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on Persistent Organic
Pollutants**

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Item 6 (b) of the provisional agenda*

Consideration of draft risk profiles: short-chained chlorinated paraffins

**Supporting document for the draft risk profile on short-chained
chlorinated paraffins**

Note by the Secretariat

1. Annex I to the present note provides information on substance identity of short-chained chlorinated paraffins, while annex II contains an updated supporting document for the draft risk profile. Both documents were considered at the fifth meeting of the Persistent Organic Pollutant Review Committee and have since been supplemented by additional information identified subsequently. In the light of the additional information, they have been revised by the Chair and the drafter of the working group established at the Committee's fifth meeting. They have been reproduced as received, without formal editing.
2. The draft risk profile on short-chained chlorinated paraffins has been made available as document UNEP/POPS/POPRC.6/11.

* UNEP/POPS/POPRC.6/1/Rev.1.

Annex I

Considerations on substance identity of short-chained chlorinated paraffins

Prepared by the intersessional working group on short-chained chlorinated paraffins established at third meeting of the Committee

I. Introduction

1. Short-chained chlorinated paraffins (SCCPs) can be described in a number of different manners; definitions may be based on the chemical structure of the substance or on the description of the starting materials and the final commercial product. This document has been developed to provide information relevant to the description of SCCPs as detailed in the UNEP nomination, as well as in various initiatives (including in various assessments and jurisdictions). Also presented is a list of chlorinated alkanes and a non-exhaustive listing of Chemical Abstract Services (CAS) numbers that have some relevance to the various definitions of SCCPs.

II. Production of chlorinated paraffins

2. Chlorinated paraffins (CPs), also referred to chemically as chlorinated alkanes, are produced by the chlorination of a hydrocarbon feedstock. The feedstock used determines the carbon chain length distribution of the product; this distribution may change with changing feedstock. In general, there are three different carbon-chain length feedstocks that are used to manufacture CPs: short-chain (C_{10-13}), medium-chain (C_{14-17}), and long-chain (C_{18-30}). In addition to the alkanes comprising the primary carbon chain lengths of the feedstock (e.g., C_{10-13}), the feedstock will contain impurities of other carbon chain lengths, as well as chemicals other than alkanes, such as olefins (alkenes) and aromatic compounds, which can also become chlorinated.

3. The chemical industry considers the term “chlorinated paraffins” to refer to products produced through chlorination of a petroleum based hydrocarbon stream that has a distribution in carbon chain lengths. Individual chlorinated alkanes are typically not considered chlorinated paraffins by industry. When a specific description is given for commercial chlorinated paraffins, it can be expected that the mixture will fall, on average, within that description, but other compounds may be present. For example, a product described as SCCP, 40% chlorine, will, on average, be composed of chlorinated alkanes that are 40% chlorine by weight and contain predominantly chain lengths between 10 and 13 carbons; the product may also contain lower and higher chlorinated alkanes as impurities.

4. Currently, it is estimated that there are essentially no alkanes with <30% chlorination being produced and likely none with greater than 75% chlorination. The Chlorinated Paraffins Industry Association (CPIA) performed a survey several years ago which covered many of the global producers. In general, the survey suggested that chain lengths and chlorine contents of global chlorinated paraffin production are similar to that in North America.

III. Approaches for defining short-chained chlorinated paraffins

5. Chemically, SCCPs may be defined as straight-chain chlorinated alkanes with a carbon chain length between 10 and 13. More restrictive definitions can specify a certain degree of chlorination (e.g., > 48%). Additionally, a definition could include the individual chain-length compounds covered by the definition, e.g., a single chain of 12 carbons, chlorinated, or could include a specific chemical definition of a class, e.g., $C_xH_{(2x-y+2)}Cl_y$ where $x = 10-13$ and $y = 1-13$.

6. For the chemical industry, the term “chlorinated paraffins” refers to a mixture of chlorinated alkanes with distributions in both carbon chain length and chlorine content. The commercial product consisting mainly of the chemically defined SCCPs would also contain impurities, such as chlorinated paraffins outside of the defined range for SCCPs, and chlorinated alkenes, branched alkanes, and aromatics, as part of the SCCPs mixture.

Definition in Stockholm Convention nomination

7. In the European Union Proposal for Listing SCCPs in the Stockholm Convention (UNEP/POPS/POPRC.2/INF/6), the nominated substance was identified as SCCPs, which "are n-paraffins that have a carbon chain length of between 10 and 13 carbon atoms and a degree of chlorination of more than 48% by weight". It was further noted that "There is a range of commercially available C₁₀₋₁₃ chlorinated paraffins and they are usually mixtures of different carbon chain lengths and different degrees of chlorination although all have a common structure in that no secondary carbon atom carries more than one chlorine" (see Chemical Identity section of proposal). The IUPAC name was given as "Alkanes, C₁₀₋₁₃, chloro". One CAS number (85535-84-8) was cited. The nomination also cited several synonyms: alkanes, chlorinated; alkanes (C₁₀₋₁₃), chloro-(50-70%); alkanes (C₁₀₋₁₃), chloro-(60%); chlorinated alkanes, chlorinated paraffins; chloroalkanes; chlorocarbons; polychlorinated alkanes; and paraffins chlorinated. The synonyms are general in nature, and encompass much more than the substance represented by either the CAS number given or C₁₀₋₁₃ chlorinated alkanes in general. The molecular formula was given as C_xH_(2x-y+2)Cl_y, where x = 10-13 and y = 1-13 (which could include individual alkanes ranging from about 16 to 78% chlorine by weight).

Definitions in other initiatives

8. SCCPs have been the subject of reviews or actions by various jurisdictions, although the scope of those reviews or actions has not necessarily always been the same. Table 1 provides the definitions of SCCPs as described in some of these initiatives.

Table 1: SCCPs definition used in various assessments, legislation and/or jurisdictions

Jurisdiction/ Overseeing body	Regulation/Directive/ Decision	Definition of SCCPs	Reference
OSPAR Commission	PARCOM Decision 95/1	Chlorinated paraffins with carbon chain lengths between and including 10 and 13 and with a chlorination degree of more than 48% by weight	
European Commission	Recommendation 199/721/EC	C _x H _(2x-y+2) Cl _y where x = 10-13 and y = 1-13 CAS No 85535-84-8 EINECS No 287-476-5	
European Commission	Directive 2000/60/EC	C ₁₀₋₁₃ -chloroalkanes CAS No 85535-84-8 EINECS No 287-476-5	
European Commission	Directive 2002/45/EC	Alkanes, C ₁₀ – C ₁₃ , chloro (short-chain chlorinated paraffins)	
United Kingdom	Updated Risk Assessment of Alkanes, C ₁₀₋₁₃ , Chloro	Chlorinated n-paraffins with a carbon chain length of 10-13. It should be noted that around 40 CAS numbers have been used to describe the whole chlorinated paraffin family at one time or another. The CAS number that is listed in IUCLID (85535-84-4) is taken to represent the commercial substance.	UK Environment Agency 2007
European Union	European Union Risk Assessment Report, Alkanes, C ₁₀₋₁₃ , Chloro-, CAS-No.: 85535-84-8, EINECS-No: 287-476-5	Alkanes, C ₁₀₋₁₃ , chloro C _x H _(2x-y+2) Cl _y , where x = 10-13 and y = 1-13 CAS No. 85535-84-8 EINECS No. 287-476-5 Synonyms: alkanes, chlorinated; alkanes (C ₁₀₋₁₃), chloro-(50-70%); alkanes (C ₁₀₋₁₂), chloro-(60%); chlorinated alkanes, chlorinated paraffins; chloroalkanes; chlorocarbons; polychlorinated alkanes; paraffins-chlorinated Notes that there is a range of commercially available C ₁₀₋₁₃ chlorinated paraffins.	EC 2000

