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Stockholm Convention on Persistent Organic Pollutants

Persistent Organic Pollutants Review Committee
Seventeenth meeting
Geneva, 24–28 January 2022*
Item 4 (c) (iii) of the provisional agenda**

Technical work: consideration of chemicals proposed for listing in Annexes A, B and/or C to the Convention: long-chain perfluorocarboxylic acids, their salts and related compounds

Proposal to list long-chain perfluorocarboxylic acids, their salts and related compounds in Annexes A, B and/or C to the Stockholm Convention on Persistent Organic Pollutants

Note by the Secretariat

I. Introduction

1. Canada has submitted a proposal to list long-chain perfluorocarboxylic acids (PFCAs), their salts and related compounds in Annexes A, B and/or C to the Stockholm Convention on Persistent Organic Pollutants, pursuant to paragraph 1 of Article 8 of the Convention (see annex to the present note). The proposal is being circulated as submitted and has not been formally edited. The Secretariat's verification of whether the proposal contains the information specified in Annex D to the Convention is set out in document UNEP/POPS/POPRC.17/INF/6.

II. Proposed action

2. The Committee may wish:

(a) To consider the information provided in the present note;

(b) To decide whether it is satisfied that the proposal fulfils the requirements of Article 8 of and Annex D to the Convention;

(c) To develop and agree on, if it decides that the proposal fulfils the requirements referred to in subparagraph 2 (b) above, a workplan for preparing a draft risk profile pursuant to paragraph 6 of Article 8 of the Convention.

* Subject to a final decision by the Bureau of the Persistent Organic Pollutants Review Committee in October 2021, taking into account the situation regarding the coronavirus disease (COVID-19) pandemic.

** UNEP/POPS/POPRC.17/1.

Annex[1]

Proposal to list long-chain perfluorocarboxylic acids (PFCAs), their salts and related compounds in Annexes A, B and/or C to the Stockholm Convention on Persistent Organic Pollutants

1. Introduction

1. Long-chain perfluorocarboxylic acids (PFCAs), their salts and related compounds are members of the per- and polyfluoroalkyl substances (PFAS) chemical class. Long-chain PFCAs and their salts are infrequently used in products. Nonetheless, the ammonium salt of C9 PFCA was identified as being used for surfactant applications and in the production of fluoropolymers. Substances that are related compounds to long-chain PFCAs have, however, been used in a range of applications, including in coating products, fabric/carpet protectors, textile impregnation agents and firefighting foams. C9 – C14 PFCAs, their salts and related compounds may also be unintentionally produced during the manufacturing of PFAS, including those containing a carbon chain of less than nine carbon atoms, and in other industrial processes. As a result, long-chain PFCAs may be present in certain products and articles as impurities.

2. C9 – C14 PFCAs are found ubiquitously in the environment and wildlife, including in remote areas, and have been measured in human biomonitoring data collected around the world. C15 to C21 PFCAs have also been measured in environmental media (i.e., soil and snow samples from Sweden, coastal surface seawater samples from Antarctica) (Plassman and Berger 2013; Cai et al. 2012b), whereas C16 and C18 PFCAs have been measured in top predator wildlife species (e.g., polar bears and peregrine falcons) (Gebbink et al. 2009; Greaves et al. 2012, 2013; Boisvert et al. 2019; Gebbink and Letcher 2012; Sun et al. 2020).

3. This proposal specifically addresses the information requirements and screening criteria of Annex D in the Stockholm Convention on Persistent Organic Pollutants (POPs) and summarizes relevant evidence relating to the screening criteria for persistence, bioaccumulation, long-range environmental transport and adverse effects. The proposal is based on risk assessments for long-chain PFCAs prepared by Environment and Climate Change Canada (Environment Canada 2012) and the European Union (EU) (ECHA 2018a,b, 2020), information from peer-reviewed scientific journals as well as grey literature.

2. Chemical identity

4. This proposal concerns the PFCAs with carbon chain lengths from 9 to 21 inclusive, their salts and related compounds. Long-chain PFCAs and their salts are a homologous series of substances with the molecular formula of $C_nF_{2n+1}CO_2H$ (where $8 \leq n \leq 20$). “Perfluorinated” refers to fluorochemicals in which the hydrogen atoms directly attached to the carbon atoms are all replaced with fluorine atoms. The term “polyfluorinated” indicates that only some of the hydrogen atoms have been replaced with fluorine atoms.

5. This proposal broadly defines related compounds as any substance that is a precursor and may degrade or transform to long-chain PFCAs, where the perfluorinated alkyl moiety has the formula C_nF_{2n+1} (where $8 \leq n \leq 20$) and is directly bonded to any chemical moiety other than a fluorine, chlorine or bromine atom. Environment and Climate Change Canada has identified a non-exhaustive list of perfluoroalkyl compounds as being part of the group of long-chain PFCAs, their salts and related compounds (Environment Canada 2012). ECHA (2018a,b) identified several C9 – C14 PFCA-related substances which they considered to have the potential to degrade or be transformed to C9 – C14 PFCAs. Related compounds to long-chain PFCAs include fluorotelomer alcohols (FTOHs) and

[1] The information in the proposal is circulated as submitted to the Secretariat. The views expressed in this proposal do not necessarily reflect the views of the Secretariat, the United Nations Environment Programme (UNEP) or the United Nations (UN). The designations employed and the presentation of the material do not imply the expression of any opinion whatsoever on the Secretariat, UNEP or the UN concerning geo-political situations or the legal status of any country, territory, area, city or their authorities.

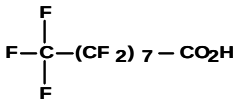
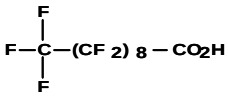
fluorotelomer derivatives[2], including side-chain fluorinated polymers and polyfluoroalkyl phosphoric acid mono-/diesters (monoPAPs/diPAPs).

6. Table 1 lists the chemical identity of the long-chain PFCAs. Table 2 lists the available experimental and calculated physical and chemical data for this group. Information relating to the physicochemical properties is limited to the C9 – C15 chain lengths.

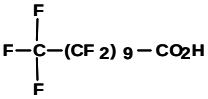
7. Linear isomers appear to be predominant for long-chain PFCAs detected in biota as they may have significantly slower elimination rates and/or may be present at higher exposure concentrations than branched isomers (Conder et al. 2008). It has been suggested that carbon-carbon conformation changes occur as the chain length increases, with longer chains becoming more helical and where the chain may fold back on itself or not be completely linear, resulting in smaller cross-sectional diameter molecules (Wang and Ober 1999, Ellis et al. 2004). The chain conformation is described as being zigzag for chain lengths of 8 carbons and less, a mixture of zigzag and helical for chain lengths of 8 to 12 carbons, and fully helical for chain lengths greater than 12 carbon atoms (Ellis et al. 2004). The conformation changes and increasing length of the perfluorinated chain may also make the molecule more hydrophobic with decreasing water solubility, which can also contribute to increased bioavailability and bioaccumulation for the long-chain PFCAs (Ellis et al. 2004; Hurley et al. 2004).

8. Fluorotelomers are a subgroup of perfluorinated substances that are produced by a process called telomerization and can occur in a range of fluorocarbon chain lengths. FTOHs are not fully fluorinated, since they have a 2-carbon hydrocarbon chain linked to the perfluorinated carbon chain (Environment Canada 2012). FTOHs with x number of perfluorinated carbons produce intermediates such as fluorotelomer unsaturated carboxylates (x:2 FTUCA) and fluorotelomer carboxylic acids (x:2 FTCA) that can further degrade to long-chain PFCAs (Environment Canada 2012). Substances containing $F(CF_2)_n(CH_2)_{2-}$ groups can also be considered potential related compounds to C9 – C14 PFCAs as they will likely result in the release of x:2 FTOHs in the environment (ECHA 2018a,b).

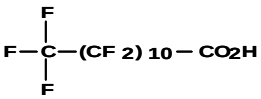
Table 1: Chemical identity of long-chain PFCAs

C9 PFCA	
CAS number:	375-95-1
IUPAC name:	2,2,3,3,4,4,5,5,6,6,7,7,8,8,9,9,9-heptadecafluorononanoic acid
Chemical Abstracts Index name:	Nonanoic acid, heptadecafluoro-
Molecular formula:	$C_9HF_{17}O_2$
Structural formula:	
Synonyms:	PFNA; C 1800; Heptadecafluorononanoic acid; Perfluorononanoic acid; Perfluoropelargonic acid
C10 PFCA	
CAS number:	335-76-2
IUPAC name:	2,2,3,3,4,4,5,5,6,6,7,7,8,8,9,9,10,10,10-nonadecafluorodecanoic acid
Chemical Abstracts Index name:	Decanoic acid, nonadecafluoro-
Molecular formula:	$C_{10}HF_{19}O_2$
Structural formula:	
Synonyms:	PFDA; Nonadecafluoro-n-decanoic acid; Nonadecafluorodecanoic acid; Perfluoro-n-decanoic acid; Perfluorocapric acid; Perfluorodecanoic acid
C11 PFCA	

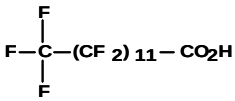
[2] The following fluorotelomer derivatives were identified as C9 – C14 PFCA precursors: fluorotelomer iodide, fluorotelomer stearate monoester/fluorotelomer citrate trimester, polyfluorinated olefins, silanes and amides, fluorotelomer (meth)acrylates, polyfluoroalkyl phosphoric acid mono-/diesters (monoPAP/diPAP), fluorotelomer urethane (monomers), ethoxylates and sulfonate, fluorotelomer thioether amido sulfonate, perfluoroalkyl phosphinic acids and side-chain fluorinated polymers (ECHA 2018b).

