS were analysed according to Standard Operating Procedure (SOP) A1114 of WFSR. To egg and milk samples, 0.62 M lead acetate was added to precipitate proteins. To samples of fruits, vegetables, meat, fish, cereals, and vegetable fats and oils, 200 mM sodium hydroxide was added to destruct organic material. Subsequently, a methanol extraction in acid medium was conducted, after which samples were cleaned up by solid phase extraction. Extracts from fruits and vegetables samples were redissolved in methanol, and the other extracts were dissolved in a mixture of 45% acetonitrile and 55% 20 mM ammonium acetate. After

[2] WFSR, Resultaten PFAS onderzoek in voedingsmiddelen, 2217163/WFSR

addition of the injection standard, samples were analysed by liquid chromatography with tandem mass spectrometry (LC-MS/MS).[3] The LC-column was a Phenomenex Luna Omega 1.6u PS C18 100A (100 x 2,1 µm) with a Phenomenex Gemini C18 column (50 x 3 mm; 5 µm) isolator column. The two mobile phases of the LC-MS/MS for the fruit and vegetable samples were 20 mM ammonium acetate in water and 100% methanol (0.5 mL/min). The two mobile phases for the other samples were 200 mM ammonium acetate in water and 100% acetonitrile (0.5 mL/min).

PFAS concentrations were quantified by means of matrix-based calibration curves.[4] For this, the food products were divided into different matrix types on the basis of their characteristics. Most PFAS occur in food as linear isomers, but some may also be present as a mixture of linear and branched isomers (i.e. PFOS, PFOA and PFHxS). PFAS were quantified by means of a reference standard for the linear isomer. The concentrations of the branched isomers of only PFOS were quantified by means of the linear PFOS isomer due to lack of standards for branched PFOS isomers at the time of analysis. A small inaccuracy may result from this in the reported PFOS concentrations (linear + branched), but this approach was assumed to be the most accurate to quantify the concentrations of PFOS. For the other PFAS, the concentrations of the linear isomers were reported.

The LC-MS/MS operated in multiple reaction monitoring, which includes monitoring of each individual PFAS at two ion transitions, except for PFBA and PFPeA for which only one ion transition was monitored. The identity of a PFAS was determined by calculating the ratio of these two ion transitions, which should be similar to the ratio of the respective PFAS reference standard. When the second ion transition was not or insufficiently visible, while the first ion transition showed presence of the PFAS, this ratio was too low. Thus, the identity of the particular PFAS could not be confirmed, even if it may appear from the first ion transition that it concerned the respective PFAS. In that case, the presence of the PFAS was indicated as below one of two analytical limits, depending on the quantity:

- Limit of detection (LOD): the lowest quantity at which the PFAS could be detected, but not quantified.
- Limit of quantification (LOQ): the lowest quantity at which the PFAS could be quantified with acceptable precision and accuracy.

The lowest quantity at which the PFAS could be quantified and at which its identity could be confirmed was the limit of confirmation (LOC). When the quantity of the PFAS in a sample was between the LOQ and the LOC, the identity of the PFAS could also not be confirmed with 100% certainty, but it was deemed highly unlikely that it would not concern the respective PFAS. The LOD was lower than the LOQ, and the LOQ was lower or equal to the LOC (LOD < LOQ ≤ LOC). Example peaks of ion transitions belonging to the three analytical limits are shown in Appendix A.

[3] LC-MS/MS, LC: Shimazdu ExionLC AD; MS: Sciex 7500
[4] The matrix contains all substances of a sample, except for the substance to be analysed.

The LOD varied from 0.05-90 picogram (pg) per gram and the LOQ and LOC from 0.1-180 pg per gram, depending on the PFAS and matrix type. Since not all food products belong to the same matrix, these limits differed between the various food products.[5] Variation in analytical limits can also be explained by the specific PFAS, since not every PFAS behaves similarly in LC-MS/MS due to their different physical/chemical properties. Generally, higher analytical limits were obtained for fish, vegetable fats and oils, cereals, eggs, and meat. This was mainly due to matrix interferences. For PFOA and PFUnDA, this was the result of a contamination, which could occur unintentionally.

2.2 Drinking water

To calculate the exposure to PFAS, PFAS concentrations in drinking water were obtained from all ten Dutch drinking water companies, which cover the whole of the Netherlands. Section 2.2.1 describes the sampling of drinking water, and the analysis of PFAS in drinking water is described in section 2.2.2.

2.2.1 Sampling

Drinking water was sampled in 2022 and a total of 777 samples were taken. PFAS concentrations are known to differ between drinking water produced from groundwater and drinking water produced from surface water (van der Aa et al., 2021). Therefore, the drinking water samples were divided into two drinking water types based on the water source as in van der Aa et al. (2021):

- Drinking water produced from phreatic groundwater and (semi) confined groundwater were categorised as drinking water produced from groundwater; and
- Drinking water produced from surface water, infiltrated surface water and riverbank filtrate were categorised as drinking water produced from surface water.

The 777 samples comprised 316 samples of drinking water produced from groundwater and 461 samples of drinking water produced from surface water.

2.2.2 Chemical analysis of PFAS

In 2022, drinking water samples were analysed for 38 PFAS by the laboratories of the following Dutch drinking water companies: Vitens, *Het Waterlaboratorium* (HWL), *Waterlaboratorium Noord* (WLN) and *Aqualab Zuid*. A few samples were analysed by WFSR and *Vrije Universiteit Amsterdam*. The laboratories are accredited by the Dutch Accreditation Council for the internationally recognized standard NEN-EN-ISO/IEC 17025. Additionally, NEN 7777 and NEN 7779 are followed, which cover interlaboratory procedures to determine performance characteristics of analytical methods such as repeatability, specificity, LOD and LOQ.

The analysis of PFAS in drinking water is not yet accredited as the drinking water limit for PFAS will not come into force until 12 January 2026. The analytical methods for PFAS analysis in drinking water are

[5] Samples of a food product were sometimes analysed in different runs, resulting also in differences in analytical limits between samples of the same food product.

still under development and differ between laboratories. All methods include acidification of the drinking water samples, followed by addition of the reference material (for the analysis of the linear isomers) and direct injection on LC-MS/MS. Aqualab Zuid, HWL and WLN have a description of their analytical methods on their websites.[6] For Vitens, the method description is not publicly available, but was shared for reference with RIVM.

Of the 38 analysed PFAS, 20 PFAS were included in the exposure calculations. No information was available on the human health relevance for the remaining 18 (i.e. no relative potency factors have been derived for these PFAS; see section 3.4). The 20 PFAS included the 17 PFAS that were analysed in food products (see Table 2), as well as 3H-perfluoro-3-[(3-methoxy-propoxy)propanoic acid] (ADONA), perfluoropentanesulfonic acid (PFPeS) and trifluoroacetic acid (TFA). By including these three PFAS in the assessment, the exposure to PFAS could be calculated most accurately on the basis of the available concentration data in food and drinking water.

Not all 20 PFAS were analysed in each drinking water sample. Table 3 presents the number of analysed samples per PFAS and drinking water type. Also, in drinking water, some PFAS can occur as a mixture of linear and branched isomers. The concentrations of the linear isomers were reported for all PFAS. For PFOS, PFOA and PFHxS, the concentrations of the branched isomers were also reported in part of the samples (see Table 3). The concentrations of the branched isomers were quantified by means of the linear isomer due to lack of standards for branched isomers at the time of analysis. As in the food products, this may have resulted in a small inaccuracy of the reported concentrations for PFOS, PFOA and PFHxS.

The LOQ varied from 0.1-25 nanogram (ng) per litre, depending on the PFAS and drinking water type. The LOQ reported by the laboratories was comparable to the LOC for food products, i.e. the lowest quantity at which the PFAS could be quantified and at which its presence was confirmed. To ensure uniformity between the different laboratories, no LODs were reported. LOCs were likewise not reported, because this limit is not part of the NEN 7777 guidelines.

[6] *Aqualab Zuid*: https://www.aqualab.nl/pdc-v243; *Waterlaboratorium Noord*: https://wln.nl/kwaliteit/ME.pfc-lcNL.html; *Het Waterlaboratorium*: https://www.hetwaterlaboratorium.nl/over-ons/actueel/alles-uit-de-kast-voor-pfas-analyse,

Table 3 Overview of analysed PFAS in samples of drinking water produced from groundwater and from surface water as included in the exposure calculations

PFAS	PFAS abbreviation	Number of analysed samples per drinking water type	
		Groundwater (n = 316)	Surface water (n = 461)
Sulfonic acids			
Perfluorobutane sulfonate	PFBS	311	407
Perfluorohexane sulfonate	PFHxS[a]	311	404
Perfluoroheptane sulfonate	PFHpS	311	407
Perfluorooctane sulfonic acid	PFOS[b]	259	412
Perfluorodecane sulfonate	PFDS	311	404
Perfluoropentanesulfonic acid	PFPeS[c]	309	404

PFAS	PFAS abbreviation	Number of analysed samples per drinking water type	
		Groundwater (n = 316)	Surface water (n = 461)
Carboxylic acids			
Perfluorobutanoic acid	PFBA	310	397
Perfluoropentanoic acid	PFPeA	310	404
Perfluorohexanoic acid	PFHxA	310	404
Perfluoroheptanoic acid	PFHpA	311	407
Perfluorooctanoic acid	PFOA[d]	309	407
Perfluorononanoic acid	PFNA	311	407
Perfluorodecanoic acid	PFDA	311	407
Perfluoroundecanoic acid	PFUnDA	310	407
Perfluorododecanoic acid	PFDoDA	310	404
Perfluorotridecanoic acid	PFTrDA	309	401
Perfluorotetradecanoic acid	PFTeDA	195	223

PFAS	PFAS abbreviation	Number of analysed samples per drinking water type	
		Groundwater (n = 316)	Surface water (n = 461)
Ether carboxylic acids			
Hexafluoropropylene oxide dimer acid	HFPO-DA (GenX)	311	407
Dodecafluoro-3H-4,8-Dioxanon-anoic Acid/ 3H-perfluoro-3-[(3-methoxy-propoxy)propanoic acid]	DONA/ADONA[c]	291	376

PFAS	PFAS abbreviation	Number of analysed samples per drinking water type	
		Groundwater (n = 316)	Surface water (n = 461)
Ultra-short-chain PFAS			
Trifluoroacetic acid	TFA[c]	5	147

n: number; PFAS: per- and polyfluoroalkyl substances
[a] 36% of the 715 samples were analysed for both linear and branched PFHxS.
[b] 73% of the 671 samples were analysed for both linear and branched PFOS.
[c] These PFAS were not analysed in food products (see Table 2 in section 2.1.2).
[d] 35% of the 716 samples were analysed for both linear and branched PFOA.

3 Methodology of the exposure calculations

To calculate the exposure to PFAS through food and drinking water, the measured PFAS concentrations in the food products and drinking water were combined with Dutch food consumption data. The exposure was calculated using the relative potency factor (RPF) approach.

3.1 Food consumption data

The exposure calculations were performed by means of food consumption data collected in the DNFCS of 2012-2016 (van Rossum et al., 2020). Food consumption data was obtained by means of two non-consecutive 24-hour dietary recalls in accordance with the EFSA Guidance for the collection of national food consumption data (EFSA, 2009). For participants aged 1-8 and 71-79 years, this information was combined with a food diary. The survey provided information on the amount of food and drinks consumed on two days by 4313 Dutch children and adults aged 1-79 years.

3.2 Concentration data

3.2.1 Food

Appendix B provides an overview of the individual PFAS concentrations in the food product samples. Concentration data was reported in five ways (see also section 2.1.2):

- < LOD: the PFAS could not be detected;
- < LOQ: the PFAS was detected, but the exact concentration could not be quantified, and its identity could not be confirmed;
- < LOC: the concentration of the PFAS could be quantified and its identity could not be confirmed, but was highly likely;
- c (numerical concentration; ≥ LOC): the specific PFAS concentration could be quantified and the identity of the PFAS was confirmed; and
- n.d. (not determined): the PFAS could not be determined due to a high background signal (this was different from a PFAS that could not be detected, i.e. reported as < LOD; see section 4.1).

PFAS concentrations that were reported as 'c' were included in the exposure calculations as such, and PFAS that could not be determined in any or some of the samples were not included in those samples. Concentrations reported below their analytical limits (i.e. LOD, LOQ or LOC) were included according to two scenarios: a lower bound (LB) scenario and an upper bound (UB) scenario. These two scenarios reflect the most optimistic and conservative assumptions, respectively, about the presence of a PFAS reported at a concentration below these limits. Table 4 shows the allocated concentrations for each reported concentration and scenario. When the reported concentration of a PFAS was below the LOD or the LOQ, the LB concentration was assumed to be 0 pg per gram, because the identity of the PFAS could not be confirmed, and in the most optimistic scenario the PFAS was not present in the sample. When the concentration was reported as below the LOC, the LB concentration was assumed to equal the LOQ (and not 0 pg per gram, because it was deemed highly likely that it would concern the respective

PFAS (see section 2.1.2)). In the UB scenario, the PFAS concentrations reported below their analytical limits were assumed to equal these limits. Furthermore, a concentration reported as below the LOC was interpreted as below the LOQ, when the corresponding LOQ was equal to the LOC.

Table 4 PFAS concentrations allocated to the reported concentrations in food products in a lower bound and upper bound scenario

Reported concentration	Scenario	
	Lower bound	Upper bound
< LOD	0	LOD
< LOQ	0	LOQ
< LOC	LOQ	LOC
c	c	c
n.d.	Not included	Not included

c: numerical concentration; LOC: limit of confirmation; LOD: limit of detection; LOQ: limit of quantification; n.d.: not determined; PFAS: per- and polyfluoroalkyl substances

3.2.2 Drinking water

An overview of the concentration data for 20 PFAS in Dutch drinking water in 2022 is shown in Appendix C. Concentration data was reported in three ways (see also section 2.2.2):

- < LOQ: the PFAS was analysed, but the exact concentration could not be quantified, the concentration is between 0 and the LOQ;
- c (numerical concentration; ≥ LOQ): the PFAS concentration could be quantified; and
- n.a. (not analysed): the PFAS was not analysed in the specific sample.