Jurisdiction/ Overseeing body	Regulation/Directive/ Decision	Definition of SCCPs	Reference
European Commission	Risk Profile and Summary Report for Short-chained Chlorinated Paraffins (SCCPs)	Short-chained chlorinated paraffins (SCCPs) are n-paraffins that have a carbon chain length of between 10 and 13 carbon atoms and a degree of chlorination of more than 48% by weight. There is a range of commercially available C ₁₀₋₁₃ chlorinated paraffins and they are usually mixtures of different carbon chain lengths and different degrees of chlorination although all have a common structure in that no secondary carbon atom carries more than one chlorine. Alkanes, C ₁₀₋₁₃ , chloro C _x H _(2x-y+2) Cl _y , where x = 10-13 and y = 1-13 CAS No. 85535-84-8 EINECS No. 287-476-5 Synonyms: alkanes, chlorinated; alkanes (C ₁₀₋₁₃), chloro-(50-70%); alkanes (C ₁₀₋₁₂), chloro-(60%); chlorinated alkanes, chlorinated paraffins; chloroalkanes; chlorocarbons; polychlorinated alkanes; paraffins-chlorinated	EC 2005
Canada	Risk assessment (Follow-up Report for Chlorinated Paraffins)	Chlorinated paraffins are chlorinated derivatives of n-alkanes, having carbon chain lengths ranging from 10 to 38 and a chlorine content ranging from 30 to 70% by weight. Short-chain chlorinated paraffins have carbon chains containing 10-13 carbon atoms.	Environment Canada, Health Canada 2008
Australia	Short Chain Chlorinated Paraffins (SCCPs)-Priority Existing Chemical Assessment Report No. 16	Short chain chlorinated paraffins contain between 10 and 13 carbon molecules. The assessment covers SCCPs generally; however, the following substances were specifically cited in the declaration notice: CAS No. 68920-70-7, Alkanes, C6-18, chloro CAS No. 85535-84-8, Alkanes (C10-13), chloro (50-70%) CAS No. 71011-12-6, Alkanes, C12-13, chloro CAS No. 85536-22-7, Alkanes, C12-14, chloro CAS No. 85681-73-8, Alkanes, C10-14, chloro CAS No. 108171-26-2, Alkanes, C10-12, chloro	NICNAS 2001
Switzerland	Ordinance on Risk Reduction related to the Use of Certain Particularly Dangerous Substances, Preparations and Articles (Ordinance on Risk Reduction related to Chemical Products (ORRChem))	From Annex 1.2: Paraffin chlorination products containing 10 to 13 carbon atoms (alkanes, C ₁₀ – C ₁₃ , chloro-) are short-chain chlorinated paraffins. Official French text: “Les produits de la chloration de la paraffine contenant 10 à 13 atomes de carbone (alcanes, C10 à C13, chloro-) sont considérés comme des paraffines chlorées à chaînes courtes.”	http://www. admin.ch/ch/ f/rs/c814_81 .html (official French text of ORRChem)
Japan	Chemical Substance Control Law	The following chemicals are listed in the existing chemical list: Chlorinated paraffins (C=20to32); Chlorinated normal paraffins (C=8to22); Mono or di chloro alkane (C=6to24) No other definition, other than the above	Japan submission via email

Jurisdiction/ Overseeing body	Regulation/Directive/ Decision	Definition of SCCPs	Reference
		names, is given.	
Japan	vPvB regulation	The following chemical is listed: chlorinated paraffins (C=11, Cl = 7-12). No other definition for CPs is given.	Japan submission via email
United States	Toxics Release Inventory	The USEPA does not use the term "SCCPs" to define a category of chemicals subject to regulations. However, under the Toxics Release Inventory, the polychlorinated alkanes, C10-13, category is defined as: $C_xH_{(2x-y+2)}Cl_y$ Where x = 10-13, y = 3-12, and the average chlorine content ranges from 40-70% with the limiting molecular formulas set at $C_{10}H_{19}Cl_3$ and $C_{13}H_{16}Cl_{12}$.	USEPA 1999
United States	Action Plan for Short-Chain Chlorinated Paraffins (SCCPs) and Other Chlorinated Paraffins	SCCPs include the chlorinated paraffins that meet the following definition: $C_xH_{(2x-y+2)}Cl_y$ where x = 10-13, y = 3-12, and the average chlorine content ranges from approximately 40-70% with the limiting molecular formulas set at $C_{10}H_{19}Cl_3$ and $C_{13}H_{16}Cl_{12}$. Any individual chemical meeting this definition is considered an SCCP. Furthermore, this action plan covers any chemical substance or mixture that contains a chemical that meets the definition.	USEPA 2009

9. Table 1 illustrates the differences in the degree of specificity in definitions, and the potential variability in interpretation of substance identity. For example:

(a) In some cases, a definition may be based on the chemical nature of the substance, as specified by a chemical formula and description, and may also refer to a general description of a class or mixture, as specified by a CAS number (e.g., EC recommendation 199/721/EC).

(b) In other cases, the jurisdiction specifies that there are numerous CAS numbers or substance groupings or classes considered within the definition of SCCPs given (e.g., EC 2005, NICNAS 2001), or that a CAS number can be considered representative of SCCPs only (e.g., UK Environment Agency 2007), indicating that the definition is based on the chemical nature of the SCCPs, not on a specific mixture.

(c) Possible ambiguity in the definitions given has been noted. For example, both the Stockholm Convention Nomination (UNEP/POPS/POPRC.2/INF/6) and EC (2005) define SCCPs as C_{10-13} , >48% chlorine by weight and by the chemical formula $C_xH_{(2x-y+2)}Cl_y$, where x = 10-13 and y = 1-13, which translates to about 16-78% chlorine by weight (i.e., some compounds with <48% chlorine by weight).

(d) Group definitions or names put forward as synonyms may in fact describe different individual chemicals, classes, or mixtures (e.g., "chlorocarbons" are not chemically equivalent to "chloroalkanes" [the latter being just one possible form of chlorocarbons]).

IV. Potential range of individual chemical compounds

10. Based on the chemical formula $C_xH_{(2x-y+2)}Cl_y$, where x = 10-13, and the only restriction on y being that it is ≥ 1 , a large number of chlorinated alkanes ranging in chlorine content from 16 to 87% by weight are theoretically possible (see Table 2). However, not all homologues would be expected to be produced by industrial manufacturing processes.

Table 2: Possible C₁₀₋₁₃ chlorinated n-alkanes. Shaded entries in Cl columns indicate chlorine content from Cl=1 to Cl=13; shaded entries in % wt Cl columns indicate percentages greater than 48%; shaded rows indicate alkanes with equal numbers of C and Cl.

C	H	Cl	Mol. Wt.	Wt. Cl	% wt. Cl
10	21	1	176.7	35.5	20.1
10	20	2	211.2	70.9	33.6
10	19	3	245.6	106.4	43.3
10	18	4	280.1	141.8	50.6
10	17	5	314.5	177.3	56.4
10	16	6	349.0	212.7	61.0
10	15	7	383.4	248.2	64.7
10	14	8	417.8	283.6	67.9
10	13	9	452.3	319.1	70.5
10	12	10	486.7	354.5	72.8
10	11	11	521.2	390.0	74.8
10	10	12	555.6	425.4	76.6
10	9	13	590.1	460.9	78.1
10	8	14	624.5	496.3	79.5
10	7	15	659.0	531.8	80.7
10	6	16	693.4	567.2	81.8
10	5	17	727.9	602.7	82.8
10	4	18	762.3	638.2	83.7
10	3	19	796.7	673.6	84.5
10	2	20	831.2	709.1	85.3
10	1	21	865.6	744.5	86.0
10	0	22	900.1	780.0	86.7

C	H	Cl	Mol. Wt.	Wt. Cl	% wt. Cl
12	25	1	204.8	35.5	17.3
12	24	2	239.2	70.9	29.6
12	23	3	273.7	106.4	38.9
12	22	4	308.1	141.8	46.0
12	21	5	342.6	177.3	51.7
12	20	6	377.0	212.7	56.4
12	19	7	411.5	248.2	60.3
12	18	8	445.9	283.6	63.6
12	17	9	480.3	319.1	66.4
12	16	10	514.8	354.5	68.9
12	15	11	549.2	390.0	71.0
12	14	12	583.7	425.4	72.9
12	13	13	618.1	460.9	74.6
12	12	14	652.6	496.3	76.1
12	11	15	687.0	531.8	77.4
12	10	16	721.5	567.2	78.6
12	9	17	755.9	602.7	79.7
12	8	18	790.3	638.2	80.7
12	7	19	824.8	673.6	81.7
12	6	20	859.2	709.1	82.5
12	5	21	893.7	744.5	83.3
12	4	22	928.1	780.0	84.0
12	3	23	962.6	815.4	84.7
12	2	24	997.0	850.9	85.3
12	1	25	1031.5	886.3	85.9
12	0	26	1065.9	921.8	86.5

C	H	Cl	Mol. Wt.	Wt. Cl	% wt. Cl
11	23	1	190.8	35.5	18.6
11	22	2	225.2	70.9	31.5
11	21	3	259.6	106.4	41.0
11	20	4	294.1	141.8	48.2
11	19	5	328.5	177.3	54.0
11	18	6	363.0	212.7	58.6
11	17	7	397.4	248.2	62.4
11	16	8	431.9	283.6	65.7
11	15	9	466.3	319.1	68.4
11	14	10	500.8	354.5	70.8
11	13	11	535.2	390.0	72.9
11	12	12	569.7	425.4	74.7
11	11	13	604.1	460.9	76.3
11	10	14	638.5	496.3	77.7
11	9	15	673.0	531.8	79.0
11	8	16	707.4	567.2	80.2
11	7	17	741.9	602.7	81.2
11	6	18	776.3	638.2	82.2
11	5	19	810.8	673.6	83.1
11	4	20	845.2	709.1	83.9
11	3	21	879.7	744.5	84.6
11	2	22	914.1	780.0	85.3
11	1	23	948.5	815.4	86.0
11	0	24	983.0	850.9	86.6

C	H	Cl	Mol. Wt.	Wt. Cl	% wt. Cl
13	27	1	218.8	35.5	16.2
13	26	2	253.3	70.9	28.0
13	25	3	287.7	106.4	37.0
13	24	4	322.1	141.8	44.0
13	23	5	356.6	177.3	49.7
13	22	6	391.0	212.7	54.4
13	21	7	425.5	248.2	58.3
13	20	8	459.9	283.6	61.7
13	19	9	494.4	319.1	64.5
13	18	10	528.8	354.5	67.0
13	17	11	563.3	390.0	69.2
13	16	12	597.7	425.4	71.2
13	15	13	632.2	460.9	72.9
13	14	14	666.6	496.3	74.5
13	13	15	701.0	531.8	75.9
13	12	16	735.5	567.2	77.1
13	11	17	769.9	602.7	78.3
13	10	18	804.4	638.2	79.3
13	9	19	838.8	673.6	80.3
13	8	20	873.3	709.1	81.2
13	7	21	907.7	744.5	82.0
13	6	22	942.2	780.0	82.8
13	5	23	976.6	815.4	83.5
13	4	24	1011.0	850.9	84.2
13	3	25	1045.5	886.3	84.8
13	2	26	1079.9	921.8	85.4
13	1	27	1114.4	957.2	85.9
13	0	28	1148.8	992.7	86.4

V. SCCP definitions and their relationship to CAS numbers

11. Table 3 provides a partial list of CAS numbers that could represent compounds that contain chlorinated alkanes that fall within the defined range of SCCPs. Given the complexity of this class of substance, an exhaustive listing of CAS numbers is not possible. Nevertheless, the following information is provided as an example.