CAS number:	2058-94-8
IUPAC name:	2,2,3,3,4,4,5,5,6,6,7,7,8,8,9,9,10,10,11,11,11-henicosafluoroundecanoic acid
Chemical Abstracts Index name:	Undecanoic acid, heneicosafluoro-
Molecular formula:	$C_{11}HF_{21}O_2$
Structural formula:	
Synonyms:	PFUnDA; Heneicosafluoroundecanoic acid; Perfluoroundecanoic acid; Perfluoroundecylic acid; PFUnA

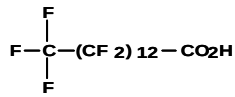
C12 PFCA

CAS number:	307-55-1
IUPAC name:	2,2,3,3,4,4,5,5,6,6,7,7,8,8,9,9,10,10,11,11,12,12,12-tricosafluorododecanoic acid
Chemical Abstracts Index name:	Dodecanoic acid, tricosafluoro-
Molecular formula:	$C_{12}HF_{23}O_2$
Structural formula:	
Synonyms:	PFDoDA; Perfluorododecanoic acid; Perfluorolauric acid; PFDoA

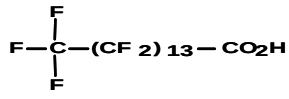
C13 PFCA

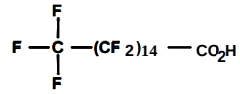
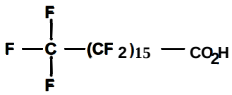
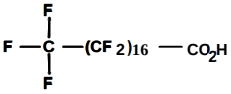
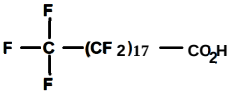
CAS number:	72629-94-8
IUPAC name:	2,2,3,3,4,4,5,5,6,6,7,7,8,8,9,9,10,10,11,11,12,12,13,13,13-pentacosafluorotridecanoic acid
Chemical Abstracts Index name:	Tridecanoic acid, pentacosafluoro-
Molecular formula:	$C_{13}HF_{25}O_2$
Structural formula:	
Synonyms:	PFTrDA; Perfluorotridecanoic acid; PFTrA

C14 PFCA

CAS number:	376-06-7
IUPAC name:	2,2,3,3,4,4,5,5,6,6,7,7,8,8,9,9,10,10,11,11,12,12,13,13,14,14,14-heptacosafluorotetradecanoic acid
Chemical Abstracts Index name:	Tetradecanoic acid, heptacosafluoro-
Molecular formula:	$C_{14}HF_{27}O_2$
Structural formula:	
Synonyms:	PFTDA; Perfluoromyristic acid; Perfluorotetradecanoic acid; PFTeA

C15 PFCA

CAS number:	141074-63-7
IUPAC name:	2,2,3,3,4,4,5,5,6,6,7,7,8,8,9,9,10,10,11,11,12,12,13,13,14,14,15,15,15-nonacosafluoropentadecanoic acid
Chemical Abstracts Index name:	Pentadecanoic acid, nonacosafluoro-
Molecular formula:	$C_{15}HF_{29}O_2$
Structural formula:	

Synonyms:	PFPeDA; Perfluoropentadecanoic acid
C16 PFCA	
CAS number:	67905-19-5
IUPAC name:	2,2,3,3,4,4,5,5,6,6,7,7,8,8,9,9,10,10,11,11,12,12,13,13,14,14,15,15,16,16,16-hentriacontafluorohexadecanoic acid
Chemical Abstracts Index name:	Hexadecanoic acid, hentriacontafluoro-
Molecular formula:	$C_{16}HF_{31}O_2$
Structural formula:	
Synonyms:	PFHxDA; Perfluoropalmitic acid; Perfluorohexadecanoic acid; Hexadecanoic acid
C17 PFCA	
CAS number:	57475-95-3
IUPAC name:	2,2,3,3,4,4,5,5,6,6,7,7,8,8,9,9,10,10,11,11,12,12,13,13,14,14,15,15,16,16,17,17,17-tritriacontafluoroheptadecanoic acid
Chemical Abstracts Index name:	Perfluoroheptadecanoic acid
Molecular formula:	$C_{17}HF_{33}O_2$
Structural formula:	
Synonyms:	PFHpDA
C18 PFCA	
CAS number:	16517-11-6
IUPAC name:	2,2,3,3,4,4,5,5,6,6,7,7,8,8,9,9,10,10,11,11,12,12,13,13,14,14,15,15,16,16,17,17,18,18,18-pentatriacontafluorooctadecanoic acid
Chemical Abstracts Index name:	Octadecanoic acid, pentatriacontafluoro-
Molecular formula:	$C_{18}HF_{35}O_2$
Structural formula:	
Synonyms:	PFODA; Perfluorostearic acid; Perfluorooctadecanoic acid; Octadecanoic acid
C19 PFCA	
CAS number:	133921-38-7
IUPAC name:	2,2,3,3,4,4,5,5,6,6,7,7,8,8,9,9,10,10,11,11,12,12,13,13,14,14,15,15,16,16,17,17,18,18,19,19,19-heptatriacontafluorononadecanoic acid
Chemical Abstracts Index name:	Perfluorononadecanoic acid
Molecular formula:	$C_{19}HF_{37}O_2$
Structural formula:	
Synonyms:	PFNDA
C20 PFCA	
CAS number:	68310-12-3
IUPAC name:	2,2,3,3,4,4,5,5,6,6,7,7,8,8,9,9,10,10,11,11,12,12,13,13,14,14,15,15,16,16,17,17,18,18,19,19,20,20,20-nonatriacontafluoroicosanoic acid
Chemical Abstracts Index name:	Perfluoroeicosanoic acid

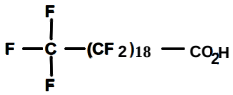
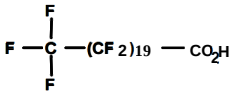
Molecular formula:	$C_{20}HF_{39}O_2$
Structural formula:	
Synonyms:	Eicosanoic acid, nonatriacontafluoro- (9CI); Nonatriacontafluoroeicosanoic acid
C21 PFCA	
CAS number:	–
IUPAC name:	–
Chemical Abstracts Index name:	Heneicosanoic acid, 2,2,3,3,4,4,5,5,6,6,7,7,8,8,9,9,10,10,11,11,12,12,13,13,14,14,15,15,16,16,17,17,18,18,19,19,20,20,21,21,21-hentetracontafluoro-
Molecular formula:	$C_{21}HF_{41}O_2$
Structural formula:	
Synonyms:	Perfluoroheneicosanoic acid

Table 2: Overview of available physicochemical properties of long-chain PFCAs

Property	*Value*	*Type*	*Reference*
C9 PFCA			
Molecular mass (g/mol)	464.08	–	–
Melting point (°C)	77	Experimental	Fontell and Lindman 1983
	71	Experimental	Blancou et al. 1976
	71-72	Experimental	Herbst et al. 1985
	65	Experimental	Beneficemalouet et al. 1991
	59.3-61.1	Experimental	Kunieda and Shinoda 1976
	69-71	Experimental	Ishikawa et al. 1983
Boiling point (°C)	203.4	Calculated	Kaiser et al. 2005
Vapour pressure (Pa) at 25°C	0.10 (log p_L*)	Experimental	Arp et al. 2006
	1.1 – 99.97 kPa (at 99.6 – 203°C)	Calculated	Kaiser et al. 2005
Water solubility	< 0.2 percent weight at 60°C	Experimental	Fontell and Lindman 1983[1]
	1.3 g/L (critical micelle concentration)	Experimental	Kunieda and Shinoda 1976[1]
pK_a (dimensionless)	< 0.8	Calculated	Goss 2008
log K_{oc} (dimensionless)	2.3 – 2.48	Experimental	Higgins and Luthy 2006
C10 PFCA			
Molecular mass (g/mol)	514.08	–	–
Melting point (°C)	87.4-88.2	Experimental	Bernett and Zisman 1959
	87.4-88.2	Experimental	Bernett and Zisman 1959
	83.5-85.5	Experimental	Mukerjee and Handa 1981
	76.5	Experimental	Ikawa et al. 1988
	87.4-88.2	Experimental	Hare et al. 1954
Boiling point (°C)	218	Experimental	Kauck and Diesslin 1951
	219.4	Calculated	Kaiser et al. 2005
	203.4	Calculated	Kaiser et al. 2005
	218	Experimental	Sigma Aldrich 2004
Vapour pressure (Pa) at 25°C	-0.64 (log p_L*)	Experimental	Arp et al. 2006
	3.1 – 99.97 kPa (at 129.6 – 218.9°C)	Calculated	Kaiser et al. 2005
Water solubility (g/L)	5.14	Experimental	Kauck and Diesslin 1951
	0.40 (critical micelle concentration)	Experimental	Bernett and Zisman 1959[1]

Property	*Value*	*Type*	*Reference*
	0.46 (critical micelle concentration at 30°C)	Experimental	Klevens and Raison 1954[1]
pKa (dimensionless)	2.58	Calculated	Moroi et al. 2001
log K_{oc} (dimesionless)	2.65 – 2.87	Experimental	Higgins and Luthy 2006
C11 PFCA			
Molecular mass (g/mol)	564.1	–	–
Melting point (°C)	112-114	Experimental	Huang et al. 1987
	97.9-100.3	Experimental	Kunieda and Shinoda 1976
Boiling point (°C)	238.4 at 101.325 kPa	Calculated	Kaiser et al. 2005
Vapour pressure (Pa) at 25°C	-0.98 (log p_L*)	Experimental	Arp et al. 2006
	0.6 – 99.97 kPa (at 112 – 237.7°C)	Calculated	Kaiser et al. 2005
Water solubility (g/L) at 25°C	1.2E-4; pH 1	Calculated	ECHA 2018b
	9.0E-4; pH 2	Calculated	ECHA 2018b
	8.5E-3; pH 3	Calculated	ECHA 2018b
	0.056; pH 4	Calculated	ECHA 2018b
	0.14; pH 5	Calculated	ECHA 2018b
	0.16; pH 6-10	Calculated	ECHA 2018b
pKa (dimensionless)	0.52 ± 0.10	Calculated	ECHA 2018b
log K_{oc} (dimesionless)	3.19 – 3.41	Experimental	Higgins and Luthy 2006
C12 PFCA			
Molecular mass (g/mol)	614.1	–	–
Melting point (°C)	112.6 – 114.7	Experimental	Bernett and Zisman 1959
	112.6-114.7	Experimental	Hare et al. 1954
	112-114	Experimental	Huang et al. 1987
Vapour pressure	0.9 – 99.96 kPa (at 127.6 – 247.7°C)	Calculated	Kaiser et al. 2005
Water solubility (g/L) at 25°C	4.81E-6 mg/L	Estimated	ECHA 2018b
	2.9E-5; pH 1	Calculated	ECHA 2018b
	2.2E-4; pH 2	Calculated	ECHA 2018b
	2.0E-3; pH 3	Calculated	ECHA 2018b
	0.014; pH 4	Calculated	ECHA 2018b
	0.034; pH 5	Calculated	ECHA 2018b
	0.039; pH 6	Calculated	ECHA 2018b
	0.040; pH 7	Calculated	ECHA 2018b
	0.041; pH 8-10	Calculated	ECHA 2018b
pKa (dimensionless)	0.52 ± 0.10	Calculated	ECHA 2018b
C13 PFCA			
Molecular mass (g/mol)	664.0989	–	–
Melting point (°C)	117.5-122	Experimental	Kunieda and Shinoda 1976
Boiling point (°C)	260.7±35.0 °C (at 101.32 kPa)	Calculated	ECHA 2018b
Vapour pressure (Pa) at 25°C	0.479	Calculated	ECHA 2018b
Water solubility (g/L) at 25°C	7.3E-6; pH 1	Calculated	ECHA 2018b
	5.5E-5; pH 2	Calculated	ECHA 2018b
	5.1E-4; pH 3	Calculated	ECHA 2018b
	3.5E-3; pH 4	Calculated	ECHA 2018b
	8.6E-3; pH 5	Calculated	ECHA 2018b
	0.0100; pH 6-10	Calculated	ECHA 2018b
pKa (dimensionless)	0.52 ± 0.10	Calculated	ECHA 2018b
C14 PFCA			
Molecular mass (g/mol)	714.12	–	–
Melting point (°C)	130.4	Experimental	Lehmler et al. 2001
	130	Experimental	Kunieda and Shinoda 1976
Vapour pressure (Pa) at 25°C	0.183	Calculated	ECHA 2018b
	1.9E-6; pH 1	Calculated	ECHA 2018b