No analytical data for drinking water was reported as it was not determined (n.d.).

Also, the reported PFAS concentration data for drinking water was included in the exposure assessment according to a LB and UB scenario. Table 5 shows the allocated concentrations for each reported concentration and scenario.

Table 5 PFAS concentrations allocated to the reported concentrations in drinking water in a lower bound and an upper bound scenario

Reported concentration	Scenario	
	Lower bound	Upper bound
< LOQ	0	LOQ
c	c	c
n.a.	Not included	Not included

c: numerical concentration; LOQ: limit of quantification; n.a.: not analysed; PFAS: per- and polyfluoroalkyl substances

The calculated LB and UB exposure show the range of the possible exposure to PFAS, considering the uncertainties about the PFAS concentrations in the samples with a reported concentration below an analytical limit. It is expected that the LB and UB exposure will underestimate and overestimate the actual dietary exposure to PFAS, respectively (see Box 1).

Box 1: *Concentrations according to a lower bound (LB) and an upper bound (UB) scenario*

LB PFAS concentrations will **underestimate** the true concentrations, since it cannot be excluded that the non-detected (< LOD) and non-quantified (< LOQ) PFAS are in reality present in the samples, and that the non-confirmed (< LOC) PFAS are present at a higher concentration than the LOQ.

UB PFAS concentrations will **overestimate** the true concentrations, since it is unlikely that all non-detected, all non-quantified and all non-confirmed PFAS are present in the samples at a concentration equal to the relevant analytical limit.

Because PFAS concentrations differ between drinking water produced from groundwater and drinking water produced from surface water (see section 2.2.1), the exposure to PFAS was calculated separately for these two types of drinking water. As a result, the exposure to PFAS through food and drinking water was calculated for two scenarios and for two drinking water types: LB-groundwater, UB-groundwater, LB-surface water and UB-surface water.

3.3 Allocating PFAS concentrations to consumed foods and drinking water

To assess the exposure to PFAS, the PFAS concentrations in the sampled food products and drinking water were allocated to consumed foods and drinks recorded in the DNFCS. This allocation is described in section 3.3.1 for food and in section 3.3.2 for drinking water.

3.3.1 Allocation to consumed foods

PFAS concentrations in the sampled food products were directly allocated to identical foods reported in the DNFCS. However, such a direct allocation was not always possible, because not all reported foods were sampled. In addition, most of the sampled food products consisted of one ingredient (e.g. fruits, vegetables, and milk; see Table 1 in section 2.1.1), whereas many reported foods are composite foods consisting of at least two ingredients (e.g. pizzas and salads). Furthermore, foods may be processed before consumption. For example, cooking vegetables may result in a weight decrease due to water loss, and someone reporting the consumption of 100 grams of cooked spinach, may actually have eaten 167 grams of raw spinach. Not addressing these various aspects will result in an underestimation of the exposure.

To avoid a potential underestimation, PFAS concentrations in the sampled food products were allocated to similar non-sampled food products (see section 3.3.1.1). Furthermore, the concentrations in the sampled food products and similar non-sampled foods products were used to calculate PFAS concentrations in composite foods and in processed counterparts of the sampled and non-sampled food products by using a food conversion model (see section 3.3.1.2).

3.3.1.1 Allocating PFAS concentrations to similar non-sampled food products

PFAS concentrations of sampled fruits (and nuts) and vegetables were allocated to similar non-sampled fruits and vegetables by first classifying them according to the categorisation of Annex I of the pesticide residues Regulation (EU) 396/2005. Each of these food products belong to a certain subgroup and a corresponding broader main food group (see Table 1 in section 2.1.1). For instance, oranges belong to the subgroup 'citrus fruit', which in turn belongs to the main food group 'fruits (and nuts)'. PFAS concentrations of sampled fruits and vegetables were allocated to non-sampled fruits and vegetables within the same subgroup. For example, PFAS concentrations in oranges and mandarins were assigned to lemons. When no food product had been sampled within a subgroup, PFAS concentrations of all sampled food products belonging to the corresponding main food group were allocated to the non-sampled food product. For example, PFAS concentrations of all fruits of the main food group 'fruits (and nuts)' were allocated to cherries as no stone fruits had been sampled.

PFAS concentrations in other sampled food products were allocated to similar non-sampled food products as shown in Table 6. Some sampled food products (e.g. canned salmon) were not linked to non-sampled food products, because no relevant similar non-sampled food product could be identified.

Table 6 Allocation of PFAS concentrations from sampled food products to non-sampled food products

Sampled food products	**Allocated to**	**Non-sampled food products**
Beans, canned or jarred	→	Processed and/or dry beans
Peas, canned or jarred	→	Processed and/or dry peas
Bread, wheat and whole grain	→	Bread and rolls
Wheat flour	→	Cereals (excluding rice) and flour
Peanut butter and peanut sauce	→	Peanuts and peanut oil
Margarine, traditional	→	Cooking and frying fat
Olive oil, margarine (traditional and blended), peanut butter and peanut sauce, sunflower oil	→	Vegetable fats and oils
Cod, pangasius and tilapia, salmon (only fresh), tuna (only frozen and fresh)	→	Fish and fish products (including crustaceans, such as shrimps)
Chicken meat	→	Poultry meat
Minced meat (beef with/without pork), chicken, pork, beef	→	Mixed fresh meat
Minced meat (beef without pork), beef	→	Cow, ox or bull fresh meat
Pork liver-type sausages	→	Liver-type sausages
Coffee (prepared)	→	Coffee beans[a]

Sampled food products	Allocated to	Non-sampled food products
	→	Coffee beverages[b]
Tea (prepared)	→	Tea leaves[a]
Milk, cow, semi-skimmed	→	Milk
Eggs, chicken	→	Eggs
Sugar	→	Glucose syrup

[a] A factor of 22 was applied to the PFAS concentrations in prepared coffee and a factor of 120 to the PFAS concentrations in prepared tea to obtain the concentrations in coffee beans and tea leaves, respectively. This approach was taken, because the consumed amount of prepared coffee and tea could be reported in the Dutch National Food Consumption Survey separately as the consumed amount of coffee beans and tea leaves, respectively, and that of drinking water.
[b] For example, café Americano and cappuccino.

3.3.1.2 Calculation of PFAS concentrations in processed food products and composite foods

PFAS concentrations in composite foods (e.g. apple pie) and in processed counterparts of the sampled and non-sampled food products (e.g. cooked spinach) were calculated using the Dutch food conversion model (van Dooren et al., 1995). This model was developed in 1995 and has been updated with every new DNFCS to include new foods. The food conversion model describes all reported foods in the DNFCS (such as apple pie) in terms of mass percentages of their raw ingredients (e.g. wheat, apple, grape and egg for apple pie). It thereby also accounts for changes in food weight due to processing.

To calculate the PFAS concentrations in composite foods, sampled and non-sampled food products were first linked to relevant raw ingredients listed in the food conversion model. The PFAS concentration in a composite food was then calculated by multiplying the mean PFAS concentrations per raw ingredient with its mass percentage and summing them across ingredients (see Box 2 for an example). This resulted in a mean PFAS concentration per composite food (see also section 3.4).

Box 2: *Example of calculating a PFAS concentration in 'red cabbage with apple pieces, jarred'*

In the food conversion model, 'red cabbage with apple pieces, jarred' consists for 98% of raw red cabbage, 5% of apple, and 10% of sugar. The mean concentrations of the individual PFAS in brassica vegetables, apple and sugar were multiplied with the relevant mass percentages and summed to obtain the mean concentration of the respective PFAS in 'red cabbage with apple pieces, jarred'.

PFAS concentration in a processed food product was calculated by linking it to the relevant sampled or non-sampled food product and correcting the concentration for possible weight loss due to processing (see Box 3 for an example).

Box 3: *Example of calculating a PFAS concentration in 'cooked spinach'*

In the food conversion model, 'cooked spinach' consists for 167% of raw spinach. The mean concentrations of the individual PFAS in raw spinach were multiplied with 1.67 to obtain the mean concentration of the respective PFAS in 'cooked spinach'.

3.3.2 Allocation to consumed drinking water

PFAS concentrations analysed in drinking water were allocated to drinks containing drinking water as an ingredient in the food conversion model (see section 3.3.1.2), such as lemonade, or directly to the reported consumption of drinking water.

PFAS were analysed in prepared coffee and tea (see Table 1 in section 2.1.1). These drinks were prepared with drinking water available in Wageningen, which is produced from groundwater. The reported PFAS concentrations may thus not reflect the PFAS concentrations in these drinks prepared with drinking water from other areas in the Netherlands. Therefore, the PFAS concentrations in prepared coffee and tea were recalculated to consider the PFAS concentrations in drinking water produced from groundwater and from surface water as provided by the Dutch drinking water companies (see section 2.2).

The concentration data provided by the Dutch drinking water companies showed that drinking water from the area of Wageningen did not contain any PFAS at or above the LOQ in 2022.[7] Therefore, all quantified PFAS (≥ LOC) in the prepared coffee and tea samples were assumed to be derived from the coffee beans and tea leaves, the materials used to prepare these drinks (e.g. coffee filter, tea bag, glassware used or boiling water tap), and/or other unknown sources of contamination. To obtain concentrations in prepared coffee (including coffee beverages, see Table 6) and tea as reported in the DNFCS, the PFAS concentrations in drinking water from groundwater and from surface water as provided by the Dutch drinking water companies were added to the PFAS concentrations in the prepared coffee and tea samples analysed by WFSR.

In conclusion, all relevant reported foods and drinks in the DNFCS were assigned a PFAS concentration, which constituted the input for the exposure calculation.

3.4 Calculation of summed PFAS concentrations

In 2020, EFSA defined a TWI of 4.4 ng/kg body weight for the sum of PFOA, PFNA, PFHxS and PFOS, the so-called EFSA-4. This TWI was based on the observed critical effect on the immune system after long-term exposure to PFAS (see chapter 1). As people are expected to be exposed to more PFAS than the EFSA-4, and these other PFAS also have an effect on the immune system (Bil et al., 2023), more PFAS than the EFSA-4 should be considered in the risk assessment. Furthermore, EFSA assumed that the EFSA-4 are equally harmful to the immune system

[7] The water, used by WFSR to prepare the coffee and tea samples, was analysed for PFAS as validation, and also showed no PFAS at or above the LOQ (based on personal communication).

(i.e. equipotent), while differences in potency for immune effects exist (Bil et al., 2023).

To address these two observations, RIVM decided to use the relative potency factor (RPF) approach in the risk assessment of PFAS (RIVM, 2021). Using this approach, the ability of each PFAS to cause an effect on the immune system is expressed relative to that of the 'index' PFAS PFOA, resulting in an RPF for each PFAS. For example, an RPF of 2 means that this PFAS is twice as potent to cause the effect as PFOA. Table 7 lists the RPFs of the twenty analysed PFAS, of which nineteen RPFs were derived by Bil et al. (2021), and one RPF (for TFA) by RIVM (2023).

Table 7 Overview of analysed PFAS and their relative potency factor

PFAS[a]	Relative potency factor[b]
Sulfonic acids	
PFBS	0.001
PFPeS[d]	0.6
PFHxS[c]	0.6
PFHpS	2
PFOS[c]	2
PFDS	2

PFAS[a]	Relative potency factor[b]
Carboxylic acids	
PFBA	0.05
PFPeA	0.05
PFHxA	0.01
PFHpA	1
PFOA[c]	1
PFNA[c]	10
PFDA	10
PFUnDA	4
PFDoDA	3
PFTrDA	3
PFTeDA	0.3

PFAS[a]	Relative potency factor[b]
Ether carboxylic acids	
ADONA[d]	0.03
HFPO-DA (GenX)	0.06

PFAS[a]	Relative potency factor[b]
Ultra-short-chain PFAS	
TFA[d]	0.002

PFAS: per- and polyfluoroalkyl substances
[a] The names of the PFAS can be found in Table 3 in section 2.2.2.
[b] Relative potency factors (RPFs) as derived by Bil et al. (2021) and RIVM (2023). The RPFs for PFDA, PFHpA, PFHpS, PFPeA, PFPeS and PFTrDA were derived as a range by Bil et al. (2021). In these cases, the highest RPF of the range was used to calculate the summed concentrations as recommended by RIVM (2021).
[c] PFAS belonging to the EFSA-4.
[d] PFAS that were only analysed in drinking water.

Using these RPFs, individual PFAS concentrations in a sample are expressed as PFOA equivalents (PEQ), and consequently summed into one summed concentration of PFAS expressed as PEQ per sample. Co-occurrence of the PFAS concentrations in a sample is preserved in this way. These summed concentrations are subsequently used to calculate the exposure to PFAS expressed as PEQ. Box 4 provides an example of how a summed PFAS concentration could be calculated using RPFs. The RPFs are assumed to be applicable to both the linear and branched isomers of a PFAS. A more extensive explanation on the risk assessment of PFAS is provided in RIVM (2021).

Box 4: *Example of a calculation of a lower bound summed PFAS concentration using RPFs*

A fictitious sample contains PFOA, PFHxA and PFOS at 0.05, 1.0 and 0.01 pg per gram. The other PFAS are reported at a concentration below the LOD. PFOA is the reference compound. The RPFs are 1 for PFOA, 0.01 for PFHxA and 2 for PFOS.

The summed concentration in PFOA equivalents (PEQ) for this fictitious sample according to the lower bound scenario is: (0.05 x 1) + (1.0 x 0.01) + (0.01 x 2) = 0.08 pg PEQ per gram. The PFAS with a reported concentration below the LOD are assumed not to be present in the sample (0 pg per gram) in this scenario.

To calculate the summed concentrations of PFAS, expressed as PEQ, in food product samples, the reported concentrations for each individual PFAS were first imputed according to the LB and UB scenario (see Table 4 in section 3.2.1). Subsequently, the individual PFAS concentrations were multiplied by their respective RPFs (see Table 7), and the concentrations expressed as PEQ were summed to obtain the LB and UB summed PFAS concentrations for each sample.

A different approach was taken for the drinking water samples, because the number of analysed PFAS per sample varied (ranging from 1 PFAS up to 20 PFAS per sample; see section 2.2.2). First, the concentrations for each individual PFAS were imputed according to the LB and UB scenario (see Table 5 in section 3.2.2), and a mean concentration of each PFAS was then calculated for both scenarios and both drinking water types. These mean concentrations were multiplied by the relevant RPF, and subsequently summed to obtain one mean LB and one mean UB summed PFAS concentration expressed as PEQ for drinking water from groundwater and for drinking water from surface water.

The summed PFAS concentrations were used to calculate the LB and UB exposure to PFAS (see section 3.5).

3.5 Exposure calculation

PFAS are associated with adverse effects on the immune system that occur after a long period of exposure. Therefore, the long-term dietary exposure to PFAS was calculated and the relevant population was the total age group of 1-79 years, as available in the DNFCS (see

section 3.1). In this report, the exposure to PFAS refers to the summed exposure to all PFAS considered, expressed as PEQ.

To calculate the long-term exposure to PFAS, the Observed Individual Mean (OIM) model was used as implemented in the Monte Carlo Risk Assessment (MCRA) software (version 9.2) that is available for registered users (https://mcra.rivm.nl/). The OIM model is the same model that EFSA applied to calculate long-term dietary exposure to PFAS in Europe (EFSA, 2020).

Using the OIM model, daily consumption patterns of the foods (including drinks), recorded by the 4313 individuals in the DNFCS, were multiplied by the mean summed PFAS concentrations expressed as PEQ per consumed food, and summed over the foods per day per individual. Mean summed concentrations were used, because differences in concentrations within foods are expected to level out in the long run. This resulted in 8626 daily total PFAS exposure levels, because for each individual two days were recorded in the DNFCS (see section 3.1). Because long-term exposure to PFAS is of interest, the daily exposure levels were averaged over these two days for each individual. Subsequently, these average daily exposure levels were divided by the individual's body weight to express the exposure in PEQ per kg body weight per day. These daily exposure levels were multiplied by seven for comparison with the TWI of the EFSA-4, which is expressed per week. The resulting distribution of 4313 average individual weekly exposure levels of PFAS expressed as PEQ per kg body weight was characterised by calculating the mean, median (50^{th} percentile; P50) and high (95^{th} percentile; P95) long-term exposure. The exposure distribution was calculated for the LB and UB scenario (see Tables 4 and 5 in section 3.2) and for two drinking water types. This resulted in the calculation of twelve exposure parameters: two (LB and UB) x two (groundwater and surface water) x three (mean, median and high).

The contribution of each food group and drinking water type, and that of each individual PFAS to the exposure distribution were also calculated. For the latter, the exposure to each individual PFAS, expressed as PEQ, was calculated.

To quantify the uncertainty in the exposure parameters due to the sampling size of the food consumption and concentration databases, the bootstrap approach was used (Efron, 1979; Efron and Tibshirani, 1993). For this, 100 food consumption databases and 100 concentration databases were generated by resampling with replacement from the original database.[8] Thus, a total of 100 two-day mean exposure distributions were generated. The differences between these distributions reflect the uncertainty around the true distribution of exposure. The mean, median and P95 were calculated on the basis of each distribution. The uncertainty was reported as the 95% confidence interval around these exposure parameters. Regarding the contributions, the mean contribution of the 100 distributions was reported. The

[8] With this method 100 bootstrap databases are generated of the same size as the original database for both the food consumption and concentration database by sampling with replacement from the original datasets. These bootstrap databases are considered as databases that could have been obtained from the original population if another sample was randomly drawn.

uncertainty in the exposure parameters due to the concentrations of PFAS in drinking water from groundwater and from surface water could not be considered, because only one mean concentration for the two drinking water types was available in the concentration databases (see section 3.4).

4 Results

4.1 Missing concentrations of PFAS in food products

In total, 17 PFAS were analysed in the food products (see section 2.1). However, PFBA could not be determined in any of the food products due to a high background signal. For the same reason, other PFAS could not be determined in some food products or in some samples of a food product (see Table 8). HFPO-DA (GenX) and PFNA could be determined in all food products. If a PFAS could not be determined, this resulted in a missing concentration for that PFAS in a sample. This was not the same as for a PFAS that could not be detected (i.e. reported at a concentration below the LOD).

Table 8 Overview of the food products or samples of a food product in which the respective PFAS could not be determined[a], resulting in missing concentrations

PFAS[b]	Food products or samples of a food product in which the respective PFAS could not be determined
Sulfonic acids	
PFBS	Onions, leeks, and a subset of the samples of some vegetables
PFHxS[c]	One sample of rice grains
PFHpS	Onions, leeks, potatoes, French fries, and a subset of the samples of some vegetables
PFOS[c]	Onions and leeks
PFDS	A subset of the samples of some fruits and vegetables

PFAS[b]	Food products or samples of a food product in which the respective PFAS could not be determined
Carboxylic acids	
PFBA	All food products
PFPeA	Approximately 40% of the samples, i.e. not in vegetables (except for potatoes), prepared coffee, bread, sweet corn, canned or jarred beans and peas, and in 50% of the milk samples
PFHxA	Almost 20% of the samples, i.e. not in bread, potatoes, French fries, sweet corn, and a subset of the samples of some vegetables
PFHpA	Bread
PFOA[c]	Bread
PFNA[c]	N.a.
PFDA	Approximately 10% of the samples, i.e. not in onions, leeks and most leafy vegetables
PFUnDA	Approximately 30% of the samples, i.e. not in cereal products, meat, most vegetable fats and oils, and a small subset of the samples of some vegetables
PFDoDA	A subset of the samples of some vegetables
PFTrDA	Approximately 35% of the samples, i.e. not in vegetables (except for peas, onions and leeks) and a subset of the samples of some fruits

PFAS[b]	Food products or samples of a food product in which the respective PFAS could not be determined
Carboxylic acids	
PFTeDA	Approximately 40% of the samples, i.e. not in vegetables, sweet corn, canned or jarred beans or peas, French fries, a subset of the samples of some fruits and eggs

PFAS[b]	Food products or samples of a food product in which the respective PFAS could not be determined
Ether carboxylic acids	
HFPO-DA (GenX)	N.a.

N.a.: not applicable; PFAS: per- and polyfluoroalkyl substances
[a] Some PFAS could not be determined due to a high background signal or because of their chemical properties
[b] The names of the PFAS can be found in Table 2 in section 2.1.2.
[c] PFAS belonging to the EFSA-4 (see section 3.4).

The effect of these missing concentrations on the calculated exposure is discussed in section 5.1.

4.2 Summed concentrations of PFAS

The summed PFAS concentration in each sampled food product was calculated by summing the concentration of each individual PFAS, expressed as PEQ, according to the LB and UB scenario (see sections 3.2 and 3.4). PFAS that could not be determined were excluded in the summed concentration (see Table 4 in section 3.2.1). Since PFBA could not be determined in any of the food products, the summed concentrations in the food product samples were based on a maximum of 16 PFAS. In approximately 75% of these samples, the summed PFAS concentration was based on fewer than 16 PFAS. For example, seven PFAS could not be determined in a sample of leeks, and therefore, the concentration of this sample was based on ten PFAS. For drinking water, one mean summed PFAS concentration was calculated for each scenario and per drinking water type, because not all twenty PFAS analysed in drinking water were analysed in all samples (see section 3.4).

Appendix D provides an overview of the LB and UB summed concentrations per sampled food product, expressed as PEQ. In Table 9, the mean LB and UB summed concentrations of each food product and both drinking water types are listed.

Table 9 Overview of the mean lower bound (LB) and upper bound (UB) summed PFAS concentrations, expressed as PEQ, per food product and the two drinking water types (the main food group is mentioned in the grey cells)

Subgroup (if applicable)	**Food products (n of samples; n of determined PFAS)[a] and drinking water**	**Mean summed PFAS concentration for two scenarios in pg PEQ per gram[b,c]**	
		LB	**UB**
Vegetables			
Root and tuber vegetables	Beetroots (5; 13)	5.2	47
	Carrots (5; 13)	2.6	46
	Potatoes (15; 12)	0.91	18
Leafy vegetables	Crisp lettuces (5; 12)	3.8	28
	Curly endives (5; 12)	23	46
	Lettuces, excluding crisp lettuces (5; 12)	50	71
	Spinaches (10; 12)	30	53
	Belgian endives (10; 9-11)	5.7	50
Brassica vegetables	Broccoli (5; 9-11)	21	82
	Cauliflowers (5; 9-11)	0.005	42
Bulb vegetables	Onions (12; 10)	8.2	47
Legumes	French beans (10; 9-11)	12	57
	Garden peas (4; 15)	0.79	22
Fruiting vegetables	Cucumbers (9; 9-11)	0.00089	42
	Sweet peppers (11; 9-11)	2.9	46
	Tomatoes and cherry tomatoes (10; 9-11)	0.43	45
Stem vegetables	Leeks (11; 10)	3.4	46
Fungi	Mushrooms (10; 9-11)	38	82

Subgroup (if applicable)	**Food products (n of samples; n of determined PFAS)[a] and drinking water**	**Mean summed PFAS concentration for two scenarios in pg PEQ per gram[b,c]**	
		LB	**UB**
Processed vegetables			
	Beans, canned or jarred (4; 15)	0.010	20
	French fries (3; 12)	0.35	19
	Peas, canned or jarred (2; 15)	7.2	30
	Sweet corn, canned (9; 11)	0	45

Subgroup (if applicable)	**Food products (n of samples; n of determined PFAS)[a] and drinking water**	**Mean summed PFAS concentration for two scenarios in pg PEQ per gram[b,c]**	
		LB	**UB**
Fruits (and nuts)			
Berries and small fruits	Grapes (10; 13-16)	1.3	35
	Strawberries (10; 13-16)	4.0	45
Pome fruits	Apples (5; 13-16)	2.4	44

Subgroup (if applicable)	Food products (n of samples; n of determined PFAS)[a] and drinking water	Mean summed PFAS concentration for two scenarios in pg PEQ per gram[b,c]	
		LB	UB
Fruits (and nuts)			
	Pears (5; 13-16)	1.5	36
Miscellaneous fruits with inedible peel	Bananas (10; 13-16)	2.7	47
Citrus fruits	Mandarins (5; 13-16)	0.48	42
	Oranges (5; 13)	11	42

Subgroup (if applicable)	Food products (n of samples; n of determined PFAS)[a] and drinking water	Mean summed PFAS concentration for two scenarios in pg PEQ per gram[b,c]	
		LB	UB
Cereals and cereal products			
	Bread (11; 11)	0.38	59
	Rice grains (10; 14-15)	4.3	83
	Wheat flour (11; 15)	6.2	86

Subgroup (if applicable)	Food products (n of samples; n of determined PFAS)[a] and drinking water	Mean summed PFAS concentration for two scenarios in pg PEQ per gram[b,c]	
		LB	UB
Vegetable fats and oils			
	Margarine, blended (6; 15-16)	0	102
	Margarine, traditional (5; 15-16)	0	99
	Olive oil (11; 15-16)	0	98
	Peanut butter and peanut sauce (9; 15-16)	16	124
	Sunflower oil (10; 15)	3.1	91

Subgroup (if applicable)	Food products (n of samples; n of determined PFAS)[a] and drinking water	Mean summed PFAS concentration for two scenarios in pg PEQ per gram[b,c]	
		LB	UB
Fish and fish products			
	Cod (10; 16)	1843	2737
	Fish fingers (10; 16)	613	1251
	Pangasius and tilapia (8; 16)	21	743
	Salmon (4; 16)[d]	17	745
	Salmon, canned (4; 16)[d]	870	1578
	Salmon, smoked (3; 16)[d]	21	746
	Tuna (9; 16)[e]	586	1309

Subgroup (if applicable)	Food products (n of samples; n of determined PFAS)[a] and drinking water	Mean summed PFAS concentration for two scenarios in pg PEQ per gram[b,c]	
		LB	UB
Meat and meat products			
	Beef (2; 15)	40	189
	Chicken (13; 15)	4.4	155
	Minced meat (12; 15)	20	173
	Pâté (5; 15)	60	238
	Pork (8; 15)	30	179
	Pork liver-type sausages (5; 15)	64	221

Subgroup (if applicable)	Food products (n of samples; n of determined PFAS)[a] and drinking water	Mean summed PFAS concentration for two scenarios in pg PEQ per gram[b,c]	
		LB	UB
Drinks			
	Coffee (12; 15)[f]	17	48
	Drinking water[g] from - Groundwater - Surface water	 1.5 9.2	 50 27
	Natural mineral water, bottled (7; 16)	0.42	3.4
	Tea (13; 16)[f]	16	44

Subgroup (if applicable)	Food products (n of samples; n of determined PFAS)[a] and drinking water	Mean summed PFAS concentration for two scenarios in pg PEQ per gram[b,c]	
		LB	UB
Dairy			
	Milk (8; 15-16)	19	72

Subgroup (if applicable)	Food products (n of samples; n of determined PFAS)[a] and drinking water	Mean summed PFAS concentration for two scenarios in pg PEQ per gram[b,c]	
		LB	UB
Eggs			
	Eggs (9; 15-16)	78	241

Subgroup (if applicable)	Food products (n of samples; n of determined PFAS)[a] and drinking water	Mean summed PFAS concentration for two scenarios in pg PEQ per gram[b,c]	
		LB	UB
Sugar			
	Sugar (10; 16)	0	29

LB: lower bound; n: number; PEQ: PFOA equivalents; PFAS: per- and polyfluoroalkyl substances; pg: picogram; UB: upper bound

[a] In case of a range, the number of determined PFAS differed between samples of the same product.
[b] Allocation of PFAS concentrations according to the LB and UB scenario is explained in section 3.2.
[c] Summed PFAS concentrations in drinks were expressed as pg PEQ per gram by converting the results expressed as ng per litre and assuming a density of 1 gram per millilitre for water.
[d] Since the summed PFAS concentrations differed between raw, canned and smoked salmon, the results of these food products are presented separately.
[e] Summed PFAS concentrations were similar between fresh, canned and frozen tuna, and therefore, these results are not presented separately.
[f] The results for coffee and tea refer to the prepared coffee and tea analysed by WFSR (see Table 1 in section 2.1.1). In the exposure calculations, the PFAS concentrations in these samples were combined with the PFAS concentrations in drinking water from groundwater and from surface water (see section 3.3.2).
[g] The sampling and analysis of the drinking water samples differed from the food samples (see sections 2.2.1 and 2.2.2).

The mean LB summed PFAS concentrations varied from 0 pg PEQ per gram in canned sweet corn, olive oil, traditional margarine, blended margarine and sugar (the reported PFAS concentrations in these food products were all below the LOD or the LOQ) to 1843 pg PEQ per gram in cod. Clearly, fish contained the highest mean LB summed PFAS concentrations. Not only cod, but also canned salmon (870 pg PEQ per gram), fish fingers (613 pg PEQ per gram) and tuna (586 pg PEQ per gram) had high mean LB summed concentrations.

Examining the difference between the mean LB and UB summed concentrations showed that this difference was, for example, relatively small for spinaches and cod, and much larger for chicken and salmon (see Table 9). These various differences can be explained by the proportion of reported PFAS concentrations below the LOD, LOQ or LOC, which was small for spinaches and cod, and large for chicken and salmon (see Appendix B). Furthermore, the mean LB summed PFAS concentration in drinking water from groundwater was lower than in drinking water from surface water, whereas this was the other way around for the mean UB summed concentrations (see Table 9). This was due to a higher number of reported concentrations below the LOQ and higher reported LOQs for drinking water from groundwater.

4.3 PFAS exposure

The long-term exposure to PFAS, expressed as PEQ, through food and drinking water was calculated using the mean LB and UB summed concentrations from Table 9. The results are shown in Table 10.

Table 10 Long-term lower bound and upper bound exposure to PFAS, expressed as PEQ, through food and two drinking water types for the Dutch consumer aged 1-79 years [a]

Scenario	**PFAS exposure in ng PEQ/kg body weight per week[b]**		
	Mean	**Median (P50)**	**High (P95)**
Food and drinking water from groundwater			
Lower bound	4.6 (4.4-4.7)	3.3 (3.2-3.4)	12 (11-13)
Upper bound	26 (26-27)	23 (23-24)	51 (49-52)

Scenario	PFAS exposure in ng PEQ/kg body weight per week[b]		
	Mean	Median (P50)	High (P95)
Food and drinking water from surface water			
Lower bound	5.9 (5.7-6.1)	4.6 (4.5-4.7)	14 (13-15)
Upper bound	22 (22-23)	19 (19-20)	45 (42-46)

ng: nanogram; P50: 50th percentile; P95: 95th percentile; PEQ: PFOA equivalents; PFAS: per- and polyfluoroalkyl substances

[a] Allocation of PFAS concentrations according to the lower bound and upper bound scenario is explained in section 3.2.

[b] 2.5% lower - 97.5% upper confidence limits of the exposure parameters are reported between brackets (see section 3.5).

The lowest and highest calculated exposure to PFAS through food and drinking water from groundwater were 3.3 (median LB) and 51 (high UB) ng PEQ/kg body weight per week, respectively. The corresponding exposure through food and drinking water from surface water was 4.6 and 45 ng PEQ/kg body weight per week.

Considering the uncertainty regarding the exposure parameters due to the sample size of the food consumption and concentration databases (see section 3.5), the high UB exposure to PFAS was hardly affected: 52 ng/kg body weight per week through food and drinking water from groundwater and 46 ng/kg body weight per week through food and drinking water from surface water. The same was true for the other exposure parameters (see Table 10). This result shows that the uncertainty around the true distribution of exposure based on the available food consumption and concentration data was small. Note that the uncertainty due to concentrations of PFAS in drinking water could not be considered (see section 3.5).

4.4 Contribution of food groups to exposure

Figure 1 shows the percentage contribution of drinking water and the food groups that contributed for at least 5% to the long-term LB and UB exposure to PFAS through food and the two drinking water types. Each food group includes all consumed food products belonging to that food group (sampled and non-sampled food products) eaten as such, or as ingredient of composite foods (see section 3.3.1). An overview of the percentage contribution of each food group and drinking water type to the exposure distribution is listed in Appendix E.

Based on these percentages, the mean LB and UB exposure to PFAS, expressed as PEQ, through each food group and through drinking water from groundwater and from surface water was also calculated. This was carried out by multiplying the percentage contribution of each food group and drinking water type with the relevant mean LB and UB exposure to PFAS as listed in Table 10. These results are also shown in Appendix E and will not be further addressed.

Below, the percentage contribution to the LB exposure is discussed, and not to the UB exposure, because this reflects the contribution based on

concentrations above the LOQ and does not depend on concentrations assigned to those reported below the LOD and LOQ.

Overall, food contributed more to the LB exposure than drinking water: 94% versus 6%, respectively, for drinking water from groundwater, and 73% versus 27%, respectively, for drinking water from surface water (see Figure 1). Drinking water from surface water had a higher mean LB summed PFAS concentration than groundwater (1.5 versus 9.7 pg PEQ per gram; see Table 9 in section 4.2), resulting in a higher contribution of drinking water to the LB exposure.

Examining the contribution of the various food groups to the LB exposure showed that, with 30%, 'fish and fish products' contributed most to the LB exposure through food and drinking water from groundwater. Fish had the highest summed PFAS concentrations (see Table 9 in section 4.2). The second highest contributor was the food group 'drinks (excluding drinking water)', with 23%. The contribution of this food group was mainly due to PFAS (i.e. PFUnDA; see Appendix B) in coffee beans and tea leaves, materials used during preparation of the drinks, and/or PFAS contamination during preparation of these drinks (and not from the drinking water used for their preparation; see section 3.3.2). 'Drinks (excluding drinking water)' did not have high summed PFAS concentrations (see Table 9 in section 4.2), but contributed largely, because they are regularly consumed in relatively high quantities. The third main food contributor to the LB exposure through food and drinking water from groundwater was 'dairy' with 17% due to PFAS present in milk. 'Meat and meat products' was the fourth highest contributor with 8% due to relatively high summed PFAS concentrations in all meat products, except for chicken. Also '(processed) vegetables' contributed at least 5% to the LB exposure.

For the LB exposure to PFAS through food and drinking water from surface water, 'fish and fish products also contributed most to the exposure through food with 24%, followed by 'drinks (excluding drinking water)' with 18%, 'dairy' with 13% and 'meat and meat products' with 6% (see Figure 1).

All other food groups contributed less than 5% to the LB exposure to PFAS through food and both drinking water types (see Figure 1).The food products that contributed most to the LB exposure within the three main contributing food groups, were:

- cod with 40%, fish fingers with 9% and shrimps with 8% within the food group 'fish and fish products';
- 'coffee prepared' with 54% and 'tea prepared' with 46% within the food group 'drinks (excl. drinking water)'; and
- milk with 45%, yoghurt with 24% and cheese with 11% within the food group 'dairy'.

Since it concerns the contributions within a food group, these percentages were independent from the drinking water type.

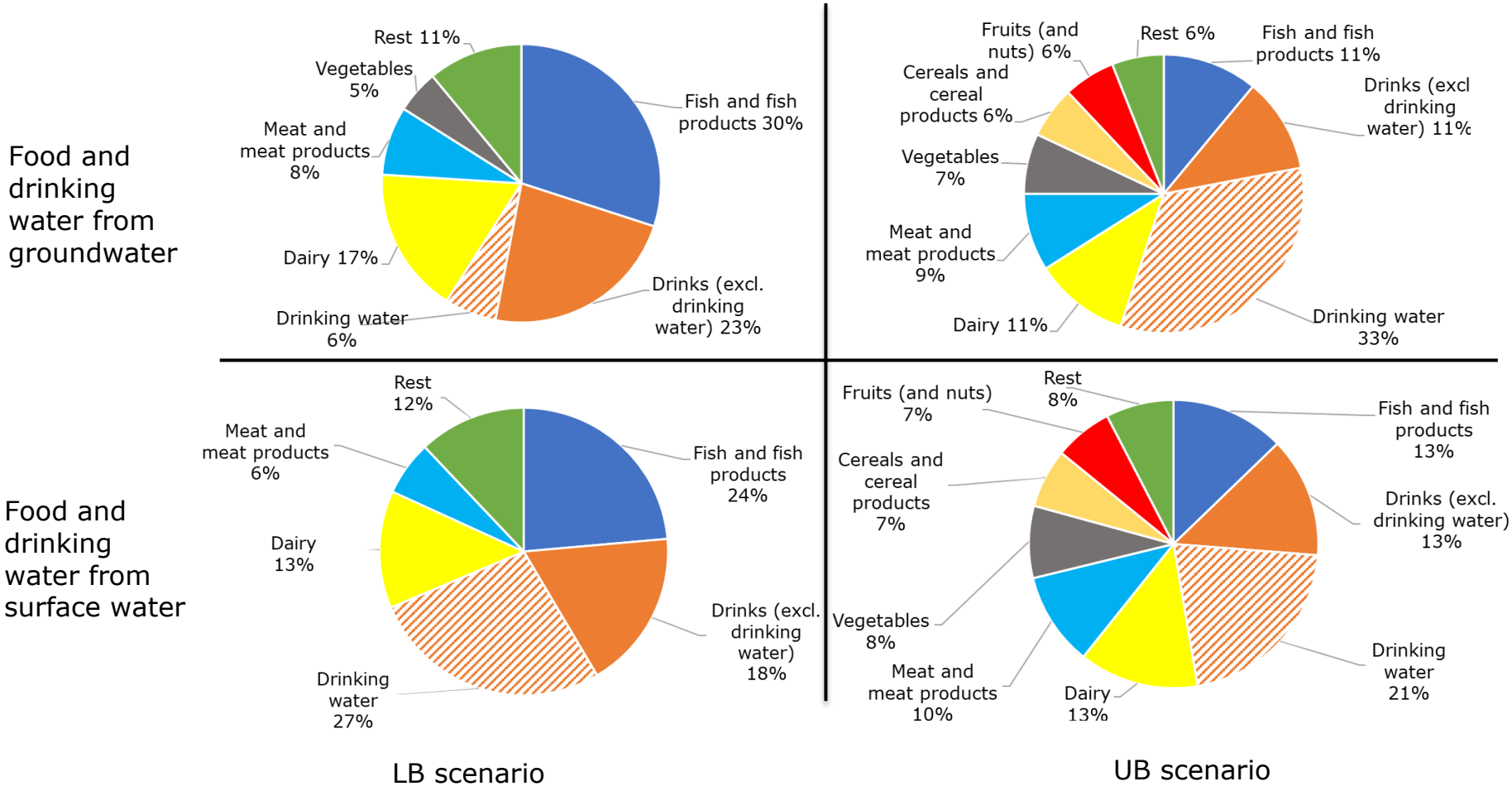

Figure 1 Percentage contribution of drinking water[a] and food groups[b] that contributed at least 5% to the long-term lower bound (LB) and upper bound (UB) summed exposure to PFAS, expressed as PEQ, through food and two drinking water types for the Dutch consumer aged 1-79 years [c,d]

LB: lower bound; PEQ: PFOA equivalents; PFAS: per- and polyfluoroalkyl substances; UB: upper bound
[a] If only the 16 PFAS (excluding also PFBA that could not be determined in any of the food products) that could be determined in both food and drinking water were considered, the contribution of drinking water to the LB exposure would decrease to 3% for drinking water from groundwater and to 22% for drinking water from surface water (see also section 5.2).
[b] Food group 'vegetables' also includes 'processed vegetables'.
[c] Allocation of PFAS concentrations according to the LB and UB scenario is explained in section 3.2.
[d] Mean LB and UB exposure to PFAS, expressed as PEQ, through each food group and through each drinking water type is listed in Appendix E.

4.5 Contribution of individual PFAS to exposure

The percentage contribution of each individual PFAS that contributed at least 5% to the long-term LB and UB exposure to PFAS through food and the two drinking water types is shown in Figure 2. An overview of the contribution of each PFAS is listed in Appendix E. This appendix also lists the mean LB and UB exposure to each individual PFAS, expressed as PEQ, through food and drinking water, which were calculated by multiplying the percentage contribution for each PFAS with the relevant mean LB and UB exposure to PFAS (see Table 10). As for the contribution of the food groups and drinking water, only the contribution of the individual PFAS to the LB exposure distribution is discussed.

PFUnDA, PFOS and PFDA contributed most to the LB exposure to PFAS through food and both drinking water types. PFOS is the only PFAS of these three PFAS that belongs to the EFSA-4. The contribution of these three PFAS to the LB exposure was 42%, 20% and 16%, respectively, through food and drinking water from groundwater and 32%, 19% and 12%, respectively, through food and drinking water from surface water.

PFUnDA and PFDA were important contributors, because of their high potency (RPF of 4 and 10, respectively) and high concentrations in fish. PFUnDA was also found in relatively high concentrations in two coffee and three tea samples (see Appendix B and Table 9 in section 4.2). Combined with a relatively high consumption of coffee and tea, this also resulted in a high contribution of PFUnDA to the LB exposure. PFOS is also relatively potent with an RPF of 2, and the highest PFOS concentrations were measured in fish and meat.

For the LB exposure through food and drinking water from groundwater, the fourth highest contributor was PFTrDA with 6%. PFTrDA is also relatively potent (RPF of 3) and was present in high concentrations in fish. PFOA was the fourth highest contributor with 9% to the LB exposure to PFAS through food and drinking water from surface water.

PFNA and TFA contributed both with 6%, and PFHpA contributed with 5% to the LB exposure in the surface water scenario. The contribution of TFA was the result of high concentrations of this PFAS in drinking water from surface water (see Appendix C). Due to a low potency of this PFAS (RPF of 0.002), this contribution was lower than would have been expected on the basis of the reported concentrations.

All other PFAS contributed less than 5% to the LB exposure through food and the two drinking water types.

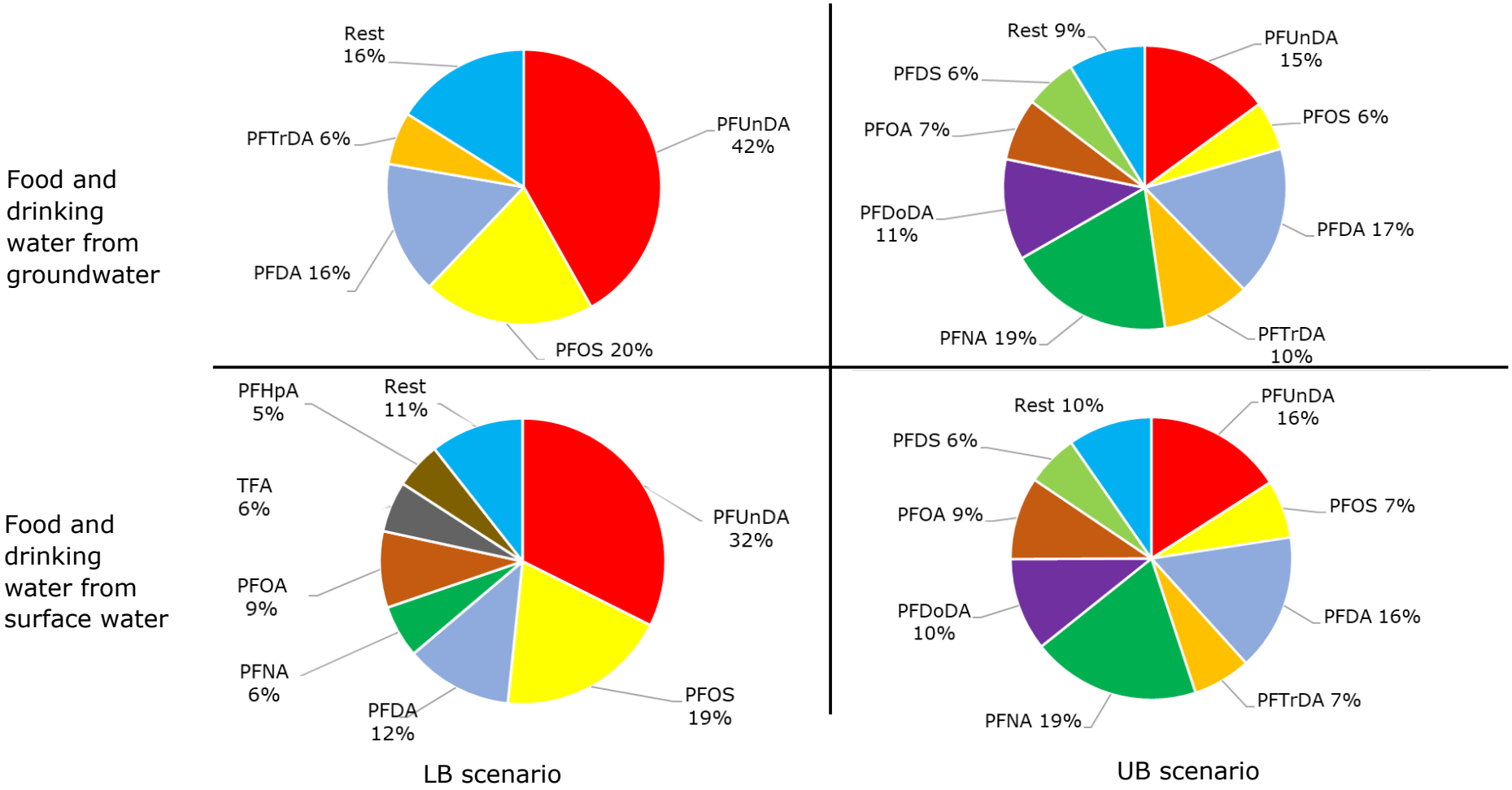

Figure 2 Percentage contribution of individual PFAS[a] to the long-term lower bound (LB) and upper bound (UB) exposure to PFAS, expressed as PEQ, via food and two drinking water types for the Dutch consumer aged 1-79 years [b,c]

LB: lower bound; PEQ: PFOA equivalents; PFAS: per- and polyfluoroalkyl substances; UB: upper bound
[a] The names of the PFAS can be found in Table 3 in section 2.2.2.
[b] Allocation of PFAS concentrations according to the LB and UB scenario is explained in section 3.2.
[c] Mean LB and UB exposure to the individual PFAS, expressed as PEQ, through food and two drinking water types is listed in Appendix E.

5 Uncertainties in the exposure assessment

The calculated exposure to PFAS through food and two drinking water types was influenced by different sources of uncertainty, including:

- concentration data;
- RPF approach;
- allocation of PFAS concentrations to consumed foods;
- model used to calculate the exposure;
- effect of processing; and
- food consumption data.

These sources are discussed below.

5.1 Concentration data

One important source of uncertainty related to the concentration data was the large number of non-detected (< LOD) and non-quantified concentrations (< LOQ). This uncertainty was quantified by calculating the LB and UB exposure. As explained in Box 1 (see section 3.2), the LB exposure will underestimate and the UB exposure will overestimate the exposure to PFAS on the basis of the reported concentrations in food and drinking water. For example, the high (P95) exposure through food and drinking water from groundwater could range from 12 (LB) to 51 (UB) ng PEQ/kg body weight per week (see Table 10 in section 4.3). The actual high exposure calculated with the available data will lie between these two results. This uncertainty can be decreased by lowering the analytical limits. In addition, the UB exposure could be refined by, for example, excluding certain PFAS with a reported concentration below the LOD from the assessment, on the basis of their chemical properties. For example, some PFAS are fat soluble, and may truly not be present, when reported below the LOD or the LOQ, in food products with a polar matrix. The UB exposure was not refined, because the LB exposure already exceeded the TWI (see chapter 7).

Some PFAS concentrations in food products were reported as below the LOC. These PFAS were either assigned a concentration equal to the LOQ (LB) or the LOC (UB). However, the presence of the specific PFAS in those samples could not be confirmed, albeit highly likely (see section 2.1.2), which could have resulted in an overestimation of the LB and UB exposure if the PFAS was not present in the food product. Since less than 1% of the PFAS concentrations was reported to be below the LOC, this uncertainty was considered to have had a negligible effect on the LB and UB exposure.

Another important source of uncertainty relating to the concentration data in food was that not all 17 PFAS could be determined in all food products. PFBA could not be determined in any of the food products, and several other PFAS could not be determined in some food products or in some samples of a food product, resulting in missing concentrations (see Table 8 in section 4.1). Only HFPO-DA (GenX) and PFNA could be determined in all food products.

Table 11 lists per individual PFAS if there is a risk of underestimating the exposure due to these missing concentrations. Overall, underestimation of the LB and UB exposure as a result of this uncertainty cannot be excluded, especially due to missing concentrations of PFDA and PFUnDA, two PFAS with high RPFs (10 and 4, respectively).

For each food product, 2-15 samples were analysed with at least 10 samples included for regularly consumed food products (e.g. potatoes and bread; see Table 1 in section 2.1.1). The food group 'French fries' was an exception with only three analysed samples. However, since the PFAS concentrations in potatoes were similar to those in the three samples of French fries, the PFAS concentrations in these three samples were considered sufficiently representative for French fries. Several food products were sampled less than 10 times. Priority was given during sampling to include a variety of food products instead of sampling only a few food products in large numbers. Thus, it was expected that a more realistic exposure to PFAS could be obtained within the boundary conditions of the project (available budget).

In total, 12 prepared coffee samples and 13 prepared tea samples were analysed for PFAS (see Table 1 in section 2.1.1). These samples were prepared across five separate days in five batches by using the same materials (e.g. boiling water tap, coffee filter and glassware). One batch consisting of two coffee and three tea samples had higher concentrations of PFUnDA compared with the samples of the other four batches: 11-22 pg per gram prepared drink versus < LOD-3.1 pg per gram prepared drink (see Appendix B). The reason for these higher concentrations is unclear but it is most likely related to the preparation of this batch, because all samples were analysed on the same day. Because the source of the higher PFUnDA concentrations is unknown and it could not be excluded that these higher concentrations can occur when coffee or tea is prepared at home or outdoors (e.g. in a restaurant), these concentrations were included in the exposure calculations. However, more research is needed to understand the source and thus relevance of these higher PFUnDA concentrations.

The concentrations of the linear isomers of PFAS in food and drinking water were reported (see sections 2.1.2 and 2.2.2). In addition, the concentrations of the branched isomers of PFOS were included in the PFAS concentrations for food, and those for PFOS, PFOA and PFHxS for part of the drinking water samples (see section 2.2.2). Missing concentration data for the branched isomers of these three PFAS in drinking water may have resulted in an underestimation of the summed PFAS concentrations in those drinking water samples. This also held true for PFOA and PFHxS in all food product samples. However, the underestimation for food is low, as branched isomers of these two PFAS in food are expected to be low (personal communication WFSR). Not considering the branched isomers for the other PFAS in both food and drinking water is expected to have resulted in a negligible underestimation of the PFAS concentration, since they occur (mainly) as linear isomers (personal communication WFSR; Sadia et al., 2023).

High concentrations of TFA were found in the two drinking water types (see Appendix C). As a result, TFA contributed considerably to the LB

Table 11 Overview of the probability of underestimating the lower bound and upper bound exposure through food due to not or only partly considering certain PFAS in the assessment

PFAS[a]	RPF	Not determined in[b]	Probability of underestimating the exposure
Sulfonic acids			
PFBS	0.001	Several samples	Small: low RPF[d]
PFHxS[c]	0.6	One sample of rice grain (out of 10)	Negligible
PFHpS	2	Onions, leeks, potatoes and French fries and some samples of some vegetables	Small: PFHpS concentrations in beetroots and carrots (also root and tuber vegetables such as potatoes) were below the LOD or LOQ; onions and leeks are consumed regularly, but not in high amounts; and, other vegetables were considered in the assessment, only on the basis of fewer samples per vegetable
PFOS[c]	2	Onions and leeks	Negligible: onions and leeks are regularly consumed vegetables, but not in high amounts
PFDS	2	Some samples of fruits and vegetables	Negligible: all fruits and vegetables were considered in the assessment only based on fewer samples per fruit or vegetable

PFAS[a]	RPF	Not determined in[b]	Probability of underestimating the exposure
Carboxylic acids			
PFBA	0.05	All food products	Small: low RPF[d]
PFPeA	0.05	Several samples	Small: low RPF[d]
PFHxA	0.01	Several samples	Small: low RPF[d]
PFHpA	1	Bread	Negligible: PFHpA concentrations in wheat flour (important constituent of bread) were below the LOD, except for one concentration that was below the LOQ
PFOA[c]	1	Bread	Negligible: PFOA concentrations in wheat flour (important constituent of bread) were below the LOD
PFNA[c]	10	N.a.	N.a.
PFDA	10	Onions, leeks and most leafy vegetables	High: in the four samples of lettuces in which PFDA could be determined, one concentration was above the LOC; high RPF
PFUnDA	4	Cereal products, meat, almost all oils and fats, and a small subset of the samples of some vegetables	High: 'meat and meat products' is a highly consumed food group; high RPF; and, not analysed in cereal products, which are also widely consumed. However, since the concentrations in sweet corn (which is categorised as a vegetable but can also be regarded as a cereal) were below the LOD, it is expected that the PFUnDA concentrations in cereal products will also be low.

PFAS[a]	RPF	Not determined in[b]	Probability of underestimating the exposure
Carboxylic acids			
PFDoDA	3	Some samples of vegetables	Negligible: all fruits and vegetables were considered in the assessment, but on the basis of fewer samples per fruit or vegetable
PFTrDA	3	Some samples of some fruits and vegetables (except for peas, onions and leeks)	Small: PFTrDA concentrations in samples of fruits and vegetables in which it was determined were below the LOD
PFTeDA	0.3	(Processed) vegetables and in some fruits, and some samples of eggs	Small: PFTeDA concentrations in produce from vegetable gardens within the vicinity of a PFAS source were all below the LOD, except for one concentration that was below the LOQ and one numerical concentration (Boon and te Biesebeek, 2022a & b). Fruits and eggs were considered in the assessment, but only on the basis of fewer samples per fruit or egg.

PFAS[a]	RPF	Not determined in[b]	Probability of underestimating the exposure
Ether carboxylic acids			
HFPO-DA (GenX)	0.06	N.a.	N.a.

PFAS[a]	RPF	Not determined in[b]	Probability of underestimating the exposure
Total			**Underestimation of the exposure to PFAS cannot be excluded, especially due to missing concentrations of PFDA and PFUnDA**

N.a.: not applicable; PFAS: per- and polyfluoroalkyl substances; RPF, relative potency factor

[a] The names of the PFAS can be found in Table 2 in section 2.1.2.

[b] Table 8 in section 4.1 provides a more detailed description of food products or samples of food products that were not analysed.

[c] PFAS belonging to the EFSA-4 (see section 3.4).

[d] An RPF was considered low when it was ≤ 0.05.

exposure to PFAS through only drinking water, despite a low potency of this PFAS (RPF of 0.002): 53% for drinking water from groundwater and 21% for drinking water from surface water. However, the concentrations of TFA in drinking water from groundwater were uncertain, because of the limited number of samples in which this PFAS was analysed (see Table 3 in section 2.2.2). In addition, drinking water companies tend to monitor on the basis of risk. Therefore, it is recommended that in a follow-up, more samples of drinking water from groundwater, which have been sampled objectively, are analysed for TFA.