Table 3: CAS number and name of some chlorinated paraffin mixtures

CAS #	Substance Name
51990-12-6	Chlorowax
61788-76-9	Alkanes, chloro
63449-39-8	Paraffin waxes and Hydrocarbon waxes, chloro
68188-19-2	Paraffin waxes and Hydrocarbon waxes, chloro, chlorosulfonated
68476-48-2	Hydrocarbons, C2-6, chloro
68606-33-7	Hydrocarbons, C1-6, chloro
68911-63-7	Alkanes, chloro, sulfurized
68920-70-7	Alkanes, C6-18, chloro
68938-42-1	Paraffin waxes and Hydrocarbon waxes, chloro, reaction products with naphthalene
68955-41-9	Alkanes, C10-18, bromo chloro
68990-22-7	Alkanes, C11-14, 2-chloro
71011-12-6	Alkanes, C12-13, chloro
72854-22-9	Paraffin waxes and Hydrocarbon waxes, chloro, sulfonated, ammonium salts
73138-78-0	Paraffins (petroleum), normal C5-20, chlorosulfonated, ammonium salts
84082-38-2	Alkanes, C10-21, chloro
84776-06-7	Alkanes, C10-32, chloro
85422-92-0	Paraffin oils, chloro
85535-84-8	Alkanes (C10-13), chloro (50-70%)
85535-85-9	Alkanes (C14-17), chloro (40-52%)
85535-86-0	Alkanes (C18-28), chloro (20-50%)
85536-22-7	Alkanes, C12-14, chloro
85681-73-8	Alkanes, C10-14, chloro
97553-43-0	Paraffins (petroleum), normal C>10, chloro
97659-46-6	Alkanes, C10-26, chloro
104948-36-9	Alkanes, C10-22, chloro
106232-85-3	Alkanes, C18-20, chloro
108171-26-2	Alkanes, C10-12, chloro
108171-27-3	Alkanes, C22-26, chloro

12. Some CAS numbers denote individual chlorinated alkanes, not mixtures. Examples of these are in Table 4.

Table 4: CAS number and name of some individual chlorinated alkanes

CAS #	Substance Name
112-52-7	1-chlorododecane
1002-69-3	1-chlorodecane
2162-98-3	1,10-dichlorodecane
3922-28-9	1,12-dichlorododecane

VI. Concluding summary

13. The following can be stated regarding the definitions of SCCPs in various initiatives:

(a) There is a difference between commercial definitions of SCCPs containing, on average, a given composition, and a strict chemical description of individual compounds or chemical class.

(b) Definitions (whether based on commercial products or identity of component alkanes) can be either very specific (e.g., specifying chlorine content) or more general (e.g., broad class definition such as "C10-C13 chloroalkanes"). Both approaches have been used internationally in a range of regulatory and non regulatory initiatives

(c) With regard to the Stockholm Convention nomination, the descriptions provided include, in different instances, either substances with a range of chlorination of 16-78% chlorination, or >48% chlorination. The descriptions also refer to a specific product (CAS number 85535-84-8), as well as to a variety of products listed as synonyms, which in fact encompass a much wider range of compounds than represented by the CAS number.

VII. References

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Annex II

Updated supporting document for risk profile on short-chained chlorinated paraffins

Prepared by the intersessional working group on short-chained chlorinated paraffins established at third meeting of the Committee, and amended based on additional information

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Executive Summary

Releases of short-chain chlorinated paraffins (SCCPs) can occur during production, storage, transportation, and use of SCCPs. Facility wash-down and spent metalworking / metal cutting fluids are sources to aquatic ecosystems. Although data are limited, the major sources of release of SCCPs are likely the formulation and manufacturing of products containing SCCPs, such as polyvinyl chloride (PVC) plastics, and use in metalworking fluids. While historical use of SCCPs was high in several countries, major reductions have been noted in recent years.

SCCPs are not expected to degrade significantly by hydrolysis in water, and degradation studies and dated sediment cores indicate that they persist in sediment longer than 1 year. SCCPs have atmospheric half-lives ranging from 0.81 to 10.5 days, indicating that they are also relatively persistent in air. The Henry's law constant indicates that SCCPs can remobilise from water to air or from moist soils to air as a result of environmental partitioning, thus facilitating atmospheric partitioning and transport. SCCPs have been detected in a diverse array of environmental samples (air, sediment, water, wastewater, fish and marine mammals) and in remote areas such as the Arctic (which is additional evidence of long range transport). In addition, Arctic Contamination Potential (ACP) modelling and OECD LRTP screening tools suggests that SCCPs have moderate ACP when emitted to air and have properties similar to known POPs that undergo long range transport.

Available empirical (laboratory and field) and modelled data all indicate that SCCPs can accumulate in biota. Laboratory derived BCFs ranged from 1900 – 138 000, depending on the species and congener tested. Field derived BAFs for lake trout ranged from 16 440 – 26 650 wet wt., and modelled BAFs were >5000 for all SCCPs. For some food webs, BMFs were >1, indicating biomagnification. High concentrations of SCCPs in upper trophic level organisms, notably in marine mammals and aquatic freshwater biota (e.g., beluga whales, ringed seals and various fish), is additional evidence of bioaccumulation. SCCPs have also been measured in the breast milk of Inuit women in Northern Quebec. Based on these data, SCCPs are considered bioaccumulative according to the criteria listed in Annex D of the Stockholm Convention.

There is evidence that SCCPs are toxic to sensitive freshwater and marine aquatic organisms at relatively low concentrations. The most sensitive organism, *Daphnia magna*, has a chronic NOEC of 5 µg/L. A chronic NOEC of 7.3 µg/L is reported for the mysid shrimp. Risk quotients show that species including pelagic, benthic, microorganisms may be at risk from exposure to SCCPs.

The International Agency for Research on Cancer considers some SCCPs (average C₁₂, average 60% chlorination) to be possible carcinogens (groups 2B), although questions have been raised regarding the mechanisms for induction of tumours and the relevance for human health of the studies on which this classification was derived. The Science Committee on Toxicity, Ecotoxicity and the Environment suggests that the finding of lung tumours in male mice may be of importance for humans, but this information would not alter the conclusion of its risk characterisation that the use of short-chain chlorinated paraffins poses no significant risk for consumers or for man exposed via the environment (CSTEE, 1998). The EU Risk Assessment Report (EC 2000) summarized the effect of SCCPs in mammalian species. Rodent studies showed dose related increases in adenomas and carcinomas in the liver, thyroid, and kidney. They concluded that there was insufficient evidence to conclude that the carcinogenicity observations in the liver and thyroid in mice and the benign tumours in the kidney of male rats were a male rat specific event and consequently the concern for humans could not be ruled out. Recent investigations have demonstrated that the mechanism for development of kidney tumours does not follow the classic profile of male-rat specific nephropathy; however, the study could not conclude if the mechanism was rat-specific or not. The EU risk assessment (EC 2000) also noted that although there was an increase in alveolar/bronchiolar carcinomas in mice the results were within historical control ranges and the controls had a greater incidence of adenomas of the lung than the treated animals. The EU concluded that there was no significance for human health that could be read into this pattern of results. An independent technical peer review on SCCPs submitted under the UNECE-LRTAP POPs Protocol indicated that aboriginal people living in the Arctic and consuming contaminated animals may be exposed to SCCPs at concentrations greater than the WHO health guideline of 11 µg/kg bw for neoplastic effects (tumour formation) (UNECE-LRTAP POPs Protocol, 2007). A tolerable daily intake (TDI) for SCCPs of 100 µg/kg-bw per day is given by IPCS (1996). Although expert groups have different opinions as to the interpretation of these data, evidence of toxicity and exposure suggests that humans could be at risk.

The increasing regulation of SCCPs in a few geographical areas have resulted in a decrease in SCCPs currently in use and released into the environment. However, evidence suggests that significant amounts are still in use and being released in several countries. The available empirical and modelled data strongly indicate that SCCPs are persistent, bioaccumulative, and toxic to aquatic organisms at low concentrations, and undergo long range environmental transport. In December 2006, the Parties to the UNECE POPs Protocol agreed that SCCPs should be considered as a POP as defined under the Protocol.