Property	*Value*	*Type*	*Reference*
Water solubility (g/L) at	1.4E-5; pH 2	Calculated	ECHA 2018b
25°C	1.3E-4; pH 3	Calculated	ECHA 2018b
	9.3E-4; pH 4	Calculated	ECHA 2018b
	2.2E-3; pH 5	Calculated	ECHA 2018b
	2.6E-3; pH 6-10	Calculated	ECHA 2018b
pKa (dimensionless)	0.52 ± 0.10	Estimated	Wang et al. 2011
C15 PFCA			
Molecular mass (g/mol)	764.1129	–	–

[1] Solubility values refer to an aqueous phase containing a mixture of protonated acid and perfluorocarboxylate anion, at an "autogenous" pH. If the pH is reduced by addition of, for example, a mineral acid, the proportion of protonated acid will increase and the overall solubility will decrease.

Abbreviations: K_{oc}, organic carbon partition coefficient; pK_a, acid dissociation constant, p_L*: saturated subcooled liquid vapor pressure.

3. Global consumption and use of long-chain PFCAs, their salts and related compounds

9. There are no known natural sources of long-chain PFCAs, their salts and related compounds (Kissa 1994). Their presence in the environment is due solely to human activity. Long-chain PFCAs can be released to the environment from direct and indirect sources. Direct sources result from the manufacture and, to a lesser extent, the use of PFCAs, while indirect sources are those where long-chain PFCAs are present as chemical reaction impurities in products or where related compounds to the long-chain PFCAs emitted to the environment have degraded to long-chain PFCAs through biotic or abiotic degradation (ECHA 2018b; Environment Canada 2012).

10. Based on available information, long-chain PFCAs and their salts are infrequently used intentionally in products. Nonetheless, the ammonium salt of C9 PFCA (ammonium perfluorononanoate or APFN) was identified as being used for surfactant applications and in the production of fluoropolymers, primarily polyvinylidene fluoride (PVDF) (Prevedouros et al. 2006). Fluoropolymers, such as PVDF, have many applications including use in cables, wires and electronics, as fire- or weather-resistant coatings for materials in construction-related applications, in the pulp and paper industry, and in nuclear waste processing (Banks 1994; Ebnesajjab 2013). The manufacture of APFN leads to a different mixture of PFCAs; Armitage et al. (2009) described the homologue profile for commercial APFN to consist primarily of C9 PFCA (73.6%), C11 PFCA (20.0%) and C13 PFCA (5.0%). Available information indicates that C9 – C14 PFCAs may also be used in electronic articles (e.g., semiconductors) and medical devices (i.e., UV-hardened dental restorative materials, manufacturing of contact lenses) (Swedish Chemicals Agency 2015; ECHA 2018b).

11. Related compounds to long-chain PFCAs, such as FTOHs and side-chain fluorinated polymers, have been reported to be used in a range of applications, including use in coating products, fabric protectors, textile impregnation agents, firefighting foams, carpet protectors, cleaning products, polishing agents, products for motor vehicle repair, paints, lacquers and varnishes (Banks 1994; Nordic Council of Ministers 2015, as summarized in NICNAS, 2019). Additionally, fluorotelomer epoxides, olefins or alcohols are used as building blocks in the production of fluorotelomer-based substances. These substances are used in commercial products to provide oil-, grease-, water- and stain-repellent properties to other substrates. Some fluorotelomer-based substances can be further exploited as monomers to generate polymeric fluorotelomer substances with the same characteristic properties (Environment Canada 2012; Kannan et al. 2011).

12. In addition, C9 – C14 PFCAs, their salts and related compounds may be unintentionally produced during the manufacturing of PFAS, including those containing a carbon chain of less than nine carbon atoms. During the manufacturing of the perfluorohexanoic acid- (C6 PFCA) based substances, the fraction containing mainly long-chain PFCAs (referred to as the C8-faction) 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 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 are also 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), and in other industrial processes, such as the manufacture of polytetrafluoroethylene (PTFE) powders and the polymerisation of fluoropolymers (ECHA 2018b, 2020). As a result, long-chain PFCAs may be present in certain products and articles as impurities.

13. Long-chain PFCAs and related compounds have been detected in various products. C9 – C14 PFCAs and related compounds, diPAPs and FTOHs (10:2 and 8:2), were detected in paper-based food contact materials (Schaider et al. 2017; Xenia et al. 2011; Vestergren et al. 2015; Kotthoff et al. 2015). FTOHs (10:2 and 8:2) have also been found in and as emissions from non-stick cookware (Sinclair et al. 2007; Herzke et al. 2012) and in certain cleaning agents (Kotthoff et al. 2015; Dinglasan-Panlilio and Mabury 2006). C4 – C14 PFCAs and FTOHs (8:2 and/or 10:2) were detected in building materials, such as coatings and foil for facades or glass-substituents (Janousek et al. 2019; Bečanová et al. 2016). FTOHs (8:2 and/or 10:2) were measured in industrially applied polymeric (carpet protector) and surfactants (used in caulks, paints, coatings, adhesives, floor waxing) (Dinglasan-Panlilio and Mabury 2006). Gewurtz et al. (2009) found C9 – C14 PFCAs and FTUCAs (10:2 and 8:2) in window films. C9 – C14 PFCAs and FTOHs (10:2 and 8:2) have been detected in aqueous film-forming foams (Herzke et al. 2012; Swedish Chemicals Agency 2015), and C9 – C21 PFCAs have been measured in ski waxes or their raw materials (Kotthoff et al. 2015; Plassman and Berger 2013). C9 – C14 PFCAs and some related compounds, including diPAPS, were reported to be found in cosmetics, sun creams and/or body lotions (reviewed in ECHA 2018b). In addition, long-chain PFCAs (C9 – C14, C16), FTOHs (10:2 and 8:2), FTCAs and FTUCAs have been detected in apparel, including in adults and/or children outerwear and baby/children's bibs (Gremmel et al. 2016; Berger and Herzke 2006; Commission for Environmental Cooperation 2017). A study conducted by Kotthoff et al. (2015) also reported detections of C9 – C14 PFCAs and FTOHs (10:2 and 8:2) in outdoor textiles (e.g., jackets, gloves). An analysis of the same samples conducted for a limited number of items indicated a correlation between FTOH (10:2 and 8:2) and PFCA (C10 and C8) concentrations (r=0.957; p=0.0013) (Kotthoff et al. 2015). C9 – C14 PFCAs and/or FTOHs (10:2 and 8:2) have also been detected in home textiles (e.g., curtains, bed covers/linens, quilts, carpets) (Commission for Environmental Cooperation 2017; Vestergren et al. 2015; Herzke et al. 2012), as well as impregnation/water proofing agents (Herzke et al. 2012; Kotthoff et al. 2015). C4 – C14 PFCAs and/or FTOHs (8:2 and/or 10:2) were also measured in other types of fabric/textiles (i.e., awning, seat cover for public transportation, maritime application) (Janousek et al. 2019).

14. Estimates of the global production and consumption of long-chain PFCAs, their salts and related compounds have been reported in the literature. Worldwide total manufacturing volumes of APFN for the production of PVDF for the years 1975 to 2004 was estimated to be in the range of 800 to 2300 tonnes (Prevedourous et al. 2006). For the year 2004, APFN volumes were estimated to range between 15 and 75 tonnes (PERFORCE 2004; Posner et al. 2009). Wang et al. (2014) estimated the APFN usage in Japan, Western Europe and the US to range between 8 and 107 tonnes per year for the years 1975 to 2015. No current intentional manufacturing or use of C9 – C14 PFCAs, their salts or related compounds has been identified in the EU (ECHA 2018b). These substances were reported as being mainly manufactured unintentionally during the manufacturing of PFCAs containing a carbon chain length of less than nine carbon atoms. Based on two industry surveys conducted under the authority of the *Canadian Environmental Protection Act*, 1999 (CEPA) (Canada 1999) for the years 1997–2000, and 2004, long-chain PFCAs were not reported to be manufactured or imported into Canada. However, in both surveys, between 1 and 100 tonnes of a number of related compounds to the long-chain PFCAs were reported to be imported into Canada (Environment Canada 2001, 2005). In addition, substances imported within manufactured items, incidentally or not, were not accounted for as they were not reported through these surveys.

15. In response to a survey conducted by the OECD for the year 2009 (OECD 2011), four companies in two countries[3] reported manufacture of long-chain PFCAs, their salts and related compounds, and fluorotelomers (10:2 – 18:2). Nine long-chain PFCAs (C9 – C12), their salts and related compounds, and fourteen fluorotelomers (10:2 – 18:2), were reportedly contained in products, whether as part of the formulation or as residue (impurity). Total volumes of < 1.5 tonnes of long-chain PFCAs and related compounds in products, and < 15 tonnes of longer-chain fluorotelomers and related compounds in products, were reported. The majority of the substances were reported to have uses as raw materials (for surface treatment agents, water/oil repellent and soil repellent), fluoropolymer polymerisation aids, or manufacturing intermediates (OECD 2011).

16. Worldwide production of fluorotelomers was estimated at approximately 9100 tonnes (reported as 20 million pounds) in 2006, and the United States (US) was considered to account for more than 50 percent of the production (US EPA 2009). Textiles and apparel were considered to account for approximately 50 percent of the volume, with carpet and carpet care products accounting for the next largest share in consumer product uses. Coatings, including those for paper products, were identified as the third largest category of consumer product uses (US EPA 2009). For the years 2012 to 2015, annual national aggregate production volumes of < 454 tonnes were reported in the US for

[3] The names of the companies or Countries were not specified in the OECD report.

each of the FTOHs (8:2, 10:2, 12:2 and 14:2) (CDR 2020). Wang et al. (2014) estimated the global annual production of fluorotelomer-based products to range between 2500 and 20 000 tonnes for the years 1961 to 2004 and was estimated at 45 000 tonnes per year for the period 2005 to 2030.

17. Information collected for the years 2004 and 2005 indicate that eight products containing related compounds to long-chain PFCA (i.e., used for automotive painting, glass treatment and ink cartridges, or as water/oil repellents for textiles, carpets and masonry/cement surfaces) were imported into Australia during that period, for a total volume of up to 33 tonnes (reported as 33 300 kg) per annum (NICNAS 2019). Two compounds related to PFCAs were also imported into Australia in 2005: a perfluorinated furan compound used as an analytical reagent (0.00025 tonnes) and a polymer containing a perfluoroalkylethyl ester moiety used to formulate coatings for wood boards of internal wall cladding (0.15 tonnes).

4. National and international administrative actions on long-chain PFCAs

18. In 2009, the United States Environmental Agency (US EPA) published an Action Plan for addressing potential concerns with long-chain perfluorinated chemicals (PFCs), including long-chain PFCAs[4], and identified long-chain PFCAs as persistent, bioaccumulative and toxic (PBT) (US EPA 2009). In July 2020, the US EPA released its final rule regarding a Significant New Use Rule (SNUR) under the Toxic Substances Control Act (TSCA) for long-chain perfluoroalkyl carboxylate and perfluoroalkyl sulfonate chemical substances. The final rule amends previous SNURs for these substances and requires manufacturers or importers of long-chain perfluoroalkyl carboxylate (PFAC) chemical substances, their salts and precursors to notify the US EPA before conducting certain activities[5] (US EPA 2020).

19. In Canada, an ecological risk assessment for long-chain PFCAs, their salts and their precursors was published in 2012 (Environment Canada 2012). The assessment concluded that these substances are entering or may enter the environment in a quantity or concentration or under conditions that have or may have an immediate or long-term harmful effect on the environment or its biological diversity. Consequently, long-chain PFCAs, their salts and their precursors[6] were placed on Schedule 1 – List of Toxic Substances of CEPA. Since 2016, the *Prohibition of Certain Toxic Substances Regulations, 2012* (Canada 2012) prohibit the manufacture, use, sale, offer for sale or import of long-chain PFCAs, their salts and their precursors, and products containing them, with a limited number of exemptions. A consultation document, proposing regulatory amendments to these Regulations to further restrict long-chain PFCAs their salts and their precursors in Canada, was published in December 2018 (Canada 2018).

20. In Australia, the National Industrial Chemicals Notification and Assessment Scheme (NICNAS) developed an action plan to assess and manage chemicals that may degrade to PFCAs, perfuoroalkyl sulfonates and similar chemicals (NICNAS 2020). NICNAS has published tier II human health and environmental risk assessments of indirect precursors to long-chain PFCAs (NICNAS 2017 and NICNAS 2019, respectively). The precursors in this group were assessed as having the potential to cause adverse outcomes for the environment and human health. Consequently, it was recommended that NICNAS consult with industry and other stakeholders to consider strategies, including regulatory mechanisms available under the *Industrial Chemicals (Notification and Assessment) Act 1989* to encourage the use of safer chemistry (NICNAS 2017, 2019).

[4] Long-chain PFCAs are encompassed by the term “long-chain perfluoroalkyl carboxylate (PFAC)” in the US EPA Action Plan. The PFAC sub-category is broader and includes PFOA and other higher homologues.

[5] This final SNUR will require persons to notify EPA at least 90 days before commencing the manufacturing (including importing) or processing of a subset of long-chain PFAC chemical substances for any use that was not ongoing after December 31, 2015; the manufacturing (including importing) or processing of all other long-chain PFAC chemicals substances for which there were no ongoing uses as of January 21, 2015 (the date of the original 2015 proposal); the import of a subset of long-chain PFAC chemicals as part of a surface coating on articles.

[6] Perfluorocarboxylic acids that have the molecular formula $C_nF2_{n+1}CO_2H$ in which $8 \leq n \leq 20$ and their salts, and compounds that consist of a perfluorinated alkyl group that has the molecular formula C_nF2_{n+1} in which $8 \leq n \leq 20$ and that is directly bonded to any chemical moiety other than a fluorine, chlorine or bromine atom were added to Schedule 1.

21. In Norway, long-chain PFCAs (C9 – C14) were included on the national priority list (ECHA 2017; OECD 2020). In the EU, C9 and C10 PFCAs and their salts are classified[7] according to the Globally Harmonized System (GHS) criteria provided under Regulation (EC) No 1272/2008 on classification, labelling and packaging of substances and mixtures (ECHA 2018b). In addition, six long-chain PFCAs and their salts were identified as Substances of Very High Concern (SVHC) and added to the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH) Candidate List, as they were identified as persistent, bioaccumulative and toxic (PBT) and toxic for reproduction (C9 and C10 PFCAs), or very persistent and very bioaccumulative (vPvB) (C11 – C14 PFCAs) (ECHA 2018b). In 2017, Germany and Sweden submitted a joint proposal for restrictions on the manufacture, placing on the market and use of some of the long-chain PFCAs (C9 – C14), their salts and related compounds in the EU (ECHA 2017). A number of derogations from these restrictions on C9 – C14 PFCAs were proposed based on opinions on the restriction proposal to the European Commission prepared by the European Chemicals Agency (ECHA) Committee for Risk Assessment (RAC) and Committee for Socio-economic Analysis (SEAC) (ECHA 2018a, 2020).The RAC and SEAC opinion notes that, due to the upcoming restriction of PFOA, its salts and related compounds within the EU, an EU-wide restriction on long-chain PFCAs is necessary to prevent the possible substitution of PFOA with long-chain PFCAs (ECHA 2018b).

5. Information on long-chain PFCAs and how they fulfil the Annex D screening criteria

5.1. Persistence

22. Long-chain PFCAs are carboxylic acids bonded to a fully fluorinated carbon chain, with total carbon numbers from 9 to 21. This carbon-fluorine bond is one of the strongest existing covalent bonds (about 108 – 120 kcal/mole) (Dixon 2001; Parsons et al. 2008), making the bond extremely stable and generally resistant to degradation by acids, bases, oxidants, reductants, photolytic processes, microbes and metabolic processes. Fluorine also has the highest electronegativity of all elements in the periodic table. The presence of fluorine instead of hydrogen on the carbon chain alters the thermal, chemical and biological characteristics of the molecule. The strong carbon-fluorine bond and high density of electron-rich repellent fluorine atoms protects the carbon backbone and results in inertness to both heat and chemical reagents (Hakli et al. 2008; Colomban et al. 2014; Parsons et al. 2008). Moreover, this contributes to a high ionization potential, low polarizability, low inter- and intra-molecular interactions and low surface tension. Therefore, long-chain PFCAs are considered stable in the environment. For example, C9 PFCA did not biodegrade under the OECD 301F method (Stasinakis et al. 2008). Other studies demonstrate that long-chain PFCAs do not degrade under environmentally relevant conditions (e.g., Hori et al. 2005a; Hori et al. 2005b; Hori et al. 2008; Qu et al. 2016; Liu et al. 2017). 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).

23. Long-chain PFCAs have met the regulatory cirteria for persistence in different jurisdictions. In the European Union, C9 – C10 PFCAs have been identified as persistent and C11 – C14 PFCAs have been identified as very persistent in accordance with the criteria set out in the REACH regulation (ECHA, 2012a, b, c, d, 2015, 2016). In Canada, the ecological screening assessment for long-chain PFCAs, their salts and their precursors (Environment Canada 2012) concluded that long-chain PFCAs and their salts meet the criteria for persistence as set out in the *Persistence and Bioaccumulation Regulations* (Canada 2000).

Conclusion on persistence according to the criteria in Annex D

24. Based on the available empirical data and physicochemical properties, it is concluded that the long-chain PFCAs meet the Annex D criteria for persistence.

5.2. Bioaccumulation

25. The unique characteristics and physicochemical properties of long-chain PFCAs are relevant to the potential for bioaccumulation. Long-chain PFCAs are non-volatile substances with combined properties of ionization, lipophobicity, hydrophobicity, and hydrophilicity over different portions of

[7] C9 PFCA and its salts are classified as Carcinogenicity 2, Reproductive toxicity 1B and Effects on or via Lactation, Acute Toxicity 4, Specific Target Organ Toxicity Repeated Exposure (STOT RE) 1 and Eye Damage 1. C10 PFCA and its salts are classified as Carcinogenicity 2, Reproductive toxicity 1B and Effects on or via Lactation.

the molecule. The carboxylate functional group attached to the perfluorinated chain also imparts polarity to the molecule. Due to these properties, the hydrophobic and lipophilic interactions between long-chain PFCAs and the substrate are not the main mechanism that govern their bioaccumulation, which is unlike most organic chemicals (Hekster et al. 2002). Hydrophobicity is unlikely to be the main driving force for partitioning to tissues, as the lipophobic tendencies oppose this partitioning process; instead, electrostatic interactions may be more important (Hekster et al. 2002). Additionally, the presence and metabolic transformation of related compounds to long-chain PFCAs in wildlife can add to the body burden of long-chain PFCAs, themselves, in wildlife (Nabb et al. 2007; Letcher et al. 2014).

26. 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.

27. Both bioconcentraton 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*). 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-day depuration period (Martin et al. 2003a). For market-size rainbow trout, the BAF for C9 PFCA was < 0.4 after a 28-day 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). 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).

28. 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 (C12 PFCA) and plasma BAFs from 631 (C9 PFCA) to 5 011 872 (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. Whole body BCFs for European perch (*Perca fluviatilis*) from Lake Halmsjön (Sweden) ranged from 42 to 54 (C9 PFCA) and 140 to 220 (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 (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 922 (C9 PFCA), 45 to 5318 (C10 PFCA), 26 to 18 902 (C11 PFCA), and 91 to 6635 (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) and C10 PFCA (331 to 5623) (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), C10 PFCA (1230 to 69 183) and C11 PFCA (589 to 7762). 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). 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. BAFs were determined for plankton species in Taihu Lake (China) that ranged from 462 (C10 PFCA) to 17788 (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 (C11 PFCA) and, for sprat, BAFs ranged from > 59 (C15 PFCA) to 158 489 (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 (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).

29. Extrapolating BCF/BAF data from fish and aquatic invertebrates to birds and terrestrial/marine mammals can underestimate the bioaccumulation potential for long-chain PFCAs. Generally, bioaccumulation occurs by the same mechanism for neutral organic chemicals that are non-polar and non-volatile (e.g., PCBs) in water-breathing organisms (e.g., fish and aquatic invertebrates) and air-breathing organisms (e.g., terrestrial/marine mammals or birds). As neutral chemicals have low elimination rates to both water and air, this results in similar bioaccumulation potential for both air-breathing and water-breathing organisms (Kelly et al. 2004; Mackay and Fraser 2000). However, long-chain PFCAs that are ionizing, polar, and non-volatile have higher water solubility compared to neutral chemicals. For water-breathing organisms, this can result in a more rapid elimination of long-chain PFCAs to the water phase and a subsequent reduction in bioaccumulation potential. The moderate water solubility of long-chain PFCAs causes a relatively high tendency to escape from the gills into water, whereas the escaping tendency of long-chain PFCAs to the air, across the alveolar membrane of the lung, would be relatively low because of their low vapor pressure and negative charge. As bioaccumulation in air-breathing organisms is driven primarily by volatility rather than polarity, the non-volatile nature of long-chain PFCAs promotes a relatively slow elimination to air, resulting in higher bioaccumulation potential in air-breathing organisms (Kelly et al. 2004). That is, fish gills provide an additional mode of elimination for long-chain PFCAs that birds and terrestrial/marine mammals do not possess (Martin et al. 2003a). Additionally, extrapolating BCF/BAF data from fish to marine/terrestrial mammals should not be performed due to the biological differences between higher and lower trophic levels (e.g., feeding rates, assimilation efficiency, and depuration rates) (Martin et al. 2003a). As such, field biomagnification factors (BMF, unitless) and trophic magnification factors (TMF, unitless) are more relevant in determining the overall bioaccumulation potential for long-chain PFCAs.

30. 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 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 (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 (Simmonet-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 (Simmonet-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. 2009b). 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. 2004; Kelly et al. 2009; Loi et al. 2011; 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.

31. There are no biomagnification or trophic magnification data for long-chain PFCAs with chain lengths greater than C16. However, it is expected that C17 – C21 PFCAs can also biomagnify in birds and terrestrial/marine mammals. C17 – C22 PFCAs have been measured in snow and soil (Plassman and Berger 2013) and C15 – C18 PFCAs have been measured in top predator species such as the peregrine falcon (*Falco peregrinus*) and herring gulls (*Larus argentatus*) from the Great Lakes region (Canada) (Sun et al. 2020) as well as East Greenland polar bears (Greaves et al. 2013; Boisvert et al. 2019). Wang and Ober (1999) have suggested that the carbon-carbon conformation changes as the chain length increases, with longer chains becoming more helical that results in smaller cross-sectional diameter molecules with a greater ability to accumulate in organisms. The rigid molecular structure of long-chain PFCAs with linear perfluoroalkyl chains (D_{eff} = 0.61 – 0.96 nm in C8 to C18 PFCAs) can enable them to pass through biological membranes more easily (Anliker et al. 1988), thus leading to greater bioaccumulation (Dimitrov et al. 2003). These physicochemical properties would result in an underestimation of the steric bulk effect of the molecule size (e.g., D_{max}) based on molecular weight and the under-estimation of the bioaccumulation potential of longer-chain PFCAs, even with their relatively large molecular weights (Inoue et al. 2012).

32. In humans, there is evidence that long-chain PFCAs can i) be distributed to many tissues (e.g., brain, liver, kidney) and fluids, ii) pass through the placenta into the fetus, and iii) be transferred through breast milk. Long-chain PFCAs (C9 – C14) have been detected globally in human blood, urine, umbilical cord, breast milk, hair, and nail samples including those collected from remote locations such as Greenland and Northern Canada (e.g., Guruge et al. 2005; Tao et al. 2008; Olsen et al. 2011; Fujii et al. 2012; Motas Guzman et al. 2016; Wu et al. 2017; Cao et al. 2018; Lee et al. 2018; ECHA 2018a; Wang et al. 2018; Caron-Beaudoin et al. 2020; Li et al. 2020b; Li et al. 2020a; Liu et al. 2020). In cycle 2 (2009–2011) and cycle 5 (2016–2017) of the Canadian Health Measures Survey (CHMS), geometric mean plasma concentrations of C9 to C11 PFCAs ranged from 0.12 to 0.82 µg/L

in participants aged 12-79 (Health Canada 2019). Of note, concentrations of long-chain PFCAs in the serum of pregnant Inuit women (C9 – C11) and First Nation Anishinabe youth (C9) were higher than these CHMS values (Caron-Beaudoin et al. 2019; Caron-Beaudoin et al. 2020).The geometric mean serum concentration of C9 PFCA as measured in the 2013-2014 National Health and Nutrition Examination Survey (NHANES) was 0.91 µg/L in participants aged 12 and older (Graber et al. 2019). The results from 29 biomonitoring studies in the EU showed C9 – C14 PFCA serum concentrations to be in the in the high pg/ml to low ng/ml (most samples) range (ECHA 2018a). PFCAs are believed to be metabolically inert (Kudo 2015). Vanden Heuvel et al. (1991) exposed male and female rats to a single intraperitoneal dose of 5 mg/kg of C10 PFCA. After 28 d, there was no evidence that C10 PFCA defluorinated or conjugated to form persistent hybrid lipids nor were polar metabolites of C10 PFCA detected in the urine or bile of male or female rats. Due in part to their strong protein binding affinity, long-chain PFCAs are eliminated very slowly from the human body. The mean half-lives for C9 PFCA are estimated to range from 2.5 to 4.3 years in humans whereas the mean half-lives for both C10 and C11 PFCA range from 4.5 to 12 years (Zhang et al. 2013). Chen et al. (2020) found that PFCAs with longer carbon chains bind more strongly to proteins. Moreover, several studies in animals suggest that the longer the carbon chain length, the more slowly the PFAS is eliminated, and thus, the more bioaccumulative it is (Ohmori et al. 2003; Kudo et al. 2006; Yeung et al. 2009). On this basis, it is reasonable to expect that the PFCAs with chain lengths > C12 may have even longer half-lives than C9 to C11 PFCAs.

33. Long-chain PFCAs have met regulatory criteria for bioaccumulation in some jurisdictions. Long-chain PFCAs (C9 – C14) have been assessed in the European Union and identified as bioaccumulative (C9 and C10 PFCAs) or very bioaccumulative (C11 to C14 PFCAs) in accordance with the criteria set out in the REACH regulation (ECHA 2012a,b,c,d, 2015, 2016). In Canada, the ecological risk assessment for long-chain PFCAs, their salts and their precursors (Environment Canada 2012) used a weight of evidence approach based on BMF/TMF data to conclude that long-chain PFCAs and their salts accumulate and biomagnify in birds, and terrestrial/marine mammals. Canada's *Persistence and Bioaccumulaton Regulations* (Canada 2000) are based on bioaccumulation data (i.e., BCF/BAF, log K_{ow}) for freshwater aquatic species (fish) and are best suited for addressing substances that preferentially partition to lipids. However, as long-chain PFCAs preferentially partition to proteins in the liver, blood, and kidneys of birds and terrestrial/marine mammals, Canada's *Persistence and Bioaccumulation Regulations* do not fully reflect the bioaccumulation potential for long-chain PFCAs.

Conclusion on bioaccumulation according to the criteria in Annex D

34. BCFs and BAFs > 5000 have been reported for C9 – C14 long-chain PFCAs in freshwater and marine aquatic organisms. C9 – C16 PFCAs have been reported to biomagnify in the food chain for birds and terrestrial/marine mammals with BMFs or TMFs > 1. Furthermore, C18 PFCA has been measured in the environment and in some top predator species such as polar bears, herring gulls and peregrine falcons. In humans, C9 – C14 PFCAs have been detected in various tissues and fluids. The elimination of C9 – C11 PFCAs is very slow resulting in long estimated half-lives. Based on reported bioaccumulation values and environmental monitoring data, long-chain PFCAs meet the Annex D criterion for bioaccumulation.

5.3. Potential for long-range environmental transport

35. Long-chain PFCAs, their salts and related compounds are measured in remote areas, such as the Antarctic and the Canadian Arctic that are far from known manufacturing sites in both biotic and abiotic samples. Long-range transport pathways include atmospheric and oceanic transport of long-chain PFCAs and/or related compounds. Examples of related compounds include fluorotelomer alcohols (e.g., 8:2 FTOH, 10:2 FTOH, 12:2 FTOH) and their fluorotelomer acid derivatives (e.g., 10:2 FTA; 10:2 FTUCA).