5.2 Relative potency factor (RPF) approach

Summed concentrations of all individual PFAS per sample were calculated by applying the RPF approach. The RPFs for PFDA, PFHpA, PFHpS, PFPeA, PFPeS and PFTrDA were uncertain, and were reported as a range (Bil et al., 2021; see footnote b of Table 7 in section 3.4). For these PFAS, the highest RPF of the range was taken to calculate the summed concentrations, as recommended by RIVM (2021). This could have resulted in an overestimation of the exposure, in particular regarding PFDA, which contributed approximately 12-16% to the LB exposure (see Appendix E).

The exposure to PFAS was calculated on the basis of 16 PFAS analysed in food (excluding PFBA, which could not be determined in any of the food product samples) and on 20 PFAS analysed in drinking water. Of the four PFAS that were only analysed in drinking water, PFBA, PFPeS and TFA were present in quantifiable amounts (≥ LOQ) in all, or part of the drinking water samples (see Appendix C). ADONA, the fourth PFAS that was only analysed in drinking water, could not be quantified in any of the samples. If the exposure assessment had been based on the 16 PFAS that were analysed in both food and drinking water, the contribution of drinking water to the LB exposure would decrease from 6% to 3% for drinking water from groundwater, and from 27% to 22% for drinking water from surface water.

Not all analysed PFAS in drinking water were included in the exposure assessment, because no RPFs have been derived (yet) for these PFAS. It is expected that RPFs will become available for more PFAS in the future. When more PFAS are included in the exposure calculation, this will potentially result in a higher exposure. PFAS with quantified concentrations in drinking water (≥ LOQ) that were excluded from the exposure assessment, because no RPFs were available, included known precursors of PFAS, such as perfluorooctanesulfonamide (PFOSA) and 6:2 fluorotelomer sulfonic acid (6:2 FTS). These precursors can be degraded to one of the 24 PFAS for which an RPF is available. Smit and Verbruggen (2022) assigned precursors the RPF of their degradation product. This was part of a worst-case approach as full degradation of the precursor was assumed. Here, precursors were not included in the exposure assessment, because degradation was not expected in the drinking water distribution network. Exclusion of the precursors from the assessment is therefore expected to have had a negligible effect on the calculated LB and UB exposure to PFAS.

5.3 Allocating PFAS concentrations to consumed foods

Not all food products that could contain PFAS were analysed for PFAS in the current study. This was addressed by assigning PFAS concentrations from sampled food products to similar non-sampled food products, and by using the Dutch food conversion model to calculate PFAS concentrations in composite foods and processed counterparts of the food products (see section 3.3). Thus, more than 99% of the consumed amount reported in the DNFCS was considered in the exposure assessment.

By assigning concentrations to non-sampled food products, an underestimation of the exposure to PFAS was generally minimised. However, if the mean concentration of a sampled food product was systematically higher or lower than the (unknown) mean concentration of a non-sampled food product, this may potentially result in an over- or underestimation of the exposure, respectively. The extent to which the exposure was affected by this uncertainty is not clear. We expect this uncertainty to have been levelled out due to the diverse range of non-sampled food products.

Calculating PFAS concentrations in composite foods with the food conversion model is potentially a source of uncertainty in an exposure assessment. It is uncertain whether the calculated PFAS concentrations reflect the actual PFAS concentrations in the foods as they are consumed. The advantage of a food conversion model is that non-sampled composite foods can be included in the assessment on the basis of analysed concentrations in ingredients of these foods. One disadvantage is that the concentrations are calculated with a model and are not based on analysed concentrations. Overall, we expect it to be highly likely that these uncertainties also levelled out in the final exposure calculations, considering the large number of foods included in the assessment via the food conversion model.

5.4 OIM calculation model

The long-term exposure to PFAS was calculated with the OIM model. This model uses the mean consumption over the survey days of each individual to assess the individual's long-term exposure. Due to the limited duration of the dietary survey (i.e. two days), the OIM model tends to overestimate the high (P95) exposure, while the mean and median exposure are considered to be correctly estimated (Boon and van der Voet, 2015). Other models are available to estimate the high exposure more realistically. However, the current data was not fit for this purpose (i.e. the logarithmic transformed daily positive exposure distribution was not normally distributed), and therefore, the OIM model was applied. This is considered to be a conservative approach.

5.5 Effect of processing and food contact materials

Only limited information is available in the literature on the effect of food processing on PFAS concentrations. There is an inconsistent view whether processing reduces or increases PFAS concentrations in food products as consumed (EFSA, 2020). As the information on processing is limited and inconsistent, it is unclear how processing has affected the PFAS concentrations, especially in (highly) processed foods. For

example, the concentrations in the food products belonging to the food group 'dairy' (e.g. low-fat yoghurt, cheese and soft cheese) were calculated on the basis of their mass percentage of milk and the PFAS concentrations in milk, without considering the possible influence of processing. As dairy products are widely consumed in the Netherlands, this could have led to an overestimation or underestimation of the exposure to PFAS. Similar considerations apply to other food groups. The effect of processing of foods (e.g. cooking) regarding changes in food weight due to preparation (e.g. shrinking of vegetables due to cooking) was considered in the exposure calculations by the food conversion model. In that case, it was assumed that the PFAS remains in the processed food and is not (partly) removed via the cooking water, or vice versa that the PFAS remains in the cooking water and does not move into the processed food. This is a source of uncertainty that could have resulted in an overestimation and underestimation of the exposure, respectively (see also sections 3.3 and 5.3). This latter uncertainty mainly concerns vegetables. For several other processed foods that absorb water during cooking, such as pasta and rice, the cooking water is included in the consumed food via the food conversion model. Uncertainties due to processing can be addressed by analysing PFAS in processed food products, such as cooked spinach, low-fat yoghurt and cheese.

Food contact materials are any materials that could come into contact with food, for example kitchenware, such as pans and baking tins, and packaging materials. PFAS can be used in food contact materials for their grease repellent properties. Therefore, food contact materials can be a source of exposure due to migration of PFAS from the food contact material into food. The chemical analysis of PFAS in the current study was conducted in food products as they are available to consumers, including both packed and unpacked food products. For example, canned and fresh salmon were sampled. The summed PFAS concentration in canned salmon was much higher than in fresh salmon (see Table 9 in section 4.2). However, whether this difference was due to PFAS from packaging materials cannot be established because the current study was not designed to draw conclusions on the source of PFAS in the food. In addition, the possible contribution of PFAS from packaging materials used for composite foods and PFAS that may end up in food due to the use of kitchenware during food preparation were not considered in the exposure assessment. According to EFSA, the contribution of food contact materials containing PFAS to the exposure is small compared with other sources of exposure (EFSA, 2020).

5.6 Food consumption data

The food consumption data used in the exposure assessment was the most recent available Dutch consumption data when performing the assessment. This data was collected from 2012 to 2016 (van Rossum et al., 2020). Recently, a new DNFCS was published, covering the period of 2019-2021. This DNFCS shows that Dutch people are eating more fruits and vegetables, and unsalted nuts, but less fish, and less red and processed meat than during the survey of 2012-2016.[9] A new dietary exposure calculation is needed to elaborate on the effect of these dietary

[9] https://www.wateetnederland.nl/

changes on the exposure to PFAS. However, it is not expected that these changes will influence the current risk assessment to such an extent that the conclusion on possible risks will change significantly; in particular, because the potential changes in the LB and UB exposure are expected to be small in relation to the exceedance of the TWI.

5.7 Conclusion on uncertainties

On the basis of the uncertainties described above, it is concluded that the LB exposure to PFAS through food and the two drinking water types is expected to be an underestimation of the actual exposure to PFAS. This conclusion is mainly based on the observation that several PFAS could not be determined in various food samples, and that not all relevant PFAS were included in the exposure calculations. The UB exposure to PFAS is expected to be an overestimation of the actual exposure to PFAS. The assumption that PFAS reported at concentrations below the LOD and the LOQ were present at the respective analytical limit values is considered to more than balance the likely underestimation due to non-determined or not-included PFAS. Given the uncertainties and assumptions of the assessment, the LB exposure is considered to be closer to the actual exposure than the UB exposure.

6 Comparison with previously reported exposure assessments in the Netherlands

This report describes the exposure assessment of PFAS through food and drinking water for Dutch consumers aged 1-79 years. In this chapter, the results are discussed in relation to the PFAS exposure assessment from 2021 (van der Aa et al., 2021; see section 6.1) and compared with three other recent Dutch studies on dietary exposure to PFAS (i.e. Boon and te Biesebeek, 2022a & b; Zwartsen and Boon, 2022; see section 6.2). In addition, the current results are compared with the findings of the PFAS exposure assessment of EFSA in 2020 (EFSA, 2020; see section 6.3).

6.1 Exposure assessment in 2021

In 2021, RIVM assessed the summed exposure to PFAS through food and the same two drinking water types on the basis of the RPF approach (van der Aa et al., 2021). Table 12 shows the results of the current exposure calculations compared with the assessment from 2021. The LB exposure in the current study is lower than calculated in 2021 for all three exposure parameters. The UB exposure in the current study is higher compared with 2021 for all parameters.

Table 12 Long-term lower bound (LB) and upper bound (UB) exposure to PFAS through food and two drinking water types[a] reported in 2021 and in the current study (2023) for the Dutch consumer aged 1-79 years [a]

Drinking water type	Exposure parameter	PFAS exposure (ng PEQ/kg body weight per week)			
		2021[b,c]		2023[d]	
		LB	UB	LB	UB
Groundwater	Mean	8.4	23	4.6	26
	Median (P50)	6.3	20	3.3	23
	High (P95)	20	45	12	51
Surface water	Mean	9.9	21	5.9	22
	Median (P50)	7.8	18	4.6	19
	High (P95)	21	43	14	45

LB: lower bound; ng: nanogram; P50: 50th percentile; P95: 95th percentile; PEQ: PFOA equivalents; PFAS: per- and polyfluoroalkyl substances; UB: upper bound

[a] Allocation of PFAS concentrations according to the LB and UB scenario is explained in section 3.2.

[b] Exposure from 2021 was the summed exposure to the EFSA-4 (i.e. PFOA, PFNA, PFHxS and PFOS) using the RPF approach (van der Aa et al., 2021).

[c] Exposure from 2021 (except for the high UB exposure through food and drinking water from surface water) was outside the 95% confidence intervals of the PFAS exposure from 2023 (see Table 10 in section 4.3).

[d] Exposure in the current study was based on 16 PFAS in food (see Table 8 in section 4.1; PFBA was excluded as it could not be determined (see section 4.1)) and 20 PFAS in drinking water (see section 2.2).

The following differences should be considered when comparing the current PFAS exposure with the exposure from 2021:

- the period of sampling;
- the kind and number of samples; and
- the number of analysed PFAS.

The food consumption data used in both assessments was the same.

In the current assessment, food products were sampled in 2021, whereas the food products used in the 2021 assessment were sampled in 2009 (van der Aa et al., 2021). The current exposure assessment thus provides a more up-to-date calculation of the PFAS exposure. The analysis of PFAS in recently sampled food products was recommended in 2021 (van der Aa et al., 2021).

The summed PFAS concentrations in food used in the exposure assessment from 2021 were derived from total diet study (TDS) samples (van der Aa et al., 2021). These samples are composite samples in which separate food products are pooled before analysis, resulting in one summed PFAS concentration for each composite sample. Each composite sample represents a food group, such as fruits, vegetables or fish. Due to pooling, information about potential differences in summed PFAS concentrations between individual food products within a food group is lost. For example, a fictitious TDS sample for fruits consists of apples, oranges and bananas. The summed PFAS concentrations are 1 pg PEQ per gram in apples, 2 pg PEQ per gram in oranges, and 27 pg PEQ per gram in bananas. This results in a (mean) summed PFAS concentration for this TDS sample of 10 pg PEQ per gram. The high summed PFAS concentration in bananas is levelled out by the low summed PFAS concentrations in apples and oranges, and vice versa. In the current assessment, all individual samples were analysed separately, and thus the number of samples per food product was higher compared with the single TDS sample per food group in the assessment from 2021. Consequently, consumed foods could be linked more precisely to the sampled food products, and the exposure could be calculated with more accuracy compared with the previous exposure assessment.

Another important difference between the two assessments is the number of PFAS considered. In the current assessment, 16 PFAS in food and 20 PFAS in drinking water were included, while the 2021 assessment was based on the four PFAS on which the TWI is based, the so-called EFSA-4 (see section 3.4). By considering more PFAS, a more accurate LB exposure to PFAS was calculated (see Table 12). And even though more PFAS were included, the LB exposure was lower than reported in 2021. The UB exposure was higher than reported in 2021 due to the inclusion of more PFAS at their analytical limit values. When only the EFSA-4 were considered as in 2021, a mean UB exposure through food and drinking water from groundwater or surface water of 8.4 and 7.9 ng PEQ/kg body weight per week, respectively, could be calculated on the basis of the percentage contribution of the EFSA-4 to the current UB exposure distribution (see Appendix E). The UB exposure was a factor 2.7 lower than reported in 2021. The LB exposure to the EFSA-4 was 1.3 and 2.0 ng PEQ/kg body weight per week, respectively, and a factor 6.5 and 5.0 lower than reported in 2021.

On the basis of the above considerations, it is concluded that the exposure to PFAS for the Dutch consumer was calculated more realistically in the current report compared with the assessment from 2021. This is mainly due to the availability of concentrations of more

individual PFAS in recent food samples. In addition, a large number of different food products with a relatively large number of samples per food product was analysed for the current assessment allowing for a more precise allocation of PFAS concentrations to consumed foods.

6.2 Recent exposure assessments in specific areas in the Netherlands

Study in the Western Scheldt
In 2022, RIVM calculated the exposure to PFAS from contaminated fish, fish products and sea vegetables sampled in the Western Scheldt (Zwartsen and Boon, 2022). Water in this estuary contains high concentrations of PFAS due to emission of industrial wastewater. The same set of PFAS was analysed as in the current study, but in different types of fish compared with the current assessment. Summed LB PFAS concentrations varied from 5,200 to 132,000 pg PEQ per gram (equal to 5.2 and 132 ng PEQ per gram, as reported in Zwartsen and Boon (2022)) in the various types of fish. These concentrations are extremely high compared with the current findings in fish ranging from 17 to 1843 pg PEQ per gram in the LB scenario.

Two studies in allotments in Helmond and in the environment of Dordrecht
Also in 2022, RIVM calculated the exposure to the same set of PFAS from the consumption of fruits and vegetables from allotments in the vicinity of two chemical plants in Helmond and Dordrecht that emit or emitted PFAS into the environment (Boon and te Biesebeek, 2022a & b). One of the allotments was a so-called reference location, which was located relatively far southeast from the plant in Dordrecht, and was, as such, not expected to be contaminated with PFAS. PFAS concentrations in the products from this reference location were assumed to represent the concentrations in fruits and vegetables sold in shops. No information on PFAS concentrations in such products were available at that time. To check whether that assumption was correct, the PFAS concentrations in products from the reference location were compared with those analysed in fruits and vegetables in the current study.

In the scenario according to a low concentration level (similar to the LB scenario), the mean summed PFAS concentrations in fruits from the reference location varied from 0.35 pg PEQ per gram in stone fruits to 42 pg PEQ per gram in strawberry. In the current analysis, these concentrations ranged from 0.48 pg PEQ per gram in mandarins to 11 pg PEQ per gram in oranges (see Table 9 in section 4.2). The analysed summed PFAS concentrations in fruits in the current assessment are lower, but this was mainly due to one high mean summed PFAS concentration in strawberry in the reference allotment. This high concentration in strawberry was based on only two samples with very different summed concentrations (i.e. 0.1 and 84 pg PEQ per gram).

The LB summed concentrations for vegetables from the reference location ranged from 0 pg PEQ per gram in Brussels sprouts and courgette to 56 pg PEQ per gram in kale. These findings are in the same order of magnitude as the current findings (i.e. 0.0009 pg PEQ per gram

in cucumbers to 50 pg PEQ per gram in lettuces, excluding crisp lettuces).

Generally, the findings in the reference allotment were comparable to the current PFAS concentrations in fruits and vegetables, confirming the assumption that PFAS concentrations in the products from the reference location represented the concentrations in fruits and vegetables sold in shops.

6.3 Exposure assessment of EFSA

In 2020, EFSA calculated the dietary exposure to PFAS for various European countries, including the Netherlands (EFSA, 2020). The exposure was based on monitoring results for the EFSA-4 in diverse food samples and drinking water from 16 European countries (no Dutch monitoring data). As in the current study, high quantified concentrations of PFAS were reported in fish. The exposure assessment of EFSA was limited to the EFSA-4 and EFSA assumed equipotency for these four PFAS.

The mean LB exposure to the EFSA-4, calculated by EFSA for the Dutch population, ranged from 3.8 to 10 ng/kg body weight per week across different age groups (1-75+ years). These exposure results were based on food consumption data from different DNFCSs that covered the period of 2006 up to 2012. In the paragraph below, the exposure reported by EFSA is compared with the LB exposure calculated in the current study. This is not a straightforward comparison, because in the current study, the exposure was calculated using the RPF approach and was based on more PFAS (see section 3.4). Furthermore, the exposure was calculated for the total age group of 1-79 years. Therefore, this comparison should only be considered as indicative and to give a sense of the order of magnitude of the difference between the exposure results.

To compare the results of the current study with those reported by EFSA, the long-term exposure for the total population was first calculated on the basis of the reported exposure for each age group by EFSA. For this, the mean LB exposure per age group was multiplied by the number of years per age group and subsequently divided by the total number of years across all age groups (see Appendix F). In this calculation, the age group of 75+ years was included as five years (75-79 years). The total number of years then equalled 79 and was the same as the number of years covered in the current study. The mean LB long-term exposure to the EFSA-4 thus calculated was 4.6 ng/kg body weight per week, based on equipotency. This exposure is similar to the mean LB exposure of 4.6 ng PEQ/kg body weight per week through food and drinking water from groundwater calculated in the current study, and lower than the mean LB exposure of 5.9 ng PEQ/kg body weight per week through food and drinking water from surface water, both calculated with the RPF approach for 20 PFAS (see Table 10 in section 4.3).

The mean LB exposure of 4.6 ng/kg body weight per week based on the EFSA assessment could also be compared with the mean LB exposure

for the EFSA-4 based on the results of the current study (see section 6.1). This showed that the mean LB exposure to the EFSA-4 was considerably lower in the current study, even when applying the RPF approach: 1.3 and 2.0 ng PEQ/kg body weight per week, depending on the drinking water type.

Several differences exist between the two assessments, such as the inclusion of more PFAS and the use of the RPF approach in the current study as opposed to only including the EFSA-4 and the assumption of equipotency in the EFSA assessment. Due to the use of recent concentration data obtained with more sensitive analytical methods for more PFAS and more up-to-date food consumption data, the reported exposure in this report is expected to reflect the exposure to PFAS in the Netherlands more accurately than the exposure reported by EFSA.

7 Risk assessment

The long-term exposure (see Table 10 in section 4.3) was compared with the TWI of PFAS of 4.4 ng/kg body weight as derived by EFSA (EFSA, 2020). For this comparison, it was assumed that the TWI is expressed as PEQ (RIVM, 2022). Furthermore, it was assumed that people consumed these foods and drinking water during their whole life, and that PFAS were present at the calculated mean summed concentrations throughout their life. A contaminant with a high (P95) exposure below the TWI is regarded as safe.

The mean and median (P50) LB and UB exposure to PFAS through food and both drinking water types were close to the TWI (see Figure 3). The high (P95) LB and UB exposure exceeded the TWI about three-fold and eleven-fold, respectively, indicating that the exposure to PFAS in the Netherlands can lead to adverse health effects.

The LB exposure is expected to be closer to the true exposure than the UB exposure (see section 5.7).

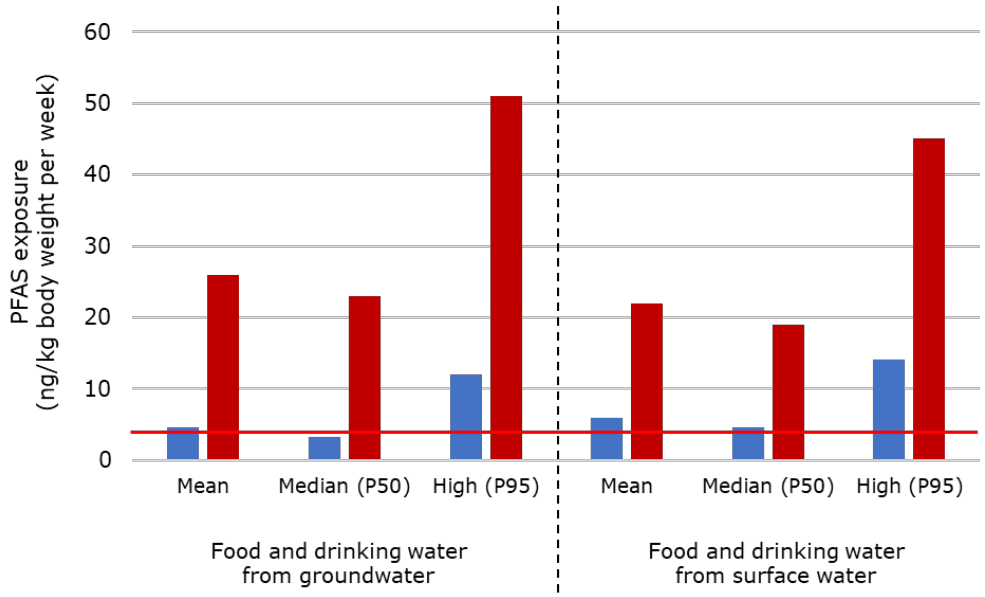

Figure 3 Mean, median (P50) and high (P95) lower bound (LB; blue bars) and upper bound (UB; red bars) long-term exposure to PFAS, expressed as PEQ, through food and two drinking water types for the Dutch consumer aged 1-79 years and compared with the TWI (red line; 4.4 ng/kg body weight) [a]

LB: lower bound; ng: nanogram; P50: 50th percentile; P95: 95th percentile; PEQ: PFOA equivalents; PFAS: per- and polyfluoroalkyl substances; TWI: tolerable weekly intake; UB: upper bound

[a] Allocation of PFAS concentrations according to the LB and UB scenario is explained in section 3.2.

8 Options to reduce exposure to PFAS through food and drinking water, and recommendations

The high (P95) dietary exposure to PFAS exceeded the TWI (see chapter 7). Below, we discuss several options to reduce the exposure to PFAS through food and drinking water, including legal measures and food advice (see section 8.1). Also, we present recommendations on the basis of the uncertainties described in chapter 5 (see section 8.2).

8.1 Options to reduce exposure to PFAS through food and drinking water

Legal measures
Several legal measures have already been taken aiming to reduce the exposure to PFAS through food and drinking water. As of 1 January 2023, the European Commission has set maximum levels (MLs) for the EFSA-4, individually and summed on the basis of equipotency, in eggs, fish and fish products, and meat and edible offal (Regulation (EC) No 2022/2388). Concurrently, the European Commission has recommended indicative levels for the EFSA-4 in fruits, vegetables, starchy roots and tubers, milk and baby food.[10] These levels *'should not affect the possibility to place on the market any food, but investigations should be carried out when the concentration of PFAS in a foodstuff exceeds those levels'*.

Comparing the analytical results for the EFSA-4 in the current study, individually and summed on the basis of equipotency, with the MLs for eggs, fish and meat showed that these results were all below these limits. For the indicative levels, the EFSA-4 concentrations in milk in the current study, the third main contributor as 'dairy' to the dietary exposure to PFAS (see Figure 1 in section 4.4), were all below these indicative levels. The analytical results for PFOA in some samples of fruits and vegetables were higher than the relevant indicative levels. Overall, the MLs and indicative levels will contribute to a lowering of PFAS concentrations in all food over time due to measures taken to comply with these levels, and thus to lowering the exposure to these contaminants through food. However, when PFAS concentrations meet these MLs, the exposure to PFAS will not necessarily be below the TWI. An exposure assessment is needed to ascertain this.

The MLs are based on the EFSA-4. In the current study, more PFAS were considered. Of the three PFAS that contributed most to the dietary exposure, two PFAS did not belong to the EFSA-4: PFUnDA (RPF of 4) and PFDA (RPF of 10) (see Figure 2 in section 4.5). When the EU MLs for PFAS are going to be reviewed in the future, inclusion of these PFAS in the MLs could be considered.

On the basis of the European Drinking Water Directive (EU) 2020/2184, the Dutch government implemented a drinking water limit value of 100 ng per litre for the sum of 20 PFAS in Dutch legislation. This legal

[10] Commission Recommendation (EU) 2022/1431 of 24 August 2022

value will come into force by 12 January 2026. Dutch drinking water already complies with this drinking water limit value (van der Aa et al., 2022). In addition, the Dutch government expressed the intent to lower the legal drinking water limit value for PFAS to 4.4 ng PEQ per litre in the future on the basis of EFSA (2020) and van der Aa et al. (2021). 11 van der Aa et al. (2022) showed that PFAS concentrations in two thirds of the drinking water samples from surface water, and in less than 5% of the drinking water samples from groundwater exceeded this limit value. As shown in the current study, drinking water from groundwater contributed less to the LB exposure to PFAS than drinking water from surface water (see Figure 1 in section 4.4). Implementation of the lower drinking water limit value will stimulate to take measures to decrease PFAS concentrations in drinking water from surface water and thus to a decrease in the PFAS exposure. Drinking water companies have already taken measures to reduce the PFAS concentrations in drinking water from surface water by intensifying treatment, such as more frequently regenerating activated carbon filters and increasing the dosage of powdered activated carbon (van der Aa et al., 2021). Comparing the median LB exposure to the EFSA-4 through drinking water from surface water reported in 2021 to the current study showed that these measures have contributed to an estimated 40% decrease in exposure (results not shown).

Other measures to decrease the exposure to PFAS have been discussed by the Dutch government and include banning the production and use of PFAS, as proposed in the PFAS restriction proposal of the European Commission, searching for alternatives of PFAS, and minimising further emissions of PFAS.[12,13] These measures will, over time, likewise result in a decrease in exposure to PFAS through food and drinking water as the amount of PFAS in the environment will decrease. As certain PFAS are extremely persistent, the decrease of PFAS in the environment will take time, and as such, contamination of food and drinking water is expected to decrease only slowly.

Consumer advice
Of the food groups included in the assessment, 'fish and fish products' contributed most, with 24-30%, to the LB exposure (see Figure 1 in section 4.4). In the upper 5% of the LB exposure distribution where the exposure exceeded 12-14 ng/kg body weight per week (see Table 10 in section 4.3), the contribution equalled even 75% through food and drinking water from surface water and 81% through food and drinking water from groundwater. A decrease in the consumption of this food group could result in a decrease in exposure to PFAS. This could be an argument to advise people to consume less fish. However, decreasing the consumption of fish is not advisable, because this food group is also a healthy part of our diet. The current advice of consuming one serving of fish per week already balances possible health risks and beneficial properties of fish consumption.[14]

[11] https://www.rijksoverheid.nl/documenten/kamerstukken/2022/10/19/brief-voor-het-commissiedebat-pfas-en-gezondheid-van-3-november-2022
[12] https://www.tweedekamer.nl/kamerstukken/brieven_regering/detail?id=2022Z19862&did=2022D42648
[13] https://echa.europa.eu/fi/-/echa-publishes-pfas-restriction-proposal
[14] https://www.voedingscentrum.nl/encyclopedie/vis.aspxONVZ#:~:text=Er%20zijn%20sterke%20aanwijzingen%20dat,lager%20risico%20op%20bepaalde%20hartziekten

In the current study, the summed PFAS concentrations in cod and tuna, which were wild-caught, were higher compared to fresh salmon and pangasius, which were farmed (see Table 9 in section 4.2). For tilapia, this information was not available. It could be explored if this provides a basis for promoting the consumption of farmed fish instead of wild-caught fish. However, this observation was only based on the samples of fresh fish bought at the supermarket. This information was not available for the fish included in the study that were bought at the market or from fishmongers. Since the available data on PFAS in wild-caught and farmed fish in this study is very limited, as well as in the EFSA opinion on PFAS (EFSA, 2020), it is not possible to advise on consumption of wild-caught versus farmed fish to reduce the exposure to PFAS.

This report also shows that the mean summed concentrations of PFAS in bottled mineral water, which is mainly produced from groundwater, are lower than in drinking water (see Table 9 in section 4.2). However, the variation in drinking water at different locations can be large, and consequently, the exposure to PFAS through drinking water at the local level can differ greatly from what is reported in this report. Therefore, the results of the current study are not suitable to determine, for an individual consumer, whether drinking predominantly bottled water would decrease the exposure to PFAS. This depends on the quality of the drinking water that is locally available. In addition, other factors should be considered before advising drinking bottled water instead of drinking tap water, such as costs, other contaminants and increased use of plastic. The results do show, however, that drinking water from surface water is likely to contain more PFAS than bottled water.

Other main food groups contributing to the dietary exposure were ‘dairy’ and ‘drinks (excluding drinking water)’. Reduction in the consumption of foods belonging to these food groups may also be an option to reduce the dietary exposure to PFAS. However, PFAS are present in most foods, so replacing the consumption of these foods with that of other foods that also contain PFAS may not result in a meaningful decrease of exposure to PFAS. The general advice to eat a varied diet for the lowest exposure to contaminants is thus very relevant for PFAS. This way, people will not eat foods with a high PFAS concentration too often.

8.2 Recommendations

Chapter 5 describes the uncertainties of the exposure assessment of PFAS through food and drinking water. An important uncertainty related to the reported exposure and the contributions of food groups and individual PFAS to the exposure distribution is the source of the relatively high concentrations of PFUnDA in coffee and tea. It is unclear what the source is of these relatively high concentrations, and more research is needed to establish this.

As indicated, it is expected that PFAS concentrations in food and drinking water will decrease over time, albeit slowly. It could therefore be relevant to repeat the exposure assessment to PFAS through food and drinking water in the future. For a new exposure assessment, it is recommended:

- to analyse the branched isomers for PFOA, PFOS and PFHxS and other relevant PFAS in all food and drinking water samples;
- to harmonise the analysis of linear and branched PFAS between the drinking water laboratories, *Vrije Universiteit Amsterdam* and WFSR as much as possible;
- to further develop the analytical methods for the analysis of PFAS in food products in which these PFAS could not be determined due to a high background signal, such as PFBA (see section 4.1);
- to include all PFAS for which an RPF has been derived in the analysis of food and drinking water; and
- to examine whether other food products (e.g. dairy products) should be sampled on the basis of the monitoring data that is expected to become available in the next years due to Commission Recommendation (EU) 2022/1431.[8]

9 Conclusion

The long-term summed exposure to PFAS, expressed as PFOA equivalents (PEQ), through food and drinking water was calculated for the Dutch population aged 1-79 years. The exposure to PFAS was calculated according to a lower bound (LB) and an upper bound (UB) scenario and with drinking water produced from either groundwater or surface water (see Table 10 in section 4.3). The LB and UB scenarios reflect the most optimistic and conservative assumptions, respectively, about the presence of a PFAS reported at a concentration below an analytical limit. The LB exposure was expected to be closer to the actual exposure than the UB exposure. The calculated exposure was compared with the TWI of PFAS of 4.4 ng/kg body weight derived by EFSA, revealing that the high (P95) exposure (i.e. representing consumers with a long-term high exposure to PFAS) exceeded the TWI.

The mean LB exposure to PFAS was 4.6 ng PEQ/kg body weight per week through food and drinking water produced from groundwater and 5.9 ng PEQ/kg body weight per week through food and drinking water produced from surface water. The high (P95) LB exposure was 12 and 14 ng PEQ/kg body weight per week, respectively. The high (P95) LB exposure exceeded the TWI approximately three-fold. Therefore, exposure to PFAS through food and drinking water can result in adverse health effects. The TWI was also exceeded by the mean (i.e. 22 and 26 ng PEQ/kg body weight per week) and high (P95) (i.e. 45 and 51 ng PEQ/kg body weight per week) UB exposure to PFAS. The exposure to PFAS reported in the current study was approximately 40% lower than the exposure reported in 2021, even though 20 PFAS were included in the present study as opposed to four in 2021 (van der Aa et al., 2021).

In the current study, food contributed most to the exposure compared to drinking water (more than 70%). From the food groups included in the assessment, ‘fish and fish products’ contributed most to the exposure, followed by ‘drinks’ and ‘dairy’. The main contributing food products within these three food groups were cod, coffee and tea, and milk, respectively. The contribution of coffee and tea to the exposure was due to relatively higher PFUnDA concentrations in 5 out of 25 coffee and tea samples. As the source for these higher concentrations was unclear, and because coffee and tea are regularly consumed, further research is needed to establish the potential sources of PFUnDA in these drinks.

Drinking water from groundwater contributed less to the long-term LB exposure to PFAS than drinking water from surface water, because drinking water from surface water contained higher PFAS concentrations.

Examining the contribution of the individual PFAS to the exposure through food and drinking water from both groundwater and surface water revealed that the contributions were highest for PFUnDA, PFOS and PFDA, which together accounted for 63-78% of the LB exposure.

Considering that PFAS is present in most foods, the general advice to eat a varied diet to achieve the lowest exposure to contaminants is also important for PFAS. This way, people will not eat foods with a high PFAS concentration too often.

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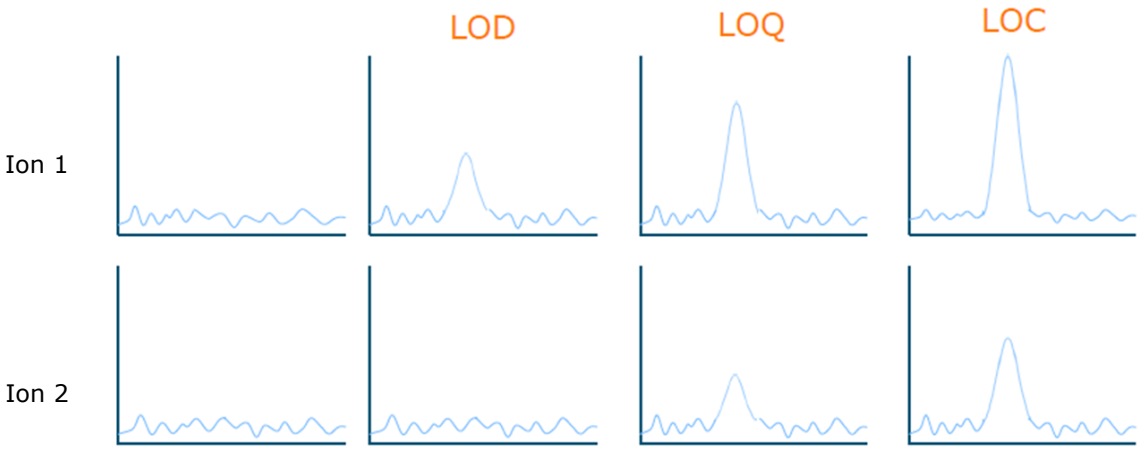
Appendix A Example peaks of ion transitions in food products



LOC: limit of confirmation; LOD: limit of detection; LOQ: limit of quantification

Appendix B PFAS concentrations in the food product samples

https://www.rivm.nl/bibliotheek/rapporten/2023-0011-bijlage.pdf

Appendix C PFAS concentrations in drinking water

https://www.rivm.nl/bibliotheek/rapporten/2023-0011-bijlage.pdf

Appendix D Lower bound and upper bound summed PFAS concentrations in each food product sample

https://www.rivm.nl/bibliotheek/rapporten/2023-0011-bijlage.xlsx

Appendix E Percentage contribution of the food groups, drinking water and the individual PFAS to the long-term lower bound (LB) and upper bound (UB) exposure distribution to PFAS through food and two drinking water types, and mean LB and UB exposure to PFAS, expressed as PEQ, through each food group, drinking water type, and individual PFAS, expressed as PEQ, through food and drinking water

Food groups and drinking water	**Percentage contribution per scenario and drinking water type[a]**				**Mean exposure to PFAS per scenario and drinking water type (ng PEQ/kg body weight per week)[a,b]**			
	From groundwater		**From surface water**		**From groundwater**		**From surface water**	
	LB	**UB**	**LB**	**UB**	**LB**	**UB**	**LB**	**UB**
Fish and fish products	30	11	24	11	1.4	2.9	1.4	2.4
Drinks	29	45	45	35	1.3	12	2.7	7.8
Drinks (excl. drinking water)	*23*	*11*	*18*	*14*	*1.1*	*2.9*	*1.1*	*3.1*
Drinking water	*5.9*	*33*	*27*	*21*	*0.27*	*8.7*	*1.6*	*4.7*
Dairy	17	11	13	13	0.78	2.9	0.77	2.9
Meat and meat products	7.9	8.9	6.1	10	0.36	2.3	0.36	2.2
Vegetables[c]	5.4	6.8	4.2	8.1	0.25	1.8	0.25	1.8
Eggs	3.7	2.0	2.9	2.4	0.17	0.53	0.17	0.53
Fruits (and nuts)	3.0	5.5	2.3	6.5	0.14	1.4	0.14	1.4
Vegetable fats and oils	2.4	3.5	1.9	4.1	0.11	0.92	0.11	0.91
Cereals and cereal products	1.1	5.6	0.86	6.6	0.051	1.5	0.051	1.5
Sugar	0	0.96	0	1.1	0	0.25	0	0.24

LB: lower bound; ng: nanogram; PEQ: PFOA equivalents; PFAS: per- and polyfluoroalkyl substances; UB: upper bound

[a] The allocation of PFAS concentrations according to the LB and UB scenario is explained in section 3.2.

[b] Mean LB and UB exposure was calculated by multiplying the percentage contribution for each food group and each drinking water type with the relevant mean exposure to PFAS (see Table 10 in section 4.3).

[c] The food group 'vegetables' also includes 'processed vegetables'.

PFAS[a]	Percentage contribution per scenario and drinking water type[b]				Mean exposure to PFAS per scenario and drinking water type (ng/kg body weight per week)[c]			
	From groundwater		From surface water		From groundwater		From surface water	
	LB	UB	LB	UB	LB	UB	LB	UB
PFUnDA	42	15	32	16	1.9	3.9	1.9	3.6
PFOS	20	5.6	19	6.6	0.92	1.5	1.1	1.5
PFDA	16	17	12	16	0.74	4.5	0.71	3.6
PFTrDA	6.2	10	4.8	6.7	0.29	2.6	0.28	1.5
PFNA	4.1	19	5.9	19	0.19	5	0.35	4.2
PFDoDA	3.9	12	3.0	10	0.18	3.2	0.18	2.2
PFOA	3.3	7.1	8.6	9.5	0.15	1.9	0.51	2.1
TFA	3.1	0.54	5.7	1.5	0.14	0.14	0.34	0.33
PFHpA	0.89	1.7	5.3	2.6	0.041	0.45	0.31	0.58
PFHxS	0.29	0.71	1.0	0.76	0.013	0.19	0.059	0.17
PFHpS	0.17	3.5	0.14	3.0	<0.01	0.92	0.008	0.67
PFTeDA	0.093	1.5	0.076	0.95	<0.01	0.39	<0.01	0.21
PFBA	0.080	0.096	0.62	0.17	<0.01	0.025	0.037	0.038
PFPeA	0.055	0.18	0.54	0.30	<0.01	0.047	0.032	0.067
PFPeS	0.015	0.38	0.085	0.14	<0.01	0.1	<0.01	0.031
PFHxA	0.011	0.023	0.097	0.042	<0.01	<0.01	<0.01	<0.01
HFPO-DA (GenX)	0.0033	0.95	0.12	0.10	<0.01	0.25	<0.01	0.022
PFDS	0.00019	5.9	0.12	5.9	<0.01	1.552	<0.01	1.31
PFBS	0.0012	0.0022	0.011	0.0043	<0.01	<0.01	<0.01	<0.01
ADONA	0	0.023	0	0.0067	0	<0.01	0	<0.01

LB: lower bound; ng: nanogram; PEQ: PFOA equivalents; PFAS: per- and polyfluoroalkyl substances; UB: upper bound

[a] The names of the PFAS can be found in Table 3 in section 2.2.2.

[b] The allocation of PFAS concentrations according to the LB and UB scenario is explained in section 3.2.

[c] Mean LB and UB exposure was calculated by multiplying the percentage contribution for each individual PFAS with the relevant mean exposure to PFAS (see Table 10 in section 4.3).

Appendix F Conversion of the dietary exposure to the EFSA-4 per age group for the Netherlands, as reported by EFSA (2020), into long-term exposure for the Dutch population

Age group (age range in years)	**Mean LB exposure (ng/kg body weight per week[a]**	**Number of years included in age range**	**Mean LB exposure (ng/kg body weight per week) x number of years**
Toddlers (1-2)	10.3	2	20.5
Other children (3-9)[b]	5.8	7	40.9
Adolescents (10-17)	3.8	8	30.6
Adults (18-64)	4.0	47	189
Elderly (65-74)[b]	5.5	10	54.6
Very elderly (75-79)	5.0	5	25.1
Total		**79**	**361**
Mean long-term exposure (ng/kg body weight per week)			**4.6**[c]

LB: lower bound; ng: nanogram

[a] Obtained from (EFSA, 2020), based on equipotency

[b] For these age groups, exposure was reported based on food consumption data derived from two DNFCSs (EFSA, 2020). The exposure based on the most recent DNFCS was used in the calculation.

[c] Calculated as 361÷79

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