Concentrations currently measured in the environment are generally below levels that have been associated with effects in laboratory studies without the incorporation of safety factors. However, concentrations in fish at some urban sites (i.e., Hamilton Harbour and the Detroit River) are within the range of concentrations causing histological effects. SCCPs are measured in Arctic biota, presumably because of their high bioaccumulative potential. Notably, SCCPs are present in Arctic marine mammals, which are in turn food for northern indigenous people. Recent information indicates that concentrations in marine mammals may be decreasing, at least in some locations, though further data is needed to corroborate this. Elevated levels have also been measured in human breast milk, including in remote communities. In view of their persistence, bioaccumulation and toxicity to a range of organisms, and their potential for long range transport, it is considered that SCCPs are likely to cause significant adverse effects. Additionally, simultaneous exposure to SCCP and its analogue MCCP would increase the risks because of similar toxicity profiles of SCCP and MCCP.

Based on the available evidence, it is concluded that SCCPs are likely, as a result of their long-range environmental transport, to lead to significant adverse environmental and human health effects, such that global action is warranted.

1. Introduction

The European Community and its Member States being Parties to the Stockholm Convention nominated on July 26, 2006, Short Chain Chlorinated Paraffins (SCCPs) to be listed in Annexes A, B, or C of the Convention (UNEP/POPS/POPRC.2/INF/6, summarized in UNEP/POPS/POPRC.2/14).

1.1 Chemical Identity of the Proposed Substance

IUPAC Name: Alkanes, C₁₀₋₁₃, chloro

CAS No: 85535-84-8

EINECS No: 287-476-5

Synonyms

chlorinated alkanes (C₁₀₋₁₃)

chloro (50-70%) alkanes (C₁₀₋₁₃)

chloro (60%) alkanes (C₁₀₋₁₃)

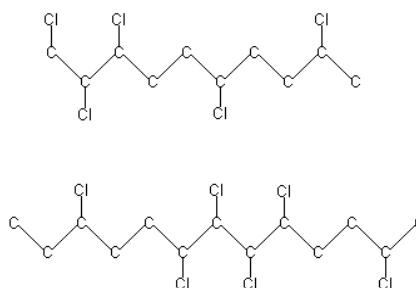
chlorinated paraffins (C₁₀₋₁₃)

polychlorinated alkanes (C₁₀₋₁₃)

paraffins chlorinated (C₁₀₋₁₃)

The proposal nominates short-chain chlorinated paraffins (Alkanes, C₁₀₋₁₃, chloro) with greater than 48% chlorination for listing as a POP under the Stockholm Convention. The proposal identified the substance as CAS No. 85535-84-8 and EINECS No. 287-476-5 (Alkanes, C₁₀₋₁₃, chloro). This CAS # represents the commercial SCCP product that is produced by the chlorination of a single hydrocarbon fraction consisting of n-alkanes that have a carbon chain length distribution consisting of 10, 11, 12 and 13 carbon atoms; however, this CAS # does not specify the degree of chlorination of the SCCP. Within the nomination, SCCPs are defined as C₁₀₋₁₃ and >48% chlorine by weight in section 1, and then as C₁₀₋₁₃ and 1-13 chlorine (~16-78% by weight) in section 1.2. The Stockholm Convention nomination for listing is directed at SCCP products that contain more than 48% by weight chlorination. Examples of two molecules that can be found within an SCCP product are presented in Figure 1.

Figure 1: The structure of two examples of SCCP compounds (C₁₀H₁₇Cl₅ and C₁₃H₂₂Cl₆)



1.2 Conclusion of the Review Committee Regarding Annex D Information

The Persistent Organic Pollutants Review Committee (POPRC) has evaluated the SCCPs proposal against the criteria listed in Annex D of the Stockholm Convention at the second meeting of the POPRC (Geneva, 6-10 November 2006). The Committee decided that SCCPs meet the screening criteria listed in Annex D of the convention (UNEP/POPS/POPRC.2/17 – Decision POPRC-2/8 Annex 1).

1.3 Data Sources

The risk profile for SCCPs builds on information gathered by the EU in its proposal of SCCPs to the POPRC (UNEP/POPS/POPRC.2/INF/6). The risk profile also incorporates information collected from risk assessment documents prepared by Canada (Environment Canada) and the United Kingdom (DEFRA). Annex E information submissions (2007 and 2010) from several POPRC parties and observers submitted following POPRC3 and POPRC5 were also reviewed and any additional information incorporated as appropriate. Information provided by Parties and observers during POPRC3 and POPRC5 has also been incorporated as appropriate.

1.4 Status of the Chemical under International Conventions

In August, 2005, the European Community proposed SCCPs to be added to the UNECE Convention on Long Range Transboundary Air Pollution, Protocol on Persistent Organic Pollutants (LRTAP). SCCPs were proposed to meet the criteria of decision 1998/2 of the Executive Body for persistence, potential to cause adverse effects, bioaccumulation and potential for long range transport. At the 24th session of the Executive Body in December 2006, the Parties to the UNECE POPs Protocol agreed that SCCPs should be considered as a POP as defined under the Protocol, and requested that the Task Force continue with the Track B reviews of the substances and explore management strategies for them. SCCPs were added to Annexes I and II of the 1998 Aarhus Protocol in December 2009 at the 27th session of the Executive Body. Annex II restricts SCCP uses to fire retardants in rubber used in conveyor belts in the mining industry and in dam sealants, and states that action to eliminate these uses should occur once suitable alternatives are available.

In 1995, OSPAR Commission for the Protection of Marine Environment of the North-East Atlantic adopted a decision on SCCPs (Decision 95/1). This established a ban on the use of SCCPs in all areas of application. Under this decision, all sale and use of SCCPs were to be prohibited by the end of 1999. Exemptions were to allow for the use of SCCPs in dam sealants and underground conveyor belts until 2004.

In 2006, OSPAR prepared an overview assessment of the implementation of PARCOM Decision 95/1 on SCCPs (OSPAR, 2006). The assessment was based on national implementation reports received from nine of 15 Contracting Parties which have been requested to submit, in the 2005/2006 meeting cycle, reports on the national measures taken. All reporting Contracting Parties have taken measures to implement PARCOM Decision 95/1. Some Contracting Parties reported a full ban of all or certain uses of SCCPs and reductions of other uses. In general, Contracting Party measures have addressed those uses covered by Directive 2002/45/EC.

Similar to OSPAR, the Baltic Marine Environment Protection Commission (HELCOM) has included SCCPs on their list of harmful substances. On November 15, 2007, HELCOM included SCCPs in the HELCOM Baltic Sea Action Plan. Contracting Parties to HELCOM have agreed, starting in 2008, to work for strict restriction on the use in the whole Baltic Sea catchment area of the Contracting States of several hazardous substances, including SCCPs. Hazardous substances are those found to be PBT or vPvB (Annex E 2010 submission from Lithuania).

The major use of SCCPs in the EU in the 1990's was in metal working fluids (Euro Chlor 1995, OSPAR 2001). Use in metal working fluids, as well as in leather fat liquors, was restricted in directive 2002/45/EC. SCCP in plastics is another main use in Europe that was not covered by this directive.

2 Summary information relevant to the risk profile

2.1 Physical/chemical properties

Tomy et al. (1998a), Muir et al. (2000) and OSPAR (2001) have reviewed the published information to 1998 on physical properties and environmental degradation of chlorinated paraffins. The EU review of SCCPs (EC 2000) includes physical property and degradation information published prior to 1997. There appear to be few new physical property data published since 1998. Environmentally relevant physical properties for SCCPs are summarized in Table 1.

Table 1: Environmentally relevant physical properties of SCCP congeners and mixtures of isomers

SCCP congener	% Cl	Vapour pressure (Pa)	Henry's law constant (Pa·m ³ /mol)	Water solubility (µg/L)	log K _{OW} ¹	log K _{OA} ¹	Reference ²
C ₁₀ H ₁₈ Cl ₄	50	0.028	17.7	328, 630, 2370	5.93	8.2	1,2,3,4
C ₁₀ H ₁₇ Cl ₅	56	0.0040–0.0054	2.62–4.92	449–692	–	8.9–9.0	1,2,3
C ₁₀ H ₁₆ Cl ₆	61	0.0011–0.0022	–	–	–	–	1
C ₁₀ H ₁₃ Cl ₉	71	0.00024	–	400	5.64	–	4
¹⁴ C ₁₁	59	–	–	150	–	–	5
C ₁₁ H ₂₀ Cl ₄	48	0.01	6.32	575	5.93	8.5	1,2,3
C ₁₁ H ₁₉ Cl ₅	54	0.001–0.002	0.68–1.46	546–962	6.20–6.40	9.6–9.8	1,2,3
C ₁₁ H ₁₈ Cl ₆	58	0.00024–0.0005	–	–	6.40	–	1,3
C _{11.5}	60	–	–	–	4.48–7.38	–	6
¹⁴ C ₁₂ H ₂₁ Cl ₅	51	0.0016–0.0019	1.37	–	–	–	1
C ₁₂ H ₂₀ Cl ₆	56	–	–	–	6.61	–	3
¹⁴ C ₁₂ H ₂₀ Cl ₆	56	0.00014–0.00052	–	–	6.2	–	1,7
C ₁₂ H ₁₉ Cl ₇	59	–	–	–	7.00	–	3
C ₁₂ H ₁₈ Cl ₈	63	–	–	–	7.00	–	3
C ₁₂ H ₁₆ Cl ₁₀	67	–	–	–	6.6	–	7
C ₁₃ H ₂₃ Cl ₅	49	0.00032	–	78	6.14	9.4	3
C ₁₃ H ₂₂ Cl ₆	53	–	–	–	6.77–7.00	–	3
C ₁₃ H ₂₁ Cl ₇	58	–	–	–	7.14	–	3
C ₁₃ H ₁₆ Cl ₁₂	70	2.8 × 10 ⁻⁷	–	6.4	7.207	–	4
C _{10–13}	49	–	–	–	4.39–6.93	–	6
C _{10–13}	63	–	–	–	5.47–7.30	–	6
C _{10–13}	70	–	–	–	5.68–8.69	–	6
C _{10–13}	71	–	–	–	5.37–8.69	–	6