36. Global modelling indicates that long-chain PFCAs, their salts and/or related compounds have the potential to be transported over long distances (Wallington et al. 2006; Wania 2007; Yarwood et al. 2007; Thackray et al. 2020). The presence of long-chain PFCAs in remote areas can be described as a precursor (e.g., FTOH) emitted to the atmosphere ultimately yielding long-chain PFCAs through biotic or abiotic degradation. Wallington et al. (2006) used a three-dimensional global atmospheric chemistry model (IMPACT) to show that 8:2 FTOH degrades in the atmosphere to form C9 PFCA. Young et al. (2007) suggested that the presence of C9 – C11 PFCAs on several Canadian High Arctic ice caps (Melville Ice Cap (Northwest Territories), Agassiz Ice Cap (Nunavut), and Devon Ice Cap (Nunavut)) is indicative of atmospheric oxidation of volatile precursors. Young et al. (2007) detected C9 PFCA (0.005 – 0.246 ng/L), C10 PFCA (ND – 0.022 ng/L), and C11 PFCA (ND – 0.027 ng/L). Ellis et al. (2004) showed that the atmospheric lifetime of FTOHs, as determined by their reaction with hydroxy radicals, was approximately 20 d which would allow precursors to be slowly oxidized

by atmospheric radical species to give fluorinated acids that would then be deposited in remote areas by precipitation (Waterland and Dobbs 2007). Atmospheric measurements confirm modelling results in that volatile precursors can reach Arctic and Antarctic latitudes where they may be biodegraded or transformed to long-chain PFCAs (Shoeib et al. 2006; Jahnke et al. 2007; Stock et al. 2007; Young et al. 2007; Cai et al. 2012a; Kwok et al. 2013; Wang et al. 2015b; Casal et al. 2017; MacInnis et al. 2019; Pickard et al. 2018; Wong et al. 2018; Joerss et al. 2020). For example, Shoeib et al. (2006) collected air samples during an ice-breaker crossing of the North Atlantic and Canadian Archipelago to investigate concentrations of FTOHs. The highest concentrations were for 8:2 FTOH (5.8 – 26 pg/m^3) followed by 10:2 FTOH (1.9 – 17 pg/m^3). Cai et al. (2012a) measured 10:2 FTOH, 12:2 FTOH, and 10:2 FTA in ambient air from the Japan Sea to the Arctic Ocean with concentrations of 1.8 – 47 pg/m^3 in the gas-phase and 0.1 – 2.5 pg/m^3 in the particle-phase. In the Antarctic Peninsula, 8:2 FTOH, 10:2 FTOH, and 12:2 FTOH and their derivatives (e.g., 8:2 FTA) were measured in the snow and air (Wang et al. 2015b). On Livingston Island (Antarctica), C9 – C14 PFCAs were measured in snow (ND – 0.04 ng/L) (Casal et al. 2017). In the Canadian Arctic, C9 – C14 PFCAs were measured in Lake Hazen (Nunavut, Canada) snowpack with concentrations ranging from < 0.002 to 3.1 ng/L (MacInnis et al. 2019). Stock et al. (2007) measured C9 – C12 PFCAs (0.2 – 19 ng/L) and their FTOH acid derivatives (i.e., 8:2 FTUCA and 10:2 FTUCA) in Resolute Lake, Char Lake, and Amituk Lake on Cornwallis Island (Nunavut, Canada). By modelling air mass transport densities and comparing temporal trends in deposition with production changes of possible sources, Pickard et al. (2018) determined that the deposition of long-chain PFCAs on the Devon Ice Cap was dominated by atmospheric formation from volatile precursors. Pickard et al. (2018) sampled a 15-m ice core representing 38 years of deposition (1977 – 2015) from the Devon Ice Cap and detected C9 – C13 PFCAs with concentrations that ranged from 0.00321 – 0.751 ng/L.

37. Oceanic transport of long-chain PFCAs, their salts and related compounds has also been proposed as a pathway to account for the presence of long-chain PFCAs in remote areas. As perfluorinated alkyl acids, their salts and conjugate bases are highly water-soluble with no appreciable vapor pressure, their presence in the atmosphere is described as their transfer from the surface ocean by sea spray aerosols (Webster and Ellis 2010; Reth et al. 2011; Johansson et al. 2019). Reth et al. (2011) determined that their surface-active properties result in enrichment on the "*surface of bursting bubbles*". Reth et al. (2011) examined the water-to-air transfer of C6 – C14 PFCAs in a laboratory-scale sea spray simulator and found that the sequestration of the perfluorinated alkyl acids, their salts and conjugate bases out of bulk water to the air-water surface increased exponentially with the length of the perfluorinated alkyl chain. Measurements of long-chain PFCAs in oceans suggest that oceanic transport does play a role in the transport of long-chain PFCAs to remote regions (Ahrens et al. 2010; Benskin et al. 2012; Cai et al. 2012a; Cai et al. 2012b; Zhao et al. 2012; Gonzalez-Gaya et al. 2014; Casal et al. 2017; Yeung et al. 2017; Li et al. 2018; Gonzalez-Gaya et al. 2019; Zhang et al. 2019). For example, Gonzalez-Gaya et al. (2019) measured C9 PFCA (ND – 1.15 ng/L) and C10 PFCA (ND – 2.19 ng/L) at depths of 20 -160 m in the Atlantic, Indian, and Pacific Oceans. Surface water samples collected from the Greenland Sea and the East Atlantic Ocean had C9 PFCA concentrations that ranged from < 0.012 – 0.039 ng/L, C10 PFCA concentrations < 0.021 ng/L, C11 PFCA concentrations that ranged from ND – < 0.013 ng/L, and C12 PFCA concentrations < 0.025 ng/L (Zhao et al. 2012). C16 PFCA concentrations ranged from < 0.007.5 – 0.0082 ng/L in coastal surface seawater samples taken along near the South Shetland Islands (Maritime Antarctica) (Cai et al. 2012b). On Livingston Island (Antarctica), C9 – C14 PFCAs measured in seawater ranged from ND – 0.11 ng/L (Casal et al. 2017).

38. Long-chain PFCAs (primarily C9 to C18 PFCAs), have been measured in Antarctic and Arctic biota such as, penguin (e.g., *Pygoscelis papua*), polar bear (*Ursus maritimus*), Arctic fox (*Vulpes lagopus*), reindeer (*Rangifer tarandus*), Alaskan sea otter (*Enhydra lutris kenyoni*) and muskox (*Ovibos moschatus*) (Bossi et al. 2005; Smithwick et al. 2005a; Smithwick et al. 2005b; Smithwick et al. 2006; Tao et al. 2006; Butt et al. 2007a; Butt et al. 2007b; Butt et al. 2008; Dietz et al. 2008; Hart et al. 2009; Katz et al. 2009; Schiavone et al. 2009; Bengtson Nash et al. 2010; Müller et al. 2011; Greaves et al. 2012; Llorca et al. 2012; Rotander et al. 2012; Aas et al. 2014; Bossi et al. 2015; Lescord et al. 2015; Routti et al. 2015; Munoz et al. 2017a; Routti et al. 2016; Routti et al. 2017; Tartu et al. 2017; Boisvert et al. 2019; Costantini et al. 2019; Roscales et al. 2019). For example, in East Greenland polar bears, C15 PFCA have been measured in polar bear liver (0.73 – 0.89 ng/g ww), blood (1.22 – 1.48 ng/g ww), brain (9.9 – 10.9 ng/g ww), muscle (0.58 – 0.72 ng/g ww), and adipose tissue (0.5 – 0.64 ng/g ww) (Greaves et al. 2012). C16 and C18 PFCAs have also been measured in East Greenland polar bear liver (0.1 – 0.2 ng/g ww for C16 PFCA and 0.2 – 0.4 ng/g ww for C18 PFCA) and ringed seal (*Phoca hispida*) liver (ND – 0.2 ww for C16 PFCA and 0.1 – 0.5 ng/g ww for C18 PFCA) (Boisvert et al. 2019). In the northern Yukon (Canada), C9 – C13 PFCAs were measured in caribou (*Rangifer tarandus groenlandicus*) liver (< 0.5 – 3.20 ng/g ww) and wolf (*Canis lupus*) liver (0.19 – 7.79 ng/g ww) (Katz et al. 2009; Müller et al. 2011). In East and South Greenland,

C9 – C13 PFCAs were measured in reindeer liver (ND – 2.06 ng/g ww) and muskox liver (0.21 – 5.25 ng/g ww) (Bossi et al. 2015). In Antarctica, C9 – C12 PFCAs were measured in the Weddell seal (*Leptonychotes weddellii*) liver (< 0.01 – 0.23 ng/g ww) (Routti et al. 2015). In Antarctica, C9 – C12 PFCAs were measured in the eggs (< 0.1 – 2.5 ng/g ww), blood (< 0.5 ng/ml), and muscle (< 1.4 ng/g ww) of the Adelie penguin (*Pygoscelis adeliae*) and in the eggs (0.1 – 0.5 ng/g ww) and muscle (ND – 0.34 ng/g ww) of the Gentoo penguin (*Pygoscelis papua*) (Schiavone et al. 2009; Tao et al. 2006; Bengtson Nash et al. 2010; Llorca et al. 2012).

Conclusion on long-range transport according to the criteria in Annex D

39. Long-chain PFCAs, their salts and related compounds were detected in remote areas, including the Antarctic and the Canadian Arctic, where they are found in environmental matrices and in biota. Available research indicates that the presence of these substances in remote areas result from the atmospheric and oceanic transport of volatile precursors and/or the acids, themselves. Based on the available data, it is concluded that long-chain PFCAs, their salts and related compounds meet the criteria for long-range transport in Annex D to the Stockholm Convention.

5.4. Adverse effects

40. PFAS, in general, have been shown to activate the peroxisome proliferating receptors (PPARs) in multiple species (Ishibashi et al. 2008b; Hickey et al. 2009; Ishibashi et al. 2011; Kurtz et al. 2019; Routti et al. 2019a). PPAR-α plays a critical physiological role as a lipid sensor and a regulator of lipid metabolism. Within the cytochrome P450 enzymes, the CYP4A family members are integral to several metabolic functions, including detoxifying xenobiotic compounds. PPARs regulate CYP4A expression, which in turn acts as a modulator with other PPARα target genes involved in lipid homeostasis. Activation of the PPARα−CYP4A pathway could result in altered liver function, developmental toxicity, immunotoxicity, and feeding disorders (Kurtz et al. 2019; Kubota et al. 2011). C9 and C10 PFCAs have been shown to induce hepatic CYP4A-like proteins via PPAR-α signaling in Lake Baikal seals (Ishibashi et al. 2008b). PPAR-α mRNA expression and CYP4a protein expression in kidneys of cetaceans have also been positively correlated with C10, C11, C13 and C14 PFCAs (Kurtz et al. 2019).

41. In laboratory toxicity studies assessing traditional endpoints (i.e., growth, reproduction, and lethality), long-chain PFCAs (up to C14 PFCA) show low to moderate toxicity depending on species sensitivity. 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).

42. In laboratory toxicity studies assessing non-traditional endpoints, effects associated with exposure to long-chain PFCAs (up to C13 PFCA) included developmental effects, behavioural effects, hepatotoxicity, immunotoxicity, neurotoxicity, genotoxicity, changes in gene expression, chemosensitivity or altered thyroid function. Species tested include common cormorant (*Phalacrocorax carbo*), zebrafish, rainbow trout, African clawed frog, rare minnow (*Gobiocypris rarus*), mussels (*Pema viridis*; *Mytilus californianus*) and chickens (*Gallus gallus*) (Matsubara et al. 2006; Stevenson et al. 2006; Liu et al. 2008a; Liu et al. 2008b; O'Brien et al. 2009; Wei et al. 2009; Yeung et al. 2009; Nobels et al. 2010; Tichy et al. 2010; Benninghoff et al. 2011; Liu et al. 2011b; Vongphachan et al. 2011; Benninghoff et al. 2012; O'Brien et al. 2013; Zhang et al. 2012a; Zhang et al. 2012b; Zheng et al. 2012; Kim et al. 2013; Ulhaq et al. 2013a; Ulhaq et al. 2013b; Jo et al. 2014;

Liu et al. 2014a; Liu et al. 2014b; Yang et al. 2014; Liu et al. 2015; Lu et al. 2015; Gorrochategui et al. 2016; Jantzen et al. 2016a; Jantzen et al. 2016b; Zhang et al. 2016; Guo et al. 2018; Zhang et al. 2018a; Menger et al. 2020). For example, 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; Jantzen et al. 2016b). 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).

43. For the field-based wildlife studies, it is difficult to uniquely distinguish effects from exposure to long-chain PFCAs, as exposures from mixtures of other PFAS (e.g., PFOS or PFOA) and other contaminants cannot be excluded (Knudsen et al. 2007; Letcher et al. 2010; Bourgeon et al. 2017; Liu et al. 2018b; Routti et al. 2019b). PFAS (including related compounds) are also often summed as a group and statistically correlated with the effect observed. 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 38 organochlorine compounds, including polychlorinated biphenyls (PCBs), polybrominated diphenyl ethers (PBDEs), and 10 phenolic compounds as well as 8 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 5 classes of PCB metabolites, 4 classes of perfluorinated sulfonates, and 4 classes of other polychlorinated compounds (i.e., chlorinated aromatics, tetrachloro aromatic sulfate, heptachlorinated hydroxylated nitroaromatics and hexachlorinated 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). Thus, mixtures can be confounding when determining whether a singular substance or group of substances is affecting the health and condition of the wildlife species under investigation. As such, a direct cause-effect correlation is difficult, as statistical correlations, by themselves, do not imply causal relationships. Recognizing this uncertainty, several field-based wildlife studies have shown statistical correlations with observed effects for long-chain PFCA mixtures (from C9 to C16) in various wildlife species, including top predators (Houde et al. 2006c; Erikstad et al. 2009; Peng et al. 2010; Miljeteig et al. 2012; Houde et al. 2013; Aas et al. 2014; Ask 2015; Bustnes et al. 2015; Elliott et al. 2019; Persson and Magnusson 2015; Eggers Pedersen et al. 2016; Blévin et al. 2017; Soloff et al. 2017; Tartu et al. 2017; Bangma et al. 2018; Grønnestad et al. 2018; Lopez-Antia et al. 2019; Briels et al. 2019; Costantini et al. 2019; Groffen et al. 2019; Kurtz et al. 2019; Lasters et al. 2019; Blévin et al. 2020; Guillette et al. 2020; Sun et al. 2020). 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.

44. Current environmental monitoring data indicate that concentrations for long-chain PFCAs are generally at the nanogram level (ng/g or ng/L) in biota. These concentrations are below the available tested toxicity thresholds, which are generally at the microgram (µg/g) or milligram level (mg/L), for both traditional and non-traditional toxicity endpoints and with varying sensitivity across species. There are unique concerns about highly persistent and bioaccumulative substances such as long-chain PFCAs and their salts. Current science cannot accurately predict the full extent of ecological effects for long-chain PFCAs as they are acknowledged to have the potential to cause serious and irreversible impacts to wildlife populations in the long-term (MacLeod et al. 2014; Ahrens and Bundschuh 2014). Long-chain PFCAs are persistent and remain in the environment for a very long time which increases their probability, magnitude and duration of exposure to wildlife. Long-chain PFCAs are also subject to long-range transport which can also result in regional or global contamination. As such, releases of long-chain PFCAs can lead to elevated concentrations in organisms over wide areas. Long-chain PFCAs may also biomagnify through the food chain, resulting in increased internal concentrations for top predator species. Since they are widespread, several different long-chain PFCAs can be present simultaneously in the tissues of organisms, increasing the likelihood and potential severity of harm. Increasing temporal concentration trends (i.e., doubling times) in wildlife, including top predator species, suggest that long-chain PFCAs can approach toxicity thresholds resulting in harm for wildlife populations in the future.

45. Temporal trends for long-chain PFCAs (up to C15 PFCA) have been reported in wildlife (including top predator species found in remote regions such as polar bears) (De Silva and Mabury 2004; Bossi et al. 2005; Smithwick et al. 2005a; Smithwick et al. 2006; Butt et al. 2007a; Butt et al. 2007b; Verreault et al. 2007; Butt et al. 2008; Dietz et al. 2008; Tomy et al. 2009a; Holmström et al. 2010; O'Connell et al. 2010; Miller et al. 2015; Gewurtz et al. 2016; Lam et al. 2016; Dassuncao et al. 2017; Smythe et al. 2018; Falk et al. 2019; Gui et al. 2019; Wu et al. 2020). From 1972 to 2002, mean doubling times for concentrations in polar bear livers from North American Arctic regions ranged from 5.8 to 9.1 years for C9 – C11 PFCAs (Smithwick et al. 2006). From 1984 to 2006, 128 sub-adult (3–5 years old) Greenland polar bears showed annual increases for C9 PFCA (6.1%), C10 PFCA (4.3%), C11 PFCA (5.9%), C12 PFCA (52%), and C13 PFCA (8.5%) (Dietz et al. 2008). From 1974 to 2007, C9 – C15 PFCA doubling times ranged from 5.6 to 9.0 years in peregrine falcon (*Falco peregrinus*) eggs collected from Sweden (Holmström et al. 2010). Temporal trends for the harbor porpoise (*Phocoena phococena*) populations from the Baltic Sea and North Sea showed that C9 – C13 PFCA concentrations increased significantly from 1991 to 2008 (Huber et al. 2012). Liver and serum mean concentrations of C9 and C10 PFCAs in the Baikal seal (*Pusa sibirica*) (Lake Baikal, Russia) collected in 2005 were 1.2 and 1.7-fold greater than liver and serum concentrations from 1992 (Ishibashi et al. 2008a). For the years 1980 to 2010, the ∑PFCAs (including C8 – C12 PFCAs) in livers of male beluga whales (Nunavut, Canada) showed an annual increase of 1.8 +/- 0.5 ng/g ww (Tomy et al. 2009a). For the years 1986 to 2013, C9 – C13 PFCA concentrations in the muscle tissue of North Atlantic male pilot whales (*Globicephala melas*) (caught in Faroe Islands) increased 2.8% to 8.3% per year (Dassuncao et al. 2017).

46. For human health, both animal data and epidemiological data are available for assessing the adverse effects of long-chain PFCAs. In terms of animal data, information is available for C9 – C14, C16 and C18 PFCAs, with C9, C10 and C12 PFCAs being the most studied. Hepatotoxicity, developmental/reproductive toxicity, immunotoxicity, and thyroid toxicity have been commonly reported in animal models. Other effects reported to a lesser extent include renal, cardiovascular and neurological effects, metabolic disruption, body and organ weight changes, and mortality.

47. *In vivo* data in rodents provide evidence of hepatotoxicity after acute, short-term, subchronic and/or chronic exposure to C9 – C12, C14, C16 and C18 PFCAs. Effects include liver weight alterations, hepatocellular hypertrophy, histopathological changes, alterations in liver gene expression, and clinical chemistry changes (Cheng and Klaassen 2008a; Cheng and Klaassen 2008b; Stump et al. 2008; Maher et al. 2008; Zhang et al. 2008; Ding et al. 2009; Mertens et al. 2010; Wolf et al. 2010; Fang et al. 2012a; Fang et al. 2012b; Fang et al. 2012c; Hirata-Koizumi et al. 2012; Takahashi et al. 2014; Fang et al. 2015; Hirata-Koizumi et al. 2015; Kato et al. 2015; Wang et al. 2015a; Liu et al. 2016; Das et al. 2017; Frawley et al. 2018; Zhang et al. 2018b; NTP 2019).

48. Several long-chain PFCAs (C9, C11, C12, C14, C16 and C18) have been shown to induce reproductive and developmental toxicity in rodents after short-term, subchronic and/or chronic oral exposure (gavage). Effects observed included reproductive organ weight alteration, testicular toxicity, and altered reproductive hormone levels. Developmental effects included postnatal mortality, reduced body weight, and developmental delays (Shi et al. 2007; Feng et al. 2009; Lau et al. 2009; Shi et al. 2009; Wolf et al. 2010; Hirata-Koizumi et al. 2012; Rogers et al. 2014; Takahashi et al. 2014; Das et al. 2015; Hirata-Koizumi et al. 2015; Kato et al. 2015; Hadrup et al. 2016; Singh and Singh 2018; Chen et al. 2019; NTP 2019; Singh and Singh 2019a; Singh and Singh 2019b; Singh and Singh 2019c).

49. Effects on the immune system induced by exposure to C9 – C11 PFCAs are reported in rodents after acute, short-term and/or chronic oral exposure (gavage or drinking water), or after intraperitoneal administration. The effects observed include splenic and thymic atrophy, reduced phagocytic function of macrophages, altered balance of immune cells, and inhibition of cytokine production (Fang et al. 2008; Fang et al. 2009; Fang et al. 2010; Rockwell et al. 2013; Bodin et al. 2016; Rockwell et al. 2017; Frawley et al. 2018; NTP 2019).

50. Short-term studies performed in rats and mice provide evidence that oral (gavage) exposure to C9, C10 and C14 PFCAs induce altered thyroid weight and histopathological alterations in the thyroid gland (Fang et al. 2009; Hirata-Koizumi et al. 2015; NTP 2019).

51. An extensive number of epidemiological studies have investigated the health effects associated with human exposure to long-chain PFCAs. These include cross-sectional, cohort and case-control studies, as well as randomized trials which included a double-blind community study. Although some studies have reported null, equivocal or even negative associations with PFAS exposure (i.e., protective effects), many studies have established positive associations between exposure to C9 – C14 PFCAs and various health related outcomes. These outcomes involve organ systems that are similar to those examined in experimental studies with animals. Two reviews examining the human health effects of exposure to PFAS found: a) a link between exposure to C9 – C10 PFCAs and increased serum lipid levels (ATSDR 2018), b) suggestive evidence of a link between C10 PFCA and immune effects (decreased antibody responses to vaccines) (ATSDR 2018) and c) limited evidence of a negative association between C10 PFCA and diphtheria antibody levels after vaccination of children or adults (Kirk et al. 2018).

52. 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 (e.g., Zeng et al. 2015; Salihovic et al. 2018; Bassler et al. 2019). Other studies observed relationships between exposure to C9, C11 – C14 PFCAs and reduced kidney function (e.g., decreased estimated glomerular filtration rate, chronic kidney disease, biomarkers of renal failure) (e.g., Stanifer et al. 2018; Wang et al. 2019; Scinicariello et al. 2020). Effects on the thyroid were investigated in relation to C9 – C14 PFCAs and associations were noted with outcomes including altered levels of hormones, thyroglobulin, and thyroid peroxidase antibodies (e.g., Ballesteros et al. 2017; Aimuzi et al. 2019; Itoh et al. 2019). Several studies evaluated possible associations between exposure to C9 – C14PFCAs and reproductive outcomes in adolescents/adults. Associations were noted with altered hormone levels, issues related to menstruation, menopause and female reproductive health (e.g., Jensen et al. 2015; Tsai et al. 2015; Lum et al. 2017; Ding et al. 2020). In terms of developmental endpoints, associations were also observed with birth size, bone development, reproductive outcomes, neurobehavioural and neuropsychological endpoints (e.g., Lien et al. 2016; Lind et al. 2016; Buck Louis et al. 2018; Wikstrom et al. 2019). With regards to the immune system, epidemiological studies noted positive associations between exposure to C9 – C12, C14 PFCAs and the incidence of infectious diseases, alterations of immune marker levels, asthma and allergic diseases and decreased efficacy of vaccinations (e.g., Dong et al. 2013; Zhu et al. 2016; Grandjean et al. 2017; Chen et al. 2018; Impinen et al. 2018). In terms of cardiovascular outcomes, associations were observed between exposure to C9 – C13PFCAs and cardiovascular disease and hypertension (e.g., Bao et al. 2017; Huang et al. 2018; Mobacke et al. 2018). In addition, links between C9, C10, C12 PFCAs and obesity and metabolic disorders (diabetes and related outcomes) have been noted (e.g., Liu et al. 2018b; Rahman et al. 2019; Duan et al. 2020). A small number of epidemiological studies evaluated possible associations between exposure to C9 – C13 PFCAs and cancer. One study reported a positive association while other studies reported weak or null-associations. In another study, Temkin et al. (2020) applied the Key Characteristics of Carcinogens framework and a weight of evidence approach (consideration of epidemiological data, *in vivo* data in animals and *in vitro* data), and found C9 – C14 PFCAs to exhibit multiple key characteristics of carcinogens.

53. The potential mechanisms of toxicity of C9 – C14 PFCAs have been examined and include, but are not limited to, the alteration of cell proliferation, modulation of receptor-mediated effects

(PPARs), inflammation, oxidative stress, cellular toxicity, and binding to human serum albumin (Wolf et al. 2012; Yang et al. 2010; Oshida et al. 2015; Gorrochategui et al. 2016; Ren et al. 2016; Rosen et al. 2017; Gao et al. 2020a). In addition, structure-activity relationships for long-chain PFCAs have been investigated and data from *in vivo* studies in aquatic species and mammals indicate that the activity/toxicity of PFCAs tend to increase with chain length up to C11/C12 (Kudo et al. 2006; ; Liu et al. 2014a; Das et al. 2015; NTP 2019). *In vitro* data in mammalian cells indicate a similar trend of increasing toxicity with increasing chain length up to C18 (e.g., Buhrke et al. 2013; Gorrochategui et al. 2014; Rand et al. 2014; Yang et al. 2017; Lee and Kim 2018; Ojo et al. 2020).

54. In the European Union, C9 and C10 PFCAs and their salts are classified according to the Globally Harmonized System (GHS) criteria (Regulation (EC) No. 1272/2008) for their carcinogenic potential (Carc. 2: Suspected of causing cancer), reproductive toxicity (Repr. 1B: Adverse effects on sexual function and fertility or on development) and effects on or via lactation. C9 PFCA is also classified for its acute toxicity (Category 4), toxicity on the liver, thymus, and spleen (STOT RE 1: Specific target organ toxicity – repeat exposure) and eye damage (Category 1). In accordance with Article 57 (c) of the REACH Regulation, C9 and C10 PFCAs are classified as toxic for reproduction (ECHA 2015, 2016, 2018b).

Conclusion on adverse effects according to the criteria in Annex D

55. Available experimental and epidemiological evidence indicate that long-chain PFCAs can cause adverse effects to the environment and wildlife and can potentially cause adverse effects in humans. Experimental, field and/or epidemiological data are lacking to evaluate the toxicity for some of the long-chain PFCAs (i.e., no ecotoxicological data for C15 – C21 PFCAs and no human health toxicological data for C15, C17, C19, C20 and C21 PFCAs). However, considering that long-chain PFCAs have a similar structure, it is expected that all long-chain PFCAs may have similar adverse effects (although toxic potency may vary with chain length). In addition, increasing temporal concentration trends in top predator wildlife species, including polar bears from remote region, suggest that long-chain PFCAs can approach ecotoxicity thresholds that would result in an increased potential for harm for wildlife populations in the future. Based on the criteria listed in paragraph 1 (e) (i, ii) of Annex D, there is sufficient evidence that C9 – C14 PFCAs are associated with adverse effects to human health and the environment, and that C15 – C21 PFCAs have the potential to be associated with adverse effects to human health and the environment.

6. Statement of the reasons for concern and need for global action

56. Based on the existing data, long-chain PFCAs and their salts can be considered to meet the screening criteria in Annex D of the Stockholm Convention for persistence, bioaccumulation, long-range transport and adverse effects.

57. Due to the ongoing use of PFCAs and the many applications for compounds related to PFCAs, long-chain PFCAs are directly or indirectly emitted into the environment from human activities. Long-chain PFCAs and their salts are persistent, bioaccumulative, have evidence of adverse effects to human health or to the environment, and have the potential to undergo long-range environmental transport in air and water, making emissions of these substances a transboundary pollution problem. Globally, the occurrence and distribution of long-chain PFCAs is demonstrated for wildlife, and environmental matrices, including measurements in remote areas such as the Arctic and Antarctic (see Sections 5.2 and 5.3). Furthermore, long-chain PFCAs have been detected in surface and ground water, as well as in food grown with contaminated soil or water (e.g. Herzke et al. 2013; Loos et al. 2009, 2010; Pan et al. 2018; Fernandes et al. 2012, 2015). In humans, long-chain PFCAs can be distributed to numerous tissues and fluids and can pass through the placenta into the fetus and be transferred through breast milk. This is of particular concern given that certain long-chain PFCAs were shown to have half-lives in humans in the order of years.

58. The evidence for adverse effects in humans relates to observed effects on the following toxicological endpoints: hepatotoxicity, developmental/reproductive toxicity, immunotoxicity, thyroid toxicity and others (e.g., cardiovascular, metabolic, renal toxicity).

59. Effects in wildlife include developmental effects, behavioural effects, hepatotoxicity, immunotoxicity, neurotoxicity, chemosensitivity, altered gene expression and altered thyroid function. In addition, there are unique concerns about the highly persistent and bioaccumulative long-chain PFCAs and their salts. Current science cannot accurately predict the full extent of the ecological effects for these substances but they are acknowledged to have the potential to cause serious and irreversible impacts to wildlife populations in the long-term (MacLeod et al. 2014; Ahrens and Bundschuh 2014). Persistent long-chain PFCAs and their salts remain in the environment for a very long time increasing their probability, magnitude and duration of exposure to wildlife. Persistent

long-chain PFCAs and their salts are subject to long-range transport and can result in regional or global contamination (see Section 5.3). As such, releases of these substances can lead to elevated concentrations in organisms over wide areas over a long period of time. Long-chain PFCAs and their salts also biomagnify through the food chain, resulting in increased internal concentrations for top predators (see Section 5.2). Since long-chain PFCAs are widespread in the environment, several different long-chain PFCAs can be present simultaneously in the tissues of organisms, increasing the likelihood and potential severity of harm. Increasing temporal concentration trends (see Section 5.4, paragraph 44) in wildlife, including top predator species, suggest that long-chain PFCAs can approach toxicity thresholds resulting in harm for wildlife populations in the future.

60. Based on the available information, adverse effects have been observed in wildlife. In addition, available epidemiological and toxicity data indicate the potential for damage to human health. Therefore, international action on long-chain PFCAs, their salts and their related compounds is warranted.

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