[Note: this document will be issued as an information document, currently referenced in the draft risk profile as “UNEP/POPS/POPRC.18/INF/X]

Additional information on the draft risk profile on long-chain perfluorocarboxylic acids (PFCAs), their salts and related compounds

2.1.2 Uses

Unintentional production of long-chain PFCAs

1. Long-chain (C9 – C21) PFCAs and related compounds may be unintentionally produced during the manufacturing of per- and polyfluoroalkyl substances (PFAS), including those containing a carbon chain of less than nine carbon atoms.

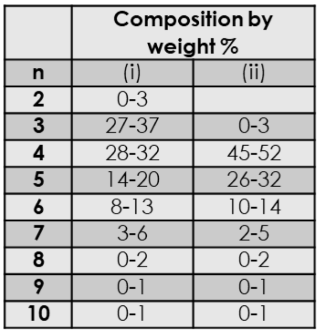
2. The manufacture of ammonium perfluorononanoate (APFN) leads to a technical mixture of PFCAs; Prevedouros et al. (2006) described the homologue profile for commercial APFN to consist primarily of C9 PFCA (73.6%), C11 PFCA (20.0%) and C13 PFCA (5.0%).

3. During the manufacturing of the perfluorohexanoic acid- (C6 PFCA) based substances, the fraction containing mainly long-chain PFCAs (referred to as the C8-fraction) can include up to 30% C9 – C14 PFCAs and related compounds (ECHA 2018b). The other fraction (the C6-fraction) has a reduced concentration of C9 – C14 PFCAs, in the low parts per million (ppm) range (ECHA 2018b). These fractions can be reworked or further processed to reduce the concentration of C9 – C14 PFCAs in mixtures and articles placed on the market (ECHA 2018b). C9 – C14 PFCAs can also be an impurity produced during the manufacturing of perfluorooctanoic acid (PFOA, C8 PFCA) (i.e., up to 0.21% C9 – C14 PFCAs) and PFOA-related compounds (i.e., 20 to 45% C9 – C14 related compounds to long-chain PFCAs) (ECHA 2018b).

Composition of fluorinated starting materials

4. Based on the available commercial information, starting materials that may be used for the production of compounds related to long-chain PFCAs consist of fluorotelomer alchohol mixtures of fluorinated chain lengths ranging from 4 to 20 carbons (see Table 1).

Table 1. Description of starting material used for the production of compounds related to long-chain PFCA

<table>
<tr><th>Use</th><th>Description of the starting material</th><th>Reference</th></tr>
<tr><td>Fluorinated lubricant additives</td><td>“[…] suitable fluorinated alcohols […] may be selected from the following species:
• $F(CF_2)_xCH_2OH$, wherein x is from 1 to about 20 […];
• $H(CF_2)_xCH_2OH$, wherein x is from 1 to about 20 […];
• $F(CF_2CF_2)_xCH_2CH_2OH$, wherein x is from 1 to about 10 […];
• $F(CF_2CF_2)_x(CH_2CH_2O)_yOH$, a telomer ethoxylate alcohol wherein x is from 1 to about 10 and y is from 1 to about 20 […]”</td><td>Beatty 2003</td></tr>
<tr><td>Fluorochemical oil and water repellents</td><td>Compositions of fluoroalcohols of formula $F(CF_2CF_2)_nCH_2CH_2OH$:

<table>
<tr><th></th><th colspan="2">Composition by weight %</th></tr>
<tr><th>n</th><th>(i)</th><th>(ii)</th></tr>
<tr><td>2</td><td>0-3</td><td></td></tr>
<tr><td>3</td><td>27-37</td><td>0-3</td></tr>
<tr><td>4</td><td>28-32</td><td>45-52</td></tr>
<tr><td>5</td><td>14-20</td><td>26-32</td></tr>
<tr><td>6</td><td>8-13</td><td>10-14</td></tr>
<tr><td>7</td><td>3-6</td><td>2-5</td></tr>
<tr><td>8</td><td>0-2</td><td>0-2</td></tr>
<tr><td>9</td><td>0-1</td><td>0-1</td></tr>
<tr><td>10</td><td>0-1</td><td>0-1</td></tr>
</table></td><td>Sherman et al. 2001</td></tr>
</table>

2.1.3 Releases to the environment

Table 2. Detection of long-chain PFCAs and their related compounds in environmental matrices and other matrices from impacted sites

Matrix	Country/ Region		Year(s)	Study site	Type of location or samples	Concentration	Mean	Reference
	Wastewater treatment plants (WWTPs)							
Sludge	Switzerland		2011	WTTP	45 WWTPs	C9 PFCA: 0.9 – 23 µg/kg of dry matter C10 PFCA: 0.9 – 73 µg/kg of dry matter		Alder and von der Voet 2014
Wastewater	United States		2005	WTTP	2 WWTPs	C9 PFCA: 0.59-54 ng/L C10 PFCA: <0.5-18 ng/L C11 PFCA: <LOD-1.9 ng/L C12 PFCA: <LOD		Loganathan et al. 2007
Sludge	United States		2005	WTTP	2 WWTPs	C9 PFCA: <2.5-67 ng/g dw C10 PFCA: 12-201 ng/g dw C11 PFCA: 5.9-37 ng/g dw C12 PFCA: 7.2-48 ng/g dw		Loganathan et al. 2007
Wastewater (final effluent)	United States		2004	WWTP	1 WWTP	C9 PFCA: 1.5-5.9 ng/L C10 PFCA: 0.6-5.1 ng/L	C9 PFCA: 3.4 ng/L C10 PFCA: 2.3 ng/L	Schultz et al. 2006
Sludge (digested)	United States		2004	WWTP	1 WWTP	C9 PFCA: 9.2-10.3 ng/g dw C10 PFCA: 5.4-6.4 ng/g dw C11 PFCA: 5.9-8.4 ng/g dw C12 PFCA: 3.6-4.2 ng/g dw	C9 PFCA: 9.9 ng/g dw C10 PFCA: 5.9 ng/g dw C11 PFCA: 6.8 ng/g dw C12 PFCA: 3.8 ng/g dw C13 PFCA: <3 ng/g dw	Schultz et al. 2006
Biosolids	United States		2020	Agricultural sites	Class B biosolids samples collected from a wastewater reclamation facility	C9 PFCA: n.d.-2 µg/kg C10 PFCA: 12-13 µg/kg C11 PFCA: 1.8-2.4 µg/kg C12 PFCA: 6.5-8 µg/kg C13 PFCA: n.d. C14 PFCA: n.d.-3.3		Pepper et al. 2021
WWTP influent	Mexico		2019	WWTP	1 WWTP	C11 PFCA: 24.1 (±2.5)-35.2 (±2.4) ng/L		Rodríguez-Varela et al. 2021
WWTP effluent	Mexico		2019	WWTP	1 WWTP	C11 PFCA: 25.5 (±1.8)-31.1 (±3.3) ng/L		Rodríguez-Varela et al. 2021
Wastewater	Mexico		2019	WWTP	Irrigation canal receiving raw wastewater	C11 PFCA: 38.3 (±3.4)-76.8 (±1.4) ng/L		Rodríguez-Varela et al. 2021
WWTP influent	Denmark		Not specified	WWTP	11 samples from 6 municipal WWTPs	C9 PFCA: <0.8-8.4 ng/L C10 PFCA: <1.6 ng/L		Bossi et al. 2008
WWTP effluent	Denmark		Not specified	WWTP	11 samples from 6 municipal WWTPs	C9 PFCA: <0.8-3.1 ng/L C10 PFCA: <1.6-3.6 ng/L		Bossi et al. 2008
Sludge	Denmark		Not specified	WWTP	7 municipal WWTPs	C9 PFCA: 0.4-8.0 µg/kg dw C10 PFCA: 1.2-32 µg/kg dw C11 PFCA: 0.4-4.4 µg/kg dw		Bossi et al. 2008

Matrix	Country/ Region		Year(s)	Study site	Type of location or samples	Concentration	Mean	Reference
Effluent water	Denmark		Not specified	WWTP	7 samples from 4 industrial WWTPs from textile, large chemical and wood floor production industries, and a facility handling various waste products	C9 PFCA: <0.8-76.0 ng/L C10 PFCA: <1.6-35.7 ng/L C11 PFCA: <2.2-18.8 ng/L		Bossi et al. 2008
Biosolids	Australia		Not specified	WWTP	Samples from 19 WWTPs	C9 PFCA: n.d.-4.9 ng/kg dw C10 PFCA: <MRL-34 ng/kg dw C11 PFCA: n.d.-3.0 ng/kg dw C12 PFCA: <MRL-18 ng/kg dw C13 PFCA: n.d.-1.8 ng/kg dw C14 PFCA: <MRL-4.2 ng/kg dw 8:2 FTSA: n.d.-4.0 ng/kg dw 10:2 FTSA: n.d.-1.9 ng/kg dw 8:2 diPAP: n.d.-240 ng/kg dw	C9 PFCA: 0.90 (±1.1) ng/kg dw C10 PFCA: 14 (±11.2) ng/kg dw C11 PFCA: 0.60 (±0.8) ng/kg dw C12 PFCA: 5.9 (±5.4) ng/kg dw C13 PFCA: 0.5 (±0.5) ng/kg dw C14 PFCA: 1.2 (±1.3) ng/kg dw 8:2 FTSA: 0.7 (±1.3) ng/kg dw 10:2 FTSA: 0.7 (±0.7) ng/kg dw 8:2 diPAP: 67 (±76) ng/kg dw	Moodie et al. 2021
Air	Canada		2009	WWTPs	Air samples collected using sorbent-impregnated polyurethane foam (SIP) disk passive air samplers (PAS), deployed for 63 days around a municipal WWTP	C9 PFCA: 0.88-4.84 pg/m^3 C10 PFCA: 0.57-8.82 pg/m^3 C11 PFCA: <0.04-5.83 pg/m^3 C12 PFCA: <0.24-3.44 pg/m^3 C14 PFCA: <0.28-1.43 pg/m^3 8:2 FTOH: 144-10 309 pg/m^3 10:2 FTOH: 70.4-1111 pg/m^3		Ahrens et al. 2011
Air	Canada		2013-2014	WWTPs	Air samples collected using SIP disk PAS, installed at WWTPs	C9 PFCA: BDL-77.9 pg/m^3 C10 PFCA: n.d.-84.2 pg/m^3 C11 PFCA: n.d.-15.9 pg/m^3 C12 PFCA: n.d. 101 pg/m^3 C13 PFCA: n.d.-0.966 pg/m^3 C14 PFCA: n.d.-5.13 pg/m^3 8:2 FTOH: 12.3-1440 pg/m^3 10:2 FTOH: 6-84.7 pg/m^3		Shoeib et al. 2016
Air	China		2013	WWTPs	Air samples collected collected using SIP disk PAS, installed at two WWTPs	C9 PFCA: 7.98-26.7 pg/m^3 C10 PFCA: 2.34-17.0 pg/m^3 C11 PFCA: 0.95-4.28 pg/m^3 C12 PFCA: 0.47-3.21 pg/m^3 8:2 FTOH: 46.1-122 pg/m^3 10:2 FTOH: 7.49-39.2 pg/m^3		Yao et al. 2016
WWTPs influent	Australia		2016	WWTPs	76 samples collected from 76 municipal WWTPs	C9 PFCA: 1.6-3.3 ng/L C10 PFCA: 2.0-6.3 ng/L C11 PFCA: n.d. C12 PFCA: n.d. 8:2 FTSA: 2.3-59 ng/L	C9 PFCA: 2.1 (±0.61) ng/L C10 PFCA: 3.4 (±1.3) ng/L C11 PFCA: n/a C12 PFCA: n/a 8:2 FTSA: 15 (±14) ng/L	Nguyen et al. 2022
Wastewater	Austria		Not specified	Not specified	Number of samples analysed: C9 PFCA (5), C10 PFCA (9) C11 PFCA (10) C12 PFCA (10)	C9 PFCA: n.d.-0.0018 µg/L C10 PFCA: n.d.- 0.0024 µg/L C11 PFCA: <LOQ C12 PFCA: <LOQ		Austria Annex E information, 2022
Sewage sludge	Austria		Not specified	Not specified	2 samples analyzed	C9 PFCA: n.d.-0.77 µg/kg TM C10 PFCA: 1.1-7.7 µg/kg TM		Austria Annex E information,

Matrix	Country/ Region		Year(s)	Study site	Type of location or samples	Concentration	Mean	Reference
						C11 PFCA: n.d.-2.1 µg/kg TM C12 PFCA: 0.77-2.7 µg/kg TM		2022
Sewage sludge compost	Austria		Not specified	Not specified	2 samples analyzed	C9 PFCA: 0.53-0.93 µg/kg TM C10 PFCA: 1.9-3.4 µg/kg TM C11 PFCA: n.d.-3.7 µg/kg TM C12 PFCA: 0.44-0.65 µg/kg TM		Austria Annex E information, 2022
	Landfills, incineration plants							
Leachate	United States		2013-2014	Landfills	18 landfills sites	10:2 FTCA: ND-0.3 µg/L 8:2 FTUCA: ND-0.02 µg/L Note: C11 – C18 PFCAs also detected above the LOD in <20% of samples, but concentrations were not specified.	C9 PFCA: 0.005 – 0.1 µg/L C10 PFCA: 0.003 – 0.1 µg/L 8:2 FTCA: 0.01 – 0.4 µg/L	Lang et al. 2017
Leachate	China		2015-2017	Municipal solid wastes (MSW) incineration plants	3 MSW incineration plants	C9 PFCA: n.d. C10 PFCA: 0.362-1.26 ng/ml C11 PFCA: 0.0894-0.142 ng/ml C12 PFCA: 0.371-0.704 ng/ml C13 PFCA: 0.138-0.156 ng/ml C14 PFCA: 0.140-0.261 ng/ml 8:2 diPAP: 0.267-0.323 ng/L		Liu et al. 2021
Fly ash	China		2015-2017	Municipal solid wastes (MSW) incineration plants	3 MSW incineration plants	C9 PFCA: 0.111-0.441 ng/g C10 PFCA: 0.0218-0.0915 ng/g C11 PFCA: n.d.-0.0195 ng/g C12 PFCA: 0.0109-0.0158 ng/g C13 PFCA: n.d.-0.0358 ng/g C14 PFCA: 0.0311-0.0540 ng/g 8:2 diPAP: n.d. 0.120 ng/g		Liu et al. 2021
Bottom ash	China		2015-2017	Municipal solid wastes (MSW) incineration plants	3 MSW incineration plants	C9 PFCA: 0.243-0.403 ng/g C10 PFCA: 0.0298-0.0578 ng/g C11 PFCA: 0.0165-0.0790 ng/g C12 PFCA: 0.0944-0.121 ng/g C13 PFCA: n.d.-0.0755 ng/g C14 PFCA: n.d.-0.0263 ng/g 8:2 diPAP: 0.119-0.250 ng/g		Liu et al. 2021
Soil	South Korea		2017	Landfills	8 soil samples collected from vacant lots in municipal and industrial landfill sites	C9 PFCA: n.d.-0.479 ng/g dw C10 PFCA: 0.058-2.85 ng/g dw C11 PFCA: n.d.-1.03 ng/g dw C12 PFCA: n.d.-3.16 ng/g dw C13 PFCA: n.d.-0.985 ng/g dw C14 PFCA: n.d.-0.812 ng/g dw	C9 PFCA: 0.252 ng/g dw C10 PFCA: 0.614 ng/g dw C11 PFCA: 0.275 ng/g dw C12 PFCA: 0.460 ng/g dw C13 PFCA: 0.192 ng/g dw C14 PFCA: 0.162 ng/g dw	Sim et al. 2021
Leachate	Canada		2010	Landfills	33 samples of leachate (flow-through or recirculated) from two municipal landfills		C9 PFCA: 15(±1.4)-450(±80) ng/L C10 PFCA: 109(±0.42)-1100(±140) ng/L C11 PFCA: <3.0-120(±100) ng/L C12 PFCA: <1.4-8.8(±0.75) ng/L C14 PFCA: <1.5-5.1(±1.6) ng/L 8:2 FTCA: <8.6-5200(±30) ng/L 10:2 FTCA: <2.7-775(±530) ng/L 8:2 FTUCA: <2.9-2100(±6.0) ng/L	Benskin et al. 2012

Matrix	Country/ Region		Year(s)	Study site	Type of location or samples	Concentration	Mean	Reference
							10:2 FTUCA: <1.7-430(±250) ng/L	
Percolate	Denmark		Not specified	Landfills	3 samples from 2 landills	C9 PFCA: <0.8 ng/L C10 PFCA: <1.6 ng/L C11 PFCA: <2.2 ng/L		Bossi et al. 2008
Leachate	Germany		Not specified	Landfills	Treated leachate from 22 landfill sites	C9 PFCA: n.d.-80.06 ng/L C10 PFCA: n.d.-55.09 ng/L C11 PFCA: n.d.-2.98 ng/L C12 PFCA: n.d.-2.45 ng/L C13 PFCA: n.d.-0.62 ng/L C14 PFCA: n.d.-0.39 ng//L C15 PFCA: n.d.-0.42 ng/L C16 PFCA: n.d.-1.91 ng/L C17 PFCA: n.d.-1.04 ng/L C18 PFCA: n.d.-2.96 ng/L		Busch et al. 2010
Leachate	Spain		2015	Landfills	6 samples from 4 municipal solid waste landfill sites	C9 PFCA: <LOD C10 PFCA: <LOD C11 PFCA: <LOD C12 PFCA: <LOD C13 PFCA: <LOD C14 PFCA: <LOD-68.4 ng//L		Fuertes et al. 2017
Leachate	Japan		2019-2021	Landfills	Industrial waste landfills	C9 PFCA: 12-1200 ng/L C10 PFCA: 14-18 ng/L C11 PFCA: 13-120 ng/L C12 PFCA: 5.4-8.3 ng/L C13 PFCA: n.d. C14 PFCA: n.d. C16 PFCA: n.d. C18 PFCA: n.d.	C9 PFCA: 500 (±350) ng/L C10 PFCA: 16 (±3.0) ng/L C11 PFCA: 86 (±48) ng/L C12 PFCA: 6.8 (±2.0) ng/L	Kameoka et al. 2021
Leachate	Japan		2019-2021	Landfills	Municipal solid waste landfills	C9 PFCA: 4.2-12 ng/L C10 PFCA: 18 ng/L C11 PFCA: 8.7-9.1 ng/L C12 PFCA: n.d. C13 PFCA: n.d. C14 PFCA: n.d. C16 PFCA: n.d. C18 PFCA: 110 ng/L	C9 PFCA: 7.2 (±3.4) ng/L C10 PFCA: 18 ng/L C11 PFCA: 8.9 (±0.23) ng/L C18 PFCA: 110 (±0.058) ng/L	Kameoka et al. 2021
Air	Canada		2009	Landfils	Air samples collected using SIP disk PAS, deployed for 55 days at 2 municipal solid waste landfill sites	C9 PFCA: 0.97-15.8 pg/m^3 C10 PFCA: 0.84-18.9 pg/m^3 C11 PFCA: <0.04-17.4 pg/m^3 C12 PFCA: 0.71-17.4 pg/m^3 C14 PFCA: <0.28-4.30 pg/m^3 8:2 FTOH: 223-17 381 pg/m^3 10:2 FTOH: 125-2151 pg/m^3		Ahrens et al. 2011
Air (gas-phase)	Germany		2009	Landfills	Air samples collected from two landfills	8:2 FTOH: 17.6-433.6 pg/m^3 10:2 FTOH: 5.7-92.7 pg/m^3 12:2 FTOH: 2.3-38.0 pg/m^3 8:2 FTA: 0.2-12.6 pg/m^3 10:2 FTA: n.d.-7.3 pg/m^3		Weinberg et al. 2011