¹ Octanol–air partition coefficient calculated from K_{OW}/K_{AW}, where K_{OW} is the octanol–water partition coefficient and K_{AW} is the air–water partition coefficient or unitless Henry's law constant (K_{AW} = HLC/RT, where HLC = Henry's law constant, R = gas constant 8.319 Pa·m³/mol K⁻¹ and T = 293 K).

² References: 1. Drouillard et al. (1998a), measured data; 2. Drouillard et al. (1998b), estimated data; 3. Sijm and Sinnige (1995), measured data; 4. BUA (1992), estimated data; 5. Madeley et al. (1983a), measured data; 6. Renberg et al. (1980), thin-layer chromatography (TLC) – K_{OW} correlation; 7. Fisk et al. (1998a), measured data.

Limited information is available on the physical/chemical properties of various SCCP congeners and mixtures (Renberg et al. 1980, Madeley et al. 1983a, BUA 1992, Sijm and Sinnige 1995, Drouillard et al. 1998a, Drouillard et al. 1998b, Fisk et al. 1998a). Estimated and measured vapour pressures (VPs) range from 0.028 to 2.8 × 10⁻⁷ Pa (Drouillard et al. 1998a, BUA 1992). Major components of SCCP products with 50–60% chlorine are predicted to have subcooled liquid VPs ranging from 1.4 × 10⁻⁵ to 0.066 Pa at 25°C (Tomy et al. 1998a). Henry's Law Constants (HLCs) ranged from 0.7 - 18 Pa·m³/mol (Drouillard et al. 1998a), suggesting that SCCPs can remobilise from water to air as a result of environmental partitioning. Measured water solubilities of individual C_{10–12} chlorinated alkanes ranged from 400 – 960 µg/L (Drouillard et al. 1998b), while estimated solubilities of C₁₀ and C₁₃ chlorinated alkane mixtures ranged from 6.4 - 2370 µg/L (BUA 1992). The logarithms of the octanol–water partitioning coefficient (log K_{OW}) were generally greater than five, ranging from 4.48 – 8.69. The logarithms of the octanol–air partitioning coefficients (log K_{OA}) were estimated using available K_{OW} and HLC values. This was possible for only a limited number of congeners; the values ranged from 8.2 – 9.8.

2.2 Sources

2.2.1 Production

Total reported annual usage of all chlorinated paraffins (CPs) (i.e., all carbon chain lengths) in Canada (production + imports – exports) was approximately 3000 tonnes in 2000 and 2001 (Environment Canada 2003a). The Canadian sales pattern for SCCPs for 1994–1998 (as a proportion of total usage of chlorinated paraffins) was similar to the European sales pattern, rather than the North American sales pattern, which is dominated by the United States (Table 2).

Whether these sales patterns are the same at present is not known. North American demand for total CPs fluctuates depending on the strength of the economy (Camford Information Services 2001). Information submitted by the CPIA (Annex E 2010 submission) indicate that sales in the EU increased in 1999 to just under 6000 tonnes, followed by a steady decrease in sale until 2009, with less than 1000 tonnes sold for the years 2005-2009. Annex E (2010) information submitted by Poland reports that 237.88 tonnes of SCCPs were sold in the EU in 2007. Overall, the decline in SCCP use within the EU is in part owing to the phasing out of production and use in Germany (Stolzenberg 1999; OSPAR 2001) and the EU marketing and Use Directive. Information on the sales of SCCPs from 1994-2004 provided by the CPIA (Annex E 2010 submission) for the four main applications of SCCPs (metal working lubricants; flame retardants in textile and rubber; paints, sealants and adhesives; and leather fat liquors) demonstrate a large decrease in sales in Europe (EU15 in 1994-2003, EU25 in 2004-2006 and EU27 in 2007-2009) between 1994 and 2003, from approximately 9500, 1500, 1800, and 500 tonnes/year for each application, respectively, to less than approximately 500 tonnes each by 2003.

Table 2: Sales of CPs in the EU and North America during the 1990s

CP group	EU ¹			North America ²		
	Year	(tonnes/year)	% total CPs sales	Year	(tonnes/year)	% total sales
SCCPs	1994	13200				
	1997	7370				
	1998	4080	6.4	1998	7900	20.6

¹ OSPAR (2001).

² CPIA (2000).

In addition to production in the US and the EU, it should be noted that there are CP (of various chain length) producers in Russia, India, China, Japan, Brazil and Slovakia. In some cases, total CPs are produced in Asia under licence to the European manufacturer. It is unclear to what extent imports from these countries are accounted for in the information provided by industry associations such as Eurochlor and CPIA (see Table 2). There is no production of SCCPs in Canada (Camford Information Services 2001). The Republic of Mauritius (Comments submitted on April 7, 2008 POPRC SCCPs risk profile), Bulgaria, Costa Rica, Ecuador, Lithuania and Sri Lanka (Annex E 2010 submissions) do not produce SCCPs. Monaco neither produces nor uses SCCPs (Annex E 2010 submission).

Information (submitted under Annex E 2007 of the Stockholm convention and as comments to the draft profile) indicates that SCCPs are produced in Novaky, Slovakia, with quantities ranging from 100 to 584 tonne/year between 1999 and 2006. Japan also noted that there is a possibility of 1% C₁₀ – C₁₃ homologues in medium chained chlorinated paraffin production. Germany noted that there has been no production in Germany since 1995. Prior to 1995, Clariant, Hoechst, and Huels produced SCCPs in Germany. Hoechst produced between 9300 - 19300 tonnes/year in Germany between the years 1993 and 1995.

As noted in the Annex E (2007) information submitted by the USA, some chlorinated paraffins are on the Toxic Substances Control Act (TSCA) inventory and are subject to the Environmental Protection Agency's (EPA's) TSCA inventory update reporting rule under which production and import information is collected. The CAS numbers used in the United States are not specific to SCCPs, hence the information collected includes other chain-length chlorinated paraffins. For 2002, the production and import volumes reported for CAS# 63449-39-8 (paraffin waxes and hydrocarbon waxes, chloro) were in the range of 50 million – 100 million pounds (23 million – 45 million kg), and for CAS # 61788-76-9 (alkanes, chloro; chloroparaffins) in the range of 50 million – 100 million pounds (23 million – 45 million kg). In 1994, for CAS # 68920-70-7, (alkanes, C₆₋₁₈, chloro) production and import volume in the range of 1 million – 10 million pounds (0.45 million – 23 million kg) were reported. The Chlorinated Paraffins Industry Association (CPIA) submitted Annex E (2010) information on the yearly production of SCCPs in North America from 2000 to 2009. Production was approximately 3700 tonnes in 2000, peaked at approximately 4000 tonnes in 2001, and steadily declined to approximately 800 tonnes in 2009.

Annex E (2007) information submitted by Brazil indicates that 150 tonnes/year of SCCPs are produced in Brazil while Slovakia reported production quantities of 560, 354, 480 and 410 tonnes for 2004, 2005, 2006 and 2007, respectively. Twenty manufacturers in India have a combined installed capacity of 110,000 tonnes of CPs per annum.

An exposure assessment of SCCPs in Australia found that over the two-year period of March 1998 to March 2000 approximately 360 tonnes of the materials were imported from Britain and the United States (NICNAS 2001). In the same period approximately 6 tonnes were exported overseas. A follow-up assessment found that one of the companies had ceased imports of SCCPs by 2002 (NICNAS 2004).

In 2006, the Republic of Korea imported approximately 156 tonnes of SCCPs (CAS No. 85535-84-8) (Comments submitted on April 7, 2008 POPRC SCCPs report). In 2005 and 2006, Croatia imported 2392-2790 and 1243-1450 kg, respectively, of SCCPs as a component ($13\pm1\%$) of flame retardant (Annex E 2010 submission). Bulgaria reports importation quantities of SCCPs from 1996-2009 (Annex E 2010 submission). Import quantities were highest in 1997 (379 tonnes), then decreased to 4 tonnes for 2007, with no importation in 2008 and 2009 (2009 data is preliminary). Argentina imported SCCPs in quantities of 40.02 tonnes in 2008 at one location and 53.688 tonnes in 2009 (sum of two locations) (Annex E 2010 submission).

Importation quantities for chlorinated paraffins (no specified chain length) were provided by Ecuador (Annex E 2010 submission). From 1990-2010, 8067 tonnes of chlorinated paraffins were imported, with 4493 tonnes of this during the period 2005-2010. The major supplier of chlorinated paraffins (no specified chain length) to Ecuador during the 1990-2010 time period was the United Kingdom (6255 tonnes), followed by the United States (847 tonnes), India (282 tonnes), Taiwan (209 tonnes), Belgium (165 tonnes), Germany (142 tonnes) and China (100 tonnes). A number of other countries supplied chlorinated paraffins in quantities less than 100 tonnes. Mexico also reported importation quantities for chlorinated paraffins (no specific chain length) (Annex E 2010 submission) of 17,912.5 tonnes from 2002-2010.

Bulgaria (Annex E 2010 submission) reports the quantity of SCCPs exported from the country in plastic articles for the years 1997, 1998, 2003, 2008 and 2009 (preliminary data in 2009). The highest quantity of SCCP containing articles exported was in 1997 (4229 tonnes); the remaining years had a much lower volume, ranging from 4-39 tonnes.

2.2.2 Uses

Nearly all reported usage of SCCPs in Canada was for metalworking applications (Environment Canada 2003a). Minor uses included use as a flame retardant in plastics and rubber. European use pattern data for SCCPs from the years 1994 and 1998 are given in Table 3.

Table 3: Applications of SCCPs in Europe

Application	1994 data ¹		1998 data ²	
	tonnes/year	% of total use	tonnes/year	% of total use
Metalworking lubricants	9 380	71.02	2 018	49.5
PVC plasticizers	Note 3	Note 3	13	0.3
Paints, adhesives and sealants	1 845	13.97	713	17.5
Leather fat liquors	390	2.95	45	1.1
Rubber/flame retardants/textiles/polymers (other than PVC) ³	1 493	11.31	638	15.7
Other	100	0.75	648	15.9
Total	13 208	100	4 075	100

¹ Data from Euro Chlor (1995).

² Data from OSPAR (2001) from Western Europe.

³ The given data did not include information specifically on usage in PVC.

The use of SCCPs in the EU in metalworking (and also in fat liquoring of leather) is now subject to marketing and use restrictions. EU Directive 2002/45/EC, which was adopted in June 2002, restricts the concentration of SCCPs in metalworking and leather fat liquoring preparations to 1% or less. The use of SCCPs in these applications has decreased significantly since the release estimates initially used in the European risk assessment of SCCPs (EC 2000) were obtained (U.K. Environment Agency 2003c). Based on 2004 consumption data, estimated total releases of SCCPs in the EU25 are 4.7-9.5, 7.4-19.6, 0.6-1.8, and 8.7-13.9 tonnes/year to surface water, waste water, air and industrial/urban soil, respectively (Annex E 2010 submission from Germany citing European Chemicals Agency Prioritisation and Annex XIV Background Information for Alkanes, C10-23, chloro).

As noted in the Annex E (2007) submission, 70 tonnes of SCCPs were used in Switzerland in 1994 and while newer data does not exist, it is estimated that uses have reduced by 80%. The most widespread use of SCCPs in Switzerland was in joint sealants where it was often used instead of PCBs in buildings. Canton Basel-Town (Kantonales Laboratorium Basel-Stadt 2001) found that 15 out of 44 joint sealants used in schools and kindergartens contained SCCPs with a content of 2-34%. Canton Argau found that 18 out of 54 joint sealants sampled (years 1960-1976) and 7 out of 29 joint sealants sampled (years 1974-2002) contained SCCPs (Kanton Aargau 2003).

As noted in the Annex E (2007) submission by Germany, the most important uses (74% of the total) of SCCPs were banned by the EU directive 2002/45/EC. SCCPs have been used as a PCB substitute in gaskets (e.g., splices, in buildings) and this may be a source when buildings are renovated. In Poland, SCCPs are used as a flame retardant in rubber conveyor belts (Annex E 2010 submission). Poland produced 16.4% (approximately 39 tonnes) of the 237.88 tonnes of conveyor belts sold in the EU in 2007, and used 23.062 tonnes of conveyor belts. In Romania, approximately 23,700 kg of SCCPs are used as a plasticizer (Annex E 2010 submission). Annex E (2007) information submitted by Brazil indicates that 300 tonnes/year is used in Brazil for the purposes of flame retardant in rubber, car carpet and accessories. It was noted that leather processing and use in paints was not relevant. In 2006, in the Republic of Korea, SCCPs were mainly used in lubricant and additive agents. The release pattern and quantitative data are not available (Comments submitted on April 7, 2008 POPRC SCCPs report). The Republic of Mauritius do use SCCPs (Comments submitted on April 7, 2008 POPRC SCCPs report). Honduras indicates that SCCPs are present in imported products and feedstocks, though the quantity is unknown; uses include in PVC production, additive in paints and sealants, the automotive industry, and in electrical cables (Annex E 2010 submission). In Argentina, SCCPs are mainly imported for the plastics industry (Annex E 2010 submission).

Use of SCCPs in Australia decreased by 80% between 1998/2000 to 2002 to approximately 25 tonnes per annum of SCCPs in the metal working industry. In the 2 year period between 1998 to 2000, 70% of the total import was used as lubricants in the metal working industry (NICNAS 2004).

2.2.2.1 Plastics and Rubber

SCCPs are not used in PVC in the EU (U.K. Environment Agency 2003a). CPs with high chlorine contents (e.g., 70% by weight) can be used as flame retardants in natural and synthetic rubbers (Zitko and Arsenault 1974). All chain lengths of CPs appear to be used in rubber where they have a plasticizing and flame retarding function. An important use for flame retarded rubber appears to be in conveyor belts for mining applications, but the rubber is also used in other applications. In Canada, 8% of CP (SCCP and other chain lengths) usage is as a flame retardant in heavy-duty rubber (Government of Canada 1993a). The amount of CP added is generally in the range 1–4% by weight (Zitko and Arsenault 1974), but can be up to 15% by weight for some applications (BUA 1992).

The results of a survey for the British Rubber Manufacturers' Association was carried out (BRMA 2001) and found that 10.1 – 16.8 % of CPs in conveyor belting rubber was in the form of SCCP with approximately 48-51 tonnes / year being used at the site. Other unidentified CPs (probably SCCPs) included 6.5% (6 tonnes/year) used in shoe soles, and 13% (1.2 tonnes/year) used in industrial sheeting (U.K. Environment Agency 2001).

An assessment in Australia found that over the two-year period of March 1998 to March 2000, 4% of imported SCCPs were used in the production of rubber products (NICNAS 2001).

2.2.2.2 Adhesives/sealants

Various CPs, including SCCPs, are used as plasticizers/flame retardants in adhesives and sealants. Examples include polysulphide, polyurethane, acrylic and butyl sealants used in building and construction and in sealants for double- and triple-glazed windows. The CPs are typically added at amounts of 10–15% by weight of the final sealant, but could be added at amounts up to 20% by weight of the final sealant in exceptional cases (U.K. Environment Agency 2007).

2.2.2.3 Paints

CPs (of various chain lengths) are used as plasticizers, binders and flame retardants in paints. The concentrations used are usually in the range 5–15% by weight. They are reported to be used in marine paints based on chlorinated rubber. Such paints may contain CPs with 70% chlorine by weight as binder and CPs with 40% chlorine by weight as plasticizer (Zitko and Arsenault 1974). Use of SCCPs in paints was noted in various assessments (EC 2000, U.K. Environment Agency 2007, NICNAS 2001). For paints and coatings, there is a general move away from CP-containing products to higher solids/lower volatile organic compound alternative coatings such as epoxies as a result of increased controls on emissions of volatile organic compounds (U.K. Environment Agency 2001).

2.2.3 Releases to the environment

There is currently no evidence of any significant natural source of CPs (including SCCPs) (U.K. Environment Agency 2003b). Anthropogenic releases of CPs into the environment may occur during production, storage, transportation, industrial and consumer usage of CP-containing products, disposal and burning of waste, and land filling of products such as PVC, textiles, painted materials, paint cans and cutting oils (Tomy et al. 1998a). The possible sources of releases to water from manufacturing include spills, facility wash-down and storm water runoff. CPs (including SCCPs) in metalworking/metal cutting fluids may also be released into aquatic environments from drum disposal, carry-off and spent bath use (Government of Canada 1993b). These releases are collected in sewer systems and ultimately end up in the effluents of sewage treatment plants. Information on percentage releases to sewage treatment plants or on removal

efficiency is not currently available. Ecuador notes that the cleaning of metallurgical facilities results in releases to aquatic ecosystems (Annex E 2010 submission).

The major source of releases of SCCPs in the EU was from metalworking applications (EC 2000). Another significant source of release of CPs to the environment is from losses during the service life of products containing CP polymers (PVC, other plastics, paints, sealants, etc.) (EC 2000, U.K. Environment Agency 2003b). These releases are predicted to be mainly to urban/industrial soil and to wastewater.

Data since 1999 reported to Canada's National Pollutant Release Inventory (NPRI) found that very small amounts of CPs (short, medium and long chain) are being released to the Canadian environment by companies that meet the NPRI reporting requirements (NPRI website accessed August 9, 2007; www.ec.gc.ca/pdb/querysite/query_e.cfm). In 2001-2002, the NPRI reported 1.45 tonnes CPs for disposal to landfill and 1.94 tonnes recycling by recovery of organics from two companies in Ontario. Both of these companies use SCCPs as a formulation component in the manufacture of wires and cables and of paints and coatings, respectively. In 2005, NPRI reported that one company in Ontario disposed 0.023 tonnes of Alkanes 10-13, chloro (CAS# 85535-84-8) off-site and 2.967 tonnes were recycled off-site.

In the USA, SCCPs are subject to the Toxic Release Inventory (TRI) reporting as part of a broader category of polychlorinated alkanes (all saturated C₁₀₋₁₃ species with an average chlorine content of 40-70%). Information submitted by the USA during the comment period indicated that, for the 2005 TRI data release year, a total of 42,779 pounds (19,404 kg) of polychlorinated alkanes was reported for on and off site disposal or other releases by U.S. industries subject to reporting. Of the 42,779 pounds, 1,527 pounds (693 kg) were fugitive air emissions; 1,941 pounds (880 kg) were point source air emissions; and 7 pounds (3.2 kg) were surface water discharges (TRI release year 2005 data set frozen on 11/15/2006, released to public 03/22/2007, available at www.epa.gov/tri).

Annex E (2010) submission information from Poland indicate that, in 2000, 9.672 kg/year SCCPs was emitted by Poland and approximately 29.5 kg/year by the EU27 + Norway.

2.2.3.1 Releases from production

Default release estimates from production can be obtained using the emission factors contained in Appendix 1 of the EU (2003) Technical Guidance Document (TGD). These are carried out for a typical production site, assuming production of around 10 000–20 000 tonnes/year. The default emission factors (Table A1.1 of Appendix 1 of the TGD: Main Category 1c; vapour pressure <1 Pa) are 0 to air and 0.003 (0.3%) to wastewater.

There are no producers of SCCPs in Canada and there was previously only one producer of MCCPs and LCCPs in Canada (Pioneer Chemicals Inc. - PCI, Canada). While PCI's production capacity in the year 2000 was 8500 tonnes (Camford Information Services 2001), it is currently not producing any chlorinated paraffins.

Releases from formulation of metalworking fluids

Losses of CPs, including SCCPs, could occur during blending of metalworking fluids. It has been estimated that the likely loss of lubricant at a formulation site would typically be in the region of 1%, with a maximum of 2% (EC 2000). Most of these losses would be controlled losses, such as off-specification material that could not be reused, and would be collected and sent for disposal. The largest consumer of CPs in Sweden (1400 tonnes/year) has estimated its emissions as 0.06 g/kg CP consumed (KEMI 1991). The European assessment (EC 2000) estimated that the loss of SCCPs was 23 tonnes/year in Europe in the mid-1990s. According to the EU TGD (EU 2003), default emission factors for all CPs for the formulation of metalworking fluids are 0.005% to air and 0.25% to wastewater before any on-site treatment, which small blending businesses may not have.

Releases from production of rubber

SCCPs are used as a flame retardant, softener or process oil in rubber. SCCPs used as flame retardants are added to rubber in a proportion of 1–10%. The U.K. draft risk assessment (U.K. Environment Agency 2003b) discussed the release estimates in the TGD (EU 2003) and Use Category Document (BRE 1998) for plastics additive substances used in the polymer industry, such as MCCPs: the release factors for flame retardants (Use Category 22) during the polymer processing step for thermoplastics are 0.1% to air (boiling point <300°C/unknown; vapour pressure <1 Pa) and 0.05% to wastewater (Table A.3.11 of Appendix 1 of the TGD). For thermosetting resins, the release factor to air is 0 and the release factor to wastewater is 0.0005 (0.05%). The EU updated risk assessment of SCCPs (U.K. Environment Agency 2003a) also uses release factors for SCCPs from rubber production from the Use Category Document (BRE 1998). The release factors used are not specified in the assessment, but are assumed to be the same as or similar to those for MCCPs.

Releases from production of textiles and polymeric materials

SCCPs are used in fire proofing of cellulosic textiles and in other polymers, representing about 17% of SCCP use in 1998 (EC 2000). In some applications (e.g., waterproofing fabrics) small amounts of CPs could be applied directly to the textile in an emulsion, which may cause releases to wastewater (EC 2000, U.K. Environment Agency 2001). Releases could occur from washing of these textiles, which are generally used in furniture and other interior decorations.

Releases from production of paints and sealants

Some SCCPs are used in paints to a small extent. Losses to air and wastewater from formulation of SCCP-containing paints and coatings are estimated to be insignificant (U.K. Environment Agency 2003a). Losses to wastewater during the manufacture of sealants are reported to be low or zero. Scrap material and machine cleaning can account for up to 5% solid waste.

2.2.3.2 Releases from use of CP-containing productsReleases from PVC, paints, adhesives and sealants

Losses from the use of SCCPs in paint and sealants are generally regarded as much lower than those from metalworking. The updated SCCP assessment (U.K. Environment Agency 2003a) discusses these losses, but does not give percent losses per unit volume or weight of paint. A considerable amount of the CP-containing paints may end up as waste during the application process and therefore be disposed of in landfill sites. Immediate losses of CPs to air and wastewater during paint application should be minimal due to the low vapour pressures and water solubilities of CPs. However, there will be gradual losses to air and water after application. The draft U.K. assessments of MCCPs and LCCPs (U.K. Environment Agency 2001, 2003b) examined this issue in more detail, as discussed below, and its conclusions should be applicable to SCCPs, given their structural similarities.

Losses from volatilization — Rubber

SCCPs are reported to be used in rubber, with applications mostly in high-density conveyor belts (EC 2000). Releases from use of SCCPs in polymers such as rubber or PVC may also occur via volatilization or from loss of polymeric material as particles during wear and abrasion of the products. Belts may also be recycled into other products at the end of their life. Volatilization of 0.05% of the MCCPs during the lifetime of the product was assumed (U.K. Environment Agency 2003b), and it is reasonable to assume that this would be the case for SCCPs as well.

Losses from leaching —Paints and varnishes

For the industrial application of paints containing MCCPs and LCCPs, the U.K. risk assessments used default emission factors taken from the Use Category Document (BRE 1998), which are 0 to air and 0.1% to wastewater (U.K. Environment Agency 2003b). The same emission factors apply to domestic use of paints. The domestic usage of this type of paint is unknown but likely to be very limited (U.K. Environment Agency 2001, 2003b).

Leaching of MCCPs and LCCPs following application of paints and sealants could be relatively high. The U.K. draft risk assessment assumed a release factor for MCCPs to water for outdoor use in paints and sealants of 0.15% per year over 5–7 years and the same fractional release over the 20- to 30-year lifetime of sealants. No estimate of leaching loss from paints was available in the EU assessment of SCCPs (EC 2000); however, it can be assumed to be similar to that for MCCPs/LCCPs.

Releases from use of metalworking fluids

Losses of SCCPs due to carry-off from workpieces were estimated to be 2.5 kg/site per year for a small user (100-L capacity) and 2500 kg/site per year for a large user (95 000-L capacity) based on the early 1990s (Government of Canada 1993b). The estimated annual losses of CPs from cutting fluid, based on the replacement rates, are thought to be 48% for a large machine shop, 75% for a medium-sized machine shop and 100% for a small machine shop (EC 2000). Not all of this loss is to wastewater. Releases of SCCPs from use in metalworking fluids to wastewater streams in the EU were estimated to be 18% of use (or 733 tonnes/year in 1998) (EC 2000). In addition, the EU assessment estimated that about 3% of metalworking use would be disposed of in landfill. A breakdown of releases of CPs from small and large machine shops is provided in Table 4.

Table 4: Total losses of CPs for a large and small machine shop using oil-based cutting fluids (from EC 2000; U.K. Environment Agency 2001, 2003b)

	Large facility with swarf reprocessing		Small facility with no swarf reprocessing	
Misting/evaporation	2%	to air	2%	to air
Overalls	1%	to water	2%	to water
Leaks	1%	to water ¹	3%	to water ¹
Dragout/swarf	27%	incinerated	81%	incinerated

	Large facility with swarf reprocessing		Small facility with no swarf reprocessing	
	3%	to landfill	9%	to landfill
Dragout/workpiece	1%	to water	1%	to water
	2%	chemical waste	2%	chemical waste
Internal reprocessing	1%	to water ¹		
External reprocessing	10%	reused/discarded as waste oil		
Total losses	48%		100%	

¹ These losses may be further minimized by collecting the cutting fluid for reuse.

The U.S. EPA (1993) assessment estimated that approximately 90% of CPs used in water-based metalworking fluids were removed via pre-discharge separation or treatment, so that only 10% of CPs in water-based metalworking fluids were assumed to be discharged to wastewater. Assuming removal from degradation and sorption to solids, actual releases from WWTPs were much less than estimated in the U.K. Environment Agency (2001, 2003b) draft assessments. In a critique of the EU assessment, CPIA (2000) indicated that estimates developed by the U.S. EPA are more relevant and appropriate for use in the USA than the estimates derived by the United Kingdom for the EU.

The EU estimates that 80% of SCCPs are used in oil-based metalworking fluids and only 20% in water-based applications (EC 2000).

U.K. Environment Agency (2003b) estimated releases for metal cutting/metalworking fluid additives using release factors from Appendix 1 of the TGD (EU 2003) to be 18.5% from oil-based fluids and 31.6% from water-based (emulsifiable) fluids. Emissions to air are estimated at 0.02% from both types of fluids. These release estimates from the TGD are worst-case, default estimates.

Releases from leather fat liquid and carbonless copy paper

According to industry data, there are no uses of CPs in North America in leather fat liquid and carbonless copy paper. The use of SCCPs in leather fat liquors has been restricted in the EU. Beginning January 6, 2004, EU Directive 2002/45/EC restricts the concentration of SCCPs in leather fat liquoring preparations to 1% or less. The updated risk assessment of SCCPs (U.K. Environment Agency 2003a) assumes no use of SCCPs in leather fat liquor.

Releases from other uses

CPIA (2002), in a review of the Canadian SCCP United Nations Economic Commission for Europe (UNECE) information dossier (Environment Canada 2003b), has indicated that releases from gear oil packages, fluids used in hard rock mining and equipment use in other types of mining, fluids and equipment used in oil and gas exploration, manufacture of seamless pipe, metalworking and operation of turbines on ships may explain the presence of SCCPs in remote environments. However, there is currently insufficient information to assess the relative importance of these activities to the presence of SCCPs in the Arctic.

2.2.3.3 Releases during disposal

Landfilling is a major disposal route for polymeric products in Canada. CPs would be expected to remain stabilized in these products, with minor losses to washoff from percolating water. Leaching from landfill sites is likely to be negligible owing to strong binding of CPs to soils. Minor emissions of these products, which are effectively dissolved in polymers, could occur for centuries after disposal (IPCS 1996).

The releases and bioavailability of CPs from polymers that are landfilled or from losses of polymeric material as particles during wear and abrasion of flooring, rubber products, etc., are unknown. These releases could be sources of input of CPs to air and soils in urban and industrial areas (U.K. Environment Agency 2001, 2003a,b).

Polymer-incorporated CPs could also be released during recycling of plastics, which may involve processes such as chopping, grinding and washing. If released as dust from these operations, the CPs would be adsorbed to particles because of high sorption and octanol-air partition coefficients.

2.2.3.4 Overall emissions in Europe and estimates for North America

Fractional losses of SCCPs to wastewater and surface waters have been estimated based on EU data (EC 2000) and are summarized in Table 5. Behaviour similar to that of MCCPs (U.K. Environment Agency 2003b) is assumed. Overall most releases of SCCPs are expected to be associated with metal working operations, however there is potential for widespread release in small amounts associated with uses in products (e.g. paints, textiles, rubber).

Table 5: Estimated fractional losses of SCCPs in the EU to wastewaters, surface waters and the terrestrial environment

Application	Release to each compartment		
	Wastewater ¹	Surface water ²	Terrestrial ³
Metalworking lubricants	18%	1.4%	17.8%
Paints and sealants	0.1%	0.015%	Unknown — Landfilling of used material
Rubber/flame retardants/ textiles/polymers (other than PVC)	0.1%	0.05–0.4%	Unknown — Landfilling of used material

¹ Wastewater during use (metalworking fluids) or product formulation (paints/polymers).

² For metalworking fluids, surface water = $0.08 \times$ wastewater. For PVC and paints/adhesives/sealants, direct losses to surface water are included.

³ Terrestrial = soil + landfilling/burial, assuming landfilling or sludge spreading, except for PVC and paints/adhesives/sealants, where direct losses to urban/industrial soils need to be considered.

2.3 Environmental Fate

2.3.1 Persistence

2.3.1.1 Persistence in Air

SCCPs were detected in four individual samples of air collected at Alert at the northern tip of Ellesmere Island in the high Arctic (Tomy 1997; Bidleman et al. 2001). Concentrations ranged from <1 to 8.5 pg/m^3 in gas-phase samples (see Table 12). The occurrence of SCCPs at a remote site like Alert indicates that the substance is subject to long range transport. The occurrence of long range transport is one line of evidence suggesting that SCCPs are persistent chemicals.

Tomy et al. (2000) reported SCCPs in the blubber of ringed seal from Eureka, southwest Ellesmere Island, beluga whales from northwest Greenland and the Mackenzie Delta and walrus from northwest Greenland at concentrations ranging from 199 to 626 ng/g wet wt. Tomy et al. (2000) also observed that the concentration profiles for the Arctic marine mammals show a predominance of the shorter carbon chain length congeners, i.e., the C_{10} and C_{11} formula groups. Drouillard et al. (1998a) showed that these congeners are the more volatile components of SCCP mixtures, which show a trend of decreasing VPs with increasing carbon chain length and degree of chlorination. SCCPs have also been detected in sediments from Hazen Lake and Lake DV09 in the Arctic, at levels of 7 and 17 ng/g dry wt., respectively (Tomy et al. 1998a, Stern and Evans 2003).

Although CPs do not degrade by direct photolysis in air, they theoretically would be subject to attack via hydroxyl radicals in the troposphere (Bunce 1993). Estimated atmospheric half-lives for SCCPs based on reaction with hydroxyl radicals are shown in Table 6 and range from 0.81 to 10.5 days, using the default atmospheric hydroxyl radical concentration of $1.5 \times 10^6 \text{ molecules/cm}^3$ during sunlight hours in AOPWIN (v. 1.86) computer program (Meylan and Howard 1993; Atkinson 1986, 1987). In the risk assessment methodology used in the EU, a lower hydroxyl radical concentration of $5 \times 10^5 \text{ molecules/cm}^3$ is generally used as a daily (24-hour) average, which is typically found in relatively unpolluted air. Use of the latter hydroxyl radical concentration results in atmospheric half-lives ranging from 1.2 to 15.7 days (Table 6). It should be noted that hydroxyl radical reaction rates vary temporally with average daily sunlight, and $5 \times 10^5 \text{ molecules/cm}^3$ may not be typical of northern latitudes since hydroxyl radical concentrations decline with latitude. In addition, the high adsorption of CPs to atmospheric particles at low temperatures, typical of Canadian conditions, may limit the atmospheric oxidation pathway.

Table 6: Estimated atmospheric half-lives for SCCPs calculated using the Syracuse Research Corporation AOPWIN computer program

Example structure	Chlorine content (% by weight)	Estimated k_{OH} ($\text{cm}^3/\text{molecule per second}$)	Estimated atmospheric half-life (days)	
			$[OH]^1 = 1.5 \times 10^6 \text{ molecules/cm}^3$	$[OH] = 5 \times 10^5 \text{ molecules/cm}^3$
$C_{10}H_{21}Cl$	20.1	9.75×10^{-12}	1.1	1.6
$C_{10}H_{20}Cl_2$	33.6	8.16×10^{-12}	1.3	2.0
$C_{10}H_{19}Cl_3$	43.4	6.57×10^{-12}	1.6	2.4
$C_{10}H_{18}Cl_4$	50.7	5.17×10^{-12}	2.1	3.1
$C_{10}H_{17}Cl_5$	56.4	5.22×10^{-12}	2.0	3.1
$C_{10}H_{16}Cl_6$	61.0	3.77×10^{-12}	2.8	4.3
$C_{10}H_{15}Cl_7$	64.8	3.68×10^{-12}	2.9	4.4

C ₁₀ H ₁₄ Cl ₈	68.0	2.63×10^{-12}	4.1	6.1
C ₁₀ H ₁₃ Cl ₉	70.6	1.59×10^{-12}	6.7	10.1
C ₁₀ H ₁₂ Cl ₁₀	72.9	1.02×10^{-12}	10.5	15.7
C ₁₃ H ₂₇ Cl ₁	16.2	13.2×10^{-12}	0.81	1.2
C ₁₃ H ₂₆ Cl ₂	28.0	10.9×10^{-12}	0.98	1.5
C ₁₃ H ₂₅ Cl ₃	37.0	9.35×10^{-12}	1.1	1.7
C ₁₃ H ₂₄ Cl ₄	44.1	7.76×10^{-12}	1.4	2.1
C ₁₃ H ₂₃ Cl ₅	49.8	7.11×10^{-12}	1.5	2.3
C ₁₃ H ₂₂ Cl ₆	54.5	5.94×10^{-12}	1.8	2.7
C ₁₃ H ₂₁ Cl ₇	58.4	4.96×10^{-12}	2.2	3.2
C ₁₃ H ₂₀ Cl ₈	61.7	4.87×10^{-12}	2.2	3.3
C ₁₃ H ₁₉ Cl ₉	64.6	4.39×10^{-12}	2.4	3.7
C ₁₃ H ₁₈ Cl ₁₀	67.1	3.34×10^{-12}	3.2	4.8
C ₁₃ H ₁₇ Cl ₁₁	69.3	2.30×10^{-12}	4.7	7.0
C ