UNITED NATIONS

UNEP

SC

UNEP/POPS/POPRC.12/INF/5

Distr.: General
18 July 2016

English only

Stockholm Convention on Persistent Organic Pollutants

Persistent Organic Pollutants Review Committee
Twelfth meeting
Rome, 19–23 September 2016
Item 4 (a) (ii) of the provisional agenda*

Technical work: consideration of draft risk profiles: pentadecafluorooctanoic acid (CAS No: 335-67-1, PFOA, perfluorooctanoic acid), its salts and PFOA-related compounds

Additional information related to the draft risk profile on pentadecafluorooctanoic acid (CAS No: 335-67-1, PFOA, perfluorooctanoic acid), its salts and PFOA-related compounds

Note by the Secretariat

As referred to in the note by the Secretariat on a draft risk profile on pentadecafluorooctanoic acid (CAS No: 335-67-1, PFOA, perfluorooctanoic acid), its salts and PFOA-related compounds (UNEP/POPS/POPRC.12/3), the annex to the present note sets out additional information related to the draft risk profile on PFOA, its salts and PFOA-related comopunds. The present note, including its annex, has not been formally edited.

* UNEP/POPS/POPRC.12/1.

Annex

Pentadecafluorooctanoic acid (CAS No: 335-67-1, PFOA, perfluorooctanoic acid), its salts and PFOA-related compounds

ADDITIONAL INFORMATION

Draft prepared by the intersessional working group on PFOA, its salts and PFOA-related compounds under the POPs Review Committee of the Stockholm Convention

June 2016

Table of content

1. Introduction

1. The present document is a background document to the draft risk profile on PFOA, its salts and PFOA-related compounds prepared by the intersessional working group on PFOA, its salts and PFOA-related compounds of the Persistent Organic Pollutants Review Committee (third draft from April 2016).

2. Background information is provided in relation to the chapters and according to the structure of the draft risk profile (e.g. information on releases to the environment is compiled in section 2.3 of the risk profile and relevant background information is provided in section 2.3 of the present document).

1.1 Chemical identity

Background information on the chemical identity from UNEP/POPS/POPRC.11/5

3. UNEP/POPS/POPRC.11/5 provides tables of data for PFOA salts and PFOA-related compounds. The tables are reproduced here for ease of reference:

Table 1: Some examples of PFOA salts (e.g. Environment Canada and Health Canada, 2012; OECD 2007, 2011)

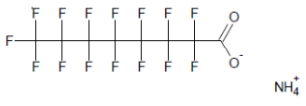
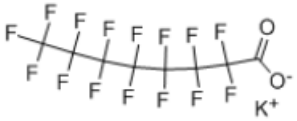
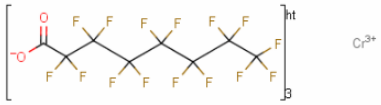
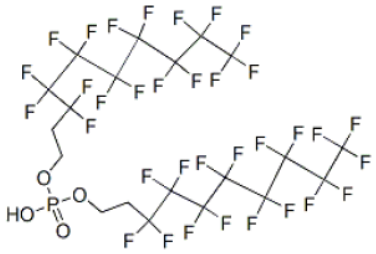
Name:	Abbreviation / Structure:	CAS number:
2,2,3,3,4,4,5,5,6,6,7,7,8,8,8-penta-deca-fluoro-octanoic acid, ammonium salt	APFO 	3825-26-1
2,2,3,3,4,4,5,5,6,6,7,7,8,8,8-penta¬deca¬fluoro-octanoic acid, sodium salt	Na-PFOA 	335-95-5
2,2,3,3,4,4,5,5,6,6,7,7,8,8,8-penta-deca-fluoro-octanoic acid, potassium salt	K-PFOA 	2395-00-8
2,2,3,3,4,4,5,5,6,6,7,7,8,8,8-penta-deca-fluoro-octanoic acid, silver salt		335-93-3
Octanoic acid, 2,2,3,3,4,4,5,5,6,6,7,7,8,8,8-pentadecafluoro-, chromium(3+)		68141-02-6
Ethanaminium, N,N,N-triethyl-, salt with pentadecafluorooctanoic acid (1:1)	-	98241-25-9

Table 2: Overview of relevant physicochemical properties of a PFOA salt (APFO)

Property	Value	Reference
Physical state at 20°C and 101.3 kPa	Solid	Kirk, 1994
Melting/freezing point	APFO: 157-165 °C (decomposition starts above 105 °C) APFO: 130 °C (decomposition)	Lines and Sutcliff, 1984 (IUCLID 2.2.) 3M Company, 1987 cited in ECHA, 2013b
Boiling point	Decomposition	Lines and Sutcliff, 1984 (IUCLID 2.2.)
Vapour pressure	APFO: 0.0081 Pa (6 x 10-6) at 20 °C, calculated from measured data	Washburn et al., 2005
Water solubility	Concentration. at saturation (g/L)	3M Company, 1987 cited in

Property	Value	Reference
	APFO: > 500	ECHA, 2013b
n-Octanol/water partition coefficient, K_{ow} (log value)	Experimental: No data Calculated: No data.	-
Dissociation constant	Dissociation Constants: pKa = 2.80 in 50% aqueous ethanol pKa = 2.5	Brace, 1962 Ylinen et al., 1990

Table 3: Examples of PFOA-related substances

Name:	Abbreviation / Structure:	CAS number:
Fluorotelomer alcohols 8:2 FTOH	8:2 FTOH 	678-39-7
Fluorotelomer acrylates 8:2 Fluorotelomer acrylate	8:2 FTAC 	27905-45-9
Polyfluoroalkyl phosphoric acid diesters 8:2 Fluorotelomer phosphate diester	8:2 diPAP 	678-41-1
Perfluorinated Iodides Perfluorooctyl iodide	PFOI 	507-63-1

4. For further information, please also consider the non-exhaustive list of PFOA, its salts and PFOA-related compounds, as provided by the Secretariat of the Stockholm Convention on POPs at: http://chm.pops.int/TheConvention/POPsReviewCommittee/Meetings/POPRC11/POPRC11Followup/tabid/4723/Default.aspx

Table 4: Overview of relevant physicochemical properties of 8:2 FTOH

Property	Value	Reference
Physical state at 20°C and 101.3 kPa	waxy solid	
Melting/freezing point	No information	Lines and Sutcliff, 1984 3M Company, 1987
Boiling point	No information	Lines and Sutcliff, 1984 (IUCLID 2.2.)
Vapour pressure	31 Pa at 25 ºC (Retention time method) 29 Pa at 45ºC (Headspace GC/AED method) 254 Pa at 25 ºC , volatile, 99.9 %	Vapour pressure seem sensitive to choice of method. Cobranchi et al 2006

Property	Value	Reference
	detected mainly in the gassousphase in the atmosphere 0.227 kPa 0.023 mmHg	Stock et al. 2004 Lei et al., 2004 Berti, 2002 cited in ECHA, 2014
Water solubility	1.4 x 10^{-4} g/L or 140 µg/L at 25 °C	Berti, 2002 cited in ECHA, 2014
n-Octanol/water partition coefficient, K_{ow} (log value)	No information	-
Dissociation constant	No information	-

1.2 Conclusion of the Review Committee regarding Annex D information

1.3 Data sources

1.4 Status of the chemical under international conventions

2. Summary information relevant to the risk profile

2.1 Sources

2.1.1 Production, trade, stockpiles

Background information on production

(For references, see draft risk profile)

5. Production of APFO for commercial use has occurred since 1947 (Simons, 1949; OECD, 2006). Historically, on average between 5-25 tonnes were produced globally per annum between 1951-1964. The global volume of production gradually increased over time with the most recent data available for production between 1995-2002 showing an average of between 200-300 tonnes per annum (OECD, 2006). ECHA 2015a provides a summary of global production volumes of PFOA, its salts and PFOA-related substances (accumulated PFOA manufacturing 1951 of 2004: 3,600 to 5,700 t (Prevedouros et al, 2006;); annual APFO/NaPFO consumption in fluoropolymer manufacturing 2011 to 2015: 127 to 731 t; annual fluorotelomer acrylates production with eight carbon fluorinated side-chains 1995 to 2004: 1650 to 2145 t; annual fluorotelomer-based products production of 45 000 t with the assumption of a share of 30 % PFOA-related substances in 2014: 13.500 t; ECHA 2015a). According to the Umweltbundesamt 2009 the world-wide production of FTOHs is estimated at 11,000 – 14,000 t/y. The available information indicates that production is still ongoing, however, no complete data on global production volumes is available. At present, and in line with commitments under the US EPA Stewardship Programme to reduce emissions in the EU, Japan and the U.S., it is believed that production has declined in these countries since no PFOA or its salts will be used in fluoropolymer manufacture after 2015. However, PFOA and its salts are expected to still be used by manufacturers who are not bound to the US EPA PFOA Stewardship Programme, for example in China where current production of PFOA is predominantly carried out. Manufacturers under the Stewardship Programme do not represent all manufacturers worldwide. For fluoropolymers the global market share of the signatory companies is estimated to be 69% in 2011 with a decreasing trend (see ECHA, 2015a).

Background information on trade

(For references, see draft risk profile)

6. Imports of PFOA and its salts to the EU are likely to be below 20t/a (based on data from 2012), with a decreasing trend since 2008. Data for the EU for 2014 shows that ~20 tonnes of PFOA was imported annually for use in the semiconductor industry (<0.05 tonnes), the photo industry (1 tonne), the manufacture of fluoropolymers (<20 tonnes), and other uses (between 0.5-1.5 tonnes) (ECHA, 2015a). Use of PFOA-related substances in the EU includes textile treatment (~1000 tonnes), fire fighting agents (>50 – 100 tonnes), paper treatment (>150 – 200 tonnes), paints and inks (>50 – 100 tonnes) and others uses (>0.1 – 0.5 tonnes) (ECHA, 2015a). According to estimates provided in the same report, imports are forecast to fall to <0.15 tonnes per annum after 2015, mainly as a result of the phasing out of the use of PFOA in the manufacturing of fluoropolymers in those companies participating in the US EPA Stewardship Programme. Moreoever, imports are expected to fall in the photo industry as a result of a shift to digital applications (ECHA, 2015a). However, the report notes a steady 5-6% annual increase in the global fluoropolymer market, primarily from manufacturers not participating in the voluntary US EPA Stewardship Programme. Additional information concerning the semiconductor industry, as provided by industry, indicates that the volume of PFOA imported by the EU annually by all semiconductor manufacturers is <8.40 kg per year (SIA, 2015). PFOA and its salts

are also imported into the EU in mixtures, in particular in fluoropolymer dispersions that are imported for further processing. The import of PTFE is expected to be within the range of 3 to 16 t/a. PFOA and its salts are also imported as residuals or impurities in articles containing fluoropolymers (produced with PFOA and its salts) or PFOA-related substances. The import volume of PFOA and its salts in fluorotelomer-based consumer products is estimated (with high uncertainty) to be below 10 t/a (ECHA, 2015a). Lastly, data for the EU for 2014 imports of PFOA-related substances shows that between 100-1,000 tonnes were imported per annum for textile and leather treatment, and that this volume of imports is likely to continue in the immediate future (ECHA, 2015a).

Background information on production volumes of PFOA and PFOA-related substances on the global market from ECHA 2015a

(For references, see ECHA 2015a)

7. Table 5 summarises available estimations on global production volumes of PFOA, PFOA-salts and PFOA-related substances.

Table 5: Summary of global production volumes of PFOA, PFOA salts and PFOA-related substances, extracted from ECHA (2015a)

	Accumulated PFOA manufacturing (Prevedouros et al., 2006)	Annual APFO/NaPFO consumption in fluoropolymer manufacturing (Wang et al., 2014)	Annual fluorotelomer acrylates production with eight carbon fluorinated side-chains (van Zelm et al., 2008).	Annual fluorotelomer-based products production 45 000 t (Wang et al., 2014), assumption 30 % PFOA-related substances
1951-2004	3,600 – 5,700 t			
2011-2015		127 - 731 t		
1995 to 2004			1,650 to 2,145 t	
Currently				13,500 t

2.1.2 Uses

2.1.3 Releases to the environment

Background information on releases to the environment

(For references, see draft risk profile)

8. Numerous direct and indirect sources of PFOA, its salts and PFOA-related substances contribute to the overall release of PFOA to the environment. Direct releases to the environment occur from the production of the raw substance, during the processing, use and disposal of the chemical, including the processing, use and disposal of products intentionally or unintentionally containing the chemical, and that of articles treated with commercial and domestic surface treatment products, as well as during the service life of these products. Main emission vectors are water, wastewater and dust particles. Indirect releases to the environment occur due to the formation of PFOA from PFOA -related substances. Certain PFOA-related substances, such as 8:2 FTOH, are volatile substances. They are released to air and waste water during manufacture of the substances themselves, from side-chain fluorinated polymers and during use and disposal of consumer articles treated with PFOA-related substances. When emitted to the atmosphere, they can be degraded to PFOA, and deposited on soil or surface waters. They are also washed out from the atmosphere via precipitation. In soil it has been shown that PFOA-related substances can be biotically degraded to PFOA (ECHA 2015a).

9. ECHA 2015a provides estimated global cumulated historical emissions and projected future emissions of PFOA based on Wang et al. (2014a), according to which the relevance of the releases from 1951 to 2002 was and is most relevant from (1) manufacturing of fluoropolymers with PFOA (2790-3850 t[1]), followed by (2) PFOA manufacture (200-570 t[1]), (3) fluoropolymer dispersion use and disposal (210-209 t[1], (4) degradation of fluorotelomer based products (50-2400 t) and finally (5) impurities in fluorotelomer based products (17-34 t). Recent degradation

[1] Plausible scenario according to Wang et al. 2014a.

studies show that up to 40% of the initial 8:2 FTOH are degraded to PFOA after 7 months. Therefore it can be assumed that after a longer time period, PFOA yield will be even higher than estimated by Wang et al. (2014a). Furthermore, Wang et al. did not quantify emissions from the degradation of side-chain fluorinated polymers (which is the major use of PFOA-related substances). The importance of indirect sources of PFOA in the environment, in particular atmospheric degradation of residuals in fluorotelomer-based products, has been highlighted by Ellis et al. (2004a) (see ECHA 2015a). In Wang et al. 2014c the authors identify emission sources of PFCAs that have not been discussed at that time: (1) PFCAs released as ingredients or impurities; (2) PFCAs formed as degradation products; and (3) sources from which PFCAs are released as both impurities and degradation products. Available information confirms that these sources were active in the past or are still active today, but due to a lack of information, it is not yet possible to quantify emissions from these sources. Some of the sources may have been significant in the past, whereas others can be significant in the long-term (e.g., (bio)degradation of various side-chain fluorinated polymers where PFCA precursors are chemically bound to the backbone (see Wang et al. 2014c).

10. The manufacturing of PFOA has been identified as a major direct source of PFOA in the environment (Armitage et al., 2009; Prevedouros et al., 2006). During the manufacturing of PFOA the substance can be emitted into the environment either via waste water or into the air. It was reported that PFOA emissions from the largest electro chemical fluorination production plant, located in the United States were approximately 5- 10% of the annual production and about 5% PFOA have been emitted to air and 95 % to water (Prevedouros et al., 2006). Prevedouros and co-workers (2006) estimated global PFOA manufacturing emissions: 45 t in 1999, 15 t in 2004, 7 t in 2006. PFOA can also be directly released from manufacturing and use of PFOA-related substances (see ECHA 2015a).Measured data show that industrial emissions from European fluoropolymer manufacturing sites have a significant influence on the PFOA levels found in European surface waters (Pistocchi and Loos, 2009 ; Loos et al., 2008 ; Dauchy et al., 2012). China stated that no data concerning releases to the environment are available (China 2015). However, a recent publication documents the first source-specific inventory for environmental releases of PFOA/PFO in China from 2004 to 2012 and estimates cumulative environmental releases reaching 250 tonnes (t) over a period of nine years (Li et al., 2015). A study conducted between August 2013 and September 2014 investigated the occurrence and distribution of legacy PFASs including PFOA as well as a PFOA alternative, hexaflouropropylene oxide dimer acid (HFPO-DA, marketed under the name GenX), in surface waters from river/estuary systems. Notably from this study, the Chinese samples were highly polluted by an industrial point source discharging mainly PFOA (Heydebreck et al., 2015). Another study found a single production facility in China that produces PTFE and other fluoropolymers with massive capacity emitting extraordinary high quantities of PFOA at 159 kg/day into a river (Wang et al., 2016). Lastly, a study investigating the spatial trend of PFOA along Xiaoqing River in China observed a positive correlation between the proximity to a fluoropolymer production facility and PFOA concentration. The range of riverine discharges of PFOA was estimated to be 23-67 t/yr, which was in agreement with the theoretical calculations of PFOA emission from fluoropolymer production (68 t/yr) (Shi et al., 2015). Substantial levels of per- and polyfluoroalkyl substances were detected in outdoor dust in mainland China with PFOA being a predominant substance (Yao et al. 2016).

11. Historic releases to the environment from production are available for the U.S. as provided by Dupont (previously a large scale user of APFO), which estimated combined releases from its plant in West Virginia of 8,636 kg into air in 1984 (Paustenbach et al. 2007;Lerner 2015), 39,463 kg into air and water (14,062 kg and 25,402 kg to air and water respectively) in 1999, 36,288 kg (14,062 kg and 22,226 kg to air and water respectively) in 2000, 4,990 kg (2,722 kg and 2,268 kg to air and water respectively) in 2003, and 771 kg (91 kg and 680 kg to air and water respectively) in 2004 (Emmett, 2006). All together, DuPont released approximately 1,136 t of PFOA into the air and water around its West Virginia plant between 1951 and 2003 (Paustenbach et al., 2007; Lerner 2005). In 1999, DuPont released more than 11,363 kg into air and water at its plant in New Jersey, USA (Lerner 2005).

12. 3M was a previous large scale producer of perfluoroalkyl substances (PFASs) from the late 1940s until 2002. 3M produced PFASs, including PFOA, at its plant in Minnesota (Minnesota State Dep, 2016; Oliaei et al., 2013). An environment assessment at this site has determined that groundwater is contaminated with both PFOA and PFOS (Minnesota State Dep, 2016). Wastes were disposed of on-site in a specially designed pit and the water treatment plant did not remove the substances, which found their way to the Mississippi River (Minnesota State Dep, 2016). Average PFOA levels in the water discharge were 2463 – 2989 ng/L (Oliaei et al., 2013). The Minnesota Pollution Control Agency estimated 3M released 4,545 kg of fluorocarbons into the Mississippi River in 2001 including 1,047 kg of PFOA (Minnesota Public Radio, 2016). In 2004 in Minnesota, PFCs were first found to have contaminated drinking water supplies in parts of the eastern Twin Cities area (Minnesota Pollution Control Agency, 2016).

13. The manufacture of fluoropolymers is considered the main direct emission source of PFOA, where APFO is used as processing aid (Armitage et al., 2009; Prevedouros et al., 2006). From fluoropolymer production sites, PFOA is emitted to air (mainly particle bound) and water (ECHA 2015a). Fluoropolymer dispersions are often used to coat metal and fabric surfaces. A significant share is sold as aqueous fluoropolymer dispersions, which still contain APFO (typical content about 2000 ppm, up to 7000 ppm). During the dispersion processing it was found that a significant share (38%) is released to the environment (e.g. to air, waste water and solid waste). When not emitted during fluoropolymer dispersion processing, PFOA can be emitted during the subsequent use and disposal of consumer

articles (ECHA 2015a). Environmental releases from the direct use of PFOA are also possible from the photo industry, semiconductor industry (see ECHA 2015a).

14. Releases to the environment during the processing of PFOA were reported by the semiconductor industry with estimated releases to wastewater at between 3-3.8% of the total quantity used. The sector reported no emissions to air from semi-conductor manufacture (ESIA 2015, SIA 2015).

15. The US EPA reported releases to the domestic indoor environment from use of products containing PFOA and its related compounds for several products purchased from retail in the US, with the largest releases from use reported for professional carpet-care liquids, pre-treated carpeting, floor waxes and stone/tile/wood sealers and household textiles and upholstery. The range of PFOA concentrations found in these domestic products was between non-detectable and 6,750 ng/g (US EPA, 2009). The German Federal Environment Agency reported that the use of outdoor jackets is one source of PFOA and PFOA-related substances into the environment. Emissions occur mainly during washing and impregnation procedures (Umweltbundesamt, 2014).

16. The use of PFOA-related substances results in direct (PFOA as impurity) and indirect emissions of PFOA (degradation of PFOA-related substances) from impurities in fluorotelomer based products and from degradation of fluorotelomer based products and from the manufacture, and use and disposal of side-chain fluorinated polymers (see (ECHA 2015a) and (Environment Canada and Health Canada 2012)). The manufacture of side-chain fluorinated polymers represents one major industrial use of PFOA-related substances. Russell et al. (2008) estimate that 2% of PFOA-related substances remain unbound in the polymeric material. PFOA-related substances are produced and traded in large amounts. PFOA-related substances are considered a relevant emission source of PFOA from several uses such as aqueous fire fighting foams, surface treated textiles, surface treated paper or from paints and inks (see ECHA 2015a). An assessment of sources of PFOA to the Baltic Sea estimated that 30% of the releases were due to transformation of fluorotelomers (Danish Ministry of Environment, 2013).

17. Waste consisting of or contaminated with PFOA, its salts or PFOA- related substances arises during all different life cycle steps of these chemicals. Releases to the environment occur from management of waste water and of solid waste. Industrial wastewater from fluoropolymer manufacturing is considered the most important point source of PFOA. In addition, PFOA and other PFASs are emitted from municipal wastewater treatment plants originating from several sources such as textile, photo industry, landfills, and electroplating (ECHA 2015a). Waste water treatment plants do not remove PFOA efficiently (Schultz et al., 2006; Bayerisches Landesamt für Umwelt, 2010). Thus, a large share remains in the water phase and enters surface water bodies (see e.g. Houtz et al. 2016). Degradation of precursor substances during the treatment can even lead to higher PFOA emissions (Schultz et al., 2006). Concentrations of some PFCs, particularly PFOA, were slightly higher in WWTP effluent than in influent, suggesting that biodegradation of some precursors contributes to the increase in PFOA concentrations in wastewater treatment processes (Loganathan et al. 2007). As shown by Vierke et al., wastewater treatment plants are also an important source of atmospheric PFASs emissions (Vierke et al., 2011). PFOA can be bound to sewage sludge. The use of sludge from municipal wastewater treatment plants for soil fertilization poses a potential source for PFOA in the environment (van Zelm et al., 2008). During solid waste management PFOA releases may result from incineration, landfilling and recycling. Yamada et al. (2005) conclude that under typical municipal waste incineration conditions no significant amounts of PFOA would be formed by incineration of a textile or paper substrate treated with a fluorotelomer based acrylic polymer. Their conclusion is questioned by Poulsen et al. (2005). It is expected that EU-wide incineration and landfilling are the most common disposal routes. From landfills, PFOA and related substances can volatilize and contaminate the atmosphere or they may leach out into soil and groundwater. Further, closed landfills may still be a potential source of PFOA leaching. It is assumed that recycling of contaminated wastes (e.g. paper containing PFOA-related substances) contributes to environmental releases and that the contaminants may again circulate through use, disposal and recycling phase of products. The best possibility to prevent emissions of PFOA and related substances is to reduce their contents in products. (see ECHA 2015a).

18. There are a number of studies available that include measurements of direct releases to the environment from the disposal of PFOA and its related compounds. For example, Muir and Scott (2003), and Boulanger et al. (2005) reported the presence of PFOA in sewage treatment plant (STP) effluents entering the Great Lakes (with measured PFOA concentrations ranging between 0.7–0.9 ng/g and 7–55 ng/L). Boulanger et al. (2005) also reported the presence of PFOA from releases to the environment from wastewater treatment plants (WWTP) (with measured PFOA concentrations ranging between 41.2-2,420 ng/L). Raw and treated landfill leachate samples were collected from 5 municipal landfill sites in China to measure the concentrations and contamination profile of perfluoroalkyl acids (PFAAs) in leachate. Total concentration of PFAAs (∑PFAAs) ranged from 7280 to 292,000 ng/L in raw leachate and from 98.4 to 282,000 ng/L in treated leachate. The dominant compounds measured were PFOA (mean contribution 28.8% and 36.8% in raw and treated leachate, respectively) and Perfluorobutanesulfonic acid (PFBS), (26.1% and 40.8% in raw and treated leachate, respectively). The Chinese national leakage of ∑PFAAs from landfill leachate to groundwater was estimated to be 3,110 kg/year (Yan et al., 2015). Ikonomou (2006) reported the presence of PFOA resulting from releases to multiple media from landfill sites in Canada. PFOA concentrations were measured at several sites with releases to soil ranging between 22-1,083 ng/g, releases to water up to 2040 ng/L, and releases to air from < 5,000 pg/m^3 upwind and < 25,000 pg/m^3 on-site. PFOA has also been detected in untreated landfill

leachates in Germany (Busch et al., 2010). Analysis of PFOS in 16 sewage sludge samples from China resulted in detection of PFOA in most sewage sludge samples with concentrations up to 4780 ng/g (oven dry weight) (Guo, 2008). Data from the U.S. for a number of agricultural sites in Alabama where sewage sludge was applied (sewage sludge sourced from a local wastewater treatment plant) showed that PFOA was recorded in 29 of the 51 samples with concentration levels of up to 11,000 ng/L and exceeding 388 ng/L in 13 of the 51 samples (US EPA, 2009).

19. Data collected from a number of OECD countries in 2008 indicates that action to mitigate releases to the environment from waste disposal has been undertaken in many waste disposal sites. Of the survey undertaken, 38 perfluorinated chemicals were released or transferred offsite during the year 2008. The majority (<225 tonnes) was disposed of by incineration, or reused or recycled (of which <201 tonnes was PFOA and related compounds). PFOA was one of the perfluorinated chemicals in the survey with the largest quantities transferred off-site for incineration or recycling (136 tonnes). No quantitative information on the release of APFO was provided in the survey responses (OECD 2011).

20. PFOA-related compounds will degrade to PFOA in sludge, soil, water and air (ECHA 2015a; ECHA 2013a; IPEN 2015), and such degradation has been noted across many products containing PFOA-related substances, including products such as grease proof paper (Dasu et al., 2013; Ellis et al., 2003; Ellis et al., 2004a; Frömel and Knepper, 2010; Gauthier and Mabury, 2005; Hilal et al., 2004; Jackson and Mabury, 2013; Rayne and Forest, 2010; Renner, 2008; Wang et al., 2005a; Wang et al., 2005b; Washington et al., 2009; Young and Mabury, 2010; Young et al., 2008; Zhang et al., 2013; Butt et al., 2014; Rankin et al., 2014; Washington et al., 2015). Release of PFOA to the environment results from its use in fire fighting foams. Aqueous film forming foams (AFFF) were developed in the early 1960s for efficiently extinguishing hydrocarbon-fuel fires. PFASs are key components in this formulation because they lower the surface tension at the air-foam interface and form a film over the hydrocarbon fuel to prevent reignition. An investigation of the occurrence and fate of 15 perfluoroalkyl substances (PFASs) and one fluorotelomer sulfonate from a firefighting training ground that was contaminated by intensive use of aqueous film forming foams (AFFF) assessed contamination levels and their spatial and vertical distribution. The kinetics of desorption of PFOS, PFOA and 6:2FTS from the concrete into an overlaying static water volume was measured under field conditions, suggesting that firefighting training pads might remain as a source of PFOA (t0.5=1 year) (Baduel et al., 2015). The Australian Government Department of Defence (Defence) and other organisations in Australia have used aqueous film forming foam (AFFF) since the 1970s to suppress liquid fuel fires. Historically, AFFF used by Defence and other organisations has contained the chemicals perfluorooctane sulfonate and perfluorooctanoic acid. These chemicals have been detected in the soils and waterways in the vicinity of RAAF Base Williamtown and the Army Aviation Centre, Oakey. Defence has reviewed its properties to identify where AFFF may have been used. At this stage, Defence has identified 16 sites for detailed environmental investigation (Australia 2016).

21. There are no known natural sources of PFOA or their related compounds (Kissa, 1994; Environment Canada and Health Canada 2012; IPEN 2015).

Background information on transformation and degradation of PFOA-related substances from ECHA 2015a

(For references, see ECHA, 2015a)

22. The following italicised text has been taken directly from ECHA (2015a).

"PFOA-related substances can be degraded to PFOA under environmental conditions. PFOA-related substances all show a similar structural feature. The non-degradable perfluorinated carbon chain (C_8F_{17}-X) attached to a degradable non-fluorinated moiety. Thus, the substances are structurally similar. Using the weight of evidence approach it seems very likely that also similar substances may degrade in a similar way in the environment. At the end of a number of degradation steps PFOA may most probably be the end product and persist in the environment.

8:2 FTOH

8:2 FTOH metabolism universally show the formation of perfluorooctanoate (PFOA) and, to a smaller fraction, perfluorononanoate (PFNA) and lower-chain-length PFCAs (Butt et al., 2014).

Dinglasan et al. investigated biodegradation of 8:2 FTOH using mixed microbial system (Dinglasan et al., 2004). The enrichment culture was obtained from sediment and groundwater from a contaminated site. By day 81, PFOA was detected at 3% of the total mass of added 8:2 FTOH. 8:2 fluorotelomer unsaturated carboxylic acid (8:2 FTUCA) was identified as major metabolite at day 81 (~50% of the total mass). Further degradation of 8:2 FTUCA may lead to an increase of PFOA concentration (see Figure below). By day 81 only 55% of products could be accounted. There may be a number of reasons for the loss: volatile metabolites may have been lost during routine sampling (loss of initial 8:2 FTOH ~20% in sterile control), volatile metabolites that were left unidentified or unsaturated metabolites which are covalently bound to biological macromolecules.

Biodegradation of ^{14}C-labelled 8:2 FTOH has been investigated in mixed bacterial culture and in activated sludge (Wang et al., 2005a; Wang et al. 2005b). The mixed bacterial culture was obtained from sludge from

an industrial wastewater treatment plant (WWTP). Meanwhile, the second study was performed with inoculums from a domestic WWTP (200-fold diluted). The results showed that 8:2 FTOH is adsorbed to sludge and degraded subsequently. A significant portion of the $_{14}$C 8:2 FTOH had volatilized from the solid/aqueous matrix and deposited onto the PTFE septa of the experimental vessels. 36% of $_{14}$C 8:2 FTOH remained in the mixed bacterial culture at day 90 (Wang et al. 2005a) and 57% of the parent still remained in the activated sludge system after 28 days (Wang et al. 2005b). In the mixed bacterial culture system the concentration of PFOA increased over 56 days and levelled off to 6% of the $_{14}$C mass balance until day 90. Approximately 25% of the sum of 8:2 fluorotelomer carboxylic acid (8:2 FTCA), 8:2 fluorotelomer unsaturated carboxylic acid (8:2 FTUCA) and 7:2 fluorotelomer secondary alcohol (7:2 sFTOH) were detected at day 90. These substances are degradation intermediates and can be further degraded to PFOA (see Figure B.4-1) (Wang et al. 2005a). In the activated sludge system 2.1% PFOA and 33% sum of 8:2 FTUCA and 8:2 FTCA of the initial $_{14}$C mass have been identified after 28 days (Wang et al. 2005b). Similar degradation pathways were observed in aerobic soil, whereby formation of PFOA were higher in the soil compared to mixed bacterial cultures and activated sludge. 10 – 40 % (average 25%) of $_{14}$C- 8:2 FTOH (half-life (primary degradation) < 7 days) was degraded to form PFOA (steady state after 7 – 56 days; test duration 197 days) (Wang et al., 2009). 10-35% of total $_{14}$C was irreversibly bound to soil, whereby PFOA was not irreversibly bound to soils.

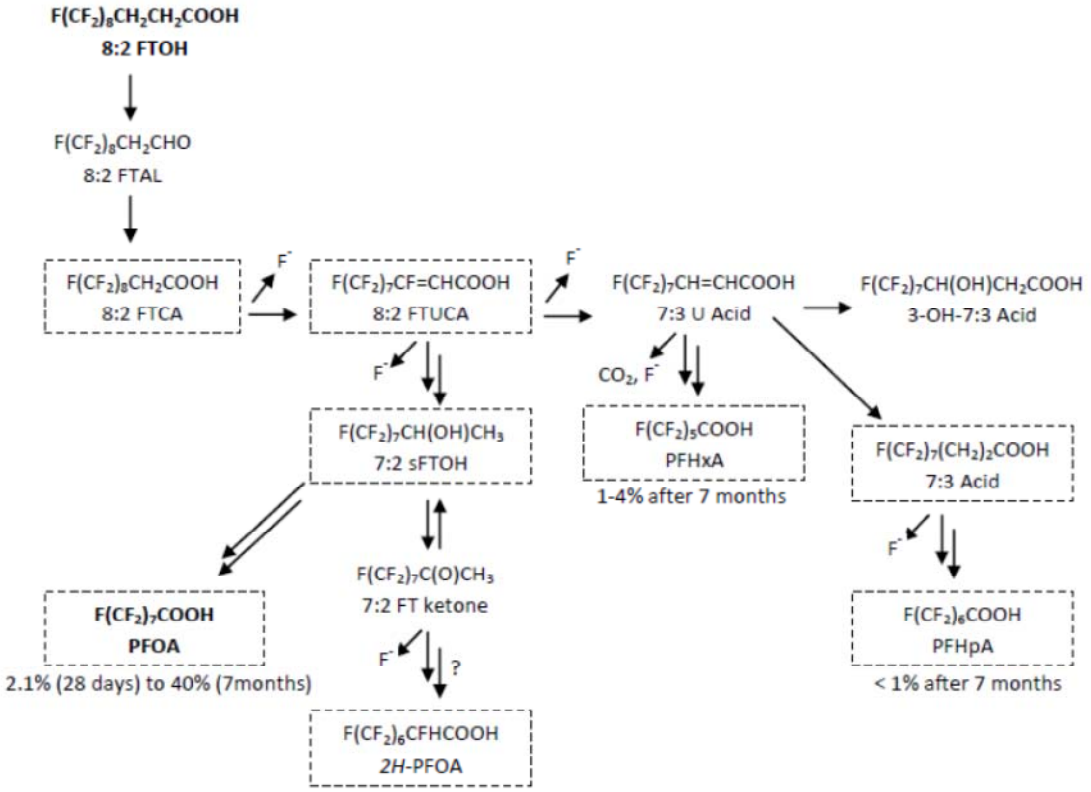


***Figure**: Aerobic degradation pathways of 8:2 FTOH in soil and activated sludge*

(Figure based on Liu and Mejia Avendano (2013)). The double arrows indicate multiple transformation steps. Defluorination reactions are indicated by release of fluoride ions (F−). Stable and semi-stable compounds are shown inside dashed boxes. 2H-PFOA has been proposed, but it has not been successfully validated as a PFOA degradation product. (Liu and Mejia Avendano, 2013). The percentages of the degradation products refer to studies by (Dinglasan et al., 2004; Wang et al., 2005a; Wang et al., 2009; Wang et al., 2005b). This figure was copied from ECHA, 2015a.

Anaerobic degradation of 8:2 FTOH under methanogenic conditions has been analysed by Zhang et al., (Zhang et al., 2013). Anaerobic digester sludge was incubated dosed with [3-$_{14}$C] 8:2 FTOH for 181 days. The half-life of 8:2 FTOH (primary degradation) is about 145 days. PFOA formation was much lower compared with the results of the aerobic sludge and soil studies (0.3 mol% of initially applied [3-$_{14}$C] 8:2 FTOH within 181 days). Approximately 39 mol% of the added 100 mol% [3-$_{14}$C] 8:2 FTOH still remained by day 181. 23 mol% of intermediate transformation products (sum of 8:2 FTCA and 8:2 FTUCA) were detected at day 181 2H, 2H, 3H, 3H-Perfluordecanic acid (7:3 acid) was detected as a stable degradation product (27 mol%). The results on anaerobic degradation obtained by Zhang may be relevant for conditions such as landfill leachate and anaerobic WTTP sludge.

Ellis and co-workers studied the kinetics of the reactions of Cl atoms and OH radicals with a series of fluorotelomer alcohols with differing chain lengths (2:2; 3:2, 4:2 FTOH) in 700 Torr of N_2 or air, diluent at 296 +/- 2K. Interestingly, the length of the perfluorinated carbon chain residue had no discernible impact on the reactivity of the molecules. The authors conclude atmospheric life-time of the FTOHs of 20 days by reaction with OH radicals (Ellis et al., 2003).

Atmospheric degradation was further studied in a smog chamber (Ellis et al., 2004). Experiments were performed in 750 Torr of air at 296 K. Reaction mixtures were subject to 0.5 to 15 min UV radiation leading to a consumption of FTOH in the range of 66 to >98%. It was shown that 8:2 FTOH is oxidized, initiated by Cl atoms which represent OH radicals, and forms PFNA, PFOA (1.5% C mass balance of 8:2 FTOH) and shorter chain PFCAs. The formation of PFOA is expected to be greater, because intermediate transformation products were still observed (e.g. 26% 8:2 FTCA, 6% 8:2 fluorotelomer aldehyde (8:2 FTAL)). The authors stress that the formation of PFOA is small but significant and postulate that FTOH degradation is likely an important source of PFOA and other PFCAs in remote areas.

The aqueous phase photo-oxidation of 8:2 FTOH in aqueous hydrogen peroxide solution, synthetic field water, and water from Lake Ontario (Canada) was investigated by Gauthier and Mabury (Gauthier and Mabury, 2005). The half-lives of 8:2 FTOH were 0.83. ± 0.20 hours (10 mM H_2O_2), 38.0 ± 6.0 hours (100µM H_2O_2), 30.5 ± 8.0 to 163.1 ± 3.0 hours (synthetic field water), and 93.2 ± 10.0 hours (Lake Ontario). The major products detected in the H_2O_2 study after 10 hours were 8:2 FTCA (~60%) and PFOA (~40%). During the experiment 8:2 FTAL was observed as a short-lived intermediate that underwent further photo-oxidation to PFOA. 8:2 FTCA was shown to undergo aqueous phase photo-oxidation leading to PFOA as the major product. It therefore appears that aqueous phase photo-oxidation of 8:2 FTOH will result in 75-100% PFOA with time. In the other test systems 1-8% (after 140-146 hours; synthetic field water) and 18% PFOA (duration not specified; Lake Ontario), respectively, were formed. Although the study is only of qualitative nature (no rate coefficients reported), it shows that fluorotelomer alcohols and other related compounds will undergo photo-oxidation in aqueous surface layers and in the atmospheric aqueous phases (cloud droplets and deliquescent particles). Since the PFOA yield from 8:2 FTOH photo-oxidation is 75-100% in the aqueous phase (compared to 3-6% in the gas phase), aqueous phase photo-oxidation may turn out to be very important in spite of the low solubility. Any quantitative statements will require multiphase modelling.

Kudo et al. (2005) investigated the biotransformation of 8:2 FTOH in male mice dosed via intraperitoneal injection and the diet. The PFOA levels in the animals continued to rise throughout the experiment. In the experiment where the male mice where exposed to 8:2 FTOH via the diet, the PFOA levels increased in a dose- and time dependent manner. The formation of PFOA was around 10 times higher than that of PFNA (Kudo et al. 2005).

Similar results were observed in a study by Martin et al. (2005) were the formation of PFOA was 10 times higher than that of PFNA when measured plasma from rats after 8:2 FtOH injection (Martin et al. 2005).

Nabb et al. (2007) investigated the in vitro metabolism of ^{14}C labelled 8:2 FTOH in rat, mouse, trout and human hepatocytes, and in rat, mouse and human liver microsomes and cytosol fractions. The 8:2 FTOH clearance rates were highest in rat, followed by mouse, humans and lowest in trout. The yield of PFOA was low. However, the author found that the 8:2 FTOH volatilized from the aqueous fraction and into the headspace of the experimental set up and was not available for biotransformation (Nabb et al. 2007).

In a study by Himmelstein et al. (2012) biotransformation of 8:2 FTOH in rats exposed via inhalation was investigated. The most abundant metabolites were 7:3 FTCA>PFOA>8:2 FTCA (Himmelstein et al. 2012).

Timed-pregnant CD-1 mice received a single dose of 8:2 FTOH (30 mg/kg bw) or vehicle by gavage on gestation day 8 (GD8). During gestation (GD9 to GD18), maternal serum and liver concentration of PFOA decreased from 789 ± 41 to 668 ± 23 ng/ml and from 673 ± 23 to 587 ± 55 ng/g, respectively. PFOA was transferred to the developing foetuses as early as 24 h post-treatment with increasing concentration from 45 ± 9 ng/g (GD10) to 140 ± 32 ng/g (GD18). The group of pups only exposed via lactation had a PFOA concentration of 57 ± 11 ng/ml at PND3 and 58 ± 3 ng/ml at PND15. 8:2 FTOH-intermediates were not assessed in this study (Henderson and Smith, 2007).

In a study by D'Eon and Mabury (2007) rats exposed to two doses of 8:2 FTOH (200 mg/kg bw) had increased concentrations of PFOA in blood with a peak of 34 ± 4 ng/g (D'Eon and Mabury 2007). Nilsson et al. (2013) measured the different metabolites FTCAs and FTUCAs of 8:2 FTOH in serum from professional skiwaxers during the skiing season in addition to summer season without skiwaxing. Several different polyfluorinated metabolites were detected in the serum, with PFOA (median of 11 skiwaxers: 110 ng/mL) being the most abundant. Due to the findings of FTCs and FTUCAs in skiwaxers blood after exposure to high levels of 8:2 FTOH via air suggest metabolism of FTOH to PFOA (Nilsson et al. 2013). The downside with this study is the lack of a control group showing possible background levels of FTOH-metabolites.

In conclusion, 8:2 FTOH mainly degrades to PFOA in sludge, soil, water and air. In vertebrates, PFOA is the main perfluoric acid formed by biotransformation of 8:2 FTOH. Emission and exposure of 8:2 FTOH will add to the overall blood concentration of PFOA in human blood stream.

8:2 Fluorotelomer derivatives

Fluorotelomer stearate monoester/fluorotelomer citrate triester

The biodegradation of 8:2 fluorotelomer stearate monoester was studied by Dasu et al., in agricultural loam soil using laboratory microcosms within 80d. Although the microcosms were closed, the oxygen concentrations were comparable to aerobic conditions. The 8:2 fluorotelomer stearate monoester was degraded with a half-life (primary degradation) of 10.3 days (first-order kinetic model fit well up to day 20). At the end of the experiment 22% of the initial8:2 fluorotelomer stearate monoester was detected. The ester bond was hydrolysed and 8:2 FTOH was rapidly formed with a half-life of 2 days. Subsequent degradation was monitored. Similar reaction products as shown Figure B.4-1 were found. PFOA, which was the major terminal product, consistently increased over time reaching 1.7 mol% by day 80 (Dasu et al., 2012). PFOA concentration has not reached plateau until day 80. Approximately 14 mol% of intermediate transformation products (sum of 8:2 FTCA and 8:2 FTUCA) were detected at day 80. Therefore, further increase of PFOA concentration with time is possible. Total mass balance decreased over time to about 38 mol% by day 80. Reasons could be irreversible sorption and decreasing extraction efficiencies of degradation products over time and formation of unidentified products.

A similar study was performed with forest soil (Dasu et al., 2013). 8:2 fluorotelomer stearate was degraded with a half-life (primary degradation) of 28.4 days (first-order kinetic model fit well up to day 46), which was slower than in the previous experiment based on agricultural soil. The major terminal metabolite was PFOA (4 mol% at 94 days). PFOA concentration has not reached plateau until day 94. Approximately 25 mol% of inital fluorotelomer stearate monoester remained at day 94. 16 mol% of intermediate transformation products (sum of 8:2 FTCA, 8:2 FTUCA, and 7:2 sFTOH) were detected at day 94. Total mass balance decreased over time to about 44 mol% by day 94.

Dasu and co-workers also studied the biodegradation of 8:2 fluorotelomer citrate in a similar experimental setup (Dasu et al., 2013). The citrate was degraded slower. Approximately 56 mol% of the initial fluorotelomer citrate remained by the end of the study (218 days). Formation of 8:2 FTOH and secondary metabolites were identical to those shown in Figure B.4- 1. 4 mol% PFOA was detected at day 218 (sum of 8:2 FTOH, 8:2 FTUCA, 8:2 FTCA, 7:2sFTOH ~6 mol%).

Polyfluoroalkyl phosphoric acid monoesters and diesters (mono-PAP, di-PAP)

Degradation of polyfluoroalkyl phosphates (6:2 diPAPs) was studied by Lee and co-workers (2010) using raw wastewater and sewage sludge. It was shown that the ester bonds were cleaved by the formation of monoPAPs (microbial hydrolysis) followed by a production of 6:2 FTOH. The authors also performed a chain length study with n:2 monoPAP (n=2,4,6,8). The production of FTOHs in the headspace and the production of FTCAs, FTUCAs and PFCAs in the aqueous phase of the bottles suggest that the monoPAPs were microbially transformed. Although the monoPAP congeners were observed to produce the corresponding FTOHs in relatively similar order (1-2% after 92 days; conservative estimates), the rate of production was observed to decrease significantly as the chain length of the monoPAP increased. Nevertheless, it can be assumed that the same transformation mechanism of 6:2 PAPs occurs to longer chain PAPs, such as 8:2 diPAPs (Lee et al., 2010). Hydrolysis of diPAP to fluorotelomeralcohol was also demostrated by D'eon and Mabury (2007) who have shown in a study with rats that metabolism of 8:2 mono and diPAP in mammals leads to the formation of 8:2 FTOH, which is then available for oxidation to PFOA. The authors suggest that exposure in rats to either 8:2 monoPAPs or 8:2 diPAPs will result in increased PFOA blood levels (D'Eon and Mabury 2007). A later study by the same authors confirms these results and suggest that biotransformation of diPAP even with low exposure could over time result in significant exposure to PFOA (D'Eon and Mabury 2011).

8:2 mono- and diPAPs are reported to undergo slow hydrolysis at environmental conditions (estimated lifetimes >26 years) resulting in 8:2 FTOH and phosphoric acid (D'eon and Mabury, 2007). It is explicitly noted that the experimental hydrolysis rates cannot be reproduced by existing models (Rayne and Forest, 2010). Mono- and diPAPs of 8:2 FTOHs, including their polymers, can therefore be considered as a class of substances leading to release of PFOA by abiotic degradation processes.

Fluorotelomer ethoxylates

Biotransformation of fluorotelomer ethoxylates was reported by Fromel & Knepper (Fromel and Knepper, 2010). WWTP effluent was used under aerobic conditions. Zonyl FSH, a commercial mixture which contains fluorotelomer ethoxylates (8:2 FTOH residues = 0.29%) with perfluorinated chain lengths between four and

12 and a degree of ethoxylation between 0 and 18 was analysed. Fluorotelomer ethoxylates were rapidly degraded (half-life (primary degradation = 1d). One significant metabolite was formed within the study duration of up to 48 days: Fluorotelomer ethoxylate carboxylate. PFOA resulted in a concentration of only 0.3 %. It can be assumed that studies with a longer time frame will result in higher PFOA concentrations.

Fluorotelomer acrylates and methacrylates

In general, carboxylic acid esters will undergo hydrolysis resulting in the corresponding alcohols and carboxylic acids. It is reported that hydrolysis of perfluorinated telomer acrylates (and methacrylates) may be fast in landfills (half-lives < 4 days; 40-50 °C and pH 4-9), but that they have half-lives in the range of years in marine systems (half-lifes = 3-5 years; 15°C and pH 8.1) (using SPARC software program). Hydrolysis of monomeric perfluorinated telomer acrylates may be a significant source to current environmental loadings of FTOHs and the corresponding PFCA. Under some saturated landfill conditions abiotic hydrolytic degradation of fluorotelomer acrylates could be occur resulting in significant fluxes of FTOHs and their degradation products into ground water and surface water (Rayne and Forest, 2010; Nielsen, 2014).

Microbial transformation (microbially mediated hydrolysis) of 8:2 fluorotelomer acrylate (8:2 FTAC) and 8:2 fluorotelomer methylacrylate (8:2 FTMAC) in aerobic soils was investigated by Royer et al. (Royer et al., 2014). 8:2 FTAC and 8:2 FTMAC were rapidly degraded with halflives of 3-5 days and 15 days, respectively. Both substances were hydrolyzed at the ester linkage as evidenced by the formation of 8:2 FTOH. 8:2 FTOH was further degraded via the known biotransformation pathway (see Figure B.4-1). 8 mol% PFOA was formed in FTAC-amended soil, and 10.3 mol% PFOA was formed in FTMAC-amended soil after 105 days, respectively. Besides the stable metabolites like PFOA, PFHpA, and PFHxA (< 3mol%), 38-45 mol% of intermediate metabolites (8:2 FTUCA, 8:2 FTCA, 7:2 sFTOH) were observed at day 105. Total mass balance decreased with incubation time with 50-75 % recovery at the end of 105 day incubation. Reasons for loss of mass balance could be: reduced extractability, increased irreversibly bound metabolites over time, or additional metabolites that were not quantified or identified.

Acrylates and methacrylates of 8:2 FTOHs, including their polymers, can therefore be considered as a class of substances leading to release of PFOA.

Polyfluorinated silanes

No relevant information concerning hydrolytic lifetimes of condensed or polymerized polyfluorinated silanes was found in the open literature. Silanes have appreciable vapour pressures and may in principle evaporate and undergo photooxidation in the atmosphere. It is also conceivable that small siloxanes may partition to the atmosphere and undergo photo-oxidation there. As reaction product PFOA will be formed (for more details see Appendix B.4.1) (Nielsen, 2014).

Polyfluorinated olefins

The atmospheric lifetimes of polyfluorinated olefins are around 8 days with 90% removal via reaction with OH radicals and 10% removal via reaction with O_3 (smog chamber experiment) (Sulbaek Andersen et al., 2005). The major product (~ 90 %) in the atmospheric photooxidation is the corresponding perfluoroalkyl aldehyde (PFAL). The atmospheric lifetimes of PFALs are estimated to be around 90 days with respect to reaction with OH. It is therefore likely that PFALs in part will partition to the atmospheric aqueous phase and undergo photooxidation there to form the corresponding PFCA (see Appendix B.4.1 for reaction equations) (Nielsen, 2014). 8:2 Fluorotelomer olefins (FTO, $F(CF_2)_3CH=CH_2$), a sub-class of polyfluorinated olefins, can therefore be considered as a class of substances leading to release of PFOA.

Polyfluorinated iodides

The hydrolysis of fluorotelomer iodides was modelled with HYDROWIN module of EPI Suite software program (Rayne and Forest, 2010; Nielsen, 2014). At 20°C the hydrolytic half-life is expected to remain constant at 126 days between pH 0 and 9 and then decrease to < 7 hours at pH 14. In marine system (pH = 8.1) the hydrolytic half-life decreased from about 8 years at 0°C to about 130 days at 20°C. The hydrolysis of fluorotelomer iodides may be contributing to substantial FTOH and PFCA inputs in aquatic systems.

The atmospheric fate of 4:2 fluorotelomer iodides was investigated in a smog chamber experiment by Young et al. (Young et al., 2008; Young and Mabury, 2010). Atmospheric lifetime of fluorotelomer iodides is expected to range from about 1 to 7 days (limited by photolysis), depending on time of year and latitude. Photolysis of fluorotelomer iodides occurs via elimination of the iodine atom leading to the formation of the fluorotelomer aldehyde. The fluorotelomer aldehyde will be further degraded (atmospheric lifetime ~4 days) to perfluoroaldehyde. Perfluoroaldehyde has a atmospheric lifetime of approximately 1 day with respect to photolysis and approximately 20 days with respect to reaction with OH-radicals. The oxidation of perfluoroaldehyde lead to the formation of PFCA. Because of their long-range potential fluorotelomer iodides contribute to the occurrence of PFCAs (e.g. PFOA) in remote areas.

Gas phase photolysis and hydrolysis of 8:2 fluorotelomer iodide will lead to the release of 8:2 FTOH and thus PFOA (see Figure B.4-1) (Rayne and Forest, 2010; Young et al., 2008; Young and Mabury, 2010).

Polyfluorinated amides

Jackson and Mabury investigated the hydrolysis of the polyfluorinated amides N-ethyl-N-(2-hydroxyethyl)perfluorooctaneamide (EtFOA) in 1 M NaOH solution (pH 14), in 5 mM Tris buffer (pH 8.5), and in 50 mM borate buffer (pH 8.5) (Jackson and Mabury, 2013). They found quantitative (98%) hydrolysis to PFOA in 1 M NaOH solution (pH 14) after 24 hours at room temperature. No hydrolysis to PFOA was observed after 8 days at pH 8.5. Rapid degradation was observed in the borate buffer, but not to PFOA unless at pH 14 (after 24 hours at room temperature):

$$C_7F_{15}C(O)NHC_2H_5 + OH^- \rightarrow C_7F_{15}C(O)O^- + C_2H_5NH_2$$

The experiments suggest that polyfluorinated amides have long hydrolytic lifetimes at environmental conditions. They do, however, hydrolyse.

Jackson et al. studied the atmospheric photo-oxidation (smog chamber experiment) of N-ethyl- perfluoro-butyramide (EtFBA, $C_3F_7C(O)NHCH_2CH_3$) as a surrogate for longer chained polyfluorinated amides and identified $C_3F_7C(O)NH_2$ as intermediate, and PFCAs and HNCO (isocyanic acid) as products (Jackson et al., 2013). They presented a general mechanism based on the observed product distribution. Atmospheric lifetime of EtFBA, with respect to reaction with OH, was estimated to be 4.4 days. Primary oxidation products reacted further to perfluorobutanoic acid (PFBA; maximum mass yield 16%). The authors predict similar reaction kinetic for N-ethyl-perfluorooctanamide (EtFOA) and EtFBA since the length of a perfluorinated chain does not affect the reaction rate with OH. The primary oxidation products of EtFOA are expected to have much longer lifetimes and could be capable of contaminating Arctic air. The primary oxidation products are expected to react further to form PFOA.

Martin et al. studied the atmospheric photo-oxidation (smog chamber experiment) of N-ethyl perfluorobutanesulfonamide (NEtFBSA, $C_4F_9S(O)_2NHCH_2CH_3$) and identified $C_4F_9S(O)_2NHC(O)CH_3$, $C_4F_9S(O)_2NHCH_2CHO$and $C_4F_9S(O)_2NHCHO$ as intermediates, and SO_2, COF_2 and PFCAs as stable products (Martin et al., 2006). Three PFCAs were detected above the level of the blank: 0.33% perfluorobutanoic acid (PFBA), 0.11% perfluoropropanoic acid (PFPrA), and 0.09 trifluoroacetic acid (TFA) of the molar balance, respectively. At the same time only 0.65% COF_2 of the starting material had unzipped. Extrapolation of these results suggests that 45% of the carbon in the perfluoroalkane chain will ultimately be incorporated into PFCAs upon complete oxidation, while the remaining fraction is expected to go to COF_2 (timeframe not given). The authors suggest that it is evident that analogous perfluorooctane sulfonamide is a potential source for PFOA. They presented a general mechanism based on the observed product distribution.

In conclusion, polyfluorinated amides will undergo slow hydrolysis resulting in the corresponding PFCA. Studies on atmospheric photo-oxidation of short-chain polyfluorinated amides show a release of the corresponding PFCA. Thus, abiotic degradation of polyfluorinated amides will result in release of PFOA.

Other potential PFOA precursors and UVCB

Nielsen (2014) stated that gas phase photolysis and aqueous phase hydrolysis of perfluorooctyl iodide will lead to the release of PFOA (see Appendix B.4.1 for reaction equations).

Other potential PFOA precursors and UVCBs cannot in general be classified as classes of substances leading to release of PFOA. However, substances containing $F(CF_2)8(CH_2)_2$-groups will most probably result in release of 8:2 FTOHs in the environment. Thus, using the weigth of evidence approach they can be considered as a class of substances leading to release of PFOA.

N-methyl perfluorooctane sulfonamidoethanol (N-MeFOSE)/ N-ethyl perfluorooctane sulfonamioethanol (N-EtFOSE)

There is some evidence that N-MeFOSE and N-EtFOSE are potential sources of PFOA to the global environment (D'eon et al., 2006; Lange, 2000, 2001; Martin et al., 2006). These substances are also PFOS-precursors, thus they are already regulated under EU POPs regulation (Commission Regulation (EU) No. 757/2010). N-MeFOSE and N-EtFOSE will therefore not be assessed further in this proposal.

Polymers

The biodegradation potential of a fluoroacrylate polymer product was studied in four aerobic soils over two years (Russell et al., 2008). It was assessed whether the FTOH side chain covalently bonded to the polymer backbone may be transformed to PFOA. The fluoroacrylate polymers contain the polymer itself and also residual raw materials and impurities ("residuals"). Major residuals present in the test substance were FTOH, fluoroacrylate monomer, FTOH acetate, and fluorotelomer olefin. Depending on soil the estimated

half-lives of the polymer ranged from 95 to >2000 years (all soils combined 1160 years). The estimated half-lives of residuals were 12 to 43 days (all soils combined 27 days). The maximum PFOA concentration ranged from 1.8 to 2.1 µmol PFOA/kg soil. The residual amount of PFOA in the test substance was 0.019 µmol PFOA/kg soil. Hence, PFOA is formed from degradation of residuals and possibly also from degradation of the side chains in the polymer. The maximum experimental PFOA concentrations are 24-28% of the theoretical amount that could be derived from 100% conversion of the residuals alone (7.55 µmol PFOA/kg soil). If all 8:2 related analytes are summed 25-32% of the theoretical amount of PFOA formed from residuals. After application of the degradation rate to the estimated total historic fluoroacrylate polymer production, use and disposal, the biodegradation of fluoroacrylate polymer and residuals is calculated to contribute less than 5 tonnes per year (based on 2007) to the global environmental concentration of PFOA.

Washington et al. also investigated the degradability of an acrylate-linked fluorotelomer polymer in soil (Washington et al., 2009). The polymer can be degraded in soil through attack on the carbon backbone and/or the ester linkage connecting the backbone to the fluoroalkyl side chains resulting in PFOA via the intermediate 8:2 FTOH. Estimated half-life of the tested coarse-grained polymer ranged from 870 to 1400 years. Modelling indicates much shorter half-lives (10-17 years) for more finely grained polymers assuming degradation is surface mediated. The authors observed degradation of PFOA with an estimated half-life of 130 days. However, this result is contradictory to other studies which stated that PFOA is not degradable in soil (Moody et al., 2003; OECD, 2006).

After extensive method development the authors investigated the degradation of two commercial acrylate-linked fluorotelomer-based polymers (containing ~ 50 % C_8 telomers and ~ 30 % C_{10} telomers) in four soils in a further study (Washington et al. 2015). The estimated half-lives ranged from 33 to 112 years. Compared with day 0, PFOA concentrations increased up to ~1264% at day 376. 8:2 FTOH concentrations even increased up to 2894%. The authors estimated a half-life of 8:2 FTOH of ~ 1200 days. Due to discrepancy to literature values (half-lives < 28 days) a follow-up 8:2 FTOH degradation experiment was performed. After spiking microcosms with 8:2 FTOH a half-life of 210 days was estimated. Because the only design difference between the both experiments was the presence of the fluorotelomer-based polymer, the authors inferred the difference in half-lives to be due to presence of the fluorotelomer-based polymer. Furthermore, the authors performed a hydrolysis experiment with the fluorotelomer-based polymer. The results showed an increase of 8:2 FTOH in the pH 10 treatments, almost doubling over the 11-day experiment, while in the pH 3 treatments and dry controls the concentration remained constant. These results suggest that fluorotelomer-based polymer can undergo OH--mediated hydrolysis.

In a further study Russell et al. evaluated the formation of PFOA from the biodegradation of a fluorotelomer-based urethane polymer product in four aerobic soils (Russell et al., 2010). The degradation of the polymer begins with the enzymatic cleavage of the fluorotelomer side-chain from the polymer backbone followed by the fractional conversion of fluorotelomer side-chains containing eight fluorinated carbons through a series of intermediates reactions forming PFOA. The maximum concentrations of PFOA (modelled; first-order reaction) formed after two years ranged between 0.5 and 1.3 µmol/kg soil (initial concentration of polymer = 77.6 µmol/kg soil; initial concentration of intermediates and PFOA = 0.032 µmol/kg soil. Including all data until day 728 in kinetic evaluation the calculated half-lives of the polymer ranged between 79 and 241 years (geomean = 132 years). Including all data until days 728 except one soil until 273 in kinetic evaluation the estimated half-lives ranged from 28 to 241 years (geo. mean 102 years). In contrast to Russell et al. 2008 the PFOA formation from residuals was negligible in this study. After application of the degradation rate to the estimated total historic production, use and disposal of fluorotelomer-based urethane polymer, the annual potential global formation of PFOA was estimated to be 0.3 - 2.5 t/a (based on 2007).

Rankin et al. investigated the biodegradability of a fluorotelomer-based acrylate polymer in soil-plant microcosm over 5.5 months in the absence/presence of wastewater treatment plant biosolid by indirect and direct analysis (Rankin et al. 2014). A unique fluorotelomer-based acrylate polymer was synthesized by aqueous dispersion following two commercial patents. The polymer was determined to be solely a homopolymer of 8:2 FTAC containing hydrogen and hexadecylthiol end groups and have primarily between 2 and 16 fluorotelomer appendages. The estimated half-lives ranged from 8 to 111 years. Incubation of the fluorotelomer-based acrylate polymer results in the accumulation of PFHxA, PFHpA, and PFOA concurrently with the reduction of 8:2 FTCA and 8:2 FTUCA. PFOA was the dominant product, constituting 57, 70, and 80% in all microcosm compartments in fluorotelomer-based acrylate polymer/soil, fluorotelomer-based acrylate polymer/plant, and fluorotelomer-based acrylate polymer/plant/biosolids, respectively. Furthermore, the biodegradation of the fluorotelomer-based acrylate polymer was observed via structural changes by direct analysis (matrix-assisted laser desorption/ionization (MALDI-TOF) time-of-flight mass spectrometry).

Hydrolytic half-lives of 8:2 fluorotelomer acrylate polymer segments was estimated using SPARC software program (Rayne and Forest, 2010). The estimated half-lives were 170-270 years in marine systems (15°C and pH 8.1) and < 1year under landfill conditions (40-50 °C and pH 4-9). Under some saturated landfill

conditions abiotic hydrolytic degradation of fluorotelomer acrylates could be occur resulting in significant fluxes of FTOHs and their degradation products (e.g. PFOA) into ground water and surface water.

Conclusion on degradation of PFOA-related substances

Studies of the 8:2 FTOH in biotic degradation studies demonstrate the formation of PFOA and to a lower extent shorter chain PFCAs. The formation of PFOA in most of these studies is rather small (<6% in 90 days) (Dinglasan et al., 2004; Wang et al. 2005a; Wang et al. 2005b). However, up to 50% of intermediate metabolites were detected at the end of the studies. These substances will further degrade to PFOA with time. Studies lasting for several months show a higher formation of PFOA. In a seven months study in aerobic soil, 8:2 FTOH degradation resulted in 10 to 40% PFOA, < 1% PFHpA and 1-4 % PFHxA (Wang et al., 2009). PFOA is created after a cascade of steps. It appears likely that one or two of these degradation steps are rather slow. This indicates that biotic degradation of 8:2 FTOH is an important source of PFOA in the environment.

In an experimental study (Ellis et al., 2004) the atmospheric degradation of 8:2 FTOH to PFOA was observed. Even if only a small amount of PFOA was released, atmospheric degradation of 8:2 FTOH is a significant global source of PFOA, especially in remote areas. The aqueous phase photo-oxidation was also investigated (Gauthier and Mabury, 2005). PFOA formation from 8:2 FTOH will result in 75 to 100%. Therefore, aqueous phase photo-oxidation may turn out to be very important in spite of the low solubility. It could be assumed that 8:2 FTOH is completely degraded to PFOA and shorter chain PFCAs. The biotic and abiotic (hydrolysis and atmospheric) degradation of 8:2 fluorotelomer derivates (e.g. Fluorotelomer stearate monoester, fluorotelomer (meth)acrylates, polyfluoroalkyl phosphoric acid monoester and diester, polyfluorinated olefins, polyfluorinated iodides, etc.) was confirmed (Dasu et al., 2012; Dasu et al., 2013; Lee et al., 2010; Royer et al., 2014, Rayne and Forest, 2010; Young and Mabury, 2010; Jackson et al., 2013; Nielsen, 2014). The 8:2 fluorotelomer derivates degrade, mainly via 8:2 FTOH, to PFOA.

Side-chain fluorinated polymers degrade very slowly in soil. Estimated half-lives ranged from 8 to 1400 years (Russell et al., 2010; Washington et al., 2009; Washington et al. 2015; Ranskin et al. 2004). Modelling data indicates much shorter half-lives (10-17 years) for finely grained polymers (Washington et al., 2009). Nevertheless, PFOA was observed as a degradation product. Therefore, side-chain fluorinated polymers are sources of PFOA in the environment.

In conclusion, all the presented PFOA-related substances are degraded to PFOA and shorter chain PFCAs by abiotic and biotic processes in the environment. For those substances where no degradation studies are available it can be assumed that based on the chemical similarity the substances will most probably be degraded in a similar way. Thus, based on the weight of evidence approach PFOA will most probably be released in the environment. Hence, these substances need to be considered as important sources of PFOA in the environment. Furthermore, they need, according to REACH, be considered as PBT-substances as well.

Table 6: Summary of degradation PFOA-related substances, extracted from Appendix B.4.1 of ECHA (2015a)

PFOA-related substance	Degradation study	Study type/compartement	Results
8:2 FTOH	Dinglasan et al., 2004	Mixed microbial system (the enrichment culture was obtained from sediment and groundwater taken from a contaminated site)	• Half-life 8:2 FTOH ~0.2 days/mg of initial biomass protein • By day 81, PFOA was detected at approximately 3% of the total mass of added 8:2 FTOH. • This production of PFOA may be attributed to the degradation of the earlier produced 8:2 FTUCA (8:2 fluorotelomer unsaturated carboxylic acid), and the authors suggest that further degradation of the 8:2 FTUCA (major metabolite at day 81, ~40%) in the system may lead to an increase in the production of PFOA. • Mass balance: By day 81, 45% loss => reasons: volatile metabolites that were left unidentified, volatile metabolites may have been lost during routine sampling ((loss of initial 8:2 FTOH ~20% in sterile control), unaccounted mass from the unsaturated metabolites being covalently bound by biological macromolecules
	Butt et al. , 2014	Review article	• Studies with the 8:2 FTOH metabolism universally show the formation of perfluorooctanoate (PFOA) and, to a smaller fraction, perfluorononanoate (PFNA) and lower-chain-length PFCAs. In general, the overall yield of PFOA is low, presumably because of the multiple branches in the biotransformation pathways, including conjugation reactions in animal systems.
	Nilsson H. et al., 2013	Human	• Has measured metabolites from 8:2 FTOH in skiwaxers. Did not find the precursor itself. Detected the following metabolites: PFOA:Range:LOD-628 µg/L Median: 110 PFNA: Range:LOD-163 µg/L Median: 12 PFHpA:Range:LOD-19.8 µg/L, Median:2.4 7:3 FTCA: Range: LOD-3.5 µg/L, Median: 0.92 µg/L 8:2 FTUCA: Range: LOD-0.64 µg/L, Median: 0.07 µg/L
	D'Eon et al. 2007	Rats, in vivo	• Rats exposed to 8:2 FTOH had increased concentrations of PFOA in blood
	Henderson and	Mice, in vivo	• Timed-pregnant CD-1 mice received a single dose of 8:2 FTOH

		detected in any of the soils)	(average PFOA formation was approximately 25 %) after 197 days • 10-35% of total ^{14}C was irreversibly bound to soils (PFOA was not irreversibly bound to the soils) • Level of PFOA reached steady state after 14, 56, and 140 days respectively (depending on soil type).
	Zhang et al., 2013	Anaerobic digester sludge (methoanogenic conditions; domestic WWTP)	• Half-life 8:2 FTOH = 145 days (primary biodegradation) • PFOA accounted for 0.3 mol% of added 100 mol% [3-^{14}C]8:2 FTOH by day 181 • Approximately 39 mol % of 8:2 FTOH still remained by day 181 • 8:2 FTCA, 8:2 FTUCA ≈ 23 mol% at day 181
	Ellis et al., 2003	Atmosphere (reaction of Cl atoms and OH radicals with 2:2; 3:2, 4:2 FTOH	• The length of the perfluorinated carbon chain residue had no discernible impact on the reactivity of the molecules • Atmospheric life-time of the FTOHs (n:2 FTOH, n ≥ 2) by reaction with OH radicals is approximately 20 days
	Ellis et al., 2004	Atmosphere (smog chamber experiment)	• 8:2 FTOH is oxidised (initiated by Cl atoms which represent OH radicals) and forms PFOA (1.5% C mass balance of 8:2 FTOH) • The overall formation of PFOA is expected to be greater because many intermediates were still observed in these samples (e.g. (8:2 FTAL (8:2 fluorotelomer aldehyde) = 6%; 8:2 FTCA = 26%), a portion of which would then form additional PFOA upon further oxidation.
	Gauthier and Mabury, 2005	Aqueous phase photo – oxidation 1.) Hydrogen peroxide solution; 2.) Synthetic field water; 3.) water from Lake Ontario, Canada	1.) • after 10 hours: ~ 40% PFOA + ~60% 8:2 FTCA which undergo further aqueous photo-oxidation leading to PFOA as major product => 75-100% transformation to PFOA expected with time • Half-life 8:2 FTOH = 0.83± 0.20hours (10 mM H_2O_2) and 38.0±6.0 hous (100µM H_2O_2) 2.) • after 140-146 hours: 1-8% PFOA • Half-life 8:2 FTOH = 30.5 ± 8.0 to 163.1 ± 3.0 hours 3.) • 18% PFOA were formed (duration not specified) • Half-life 8:2 FTOH = 93.2 ± 10.0 hours
Fluorotelomer stearate monoester (FTS)	Dasu et al., 2012	Agricultural soil	• 1.7 mol% PFOA by day 80 (major terminal product) • PFOA concentration has not reached plateau until day 80 (8:2 FTCA, 8:2 FTUCA ~ 14% mol at day 80) • Approximately 22 mol % of FTS remained on day 80

	Smith, 2007		(30 mg/kg bw) or vehicle by gavage on gestation day 8 (GD8). During gestation (GD9 to GD18), maternal serum and liver concentration of PFOA decreased from 789 ± 41 to 668 ± 23 ng/ml and from 673 ± 23 to 587 ± 55 ng/g, respectively. PFOA was transferred to the developing foetuses as early as 24 h post-treatment with increasing concentration from 45 ± 9 ng/g (GD10) to 140 ± 32 ng/g (GD18). The group of pups only exposed via lactation had a PFOA concentration of 57 ± 11 ng/ml at PND3 and 58 ± 3 ng/ml at PND15.
	Nabb et al., 2007	Hepatocytes from rats, mice and humans, in vitro studies	• The in vitro data suggest that hepatocytes from rats, mice and humans have the ability to biotransform 8:2 FTOH into several metabolites including PFOA. The yield of PFOA was low. However, the author found that the 8:2 FTOH volatilized from the aqueous fraction and into the headspace of the experimental head space and was not available for biotransformation
	Kudo et al. 2005	Mice	• The PFOA levels in the animals continued to rise throughout the experiment where the mice where exposed to 8:2 FTOH. The formation of PFOA was 10 times higher than that of PFNA
	Martin et al. 2005	Rat hepatocytes	• The formation of PFOA was 10 times higher than that of PFNA.
	Himmelstein et al. 2012	Rat	• The biotransformation of 8:2 FTOH in rats exposed via inhalation was investigated. The most abundant metabolites were 7:3 FTCA>PFOA>8:2 FTCA.
	Wang et al., 2005a	Mixed bacterial culture (culture was obtained from sludge from an industrial WWTP)	• Concentration of PFOA increase over 56 days and levelled off to 6% of the ^{14}C mass balance (90 days) • Approximately 36% of ^{14}C-8-2 FTOH remained in the mixed bacterial culture at day 90, partly due to its strong adsorption to the PTFE septa. • Sum of FTUCA, FTCA (8:2 fluorotelomer carboxylic acid), 7:2sFTOH (7:2 fluorotelomer secondary alcohol) ≈ 25% at day 90
	Wang et al., 2005b	(200-fold diluted) Activated sludge from a domestic WWTP	• 2.1 ± 0.4 % PFOA of total initial mass (^{14}C labelled) at day 28 • The parent still contributed about 57% of the mass balance at day 28, about 41% of which resulted from adsorption to the septa. It appears that the strong adsorption of the parent to the PTFE septa during the test reduced its bioavailability for microbial biodegradation. • Sum of FTUCA, FTCA≈ 33% at day 28
	Wang et al., 2009	Three different aerobic soils (8:2 FTOH was not	• Half-life 8:2 FTOH less than 7 days (primary biodegradation) • 10-40% of [3-^{14}C] 8:2 FTOH was biodegraded to form PFOA

			• Total mass balance decreased over time to about 38 mol% by day 80 (irreversible sorption and decreasing extraction efficiencies of degradation products over time and formation of unidentified products) • Half-life FTS = 10.3 days (primary degradation); Half-life 8:2 FTOH ~ 2 days
	Dasu et al., 2013	Forest soil	• ~4 mol% PFOA by day 94 • PFOA concentration has not reached plateau until day 94 (8:2 FTUCA and 8:2 FTCA and 7:2 sFTOH ~16 mol % by day 94) • Approximately 25 mol % of FTS remained on day 94 • Total mass balance decreased over time to about 44 mol% by day 94. • Half-life FTS = 5-28 days (primary degradation); Half-life 8:2 FTOH ~ 2 days
Fluorotelomer citrate trimester (TBC)	Dasu et al., 2013	Agricultural soil	• 4% mol PFOA by day 218 • Approximately 56% of TBC remained on day 218
Mono-PAP, di-PAP	D'eon and Mabury, 2007	Hydrolysis	• <0.1% degradation over a 2-week period for 8:2 diPAP and monoPAP; minimum lifetime of 26 years with respect to hydrolysis (pH 9; 50°C)
	D'eon and Mabury, 2007	Rats	• oral exposure of rats to either 8:2 monoPAPS or 8:2 diPAPS will result in increased PFOA blood levels
	D'eon and Mabury, 2011	Rats	• observed biotransformation to the PFCAs for both monoPAP and diPAP • diPAPs were bioavailable, with bioavailability decreasing as the chain length increased from 4 to 10 perfluorinated carbons • Using experimentally derived biotransformation yields, perfluorooctanoic acid (PFOA) sera concentrations were predicted from the biotransformation of 8:2 diPAP at concentrations observed in human serum. Because of the long human serum half-life of PFOA, biotransformation of diPAP even with low-level exposure could over time result in significant exposure to PFOA.
	Lee et al., 2010	Raw wastewater and sewage sludge	6:2 diPAP, 6:2 monoPAP: • The main degradation pathway of PAPs in WWTPs is likely to be microbial hydrolysis of the phosphate ester bonds to produce FTOHs • Since FTOH production was not observed in any of the control bottles, degradation observed in the experiments can be attributed to microbial transformation.
			Chain length study (n:2 monoPAP, n=4,6,8,10): • Production of FTOHs was observed in the headspace of the monoPAP-dosed bottles during microbial incubation. This hydrolysis was microbially mediated as the evolution of FTOHs was not observed in the sterile controls. The production of FTCAs, FTUCAs, and PFCAs in the aqueous phase of the experimental bottles suggests that some of the monoPAPs were microbially transformed via a concerted mechanism that involved further oxidation of the FTOH intermediate within the microbial cells. • Although the four monoPAP congeners were observed to produce the corresponding FTOHs in relatively similar order (1-2% after 92 days; (conservative estimates), the rate of production was observed to decrease significantly as the chain length of the monoPAP increased.
Fluorotelomer ethoxylates	Frömel and Knepper, 2010	Effluent of a commercial WWTP	• Commercial mixture of FTEO with a perfluoroalkyl chain length between 4 and 12 carbon atoms and a degree of ethoxylation between 0 and 18 (8:2 FTOH residues = 0.29%) • half-life (primary degradation)= 1 day (significant metabolite = FTEO carboxylates) • PFOA formation 0.3% in 48 days (degradation of residual FTOH) • Only a short-term study; Long-term studies might prove slow biotransformation of short-chained FTEOC finally ending up in the respective FTOH and thus in the respective PFCA
Fluorotelomer acrylates (8:2 FTACs) and methacrylates (8:2 FTMACs)	Rayne and Forest, 2010; Nielsen, 2014	Hydrolysis (SPARC software program)	• Degradation of Flurotelomer acrylates could be rapid: Landfills (40-50 °C, pH 4-9)) half-lives < 4 days marine systems (15°C, pH 8.1) half-lives = 3-5 years • Under dome saturated landfill conditions degradation could be resulting in significant fluxes of FTOHs and their degradation product (PFCAs) into ground and surface water
	Royer et al. 2014	Soils	• Half-lives: 3-5 days (FTCAs) and 15 days (8:2 FTMCAs) • 8 mol% PFOA was formed in FTAC-amended soil (105 days) • 10.3 mol% PFOA was formed in FTMAC-amended soil (105 days) • Beside stable metabolites like PFOA, PFHpA, and PFHxA (<3 mol%), 38-45 mol% of intermediate metabolites (8:2 FTUCA, 8:2 FTCA, 7:2 sFTOH) were observed at day 105. • Total mass balance decreased with incubation time with 50-75 % recovery (reduced extractability, increased irreversibly bound metabolites over time, or additional metabolites that were not quantified or identified.)

Polyfluorinated silanes	Nielsen, 2014	Atmosphere (theoretical consideration)	• May in principle evaporate and undergo photooxidation • PFOA will be formed as reaction product
Polyfluorinated olefins (8:2 Fluorotelomer olefin)	Sulbaek Andersen et al., 2005 Nielsen, 2014	Atmosphere (smog chamber experiment)	• Atmospheric lifetime is approximately 8 days with 90% of removal via reaction with OH and 10% via reaction with O3 • The major product (around 90 %) in the atmospheric photo-oxidation is the corresponding PFAL (perfluoroalkyl aldehyde). The atmospheric lifetimes of PFALs are estimated to be around 90 days with respect to reaction with OH. It is therefore likely that PFALs in part will partition to the atmospheric aqueous phase and undergo photo-oxidation there (product corresponding PFCA)
Polylfuorinated iodides (Fluorotelomer iodides, FTI)	Rayne and Forest, 2010; Nielsen, 2014	Hydrolysis (HYDROWIN module of EPI Suite software program)	• At 20°C the hydrolytic half-life is expected to remain constant at 126 days between pH 0 and 9 and then decrease to < 7 hours at pH 14. • Marine systems (pH =8.1): hydrolytic half-life decreases from about 8 years at 0°C to about 130 days at 20°C • suggesting FTI may be contributing to substantial FTOH and PFCA inputs in aquatic systems
	Young et al, 2008; Young and Mabury, 2010; Nielsen, 2014	Atmosphere (smog chamber experiment)	• Atmospheric lifetime of FTIs is expected to range from about 1 to 7 days (limited by photolysis), depending on time of year and latitude. • Photolysis of FTIs occurs via elimination of the iodine atom leading to the formation of the fluorotelomer aldehyde (FTAL) • FTAL atmospheric lifetime ~ 4 days (OH radicals) => Perfluoroaldehyd (atmospheric lifetime 1 day (photolysis) or 20 days (OH radicals)) => PFCA • LRT potential of FTIs => PFOA in remote areas
Polyfluorinated amides	Jackson and Mabury, 2013; Nielsen, 2014	Hydrolysis	• No hydrolysis of N-ethylperfluorooctanamide (EtFOA) to PFOA was observed at pH 8.5 after 8 days. • At pH 14, quantitative (98%) conversion of EtFOA to PFOA was observed after 24 h at room temperature. • No hydrolysis to PFOA was observed after 8 days at pH 8.5
	Jackson et al. 2013; Nielsen, 2014	Atmosphere (smog chamber experiment)	• Atmospheric lifetime of EtFBA (N-ethyl-N-(2-hydroxyethyl)perfluorooctaneamide) with respect to reaction with OH was estimated to be approximately 4.4 days. • Maximum mass yield of the corresponding PFCA (perfluorobutanoic acid PFBA) = 16% • Authors predict similar reaction kinetics for EtFOA (N-ethyl-perfluorooctanamide) as EtFBA since the length of a perfluorinated chain does not affect the reaction rate with OH • The primary oxidation products of EtFOA are expected to have much longer lifetimes with respect to reaction with OH and could be capable of contaminating Arctic air. The primary oxidation products are expected to react further to form PFOA.
	Martin et al., 2006; Nielsen, 2014	Atmosphere (smog chamber experiment)	• Atmospheric photo-oxidation of NetFBSA (N-ethyl perfluoro-butanesulfonamide): Three PFCAs were detected: 0.33% mol PFBA, 0.11% mol PFPrA (perfluoropropanoic acid), 0.09 % mol TFA (trifluoroacetic acid) ; at the same time only 0.65% of the starting material had unzipped COF_2; Extrapolation of this result suggests that 45% of the carbon in the perfluoroalkane chain will ultimately be incorporated into PFCAs upon complete oxidation, while the remaining fraction is expected to go to COF_2 (timeframe not given). • The authors suggest that it is evident that analogous perfluorooctanesulfonamide is potenital source for PFOA
Polymers	Russell et al., 2008	Soil	• Fluoroacrylate polymer • Estimated half-lives of the polymers = 95 to > 2000 years (all soils combined 1160 years) • Estimated half-lives of residual raw material and impurities ("residuals") = 12 to 43 days (all soils combined 27 days) • Major residuals in test substance were FTOH, fluoroacrylate monomer, FTOH acetate, and fluorotelomer olefin • maximum experimental PFOA concentrations are 24-28% of the theoretical amount that could be derived from 100% conversion of the residuals alone; If all 8:2 related analytes are summed 25-32% of the theoretical amount of PFOA formed from residuals.
	Renner, 2008		Comment the study from Russell et al. 2008: • Bottles may have released degradation products • Added FTOH could not be recovered • Experiment did not maintain mass balance => study from Russell et al. 2008 should not be given too much weight
	Washington et al., 2009	Soil	• Acrylate-linked fluorotelomer polymer • Estimated half-lives = 870-1400 years • Modelling for more finely grained polymers => estimated half-lives 10-17 years
	Washington et al.	Soil + hydrolysis	• Acrylate-linked fluorotelomer polymer

	2015		• Estimated half-lives = 33-112 years • PFOA concentrations increased up to ~1264% at day 376; 8:2 FTOH concentrations even increased up to 2894% (compared to day 0) • fluorotelomer-based polymer can undergo OH^--mediated hydrolysis
	Russell et al., 2010	Soil	• Fluorotelomer based urethane polymer • Including all data (until day 728) in kinetic evaluation: estimated half-lives = 79-241years (geomean = 132 years) • Including all data (until day 728) except one soil until day 273 in kinetic evaluation: estimated half-lives 28 -241 years (geomean 102 years) • Maximum PFOA concentration formed after 2 years ranged between 0.5 and 1.3 µmol/kg soil (initial conc. Polymer = 77.6 µmol/kg soil)
	Rankin et al., 2014		• Acrylate-linked fluorotelomer polymer • Estimated half-lives = 8-111 years • PFOA was the dominant product, constituting 57, 70, and 80% in all microcosm compartments in fluorotelomer-based acrylate polymer/soil, fluorotelomer-based acrylate polymer/plant, and fluorotelomer-based acrylate polymer/plant/biosolids, • Direct analysis: strucural changes of the polymer
	Rayne and Forest, 2010	Hydrolysis (SPARC software program)	• 8:2 fluorotelomer acrylate polymer segments: Landfills (40-50 °C, pH 4-9)) half-lives < 1 year marine systems (15°C, pH 8.1) half-lives = 170-270 years • Under dome saturated landfill conditions degradation could be resulting in significant fluxes of FTOHs and their degradation product (PFCAs) into ground and surface water

<u>*Degradation of Polyfluorinated silanes*</u>

Atkinson studied the kinetics of OH reactions with a series of organosilicon compounds including siloxanes and reported atmospheric lifetimes of >10 days (Atkinson, 1991). Tuazon et al. have investigated the products formed in the atmospheric degradation of volatile methylsilicon compounds (Tuazon et al., 2000). For tetramethylsilane the first steps in the photooxidation are reported to be:

$$Si(CH_3)_4 + OH \rightarrow (CH_3)_3SiC(\cdot)H_2 + H_2O$$
$$(CH_3)_3SiC(\cdot)H_2 + O_2 \rightarrow [(CH_3)_3SiCH_2(OO\cdot)] \rightarrow (CH_3)_3SiOCH_2O\cdot$$
$$(CH_3)_3SiOCH_2O\cdot + O_2 \rightarrow (CH_3)_3SiOCHO + HO_2$$
$$(CH_3)_3SiOCHO + H_2O \rightarrow (CH_3)_3SiOH + HC(O)OH$$

For telomer-substituted silanes and/or siloxanes the corresponding reactions will lead to the corresponding FTCA as product. The subsequent gas phase photo-oxidation of CF3(CF2)7CH2C(O)OH will eventually lead to some PFOA. The first steps are expected to be:

$$CF_3(CF_2)_7CH_2C(O)OH + OH \rightarrow CF_3(CF_2)_7C(\cdot)HC(O)OH + H_2O$$
$$CF_3(CF_2)_7C(\cdot)HC(O)OH + O_2 \rightarrow CF_3(CF_2)_7C(OO\cdot)HC(O)OH$$
$$CF_3(CF_2)_7C(OO\cdot)HC(O)OH + NO \rightarrow CF_3(CF_2)_7C(O\cdot)HC(O)OH$$
$$CF_3(CF_2)_7C(O\cdot)HC(O)OH \rightarrow CF_3(CF_2)_6CF_2(\cdot) + CHOC(O)OH$$

The reactions of the perfluoroalkyl radical leading to PFOA are (Wallington et al. 2006):

$$CF_3(CF_2)_6CF_2(\cdot) + O_2 \rightarrow CF_3(CF_2)_6CF_2(OO\cdot)$$
$$CF_3(CF_2)_6CF_2(OO\cdot) + NO \rightarrow CF_3(CF_2)_6CF_2(O\cdot)$$
$$CF_3(CF_2)_6CF_2(OO\cdot) + CH_3OO \rightarrow CF_3(CF_2)_6CF_2OH + CH_2O + O_2$$
$$CF_3(CF_2)_6CF_2OH \rightarrow CF_3(CF_2)_6CFO + HF$$
$$CF_3(CF_2)_6CFO + H_2O \rightarrow CF_3(CF_2)_6C(O)OH + HF$$

Established aqueous phase photo-oxidation reactions resemble the above and lead to the same product.

<u>*Degradation of Polyfluorinated olefins*</u>

$$CF_3(CF_2)_nCH=CH_2 + OH \rightarrow CF_3(CF_2)_nC(\cdot)HCH_2OH$$
$$CF_3(CF_2)_nC(\cdot)HCH_2OH + O_2 \rightarrow CF_3(CF_2)_nC(OO\cdot)HCH_2OH$$
$$CF_3(CF_2)_nC(OO\cdot)HCH_2OH + NO \rightarrow CF_3(CF_2)_nC(O\cdot)HCH_2OH + NO_2$$
$$CF_3(CF_2)_nC(O\cdot)HCH_2OH \rightarrow CF_3(CF_2)_nCHO + CH_2OH$$
$$CF_3(CF_2)_nCHO + H_2O_{(aq)} \rightarrow CF_3(CF_2)_nCH(OH)_{2(aq)}$$
$$CF_3(CF_2)_nCH(OH)_{2(aq)} + OH_{(aq)} \rightarrow CF_3(CF_2)_nC(\cdot)(OH)_{2(aq)}$$
$$CF_3(CF_2)_nC(\cdot)(OH)_{2(aq)} + O_{(aq)2} \rightarrow CF_3(CF_2)_nC(O)OH_{(aq)}"$$

Background information on the degradation of PFOA-related substances from precursors from Environment Canada and Health Canada 2012

(Section related to indirect releases; for references, see Environment Canada and Health Canada, 2012):

23. Potential sources for the formation of PFOA, such as the degradation or transformation of precursors, could lead to indirect environmental releases and contribute to the total amount of PFOA found in the environment.

24. Wallington et al. (2006) used a three-dimensional global atmospheric chemistry model (IMPACT) to indicate that n-$C_8F_{17}CH_2CH_2OH$ (i.e., 8:2 FTOH) degrades in the atmosphere to give PFOA and other PFCAs. Schenker et al. (2008) used a global-scale multispecies model (CliMoChem) to indicate that, until the year 2000, the contribution of atmospheric fluxes from perfluorooctanesulfonyl fluoride–based substances to the atmospheric deposition of PFOA in the Arctic was similar to the contribution from fluxes from FTOHs. Depending on the location and season, molar PFOA concentrations in the atmosphere are considered to be the correct order of magnitude to explain observed levels in Arctic biota (Wallington et al. 2006). The seasonal behaviour of PFOA is such that relatively high PFOA concentrations (>1.5 × 103 molecules/cm^3) extend throughout the Arctic during the Arctic summer, whereas winter PFOA concentrations are lower by an order of magnitude (Wallington et al. 2006).

25. The formation of PFOA through the thermolysis of fluoropolymers has been reported by Ellis et al. (2001, 2003). The results from these studies indicate the potential for this process to produce PFOA. However, Ellis et al. (2001, 2003) stated that this process is unlikely to release significant quantities of PFOA to the environment and would not contribute to its long-range transport. The onset of thermolysis of fluoropolymers occurs at 365°C (not an environmentally relevant temperature). These temperatures could be reached in industry and in household applications and, as such, the thermolysis of fluoropolymers could be considered a source of PFOA.

26. FTOHs have been shown to metabolize to PFOA in rats and rainbow trout (Hagen et al. 1981; Butt et al. 2010b). A β-oxidation mechanism was proposed by Hagen et al. (1981) to account for the formation of PFOA in rats. Butt et al. (2010a) found that exposure to 7:3 FTCA did not result from the formation and accumulation of PFOA in rainbow trout; however, PFOA was formed in the 8:2 FTCA and 8:2 fluorotelomer unsaturated carboxylate (FTUCA). The author proposed a beta-oxidation pathway proceeding from 8:2 FTUCA → 7:3 β-keto acid →7:2 ketone →PFOA, or that PFOA could be formed directly through the β-oxidation of the 7:3 keto acid. Butt et al. (2010b) exposed rainbow trout to 8:2 fluorotelomer acrylate (FTAc) via dietary exposure and found that 8:2 FTCA, 8:2 FTUCA and 7:3 FTCA were formed; however, the overall formation and accumulation of PFOA was low. Frömel and Knepper (2010) suggested that ultimate ethoxylate shortening of fluorotelomer ethoxylates (FTEOs) may result in FTOHs and thus contribute as a potential source of PFCs such as PFOA.

27. Dinglasan et al. (2004) showed aerobic biodegradation of 8:2 FTOH with an initial half-life of ~0.2 day per milligram of initial biomass protein followed by a second half-life of 0.8 day per milligram in a mixed microbial culture obtained from sediment and groundwater taken from a contaminated site, enriched on 1,2-dichloroethane and subsequently maintained using ethanol as the sole carbon source. This mixed culture was chosen because it was acclimated to degradation of chlorinated alkanes and alcohols and, therefore, considered to be active on fluorinated alcohols. The degradation of the alcohol occurred primarily through a mechanism leading to a telomer acid, which then underwent β-oxidation to yield PFOA, accounting for 3% of the total mass of FTOH initially at day 81. However, because this study was limited to identification and quantification of known or predicted transformation products, potential unknown transformation products were not identified. FTOHs were capable of metabolizing to PFOA in municipal wastewater treatment sludge (Pace Analytical 2001). Liu et al. (2007b) showed the microbial transformation of 8:2 FTOH to PFOA in clay soil and two pure soil bacterial cultures (Pseudomonas species).

28. Wang et al. (2005) conducted aerobic biodegradation studies of ^{14}C-labelled 8:2 FTOH in diluted activated sludge from a WWTP. Three transformation products were identified: 8:2 saturated acids, 8:2 unsaturated acids and PFOA, representing 27%, 6.0% and 2.1%, respectively, of the initial ^{14}C mass after 28 days. Results suggested that perfluorinated acid metabolites such as PFOA account for only a very small portion of the transformation products observed over the time frame considered (Wang et al. 2005). Wang et al. (2005) also suggested that the biological fate of 8:2 FTOH is determined by multiple degradation pathways, with neither β-oxidation nor any other enzyme-catalyzed reaction as a single dominant mechanism. A study by Dinglasan et al. (2005) showed that the oxidation of the 8:2 FTOH to the telomer acid occurred via the transient telomer aldehyde. The telomer acid was then further transformed via a β-oxidation mechanism, leading to the unsaturated acid and PFOA. However, a complete mass balance was not achieved, and the authors attributed this to binding of metabolites to biomass and other biological macromolecules, unaccounted metabolites, uptake of intermediates (formation of covalent linkages) or alternative degradation pathways (Dinglasan et al. 2005).

29. FTOHs may be released from polymeric materials or chemicals that incorporate FTOHs, or residual amounts of FTOHs that failed to be covalently linked to polymers or chemicals during production. FTOHs are used in fire-fighting foams, personal care and cleaning products, and oil, stain, grease and water repellent coatings on carpet, textiles, leather and paper (US EPA 2006a). FTOHs are also used in the manufacture of a wide range of products, such as paints, adhesives, waxes, polishes, metals, electronics and caulks. During the years 2000–2002, an estimated 5

× 106 kg of these compounds per year were produced worldwide, 40% of which was in North America (Dinglasan et al. 2004). Fluorotelomer-based raw materials and products are manufactured by a series of steps, beginning with Telomer A. Global Telomer A production between 2000 and 2002 was between 5000 and 6000 tonnes per year (Prevedouros et al. 2006).

30. Yoo et al. (2010) measured FTOHs in soil from fields (near Decatur, Alabama) to which sewage sludge had been applied. Sludges generated at a WWTP in Decatur, Alabama, have been applied to agricultural fields for more than decade; this WWTP received waste streams from industries that work with fluorotelomer compounds (Washington et al. 2010). Yoo et al. (2010) found that sludge-amended fields had surface soil FTOH concentrations ranging from 5 to 73 ng/g dry weight. The highest FTOH concentration was for 10:2 FTOH, which had concentrations ranging from < 5.6 to 166 ng/g dry weight. The half-lives for FTOHs ranged from 0.85 to 1.8 years, suggesting that sludge application is a possible pathway for the degradation of FTOHs to PFOA and other PFCAs. Washington et al. (2010) also found that these sludge-amended fields have high concentrations of PFCAs, including PFOA (<320 ng/g dry weight).

31. Measured vapour pressures of FTOHs range from 140 to 990 Pa. The calculated dimensionless Henry's Law constants for this class of compounds (e.g., 270 at 25°C for 8:2 FTOH) using the limited data available for water solubility and vapour pressure reveal the propensity of these compounds to partition to air (Dinglasan et al. 2004). Ellis et al. (2004a) showed the potential for FTOHs to react in the atmosphere with hydroxyl radicals to yield PFOA. Smog chamber studies indicate that FTOHs can degrade in the atmosphere to yield a homologous series of PFCAs (Ellis et al. 2004a). It is believed that oxidation of FTOHs in the atmosphere is initiated by reaction with hydroxyl radicals (Dinglasan et al. 2004; Ellis et al. 2004a). It was shown that perfluorooctyl sulphonamides react with hydroxyl radicals to yield PFOA (Hatfield et al. 2002). Although these experiments were conducted at environmentally unrealistically high concentrations of hydroxyl radicals and the results were qualitative rather than quantitative, they do show the potential for related products to react atmospherically to produce PFOA.

32. Stock et al. (2004, 2007) showed that there is a significant concentration of FTOHs present and widely disseminated in the North American atmosphere. A recent air sampling campaign detected FTOHs at tropospheric concentrations typically ranging from 17 to 135 pg/m^3, with urban locations having higher concentrations than rural areas (Martin et al. 2002; Stock et al. 2004). Loewen et al. (2008) studied atmospheric concentrations of FTOHs and lake water concentrations over an altitudinal transect in western Canada. Lake water samples were collected at Cedar Lake (a small lake near Golden, British Columbia), at Bow Lake in Banff National Park (Banff, Alberta) and at another unnamed small lake in Banff National Park (Banff, Alberta). Passive air samplers were deployed on altitudinal transects (800–2740 above sea level) from Golden, British Columbia, to Banff National Park. Loewen et al. (2008) noted that the amount of 8:2 and 10:2 FTOHs (<2.0 ng/sampler) increased with increasing altitude. Lake water concentrations of PFOA along the elevation transect were below 1 ng/L. No clear trend was evident between altitude and PFOA concentrations. Ellis et al. (2004a) and Wallington et al. (2006) indicated that telomer alcohols may be responsible in part for the presence of PFCAs in the Arctic and other non-urban areas where concentrations of peroxy radicals far exceed those of nitrogen oxides. It was noted that since the reaction of 8:2 FTOH with nitric oxide competes with the reaction that forms PFOA, the formation of PFOA should decrease with increasing nitrogen oxide concentrations. The production of PFOA is therefore suppressed in source regions that typically have nitrogen oxide concentrations of 100 parts per trillion (ppt) or greater.

33. In conjunction with the atmospheric measurements of the alcohols made by Martin et al. (2002) and Stock et al. (2004) and with the profile of the linear to branched isomers observed in Canadian Arctic samples (De Silva and Mabury 2004), Ellis et al. (2004a) concluded that telomer alcohols were a plausible source for some PFOA in remote regions. This conclusion is supported by Stock et al. (2007), who measured FTOHs in air at Cornwallis Island, Nunavut, in 2004. Mean concentrations of FTOHs ranged from 2.8 to 14 pg/m^3. This conclusion is also supported by observations of PFOA in US rainwater (Scott et al. 2003, 2006b). These results indicate that FTOHs are widely disseminated in the troposphere and are capable of long-range atmospheric transport. In addition, De Silva et al. (2009) detected branched PFOA isomers in Arctic and Lake Ontario sediment and surface water, Lake Ontario biota and humans; however, branched PFOA isomers were not detected in ringed seals and polar bears above the detection limits (3.6 ng/g).

34. Gewurtz et al. (2009) found PFOA as well as 8:2 FTUCA and 10:2 FTUCA in window film concentrations from indoor/outdoor/downtown/suburban/rural/carpet stores locations in Toronto, Ontario. Nilsson et al. (2010) found PFCs, including PFOA, in the blood of ski wax technicians (PFOA concentrations ranged from 4.8 to 535 ng/L). Nilsson et al. (2010) suggested that fluorinated organic compounds are added to glide waxes to prevent adhesion of snow, ice and dirt; fluorinated ski waxes are applied using heat of approximately 130–220oC, during which airborne particles and fumes containing a blend of gaseous organochlorine compounds are emitted. However, the authors did not analyze the glide waxes themselves to determine the presence of PFCs.

2.2 Environmental fate

2.2.1 Background information on persistence

(For references, see draft risk profile)

35. PFOA is extremely stable within the natural environment due to its chemical properties. Degradation of the substance does not occur under environmentally relevant conditions. This is expanded upon further by Siegmund et al, 2000, as reported in ECHA 2013a):

> *"When all valences of a carbon chain are satisfied by fluorine, the zig-zag shaped carbon skeleton is twisted out of its plane in the form of a helix. This situation allows the electronegative fluorine substituents to envelop the carbon skeleton completely and to shield it from chemical attack."*

36. This chemical structure means that not only is PFOA highly stable as a molecule, but it is also highly resistant to conventional chemical mechanisms of degradation such as hydrolysis, photolysis, or chemical substitution, a fact which is corroborated by the US EPA (USEPA, 2014), Australian NICNAS (2015a and 2015b), Environment Canada and Health Canada (2012) and OECD (2006). In particular, both assessments conducted by Environment Canada and Health Canada and the study by OECD noted that there was no clear degradation of PFOA under abiotic or biotic conditions.

37. The authors of ECHA (2013a) conclude that PFOA is not ready biodegradable using standard test methods. The results of one non-standard aerobic biodegradation simulation test, one non-standard anaerobic biodegradation simulation test and field monitoring data on PFOA from contaminated sites provide evidence that no biodegradation in water, soil and sediment occurs. The monitoring data show that PFOA in soil leaches over time and can be a long term contamination source to underlying groundwater.

38. Within the aquatic compartment under natural environmental conditions, PFOA has a half-life of greater than 92 years with the most likely value of 235 years and shows no obvious decay from photodegradation (Todd, 1979; Hatfield, 2001; 3M, 2001). In aquatic environments where PFOA undergoes indirect photolysis, the half-life was estimated to be longer than 349 days (Hatfield, 2001; ECHA, 2013a, 2013b). A review of half-life data and mechanisms of decay is presented in Table 7.

Table 7: Reported findings on the persistence of PFOA (adapted from Environment Canada and Health Canada, 2012)

Medium	Mechanism	Degradation Half-life	References
Water	Photolysis	No photodegradation	Todd 1979; Hatfield, 2001
Water	Indirect photolysis	> 349 days	Hatfield, 2001
Water	Hydrolysis	~ 235 years	3M, 2001
Air	Hydroxyl reaction	~ 130 days	Hurley et al., 2004
Sludge	Biodegradation	> 2.5 months	Pace Analytical , 2001
Soil/sludge	Biodegradation	> 259 days	Liou et al., 2010

39. On the basis of the available data, abiotic degradation of PFOA in the atmosphere is expected to be slow. The atmospheric lifetime of PFOA has been predicted to be 130 days (conclusion by analogy from short-chain perfluorinated acids; see ECHA 2013a; Hurley et al., 2004).

40. Persistence within the terrestrial environment mirrors the position seen in the aquatic environment, with high persistence, slow degradation and long half-lives. Both the REACH Annex XV restriction dossier (ECHA, 2015a) and REACH proposal for identification of PFOA as an SVHC (ECHA 2013a) state that PFOA was not biodegradable and that, based on high persistence, it was not possible to calculate half-lives in soil or sediment. This was the same conclusion presented within the screening dossier (UNEP/POPS/POPRC.11.5) "Screening tests for PFOA or APFO indicated little (max. 7% in 28 days) or no biodegradation (MITI, 2002; Stansinakis et al., 2008; OECD 2006; 3M, 1978)".

2.2.2 Bioaccumulation

Background information on bioconcentration studies in aquatic organisms

(For references, see draft risk profile)

41. ECHA (2013a) discusses the issue of BCF/BAF and BMF/TMF. Because of the high surfactant capacity and solubility of PFOA, fish might excrete PFOA through the gills leading to reduced uptake and bioaccumulation. This explains the often low values seen in BMF/BAF tests using fish. Equally for BMF/TMF analysis where the higher predators within the food chain are fish species it can be seen that the critical values fall below 1. For non-fish species,

particularly air-breathing terrestrial and avian species that do not have this mechanism, bioaccumulation is shown to occur. ECHA (2013a) highlights five aquatic BMF studies, where the higher tier predators were air breathing species, and all indicate bioaccumulation occurred. These include the food chains for walrus (liver)/clam, narwhal (liver)/Arctic cod, beluga (liver)/Arctic cod, beluga whale (liver)/Pacific herring (liver) and Artic cod (liver)/marine artic copepod: the BMFs for PFOA were 1.8, 1.6, 2.7, 1.3 and 2.2 respectively, indicating biomagnification (Tomy et al., 2004, 2009).

42. There are several studies investigating the isomer profile of PFOA in the aquatic organisms. In a marine food web from Taihu Lake, China, the linear isomer of PFOA is the dominant form found in the biota with a range of 91.9-100% of total PFOA detected (Fang et al., 2014). Toxicokinetics study in rainbow trout demonstrated an accumulation propensity of n-PFOA and two minor branched PFOA isomers in tissue although the branched isomers had lower accumulation ratio values than n-PFOA (De Silva et al., 2009).

43. BCF values within the aquatic environment tend to be low. Regarding fresh water organisms, Ulhaq et al. conducted a laboratory experiment with the model organism zebrafish (*Danio rerio*) exposing it to PFOA. Under steady-state conditions, the BCF were highest in liver and intestine and were approximately 100-fold of the exposure medium. During the wash-out phase, half of the accumulated concentration during the steady-state phase was excreted after 13-14 days, confirming the outcomes of the reviews quoted above (BCF are below the critical value of 5,000) (Ulhaq et al., 2015). Reviews of available studies reported in ECHA (2013a), Environment Canada and Health Canada (2012) and OECD (2011) are all below the critical value of 5,000. Based on analysis of fish and bi-valves, the studies reviewed within the REACH Annex XV restriction dossier (ECHA, 2015a) BAFs for PFOA range between 0.9 and 266. ECHA 2015a concludes that PFOA does not accumulate in water breathing animals because BCFs range from 1.8 to 8.0, BAFs range from 0.9 to 266, BMFs range from 0.02 to 7.2 whereas most of the data are below 1 and TMFs range from 0.3 to 0.58 in aquatic piscovorous food webs. The studies quoted by Environment Canada and Health Canada (2012) BCFs for PFOA on fish range from 0.04 – 27. The OECD and SIDS data (2011) for rainbow trout and carp quote values of 8 and 9.4, respectively. This supports the position put forward in the screening dossier (UNEP/POPS/POPRC.11/5) that the evidence for meeting the bioaccumulation criterion in the aquatic environment has not been met.

44. However, this is expected to be the case based on the high solubility and surfactant properties alluded to and the capacity of gill breathing species to excrete PFOA, its salts and related compounds readily, preventing significant uptake of PFOA to allow bioaccumulation within body tissues. Studies for air breathing aquatic species are scarce. However, the study by Houde et al. (2006) on bottle nosed dolphins indicates the uptake and bioaccumulation of PFOA with concentration particularly within the liver. This study quotes a BMF >1 for liver concentrations and <1 for whole body. Overall, the potential for bioaccumulation with marine mammals is confirmed by the Houde et al. (2006) study. There is evidence that PFOA biomagnifies in air-breathing mammals because BMFs range from 1.3 – 125 for selected predator prey relationships and TMFs range from 1.1 to 13 for selected food chains. Butt et al. (2008) determined regionally-based ringed seal/polar bear BMF values for PFOA that ranged from 45 – 125. These regionally based BMFs were calculated by grouping ringed seal populations to corresponding similarly located polar bear populations in the Canadian Arctic. Houde et al. (2006) reported BMFs of 1.8–13 using whole-prey homogenates and whole-body bottlenose dolphin (*Tursiops truncatus*) conjugate base concentrations in the bottlenose dolphin food web at Charleston, South Carolina. Kelly et al. (2009) found a TMF of 3.28 over the Canadian Arctic (Hudson Bay region) marine food web (macroalgae, bivalves, fish, seaduck and beluga whale). Martine van den Heuvel-Greve et al. (2009) found BMFs of 3.8 and 23, respectively, for the benthic and the pelagic food webs with harbour seals (*Phoca vitulina*) as the apex predator. BMFs for other species in the Westerschelde, Netherlands estuary ranged from 0.03 to 31.

Background information on bioaccumulation studies in terrestrial organisms

(For references, see draft risk profile)

45. A number of studies have been completed highlighting the presence of PFOA in higher-tier terrestrial species. Rüdel et al. (2011) conducted studies based on the German Environmental Specimen Bank (ESB) for three time periods (1994-1996, 2000-2002, and 2006-2009) to gain an overview of marine, limnetic and terrestrial biota for presence of perfluorinated chemicals. This detected appreciable levels of PFOA within the eggs of herring gulls measured at 6.5 to 118 ng/g (ww). Martin et al. (2004) detected the presence of PFOA within liver samples from Canadian bears with concentrations of 3-13 ng/g and found that polar bears, which occupy the highest trophic level in the Canadian Arctic, have higher levels of PFOA than all other Arctic organisms examined. Müller et al. (2011) conducted ecosystem studies within the Yukon and Nunavut (North West Territories), Canada as part of a BMF assessment for PFOA. The results of this study highlighted the presence of PFOA above detectable limits in lichen, caribou and wolves. In all three studies the potential for PFOA to bioaccumulate within terrestrial species was confirmed. A feeding study with fattening pigs (Numata et al., 2014) showed that PFAAs accumulated in plasma, fat, muscle tissue, liver, and kidney with decreasing percentage of unexcreted PFAA – plasma showed percentages of up to 51%. The authors modelled related toxicokinetics and concluded that pigs exhibit longer elimination half-lives (236 days for PFOA) than reported for most organisms.

2.2.3 Background information on potential for long-range environmental transport

(For references, see draft risk profile)

46. The screening dossier (UNEP/POPS/POPRC.11/5) highlights the high persistence and stability of PFOA, its salts and PFOA-related compounds, particularly within the air compartment. Hurley et al. (2004) (see Table 7) quote that the atmospheric lifetime of PFOA with respect to hydroxyl radicals has been predicted to be 130 days. This would be sufficient amount of time to allow PFOA to travel considerable distances from the point of emission.

47. Franklin (2002) calculated an atmospheric lifetime of PFOA to be in the order of days if PFOA is emitted from a ground source, and therefore, PFOA is likely not subject to long-range transport. However, if PFOA is produced from an atmospheric source (i.e. via precursors) and if the major loss mechanism is wet or dry deposition, then it may have a lifetime of 20-30 days before deposition (Ellis et al, 2004b). This would be sufficient amount of time to allow PFOA transport over many thousands of kilometres.

48. The proposal to list PFOA, its salts and PFOA-related compounds under the Stockholm Convention (UNEP/POPS/POPRC.11/5) also highlights the potential value of environmental models to support the long-range transport of these substances:

> "*Gomis et al., 2015 used COSMOtherm and SPARC to estimate the physicochemical properties of various novel perfluoro alkyl sulphonates in comparison to PFOS, PFOA and 8:2 FTOH. The US-EPA Epi Suite was used to predict degradation half-lives in air, water and soil (for PFOA the estimated values were 31 days in air and 720 days in water and soil). The results were then used as input parameters for the OECD Pov and LRTP Screening Tool*[2] *which has been developed with the aim of using multimedia models for estimating overall persistence (Pov) and long-range transport potential (LRTP) of organic chemicals at a screening level in the context of PBTs / POPs assessments. The OECD overall persistence (Pov) and Long-Range Transport Potential (LRTP) Screening tool gave the following result for PFOA: Pov = 1038 days; critical travel distance (CTD) = 1745 km (both like PFOS); the travel efficiency (TE) in % was higher than for PFOS with 0,0146. The in silico methods predict that PFOA, like PFOS, is globally distributed.*"

49. Modelling studies have reported oceanic transport as the dominant pathway for distributing PFOA (primarily from direct sources) to the Arctic. It is believed that ocean circulation and its variations are factors that determine the long-range distribution and fate of PFOA (Stemmler and Lammel, 2010; Armitage et al., 2009).

Presence of PFOA, its salts and PFOA-related compounds in remote areas

50. Alongside the persistent characteristics of PFOA, its salts and PFOA-related compounds and environmental modelling data to suggest the capacity for long-range transport, environmental monitoring data from a number of studies also exist to help corroborate the model estimates. Table 8 provides details of monitoring for PFOA within surface water, ice, sediment and biota from remote locations far from the point of use and emission.

Table 8: Presence of PFOA in remote areas

Sample	Value	Remarks	Reference
Surface Water			
Canadian Arctic lakes (Armituk Lake, Char Lake, Resolute Lake)	0.5 – 16 ng/L (range)	2003-2005	(Stock et al., 2007)
Sea Water / Ice			
Baydaratskaya Bay (Russian Federation)	130.7 (±77.2) pg/L (average ± std dev.)	2007	(Saez et al., 2008)
Greenland Sea	20 – 111 pg/L (range)		(Theobald et al., 2007)
Sediment			
Canadian Arctic lakes (Char Lake and Resolute Lake)	1.7 and 7.5 ng/g dw <1.1 and 2.3 ng/g dw 1.2 and <1.8 ng/g dw (range)	0-1 cm (depth) 1-2 cm 2-3 cm	(Stock et al., 2007)
Air			
High volume air samples collected from research vessels on Atlantic Ocean, Southern Ocean and Baltic Sea	0.7 pg/m^3 (average)	2007-2008	(Dreyer et al., 2009)

High volume air samples from a land-based site close to Hamburg, Germany	0.3 pg/m^3 (average)		
Birkenes Zeppelin Andøya (Norway)	0.32 pg/m^3 0.22 pg/m^3 0.19 pg/m^3 (average)	2014	(NILU, 2015)
Cornwallis Island (Canada)	1.4 pg/m^3 (average)	2004	(Stock et al., 2007)
Biota			
Polar bear (liver) (East Greenland)	0.6 – 14 ng/g ww 6.8 – 15.8 ng/g ww 11.8 – 17.6 ng/g ww (range)	1990 (year) 1995 2006	(Dietz et al., 2007)
Polar bear (liver) (North American Arctic, European Arctic)	2.4 – 36 ng/g ww (range)		(Smithwick et al., 2005)
Ringed seal (liver) (Arviat – Canadian Arctic)	<0.85 – 3.6 ng/g ww (range)		(Butt et al., 2007)

51. There is evidence of PFOA present in the Arctic. A number of studies reported detection of PFOA in the Canadian Arctic (Environment Canada and Environment Canada and Health Canada, 2012). PFOA has been detected in concentrations from the low- to mid- picograms per litre (pg/L) range in remote regions of the Arctic ice cap (US EPA, 2014), and monitoring of sea water (as well as air) in Arctic regions as part of a study on PFCs has been carried out by the Arctic Monitoring and Assessment Programme (AMAP) (Butt et al., 2010b; AMAP, 2014). From this programme, PFOA was detected in surface sea waters off the coast of Greenland, in the Labrador Sea, in the sea around Iceland/Faroe Islands and in the Russian Arctic with the average concentrations of <130 – 111 pg/L, 55 pg/L, 3.5 – 4.02 ng/L and 131 pg/L, respectively (Butt et al., 2010b). The data provided in Table 8 show the widespread presence of PFOA in Arctic regions. Another study (Wang et al., 2014b) in a remote area, namely on the Tibetan Plateau, showed that PFAAs including PFOA were detectable in snow and ice cores. Samples were taken from two different glaciers with different accumulation periods and had PFOA concentrations of up to 243 pg/L. The authors explained differences in concentrations by respective upwind sources from Europe, Asia or India. Xie et al. (2015) measured PFAS in air and snow samples from glaciers on the Norwegian island Spitzbergen. They detected 6.7 to 39 pg/m^3 in air and 330 to 690 pg/L in snow.

52. When PFOA is produced from an atmospheric source (i.e. via degradation of precursors) and when the major loss mechanism is wet or dry deposition, then it may have a lifetime of 20–30 days before deposition (Ellis et al., 2004a). Fluorotelomer alcohols such as n-$C_8F_{17}CH_2CH_2OH$ appear to be a global source of persistent bioaccumulative perfluorocarboxylic acid pollution, and present modeling results show that with current estimates of chemistry and fluxes the atmospheric oxidation of 8:2 FTOH can provide a quantitative explanation for the presence of PFCAs in remote regions (Wallington et al., 2006; Young et al., 2007). Del Vento et al. (2012) and the OECD (2008) (referenced in NICNAC, 2015a) both discuss the presence of PFOA precursors in the atmosphere around the Antarctic. These studies note the widespread presence of FTOHs, particularly C8 varieties of FTOHs detected around the Antarctic. Del Vento's study which included the sampling of air off the West Peninsula of the Antarctic noted that the longer chain FTOHs dominated with 8:2 and 10:2 species of FTOHs producing the highest concentrations, averaging of 9.9 and 7.4 pg/m^3 respectively. As a comparison, the Danish Ministry of Environment (2013) quotes a set of sea bound air sampling which detected concentrations of FTOH (8:2) in the southern oceans around Antarctica of 4.5 pg/m^3 and concentrations from European atmospheric sources of 335 pg/m^3. In another study (Wang et al., 2015) conducted in the Antarctic peninsula and across the Atlantic, snow and air samples were investigated for presence of 12 PFAS. In the snow samples 125 to 303 pg/L were detected, the air samples showed concentrations of 2.8 to 68.8 pg/m^3. In the southern hemisphere, especially around the Antarctic Peninsula, higher ratios FTOH (8:2 to 10:2 to 6:2) were found. The authors concluded that long-range transport was the reason for the occurrence of PFAS in this region.

53. PFASs were measured in background locations of the Norwegian Arctic and the Norwegian mainland, which are little effected by anthropogenic sources (NILU, 2013). PFOA was detected in all sediment, water and pooled soil samples. PFOA was detected in 30% of brown trout samples, in 100% of harbour seal liver (mean: 0.8 mean ng/g wt) and 100% eider eggs (mean: 1.62 ng/g wt), 80 % of Herring gull eggs (mean: 0.11 ng/g wt), and 30% of cod liver (mean: 0.09 ng/g wt) (as summarised in UNEP/POPS/POPRC.11/5).

54. Lescord et al. (2015) report evidence of perfluorinated and polyfluorinated compounds in lake food webs from the Canadian high Arctic. Evidence is also reported to show the contamination in eggs of the high-artic ivory gull (*Pagophila eburnea*) (Lucia et al., 2015) and other Arctic and North Atlantic mammals (Bytingsvik et al., 2012;

Rotander et al., 2012). Different PFOA isomers were detected in liver samples of polar bears from the Canadian Arctic and eastern Greenland such that Greenland polar bear samples showed a variety of branched isomers while only the linear PFOA isomer was determined in Canadian samples (de Silva and Mabury, 2004).

55. In recent sampling of snow in remote locations and water from mountain lakes, PFCs were present in nearly all the samples, including short chain PFCs. PFOA was measured in snow samples from Slovakia (between 0.107-0.348 ng/L), Switzerland (0.087 ng/L) and Italy (0.209 ng/L) (Cobbing, 2015).

56. According to Zhao et al. (2012a), PFOA (15 g/L) was occasionally found in the Southern Ocean. Furthermore, PFOA was detected in 30% of livers of albatrosses from the Southern Ocean with a concentration range of <0.6-2.45 ng/g wet wt. The study documents the existence of low but detectable levels of PFOA in fauna in remote areas of the Southern Hemisphere, suggesting distribution of these compounds on a global scale (Tao et al., 2006).

Potential mechanisms for long range transport

57. Bengtson Nash et al. (2010) and Prevedouros et al. (2006) discuss two key mechanisms to aid the long range transport of PFOA to remote regions. The first mechanism involves a release to environment either as air-borne dust particles contaminated directly by PFOA or salts of PFOA, or a direct release to surface waters. These releases then undergo long range transport through a cycle of deposition and volatisation from marine waters, with the transport in water via the marine environment likely to be the dominant delivery method to the Canadian Arctic regions. It has been suggested that hydrospheric transport will form a slow but primary input pathway of PFCs to the Antarctic region (Bengtson Nash et al., 2010). The second mechanism involves a delivery through air-borne precursors (such as FTOHs) again through contaminated dust particles and then the degradation of these materials to form PFOA *in situ* on arrival.

58. ECHA (2015a) goes on to discuss the atmospheric mechanism further:

> *"*Due *to the relative vapour pressures of APFO, PFOA, and perfluorooctanoate (PFO), the chemical form potentially most subject to gas-phase atmospheric transport is PFOA. Franklin (2002) suggested that in the presence of water in air (humidity), gaseous PFOA condenses to aerosol particles and dissociates to the corresponding PFO, resulting in a low vapour pressure. Pure PFOA at room temperature has a moderate vapour pressure (2.3Pa). The vapour pressure of APFO is much lower with (sic) 0.008Pa. APFO and PFOA dissolved in water dissociate to ions. Although the dissociated fraction is not subject to Volatisation, depending on the pH, pure PFOA is expected to volatise from water to a certain degree."*

59. This would suggest that possible cycling between atmosphere and water through wet/dry deposition and volatisation is possible and would aid the long range transport of PFOA. In a study on distribution and long-range transport of polyfluoroalkyl substances in the Artic, Atlantic Ocean and Antarctic coast, Zhao et al. (2012a) concluded that the elevated PFOA level resulted from melting snow and ice in Greenland Sea implies that the Arctic may have been driven by climate change and turned to be a source of PFASs for the marine ecosystem.

2.3 Exposure

2.3.1 Background information on environmental monitoring data

(For references, see draft risk profile)

60. Various studies demonstrate that PFOA is ubiquitously present in the environment. ECHA 2015a contains a selection of studies, which report detections of PFOA and related compounds in several compartments (surface water, deep-sea water, drinking water, wastewater treatment plant, sediment, groundwater, soil, atmosphere, dust, biota, and human) at worldwide sampling locations (see ECHA, 2015a, Table A.B.4-8 in Appendix B; see also section 2.3.1 of the INF-document). No large-scale monitoring program has been conducted for PFOA and only limited time trend studies are available. There is not sufficient information available to conclude on the trend of environmental concentrations. The few available time trend studies indicate a decreasing trend in biota. As PFOA is not degradable this decreasing trend is not proven by water and sediment samples suggesting that oceans and sediments are sinks of PFOA (ECHA 2015a).

61. Environmental monitoring data has been gathered for a number of sites still producing PFOA and its related compounds in China. Bao et al. (2011) reported maximum PFOA concentrations of 48 ng/g dry weight in sediment and 668 ng/L in river water of the River Xi along the industrial park where a production facility making PTFE and PFBS is situated. The study also found PFOA concentrations in groundwater beneath the industrial park of up to 524 ng/L and that PFOA levels in drinking water from the public water supply system in the area ranged between 1.3-2.7 ng/L. More recently, median PFOA concentrations of 26.5 ng/L were reported from 35 river water samples and 34 sediment samples collected from rivers in the Liaodong Bay basin, where two fluorine industry parks are situated (Chen et al., 2015). Very high concentrations of PFOA (up to 496,000 ng/L) were also recently detected in the Ziaoqing River (Shi et al., 2015), and PFOA was the dominant perfluoroalkyl carboxylic acid found in Tianjin (concentration range of 8.58-20.3 ng/L) and in Weifang (concentration range of 6.37-25.9 ng/L) (Yao et al., 2014). Lastly, PFASs and the isomers of PFOA were analysed in fresh snow samples collected from 19 cities in northern

China in 2013. The levels of total PFASs in the snow samples were 33.5-229 ng/L (PFOA 9.08-107 ng/L), suggesting heavy atmospheric pollution of PFASs in northern China. Isomer profiles of PFOA in the snow were in agreement with the signature of the historical 3M electrochemical fluorination products, suggesting that these products were still produced and used in China (Shan et al., 2015). The results of the Jiang et al. (2015) study suggest that, although it is widely accepted that telomerization is currently the predominant manufacturing method for PFOA, yielding an isomerically pure and linear product, electrochemical fluorination is still used by some manufacturers in China.

62. PFOA was found in the majority of ground water samples in the EU and of public drinking water systems in the U.S. (Loos et al., 2010; Post et al., 2009; US EPA, 2016). In particular, PFOA was found in 66% of 164 ground water samples collected from 23 European countries (Loos et al., 2010) and in 57% of samples from public drinking water systems (mainly representing ground water) in New Jersey, USA (Post et al., 2009). A study investigating PFOA in soil from a U.S. metropolitan area reported a general increase in PFOA concentration in soil with increasing depth, suggesting a downward movement of PFOA toward the groundwater table. This study concluded that the groundwater contamination of PFOA often follows its release to surface soils for years, if not decades, and that the aquifer can be a major source of exposure for communities living near point sources (Xiao et al., 2015). A study from the Netherlands suggested that the general atmospheric deposition of PFOA can also lead to diffuse contamination of ground water (Eschauzier et al., 2013). Landfill can also be a source of PFOA to ground water, as indicated by high levels of PFOA found in Chinese landfill leachate samples (Yan et al., 2015). Currently, there is a large-scale drinking water monitoring program funded by the U.S. EPA under the Safe Drinking Water Act called the Unregulated Contaminant Monitoring Rule (UCMR). The third round of the UCMR (UCMR3) conducted monitoring of over 4800 public water systems in the US between 2013 and 2015 for 28 chemicals including PFOA and 2 viruses. The occurrence data from UCMR3 can be found online and contains preliminary chemical analyses (US EPA, 2016).

63. The results of a Swedish study provide evidence that the historical use of aqueous film forming foams has contaminated an aquifer with PFOA as well as PFOS, and constant PFAA purification of the aquifer will be required before using it for drinking water production (Filipovic et al., 2015). High concentrations of PFOA have been found in the soil, groundwater and drinking water in a number of municipalities in Sweden. The release has occurred mainly in the vicinity of airports (civil and military) and fire-fighting training sites (KemI, 2016).

64. An assessment of the relative contribution of historic (i.e. electronically fluorinated) and contemporary (i.e. telomer) sources of PFOA in various seawater samples indicated that the dominant PFOA source(s) to the Pacific and Canadian Arctic Archipelago are either from (1) direct emissions of contemporary PFOA via manufacturing or use in Asia or (2) from atmospheric transport and oxidation of contemporary PFOA-precursors (Benskin et al., 2012).

65. Environmental monitoring data for Japan in 2012 indicate that PFOA concentrations in surface water range between 0.24-26 ng/L, and between 0.012-0.28 ng/g dry weight in sediment. Data for PFOA concentrations in air from 2013 vary between the warm and cold seasons with a higher concentration range reported for the latter (0.0032–0.19 and 0.0030–0.053 ng/m^3) (Japan, 2015).

66. In Italy, environmental monitoring data from 2008 until 2013 at a fluoropolymer plant sited in Piedmont on the river Bormida in the Po basin, indicated measured PFOA concentrations of up to 6,500 ng/L (ECHA, 2014b).

67. In Australia, environmental monitoring data for 13 landfills found that PFOA was present in all leachate samples at a concentration range of 19-2,100 ng/L with a mean concentration of 446 ng/L (Gallen et al., 2016). Also, PFOA was detected in all water (range: 4.2-6.4 ng/L), 50% of gull eggs (range: <0.6-2.6 ng/g wet wt) and 20% of sediment (0-0.16 ng/g dry wt) samples collected in the upper vicinity of the upper Parramata River near Syndey, Australia (Thompson et al., 2011a).

68. In South Africa, PFOA was detected in all sediment samples from 3 different rivers, Diep, Eerste and Salt, with the ranges of concentration of 10.7-497.5, 15.2-193.2 and 38.6-187.0 ng/g dry wt, respectively. The PFOA concentrations found in these rivers was much higher than those found in other countries such as the USA, Canada, Japan and Germany, suggesting that there is cause for concern (Mudumbi et al., 2014).

69. There is abundant evidence to show detected levels of PFOA in marine, limnetic and terrestrial biota worldwide (de Silva and Mabury, 2004; Martin et al., 2004; Smithwick et al., 2005; Houde et al., 2006; Butt et al., 2007; Dietz et al., 2007; Ishibashi et al., 2008; Sonne et al., 2008; de Silva et al., 2009; Müller et al., 2011; Rüdel et al., 2011; Bytingsvik et al., 2012; Rotander et al., 2012; NILU, 2013; Fang et al., 2014; Lescord et al., 2015; Lucia et al., 2015). As previously discussed in Sections 2.2.2 and 2.2.3, bioaccumulation of PFOA occurs across trophic levels, and the detection of PFOA in biota at remote locations suggest the capacity of PFOA for long range transport, respectively.

Background information on environmental trends from ECHA 2015a

(For references, see ECHA 2015a)

70. Decreasing trends of PFOA in environmental samples have been reported by Ahrens et al. in harbour seals from the German Bight sampled between 1999 and 2008 (Ahrens et al., 2009b). Decreasing trends were also found in Greenland ringed seals and polar bears (Riget et al., 2013). Decreasing concentrations were found in Lake Trout from

Lake Ontario (Myers et al., 2012). However, increasing concentrations were identified for suspended sediment samples of Lake Ontario and Niagara River (Myers et al., 2012). PFOA concentrations increased from 2001 to 2006 (doubling time = 2 years). Furthermore, increasing PFOA trends were found in three sediment cores from western, central, and eastern Lake Ontario (1988 to 2004; doubling time = ~4 years in the western Lake Ontario core) (Myers et al., 2012).

Background information on environmental monitoring data from ECHA 2015a

(For references, see ECHA 2015a)

Table 9: Measured levels of PFOA and PFOA-related substances from global sampling points in various compartments, extracted from ECHA (2015a)

Compartment	Location	Sampling year	Substance	Concentrations	Reference
Surface water	River Elbe North Sea German Coast Open Norwegian Coast Baltic Sea	2007	PFOA	4.36 – 4.81 ng/L (dissolved phase) 0.08 – 3.02 ng/L (dissolved phase) 0.02 – 0.07 ng/L (dissolved phase) 0.07 – 0.35 ng/L (dissolved phase) 0.25 – 4.55 ng/L (dissolved phase)	(Ahrens et al., 2010a)
	Greenland Sea Atlantic Ocean Southern Ocean	2009 2010 2010-2011	PFOA	0.045 – 0.16 ng/L <0.013 – 0.16 ng/L < 0.013 – 0.15 ng/L	(Zhao et al., 2012)
	surface water samples collected on board the research vessel Polarstern (52°N-69°S); Northern Europe – Atlantic – Southern Ocean	2008	PFOA	<0.0005 – 0.223 ng/L (dissolved phase)	(Ahrens et al., 2010b)
	surface water samples collected on board the research vessel Polarstern (67.5-80.4°N); East Greenland Arctic Ocean	2009	PFOA	< 0.012 – 0.12 ng/L (in 37 of 38 samples detected) Mean concentration: 0.051 ± 0.030 ng/L	(Busch et al., 2010b)
	> 100 individual water samples from over 100 European rivers from 27 European Countries	2007	PFOA	Frequency of detection: 97% (LOD = 1 ng/L) Maximum: 174 ng/L Median: 3 ng/L	(Loos et al., 2009)
	14 major European rivers	2005-2006	PFOA	<0.65 – 200 ng/L	(McLachlan et al., 2007)
	539 river samples collected from 41 cities in 15 countries (Asia, Europe, North America)	2004-2010	PFOA	PFOA was detected in all 41 cities in 89% of the samples (industrialized and non-industrialized) Average in each city: 0.2 – 1,630.2 ng/L	(Kunacheva et al., 2012)
	Rhine River and selected	2006	PFOA	<2 – 48 ng/L	(Skutlarek et
	tributaries, Germany Ruhr area, Germany Moehne River and selected tributaries, Germany			<2 – 3,640 ng/L <2 – 33,900 ng/L	al., 2006)
	Rhine River, Germany Ruhr River, Germany Moehne River, Germany	2008-2009	PFOA	<10 –11 ng/L <10 – 88 ng/L 48 – 160 ng/L	(Wilhelm et al., 2010)
	Tokyo Bay Offshore of Japan Coastal area of Hong Kong Coastal area of China Coastal area of Korea Sulu Sea South China Sea Western Pacific Ocean Central to Eastern Pacific Ocean North Atlantic Ocean Mid Atlantic Ocean	2002-2004	PFOA	1.8 – 192 ng/L 0.137 – 1.06 ng/L 0.637 – 5.45 ng/L 0.243 – 15.3 ng/L 0.239 – 11.35 ng/L 0.088 – 0.51 ng/L 0.16 – 0.42 ng/L 0.136 – 0.142 ng/L 0.015 – 0.062 ng/L 0.16 – 0.338 ng/L 0.1 – 0.439 ng/L	(Yamashita et al., 2005)
	26 locations between the Asian and Antarctic regions Shanghai Western Pacific Ocean Pacific Ocean Eastern Indian Ocean Indian Ocean Antarctica		PFOA	<0.001 – 0.4416±0.0064 ng/L 0.2784±0.0688 – 0.4416±0.0064 ng/L < 0.005 – 0.0213±0.0015 ng/L < 0.005 – 0.007 ng/L < 0.005 – 0.0119±0.0011 ng/L 0.0064±0.0014 – 0.011±0.0015 ng/L < 0.005 ng/L	(Wei et al., 2007)
	Conasauga River, Oostanaula River, Coosa River, Georgia, USA	2008	PFOA	< 0.07 – 204 ng/L	(Lasier et al., 2011)
	Cornwallis Island, Nunavut, Canadian Arctic Amituk Lake, Char Lake, Resolute Lake, Meretta Lake	2003,2005	PFOA	0.5 – 16 ng/L	(Stock et al., 2007)
	Winam Gulf of Lake Victoria, Kenya; Sites selected included	2006-2007	PFOA	<0.4 – 96.4 ng/L	(Orata et al., 2009)

	along rivers that flow near industries, residential estates and waste treatment facilities				
	Baydararskaya Bay, North Russian Federation within the North Pole Region (ice-core, surface to 300 cm)	2007	PFOA	0.1307±0.0772 ng/L	(Saez et al., 2008)
	Hong Kong	2009	6:2 diPAP 6:2/8:2 diPAP 8:2 diPAP PFOA	<0.010 – 0.029 ng/L <0.010 ng/L <0.010 – 0.18 ng/L 0.31 – 4.41 ng/L	(Loi et al., 2013)
Deep-sea water	Sulu sea (deep water; 1000-3000m) Central to Eastern Pacific Ocean (deep water; 4000-4400m)	2002-2004	PFOA	0.076 – 0.117 ng/L 0.045 – 0.056 ng/L	(Yamashita et al., 2005)
Drinking water/ tap water	Tarragona Province, Spain (public fountains of Reus, Tarragona, Tortosa, and Valls)	2007	PFOA	0.32 – 6.28 ng/L	(Ericson et al., 2008)
	Public buildings of the Rhine-Ruhr area, Germany Berlin, Germany Muenster, Germany	2006	PFOA	<1 – 519 ng/L 2 ng/L 4 ng/L	(Skutlarek et al., 2006)
	26 waterworks along the Ruhr River, Germany	2008-2009	PFOA	Maximum: 83 ng/L Median: 23 ng/L	(Wilhelm et al., 2010)
	Area of Lake Maggiore, Italy	2007	PFOA	1 – 2.9 ng/L	(Loos et al., 2007)
	Osaka, Japan	2006-2007	PFOA	2.3 – 84 ng/L Detected in all tap water samples	(Takagi et al., 2008)
Wastewater treatment plant	nine WWTP along the River Elbe between Lauenburg and Cuxhaven, Germany	2007	PFOA	Effluent: 12.3±1.7 – 77.6±0.3 ng/L	(Ahrens et al., 2009a)
(WWTP)	Six WWTP (domestic, commercial, and industrial) in New York State, USA	2004/2005	PFOA	Effluent: 58 -1050 ng/L	(Sinclair and Kannan, 2006)
	Rural WWTP, Kentucky, USA Urban WWTP, Georgia, USA	2005	PFOA	Influent: 22 – 184 ng/L Effluent: 122 – 183 ng/L Final solid waste: 8.3 -219 ng/g dw Influent: 2 – 30 ng/L Effluent: 6.7 – 102 ng/L Sludge before burning: 64 – 130 ng/g dw Sludge after burning: 7.0 – 35 ng/g dw	(Loganathan et al., 2007)
	two municipal WWTP in Singapore Plant A:conventional activated sludge process line (CAS) in parallel with liquid treatment module (LTM) and membrane biological reactor (MBR) Plant B: conventional activated sludge process line	2006-2007	PFOA	Influent: 11.1±1.84 – 71.3±25.3 ng/L Effluent (CAS): 15.8±2.8 – 138.7±17.4 ng/L Effluent (LTM): 17.0±3.5 – 21.8±2.6 ng/L Effluent (MBR): 30.4±5.4 – 93.8±26.6 ng/L Digested sludge: 17.4±5.4 – 45.8±10.7 ng/g dw LTM sludge: 6.0±1.2 – 13.1±3.9 ng/g dw MBR sludge: 12.1±2.3 ng/g dw Influent: 36.6±5.4 – 531.7±87.7 ng/L Effluent: 77.4±13.7 – 1057.1±205.8 ng/L Digested sludge: 46.9±8.4 – 69.0±12.2 ng/g dw	(Yu et al., 2009a)
	WWTP of Bayreuth, Germany	2007	PFOA	River - 0.1 km upstream: <0.06 -2 ng/L Effluent: 20 – 3,900 ng/L River – 1 km downstream: 3.1 – 8 ng/L	(Becker et al., 2010)
	Eight WWTP located in Shanghai, China Waste activated sludge Chemical sludge Activated sludge of aeration tank	2008	PFOA	41.0 – 71.6 ng/g dw 75.5 ng /g dw 9.21 – 18.2 ng /g dw 42.3 ng /g dw	(Li et al., 2010)

	Primary sludge				
	Nine WWTP at different locations in Lagos, Oyo and Ogun state, all in South West Nigeria Domestic WWTP Industrial WWTP Hospital WWTP	2012	PFOA	 0.0189 – 0.0415 ng/g 0.0266 – 0.4163 ng/g 0.0812 ng/g	(Sindiku et al., 2013)
Sediment	North Sea (German Bight) Western Baltic Sea	2002-2005 2005	PFOA	0.079 – 0.157 µg/kg dw 0.061 – 0.684 µg/kg dw	(Theobald et al., 2012)
	Surficial sediments (top 1-5 cm) from the outlets of various rivers and creeks in the San Francisco Bay Area; additional: sediment from the Palo Alto Mudflats and Hayward, California; Baltimore, Maryland; Corvallils, Oregon	2002-2004	PFOA	< 0.011 – 0.625 ng/g dw	(Higgins et al., 2005)
	Ariake Sea, Japan (tidal flat)	2004	PFOA	0.84 – 1.1 ng/g dw	(Nakata et al., 2006)
	(top 0-2 cm) Zhujiang River, Guangzhou, China (13 sites) Huangpu River, Shanghai, China (9 sites)	2009	PFOA	 0.09 – 0.29 ng/g dw 0.20 – 0.64 ng/g dw	(Bao et al., 2010)
	Huangpu River, Shanghai, China Sujhou River, Shanghai, China (note: a PTFE manufacture plant is located in Yangtze River Delta)	2007	PFOA	5.20 – 203 ng/g dw 20.8 ng/g dw	(Li et al., 2010)
	Cornwallis Island, Nunavut, Canadian Arctic Amituk Lake, Char Lake, Resolute Lake,	2003,2005	PFOA	<0.29 – 7.5 ng/g dw	(Stock et al., 2007)
	Hong Kong	2009	6:2 diPAP	<0.017 – 0.080 ng/g dw	(Loi et al.,
			6:2/8:2 diPAP 8:2 diPAP PFOA	<0.017 ng/g dw <0.017 – 0.870 ng/g dw <0-017 – 0.163 ng/g dw	2013)
Soil	Top soil samples (0-10 cm) around manufacturing facility in Wuhan, Hubei province, China Former manufacturing facility (ceased production of PFASs since 2002); sampling near the plant and along the Yangtze River	2009	PFOA	Average: 50.1 ng/g dw PFOA detected in 17 of 32 soils PFOA <0.05 ng/g dw at sampling points > 2 km distance from plant <0.05 – 1.82 ng/g dw PFOA detected in >50% of soils	(Wang et al., 2010)
	Shanghai, China Agricultural areas Residential and industrial areas	2007	PFOA	 3.28 – 44.0 ng/g dw 42.3 – 47.5 ng/g dw	(Li et al., 2010)
	Soil samples (0-15 cm) from: United States Japan Mexico	-	PFOA	 1.35 – 31.7 ng/g dw 1.84 -21.5 ng/g dw 0.764 ng/g dw	(Strynar et al., 2012)
Ground water	164 individual ground water samples from 23 European Countries	2008	PFOA	Frequency of detection: 66% (LOD 0.4 ng/L) Maximum: 39 ng/L Median: 1 ng/L	(Loos et al., 2010)
	Ground water recharge area, located in the central part of The Netherlands (former landfill and a nearby military base/urban area)	2011	PFOA	<0.01 – 2,060 ng/L	(Eschauzier et al., 2013)
	Wurtsmith Air Force Base, Michigan, USA (decommissioned in 1993, fire-training area 1952-1993)	1998-1999	PFOA	3,000 – 105,000 ng/L	(Moody et al., 2003)
	Naval Air Station Fallon, Nevada, USA (1950s-1988 fire-training	after 7-10 years	PFOA	<18,000 – 6,570,000±150,00 ng/L	(Moody and Field, 1999)

	activities) Tyndall Air Force Base, Florida, USA (1980-1992 fire-training activities)	of inactivity		<18,000 – 116 ng/L	
	16 ground water and spring samples from 0 to 30 m below, Tokyo	2006	PFOA	0.47 – 60 ng/L	(Murakami et al., 2009)
	Bavaria (51sampling points) Gendorf (fluoropolymers manufacturing)	-	PFOA	<1 – 4.1 ng/L (n= 23 > LOD) 29 – 4300 ng/L	(Bayerisches Landesamt für Umwelt, 2010)
Atmosphere	Ship-based samples were taken on observation deck of different research vessels during several sampling campaigns along north-south and east west transects of the Atlantic and Southern Ocean as well as in coastal areas of the Baltic Sea; Northern Hemisphere Southern Hemisphere Longyearbyen – Kiel Bremerhaven – Cape Town Bremerhaven – Cape Town Cape Town – Neumayer Station – Cape Town Rostock – Tallinn – Kiel German Bight, North Sea Las Palmas – St. John`s Recife - Dakar	2007- 2008 2007- 2008 2007 2007 2008 2008- 2009 2008 2007 2007 2008	8:2 FTOH	27 pg/m³ (n=66) (gas phase) 0.5 pg/m³ (n=63) (particle-phase) 7.8 pg/m³ (n=39) (gas phase) 0.1 pg/m³ (n=34) (particle-phase) 10 - 50 pg/m³ (gas phase) 1.5 - 39 pg/m³ (gas phase) 3.4 - 8 pg/m³ (gas phase) 1.8 - 11 pg/m³ (gas phase) 7.0 - 94 pg/m³ (gas phase) 11 - 130 pg/m³ (gas phase) 6.8 - 124 pg/m³ (gas phase) 6.2 - 29 pg/m³ (gas phase)	(Dreyer et al., 2009)
	Northern Hemisphere Southern Hemisphere Longyearbyen – Kiel Bremerhaven – Cape Town Bremerhaven – Cape Town Cape Town – Neumayer Station – Cape Town Rostock – Tallinn – Kiel German Bight, North Sea Las Palmas – St. John`s Recife - Dakar	2007- 2008 2007- 2008 2007 2007 2008 2008- 2009 2008 2007 2007 2008	8:2 FTA	1.5 pg/m³ (n=66) (gas phase) 0.0 pg/m³ (n=63) (particle-phase) 0.4 pg/m³ (n=39) (gas phase) 0.0 pg/m³ (n=34) (particle-phase) n.d. – 5.2 pg/m³ (gas phase) n.d. – 3.5 pg/m³ (gas phase) n.d. n.d. – 0.2 pg/m³ (gas phase) n.d. – 3.7 pg/m³ (gas phase) 1.7 – 15 pg/m³ (gas phase) 0.1 – 15 pg/m³ (gas phase) n.d. – 3.6 pg/m³ (gas phase)	
	expedition of the icebreaker Oden on the first leg of a cruise from Gothenburg, Sweden to Barrow, Alaska, via the North Atlantic Ocean and Canadian Archipelago (58°47.5´-74°41.0´N) Toronto	2005 2006	8:2 FTOH	4.16 – 22.7 pg/m³ (gas phase) 1.07 – 8.37 pg/m³ (particle phase) 25.1 – 59.6 pg/m³ (gas phase) 0.30 – 1.31 pg/m³ (particle phase)	(Shoeib et al., 2006)
	Hamburg, Germany (urban site) Waldhof, Germany (rural site)	2005	8:2 FTOH	62 – 275 pg/m³ 33 – 112 pg/m³	(Jahnke et al., 2007)
	Ontario, Canada Air around a WWTP and two landfill sites	2009	8:2 FTOH	192 – 10,309 pg/m³ 223 – 17381 pg/m³	(Ahrens et al., 2011)
	Ontario, Canada Air around a WWTP and two landfill sites	2009	PFOA	2.99 – 47.3 pg/m³ <0.04 – 46.2 pg/m³	
	Air samples from Northwest Europe (UK, Ireland, Norway)	2005-2006	8:2 FTOH	11.3 – 102 pg/m³ (gas phase) <1.1 – 8.5 pg/m³ (particulate phase)	(Barber et al., 2007)
			PFOA	1.54 – 552 pg/m³ (particulate phase)	

	Indoor air and outdoor air in Canada Indoor (homes in Vancouver, Canada)	2007-2008	8:2 FTOH	660 – 16,080 pg/m³ Median: 2,720 pg/m³ 100% of samples > LOD (LOD = 14 pg/m³)	(Shoeib et al., 2011)
			PFOA	3.4 – 2,570 pg/m³ Median 21 pg/m³ 100% of samples > LOD (LOD = 0.47 pg/m³)	
	Outdoor (Vancouver, Canada)		8:2 FTOH	83 – 367 pg/m³ Median: 117 pg/m³ 100% of samples > LOD (LOD = 14 pg/m³)	
			PFOA	<0.47– 9.2 pg/m³ 67% of samples > LOD (LOD = 0.47 pg/m³)	
Dust	Microenvironments in Stockholm, Sweden: Houses (n=10) Apartments (n=38) Day care centres (n=10) offices (n=10) cars (n=5)	2006-2007	PFOA	 15 – 98 ng/g; median = 54 ng/g 17 – 850 ng/g; median = 93 ng/g 31 – 110 ng/g; median = 41 ng/g 14 – 510 ng/g; median = 70 ng/g 12 – 96 ng/g; median = 33 ng/g	(Bjorklund et al., 2009)
	Ohio and North Carolina, USA Homes (n=102) Day care centres (n=10)	2000-2001	PFOA	Maximum: 1960 ng/g Median: 142 ng/g 96.4% above LOQ (LOQ = 10.2 ng/g)	(Strynar and Lindstrom, 2008)
	Ohio and North Carolina, USA Homes (n=102) Day care centres (n=10)	2000-2001	8:2 FTOH	Maximum: 1660 ng/g Median: 32.9 ng/g 53.6% above LOQ (LOQ = 28.5 ng/g)	
	Homes, Japan (n=16)	-	PFOA	69 – 3,700 ng/g Median: 165 ng/g	(Moriwaki et al., 2003)
	Manufacturing facility (production of PFOA), Wuhan, Hubei province, China	2009	PFOA	 1100 and 2790 ng/g	(Wang et al., 2010)
	Office (n=2) Product storage (n=2) Raw material stock room (n=2)) Electrolysis workshop (n=3) Sulfonation workshop (n=3) Laboratory building Road (n=3)			1090 and 1200 ng/g <10 and 2780 ng/g 27,060 – 134,630 ng/g 15,990 – 160,00 ng/g 19,400 ng/g 160 – 1.810 ng/g	
	Homes, Vancouver, Canada (FTOH n=140; PFOA n=132)	2007-2008	8:2 FTOH	9.0 – 4,670 ng/g Median 63 ng/g 100% of samples > LOD (LOD = 0.19 ng/g)	(Shoeib et al., 2011)
			PFOA	1.9 – 1,390 ng/g Median 30 ng/g 100% of samples > LOD (LOD = 1.51 ng/g)	
	Residential indoor dust (n= 102; Vancouver, Canada)	2007-2008	8:2 diPAP	Maximum: 38,206 ng/g Median: 535 ng/g 99% above LOQ (LOQ = 12 ng/g)	(De Silva et al., 2012)
			8:2/10:2 diPAP	Maximum: 13,459 ng/g Median: 213 ng/g 99% above LOQ (LOQ = 12 ng/g)	
			6:2/8:2 diPAP	Maximum: 130,071 ng/g Median: 614 ng/g 100% above LOQ (LOQ = 9 ng/g)	
Biota	Pooled serum/plasma samples Svalbard reindeer, Svalbard, Norway Reindeer, East-Finmark, Norway Reindeer, West-Finmark, Norway Reindeer, Hardangervidda, Norway	 1996 2007 1993 2005 1993 2004 2000 2007 2009	PFOA	 0.3 ng/g ww 0.1 ng/g ww 0.1 ng/g ww 0.1 ng/g ww 0.03 ng/g ww 0.07 ng/g ww 0.2 ng/g ww 0.4 ng/g ww 0.02 ng/g ww	(Norwegian Pollution Control Authority, 2009)

	Reindeer, Sørreisa, Norway Reindeer, Hattfjelldal, Norway Red deer, Stranda, Norway Moose, Ringebu/Øyer, Norway	2002 2003 2004		0.02 ng/g ww 0.03 ng/g ww <0.1 ng/g ww	
	Polar bear liver, Ittoqqortoormiit, East Greenland	1984 1985 1986 1987 1988 1989 1990 1991 1992 1993 1994 1995 1996 1999 2000 2001 2003 2004 2006	PFOA	3.2 – 9.0 ng/g ww 5.0 – 6.2 ng/g ww 7.4 – 8.0 ng/g ww 3.4 – 7.8 ng/g ww 0.6 ng/g ww 0.6 – 12.1 ng/g ww 0.6 – 14 ng/g ww 4.0 – 7.6 ng/g ww 4.0 – 7.0 ng/g ww 0.6 – 14.2 ng/g ww 6.8 – 9.0 ng/g ww 6.8 – 15.8 ng/g ww 0.6 – 18.3 ng/g ww 0.6 – 18.2 ng/g ww 0.6 – 170.8 ng/g ww 0.6 – 36.4 ng/g ww 8.8 – 18.8 ng/g ww 5.6 – 11.5 ng/g ww 11.8 – 17.6 ng/g ww	(Dietz et al., 2008)
	Polar bear liver Chukchi Sea, Alaska, USA Northwest Territories, Canada South Baffin Island, Canada High Arctic, Canada South Hudson Bay, Canada East Greenland Svalbard, Norway	- - 2002 2002 2002 1999 - 2001 -	PFOA	<2.3 – 9.04 ng/g ww 10.2 – 33.3 ng/g ww 20 – 55.8 ng/g ww 8.64 – 31.8 ng/g ww 18.6 – 31.2 ng/g ww <2.3 – 57.1 ng/g ww 11.9 – 37.5 ng/g ww	(Smithwick et al., 2005)
	Plasma of Bottlenose Dolphins Sarasota Bay, Florida, USA	2003	PFOA	0.7 – 26 ng/g ww	(Houde et al., 2005)
	Bermuda Indian River Lagoon, Florida, USA Charleston, South Carolina, USA Delaware Bay, New Jersey, USA			0.6 – 0.9 ng/g ww 1- 70 ng/g ww 4.6 – 163 ng/g ww 20 – 115 ng/g ww	
	Offshore waters of South Carolina, Georgia and Florida Loggerhead sea turtle(plasma) Kemp`s ridley sea turtle(plasma)	2003	PFOA	0.493 – 814 ng/ml 2.77 – 4.25 ng/ml	(Keller et al., 2005)
	Cormorant liver, Cabras Lafoon, (Sardinian Sea, Italy)	1997	PFOA	29 – 450 ng/g ww	(Kannan et al., 2002)
	Cormorant eggs from the Baltic Sea, island Heuwiese, Germany Cormorant eggs from the Elbe estuary, site Haseldorf, Germany Rook eggs from Saarlouis, Germany	2009	PFOA	0.7 – 1.9 ng/g ww 0.5 – 3.7 ng/g ww <0.5 – 1.2 ng/g ww	(Rüdel et al., 2011)
	Herring gull eggs (15 colonies) in the Laurentian Great Lakes, North America	2007	PFOA	<0.1 – 2.6±0.4 ng/g ww	(Gebbink et al., 2009)
	Lake Trout collected from the Great Lakes, North America	2001	PFOA	0.61±0.07 – 6.8±2.7 ng&g ww	(Furdui et al., 2007)

2.3.2 Background information on Human Exposure

(For references, see draft risk profile)

71. A large number of studies have been conducted to determine human exposure to PFOA. Among other things, findings identify common routes to exposure and vulnerable groups to exposure as well as reporting concentration levels found in samples.

72. Human exposure typically takes place “human via environment”[3] by consumption of drinking water and food, via uptake of contaminated indoor dust or from consumer products containing PFOA and its related substances. (Post et al., 2009; Shoeib et al., 2015; Wilhelm et al., 201 ; Schwanz et al., 2015; D’Hollander et al., 2015; Eriksson et al., 2013; Carlsson et al., 2014; Yamada et al., 2014). Indoor air and dust are also important sources of human PFOA exposure (Haug et al., 2011; Gebbink et al., 2015). ECHA, 2013a also comments that PFOA is primarily taken in orally, through inhalation of dust, or to a lesser degree through dermal contact. PFOA then concentrates within the

[3] Indirect exposure of humans via the environment may occur by consumption of food (fish, crops, meat and milk) and drinking water as well as inhalation of air (ECHA, 2012).

blood, liver, kidney and lungs. In some cases, elevated serum PFOA levels can be largely attributed to exposure to PFOA-related substances such as 8:2 FTOH (Nilsson et al., 2010a, 2010b).

73. Several studies have suggested that PFOA levels increase with age (Haug et al., 2010, 2011; Christensen et al., 2016) and that breast feeding, diet and indoor environment are important factors for PFOA exposure that need to be addressed in the evaluation of human exposure and accumulation of PFOA (Haug et al., 2010, 2011; Brantsater et al. 2013; ECHA, 2013a).

74. Dietary intake is considered as the most important route of human exposure to PFOA based on studies from various countries (Vestergren et al., 2008; Haug et al., 2011; Thompson et al., 2011b; Cornelis et al., 2012, Shan et al., 2016). Haug et al. (2011) assessed the relative importance of different exposure pathways of PFCs in a group of Norwegians and compared estimated intakes with internal doses obtained through biomonitoring. It was observed that food was a major PFOA exposure source, representing 67-84% of the median total intake for PFOA using different dust ingestion rates and biotransformation factors of 'precursor' compounds. Next, in an Australian study by Thompson et al. (2011b), PFOA was one of the most commonly detected perfluorinated alkyl acids in potable water, quantifiable in 40% of all samples with a maximum PFOA concentration of 9.7 ng/ L. It was reported that the daily water intake, along with an average for PFOA, was of 2.2% of estimated daily intake, up to 24% in some locations. Cornelis et al. (2012) calculated intake of PFOA by the population of Flanders, Belgium, including children and adults, from food, drinking-water, air, settled dust and soil and observed that dietary exposure dominated overall PFOA intake. For adults, average dietary intake equaled 6.1 ng/PFOA kg per day. For children, the dietary intake was about 3 times higher. Predicted intake is high when compared to assessments in other countries and to serum levels from Flanders but comparable to the intakes published by the European Food Safety Authority (EFSA) in 2008. Lastly, it was reported that linear (n-) PFOA was the predominant isomer found in fish (92.2%) and meat (99.6%) samples collected in Tianjin, China, and dietary intake contributed >99% of the estimated daily intake for the general Chinese population (Shan et al., 2016). There is also evidence of human exposure via dust in domestic settings (Eriksson and Kärrman, 2015; Tian et al., 2016).

75. The issue of human exposure via consumer products is also examined by Vestergren et al. (2015), who modelled emissions from consumer products imported from China and finds that 1.5% of PFOA discharged to wastewater and 0.3% of 8:2 FTOH emissions to air can be attributed to these imported products.

76. Based on the existing studies of serum/plasma PFOA concentrations in the general European adult population and in children worldwide (see Tables 10 and 11 below), the average median (and maximum) PFOA serum levels were calculated to be 3.5 (21) ng/mL and 6.4 (108) ng/mL for European adults and children, respectively (ECHA, 2015a). It should be mentioned that for the calculation of average median (and maximum) PFOA serum levels reported in ECHA, 2015a, one outlying value from the study by Mondal et al., (2012) was included in the calculation. By removing this outlying value, the average median (and maximum) PFOA serum levels in children is 2.52 ng/mL (15.9 ng/mL).

Table 10: List of studies reporting serum/plasma concentrations of PFOA (ng/mL) in the general European adult population, extracted from ECHA (2015a)

Location	Serum/plasma concentrations, ng/mL			Number of samples	Year	Back-calculated intake, ng/kg bw/day			Reference
	Median	Min	Max			Median	Min	Max	
Belgium	4.1	1.1	12.8	20	1998	0.47	0.13	1.5	Kannan et al. (2004)
Poland		9.7	40	25	2003		1.1	4.6	Kannan et al. (2004)
Spain	3.4	1.6	6.2	48	2006	0.39	0.18	0.72	Ericson et al. (2007)
Germany		0.7	100	521	2006		0.08	12	Höltzer et al (2008)
Germany	6.8	1.7	39.3	105	2006	0.79	0.2	4.5	Midasch et al. (2006)
Germany	5.7	0.5	19.1	356	2006	0.66	0.06	2.2	Fromme et al. (2007)
Norway	2.2			950	2003-2004	0.25			Whitworth et al. (2012)
Norway	3.6	0.5	13	175	2003	0.42	0.06	1.5	Haug et al. (2010b)
Norway	1.4	0.28	22	41	2007-2008	0.16	0.03	2.5	Haug et al. (2011a)
Denmark	3.7	0.1	19.8	665	1988-1989	0.43	0.014	2.3	Halldorsson et al. (2012)
Denmark	5.6			222	1992-1995	0.65			Vestergaard et al. (2012)
Denmark		<LOQ	41.5	1400	1996-2002			4.8	Fei et al. (2007)
The Faroe Islands	3.2			656	1999-2001	0.37			Grandjean et al. (2012)
Sweden*	5	1	24.8	66	1997-2000	0.58	0.12	2.9	Kärrman et al. (2006)
Belgium*	2.3			8 pools	2002–2005	0.27			Cornelis et al. (2012)
Belgium*	3.6			200	2008-2009	0.42			Cornelis et al. (2012)
Sweden*	2.1			3 pools with 10 in each pool	2008	0.24			Glynn et al. (2012)
Sweden*	1.9			3 pools with 10 in each pool	2009	0.22			Glynn et al. (2012)
Sweden*	1.7			3 pools with 10 in each pool	2010	0.19			Glynn et al. (2012)
Germany	4.1	2.3	6.7	20	2008	0.47	0.27	0.8	Schröter-Kermani et al. (2013)
Germany	3.2	0.8	8.7	18	2010	0.37	0.09	1	Schröter-Kermani et al. (2013)

*:mean

Table 11: List of studies reporting serum/plasma concentrations of PFOA (ng/mL) in the children worldwide, extracted from ECHA (2015a)

Location	Serum/plasma concentrations, ng/mL			Age	Number of samples	Sampling year	Comments	Reference
	Median	Min	Max					
Korea	1.94	1.68	2.46	12-19	77	2009		Ji et al. (2012)
China	1.7	0.35	11	0-1	14	2009		Zhang et al (2010b)
China	2.42	0.36	15.2	1-5	85	2009		Zhang et al (2010b)
China	2.19	0.3	6.37	5-10	85	2009		Zhang et al (2010b)
China	1.23	<0.56	3.22	10-18	19	2009		Zhang et al (2010b)
Canada	1.6*	0.4	11	0,9 - 4,5	86	2006-2008	* geometric mean	Turgeon et al. (2012)
Texas, USA	2		9.6	0-3	75	2009		Schecter et al. (2012)
Texas, USA	3.1		11	3-6	75	2009		Schecter et al. (2012)
Texas, USA	3		10.7	6-9	75	2009		Schecter et al. (2012)
Texas, USA	3		13.5	9-13	75	2009		Schecter et al. (2012)
New York, USA	3.28	0.43	5.87	9-11	83	2008-2009		Gump et al. (2011)
USA	4.4	0.4	21.7	12-15	571	1999-2004		Hoffman et al. (2010)
Ohio, USA	68.4*	0.7	1283	1-19	4943	2005-2006	* arithmetic mean	Mondal et al. (2012)
Greenland	4.06	3.33 *	4.96 *	5	456	2002-2005	* inter quartile range	Grandjean et al. (2012)
Norway	1.6			0-1	1 pool of >10 individuals	2007		Haug et al. (2009)
Norway	2.6			1-4	1 pool of >10 individuals	2007		Haug et al. (2009)
Norway	2.2			5-14	1 pool of >10 individuals	2007		Haug et al. (2009)
Germany		2	96	5-6	170	2006		Höltzer et al. 2008

77. A number of studies have also been conducted to illustrate the presence of PFOA within human populations. Based on analysis of existing studies of PFOA concentrations in the European populations, blood serum and plasma concentrations of PFOA ranged from <0.5 – 40 µg/L (Fromme et al., 2009; Vestergren and Cousins, 2009).

78. PFOA concentrations of the American general population have been measured and reported in various studies and surveys (CDC, 2015; Christensen et al., 2016). For example, the US National Health and Nutrition Examination Survey (NHANES) is a nationally representative survey of the general American population. For the survey years 2011-2012, the geometric mean of the PFOA serum concentrations within the general American population was 2.08 µg/L (95% confidence interval 1.95–2.22 µg/L) (CDC, 2015). In addition, a US study of male anglers aged 50 and older found PFOA serum levels (median of 2.50 µg/L) similar to those of the US general

population (median of 2.62 µg/L; value extracted from the 2011-2012 NHANES), but increasing age as well as alcohol consumption were observed to be predictors of PFOA levels (Christensen et al., 2016).

79. The Canadian Health Measures Survey (CHMS) is a nationally representative survey of the general Canadian population, and biomonitoring data from Cycle 2 of the CHMS (2009-2011) found that the general population (aged 12-79 years) had blood plasma concentrations of PFOA with a geometric mean of 2.3 µg/L (95% confidence interval 2.1-2.5 µg/L). PFOA will be measured again in future cycles of the CHMS (Health Canada, 2013). Children in Nunavik, Northern Quebec with a mean age of 11.3 years were found to have whole blood concentrations ranging from 1.1 to 12 µg/L (geometric mean 2.5 µg/L). PFOA has also been detected in adult Canadians (aged 20 years or older) living on reserve and people throughout the circumpolar Arctic (Wuttke et al., 2013; AMAP, 2015). In some countries such as Iceland and Norway, PFOA levels within the sampled populations were among the highest compared to other POPs such as *p,p'*-DDE, PCB153, *trans*-nonachlor (AMAP, 2015).

80. Serum PFOA levels measured in the Australian population in 2010-2011 ranged from 3.1 to 6.5ng/mL, but a comparison of serum levels in Australian adults between 2002/03-2010/11 found that human PFOA concentrations in sera decreased over time (Toms et al., 2014). Also, a study analysing the time trends of perfluorinated alkyl acids in serum from Danish pregnant women (2008 – 2013) indicated that PFOA levels have been decreasing on average with 9.1% (Bjerregaard-Olesen et al., 2016).

81. There are several reports showing gender-related differences in PFOA concentrations in countries such as South Korea and Poland (Cho et al., 2015; Góralczyk et al., 2015). However, in another US study by Emmett et al. (2006), no correlation between gender and PFOA concentration level was observed.

82. There are a few reports of detected PFOA serum levels in the occupationally exposed workers (Guruge et al., 2005; Fromme et al., 2009; Freberg et al., 2010; Gao et al., 2015; Nilsson et al., 2010b). Indoor dust and total suspended particles seem to be important occupational exposure routes in fluorochemical manufacturing and is also considered relevant in domestic settings (Gao et al., 2015; Eriksson and Kärrman, 2015; Tian et al., 2016).

83. A study that examined serum and urine samples from 36 occupational workers in a fluorochemical manufacturing plant in China (2008-2012) showed that the most important exposure routes to PFAA isomers (PFOS, PFOA, and PFHxS) in the occupational workers were considered to be the intake of indoor dust and total suspended particles. This study reported a geometric mean concentration of 371 ng/mL PFOA in serum (range 2.66–14774 ng/mL) (Gao et al., 2015). With view on developing countries, a study conducted in Sri Lanka (Guruge et al., 2005) showed that PFOA levels in human blood serum and seminal plasma from 30 volunteers of both urban and rural regions (tea workers) were 6.38 ± 6.84 ng/mL (blood serum) and 0.323 ± 0.513 ng/mL (seminal plasma), respectively. Workers from an organic tea farm showed lower serum concentrations of PFOA than workers from a conventional tea farm or from an urban area. Recent studies from Sweden and Norway reported significantly elevated PFOA levels found in workers and technicians using fluorinated ski wax (Nilsson et al., 2010b; Freberg et al., 2010). In the Swedish study, the PFOA levels in three technicians with "low" initial levels of PFOA increased from pre-season to post-season by 254, 134, and 120 %, whereas no increases in the blood levels were observed for the five technicians with "high" initial levels (Nilsson et al., 2010b). In the Norwegian study, serum samples from 13 professional male ski waxers were collected at three occasions. A statistically significant positive association between years exposed as a ski waxer and concentration of PFOA in serum was observed (Freberg et al., 2010). The results also indicate long elimination half-lives of PFOA in humans.

84. Persons living in the vicinity of fluorochemical manufacturing plants have higher PFOA levels than the general population (Emmett et al., 2006; Oliaei et al., 2013; Hoffman et al., 2011). Emmett et al. (2006) conducted a study on populations living in Ohio, USA in the vicinity of a fluoropolymer production facility with detected median PFOA concentrations of 453 ng/mL (interquartile range 176–568 ng/mL). A survey of 196 people living near a 3M PFC manufacturing facility found PFOA in all of them at 15.4 ng/ml (geometric mean) – about four times the mean US population level (Oliaei et al., 2013). A survey of 108 people living near a DuPont fluoropolymer manufacturing facility who use private wells found a mean serum PFOA concentration of 75.7 µg/L – approximately 20 times higher than the general US population. Each 1 µg/L increase in PFOA levels in drinking water was associated with an increase of 141.5 µg/L in serum (Hoffman et al., 2011).

85. Based on this, Table 12 gives an overview on a number of reported PFOA concentrations in humans.

Table 12: PFOA concentrations in humans according to selected information sources

Country or Region	Population	Matrix	Concentration(s) [µg/L]	Reference(s)
EU	EU population	Serum and plasma	Range: <0.5 – 40	Fromme et al., 2009
Germany	Young adults	Blood	Range: 0.5 – 19	Fromme at al., 2009
U.S.	Populations living in the vicinity of production facilities	Serum	IQR: 176 – 568 Mean: 453	Emmett et al., 2006

Country or Region	Population	Matrix	Concentration(s) [µg/L]	Reference(s)
U.S.	General population (2011 – 2012)	Serum	GM: 2.08 Median: 2.62	CDC, 2015 Christensen et al., 2016
Canada	Canadian adults (≥20 years) living on reserve	Plasma	GM: 1.39	Wuttke et al., 2013
Canada	Children from Nunavik, Northern Quebec	Whole blood	Range: 1.1 – 12 GM: 2.5	AMAP, 2015
Canada	General population from 2009-2011	Plasma	GM :2.3	Health Canada, 2013
China	Workers in a fluorochemical manufacturing plant	Serum	Range: 2.66 – 14774 GM: 371	Gao et al., 2015
Sri Lanka	Tea workers	Serum	Range: 0.23 – 23.5 Mean: 6.38	Guruge et al., 2005
Sweden	Ski waxing technicians	Whole blood	Range: 4.8 – 535	Nilsson et al., 2010b
Norway	Professional ski waxers	Serum	Range: 20 – 174 Median: 50	Freberg et al., 2010

IQR: Interquartile range; GM: geometric mean

86. Mothers excrete PFOA via breast milk, which is considered an important source of exposure to breast-fed infants, whose PFOA exposure level is considerably higher than adults (Haug et al., 2011). PFOA is also transferred to the foetus through the placenta, and it has been reported that total branched PFOA isomers cross the placenta more efficiently than linear isomers (Beesoon et al., 2011). Time interval between pregnancies has been shown to be strongly correlated with increased concentrations of PFOA in pregnant women, possibly reflecting the re-accumulation of PFOA in maternal blood with increasing time between pregnancies (Brantsæter et al., 2013).

87. Concentrations of PFOA in Australian women of child-bearing age are almost twice those found in pregnant women from Germany and PFOS and PFOA concentrations are 1.5 and twice those found in adults from the USA (Toms et al., 2014).

88. A study of 4943 mother —child pairs found that children up to about 12 years of age had higher serum PFOA concentrations than their mothers with the highest child/mother ratios found in children less than 5 years-old (44% higher than their mothers) (Mondal et al., 2012). The authors attribute the differences to *in utero* exposure and exposure from breast milk and drinking water such that the longer the women nursed their babies, the higher the levels of certain chemicals, including PFOA, detected in their babies' blood. Breastfeeding is an important exposure pathway to PFOA as well as trans-placental passage (Umweltbundesamt AT, 2012, 2013; Carious et al., 2015; Forns et al., 2015; Papadopoulou et al., 2015; Hanssen et al., 2013; Mogensen et al., 2015). Also, in Norway, there was a study conducted examining PFAS concentrations including PFOA in blood plasma of pregnant women (Brantstæter et al., 2013). Median value of PFOA in blood plasma of pregnant women under 25 years were 2.70 ng/mL, and pregnant women older than 35 years had a median value of 1.93 ng/mL. The authors examined different influencing factors on PFOA blood plasma concentrations and revealed that women that had babies before and breastfed them was the most prominent influencing factor.

89. Seventy-nine milk samples were collected from breastfeeding women and 25 samples from local fresh cow milk in northern Jordan. Among the findings the study found that mean concentrations of PFOA were significantly higher in milk samples provided by older women. Concentrations were also found to be higher in samples from women who had previously breastfed compared to those who had not breastfed before; although there was considerable variation in the range of results (Al-sheyab et al., 2015). Another study conducted in Spain on perfluorinated carboxylic acids in human breast milk observed that there is a greater transfer of PFC during breastfeeding by primiparous and thus a higher exposure to these contaminants for the first child (Motas Guzmàn et al., 2016). Same conclusion derived from a survey conducted in Italy when PFOA resulted higher in milk samples provided by primiparous women, suggesting that the risk of intake might be higher for first-borns (Barbarossa et al., 2013).

90. A recent study reported the detection of PFOA in 27 out of 30 human hair samples with a PFOA concentration range of 25-74 pg/g hair and an average of 46 pg/g hair (Alves et al., 2015).

91. Humans are very slow eliminators of PFOA compared with other species such as rodents, pigs and monkeys (Olsen et al., 2007; Numata et al., 2014) with a half-life ranging between 2 and 4 years (ECHA, 2015a; Olsen et al., 2007; Russell et al., 2015). The elimination half-life of PFOA was for the first time studied in 26 retired

fluorochemical production workers who had high initial serum concentrations (Olsen et al., 2007). Depuration followed a first-order kinetic, and geometric means of half-lives were 3.5 years. The half-life range for PFOA found in highly exposed workers was later confirmed in studies of general populations from Germany and the US exposed to PFOA through contaminated drinking water. The median half-life was found to be 2.3 years (Bartell al., 2010).

2.4 Hazard assessment for endpoints of concern

Background information on adverse effects on aquatic organisms

(For references, see draft risk profile)

92. According to several information sources, data currently available for PFOA indicate some adverse effects on a number of aquatic organisms. Generally, acute aquatic toxicity is low in standard ecotoxicity tests (NICNAS, 2015a); moderate to low acute toxicities are seen in pelagic organisms, including fish, low chronic toxicities in benthic organisms (Environment Canada and Health Canada, 2012). Adverse effects include intergenerational toxicity in the first offspring generation (Ji et al., 2008) and some PFOA-mediated toxicity in freshwater algae (Elnabarawy et al., 1981; Ward et al., 1995a, 1995b, 1996a, 1996b, 1996c; Boudreau, 2002; Thompson et al., 2004 as cited in Environment Canada and Health Canada, 2012; Latała et al., 2009) and other aquatic organisms (3M Company 1987a, 1990a, 1996a, b, c; Beach 1995a cited in Environment and Health Canada 2012). Further, PFOA-mediated effects on fish development, particularly in reproduction, have been observed. Studies with other aquatic organisms such as freshwater male tilapia, marine mussels and Baikal seals showed estrogenic effects, hepatotoxicity, inflammation, and chemosensitivity. Field studies related to effects of PFOA on immune function and clinical blood parameters in dolphins and sea turtles revealed increases in indicators of inflammation and immunity (Peden-Adams et al., 2004a, 2004b). Increased pro-inflammatory responses in male Japanese medaka were also observed (Yang, 2010). Activation of peroxisome proliferator–activated receptor α were shown in Baikal seals (Ishibashi et al., 2008). Evidence indicates that PFOA could exacerbate the adverse effects triggered by certain types of pesticides (Rodea-Palomares et al., 2015).

93. According to the assessment of the Australian Government (NICNAS, 2015a) data currently available for PFOA indicate low acute aquatic toxicity (median lethal/effective concentration values > 300 mg/L) and chronic aquatic toxicity (no-observed-effect concentration values ≥ 12.5 mg/L) in standard ecotoxicity tests. However, there is existing data that demonstrates the intergenerational toxicity of PFOA in the first offspring generation (F1) fish when both the parent and offspring fish are exposed to concentrations as low as 0.1 mg/L (Ji et al., 2008). The assessment notes that direct precursors to PFOA have been classified as Chronic Aquatic Category 1 (H410) under the United Nations' Globally Harmonised System of Classification and Labelling of Chemicals (GHS), but concludes that insufficient data are presented in this assessment to classify the aquatic hazards of chemicals in this group according to the third edition of the GHS (NICNAS, 2015a).

94. According to Environment Canada and Health Canada (2012), PFOA exhibits moderate to low acute toxicities in pelagic organisms, including fish (70–2470 mg/L) in traditional toxicity studies. PFOA exhibits low chronic toxicities in benthic organisms (>100 mg/L).

95. The most sensitive pelagic organism to PFOA-mediated toxicity was found to be the freshwater algae (*Pseudokirchneriella subcapitata*). Several toxicity tests on the freshwater alga *Pseudokirchneriella subcapitata* (Elnabarawy et al., 1981; Ward et al., 1995a, 1995b, 1996a, 1996b, 1996c; Boudreau, 2002; Thompson et al., 2004 as cited in Environment Canada and Health Canada, 2012; Latała et al., 2009) determined 96-hour median effective concentration (EC_{50}) values were determined to range from 4.9 to >3330 mg/L. The 96-hour no-observed-effect concentrations (NOECs) from 1.0 to 500 mg/L. Similarly, the 96-hour LOECs ranged from 2.0 to 1000 mg/L.

96. PFOA-mediated toxicity has also been observed in other aquatic organisms such as bacteria (*Photobacterium phosphoreum*) (3M Company 1987a, 1990a, 1996a, b, c; Beach 1995a cited in Environment and Health Canada 2012), water flea (*Daphnia magna*) (3M Company 1982, 1984, 1987; Ward and Boeri 1990; Ward et al. 1995c, 1996e, f, g; Boudreau 2002; CIT 2003; Li 2008; Kim et al. 2009 cited in Environment and Health Canada 2012), freshwater planarian, freshwater snail, green neon shrimp (Li, 2008), fathead minnow (3M Company 1977, 1985a, 1987c; Elnabarawy et al., 1980; Ward et al. 1995a, c, 1996b, f, iI, j cited in Environment and Health Canada 2012), bluegill (3M Company, 1978a, b) and aquatic macrophytes (*Myriophyllum sibiricum* and *Myriophyllum spicatum*) (Hanson et al., 2005).

97. In a study by MacDonald et al. (2004) on benthic invertebrates (*Chironomus tentans*), no toxicity was observed at any of the concentrations tested, and the 10-day NOEC value was determined to be 100 mg/L.

98. PFOA-mediated effects on fish development, particularly in reproduction, have been observed. For example, a PFOA kinetic study using male and female zebrafish (*Danio rerio*) continuously exposed to 10 µg/L for 40 days indicated that PFOA can accumulate in bile and intestines, suggestive of enterohepatic circulation of PFOA as well as in maturing vitellogenic oocytes, which may present potential adverse effects on embryonic development and offspring health of fish (Ulhaq et al. 2015). Iin female and male rare minnows (*Gobiocypris rarus*), exposure to 3–30 mg/L PFOA for 28 days elicited inhibition of the thyroid hormone biosynthesis genes, induced vitellogenin

expression in males, developed oocytes in the testes of male fish and caused ovary degeneration in females (Wei et al., 2007; Wei et al., 2008).

99. Studies with other aquatic organisms such as freshwater male tilapia, marine mussels and Baikal seals showed estrogenic effects, hepatotoxicity, inflammation, and chemosensitivity (Liu et al., 2007; Stevenson et al., 2006; Ishibashi et al., 2008).

100. Field studies related to effects of PFOA on immune function and clinical blood parameters in bottlenose dolphins and sea turtles from Florida, Georgia and South Carolina revealed increases in indicators of inflammation and immunity (Peden-Adams et al., 2004a, 2004b). Yang (2010) also observed increased pro-inflammatory responses in male Japanese medaka (*Oryzias latipes*) exposed to 10-100 mg/L for 7 days.

101. Rodea-Palomares et al. (2015) examined the synergistic effects of PFOA on aquatic plant life (*cyanobacterium*) in combination with certain types of pesticides. The results demonstrated that pre-exposure to 5 mg/L PFOA, a concentration below their no observed effect concentration (NOEC), for 72 hours affected the uptake rates and the toxicity of specific pesticides such as 2,4-D, diuron and paraquat on aquatic plant life. Among the tested pesticides, paraquat demonstrated the strongest toxicity effect such that its toxicity on aquatic plant life doubled with PFOA pre-exposure. This suggests that PFOA could exacerbate the adverse effects triggered by certain types of pesticides.

Background information on adverse effects on terrestrial organisms

(For references, see draft risk profile)

102. An extensive review of available evidence for impacts on terrestrial organisms is contained in the REACH background document for the restriction proposal for PFOA (ECHA, 2015a).

103. The effects of repeated oral exposure to PFOA have been evaluated in mice (Loveless et al., 2006; Christopher and Marisa, 1977; Griffith and Long, 1980; Lau et al., 2006; Macon et al., 2011; Abbott et al., 2007; Wolf et al., 2007), rats (Metrick and Marisa, 1977; Griffith and Long, 1980; Goldenthal, 1978; Palazzolo, 1993; Biegel et al., 2001; Perkins et al., 2004) and monkeys (Goldenthal, 1978b; Griffith and Long, 1980; Thomford, 2001; Butenhoff et al., 2002). The key adverse effect observed from repeated dose animal studies is alterations to the liver, primarily increased liver weight and hepatocellular hypertrophy. Mortality was observed at high doses.

104. At lower doses, reduced body weight and increased kidney and liver weight were noted. Hepatocellular hypertrophy, degeneration and/or focal to multifocal necrosis were reported with increased severity at doses between 1.5 to 15 mg/kg bw/day in rats and mice. Hepatocellular hypertrophy was observed in all species. Increased liver weight and hepatocellular hypertrophy was also observed at 0.64 mg/kg bw/day in rats (Perkins et al., 2004; ECHA, 2015a).

105. According to the dossier, the NOAEL and LOAEL for increased liver weight and hepatocellular hypertrophy based on subchronic toxicity studies in rats are 0.056 mg/kg bw/day and 0.64 mg/kg bw/day, respectively (ECHA, 2015a; Perkins et al., 2004). The LOAEL of maternal toxicity was 1 mg/kg bw/day based on increased liver weight. (Borg and Håkansson, 2012 cited in ECHA 2015a). White et al. (2009) identified a LOAEL of 1.0 mg/kg bw/day for delayed mammary gland development in F1 (the first offspring generation). Abbott et al. (2007) calculated a NOAEL of 0.3 mg/kg bw/day for neonatal survival based on developmental exposure of mice. It should be mentioned that the NOAELs and LOAELs were identified from critical and relevant studies for the observed effects.

106. Toxicological studies in rats have shown that PFOA reduces serum lipids while it increases hepatic triglycerides, probably through the activation of PPARα (Haugom and Spydevold, 1992; Bjork et al., 2011). A study by Butenhoff et al. reported a dose-dependent increase in serum triglycerides in monkeys (exposed orally to 3-30 mg/kg bw PFOA daily for 6 months) and only a moderate and non-significant reduction in cholesterol with increasing PFOA (Butenhoff et al., 2002).

107. The induction of tumours has been demonstrated in rats exposed to PFOA. Rats exposed to 300 ppm PFOA via dietary intake up to 21 months resulted in increased incidences of liver adenomas, Leydig cell hyperplasia/adenomas and pancreatic acinar cell tumours (PACT) in male Sprague-Dawley rats (Biegel et al., 2001). Mammary fibroadenoma in the female rats was observed in another chronic PFOA exposure study although this observation has since been disputed after an independent group of pathologists (Pathology Working Group) re-examined the tissue from this study and reached a consensus that incidence of mammary gland neoplasms were not affected by chronic PFOA exposure (Hardisty et al., 2010; Butenhoff et al., 2012).

108. Following oral dosing of PFOA ammonium salt (APFO), increased liver weight in mice and altered lipid parameters in rats were observed in short-term (14-day) toxicity studies; increased liver weight was noted in a 26-week toxicity study in monkeys; and increased liver weight in dams, alterations in foetal ossification and early puberty in male pups were found in a developmental toxicity study in mice. In 2-year carcinogenicity bioassays in rats, males administered a high dose of APFO in the diet had significantly higher incidences of adenomas of the liver hepatocytes, Leydig cells in the testes and pancreatic acinar cells (ECHA 2015a).

109. Long-term toxicity studies with cynomolgus monkey showed reversible liver effects and relative liver weight increases at lowest-observed-adverse-effect level (LOAEL) of 3 mg/kg bw/day (Butenhoff et al., 2002).

110. The National Institute of Environmental Health Sciences (US) recently reviewed the evidence for the effects of PFOA on foetal growth in animals (Koustas et al., 2014). The authors concluded that there is sufficient evidence of decreased foetal growth in non-human mammalian species.

111. Animal studies show that PFOA increases the incidence of complete litter loss, postnatal mortality, decreases foetal body weight, delays ossification, changes mammary gland development and delays maturation in several developmental studies in mice (and some in rat) depending on strain, dose, time and length of exposure (Lau et al., 2006; Abbott et al., 2007; Macon et al., 2011; White et al., 2007, 2009, 2011; Wolf et al., 2007; Yang et al., 2009; Zhao et al., 2012b; Dixon et al., 2012, Suh et al., 2011; Albrecht et al., 2013). The LOAEL of maternal toxicity was determined as 1 mg/kg bw/day, and the NOAEL for neonatal survival was 0.3 mg/kg bw/day (Lau et al., 2006; Abbott et al., 2007). Some of observed PFOA-induced developmental/reproductive effects might be mediated by the peroxisome proliferator-activated receptor α (PPARα) (Zhao et al., 2012b; Albrecht et al., 2013). However, it has been mentioned that PFOA-induced alterations in mammary gland development might be dependent on steroid production in ovaries and independent of PPARα (Zhao et al., 2010).

112. According to Abbott et al. (2007), PFOA increased relative liver weight in adult female mice and weaned pups starting from a lowest dose of 0.1 mg/kg bw/day in WT pups or 1 mg/kg bw/day in WT adult females. White et al. (2009) showed increase in liver weight in both female and male offspring at doses of 0.3 mg/kg bw/day after full gestational exposure (GD 1-17).

113. Reduced neonatal survival started at the 0.6 mg/kg dose, and reduced pup weight as well as delayed eye opening above 1 mg/kg (Abbott et al., 2007). A meta-analysis of animal studies concluded that increasing concentrations of PFOA were associated with a decrease in mean pup birth weight (Koustas et al., 2014).

114. Lau et al. (2006) showed marked delay in ossification and earlier onset of sexual maturation of males as the most sensitive developmental impact of PFOA on mice (1 mg/kg bw/day). Other observed impacts comprise significant decrease in foetal body weight between 1 (Abbott at al.; 2007) and 3 mg/kg bw/day (Lau et al., 2006). An increase in litter resorption, neonatal mortality as well as a delay in eye opening was observed at 5 mg/kg bw/day (Lau et al. 2006; Abbott et al., 2007).

115. There are several studies suggesting that PFOA may alter steroid hormone production or act indirectly, via ovarian effects, as a mean of endocrine disruption (Zhao et al., 2010, 2012b; York, 2002; Butenhoff et al., 2004; Suh et al., 2011). For example, female mice exposed to 5 mg/kg bw/day of PFOA for 4 weeks exhibited enhanced mammary gland development, possibly due to increased steroid hormone production in the ovaries (Zhao et al., 2010). In addition, PFOA has also been reported to alter sexual maturation and pubertal timing in female and male offspring of rats and in multiple strains of mice (York, 2002; Butenhoff et al., 2004, Yang et al., 2009; Dixon et al. 2012), indicating a disruption of the normal steroid hormone regulation. While BALB/c and C57BL/6 female mice exposed to a daily dose of 1, 5 or 10 mg/kg bw of PFOA for 5 days/week both exhibited delayed vaginal opening, a difference in mammary gland and uterine development was observed between these two mouse strains (Yang et al., 2009). Also, increased uterine weight and histopathological changes of uterus, cervix and vagina of immature CD-1 mice exposed to low doses of PFOA (0.01 mg/kg bw/day) for 3 consecutive days was reported (Dixon et al., 2012). Lastly, NOAELs of 30 mg/kg for reproductive function of parental and first (F_1)-generation rats and 10 mg/kg for F_1-generation sexual maturation were reported in a repeated APFO exposure study (Butenhoff et al., 2004).

116. Effects of gestational exposure to PFOA on the development of the mammary gland from lactating dams and female pups at 5 mg/kg bw/day dose have been demonstrated (White et al., 2007, 2009). PFOA, when exposed in a critical window of susceptibility (GD 10-17), has been shown to induce changes in offspring mammary gland development in CD-1 mice at a dose of 0.01 mg/kg bw/day (Macon et al., 2011). In addition, chronic low dose exposure of PFOA (5 ppb in drinking water across two generations) reduced mammary gland development in F_1 as well as F_2 (Yang et al. 2009). Zhao et al. (2010) observed that PFOA stimulates mammary gland development in C57BL/6 mice.

117. Yang et al. (2009) studied the effects of peripubertal exposure to PFOA and observed hepatocellular hypertrophy (at 1 mg/kg bw/day) and delayed vaginal opening (at 5 mg/kg bw/day) in different mouse strains.

118. There are some indications of immune effects mediated by PFOA. Short-term dietary exposure to PFOA (i.e. up to 30 mg/kg bw/day via food or drinking water for 10 days) resulted in a reduction in thymus weight, decreased number of thymocytes and splenocytes and suppressed IgM antibody synthesis in C57BL/6 mice (DeWitt et al., 2008; DeWitt et al., 2009; Qazi et al., 2009). Adult offspring exposed to 2 mg/kg of PFOA given to dams from gestation through lactation exhibited altered immune responses such as reduced splenic T cells and IL-10 production from these cells (Hu et al., 2012). PFOA was also shown to increase histamine release from mast cells as well as exacerbate the IgE-dependent local allergic reaction in mice (Singh et al., 2012). Lastly, suppressed T-cell-mediated immunity in Japanese quails was observed after exposure to 10 ppm PFOA in drinking water for 8 weeks (Smits and Nain, 2013).

119. PFOA has been shown to modify gene expression in rodents. For example, gene expression profiling of livers from rats exposed to 10 mg PFOA/kg bw/day for 21 days indicated alterations in genes involved in multiple cell mechanisms such as transport and metabolism of lipids (particularly fatty acids), cell communication, adhesion, growth, apoptosis, hormone regulatory pathways, proteolysis/peptidolysis and signal transduction (Guruge et al., 2006). Also, inhibition on the expression of the placental prolactin-family hormone genes was observed in the placenta of mice exposed to 2, 10 and 25 mg/kg bw/day of PFOA at gestation days 11-16 (Suh et al., 2011).

120. The authors of the background document for the restriction of PFOA (ECHA, 2015a) emphasize the importance of assessing mice studies instead of rat studies as basis for DNEL-setting when this is based on animal studies due to the longer half-life of PFOA in mice compared to rats.

121. Studies with chicken showed no significant outcomes such as alterations in embryonic pipping success or in biochemical parameters at concentrations up to 10 µg/g ww of embryo and doses up to 1.0 mg/kg body weight for 3 weeks (O'Brien et al., 2009; Yeung et al., 2009). However, a recent study observed developmental toxicity of PFOA in cormorant (*Phalacrocorax carbo sinensis*), herring gull (*Larus argentatus*) and the domestic White Leghorn chicken (*Gallus gallus domesticus*), with chicken being the most sensitive showing 50 % reduced embryo survival at 2.5 µg/g egg PFOA (Nordén et al., 2016).

122. The soil-dwelling nematode *Caenorhabditis elegans* showed lethal effects with EC_{50} concentrations of 3.85 mM after 1 hour of exposure and 2.35 mM after 48 hours of exposure (Tominaga et al., 2004).

123. Studies with terrestrial plants such as lettuce (*Lactuca sativa*), cucumber (*Cucumis sativus*) and pakchoi (Brassica *rapa chinensis*), spring wheat, oats, potatoes, maize, and perennial ryegrass showed species-dependent adverse effects mediated by PFOA. For example, exposure to ≥1000 mg/L PFOA for 5 days almost completely inhibited lettuce and pakchoi root growth but not cucumber growth (Li, 2008). Also, visible abnormalities such as reduced growth and necrosis were observed in spring wheat, oats, potatoes and perennial ryegrass but not in maize exposed to 10-50 mg/kg PFOA/PFOS mixture in the soil (Stahl et al., 2009).

124. PFOA is primarily detected in the livers of biota such as polar bears and seals (Martin et al., 2004; Smithwick et al., 2005; Dietz et al., 2007; Sonne et al., 2008; Butt et al., 2007; Ishibashi et al., 2008), but the adverse effects of PFOA in such biota have not yet been elucidated. Ishibashi et al., (2008) showed activation of PPARα in the PFOA-exposed livers of Baikal seals at a LOEC for PFOA of 62.5 µM, but no PFOA-mediated adverse effect in the livers was reported. Also, Sonne et al., (2008) concluded that it is not clear whether chronic exposure to PFOA is associated with liver lesions in polar bears, but it is possible that at sufficient concentrations in polar bears, PFOA might induce hepatic alterations.

125. Environment Canada and Health Canada (2012) concludes that the risk quotients (PEC/PNEC) indicate low likelihood of risk to pelagic organisms, mammalian wildlife from exposures at current concentrations in the environment; however, due to the persistence of the substance, its tendency to accumulate and biomagnify in a variety of terrestrial and marine mammals, its hepatotoxicity, and the upward temporal trend of PFOA concentrations in polar bears and some other species, PFOA concentrations in polar bears may approach exposures resulting in harm. Indeed, a temporal trend analysis indicated an annual increase of 2.3% in PFOA levels in East Greenland polar bears from 1984 to 2006 (Dietz et al., 2007).

Background information on adverse effects on human health

(For references, see draft risk profile)

126. The toxicity of PFOA has been evaluated by ECHA, US EPA, the Canadian ministries, EFSA (ECHA, 2011, 2015a; Environment Canada and Health Canada, 2012; US EPA, 2006; EFSA, 2008). In the European Union, PFOA has a legally-binding harmonised classification as Carc. 2, Repr. 1B and STOT RE 1 (liver) according to Regulation (EU) No 944/2013 (index number: 607-704-00-2) amending Regulation (EC) No 1272/2008 on classification, labelling and packaging of substances and mixtures as amended by Regulation (EU) No 944/2013 (index number: 607-704-00-2).

127. PFOA is readily absorbed after exposure (ingestion) and accumulates in serum and highly perfused organs, mainly in the liver and kidney, due to PFOA primarily binding to albumin proteins in the blood. PFOA does not undergo metabolism or biotransformation in the body (Environment Canada and Health Canada, 2012; Post et al., 2012). As mentioned earlier, the half life of PFOA elimination in humans is long, ranging between 2 and 4 years (Olsen et al., 2007; Russell et al., 2015).

128. Animal studies have demonstrated the induction of tumours mediated by PFOA or APFO and hepatic activation of PPARα has been proposed as a mechanism of induction of hepatic tumours (Klaunig et al., 2003). However, the PPAR-agonist mode of action proposed for rat liver, testes and pancreatic tumours may not be relevant for humans. However, human relevance had not been definitively determined according to established frameworks a decade ago (Meek et al., 2003; Boobis et al., 2006), and PFOA compounds have also not been tested for carcinogenic potential in any laboratory animal species other than rats (Environment Canada and Health Canada, 2012). Therefore, the ECHA's Risk Assessment Committee (RAC) came to the conclusion that data on the mode of action are

insufficient to conclude that APFO-induced tumours in animals are not relevant for humans, and therefore, PFOA is classified as Carc 2, H351 (ECHA, 2011). Based on limited evidence in humans that PFOA causes testicular and renal cancer as well as limited evidence in experimental animals, IARC has classified PFOA as a Group 2B substance (possibly carcinogenic to humans) (Benbrahim-Tallaa et al., 2014).

129. PFOA is also classified as Repr 1B, H360D (May damage the unborn child) (ECHA, 2011). Positive correlation in PFOA level between maternal and cord blood samples has been reported in several birth cohort studies in Spain and Norway (Manzano-Salgado et al., 2015; Gutzkow et al., 2012). As APFO can be transferred to infants through breast-feeding, the RAC agreed on an additional classification of PFOA on lactation effects (CLP: Lact. H362: May cause harm to breast-fed children) (ECHA, 2011).

Epidemiological studies

130. Several epidemiological studies investigated exposure and related health effects in workers and different population groups. The most established epidemiological study regarding PFOA is the the C8 Health Project, a population-wide health study to determine a probable link between PFOA exposure and human disease. This project was set up as a part of the settlement of a lawsuit against DuPont brought by the communities around Parkersburg, West Virginia, USA, where its drinking water source has been contaminated with PFOA coming from the DuPont Washington Works Plant. About 69,000 subjects residing in the 6 public water districts contaminated by PFOA were included in the study, and participant information and blood samples were collected between August 2005 and July 2006 (C8 Science Panel, 2013). The main conclusions of the C8 Health Project are in the form of Probable Link reports, which summarise in each case whether the Science Panel found or did not find a link between exposure and disease. For six disease categories, the Science Panel concluded that there was a probable link to PFOA exposure: diagnosed high cholesterol, ulcerative colitis, thyroid disease, testicular cancer, kidney cancer, and pregnancy-induced hypertension. The following section covers some of the key findings from the C8 Health Project as well as other related studies.

131. Exposure of PFOA and elevated cholesterol levels (e.g. hypercholesterolemia) as well as other lipid parameters (e.g. uric acid, serum lipid) have been positively correlated in studies involving occupationally-exposed workers (Sakr et al., 2007), highly exposed community residents (Steenland et al., 2009; Steenland et al., 2010a; Frisbee et al., 2010; Fitz-Simon et al., 2013) and the general population (Eriksen et al., 2013; Nelson et al., 2010). In exposed workers, serum PFOA was positively associated with total cholesterol with an increase of 1.06 mg/dL of cholesterol for each 1 ppm increase in PFOA after adjusting for age, BMI, gender, and decade of hire (Sakr et al., 2007). PFOA exposure is associated with reduction in levels of mRNAs involved in chlolesterol transport and an increase in mRNAs involved in cholesterol mobilization in women, suggesting that exposure may lead to a hypercholesterolaemic environment with implications for human disease (Fletcher et al., 2013). PFOA was found to be associated with elevated serum levels of uric acid in Taiwanese children, especially boys (Qin et al., 2016).

132. There is evidence to suggest a potential association between PFOA exposure and increased risk of testicular and/or kidney cancers. A dose-related increase in both cancers was observed among 32,254 participants with hazard ratios of 3.17 and 1.58 for testicular and kidney cancers, respectively, for those in the highest PFOA exposure quartile (Barry et al., 2013). This observation has been consistently reported in other publications from the C8 Health Project (Vieira et al., 2013; Steenland, 2012). PFOA can induce colorectal cancer cell DLD-1 invasive ability by activating nuclear factor kappa B (Miao et al., 2015). PFOA is positively associated with liver alanine transaminase levels, a marker of hepatocellular damage (Gallo et al., 2012). However, in a review by Chang et al., (2014) on PFOA exposure and human cancer risk, it states that many positive associations with PFOA exposure were detected in community settings without occupational exposure yet were not supported by results in exposed workers. As occupational exposure to PFOA is one to two orders of magnitude higher than environmental exposure, the discrepant positive findings are likely due to chance, confounding, and/or bias. Thus, the epidemiological evidence currently does not appear to support the hypothesis of a causal association between PFOA exposure and cancer in humans. Furthermore, in a mortality study at the DuPont Washington Works plants, although a suggestive elevation in kidney cancer risk was found (relative risk=1.8), it was not statistically significant. There was also no statistically significant ($p<0.05$) excesses for any cancers reported (Leonard et al., 2008). A general population study in Denmark also showed no significant association between PFOA exposure and cancer (Eriksen et al., 2009).

133. Ulcerative colitis and rheumatoid arthritis have been positively associated with PFOA exposure among workers (Steenland et al., 2015).

134. Adverse reproductive effects have been reported in some epidemiological studies. A positive association between serum PFOA levels and the rate of menopause has been reported in several studies such that women in the highest quintile of PFOA exposure have higher chances of experiencing earlier menopause (Taylor et al., 2013; Knox et al., 2011a). Another study in Canada involving more than 1,700 women showed possible reduction in fecundity in women with higher plasma levels of PFOA (Velez et al., 2015). Serum PFOA is positively associated with pregnancy-induced hypertension (Darrow et al., 2013), and preeclampsia is weakly associated with PFOA level (Savitz et al., 2012; Stein et al., 2009).

135. Developmental effects in humans have also been reported in a number of studies but remain inconclusive. On one hand, an inverse correlation between PFOA and birth weight, ponderal index and head circumference has been reported in several mother-child cohort studies (Fei et al., 2007; Apelberg et al., 2007; Maisonet et al., 2012; Chen et al., 2012, Wu et al., 2012; Whitworth et al., 2012a). A novel meta-analysis methodology ("the Navigation Guide") was applied to review the evidence of 9 human studies to determine if there is a relationship between PFOA exposure and foetal growth, and the authors drew the conclusion that there was sufficient evidence to suggest that developmental exposure to PFOA reduces foetal growth (Johnson et al., 2014; Lam et al., 2014). On the other hand, mothers of low birth weight babies might have altered renal function such as less plasma volume expansion, therefore leading to reduced clearance of PFOA through glomerular filtration (Whitworth et al., 2012b; Verner et al., 2015; Sagiv et al., 2015). Furthermore, there are other studies that reported no significant association between maternal serum PFOA levels and birth weight (Washino et al., 2009; Stein et al., 2009; Monroy et al., 2008; Hamm et al., 2010).

136. Other reported adverse effects observed in children associated with higher levels of serum PFOA include impaired neurodevelopment (see section below), adiposity (Halldorsson et al., 2012 ; Braun et al., 2016), dyslipidemia (Geiger et al., 2014), altered renal function (Kataria et al., 2015), reduced humoral immune response (Grandjean et al., 2012; Grandjean and Budtz-Jørgensen, 2013) and lower levels of insulin-like growth factor-1 and sex hormones (Lopez-Espinosa et al., 2016). In adolescents, PFOA concentrations in serum are statistically significantly associated with higher odds of self-reported food allergies (Buser and Scinicariello, 2016). Higher prenatal serum levels of PFOA are associated with greater adiposity at age 8 and a more rapid increase in body mass index scores between 2 – 8 years-old (Braun et al., 2016). In girls, higher concentrations of PFOA are associated with a later age of puberty (Lopez-Espinosa et al., 2011; Holtcamp, 2012).

Neurotoxicity in humans

137. Impaired neurodevelopment has been associated with PFOA. An inverse relationship between prenatal PFOA concentrations in mothers and neurodevelopment as determined with the mental development index (MDI) in female (not male) offspring at 6 months of age was observed in a Japanese birth cohort (Hokkaido) study. However, this relationship was not observed with offspring at 18 months of age (Goudarzi et al., 2016). Also, no correlation between PFOA levels and birth weight was observed in the same cohort study (Washino et al., 2009). Statistically significant inverse associations between PFOA and memory impairment has been reported (Gallo et al., 2013). On the other hand, there are studies that reported no association between PFOA exposure and impaired neurodevelopment or behaviour (Chen et al., 2013; Stein et al., 2013).

Immunotoxicity

138. Reduced humoral immune response has been observed in a few studies (Grandjean et al., 2012; Looker et al., 2014; Kielsen et al., 2015). In particular, elevated PFOA serum levels are associated with reduced antibody titer rise, especially to A/H3N2 influenza virus and an increased risk of not attaining a protection threshold antibody concentration (Looker et al., 2014). In the Danish National Birth Cohort study, it was reported that prenatal exposure to PFOA is not associated with increased risk of infectious diseases leading to hospitalisation in early childhood; however, when the analysis was stratified by gender, girls showed a slightly higher risk of hospitalisation for infections associated with higher maternal PFOA levels (incidence rate ratio of 1.74 at the highest quartile compared with the lowest) (Fei et al., 2010).

139. Using data from a study of immunotoxicity in children (Grandjean et al., 2012), BMDLs were calculated to be approximately 0.3 ng/mL for PFOA, in terms of the serum concentration. Using an uncertainty factor of ten to take into account individual susceptibility, the BMDLs would result in a reference dose serum concentration of about or below 0.1 ng/mL.

Endocrine Disruption

140. Findings from studies seem to indicate a PFOA-mediated effect on the endocrine system. Prenatal exposure of PFOA may alter testosterone concentrations in females (Maisonet et al., 2015), and an inverse correlation between parathyroid hormone 2 receptor (PTH2R) and PFOA exposure was also reported in a study of 189 women (Galloway et al., 2015). As for men, a study by La Rocca et al. (2015) reported an inverse relationship between PFOA serum level in men and expression of nuclear receptors such as estrogen and androgen receptors. Early menopause in women with high PFOA levels was observed in the cross-sectional analysis from the C8 Health Project (Knox et al., 2011a, 2011b).

141. PFOA has been implicated to act as a so-called obesogene similar to other endocrine disruptive compounds that can act directly on ligands for nuclear hormone receptors or affect components in metabolic signalling pathways. Hines et al. (2009) demonstrated that mice prenatally exposed to low doses of PFOA (0.01-0.1 mg/kg) exhibited increased body weight as well as serum insulin and leptin levels in mid-life, effects that are not seen in higher doses of PFOA, suggesting that there is a critical window of exposure for low dose effects of PFOA on body weight and on insulin as well as leptin concentrations and that might lead to metabolic perturbations such as diabetes later in life. Similar to the Hines study, a human prospective cohort study showed a correlation between low dose PFOA exposure

of 655 Danish pregnant women and obesogenic effects in their offspring at 20 years of age. Maternal PFOA concentrations were positively associated with serum insulin and leptin levels and inversely associated with adiponectin levels in female offspring (Halldorsson et al., 2012). On the other hand, the C8 Health Project concluded that PFOA exposure in early life was not associated with overweight and obesity risk in adulthood (Barry et al., 2014).

142. On the other hand, according to the study of Su et al., (2016), PFOA showed a potential protective effect against glucose intolerance and the risk of diabetes.

143. Evidence from several epidemiological studies seems to suggest an association between exposure to PFOA and changes in different thyroid hormones leading to altered thyroid function inducing thyroid disease such as hypothyroidism or hyperthyroidism (Shrestha et al., 2015; Lopez-Espinosa et al., 2012; Knox et al., 2011b; Kim et al., 2011; Melzer et al., 2010). For example, PFOA-mediated disruption of thyroid function has been observed in the C8 Health Project (C8 Science Panel, 2013). A cross-sectional study amongst 52,296 adults revealed significant elevations in serum thyroxine and significant reduction in T3 uptake associated with serum PFOA in all participants (Knox et al., 2011b). A study of 157 maternal and cord serum samples from Beijing showed a ratio of PFOA concentrations in fetal versus maternal serum of 0.65 :1 indicating placental transfer along with impacts on levels of thyroid stimulating hormone, triiodothyronin, and free triiodothyronin (Yang et al., 2016). Thyroid hormones are important for normal brain maturation and development and slight differences during pregnancy or after delivery can be associated with neurological impairment (Lopez-Espinosa et al., 2012). However, there have been studies that reported inconsistent findings between PFOA exposure and thyroid diseases (i.e. inverse relation between subclinical hyperthyroidism and PFOA or no association between hypothyroidism and PFOA) (Steenland et al., 2010b; C8 Science Panel, 2013).

144. The potential of PFOA to affect estrogen receptor (ER) and androgen receptor (AR) transactivity as well as aromatase enzyme activity was analysed in an *in vitro* study, and it was shown that PFOA significantly induced ER transactivity yet antagonised AR activity in a concentration-dependent manner. In addition, when PFOA was mixed with 6 other PFCs, a mixture effect more than additive was observed on AR function, emphasising the importance of considering the combined action of PFCs in assessing related health risks (Kjeldsen and Bonefeld-Jørgensen, 2013).

145. In ECHA (2015a), prepared by the German and Norwegian authorities, multiple studies were reviewed to develop DNELs (derived no-effect levels) for the EU human health risk characterisation. The studies considered DNELs in relation to the general adult population (Abbott et al., 2007; Macon et al, 2011; Lau et al., 2006; Fei et al., 2007). On the other hand, ECHA (2015a), also considered DNELs but had different assumptions and derivations than ECHA (2015a). The discrepancy between the two sets of DNEL values is further elaborated in ECHA (2015b). These findings are summarised in the Table 13 below.

Table 13: Summary of DNELs for PFOA derived for the general population as proposed in (ECHA, 2015a) and (ECHA, 2015b)

DNEL (ng/ml) from (ECHA, 2015a)	DNEL (ng/mL) from (ECHA, 2015b)	Endpoint of interest	Reference for DNEL estimation
209	800	Reduced birth weight in mice	Lau et al., 2006
277	Not reported [a]	Neonatal survival in mice	Abbott et al., 2007
1.3	Not reported [b]	Delayed mammary gland development in mice	Macon et al., 2011
0.3	Not reported [b]	Decrease in foetal birth weight in humans	Fei et al., 2007

[a] Not reported because the DNEL generated from Abbott et al., 2007 is in the similar range as Lau et al., 2006. Therefore, the RAC supports the use of a modified DNEL based on the Lau et al., 2006 study.

[b] Not reported because there are uncertainties within the relevant studies that RAC believes it is currently not possible to set a DNEL although some effects (e.g. on the mammary gland) may be more sensitive than the animal data currently used in the risk characterisation.

146. Using data from a recent study of immunotoxicity in children (Grandjean et al., 2012), BMDLs were calculated to be approximately 0.3 ng/mL for PFOA, in terms of the serum concentration. Using an uncertainty factor of ten to take into account individual susceptibility, the BMDLs would result in a reference dose serum concentration of about or below 0.1 ng/mL. The reference dose based on mammary gland development in mice would correspond to a serum-PFOA concentration of 0.8 ng/mL (Grandjean and Clapp, 2015).