Matrix	Country/ Region		Year(s)	Study site	Type of location or samples	Concentration	Mean	Reference
Air (particle-phase)	Germany		2009	Landfills	Air samples collected from two landfills	C9 PFCA: n.d.-0.7 pg/m^3 C10 PFCA: n.d.-0.8 pg/m^3 C11 PFCA: n.d.-0.8 pg/m^3 C12 PFCA: n.d.-0.3 pg/m^3		Weinberg et al. 2011
Military bases, airports								
Groundwater	United States		1942-1990	Military bases	4 archived groundwater samples	C9 PFCA: 40-390 ng/L C10 PFCA: <LOD-17 ng/L C11 PFCA: <LOD-<3.1 ng/L C12 PFCA: <LOD C13 PFCA: <LOD C14 PFCA: <LOD		Backe et al. 2013
Groundwater	United States		1950-1993	Military bases	8 archived groundwater samples	C9 PFCA: <LOD -680 ng/L C10 PFCA: <3.1-19 ng/L C11 PFCA: <LOD-5.2 ng/L C12 PFCA: <LOD-<3.4 ng/L C13 PFCA: <LOD C14 PFCA: <LOD		Backe et al. 2013
Surface soil	Canada		2016-2017	Airports	Soil samples from aqueous film-forming foam (AFFF)-impacted sites of four airports	C9 PFCA: n.d.-13.8 µg/kg dw C10 PFCA: n.d.-15.8 µg/kg dw C11 PFCA: n.d.-8.3 µg/kg dw C12 PFCA: n.d.-9.0 µg/kg dw C13 PFCA: n.d.-1.1 µg/kg dw C14 PFCA: n.d.-1.3 µg/kg dw C16 PFCA: n.d.-0.2 µg/kg dw 8:3 FTCA: n.d.-1.2 µg/kg dw 9:3 FTCA: n.d.-9.9 µg/kg dw 11:3 FTCA: n.d.-1.8 µg/kg dw 8:2 FTUA: n.d.-0.5 µg/kg dw 8:2 FTSA: n.d.-1684.4 µg/kg dw 10:2 FTSA: n.d.-46.9 µg/kg dw 12:2 FTSA: n.d. 14:2 FTSA: n.d.-13.9 µg/kg dw		Liu et al. 2022
Subsurface soil	Canada		2016-2017	Airports	Subsurface soil samples from aqueous film-forming foam (AFFF)-impacted sites of four airports	C9 PFCA: n.d.-2.2 µg/kg dw C10 PFCA: n.d.-0.9 µg/kg dw C11 PFCA: n.d.-0.3 µg/kg dw C12 PFCA: n.d. C13 PFCA: n.d. C14 PFCA: n.d. C16 PFCA: n.d. 8:3 FTCA: n.d. 9:3 FTCA: n.d. 11:3 FTCA: n.d. 8:2 FTUA: n.d.-0.2 µg/kg dw 8:2 FTSA: n.d.-56.4 µg/kg dw 10:2 FTSA: n.d.-0.5 µg/kg dw 12:2 FTSA: n.d. 14:2 FTSA: n.d.		Liu et al. 2022
Groundwater	Canada		2016-	Airports	Groundwater samples from	C9 PFCA: n.d.-2.0 µg/L		Liu et al. 2022

Matrix	Country/ Region		Year(s)	Study site	Type of location or samples	Concentration	Mean	Reference
			2017		aqueous film-forming foam (AFFF)-impacted sites of four airports	C10 PFCA: n.d.-0.5 µg/L C11 PFCA: n.d.-0.2 µg/L C12 PFCA: n.d. C13 PFCA: n.d. C14 PFCA: n.d. C16 PFCA: n.d. 8:2 FTUA: n.d. 10:2 FTUA: n.d. 8:2 FTSA: n.d.-230.0 µg/L 10:2 FTSA: n.d.-0.5 µg/L		
	Land application of biosolids, agricultural sites							
Well water	United States		2009	Farms	21 farms with historical land application of fluorochemical industry impacted biosolids	C9 PFCA: <LOD-25.7 ng/L C10 PFCA: <LOD		Lindstrom et al. 2011
Surface water	United States		2009	Farms	21 farms with historical land application of fluorochemical industry impacted biosolids	C9 PFCA: <LOD-285.6 ng/L C10 PFCA: <LOD-838.2 ng/L		Lindstrom et al. 2011
Soil	United States		2020	Agricultural sites	72 soil samples collected at various depths	C9 PFCA: n.d.-0.61 µg/kg C10 PFCA: n.d.-4.1 µg/kg C11 PFCA: n.d.-0.41 µg/kg C12 PFCA: n.d.- 0.48 µg/kg C13 PFCA: n.d. C14 PFCA: n.d.- 0.16 µg/kg		Pepper et al. 2021
Groundwater	United States		2020	Agricultural sites	Samples collected from nine irrigation wells associated with the agricultural sites	C9 PFCA: n.d.-3.4 ng/L C10 PFCA: n.d.-19 ng/L		Pepper et al. 2021
Soil	South Korea		2017	Farmland	4 soil samples collected from farmlands	C9 PFCA: 0.69-0.379 ng/g dw C10 PFCA: 0.164-0.300 ng/g dw C11 PFCA: n.d.-0.491 ng/g dw C12 PFCA: 0.059-0.150 ng/g dw C13 PFCA: n.d.-0.172 ng/g dw C14 PFCA: n.d	C9 PFCA: 0.281 ng/g dw C10 PFCA: 0.241 ng/g dw C11 PFCA: 0.279 ng/g dw C12 PFCA: 0.103 ng/g dw C13 PFCA: 0.081 ng/g dw C14 PFCA: n.d	Sim et al. 2021
Soil	United States		2015	Agricultural site	34 surface soil samples from agricultural feedstock station with history of land application of biosolids since mid-1990s		C9 PFCA: 5.1 µg/kg (average) C10 PFCA: 26 µg/kg (average) C11 PFCA: 3.0 µg/kg (average) C12 PFCA: 6.2 µg/kg (average)	Johnson 2022
	Ski areas							
Snow	United States		2020	Skiing area	Snow samples after cross-country ski races	C9 PFCA: n.d.-211 ng/L C10 PFCA: 1.87-1180 ng/L C11 PFCA: n.d.-606 ng/L C12 PFCA: 3.74-1800 ng/L C13 PFCA: 2.38-1000 ng/L C14 PFCA: 12.9-4210 ng/L 8:2 FTSA: n.n.-7.2 ng/L		Carlson and Tupper 2020

Matrix	Country/ Region		Year(s)	Study site	Type of location or samples	Concentration	Mean	Reference
Soil	United States		2020	Skiing area	Soil samples collected after snowmelt in a skiing area	C9 PFCA: n.d. C10 PFCA: n.d.-1.75 ng/g dw C11 PFCA: n.d. C12 PFCA: n.d.-2.82 ng/g dw C13 PFCA: n.d.-3.61 ng/g dw C14 PFCA: 1.97-3.91 ng/g dw 8:2 FTSA: n.d.		Carlson and Tupper 2020
Soil	Norway		2017-2018	Skiing area	5 soil samples collected after snowmelt in a skiing area	C9 PFCA: <LOQ-0.602 ng/g dw C10 PFCA: <LOQ-1.96 ng/g dw C11 PFCA: <LOQ-0.294 ng/g dw C12 PFCA: <LOQ-0.401 ng/g dw C13 PFCA: <LOQ-0.203 ng/g dw C14 PFCA: <LOQ-0.138 ng/g dw	C9 PFCA: 0.179 (±0.177) ng/g dw C10 PFCA: 0.417 (±0.632) ng/g dw C11 PFCA: 0.134 (±0.112) ng/g dw C12 PFCA: 0.159 (±0.139) ng/g dw C13 PFCA: 0.090 (±0.067) ng/g dw C14 PFCA: 0.122 (±0.140) ng/g dw	Grønnestad et al. 2019
Snow	Sweden		2010	Skiing area	Snow samples collected after a ski competition	C9 PFCA: n.d.-19.6 ng/L C10 PFCA: n.d.-17.2 ng/L C11 PFCA: n.d.-12.8 ng/L C12 PFCA: n.d.-21.8 ng/L C13 PFCA: n.d.-22.0 ng/L C14 PFCA: n.d.-57.9 ng/L C15 PFCA: n.d.-16.8 ng/L C16 PFCA: n.d.-108 ng/L C17 PFCA: n.d.-55.9 ng/L C18 PFCA: n.d.-786 ng/L C19 PFCA: n.d.-60.6 ng/L C20 PFCA: n.d.-113 ng/L C21 PFCA: n.d.		Plassman and Berger 2013
Soil	Sweden		2010	Skiing area	Soil samples collected after snowmelt in a skiing area	C9 PFCA: n.d.-1.15 ng/g dw C10 PFCA: n.d.-3.38 ng/g dw C11 PFCA: n.d.-1.82 ng/g dw C12 PFCA: n.d.-2.48 ng/g dw C13 PFCA: n.d.-1.43 ng/g dw C14 PFCA: n.d.-2.28 ng/g dw C15 PFCA: n.d.-0.623 ng/g dw C16 PFCA: n.d.-0.709 ng/g dw C17 PFCA: n.d.-0.307 ng/g dw C18 PFCA: n.d.-1.89 ng/g dw C19 PFCA: n.d.-0.141 ng/g dw C20 PFCA: n.d.-0.175 ng/g dw C21 PFCA: n.d.-0.021 ng/g dw		Plassman and Berger 2013
	Industrial and urban areas							
River water	India		2014	Ganges River	14 samples collected in nine locations, including in industrialized areas	C9 PFCA: <MQL-0.19 ng/L C10 PFCA: <MQL-0.19 ng/L C11 PFCA: <MQL C12 PFCA: <MQL-0.05 ng/L C13 PFCA: <MQL-0.03 ng/L C14 PFCA: <MQL		Sharma et al. 2016
Groundwater water	India		2014	Ganges River bank	14 samples collected from wells in the vicinity of the	C9 PFCA: <MQL-0.22 ng/L C10 PFCA: <MQL-0.10 ng/L		Sharma et al. 2016

Matrix	Country/ Region		Year(s)	Study site	Type of location or samples	Concentration	Mean	Reference
					Ganges River bank	C11 PFCA: <MQL C12 PFCA: <MQL-0.05 ng/L C13 PFCA: <MQL-0.02 ng/L C14 PFCA: <MQL		
Surface sediment	China		Not specified	Plain river network of Changshu (Tasin Basin)	17 sampling sites located in residential, agricultural and industrial areas	C9 PFCA: 0.99-9.65 ng/g dw C10 PFCA: 1.33-24.99 ng/g dw C11 PFCA: 0.99-15.67 ng/g dw C12 PFCA: 0.27-18.32 ng/g dw C13 PFCA: <3.25-25.91 ng/g dw C14 PFCA: <0.20-11.97 ng/g dw	C9 PFCA: 4.71 ng/g dw C10 PFCA: 7.11 ng/g dw C11 PFCA: 4.86 ng/g dw C12 PFCA: 6.86 ng/g dw C13 PFCA: 8.44 ng/g dw C14 PFCA: 4.33 ng/g dw	Li and Hua 2021
Suspended particles	China		Not specified	Plain river network of Changshu (Tasin Basin)	17 sampling sites located in residential, agricultural and industrial areas	C9 PFCA: 3.26-178.37 ng/g dw C10 PFCA: 3.60-30.40 ng/g dw C11 PFCA: 1.79-85.35 ng/g dw C12 PFCA: 3.42-159.01 ng/g dw C13 PFCA: 3.96-85.69 ng/g dw C14 PFCA: 1.99-42.57 ng/g dw	C9 PFCA: 20.05 ng/g dw C10 PFCA: 13.58 ng/g dw C11 PFCA: 25.10 ng/g dw C12 PFCA: 29.97 ng/g dw C13 PFCA: 23.49 ng/g dw C14 PFCA: 16.64 ng/g dw	Li and Hua 2021
Dissolved phase	China		Not specified	Plain river network of Changshu (Tasin Basin)	17 sampling sites located in residential, agricultural and industrial areas	C9 PFCA: 0.54-48.83 ng/L C10 PFCA: 2.88-264.30 ng/L C11 PFCA: 0.18-221.87 ng/L C12 PFCA: 0.44-12.85 ng/L C13 PFCA: <0.47-8.56 ng/L C14 PFCA: <0.23-5.08 ng/L	C9 PFCA: 18.69 ng/L C10 PFCA: 35.57 ng/L C11 PFCA: 57.66 ng/L C12 PFCA: 5.04 ng/L C13 PFCA: 3.56 ng/L C14 PFCA: 1.90 ng/L	Li and Hua 2021
Colloidal phase	China		Not specified	Plain river network of Changshu (Tasin Basin)	17 sampling sites located in residential, agricultural and industrial areas	C9 PFCA: <0.21-44.36 ng/L C10 PFCA: 0.44-258.46 ng/L C11 PFCA: 0.12-210.38 ng/L C12 PFCA: 0.26-10.88 ng/L C13 PFCA: <0.24-8.18 ng/L C14 PFCA: <0.15-4.65 ng/L	C9 PFCA: 15.72 ng/L C10 PFCA: 40.25 ng/L C11 PFCA: 55.09 ng/L C12 PFCA: 3.84 ng/L C13 PFCA: 2.70 ng/L C14 PFCA: 1.77 ng/L	Li and Hua 2021
Soluble phase	China		Not specified	Plain river network of Changshu (Tasin Basin)	17 sampling sites located in residential, agricultural and industrial areas	C9 PFCA: <0.21-20.67 ng/L C10 PFCA: 0.96-26.81 ng/L C11 PFCA: 0.06-46.43 ng/L C12 PFCA: 0.18-4.98 ng/L C13 PFCA: <0.06-5.33 ng/L C14 PFCA: <0.08-1.02 ng/L	C9 PFCA: 7.47 ng/L C10 PFCA: 7.57 ng/L C11 PFCA: 11.82 ng/L C12 PFCA: 1.62 ng/L C13 PFCA: 1.27 ng/L C14 PFCA: 0.55 ng/L	Li and Hua 2021
Rain	China		2016	Fluorochemical manufacturing parks (FMPs) in Fuxin	94 multimedia samples collected in the area surrounding two FMPs	C9 PFCA: n.d.-13 ng/L C10 PFCA: 0.57-22 ng/L C11 PFCA: n.d.-2.1 ng/L C12 PFCA: 0.37-1.7 ng/L 8:2 FTUCA: n.d.-3.0 ng/L		Chen et al. 2018
Shallow groundwater	China		2016	Fluorochemical manufacturing parks (FMPs) in Fuxin	94 multimedia samples collected in the area surrounding two FMPs	C9 PFCA: n.d.-3.7 ng/L C10 PFCA: n.d-3.9 ng/L C11 PFCA: n.d C12 PFCA: n.d. 8:2 FTUCA: n.d		Chen et al. 2018
Surface reservoir and river water	China		2016	Fluorochemical manufacturing parks (FMPs) in Fuxin	94 multimedia samples collected in the area surrounding two FMPs	C9 PFCA: n.d.-32 ng/L C10 PFCA: n.d.-86 ng/L C11 PFCA: n.d.-51 ng/L C12 PFCA: n.d.-14 ng/L		Chen et al. 2018

Matrix	Country/ Region		Year(s)	Study site	Type of location or samples	Concentration	Mean	Reference
						8:2 FTUCA: n.d.-0.46 ng/L		
Surface sediment	China		2016	Fluorochemical manufacturing parks (FMPs) in Fuxin	94 multimedia samples collected in the area surrounding two FMPs	C9 PFCA: n.d.-0.43 ng/g C10 PFCA: n.d.-0.77 ng/g C11 PFCA: n.d.-9.3 ng/g C12 PFCA: n.d.-0.92 ng/g 8:2 FTUCA: n.d.-0.24 ng/g		Chen et al. 2018
Soil	China		2016	Fluorochemical manufacturing parks (FMPs) in Fuxin	94 multimedia samples collected in the area surrounding two FMPs	C9 PFCA: 0.066-9.9 ng/g C10 PFCA: 0.046-50 ng/g C11 PFCA: 0.022.-12 ng/g C12 PFCA: n.d.-42 ng/g 8:2 FTUCA: n.d.-2.7 ng/g		Chen et al. 2018
Outdoor settled dust	China		2016	Fluorochemical manufacturing parks (FMPs) in Fuxin	94 multimedia samples collected in the area surrounding two FMPs	C9 PFCA: n.d.-160 ng/g C10 PFCA: n.d.-160 ng/g C11 PFCA: n.d.-96 ng/g C12 PFCA: n.d.-100 ng/g 8:2 FTUCA: n.d.-32 ng/g		Chen et al. 2018
Leaves	China		2016	Fluorochemical manufacturing parks (FMPs) in Fuxin	94 multimedia samples collected in the area surrounding two FMPs	C9 PFCA: n.d.-220 ng/g C10 PFCA: n.d. C11 PFCA: n.d. C12 PFCA: n.d.-56 ng/g 8:2 FTUCA: n.d.		Chen et al. 2018
Air	China		2016	Fluorochemical manufacturing parks (FMPs) in Fuxin	94 multimedia samples collected in the area surrounding two FMPs	C9 PFCA: 9.9-370 pg/m^3 C10 PFCA: n.d.-650 pg/m^3 C11 PFCA: n.d.-220 pg/m^3 C12 PFCA: n.d.-120 pg/m^3 8:2 FTUCA: 7.9-340 pg/m^3		Chen et al. 2018
Air	China		2014	Textile manufacturing plant located in the Yangtze River Delta	34 multimedia samples collected in four workshops	C9 PFCA: 44-49 pg/m^3 C10 PFCA: 99-114 pg/m^3 C11 PFCA: 24-27 pg/m^3 C12 PFCA: n.d.-7 pg/m^3 C13 PFCA: n.d. C14 PFCA: n.d. 8:2 FTOH: 9.7-23.0 pg/m^3 10:2 FTOH: 2.6-2.7 pg/m^3		Heydebreck et al. 2016
WWTP effluent	China		2014	Textile manufacturing plant located in the Yangtze River Delta	34 multimedia samples collected in four workshops	C9 PFCA: 255.7-279.7 ng/L C10 PFCA: 723.9-911.9 ng/L C11 PFCA: 40.6-47.0 ng/L C12 PFCA: 0.65-0.74 ng/L C13 PFCA: n.d. C14 PFCA: n.d. 8:2 FTUCA: 628.0-742.0 ng/L 10:2 FTUCA: 41.6-52.0 ng/L		Heydebreck et al. 2016
WWTP effluent – suspended particulate matter	China		2014	Textile manufacturing plant located in the Yangtze River Delta	34 multimedia samples collected in four workshops	C9 PFCA: 15.7-18.2 ng/L C10 PFCA: 144.3-153.0 ng/L C11 PFCA: 17.6-18.4 ng/L C12 PFCA: 0.93-1.03 ng/L C13 PFCA: 1.86-1.96 ng/L C14 PFCA: 0.32-0.36 ng/L		Heydebreck et al. 2016

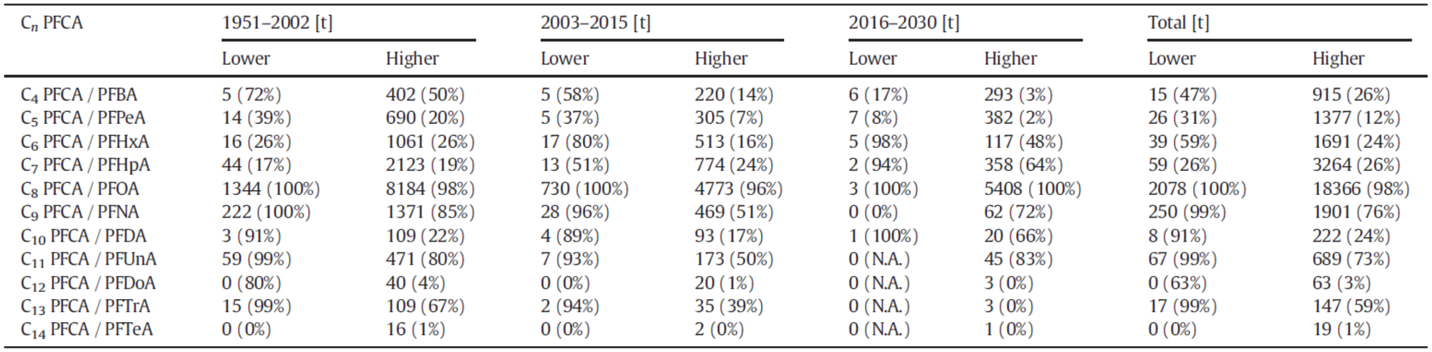
Matrix	Country/ Region		Year(s)	Study site	Type of location or samples	Concentration	Mean	Reference
						8:2 FTUCA: 37.6-48.4 ng/L 10:2 FTUCA: 37.0-40.0 ng/L		
River	China		2014	Textile manufacturing plant located in the Yangtze River Delta	34 multimedia samples collected in four workshops	C9 PFCA: 2.56-2.96 ng/L C10 PFCA: 3.06-3.82 ng/L C11 PFCA: 2.68-3.20 ng/L C12 PFCA: 0.15-0.20 ng/L C13 PFCA: n.d. C14 PFCA: n.d. 8:2 FTUCA: n.d. 10:2 FTUCA: n.d.		Heydebreck et al. 2016
River – suspended particulate matter	China		2014	Textile manufacturing plant located in the Yangtze River Delta	34 multimedia samples collected in four workshops	C9 PFCA: 0.49-0.60 ng/L C10 PFCA: 1.35-1.86 ng/L C11 PFCA: 2.90-3.68 ng/L C12 PFCA: 0.96-1.14 ng/L C13 PFCA: 1.43-1.64 ng/L C14 PFCA: 0.56-0.82 ng/L 8:2 FTUCA: n.d. 10:2 FTUCA: n.d.		Heydebreck et al. 2016
Water	South Korea		2010-2012	Nakdong River	3 sampling sites in a river located in a highly industrialized area	C9 PFCA: 0.83-4.49 ng/L C10 PFCA: 0.53-4.80 ng/L C11 PFCA: 0.28-1.13 ng/L C12 PFCA: 0.13-0.33 ng/L	C9 PFCA: 2.32 ng/L C10 PFCA: 2.13 ng/L C11 PFCA: 0.59 ng/L C12 PFCA: 0.20 ng/L	Lam et al. 2014
Water	South Korea		2010-2012	Yeongsan River	3 sampling sites in a river located in a highly industrialized area	C9 PFCA: 0.54-1.08 ng/L C10 PFCA: 0.14-1.10 ng/L C11 PFCA: 0.13-0.73 ng/L C12 PFCA: 0.10-0.31 ng/L	C9 PFCA: 0.85 ng/L C10 PFCA: 0.64 ng/L C11 PFCA: 0.41 ng/L C12 PFCA: 0.21 ng/L	Lam et al. 2014
Sediment	South Korea		2010-2012	Nakdong River	3 sampling sites in a river located in a highly industrialized area	C9 PFCA: n.d.-0.03 ng/g dw C10 PFCA: 0.02-0.07 ng/g dw C11 PFCA: 0.03-0.08 ng/g dw C12 PFCA: 0.07-0.08 ng/g dw	C9 PFCA: 0.01 ng/g dw C10 PFCA: 0.05 ng/g dw C11 PFCA: 0.06 ng/g dw C12 PFCA: 0.08 ng/g dw	Lam et al. 2014
Sediment	South Korea		2010-2012	Yeongsan River	3 sampling sites in a river located in a highly industrialized area	C9 PFCA: 0.09-0.15 ng/g dw C10 PFCA: 0.03-0.04 ng/g dw C11 PFCA: 0.02-0.04 ng/g dw C12 PFCA: 0.06-0.08 ng/g dw	C9 PFCA: 0.12 ng/g dw C10 PFCA: 0.03 ng/g dw C11 PFCA: 0.03 ng/g dw C12 PFCA: 0.07 ng/g dw	Lam et al. 2014
Soil	South Korea		2017	Industrial complexes (chemical, textile, electronics and metal)	33 soil samples collected from industrial complexes from major industrial areas	C9 PFCA: n.d.-1.52 ng/g dw C10 PFCA: 0.086-1.73 ng/g dw C11 PFCA: n.d.-1.06 ng/g dw C12 PFCA: n.d.-2.10 ng/g dw C13 PFCA: n.d.-0.952 ng/g dw C14 PFCA: n.d.-0.977 ng/g dw	C9 PFCA: 0.387 ng/g dw C10 PFCA: 0.553 ng/g dw C11 PFCA: 0.382 ng/g dw C12 PFCA: 0.435 ng/g dw C13 PFCA: 0.167 ng/g dw C14 PFCA: 0.130 ng/g dw	Sim et al. 2021
Suspended particulate matter	Germany		2005-2019	River Mulde located downstream of a large industrial park	Samples from riverine sampling sites of the Greman Environmental Specimen Bank collected between 2005 and 2019	C9 PFCA: 0.056-0.647 µg/kg dw C10 PFCA: 0.809-3.492 µg/kg dw C11 PFCA: 0.136-1.804 µg/kg dw C12 PFCA: <0.05-2.319 µg/kg dw C13 PFCA: <LOQ C14 PFCA: <LOQ C15 PFCA: <LOQ C16 PFCA: <LOQ		Göckener et al. 2022

Matrix	Country/ Region		Year(s)	Study site	Type of location or samples	Concentration	Mean	Reference
						8:2 diPAP : 2,537-44,418 µg/kg dw 8:2 FTCA: <LOQ		
Water	Japan		2010	Rivers located in the Hyogo prefecture	Samples from 41 rivers, including a site downstream of a perfluorinated compounds production facility	C9 PFCA: <0.5-39 ng/L C10 PFCA: <0.5-47 ng/L C11 PFCA <0.5-39 ng/L C12 PFCA: <0.5-4.1 ng/L		Takemine et al. 2014
Water	Japan		2011	Samondogawa River	Sample from a location downstream of a perfluorinated compounds production facility	C9 PFCA: 12 ng/L C10 PFCA: 3.5 ng/L C11 PFCA <1.5 ng/L C12 PFCA: <0.5 ng/L		Takemine et al. 2014
Water	Japan		2012	Samondogawa River	Sample from a location downstream of a perfluorinated compounds production facility	C9 PFCA: 8.1 ng/L C10 PFCA: 2.7 ng/L C11 PFCA <0.5 ng/L C12 PFCA: <0.5 ng/L		Takemine et al. 2014
Water	China		2021	Taihu Lake	32 water samples collected at various locations, including in proximity to industrial areas	C9 PFCA: 7.75-63.8 ng/L C10 PFCA: 4.55-118 ng/L	C9 PFCA: 15.9 (±11.4) ng/L C10 PFCA: 17.7 (±22.6) ng/L	Yu et al. 2022
Air	China		2013	Tianjin City	Air samples collected collected using SIP disk PAS, installed at various sites, including in urban areas	C9 PFCA: 8.57-23.7 pg/m^3 C10 PFCA: 1.47-7.67 pg/m^3 C11 PFCA: 1.13-3.23 pg/m^3 C12 PFCA: 0.31-2.11 pg/m^3 8:2 FTOH: 43.9-89.9 pg/m^3 10:2 FTOH: 14.1-39.8 pg/m^3		Yao et al. 2016
Wastewater	China		Not specified	Electroplating industrial parks	23 water samples collected in production workshops and treatment units	C9 PFCA: 2.4-714.5 ng/L C10 PFCA: 87-259 ng/L C11 PFCA: concentration not specified		Jiawei et al. 2019
Sediment	Norway		2018-2019	PFAS-coated paper products factory	Sediment samples collected downstream of the factory		C9 PFCA: 6.9 (±6.6) µg/kg dw C10 PFCA: 69.4 (±66.2) µg/kg dw C11 PFCA: 19.9 (±18.5) µg/kg dw C12 PFCA: 21.0 (±18.3) µg/kg dw C13 PFCA: 3.2 (±2.4) µg/kg dw C14 PFCA: 23.3 (±20.1) µg/kg dw C15 PFCA: 1.5 (±1.1) µg/kg dw C16 PFCA: 2.8 (±2.3) µg/kg dw 8:2 FTSA: 253 (±212) µg/kg dw 10:2 FTSA: 472 (±269) µg/kg dw 12:2 FTSA: 370 (±182) µg/kg dw 14:2 FTSA: 106 (±68.2) µg/kg dw	Langberg et al. 2020
Industrial WWTPs influent	South Korea		2018-2019	Industrial complex containing 77 industrial plants producing ceramics, electronic equipment, electroplated metals, polymers, textiles, and other items	79 samples from influent wastewater	C9 PFCA: n.d.- 13.8 ng/L C10 PFCA: n.d.-<LOQ ng/L C11 PFCA: n.d.-14.7 ng/L C12 PFCA: n.d.-26.0 ng/L C13 PFCA: n.d.-15.2 ng/L 8:2 FTSA: n.d.-2.35 ng/L	C9 PFCA: 13.5 ng/L C10 PFCA: < LOQ C11 PFCA: 14.7 ng/L C12 PFCA: 26.0 ng/L C13 PFCA: 15.2 ng/L 8:2 FTSA: 2.35 ng/L	Kim et al. 2021

Matrix	Country/ Region		Year(s)	Study site	Type of location or samples	Concentration	Mean	Reference
Industrial WWTPs effluent	South Korea		2018-2019	Industrial complex containing 77 industrial plants producing ceramics, electronic equipment, electroplated metals, polymers, textiles, and other items	66 samples from effluent wastewater	C9 PFCA: n.d.- 20.9 ng/L C10 PFCA: n.d.-9.5 ng/L C11 PFCA: <LOQ C12 PFCA: n.d.-40.2 ng/L 8:2 FTSA: n.d.-9.3 ng/L	C9 PFCA: 14.5 ng/L C10 PFCA: 8.9 ng/L C11 PFCA: <LOQ C12 PFCA: 35.0 ng/L 8:2 FTSA: 9.3 ng/L	Kim et al. 2021
Municipal WWTP influent	South Korea		2018-2019	Municipal WWTPs receiving treated wastewater from an industrial complex	Sanples from two municipal WWTPs plants	C9 PFCA: <LOQ ng/L C10 PFCA: n.d. C11 PFCA: n.d. C12 PFCA: <LOQ C13 PFCA: n.d. 8:2 FTSA: n.d.		Kim et al. 2021
Municipal WWTP effluent	South Korea		2018-2019	Municipal WWTPs receiving treated wastewater from an industrial complex	Sanples from two municipal WWTPs plants	C9 PFCA: n.d.- 7.86 ng/L C10 PFCA: <LOQ C11 PFCA: n.d. C12 PFCA: n.d. C13 PFCA: n.d. 8:2 FTSA: n.d.		Kim et al. 2021

Abbreviations: n.d., not detected; LOD, limit of detection; dw, dry weight; diPAP, polyfluoroalkyl phosphoric acid diesters; FTA, fluorotelomer acrylate; FTCA, fluorotelomer carboxylic acids; FTUCA, fluorotelomer unsaturated carboxylates; FTOH, fluorotelomer alcohols; FTSA, fluorotelomer sulfonate; MQL, method quantification limit; MRL, method reporting limit; PFCA, perfluorocarboxylic acid.

Table 3. Estimated global cumulative emissions of C4–C14 PFCA homologues (1951–2030) from quantified sources in tonnes* (Wang et al. 2014)



C_n PFCA	1951–2002 [t]		2003–2015 [t]		2016–2030 [t]		Total [t]	
	Lower	Higher	Lower	Higher	Lower	Higher	Lower	Higher
C_4 PFCA / PFBA	5 (72%)	402 (50%)	5 (58%)	220 (14%)	6 (17%)	293 (3%)	15 (47%)	915 (26%)
C_5 PFCA / PFPeA	14 (39%)	690 (20%)	5 (37%)	305 (7%)	7 (8%)	382 (2%)	26 (31%)	1377 (12%)
C_6 PFCA / PFHxA	16 (26%)	1061 (26%)	17 (80%)	513 (16%)	5 (98%)	117 (48%)	39 (59%)	1691 (24%)
C_7 PFCA / PFHpA	44 (17%)	2123 (19%)	13 (51%)	774 (24%)	2 (94%)	358 (64%)	59 (26%)	3264 (26%)
C_8 PFCA / PFOA	1344 (100%)	8184 (98%)	730 (100%)	4773 (96%)	3 (100%)	5408 (100%)	2078 (100%)	18366 (98%)
C_9 PFCA / PFNA	222 (100%)	1371 (85%)	28 (96%)	469 (51%)	0 (0%)	62 (72%)	250 (99%)	1901 (76%)
C_{10} PFCA / PFDA	3 (91%)	109 (22%)	4 (89%)	93 (17%)	1 (100%)	20 (66%)	8 (91%)	222 (24%)
C_{11} PFCA / PFUnA	59 (99%)	471 (80%)	7 (93%)	173 (50%)	0 (N.A.)	45 (83%)	67 (99%)	689 (73%)
C_{12} PFCA / PFDoA	0 (80%)	40 (4%)	0 (0%)	20 (1%)	0 (N.A.)	3 (0%)	0 (63%)	63 (3%)
C_{13} PFCA / PFTrA	15 (99%)	109 (67%)	2 (94%)	35 (39%)	0 (N.A.)	3 (0%)	17 (99%)	147 (59%)
C_{14} PFCA / PFTeA	0 (0%)	16 (1%)	0 (0%)	2 (0%)	0 (N.A.)	1 (0%)	0 (0%)	19 (1%)

* Numbers in brackets indicate the percentage of emissions from direct sources. The percentage of emissions from indirect sources can be calculated as 100% minus these percentages. N.A. – not applicable.

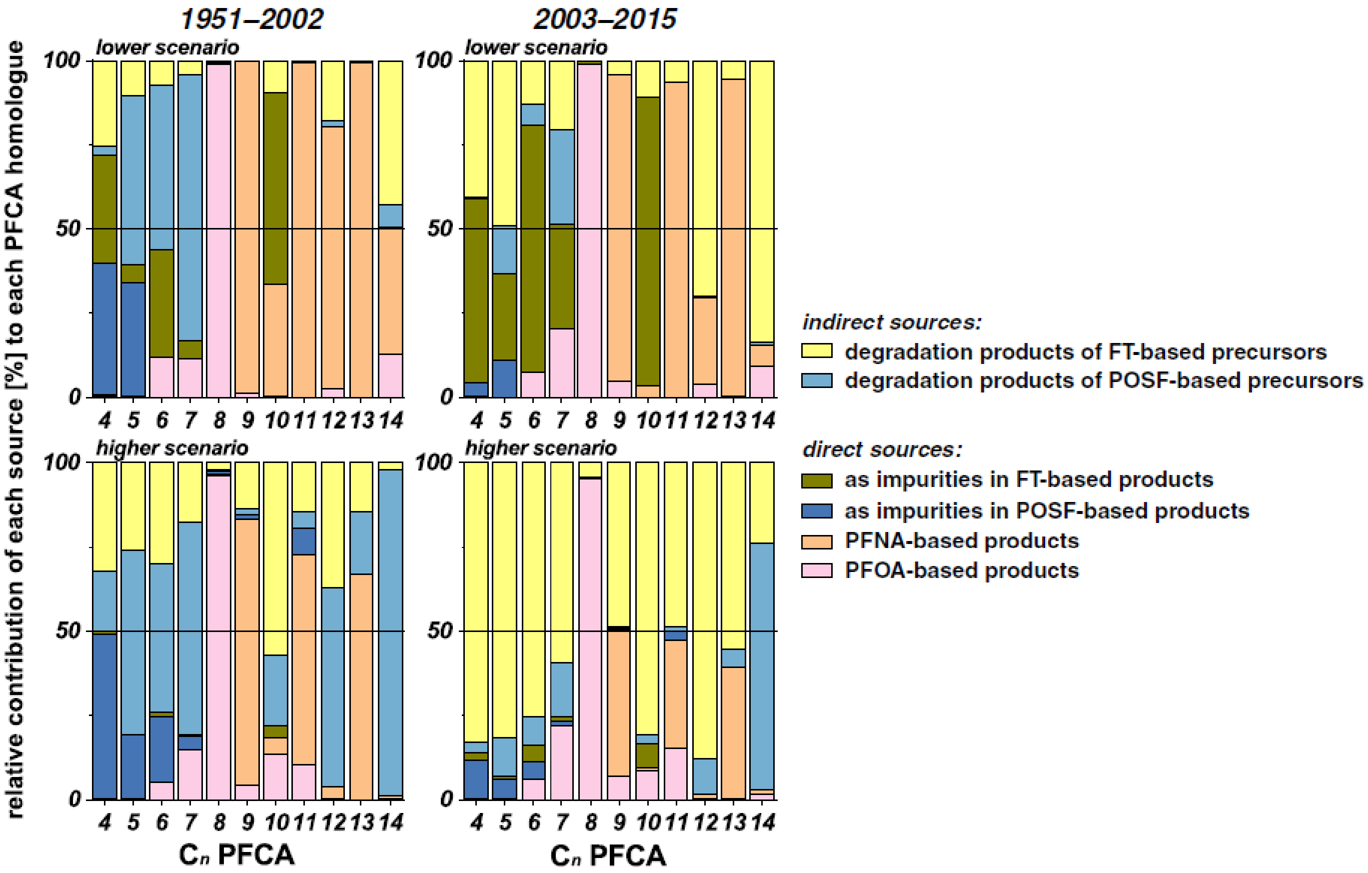


Figure 1. Relative contributions of each source to estimated total global emissions from all quantified sources for individual C4–C14 PFCA homologues in 1951–2002 (pre-phase-out) and 2003–2015 (transition after phase-out) (Wang et al. 2014)

2.2.1 Persistence

5. Examples of conditions considered not environmentally relevant include a study where 30–35% photolysis was observed for C10 PFCA at high altitudes (2500 m and 4200 m) when exposed to solar irradiation for 106 d (Taniyasu et al. 2013) and a study where C9 – C18 PFCAs underwent 38% defluorination in river water using electrooxidation (Barisci and Suri 2020).

2.2.2 Bioaccumulation

6. Octanol-water partitioning coefficient (log *K*ow) values are used to describe the partitioning from water to lipids and are also traditionally used as an indicator for bioaccumulation. Modelled log *K*ow values are available but empirical log *K*ow values are not available for long-chain PFCAs. However, meaningful log *K*ow values cannot be reliably measured or modelled for surface-active and ionizing substances such as long-chain PFCAs. Wang et al. (2011) modelled log *K*ow values for the neutral form of C9 – C14 PFCAs with log *K*ow values that ranged from 5.9 to 8.9 and which represent high bioaccumulation potential. However, Wang et al. (2011) cautioned that these values have high and unquantifiable uncertainties due to the modelling estimates being highly dependent on the chosen conformation of the neutral and anionic forms. Recent studies point to a p*K*a between 0 and 1 for PFCAs suggesting that long-chain PFCAs are almost completely ionized at environmental pH values and thus, the neutral form is unlikely to be present in the environment (Wang et al. 2011; Ng and Hungerbuhler 2014). Rather, long-chain PFCAs tend to migrate to the interface of the organic (lipid) and aqueous phases rather than partition between the two phases (Houde et al. 2006b; OECD 2002). Some portions of the perfluorinated molecule can interact with phospholipids (Armitage et al. 2012; Dassuncao et al. 2019; Droge et al. 2019) but most studies show that, at the organismal level, protein-rich tissues (i.e., yolk, liver, and blood), rather than lipids, are the primary repositories for long-chain PFCAs. The transport of these substances into cells results in binding to fatty acid-binding proteins and lipoproteins/albumin, and then sequestering into protein-rich tissues (Jones et al. 2003; Bischel et al. 2010; Woodcroft et al. 2010; Bischel et al. 2011; Ng and Hungerbuhler 2013; Cheng and Ng 2018; Zhong et al. 2019). On this basis, it is inappropriate to use log *K*ow as a descriptor for bioaccumulation and for predictive purposes (e.g., bioaccumulation models) for long-chain PFCAs (OECD 2002; Conder et al. 2008). Instead, empirical bioaccumulation data, rather than modelled data, is more relevant.

7. Both bioconcentration and bioaccumulation empirical data are available for some long-chain PFCAs. Laboratory-derived bioconcentration factors (BCF, L/kg) and bioaccumulation factors (BAF, L/kg) have been reported (up to C18 PFCA) in three freshwater fish species (i.e., zebrafish (*Danio rerio*), common carp (*Cyprinus carpio* L.) and rainbow trout (*Oncorhynchus mykiss*)) and one green mussel species (*Perna viridis*) and for saltwater species blackrock fish (*Sebastes schlegeli*). Zebrafish embryos exposed to 1 mg/L C9 PFCA for 144 hours post-fertilization had BCFs that ranged from 582 – 638 (Menger et al. 2020). Steady-state whole-body BCFs in adult zebrafish ranged from 1202 (C9 PFCA) to 257 039 (C14 PFCA) and steady-state liver BCFs ranged from 1514 (C9 PFCA) to 363 078 (C14 PFCA) (Chen et al. 2016). In common carp, whole body BCFs were determined for C11 PFCA (2300 – 3700), C12 PFCA (10 000 – 16 000), C13 PFCA (16 000 – 17 000), C16 PFCA (4700 – 4800) and C18 PFCA (320 – 430) (Inoue et al. 2012). For juvenile rainbow trout, steady-state whole-body and liver BCFs were determined for C10 – C14 PFCAs after 12 d of exposure followed by 33 d of depuration (Martin et al. 2003b). Steady-state whole-body BCFs ranged from 450 (C10 PFCA) to 23 000 (C14 PFCA). Steady-state liver BCF values ranged from 1100 (C10 PFCA) to 30 000 (C14 PFCA). Steady-state carcass BAFs for C10 – C13 PFCAs ranged from 0.04 to 1.0 in juvenile rainbow trout after 34 d exposure followed by a 41 d depuration period (Martin et al. 2003a). For market-size rainbow trout, the BAF for C9 PFCA was < 0.4 after a 28 d exposure followed by a 28 d depuration period (Goeritz et al. 2013). For the green mussel, BAFs were determined for C9 and C10 PFCAs after 56 d exposure at 1 µg/L and 10 µg/L (Liu et al. 2011a). BAFs for green mussel ranged from 109 to 144 (C9 PFCA) and 464 to 838 (C10 PFCA). Serum and BCFs for blackrock fish (*Sebastes schlegeli*) ranged from 4321 to 5239 and 667 to 811 (C10), respectively, and 13 553 to 16 370 and 1070 to 1345 (C11), respectively (Jeon et al. 2010). In summary, laboratory BCF/BAF values were variable depending on the species and age of the test organism. BCF and BAF values generally increased from C9 PFCA (<0.4 – 1514) to C14 PFCA (17 000 – 363 078) and then decreased for C16 to C18 PFCAs (20 – 4800).

8. Field-derived BCFs and BAFs in freshwater and marine aquatic organisms have been reported up to C15 PFCA. For example, whole-body BAFs were determined in 4-year old lake trout (*Salvelinus namaycush*) (Great Lakes, Canada) for C9 PFCA (1259 – 6309) and C10 PFCA (5011 to 19 952) (Furdui et al. 2007). BAFs in European chub (*Leuciscus cephalus*) (Orge River, France) had liver BAFs from 79 (C9 PFCA) to 501 187 L/kg (C12 PFCA) and plasma BAFs from 631 (C9 PFCA) to 5 011 872 L/kg (C12 PFCA) (Labadie and Chevreuil 2011).

BAFs were determined for common carp collected from a drainage canal near a sewage treatment plant outfall (Tokyo, Japan) with liver BAFs that ranged from 69 (C9 PFCA) to > 26 000 (C13 PFCA) and kidney BAFs that ranged from 2600 (C9 PFCA) to > 40 000 (C13 PFCA) (Murakami et al. 2011). BAFs were reported for common carp, tilapia (*Tilapia aurea*), snakehead (*Ophicephalus argus*), and catfish (*Clarias fuscus*) from the Pearl River Delta (China) (Pan et al. 2014). Across all species, liver BAFs for C9 – C11 PFCAs ranged from 501 to 100 000 with increasing BAF from C9 to C11. This is consistent with other studies that observed that bioaccumulation increases with fluorinated carbon chain length (Conder et al. 2008). Whole body BCFs for European perch (*Perca fluviatilis*) from Lake Halmsjön (Sweden) ranged from 42 to 54 L/kg (C9 PFCA) and 140 to 220 L/kg (C10 PFCA) (Ahrens et al. 2015). Whole-body BAFs were determined in Chinese icefish (*Neosalanx tangkahkeii taihuensis*), a top predator in Lake Chaohu (China) where values ranged from 93 (C13 PFCA) to 2041 L/kg (C9 PFCA) (Pan et al. 2019). At Baiyangdian Lake (China), BAFs were measured in five freshwater fish species (grass carp (*Ctenopharyngodon idellus*), goldfish (*Carassius auratus*), common carp, silver carp (*Hypophthalmichthys molitrix*), and northern snakehead (*Channa argus*)). Across species, BAFs were 3.9 to 1892 (C9 PFCA), 45 to 8672 (C10 PFCA), 26 to 30 475 (C11 PFCA), and 91 to 9874 mL/g ww (C12 PFCA) (Liu et al. 2019a). C9 PFCA BCFs were estimated in female crabs (species unknown, collected from South Korean fish markets) with BCF values of 440 in legs, 660 in eggs, 879 in body, and 1040 in offal (Choi et al. 2020). BAFs were determined for eel (*Anguilla Anguilla*; collected from 21 rivers, lakes and canals in the Netherlands) for C9 PFCA (105 to 1380 L/kg ww) and C10 PFCA (331 to 5623 L/kg ww) (Kwadijk et al. 2010). BAFs were determined for a variety of fish, crab, and snail species in Baiyangdian Lake (China) (Zhou et al. 2012). Across all species, BAFs were determined for C9 PFCA (59 to 60 L/kg ww), C10 PFCA (1230 to 69 183 L/kg ww) and C11 PFCA (589 to 7762 L/kg ww). BAFs were determined in a variety of copepod, mysid, and shrimp species from a macrotidal estuary in Aquitaine (France) (Munoz et al. 2019). Across all species, BAFs were determined for C9 – C11 PFCA (631 to 12 589 L/kg ww). BCFs were reported in various fish, crab, gastropod, and bivalve species collected along the western coast of Korea (Naile et al. 2013). Across all species, whole-body BCFs for C9 – C11 PFCAs ranged from 7 to 269 L/kg ww. BAFs were determined for plankton species in Taihu Lake (China) that ranged from 462 (C10 PFCA) to 17788 L/kg ww (C12 PFCA) (Fang et al. 2014). BAFs were determined for herring (*Clupea* sp.) and sprat (*Sprattus* sp.) collected from the Baltic Sea where BAFs for herring ranged from > 224 (C15 PFCA) to 218 776 L/kg ww (C11 PFCA) and, for sprat, BAFs ranged from > 59 (C15 PFCA) to 158 489 L/kg ww (C11 PFCA) (Gebbink et al. 2016). BAFs were determined for various shrimp, snail, and fish species in Lake Chaohu (China) that ranged from 118 (C9 PFCA) to 12 370 L/g (C11 PFCA) (Liu et al. 2019b). In summary, field-derived BCFs and BAFs were variable depending on the species and ranged from 3.9 (C9 PFCA) to 5 011 872 (C12 PFCA). Field-derived BCFs and BAFs also generally increased from C9 PFCA to C14 PFCA and then declined at C15 PFCA (> 59 – 224).

9. Field biomagnification or trophic magnification studies on long-chain PFCAs (up to C16 PFCA) that focused on multiple fish species and/or top predator species (i.e., birds or terrestrial/marine mammals) show higher biomagnification potential. Biomagnification factor and trophic magnification factor (TMF) values above one are considered bioaccumulative. For example, a marine food web (Liaodong Bay, China) with black-tailed gulls (*Larus crassirostris*) as the top predator species had TMFs that ranged from 1.78 to 4.88 for C9 – C14 PFCAs, based on whole body concentration estimates using muscle and liver data (Zhang et al. 2015). A eutrophic freshwater food web (Taihu Lake, China) with egrets and carnivorous fish as the top predator species had TMFs that ranged from 2.1 to 3.7 for C9 – C12 PFCAs (Xu et al. 2014). The Orge River (France) foodweb with eight freshwater fish species as top predators but with varying feeding behaviours (e.g., benthic, bentho-pelagic, omnivorous, carnivorous) had BMFs that ranged from 0.3 to 25.2 and TMFs that ranged from 1.5 to 3.0 (Simonnet-Laprade et al. 2019a). Five riverine foodwebs (France) with chub (*Squalius cephalus*) and common barbel (*Barbus barbus*) as top predator species had TMFs that ranged from 0.9 to 14.9 for C9 – C14 PFCAs (Simonnet-Laprade et al. 2019b). A marine food web in the western Canadian Arctic with ringed seal (*Phoca hispida*) and beluga whales (*Delphinapterus leucas*) as top predator species had TMFs for C9 – C11 PFCAs that ranged from 3.8 to 19.8 (Tomy et al. 2009). In other food webs, TMFs ranged from 1.00 to 8.29 for C9 – C13 PFCAs in the Lake Ontario (Canada) freshwater food web, in the Lake Taihu (China) freshwater food web, in the Hudson Bay (Canadian Arctic) marine food web, and in the subtropical food web of the Mai Po Marshes Nature Reserve (Hong Kong) (Martin et al. 2004b; Kelly et al. 2009; Loi et al. 2013; Fang et al. 2014). In East Greenland, mean BMFs for C9 – C16 PFCAs were above one for the top predator species, polar bear (*Ursus maritimus*) consuming ringed seal (*Pusa hispida*). Mean BMFs ranged from 1 to 10 for ringed seal blubber to polar bear liver for C9 – C16 PFCAs and mean BMFs ranged from 100 to 10 000 for ringed seal liver to polar bear liver for C9 – C13 PFCAs (Boisvert et al. 2019). In the Canadian Arctic, geometric mean BMFs calculated for ringed seal liver to polar bear liver for C9 – C15 PFCAs ranged from 2.2 (C13 PFCA) to 56 (C9 PFCA) (Butt et al. 2008). A western Canadian Arctic food web with seal as the top predator

species had BMFs for C10 – C12 PFCA that ranged from 0.8 to 3.1 (Powley et al. 2008). From the Yukon, Northwest Territories, and Nunavut (Canada), BMFs and TMFs were determined for two barren ground caribou (*Rangifer tarandus groenlandicus*) herds with wolf (*Canis lupus*) as the top predator species (Müller et al. 2011). Whole-body caribou/wolf BMFs for C9 – C13 PFCAs ranged from 0.8 to 5.4 and whole-body caribou/wolf TMFs ranged from 1.9 to 2.9. BMFs were determined for the bottlenose dolphin (*Tursiops truncatus*) food web at Charleston (South Carolina, US) and Sarasota Bay (Florida, US) (Houde et al. 2006a). In the Charleston food web, BMFs and TMFs for C9 – C11 PFCAs ranged from 0.1 to 8.8. In Sarasota Bay food web, BMFs for C12 PFCA ranged from 0.1 to 2.0. The Barents Sea (Svalbard) ice edge food web with predator species such as black guillemot (*Cepphus grylle*) and glaucous gull *(Larus hyperboreus*) had C9 PFCA BMFs that ranged from 8.76 to 11.6 (Haukås et al. 2007). Lake trout (*Salvelinus namaycush*), as top predator species in Lake Ontario (Canada), had adjusted whole-body BMFs (i.e., a diet-weighted BMF that accounted for the abundance of each of three forage fish species in the lake trout diet) that ranged from 1.6 to 3.4 for C9 – C14 PFCAs (Martin et al. 2004b). A temperate macrotidal estuary foodweb (Gironde Estuary, France) with seabass (i.e., common seabass, *Dicentrarchus labrax*; spotted seabass, *Dicentrarchus punctatus*) and meagre (*Argyrosomus regius*) as top predator species had TMF values that ranged from 0.88 to 1.3 for C9 – C14 PFCAs (Munoz et al. 2017b). In summary, TMF values ranged from 0.3 to 19.8 and BMF values ranged from 0.1 to 25.2 with top predator species (e.g., black-tailed gulls, egrets, carnivorous fish, ringed seal, beluga whales, polar bears and wolves) having values consistently above 1.

2.2.4 Potential for long-range environmental transport

Table 4. Environmental concentrations of long-chain PFCAs and their related compounds in locations distant from sources

Location	Compartment / Species	Concentration	Reference
Arctic			
North Atlantic and Canadian Archipelago	Air	8:2 FTOH: 5.8 – 26 pg/m^3 10:2 FTOH: 1.9 – 17 pg/m^3	Shoeib et al. 2006
Canadian and Norwegian Arctic	Air	8:2 FTOH: <0.065 – 21 pg/m^3 10:2 FTOH: <0.015 – 8.7 pg/m^3 C9 – C18 PFCA: <0.0063 – 0.77 pg/m^3	Wong et al. 2018
Japan Sea to the Arctic Ocean	Gas-phase; Particle-phase	FTOH (10:2, 12:2 and 10:2): 1.8 – 47 pg/m^3; 0.1 – 2.5 pg/m^3	Cai et al. 2012a
Livingston Island (Antarctica)	Snow	C9 – C14 PFCA: ND – 0.04 ng/L	Casal et al. 2017
Lake Hazen (Nunavut, Canada)	Snowpack	C9 – C14 PFCA: < 0.002 to 3.1 ng/L	MacInnis et al. 2019
Oceans			
Atlantic, Indian and Pacific Oceans	Depth of 20 – 160 m	C9 PFCA: ND – 1.15 ng/L C10 PFCA: ND – 2.19 ng/L	Gonzalez-Gaya et al. 2019
Greenland Sea and East Atlantic Ocean	Surface water	C9 PFCA: <0.012 – 0.039 ng/L C10 PFCA: <0.021 ng/L C11 PFCA: ND – <0.013 ng/L C12 PFCA: <0.025 ng/L	Zhao et al. 2012
South Shetland Islands (Maritime Antarctica)	Coastal surface seawater	C16 PFCA: <0.007.5 – 0.0082 ng/L	Cai et al. 2012b
Livingston Island (Antarctica)	Seawater	C9 – C14 PFCA: ND – 0.11 ng/L	Casal et al. 2017
Biota			
East Greenland	Polar bear – liver; blood; brain; muscle; adipose tissue	C15 PFCA: 0.73 – 0.89 ng/g ww; 1.22 – 1.48 ng/g ww; 9.9 – 10.9 ng/g ww; 0.58 – 0.72 ng/g ww; 0.5 – 0.64 ng/g ww	Greaves et al. 2012
East Greenland	Polar bear – liver	C16 PFCA: 0.1 – 0.2 ng/g ww C18 PFCA: 0.2 – 0.4 ng/g ww	Boisvert et al. 2019
	Ringed seal (*Phoca hispida*) – liver	C16 PFCA: ND – 0.2 ww C18 PFCA: 0.1 – 0.5 ng/g ww	

Location	Compartment / Species	Concentration	Reference
Yukon (Canada)	Caribou (*Rangifer tarandus groenlandicus*) – liver	C9-C13 PFCA: < 0.5 – 3.20 ng/g ww	Katz et al. 2009; Müller et al. 2011
	Wolf (*Canis lupus*) – liver	C9-C13 PFCA: 0.19 – 7.79 ng/g ww	
East and South Greenland	Reindeer – liver	C9-C13 PFCA: ND – 2.06 ng/g ww	Bossi et al. 2015
East and South Greenland	Muskox – liver	C9-C13 PFCA: 0.21 – 5.25 ng/g ww	
Antarctica	Weddell seal (*Leptonychotes weddellii*) – liver	C9-C12 PFCAs: < 0.01 – 0.23 ng/g ww	Routti et al. 2015
Antarctica	Adelie penguin (*Pygoscelis adeliae*) – eggs; blood; muscle	C9-C12 PFCA: < 0.1 – 2.5 ng/g ww; < 0.5 ng/ml; < 1.4 ng/g ww	Schiavone et al. 2009; Tao et al. 2006; Bengtson Nash et al. 2010; Llorca et al. 2012
	Gentoo penguin (*Pygoscelis papua*) – eggs; muscle	C9-C12 PFCA: 0.1 – 0.5 ng/g ww; ND – 0.34 ng/g ww	
Canada	Caribou and reindeer (*Rangifer tarandus*) – liver	C9-C13 PFCA: <0.008 – 5.25 ng/g ww	Roos et al. 2021
Greenland		C9-C13 PFCA: <0.01 – 35.25 ng/g ww	
Norway		C9-C13 PFCA: <0.4 – 1.83 ng/g ww	
Sweden		C9-C13 PFCA: <0.17 – 3.30 ng/g ww	

ND = not detected

2.3.1 Environmental monitoring data

Environmental concentrations of long-chain PFCAs

10. Worldwide concentrations of long-chain PFCAs are illustrated in Figure 2 below. Reported concentrations of long-chain PFCAs in biota (bird, fish, invertebrate, mammal, plant, reptile), separated by continent, are illustrated in Figure 3. The list of references used to generate these figures is provided in the Appendix to this document. The detailed reported environmental concentrations of long-chain PFCAs are provided in UNEP/POPS/POPRC.18/INF/z.

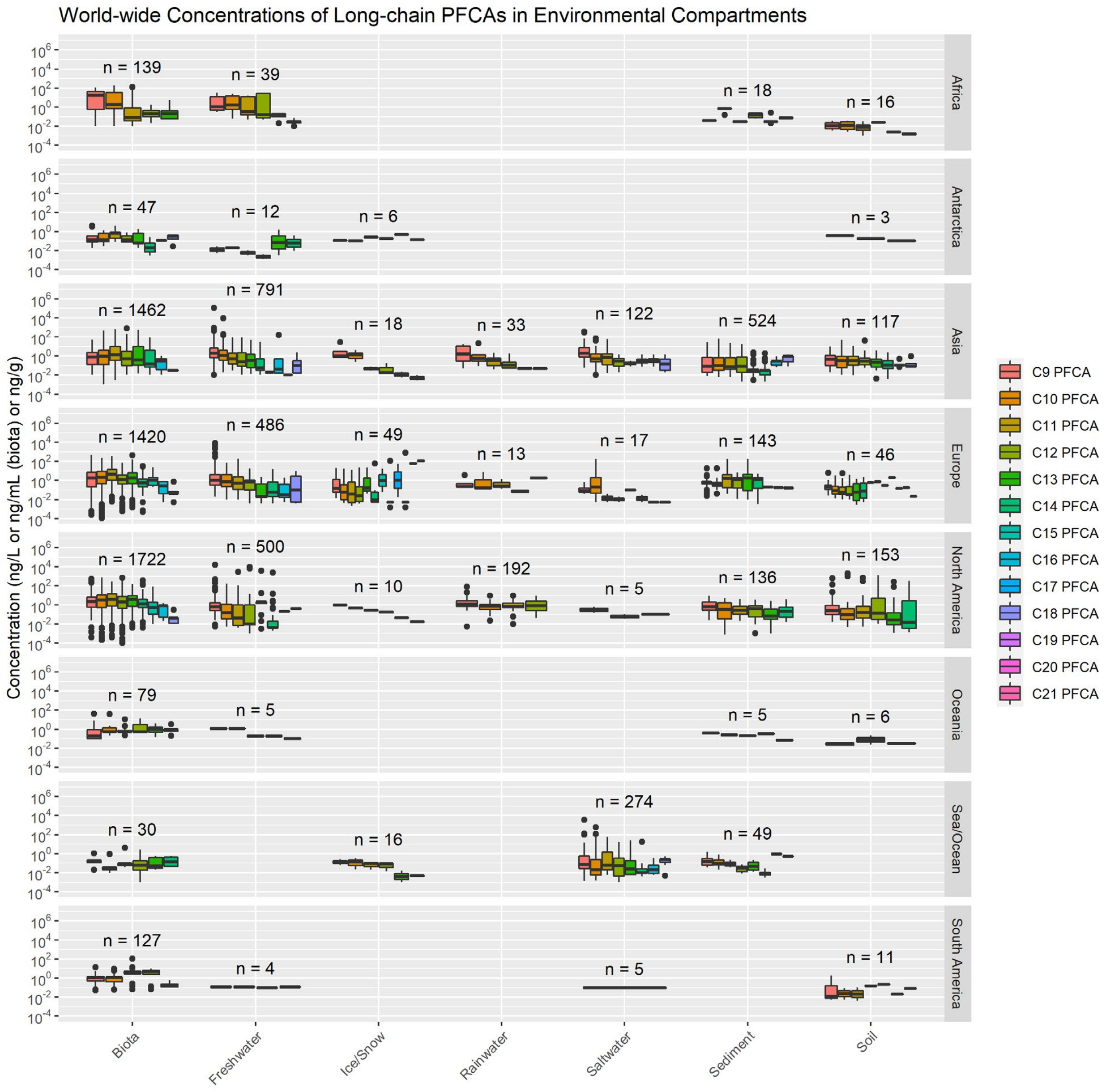


Figure 2. World-wide concentrations of long-chain PFCAs (C9 – C21) in different environmental compartments, by chain length. Tukey box plots are interpreted as follows: the numbers above the bars indicate the number of data points and the lower and upper hinges (edges) of the box represent the first and third quantiles (Q1 and Q3), which are the 25th and 75th percentiles, respectively, while the black horizontal line within the box represents the second quantile, or the 50th percentile (median). The distance between the 25th and 75th percentile is called the interquartile range (IQR). The lower whisker represents the lowest data that are within the Q1 – 1.5 x IQR threshold, and the upper whisker represents the highest data that are within the Q3 + 1.5 x IQR threshold. Data exceeding these thresholds appear as circles. However, if the minimum and maximum are within these thresholds, they represent the lower and upper whiskers and no outliers are present.

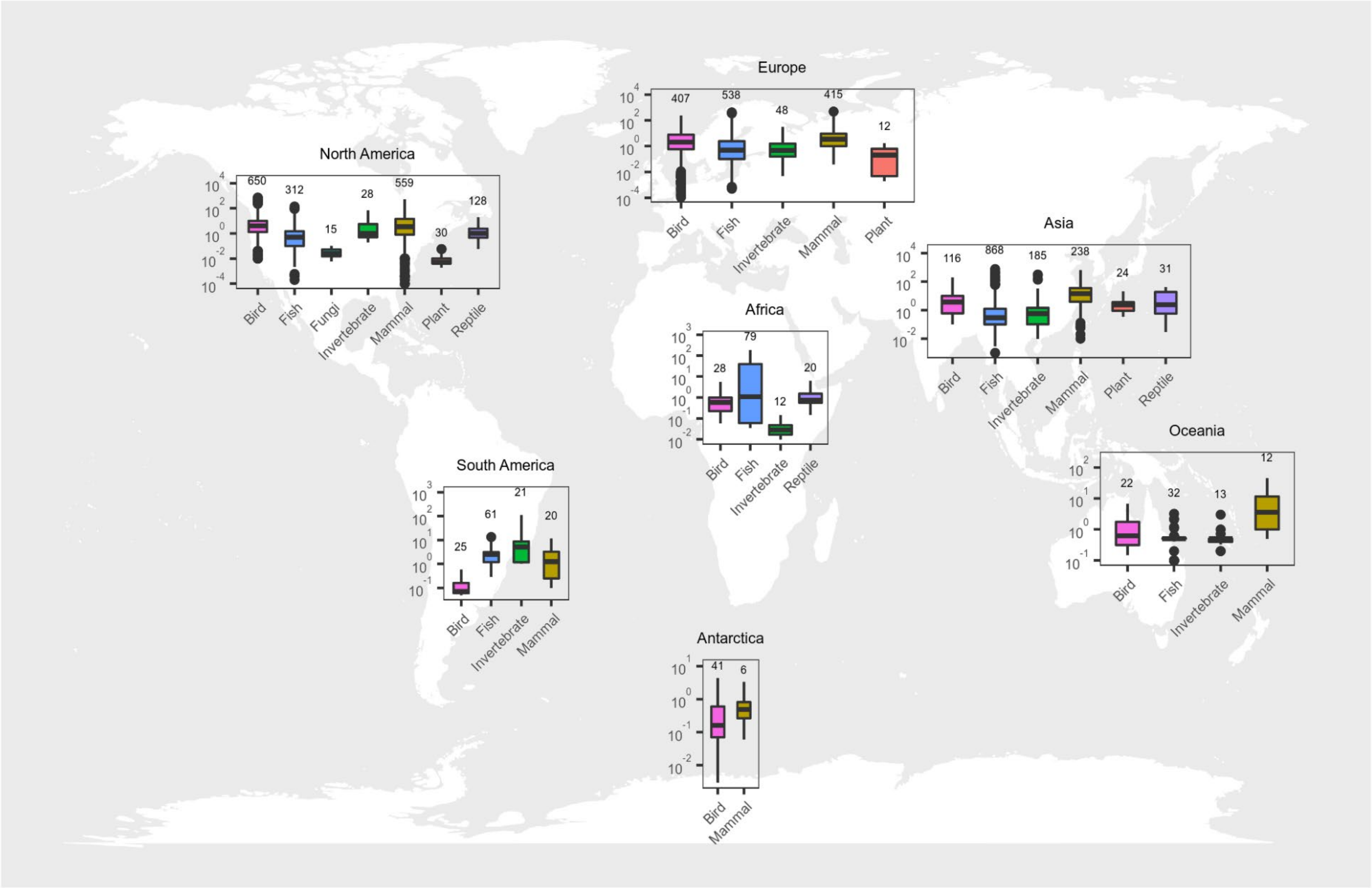


Figure 3. World-wide map representing the concentrations of long-chain PFCAs (C9 – C21) in biota (bird, fish, invertebrate, mammal, plant, reptile, fungi), separated by continent. All measurements are reported in ng/mL or ng/g.

2.3.2 Human exposure

Table 5. Concentrations of long-chain PFCAs in indoor air and dust (units are in ng/g unless otherwise specified)

				LC-PFCA concentrations in ng/g Range (median), detection frequency %							
Media	**Country/ Region**	**Year of sampling (Months)**	**Type of location (n)**	**C9**	**C10**	**C11**	**C12**	**C13**	**C14**	**C15**	**Reference**
Dust	China/ Tianjin	2015 (June-Sept)	Private homes (n=18)	0.96-13.1 (2.36), 100	n.d.-10.8 (2.22), 94	0.51-4.14 (1.91), 100	0.55-7.37 (1.71), 100	NM	NM	NM	Yao et al. 2018
Dust	China/ Tianjin	2015 (June-Sept)	Hotels (n=11)	n.d.-20.2 (2.46), 91	n.d.-1.68 (n.d.), 18	n.d.-0.82 (n.d.), 9	n.d.-0.64 (n.d.), 18	NM	NM	NM	Yao et al. 2018
Air	China/ Tianjin	2015 (June-Sept)	Private homes (n=22)	n.d.-380 pg/m^3 (38.1 pg/m^3), 95	<MDL-57.6 pg/m^3 (13.4 pg/m^3), 100	n.d.-178 pg/m^3 (18.5 pg/m^3), 91	n.d.-20.1 pg/m^3 (6.54 pg/m^3), 91	NM	NM	NM	Yao et al. 2018
Air	China/ Tianjin	2015 (June-Sept)	Hotels (n=19)	n.d.-220 pg/m^3 (13.1 pg/m^3), 95	n.d.-110 pg/m^3 (12.2 pg/m^3), 79	n.d.-142 pg/m^3 (4.92 pg/m^3), 63	n.d.-20.1 pg/m^3 (5.28 pg/m^3), 84	NM	NM	NM	Yao et al. 2018
Dust	USA/Boston, MA	2009	Offices (n=31)	10.9-639 (63.0)[a], 94	5.30-492 (46.5)[a], 97	9.22-373 (19.0)[a], 52	6.56-481 (40)[a], 87	8.67-768 (21.6)[a], 58	9.35-367 (18.6)[a], 71	NM	Fraser et al. 2013
Dust	USA/Boston, MA	2009	Private homes (n=30)	6.21-1420 (10.9)[a], 67	6.97-26.8 (NR), 43	10.8-39.4 (NR), 7	5.09-13.3 (NR), 23	10.3-10.3 (NR), 3	11.2-11.2 (NR), 3	NM	Fraser et al. 2013
Dust	USA/Boston, MA	2009	Vehicles (n=12)	4.95-101 (14.7)[a], 85	5.42-70.1 (8.40)[a], 69	5.24-6.30 (NR), 15	4.96-24.6 (6.76)[a], 77	n.d.-n.d. (NR), 0	14.3-14.3 (NR), 8	NM	Fraser et al. 2013
Dust	USA/Ohio & North Carolina	2000/01	Private homes (n=102) & daycares (n=10)	<DL-263 (7.99), 42.9	<DL-267 (6.65), 30.4	<DL-588 (7.57), 36.6	<DL-520 (7.78), 18.7	NM	NM	NM	Strynar and Lindstrom 2008
Dust	USA/ Wisconsin	2008 (Mar-Apr)	Private homes (n=39)	1.3–280 (12), 100	ND-60 (5.7), 72	ND-48 (3.1), 87	ND-41 (5.0), 95	ND-11 (2.1), 92	ND-24 (3.7), 97	NM	Knobeloch et al. 2012
Dust	Norway/ Oslo	2018 (Feb-May)	Private homes (n=41)	3.9-92 (23), 61	1.1-12 (4.1), 24	n.d.-n.d. (NR), 0	1.4-78 (19), 98	1.1-46 (6.8), 95	1.1-35 (3.3), 7	NM	Haug et al. 2011
Dust	Norway/ Oslo	2016 (Oct)	Hotel (n=2)	<4-<8.3 µg/kg dw	<43-<90 µg/kg dw	<0.93-<2 µg/kg dw	<21-<45 µg/kg dw	<24-<51 µg/kg dw	<24-<51 µg/kg dw	NM	Konieczny et al. 2017
Dust	Norway/ Tromso	2007/08 (Winter)	Private homes (n=7)	3.3-26.7 (7)	2-10.5 (7.5)	0.9-322 (96.8)	0.2-3.0 (0.8)	NM	NM	NM	Huber et al. 2011
Dust	Norway/ Tromso	2007/08 (Winter)	Office (n=1)	(10.6)	(12.1)	(1.4)	(3.7)	NM	NM	NM	Huber et al. 2011

				LC-PFCA concentrations in ng/g Range (median), detection frequency %							
Media	**Country/ Region**	**Year of sampling (Months)**	**Type of location (n)**	**C9**	**C10**	**C11**	**C12**	**C13**	**C14**	**C15**	**Reference**
Dust	Norway/ Tromso	2007/08 (Winter)	Storage room in office building (n=1)[b]	(43.4)	(22.4)	(614)	(<4.7)	NM	NM	NM	Huber et al. 2011
Dust	Norway/ Tromso	2015	Private homes (n=6)	<0.05-20.9	<0.05-6.68	<0.05-6.81	<0.05-2.97	<0.05-1.74	<0.05-1.31	NM	Bohlin Nizzetto et al. 2015
Dust	Norway	Not provided	Private homes (n=7)	n.d.-3, 71[d]	n.d.-6, 57[d]	n.d.-2, 43[d]	n.d.-5, 57[d]	n.d.-0.11, 14 [d]	n.d.-n.d., 0 [d]	NM	Padilla-Sanchez et al. 2016
Dust	Czech Republic	2013 (April-Aug)	Private homes (n=16)	n.d.-11 (<MQL), 50	n.d.-17.1 (<MQL), 31.3	n.d.-4.3 (<IQL), 6.3	n.d.-13.1 (0.5), 56.3	n.d.-3.5 (<IQL), 6.3	n.d.-14.8 (<MQL), 43.8	NM	Karaskova et al. 2016
Dust	Canada	2013 (April-Aug)	Private homes (n=20)	<MQL-195 (4.4), 95	0.9-86.2 (2.4), 100	n.d.-49.6 (1.1), 60	n.d.-61.1 (1.1), 75	n.d.-19.4 (<MQL), 29	<MQL-33.6 (1.4), 65	NM	Karaskova et al. 2016
Dust	USA	2013 (April-Aug)	Private homes (n=20)	1.1-62.9 (3.9), 100	0.4-64.0 (1.8), 100	n.d.-13.1 (1.2), 60	n.d.-9.0 (0.6), 60	n.d.-2.1 (<MQL), 15.0	<MQL-3.0 (0.8), 50	NM	Karaskova et al. 2016
Dust	UK, Australia, Germany, USA	2004	Private homes (n=39)	<MQL-832 (<MQL), 25.6	<MQL-1965 (<MQL), 38.5	<MQL-732 (<MQL), 20.5	<MQL-1048 (<MQL), 43.6	NM	NM	NM	Kato et al. 2009
Dust	Canada	2007	Private homes of pregnant women (n=18)	1.4-220 (15), 100	1.7-250 (15), 100	<0.5-240 (6.1), 100	1.4-160 (10), 100	<0.5-67 (2.4), 78	<0.5-24 (3.3), 94	NM	Beesoon et al. 2011
Air	Canada/ Vancouver, BC	2007/08	Private homes (n=39)	<DL-2166 pg/m^3 (89 pg/m^3)[e], 62	<DL-977 pg/m^3 (7.9 pg/m^3), 97	<DL-79 pg/m^3 (3.4 pg/m^3)[e], 23	<DL-263 pg/m^3 (9.8 pg/m^3)[e], 28	NM	<DL-3.7 pg/m^3 (0.16 pg/m^3)[e], 5	NM	Shoeib et al. 2011
Dust	Canada/ Vancouver, BC	2007/08	Private homes (n=132)	<DL-680 (26)[e], 70	<DL-251 (8.4)[e], 55	<DL-370 (7.8)[e], 49	<DL-301 (6.3)[e], 42	NM	<DL-478 (7.3)[e], 39	NM	Shoeib et al. 2011
Dust	USA	Not provided	Childcare facilities (n=20)[f]	0.11-13 (1.7), 100	0.22-2.4 (0.59), 100	0.05-3.0 (0.65), 100	0.26-3.1 (0.58), 100	n.d.-2.2 (0.31), 50	n.d.-4.4 (0.29), 85	NM	Zheng et al. 2020
Dust	/ Catalan	2009	Private homes (n=10)[g]	0.4-37	0.75-41	0.30-15	<DL-17	0.047-25	<DL-6.7	NM	Ericson Jogsten et al. 2012
Air	Finland/Kuopio	2014/15	Children's bedrooms (n=57)	0.95-16.5 pg/m^3 (2.41 pg/m^3), 100	1.27-20.6 pg/m^3 (4.21 pg/m^3), 100	<DL-8.24 pg/m^3 (0.75 pg/m^3), 98	<DL-5.65 pg/m^3 (0.84 pg/m^3), 96	<DL-2.22 pg/m^3 (<DL), 21	<DL-1.79 pg/m^3 (0.33 pg/m^3), 63	<DL-1.06 pg/m^3 (<DL), 7	Winkens et al. 2017

n.d. = non-detect; NR = not reported due to low percentage of detection (<50%); NM = not measured; MQL = method quantification limit; MDL = method detection limit; IQL = instrumental quantification limit; DL = detection limit

[a] Geometric mean

[b] The storage room was being used to store highly contaminated PFAS samples, technical mixtures and chemicals for several years.

[c] The main production of the manufacturing plant included perfluoroalkyl sulfonic acid, perfluoroalkyl carboxylic acid, perfluorotertiary amine and their derivatives using the electro-chemical fluorination process. Dust samples were mainly collected from inside the plant (offices, storage rooms, raw material stock rooms, electrolysis and sulfonation workshops, and a laboratory building). Three samples were collect outside next to roads near the facility.

[d] The detection frequency % was not explicitly provided by Padilla-Sanchez et al. (2016), and was calculated manually.

[e] Arithmetic mean

[f] C16 PFCA was also measured in this study, but was not detected in any dust sample.

[g] C18 PFCA was also measured in this study, but was not detected in any dust sample.

Table 6. Concentrations of long-chain PFCAs in drinking water at the tap. Tap water concentration in ng/L range, detection frequency

Location	Year	N	C9	C10	C11	C12	C13	C14	Reference
The Netherlands	2016	6	<0.03-0.28	<0.03-0.10	NM	NM	NM	NM	Gebbink et al. 2017
The Netherlands	2013-2014	37	<0.6	<0.6	<0.6	NM	NM	NM	Zafeiraki, et al. 2015
Greece	2013-2014	43	<0.6	<0.6	<0.6	NM	NM	NM	
Sweden	2012-2014	30	<10	<10	<10	<10	NM	NM	Gyllenhammar et al. 2015
Germany	Not provided	26	1.4, 4%	<1	<1	<1	<1	NM	Gellrich et al. 2013
Spain	2008	40	<0.15-58.21, 58%	<0.12-10.00, 33%	<0.07-4.23, 13%	<0.04	<0.06	NM	Ericson et al. 2009
Europe	2010	7	<MLQ-0.522	<MLQ-0.612	ND-<MLQ	<MLQ	NM	NM	Ullah et al. 2011
Canada, USA, Chile, Africa, Europe, Asia	2015-2016	59	median=0.15, max=4.5, 64%	median <0.030, max=1.0, 66%	<0.010-1.6, 14%	<0.010-1.1, 12%	<0.010-0.94, 8%	<0.010-0.62, 8%	Kaboré et al. 2018
Canada[a]	2012-2016	226	<0.5–1.2, 18%	<0.5–0.63, 2%	<1	<1	NM	NM	Kleywegt et al. 2020
France[a]	2009	41	median <1, max=11, 24%	<1	NM	NM	NM	NM	Boiteux et al. 2012
Austria[b]		10	ND-0.85, 60%	ND	ND	ND	NM	NM	Austria Annex E information 2022

LOQ = limit of quantification; MLQ = method limits of quantification; ND = not detected; NM = not measured

[a] Long-chain PFCAs were measured in treated water leaving the water treatment plant

[b] Long-chain PFCAs were measured in well water

Concentrations of long-chain PFCAs in food

11. The diet has been suggested as a principal exposure route for long-chain PFCAs (Vestergren et al. 2012; Poothong et al. 2020) and a number of studies have investigated the presence of long-chain PFCAs in food items (see EFSA 2020 Annex A4; Table 7). However, due in part to methodological challenges associated with targeted analyses in varied and complex food matrices, the measurements of long-chain PFCAs often fall below of the limit of detection/quantification (LOD/LOQ). For example, in the 2019-2021 analyses of regional and national food samples collected under the U.S. Total Diet Study, only 3 out of 532 samples had concentrations of long-chain PFCAs that were above the method detection limit. C9 PFCA was detected in a cod sample (233 ng/kg) and a frozen fish stick/patty (50 ng/kg) whereas C10 PFCA was detected in canned tuna (72 ng/kg)(FDA 2021). Similarly, concentrations of C9, C10 and C12 PFCAs were below the LOD for 31 different types of food (310 individual food samples) purchased from supermarkets in Dallas, Texas (USA) in 2009 (Schecter et al. 2010). In an analysis of 54 food composites collected during Canadian Total Diet studies from 1992 to 2004, C10 – C12 PFCAs were not detected in any food sample and C9 PFCA was detected only in beef steak at 4.5 ng/g, wet weight (Tittlemier et al. 2007). The European Food Safety Authority (EFSA) reported that 93.5% or more of their results for C9 – C16 and C18 PFCA concentrations in foods were left-censored (i.e., below the LOQ or LOD) (EFSA 2020). Fish was the best studied of all food types and several long-chain PFCAs were present in fish at higher concentrations than in other food groups with upper bound mean concentrations ranging from 0.072 μg/kg (C12 in halibut) to 5.85 μg/kg (C13 in fish offal) (EFSA 2020). Relatively high values were also noted for edible offal from game animals, with upper bound mean concentrations ranging from 0.24 μg/kg (C11) to 9.87 μg/kg (C9) and a maximum 95th percentile concentration of 22 μg/kg (C9) (EFSA 2020). In addition, there is some indication that food contact materials (e.g., paper cups, paper trays, microwave popcorn bags) may be a source of exposure to long-chain PFCAs and their related products (Yuan et al. 2016; Granby and Tesdal Haland 2018). However, data on the migration of long-chain PFCAs into food is limited. EFSA has estimated the chronic dietary exposure to 17 PFAS (including C9 – C14 PFCA) to be at the level of a few ng/kg bw/d (EFSA 2020). However, due to the left-censored nature of the data, the reliability of dietary intake estimates in general for long-chain PFCAs is considered to be low.

12. The relationship between dietary exposure and body burden of long-chain PFCAs remains uncertain with few correlations having been observed. This may be due to limitations associated with estimating dietary exposure or because serum concentrations reflect longer term exposure while dietary intake estimates tend to reflect a shorter period of time. When considering the results of a food frequency questionnaire covering a longer time period (e.g., 12 months vs 7 days or less), Haug et al. (2010a) found a significant association between estimated dietary intakes of C11 PFCA and body burden. Despite the absence of a consistent correlation between body burden and total dietary intake estimates of long-chain PFCAs, regular consumption of several dietary items (e.g., fish, eggs, meat, popcorn, junk food) has been associated with increases in internal levels of long-chain PFCAs (Averina et al. 2018; Tian et al. 2018; Susmann et al. 2019; Zhou et al. 2019; Lin et al. 2020).

Table 7. Concentrations of long-chain PFCAs in food (see also Annex A4 of EFSA 2020). LC-PFCA Concentrations – Means or Ranges in pg/g

Food Category	Country/ Region (n)	Year of sampling	Food Sample Type	C9	C10	C11	C12	C13	C14	Reference
Fish	Netherlands	2009	Fatty fish	5	4	36	10	41	3	Noorlander et al. 2011
			Lean fish	77	48	177	56	229	24	
	Sweden	1999	Fillets of fish, canned fish, shellfish	70	40	111	32	86	10	Gebbink et al. 2015
	Norway/ Oslo	2008/09	Fish sticks	<11	17	18	<13	NM	NM	Haug et al. 2010b
			Canned mackerel	<11	<31	19	<12	NM	NM	
			Salmon	10	26	4.5	<12	NM	NM	
			Cod	5.9	13	21	<7.5	NM	NM	
			Cod liver	14	39	230	<33	NM	NM	
	USA/Dallas (n=70)	2009	Salmon, tuna, catfish, tilapia, cod, sardines, fish sticks	<LOD	<LOD	NM	<LOD	NM	NM	Schecter et al. 2010
	Canada	2004	Marine fish	<1 ng/g	<2 ng/g	<1 ng/g	<0.8 ng/g	NM	<5	Tittlemier et al. 2007
			Freshwater fish	<1 ng/g	<2 ng/g	<1 ng/g	<0.9 ng/g	NM	<5	
	Canada	1998	Freshwater fish	<1 ng/g	<2 ng/g	<2 ng/g	<2 ng/g	NM	<2	
	Sweden	2010	Fillets of fish, canned fish, shellfish	72	92	316	72	123	12	Vestergren et al. 2012
	Sweden	2005	Fillets of fish, canned fish, shellfish	90	79	214	54	113	8.6	
	Sweden	1999	Fillets of fish, canned fish, shellfish	90	44	130	36	68	9.8	
	USA	2020/21	Tilapia, shrimp, salmon, catfish, cod	<MDL-233 ng/kg[a]	<MDL	NM	NM	NM	NM	FDA 2021
Crustaceans	Netherlands	2009	Muscles, shrimp, crab	58	90	157	45	268	45	Noorlander et al. 2011
Dairy	Netherlands	2009	Butter	2	6	<3	2	<19	<1	Noorlander et al. 2011
			Cheese	7	8	<16	<11	<92	<5	
			Milk	<1	1	<0.5	<0.5	<0.5	<2	
	Sweden	1999	Milk, cream, yogurt, cheese	0.5	<0.3	<1	<0.5	<0.2	<0.05	Gebbink et al. 2015
	Norway/ Oslo	2008/09	Cheese	16	6.6	4.1	<15	NM	NM	Haug et al. 2010b
			Milk	<2.1	4.0	<2.5	<2.4	NM	NM	
	USA/Dallas (n=80)	2009	Butter, milk, cheese, ice cream, frozen yogurt, yogurt	<LOD	<LOD	NM	<LOD	NM	NM	Schecter et al. 2010
	Sweden	2010	Milk, cream, yogurt, cheese	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	Vestergren et al. 2012
	Sweden	2005	Milk, cream, yogurt, cheese	<MDL	6.6	<MDL	<MDL	<MDL	<MDL	
	Sweden	1999	Milk, cream, yogurt, cheese	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	
	USA	2020/21	Ice cream, milk shake, frozen yogurt, cheese, milk, cream	<MDL	<MDL	NM	NM	NM	NM	FDA 2021
Eggs	Netherlands	2009	Chicken eggs	6	11	<19	<13	<107	<5	Noorlander et al. 2011
	Sweden	1999	Hen eggs	24	5.6	41	9.9	16	2.8	Gebbink et al. 2015
	Netherlands (n=73)	2013/14	Domestic eggs	<0.5-2.0 ng/g ww (0.9 ng/g ww), 18 [b]	<0.5-3.0 ng/g ww (0.9 ng/g ww), 32 [b]	<0.5-2.3 ng/g ww (0.9 ng/g 22), 21 [b]	NM	NM	NM	Zafeiraki et al. 2016

Food Category	Country/ Region (n)	Year of sampling	Food Sample Type	C9	C10	C11	C12	C13	C14	Reference
	Netherlands (n=22)	2013/14	Commercial eggs	<0.5-<0.5 ng/g ww	<0.5-<0.5 ng/g ww	<0.5-<0.5 ng/g ww	NM	NM	NM	Zafeiraki et al. 2016
	Greece (n=45)	2013/14	Domestic eggs	<0.5-3.0 ng/g ww (0.8 ng/g ww), 20 [b]	<0.5-8.0 ng/g ww (0.9 ng/g ww), 36 [b]	<0.5-4.5 ng/g ww (0.7 ng/g ww), 24 [b]	NM	NM	NM	Zafeiraki et al. 2016
	Greece (n=31)	2013/14	Commercial eggs	<0.5-<0.5 ng/g ww	<0.5-<0.5 ng/g ww	<0.5-<0.5 ng/g ww	NM	NM	NM	Zafeiraki et al. 2016
	Norway/Oslo	2008/09	NP	<7.4	12	9.9	<8.1	NM	NM	Haug et al. 2010b
	USA/Dallas (n=10)	2009	NP	<LOD	<LOD	NM	<LOD	NM	NM	Schecter et al. 2010
	Sweden	2010	Hen eggs	<MDL	3.3	<MDL	<MDL	<MDL	<MDL	Vestergren et al. 2012
	Sweden	2005	Hen eggs	5.6	4.9	3.3	<MDL	<MDL	<MDL	
	Sweden	1999	Hen eggs	22	15	3.8	10	14	<MDL	
	USA	2020/21	Hard boiled	<MDL	<MDL	NM	NM	NM	NM	FDA 2021
Meat	Netherlands	2009	Pork	2	2	<4	<3	<23	<1	Noorlander et al. 2011
			Beef	4	6	2	<2	<14	<0.7	
			Chicken/poultry	1	<1	<3	<2	<17	<0.8	
	Sweden	1999	Beef, pork, lamb, poultry, cured, sausage	6.7	<0.3	9.1	12.3	<0.2	7.1	Gebbink et al. 2015
	Norway/ Oslo	2008/09	Pork	5.5	16	<8.2	<8.0	NM	NM	Haug et al. 2010b
			Beef	15	23	<6.4	<6.2	NM	NM	
			Chicken	6.8	<23	13	<9.2	NM	NM	
	USA/Dallas (n=80)	2009	Beef, pork, chicken/poultry, sausage, canned chili	<LOD	<LOD	NM	<LOD	NM	NM	Schecter et al. 2010
	Canada	2004	Beef steak	4.5 ng/g	<2 ng/g	<1 ng/g	<1 ng/g	NM	<3	Tittlemier et al. 2007
			Roast beef	<1 ng/g	<2 ng/g	<2 ng/g	<1 ng/g	NM	<3	
			Ground beef	<1 ng/g	<4 ng/g	<1 ng/g	<1 ng/g	NM	<3	
			Luncheon meat, cold cuts	<1 ng/g	<2 ng/g	<1 ng/g	<1 ng/g	NM	<3	
	Sweden	2010	Beef, pork, lamb, poultry, cured, sausage	5.8	6.3	2.5	1.1	<MDL	<MDL	Vestergren et al. 2012
	Sweden	2005	Beef, pork, lamb, poultry, cured, sausage	9.2	6.4	7.8	2.1	3.8	<MDL	
	Sweden	1999	Beef, pork, lamb, poultry, cured, sausage	7.1	5.2	4.8	1.9	<MDL	<MDL	
	USA	2020/21	Beef, pork, lamb, poultry, salami	<MDL	<MDL	NM	NM	NM	NM	FDA 2021
Pastries/ Baked Goods	Netherlands	2009	Cake, almond paste, biscuits, pie	1	1	<1	<0.7	<6	<0.3	Noorlander et al. 2011
	Sweden	1999	Biscuits, buns, cakes	1.2	<0.3	<1	<0.5	<0.2	<0.05	Gebbink et al. 2015
	Sweden	2010	Biscuits, buns, cakes	<MDL	2.5	<MDL	<MDL	<MDL	<MDL	Vestergren et al. 2012
	Sweden	2005	Biscuits, buns, cakes	<MDL	2.9	1.5	<MDL	<MDL	<MDL	
	Sweden	1999	Biscuits, buns, cakes	<MDL	2.0	1.0	1.6	<MDL	<MDL	
	USA	2020/21	Biscuits, cake, muffin, cinnamon roll	<MDL	<MDL	NM	NM	NM	NM	FDA 2021
Fruits/ Vegetables	Netherlands	2009	Fruits & vegetables[c]	1	2	<2	<2	<14	<0.7	Noorlander et al. 2011
	Sweden	1999	Vegetables (fresh, frozen, and canned)	<0.3	<0.3	<1	<0.5	<0.2	<0.05	Gebbink et al. 2015

Food Category	Country/ Region (n)	Year of sampling	Food Sample Type	C9	C10	C11	C12	C13	C14	Reference
	Sweden	1999	Fruits (fresh, frozen, and canned)	0.6	<0.3	<1	<0.5	<0.2	<0.05	Gebbink et al. 2015
	Sweden	1999	Potatoes (fresh, French-fries, crisps)	<0.3	<0.3	<1	<0.5	<0.2	<0.05	Gebbink et al. 2015
	Norway/Oslo	2008/09	Lettuce	<1.0	0.78	<1.3	1.3	NM	NM	Haug et al. 2010b
			Carrot	<2.1	<1.4	<2.5	<2.4	NM	NM	
			Potato	<4.1	3.0	2.2	<4.8	NM	NM	
	Sweden	2010	Vegetables (fresh, frozen, and canned)	<MDL	2.5	<MDL	<MDL	<MDL	<MDL	Vestergren et al. 2012
	Sweden	2005	Vegetables (fresh, frozen, and canned)	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	
	Sweden	1999	Vegetables (fresh, frozen, and canned)	<MDL	3.1	<MDL	1.6	<MDL	<MDL	
	Sweden	2010	Fruits (fresh, frozen, and canned)	<MDL	2.4	<MDL	<MDL	<MDL	<MDL	
	Sweden	2005	Fruits (fresh, frozen, and canned)	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	
	Sweden	1999	Fruits (fresh, frozen, and canned)	1.9	1.8	<MDL	<MDL	<MDL	<MDL	
	Sweden	2010	Potatoes (fresh, French-fries, crisps)	<MDL	2.6	<MDL	<MDL	<MDL	<MDL	
	Sweden	2005	Potatoes (fresh, French-fries, crisps)	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	
	Sweden	1999	Potatoes (fresh, French-fries, crisps)	<MDL	1.7	<MDL	<MDL	<MDL	<MDL	
	USA	2020/21	Fruits & vegetables	<MDL	<MDL	NM	NM	NM	NM	FDA 2021
			Potatoes (boiled, baked, Fresh-fries)	<MDL	<MDL	NM	NM	NM	NM	
Fats/ Vegetable-based foods	Netherlands	2009	Vegetable oil	<0.1	<0.6	<2	<1	<11	<0.6	Noorlander et al. 2011
			Industrial oil	<0.3	2	<3	<2	<16	<0.8	
	Sweden	1999	Butter, margarine, cooking oil, mayo	3.7	<0.3	1.2	<0.5	<0.2	<0.05	Gebbink et al. 2015
	Norway/Oslo	2008/09	Margarine	<13	<8.6	<16	<16	NM	NM	Haug et al. 2010b
	USA/Dallas (n=70)	2009	Olive oil, canola oil, margarine, cereal, apples, potatoes, peanut butter	<LOD	<LOD	NM	<LOD	NM	NM	Schecter et al. 2010
	Sweden	2010	Butter, margarine, cooking oil, mayo	<MDL	<MDL	5.8	<MDL	<MDL	<MDL	Vestergren et al. 2012
	Sweden	2005	Butter, margarine, cooking oil, mayo	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	
	Sweden	1999	Butter, margarine, cooking oil, mayo	<MDL	3.8	<MDL	<MDL	<MDL	<MDL	
Grains/ Cereals	Netherlands	2009	Flour	15	9	4	4	<9	<0.4	Noorlander et al. 2011
	Sweden	1999	Flour, grain, corn flakes, pasta, bread	<0.3	<0.3	<1	<0.5	<0.2	0.3	Gebbink et al. 2015
	Norway/Oslo	2008/09	Bread	9.5	17	<15	<15	NM	NM	Haug et al. 2010b
	Canada	1998	Pizza	<1	<1	<1	<1	NM	<1	Tittlemier et al. 2007
			Microwave popcorn	<1 ng/g	<1 ng/g	<0.9 ng/g	<1 ng/g	NM	<1 ng/g	
	Sweden	2010	Flour, grain, corn flakes, pasta, bread	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	Vestergren et al. 2012
	Sweden	2005	Flour, grain, corn flakes, pasta, bread	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	
	Sweden	1999	Flour, grain, corn flakes, pasta, bread	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	
	USA	2020/21	Breads, rice, cereal, pizza	<MDL	<MDL	NM	NM	NM	NM	FDA 2021
Sugar/ Sweets/ Sauces	Sweden	1999	Sugar, chocolate, candy, sauces	<0.3	<0.3	<1	<0.5	<0.2	<0.05	Gebbink et al. 2015
	Norway/Oslo	2008/09	Strawberry jam	3.7	8.70	<13	<13	NM	NM	Haug et al. 2010b
	Sweden	2010	Sugar, chocolate, candy, sauces	<MDL	2.0	<MDL	<MDL	<MDL	<MDL	Vestergren et al. 2012
	Sweden	2005	Sugar, chocolate, candy, sauces	<MDL	2.0	1.1	<MDL	<MDL	<MDL	
	Sweden	1999	Sugar, chocolate, candy, sauces	<MDL	1.7	<MDL	<MDL	<MDL	<MDL	

Food Category	Country/ Region (n)	Year of sampling	Food Sample Type	C9	C10	C11	C12	C13	C14	Reference
	USA	2020/21	Barbeque sauce	<MDL	<MDL	NM	NM	NM	NM	FDA 2021
Soft drinks	Sweden	1999	Soft drinks, mineral water, beer	0.5	<0.3	<1	<0.5	<0.2	<0.05	Gebbink et al. 2015
	Sweden	2010	Soft drinks, mineral water, beer	<MDL	1.0	<MDL	<MDL	<MDL	<MDL	Vestergren et al. 2012
	Sweden	2005	Soft drinks, mineral water, beer	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	
	Sweden	1999	Soft drinks, mineral water, beer	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	

NM = not measured; NP = not provided; LOD = limit of detection; MDL = method detection limit

[a] The detectable value of PFNA (233 ng/kg) was found in cod, and was the only detectable value.
[b] Range (median), detection frequency
[c] Apple, orange, grape, banana, potato, onion, carrot, beat, chicory, leak, tomato, cucumber, paprika, mushroom, cauliflower, broccoli, cabbage, brussel sprouts, spinach, endive, lettuce, beans

Concentrations of long-chain PFCAs in humans

Table 8. Concentrations of long-chain PFCAs in human milk. Human milk concentration in pg/mL mean (range), % detection

Location (n)	Year	C9	C10	C11	C12	C13	C14	Reference
Czech Republic (n=232)	2017	7 (<3-29), 98.7	NM	NM	NM	NM	NM	Černá et al. 2020
France (n=48)	2007	(< LOD-64), 2	< LOQ	< LOQ	< LOQ	NM	NM	Antignac et al. 2013
France (n=30)	2010	< LOQ	< LOQ	< LOQ	< LOQ	NM	NM	Kadar et al. 2011
France (n=61)	2010-2013	< LOQ	< LOQ	< LOQ	NM	NM	NM	Cariou et al. 2015
Spain (n=10)	2007	< LOQ	< LOQ	< LOQ	< LOQ	< LOQ	NM	Kärrman et al. 2010
Spain (n=20)	2008	< LOQ	666, (< LOQ-1095), 10	NM	< LOQ	NM	NM	Llorca et al. 2010
Spain (n=10)	2012	4 (2-21), 30	43 (1.4-306), 70	88 (18-370), 60	ND	ND	ND[a]	Lorenzo et al. 2016
Spain (n=67)	2014	41 (15-70), 6	24 (< LOQ-34), 4	29 (16-57), 10	21 (16-26), 3	NM	NM	Motas Guzman et al. 2016
Sweden (n=12)	2004	17 (< 0.005–0.020), 17[b]	< LOQ	< LOQ	NM	NM	NM	Kärrman et al. 2007
Ireland (n=92)	Not provided	26 (<10-100), 69	NM	NM	NM	NM	NM	Abdallah et al. 2020
United States (n=45)	2004	7.26 (<5.2-18.4), 64	(< 7.72-11.1), 9	(<4.99-8.84), 7	(<4.40-9.74), 2	NM	NM	Tao et al. 2008a
United States (n=50)	2019	5.98[c] (2.00-36.3), 100	7.40[c] (<0.80-697), 94	4.43[c] (<0.20-18.0), 84	5.26[c] (<1.0-374), 94	3.16[c] (<1.2-313), 78	<15[c] (<15-409), 18	Zheng et al. 2021
China (n=19)	2004	(6.3-62), 100	(3.8-15), 100	(9.1-56), 100	NM	NM	NM	So et al. 2006
China (n=30)	2008-2009	15.3 (<10-47), 70.0	<15 (<15-29), 13.3	16.0 (<10-47), 56.7	<10 (<10-25), 10.0	<10 (<10-43), 23.3	NM	Fujii et al. 2012
China (n=1237)	2007	9.9 (6-76), 100	(<1.44–63), 87.5	(<1.30-196), 83	NM	NM	NM	Liu et al. 2010
China (n=50)	2009	26 (5-95), 100	20 (< 1–70), 78	26 (< 1–70), 72	<LOQ	<LOQ	NM	Liu et al. 2011b

Location (n)	Year	C9	C10	C11	C12	C13	C14	Reference
China (n=174)	2018, 2019	12 (<LOD-115), 55	12 (<LOD-138), 67	13 (<LOD-92), 84	(<LOD-11), 0.57	<LOQ	<LOQ	Jin et al. 2020
Japan (n=30)	2010	32.1 (<10-72), 90.0	21.3 (<15–65), 66.7	36.6 (<10-100), 93.3	<10 (<10-29), 16.7	15.2(<10-91), 33.3	NM	Fujii et al. 2012
Japan (n=24)	1999	(<8.82-23.9), 13	< LOQ	< LOQ	< LOQ	NM	NM	Tao et al. 2008b
Korea (n=30)	2010	14.7 (10-41), 66.7	<15 (<15-19), 13.3	19.6 (<10-51), 73.3	<10 (<10-41), 13.3	11.7 (<10-43), 50	NM	Fujii et al. 2012
Korea (n=293)[d]	Beginning 2011	19.4 (<10-127), 63	0.88 (<10-58.1), 3.1	23.7 (<10-119), 86	1.57 (<10-129), 4.1	0.70 (<10-52.1), 2.4	0.38 (<10-82.6), 0.7	Lee et al. 2018
Malaysia (n=13)	2003	(<8.82-14.9), 8	< LOQ	< LOQ	< LOQ	NM	NM	Tao et al. 2008b
Phillipines (n=24)	2000, 2004	(<8.82-25.0), 17	< LOQ	< LOQ	< LOQ	NM	NM	Tao et al. 2008b
Indonesia (n=20)	2001	(<8.82-135), 5	< LOQ	< LOQ	< LOQ	NM	NM	Tao et al. 2008b
Vietnam (n=40)	2000-2001	(<8.82-10.9), 5	< LOQ	< LOQ	< LOQ	NM	NM	Tao et al. 2008b
Cambodia (n=24)	2000	(<8.82-12.3), 13	< LOQ	< LOQ	< LOQ	NM	NM	Tao et al. 2008b
India (n=34)	2002, 2004, 2005	<8.82	< LOQ	< LOQ	< LOQ	NM	NM	Tao et al. 2008b

LOD = limit of detection; LOQ = limit of quantification; ND = not detected; NM = not measured

[a] One measurement for C14 was below the LOQ. C16 and C18 PFCAs were also measured in this study. All values for C16 were non-detects and all values for C18 were non-detects except for one which was below the LOQ.

[b] The detection frequency % was not explicitly provided but was calculated manually.

[c] Median

[d] C16 and C18 PFCAs were also measured in this study with the mean (range), % detection as follows: C16 = 0.43 (<10-96.4), 0.7; C18 = 0.27 (<10-54.2), 0.7

Table 9. Concentrations of long-chain PFCAs in plasma or serum as detected in larger scale biomonitoring programs. LC-PFCA concentrations in ng/mL Geometric mean (range), detection frequency %

Country/ Region	Year of sampling	Population (n)	C9	C10	C11	C12	C13	C14	Reference
Canada	2009–2011	CHMS, 12-79yrs (1524)	0.82, 99.4	0.20, 79.3	0.12, 59.3	NM	NM	NM	Health Canada 2021
Canada	2016–2017	CHMS 12-79yrs (1497)	0.51, 98.8	0.18, 91.4	NC, 38.5	NM	NM	NM	Health Canada 2021
Canada	2018-2019	CHMS 12-79yrs (1457)	0.44, 98.4	0.12, 69.0	NC, 39.0	NM	NM	NM	Health Canada 2021
USA	2011-2012	NHANES, 12-19yrs (344)	0.680	0.146	NC	NM	NM	NM	CDC 2021
USA	2013-2014	NHANES, 12-19yrs (402)	0.500	0.136	NC	NM	NM	NM	CDC 2021
USA	2015-2016	NHANES, 12-19yrs (353)	0.500	NC	NC	NM	NM	NM	CDC 2021
USA	2017-2018	NHANES, 12-19yrs (313)	0.400	0.153	NC	NM	NM	NM	CDC 2021
USA	2011-2012	NHANES, 20+yrs (1560)	0.890	0.209	0.146	NM	NM	NM	CDC 2021
USA	2013-2014	NHANES, 20+yrs (1766)	0.700	0.193	NC	NM	NM	NM	CDC 2021
USA	2015-2016	NHANES, 20+yrs (1640)	0.600	0.160	NC	NM	NM	NM	CDC 2021
USA	2017-2018	NHANES, 20+yrs (1616)	0.400	0.199	0.129	NM	NM	NM	CDC 2021
USA	2000-2001	Red cross blood donors (645)	0.56	0.16	NC	NC	NM	NM	Olsen et al. 2017
USA	2006	Red cross blood donors (600)	0.96	0.34	NC	NC	NM	NM	Olsen et al. 2017
USA	2010	Red cross blood donors (600)	0.83	0.27	NC	NC	NM	NM	Olsen et al. 2017

Country/ Region	Year of Sampling	Population (n)	C9	C10	C11	C12	C13	C14	Reference
USA	2015	Red cross blood donors (616)	0.43	0.15	NC	NC	NM	NM	Olsen et al. 2017
USA/ New Hampshire	2015–2016	All ages (1,578)	0.73, 85.2	0.22, 42.1	0.19, 30.0	0.08, 4.7	NM	NM	NH DHHS 2016
USA/ Ohio	2005-2007	Girls, 6-8yrs (353)	1.4, 99.9	0.3, 75.8	NM	NM	NM	NM	Pinney et al. 2014
USA/ California	2007-2009	Girls, 6-8yrs (351)	1.7, 100	0.3, 78.7	NM	NM	NM	NM	Pinney et al. 2014
USA/ Massachusetts	2007-2010	Girls, 6-10yrs (653)	1.7, 99.5	0.3, 88.2	NM	NM	NM	NM	Harris et al. 2017
9 European Countries	1979-2015	-	(<LOD-38.6)	(<LOD-11.2)	(<LOD-24.9)	(<LOD-6.5)	(<LOD-0.90)	(<LOD-0.43)	ECHA 2018a (see Appendix I for details)
Sweden	2016-2017	Riksmaten Adolescents (1098)	0.382[a b] (<LOD-2.80)	0.162 (<LOD-1.35)	0.097 (<LOD-1.01)	<LOD (<LOD-0.182)	<LOD (<LOD-0.168)	(<LOD-0.136)	Nystrom et al. 2022
Sweden	2017	Adolescents 17-21yrs (197)	0.41 (0.10-1.56), 100	0.21 (0.07-0.87), 100	0.14 (0.01-0.66), 100	0.02 (<LOD-0.09), 88	NM	NM	Norén et al. 2019
Sweden	2017-2019	First time mothers (110)	0.5 (0.13-1.59), 100	0.5 (<0.082-1.10), 94	0.5 (<0.082-0.46), 86	<LOQ	(<0.082-0.14), 8	<LOQ	Gyllenhammar et al. 2020
Germany	2014-2017	Children 3-17yrs (997-1108)	<LOQ (<LOQ-3.54), 10	<LOQ (<LOQ-3.00), 10	<LOQ (<LOQ-0.78), 1	<LOQ (<LOQ-0.96), 0	NM	NM	Duffek et al. 2020
France	2014-2016	Adults (744)	0.80, 99.5	0.34, 89.2	0.17, 99.5	NC, 22.3	NM	NM	Fillol et al. 2021
France	2014-2016	Children (249)	0.61, 99.6	0.24, 71.1	0.12, 95.6	NC, 8.0	NM	NM	Fillol et al. 2021
Greenland	2010-2015	Pregnant women (499)	1.15[a] (0.21–7.87), 100	0.71[a] (0.12–7.84), 99.9	1.42[a] (0.08–18.2), 99.7	NA	NA	NM	Hjermitslev et al. 2020
Korea/ Siheung	2008	>12 yrs (633)	2.09[a] (1.49-2.74), 100	0.91[a] (0.58-1.45), 100	1.75[a] (1.11-4.58), 100	0.92[a] (0.21-1.13), 76.3	0.39[a] (1.27-0.57), 99.7	Detection <7.4%	Ji et al. 2012
Korea/ Seoul and Gyeonggi	2012-2014	KorEHS-C 3-18 yrs (150)	0.939, 100	0.0501, 79.3	0.545, 98.7	<LOQ	NC, 32.7	<LOQ	Kang et al. 2018
Korea/ Seoul	2006-2015	HASSC Adults (786)	2.03 (<LOD-12.64)	1.29 (<LOD-5.36)	1.83 (<LOD-9.80)	0.36 (<LOD-2.87)	0.59 (<LOD-3.41)	0.15 (<LOD-7.69)	Seo et al. 2018
Japan	2009-2010	JECS Mothers (339)	1.8[a] (0.39-11) 100	0.59[a] (<LCMRL-3.1), 99.7	1.5[a] (<LCMRL-5.3), 100	0.17[a] (<LCMRL-0.76), 79.6	0.38[a] (<LCMRL-1.6), 98.8	<LCMRL	Nakayama et al. 2020
Japan	2003-2012	Hokkaido Study Mothers (2689)	1.54	0.51	1.43	0.17	0.33	<MDL	Ait Bamai et al. 2020
Australia	2016-2017	1-4yrs (400)	0.52, 100	0.26, 100	<LOQ	<LOQ	<LOQ	<LOQ	Toms et al. 2019
Australia	2016-2017	5-15yrs (400)	0.38, 100	0.24, 100	<LOQ	<LOQ	<LOQ	<LOQ	Toms et al. 2019
Australia	2016-2017	16-30yrs (400)	0.46, 100	0.26, 100	<LOQ	<LOQ	<LOQ	<LOQ	Toms et al. 2019
Australia	2016-2017	31-45yrs (400)	0.46, 100	0.25, 100	<LOQ	<LOQ	<LOQ	<LOQ	Toms et al. 2019
Australia	2016-2017	46-60yrs (400)	0.47, 100	0.27, 100	<LOQ	<LOQ	<LOQ	<LOQ	Toms et al. 2019
Australia	2016-2017	>60yrs (400)	0.56, 100	0.27, 100	<LOQ	<LOQ	<LOQ	<LOQ	Toms et al. 2019

CHMS = Canadian Health Measures Survey; HASSC = Health Assessment Study of Seoul Citizens; JECS= Japan Environment and Children's Study
KorEHS-C = Korea Environmental Health Survey in Children and Adolescents; LCMRL = lowest concentration minimum reporting level
LOD = limit of detection; LOQ = limit of quantification; NA = data not available; NC = not calculated (the proportion of results below the detection limit was too high to provide a valid result);
NHANES = National Health and Nutrition Examination Survey; NM = not measured

[a] Median

[b] Concentrations for all long-chain PFCAs in this study were measured in ng/g (as opposed to ng/mL). C15, C16 and C18 PFCAs measured in this study were detected in less than 2% of samples and had a range of values of <LOD-0.359 ng/g; <LOD-1.482ng/g and <LOD-8.520 ng/g, respectively.

[c] Concentrations for all long-chain PFCAs in this study were measured in ng/g (as opposed to ng/mL). C15, C16 and C18 PFCAs were measured in this study but were all below the LOQ.

2.4 Hazard assessment for endpoints of concern

13. Laboratory toxicity studies assessing endpoints such as growth, reproduction, and lethality include the following studies. For C9 – C12 PFCAs, the 48h EC50 values for a pelagic cladoceran (*Daphnia magna)* and a benthic cladoceran (*Chydorus sphaericus*) ranged from 12.4 – 181 mg/L with the benthic cladoceran showing greater sensitivity (Ding et al. 2012). Vitellogenin induction occurred in juvenile rainbow trout after dietary exposure to C9 – C11 PFCAs at 250 ppm (Benninghoff et al. 2011). However, in male medaka (*Oryzias latipes*) exposed to C9 PFCA (464 mg/L) or C10 PFCA (51 or 514 mg/L) induction of vitellogenesis was not observed (Ishibashi et al. 2008c). C10 PFCA had a 96h LC50 of 32 mg/L for rainbow trout, a 48h LC50 > 100 mg/L for *Daphnia magna*, and a 72h EC50 of 10.6 mg/L for green algae (*Pseudokirchneriella subcapitata*) whereas C9 PFCA had acute toxicity values > 100 mg/L for both Daphnia and algae (Hoke et al. 2012). For C9 PFCA, 72h EC50 values for green algae (*Chlorella vulgaris*), diatom (*Skeletonema marinoi*) and the blue-green algae (*Geitlerinema amphibium*) ranged from 125 to 473 mg/L (Latala et al. 2009). The 48-hour EC_{50} (based on acute lethality) for C9 PFCA for the soil-dwelling nematode (*Caenorhabditis elegans*) was 306.3 mg/L (Tominaga et al. 2004). However, multi-generation effects were seen at 0.000464 mg/L (C9 PFCA) which induced a 70% decline in nematode fecundity by the fourth generation (Tominaga et al. 2004). C12 and C14 PFCA inhibited algal (*Scenedesmus obliquus*) growth rate in a concentration-dependent manner (i.e., inhibition increased with increasing exposure concentration) and with an increase in cell membrane permeability (Liu et al. 2008a). African clawed frog (*Xenopus laevis*) embryos exposure to 10 uM to 2 mM of C9 – C11 PFCAs resulted in retardation of development, growth inhibition, and multiple edemas, with each PFCA having unique effects on development and teratogenesis at different points in time (Kim et al. 2013).

14. Additional laboratory toxicity studies assessing exposure include the following studies. Rainbow trout fry were fed 200 ppm C10 PFCA or 1000 ppm C9 PFCA for 6 months to determine the impact on hepatic tumorigenesis. Results show that C9 and C10 PFCAs can promote liver cancer, and that the mechanism of promotion may be similar to that of 17ß-estradiol (Benninghoff et al. 2012). C9 PFCA at 0.93 mg/L resulted in altered responses in locomotion and gene expression in embryo-larval zebrafish as well as biochemical and behavioural changes in young adult zebrafish exposed embryonically (Jantzen et al. 2016a,b). Zebrafish larvae exposure to C10 PFCA (0.01 – 10 mg/L) or C13 PFCA (0.01 – 10 mg/L) can modulate the production of the sex steroid hormone and related gene transcription of the hypothalamic-pituitary-gonad axis (Jo et al. 2014). Green mussels exposed to C9 PFCA (0.1 – 1000 µg/L) or C10 PFCA (0.1 – 1000 µg/L) for 7 d showed reduced immune function, but this effect was reversible (Liu and Gin 2018). Genotoxicity was observed in green mussels for C9 PFCA (EC50 values: 144 – 265 µg/L) and C10 PFCA (EC50 values: 73 – 84 µg/L) (Liu et al. 2014a). C9 and C10 PFCAs inhibited the p-glycoprotein in the marine mussel with average IC50 values of 2.2 mg/L and 3.7 mg/L, respectively, indicating that C9 and C10 PFCAs are chemo sensitizers (Stevenson et al. 2006). One-day old male chickens exposed to C10 PFCA (0.1 and 1.0 mg/kg body weight, three times a week for three weeks) had no adverse effects on body weight, organ indexes, blood clinical parameters or organ histopathology (Yeung et al. 2009).

15. As mentioned in the risk profile, field-based wildlife studies are difficult to interpret due to the exposure of mixtures of other PFAS and other contaminants. For example, a mixture of PFAS (PFHxS, PFOS, PFOA, and C9 – C14 PFCAs) was associated with the disruption of thyroid hormone homeostasis in polar bears (*Ursus maritimus*) from the Barents Sea (Bourgeon et al. 2017). However, these polar bears also had concentrations of organochlorine compounds, including polychlorinated biphenyls (PCBs), polybrominated diphenyl ethers (PBDEs), phenolic compounds as well as other PFAS that may also have contributed to the effect observed. Liu et al. (2018a) analyzed pooled polar bear serum from the Hudson Bay and Beaufort Sea subpopulations in the Canadian Arctic and found PCB metabolites, perfluorinated sulfonates, and other polychlorinated compounds. Knudsen et al. (2007) measured insecticides (e.g., mirex), PFAS, hexachlorocyclohexanes, toxaphenes, dioxins, furans, PCBs, brominated compounds, endosulfans, and mercury in northern fulmars (*Fulmarus glacialis*) from the Barents Sea. Gao et al. (2020b) measured 3108 substances (388 contaminants and 2720 metabolites) in wild crucian carp (*Carassius auratus*) from Taihu Lake (China). Further, field-based wildlife studies have shown statistical correlations with observed effects for long-chain PFCA mixtures. For example, total PFAS (includes PFOS, PFOA, PFHxS, PFOSA, and C9 – C13 PFCAs) concentrations in liver (114 – 3052 ng/g ww) may be associated with liver lesions in East Greenland polar bears (Sonne et al. 2008). Correlations were found for the ∑PFCA concentrations in brain at 88 ng/g ww (includes C6 – C8 PFCAs, C12 and C13 PFCAs) with neurochemical transmitter systems and brain-specific bioaccumulation in the East Greenland polar bears. However, results were inconclusive as to whether observed alterations in neurochemical signaling were having negative effects (Eggers Pedersen et al. 2015). C8 – C14 PFCAs and PFOS at plasma concentrations of 0.03 – 29.7 ng/L ww were associated with reduced hatching and

breeding success in adult chick-rearing black-legged kittiwakes (*Rissa tridactyla*) (Tartu et al. 2014). Positive correlations were found for PFCAs in plasma at 3.6 – 35.5 ng/g ww (includes PFOA, C9 – C14 PFCAs) with thyroid hormone concentrations in the northern fulmar and the black-legged kittiwake chicks that may result in developmental effects in young birds (Nøst et al. 2012). Concentrations of the ΣPFCAs (includes C8 – C15 PFCAs) in plasma (at 0.0002 mg/ml for ΣPFCAs) were associated with altered immune parameters in bottlenose dolphins (*Tursiops truncatus*) that may affect immune, hematopoietic, kidney and liver function (Fair et al. 2013). Nakayama et al. (2008) studied the common cormorant, a fish-eating bird that is the top predator in the Lake Biwa (Japan) ecosystem. C9 PFCA liver concentrations (< 0.005 – 0.043 µg/g-ww) were related to gene expression. Significant positive relationships were shown between C9 PFCA and glutathione peroxidase 1 (enzyme in the antioxidant system) and heterogenous nuclear ribonucleoprotein U (RNA processing). Sun et al. (2020) studied the effects of between the ΣPFCAs and body condition of peregrine falcon nestlings and found that the body condition of peregrine falcon nestlings were significantly and negatively associated with higher ΣPFCA burdens.

16. There is evidence from acute and intermediate oral laboratory studies in rats and mice that the liver is a sensitive target of C9 – C12 toxicity (ATSDR 2021). For example, rats and mice experienced increased relative liver weights, increased hepatic triglycerides and total cholesterol, and altered expression of genes related to lipid metabolism when exposed to 1 mg/kg bw/d of C9 PFCA for 14 days. In addition, at 5 mg/kg bw/d, substantial lipid accumulation in the liver and disrupted hepatic glucose metabolism were noted (Fang et al. 2012a, 2012b, 2012c; Wang et al. 2015). Increased liver weights, and hepatocellular hypertrophy, degeneration, and necrosis were observed in rats exposed for 90 days to a mixture of PFAS (about 74% of which was C9 PFCA). The NOELs were 0.025 mg/kg bw/d for males and 0.125 mg/kg bw/d for females (Mertens et al. 2010). Hepatocyte necrosis and hepatomegaly were observed in rats treated with 0.5 mg/kg bw/d of C10 PFCA for 28 days (Frawley et al. 2018). Exposure to C11 PFCA for 42 days resulted in increased liver weights in male rats at 0.3 mg/kg bw/day and in females at 1.0 mg/kg bw/day, and centrilobular hepatocellular hypertrophy was observed in both males and females at 1.0 mg/kg bw/day (Takahashi et al. 2014). Increased liver weights and hepatotoxicity (liver hypertrophy, necrosis, and inflammatory cholestasis) were noted in rats exposed for 42 days to 0.5 and 2.5 mg/kg bw/d of C12 PFCA respectively (Kato et al. 2015). Exposure to C12 PFCA induced hepatic steatosis in rats exposed to 0.2 mg/kg bw/d for 110 days. Accompanying gene expression studies provided supporting evidence that these liver effects likely occurred as a result of perturbations to fatty acid uptake, lipogenesis, and fatty acid oxidation (Ding et al. 2009).

17. The effects of long-chain PFCAs on the liver is believed to be mediated in part by peroxisome proliferator-activated receptor alpha (PPARα) activation which affects lipid homeostasis by altering the expression of genes involved in fatty acid uptake, activation, and oxidation (Cheng and Klaassen 2008a, 2008b; Maher et al. 2008; Liu et al. 2016; Zhang et al. 2018). However, studies in PPARα-null mice dosed with 10 mg/kg bw/d of C9 PFCA for 10 days also found increases in liver weight, steatosis, and increases in liver triglyceride levels (Das et al. 2017). This suggests that mechanisms other than PPARα activation are also involved.

18. There are indications that exposure to C9 – C11 PFCAs can result in effects on the immune system. In a series of studies examining the immunotoxicity of C9 PFCA, rats and mice were exposed to 1, 3 or 5 mg/kg bw/d for 14 days (Fang et al. 2008; Fang et al. 2009; Fang et al. 2010). Decreased thymus and/or spleen weights were observed in rats and mice typically at ≥3 mg/kg/day. Atrophy of the lymphoid organs were noted and effects on innate immune cell homeostasis were observed in mice as evidenced by decreased percentages of F4/80+ and CD49b+ cells in the spleen of all treated groups and decreases in CD11c+ cells in the 3 and 5 mg/kg bw/d groups (Fang et al. 2008). Thymocyte apoptosis was observed in rats at 5 mg/kg bw/d, likely due to increased serum cortisol and decreased expression of Bcl-2 (which regulates cell death). Increases in pro-inflammatory cytokines were observed at ≥3 mg/kg/day (Fang et al. 2009). C9-induced apoptosis was observed in rat splenocytes and the production of pro-inflammatory and anti-inflammatory cytokines was significantly increased and decreased respectively at 5 mg/kg bw/d (Fang et al. 2010). C9 PFCA also caused marked splenic and thymic atrophy and an altered balance of immune cell populations in the spleen and thymus of mice 14 days after administration of a single i.p. dose of 0.1 mmol/kg-bw (Rockwell et al. 2013). A follow-up study showed that a single high dose of C9 PFCA still had effects on the immune system 28 days later (Rockwell et al. 2017). In a 28-day study, rats were exposed 0.125–0.5 mg/kg/d and mice were exposed to 0.3125–5.0 mg/kg/week C10 PFCA. A reduction in immune cell populations in the spleen of mice was observed at ≥1.25 mg/kg bw/week. However, exposure to C10 PFCA had little effect on humoral- and cell-mediated immunity, developing hematopoietic cells in the bone marrow, or host resistance to influenza virus in either rats or mice (Frawley et al. 2018). Although exposure of rats to 0-25 mg/kg/day C9 and C10 PFCA for 28 days also resulted in thymic atrophy and decreased spleen and thymus weights, these changes were attributed to stress (NTP 2019). Non-obese diabetic mice were exposed during gestation,

lactation and early life to C11 PFCA in drinking water (3, 30 and 300 g/L) to determine the effect on the early stages of diabetes development (an autoimmune disorder). Exposure to C11 PFCA was associated with accelerated development of pancreatic insulitis, decreased peritoneal macrophage phagocytosis and altered splenocyte cytokine secretion, but it did not increase the incidence of diabetes (Bodin et al. 2016).

19. No clear mode of action for the immunotoxic effects of PFAS (including long-chain PFCAs) has been established. Suppressed adaptive immunity may arise from the interaction of PFAS with PPARα which alters cytokine secretion. However, other PPAR-independent mechanisms are also likely involved including the inhibition of NFkB activation, which directly suppresses cytokine production by immune cells (Corsini et al. 2012; Dewitt et al. 2015). Other possible immune toxicity mechanisms include AIM2 inflammasome activation, gene dysregulation, and signal pathway disorders (Liang et al. 2021).

20. Several long-chain PFCAs (C9 – C12, C14 and C18) have been studied for reproductive toxicity in rodents. Effects observed include altered reproductive organ weight, histological changes in reproductive tissues, altered reproductive hormone level and impaired reproductive functions. For example, exposure of male rats and mice to 5 mg/kg bw/d of C9 PFCA for 14 days resulted in decreased serum testosterone levels, increased serum estradiol levels, atrophy of the seminiferous tubules, large vacuoles between the Sertoli cells and spermatogonia in the testes, and alterations in spermatogenesis and testosterone production (Feng et al. 2009, 2010; Singh and Singh 2019a, 2019b). Short term exposure of male rats to C14 resulted in delays in Leydig cell regeneration, reduced serum testosterone level, down-regulated steroidogenic gene/protein expression and lower AKT1 and ERK1/2 phosphorylation (Zhang et al. 2021). In a longer 90-day study, degenerative changes in the seminiferous tubules and adverse effects on sperm parameters and serum levels of testosterone were observed in male mice administered 0.5 mg/kg bw/d of C9 PFCA. A significant decrease in litter size was also noted when unexposed females were mated with males treated with 0.5 mg/kg bw/d of C9 PFCA (Singh and Singh 2018). Multiple histopathologic findings in the testis were noted in rats exposed to 2.5 mg/kg bw/d of C10 PFCA for 28 days (NTP 2019). No significant reproductive findings were noted for rats exposed to C11 or C14 PFCAs in reproductive and development toxicity assays (Takahashi et al. 2014; Hirata-Koizumi et al. 2015). Decreased spermatid and spermatozoa counts in males, as well as a continuous dioestrus in unmated females was observed in rats dosed with 2.5 mg/kg bw/d of C12 PFCA for 42 days. In pregnant females dosed with 2.5 mg/kg bw/d, hemorrhages were observed at the implantation sites and only one female delivered live pups (Kato et al. 2015). Decreased serum testosterone levels were observed in rats treated with 0.2 mg/kg bw/d C12 PFCA for 110 days (Shi et al. 2009). Reduced implantation numbers, reduced total number of born pups and number of live pups occurred only at much higher exposures (1,000 mg/kg bw/d) to C18 PFCA in rats (Hirata-Koizumi et al. 2012).

21. Developmental effects related to long-chain PFCA exposure (C9 – C12, C14, C18) include postnatal mortality, reduced body weight, and developmental delays (eye opening and onset of puberty). For example, surviving pups (20% survival at weaning) born to dams exposed to 5 mg/kg bw/d of C9 PFCA during gestational day (GD) 1-17 experienced decreased postnatal growth and a dose-dependent delay in developmental landmarks (eye opening, preputial separation and vaginal opening) (Das et al. 2015). Delays in eye opening and decreased in pup body weight gain were also observed in offspring of mice dosed at 2 mg/kg bw/d C9 PFCA on GDs 1–18. Notably, these effects were not observed in transgenic mice whose PPARα was functionally knocked out, suggesting this nuclear receptor is involved in mediating C9 PFCA-induced developmental toxicity (Wolf et al. 2010). Decreases in fetal body weight were observed at 1 mg/kg bw/d in the offspring of mice exposed to C10 PFCA (Harris and Birnbaum 1989) and C11 PFCA (Takahasi et al. 2014). In rats exposed to 2.5 mg/kg bw/d of C12 PFCA, only 1 of the 12 dams delivered live pups and decreases in pup body weight gain were noted (Kato et al. 2015). Inhibition of postnatal body weight gain in pups was observed in the offspring of rats exposed to 10 mg/kg bw/d of C14 PFCA (Hirata-Koizumi et al. 2015).

22. Short-term studies performed in rats show that oral (gavage) exposure to C9, C10 and C14 PFCAs can effect the thyroid. Rats exposed up to 25 mg/kg bw/d of C9 or C10 PFCA for 28 days experienced altered thyroid weight and altered thyroid hormone levels (NTP 2019). Levels of T3 and T4 hormones increased 2- and 4-fold in female mice 30 days after being exposed to a single doses of 20 to 80 mg/kg of C10 PFCA (Harris et al. 1989). Follicular cell hypertrophy was noted in the thyroid of male rats exposed to ≥ 3 mg/kg bw/d C14 for 42 days (Hirata-Koizumi et al. 2015).

23. Several epidemiological studies evaluated hepatic endpoints and noted associations between exposure to C9 – C14 PFCAs and increased levels of serum lipid levels and clinical biomarkers of liver function. Associations were strongest for C9 and C10 PFCA whereas studies regarding C11 – C14 PFCAs were either too few in number or the

results were too inconsistent to determine if they also had an effect on serum lipid levels. In its overall analysis of the data, EFSA has concluded that epidemiological studies provide clear evidence for an association between exposure to C9 PFCA and increased serum levels of cholesterol (EFSA 2020). Similarly, the Agency for Toxic Substances and Disease Registry (ATSDR) has indicated that the preponderance of the evidence is suggestive of a link between serum levels of C9 and C10 PFCA and increased serum lipid levels, particularly for total cholesterol and LDL cholesterol (ATSDR 2021). The results of a prospective cohort study from the Faroe Islands, published after these reviews, support their findings. Serum concentrations of C9 and C10 PFCA were measured in 490 children at birth, infancy and childhood. Serum levels at ages five and nine were positively associated with lipid concentrations at age nine (Blomberg et al. 2021). Notably, cholesterol concentrations in childhood are a risk factor for adult cardiovascular disease (Daniels and Greer 2008).

24. Associations between exposure to long-chain PFCAs (C9 – C14) and immunological outcomes, including incidence of infectious diseases, efficacy of vaccinations, asthma and allergic diseases, and immune marker levels (e.g., serum cytokine levels, antibody levels) have been investigated in several epidemiological studies. In humans, the strongest evidence of immunotoxicity comes from investigations into antibody response to vaccines (see Table 10). In its evaluation of the data, ATSDR indicates that there is suggestive evidence of a link between serum C10 PFCA levels and decreased antibody responses to vaccines (ATSDR 2021). This is based largely on studies examining decreased antibody response to diphtheria and tetanus vaccines in children (Grandjean et al. 2012, 2017) and decreased response to diphtheria vaccines in adults (Kielsen et al. 2016). In a systematic review of the literature, Kirk et al. (2018) also concluded there was evidence of a negative association between C10 PFCA and diphtheria antibody levels after vaccination of children or adults. The evidence was considered to be "limited" because some of the studies were on the same cohort in the Faroe Islands, making it difficult to assess the consistency of evidence across populations. Since this systematic review, the results of a study in West African children (with substantially different lifestyles and exposure profiles), were published. The study found a doubling of serum C10 PFCA concentrations in vaccinated children to be associated with 25% lower measles antibody concentrations (Timmerman et al. 2020). In addition, another study in children from Greenland noted that for every 1 ng/g increase in C10 PFCA, the odds of not having protective levels of diphtheria antibodies were increased by 5.08 times (95 % CI: 1.32–19.51) (Timmerman et al. 2022). With respect to other long-chain PFCAs, one study noted reduced diphtheria and tetanus antibody levels in adults in relation to serum concentrations of C11 and C12 PFCA (unadjusted for potential confounders) (Kielsen et al. 2016). Another study noted reduced diphtheria antibody levels in children in relation to serum concentrations of C11 PFCA (Timmermann et al. 2022). In regards to C9 PFCA, the data were mixed with some studies showing associations with a reduced antibody response to vaccines and others not (Grandjean et al. 2012; Granum et al. 2013; Kielsen et al. 2016; Stein et al. 2016a. 2016b; Grandjean et al. 2017; Timmerman et al. 2020, 2022).

Table 10. Associations of long-chain PFCAs and antibody levels after vaccination

Type of Study	Study Population	N	Association with Antibody Response	PFCA	Positive, Negative, or No Association with Antibody Response	Reference
Cohort (INUENDO and IVAAQ)	Children	314	diphtheria and tetanus	C9	Negative associations between diphtheria antibody levels and serum C9 levels (adjusted for confounders). Weak negative association for tetanus antibody levels.	Timmermann et al. 2022
				C10	Negative associations between diphtheria antibody levels and serum C10 levels (adjusted for confounders). Weak negative association for tetanus antibody levels.	
				C11	Negative associations between diphtheria antibody levels and serum C11 levels (adjusted for confounders). Weak negative association for tetanus antibody levels.	
Randomized controlled trial	Children (inclusion, 9 months and 2 years)	237	measles	C9	Significant negative association between measles antibodies and serum C9 levels at 9-month visit after inclusion (adjusted analyses). Non-significant negative association at 2-year visit.	Timmermann et al. 2020

Type of Study	Study Population	N	Association with Antibody Response	PFCA	Positive, Negative, or No Association with Antibody Response	Reference
				C10	Significant negative association between measles antibodies and serum C10 levels at 9-month visit after inclusion (adjusted analyses). Non-significant negative association at 2-year visit.	
				C11	Significant negative association between measles antibodies and serum C11 levels at 9-month visit after inclusion (adjusted analyses). Non-significant negative association at 2-year visit.	
Birth Cohort	Children (7 and 13 year old)	516	diphtheria and tetanus	C9	No association for antibody levels at age 13 and C9 levels at age 7 or 13.	Grandjean et al. 2017
				C10	Negative association between diphtheria or tetanus antibody levels at age 13 and serum C10 levels at age 7.	
Birth Cohort	Mother-child pairs	587	diphtheria and tetanus	C9	Significant negative association between C9 and diphtheria antibodies levels at age 5. No associations between maternal or child C9 levels and tetanus antibody levels at ages 5 or 7.	Grandjean et al. 2012
				C10	Negative association between C10 levels and tetanus antibody levels at ages 5 and 7. No association between C10 and diphtheria antibody levels at ages 5 or 7.	
Birth Cohort	Mother-child pairs	56	measles, rubella, tetanus, and Haemophilus influenza type b	C9	Negative association between maternal serum C9 and rubella antibody levels in children of three years. Positive association between maternal C9 and the number of episodes of common cold for the children.	Granum et al. 2013
Cross-sectional	Adults	12	diphtheria and tetanus	C9	Negative associations between diphtheria antibody levels and serum C9 levels. No association for tetanus antibody levels.	Kielsen et al. 2016
				C10	Negative associations between diphtheria antibody levels and serum C10 levels. No association for tetanus antibody levels.	
				C11	Negative associations between serum C11 (not adjusted for potential confounders) and diphtheria and tetanus antibody levels.	
				C12	Negative associations between serum C12 levels (not adjusted for potential confounders) and diphtheria and tetanus antibody levels.	
Cross-sectional (NHANES 1999-2000 and 2003-2004)	Adolescents	1191	Measles, mumps, and rubella	C9	No associations between recent C9 serum levels and measles, mumps, or rubella antibody titers.	Stein et al. 2016a
Cohort	Adults	78	Influenza (FluMist)	C9	No associations between C9 levels and response to influenza vaccine.	Stein et al. 2016b

25. Several epidemiological studies evaluated possible associations between exposure to long-chain PFCAs (C9 – C14) and reproductive outcomes. Overall, there were only a small number of studies for each long-chain PFCA and for each endpoint. A number of epidemiological studies showed either equivocal, null, or potentially protective outcomes. However, several other studies showed positive associations. For example, associations were observed between alterations in reproductive hormones levels in women and adolescents and exposure to C9 – C12 PFCAs (Joensen et al. 2013; Tsai et al. 2015; Zhou et al. 2016, 2017; Heffernan et al. 2018). Some associations were also found between serum C9 and C10 PFCA and sperm parameters (e.g., head length, percentage of sperm with coiled tails) (Buck Louis et al. 2015). In addition, altered female reproductive health (i.e., miscarriage, increased risk of polycystic ovarian syndrome, decreased blastocyst conversion rate) was linked with C9 – C12 PFCAs (Jensen et al. 2015; McCoy et al. 2017; Wang et al. 2019). There is suggestive evidence of associations between exposure to C9 PFCA and issues related to endometriosis, earlier menopause and hysterectomy (Louis et al. 2012; Taylor et al. 2014). However, in terms of the earlier menopause, it's possible that reverse causation could be a factor (i.e., earlier menopause leads to increased PFAS levels, due to decreased elimination through menstruation)..

26. In some studies, reduced birth weight has been associated with exposure to some long-chain PFCAs (Kwon et al. 2016; Lind et al. 2017; Starling et al. 2017; Cao et al. 2018; Gyllenhammar et al. 2018; Shoaff et al. 2018; Wikstrom et al. 2019). For example, median cord blood concentrations of C9 (0.2 ng/mL), C10 (0.1 ng/mL) and C11 (0.3 ng/mL) PFCAs were inversely associated with birth weight in 268 infants that were part of the Ewha Birth and Growth Cohort in South Korea. In the same study, no associations were found for C12 (0.1 ng/mL) and C13 (0.4 ng/mL) PFCAs (Kwon et al. 2016). In the Taiwan Maternal and Infant Cohort Study of 233 maternal-infant pairs, inverse associations were noted between median maternal serum concentrations (taken during third trimester) of C9 (1.6 ng/mL), C10 (0.4 ng/mL), C11 (3.4 ng/mL), and C12 (0.4 ng/mL) PFCAs and birth weight among female infants. (Wang et al. 2016).

27. Concern about the endocrine disrupting properties of PFAS has led to research into the effects on thyroid outcomes, including thyroid hormone levels in infants (in umbilical cord blood, maternal blood and infant blood), children, adults and pregnant women, and thyroid diseases in infants. One study found an association between infant serum concentrations of C9 – C11 PFCAs and an increased incidence of congenital hypothyroidism (Kim et al. 2016). In a systematic review of thyroid outcomes in children and pregnant women a positive association was found between levels of thyroid stimulating hormone (TSH) and C9 PFCA levels in boys ≥ 11 years old (Ballesteros et al. 2017). Various associations were also found between levels of TSH, triiodothyronine (T3), or thyroxine (T4), thyroglobulin, and thyroid peroxidase antibodies in adults, pregnant women, children and infants and levels of C9 – C14 PFCAs (e.g., Ballesteros et al. 2017; Aimuzi et al. 2019; Itoh et al. 2019; Coperchini et al. 2021; ATSDR 2021). However, the associations were not always consistent across studies and a number of investigations identified no associations with effects on the thyroid (ATSDR 2021).

In other studies, associations have been observed between C9 – C11 and C13 PFCAs and reproductive outcomes (shorter anogenital distance, altered hormonal levels, and altered onset of puberty) in infants and children (Lind et al., 2016, 2017; Ernst et al. 2019; Tian et al. 2019; Yao et al. 2019; Jensen et al. 2020). In addition, associations have been noted between C9 – C10 PFCAs and altered bone development (i.e. size, mass, length, and bone density health) in children (Buck Louis et al. 2018; Jeddy et al. 2018; Khalil et al. 2018; Cluett et al. 2019). Associations have also been detected between prenatal or child serum levels of C9 – C12 PFCAs and neurobehavioral and neuropsychological endpoints (i.e. increased attention deficit hyperactivity disorder (ADHD), hyperactivity, risk of personal-social difficulties, and poor executive functions) (Lien et al. 2016; Oulhote et al. 2016; Høyer et al. 2018; Vuong et al., 2018a, 2018b; Niu et al., 2019) as well as cognitive dysfunction (Weng et al. 2020).

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Appendix

List of references used to generate Figure 1 of the Risk Profile and Figures 2 and 3 of this document

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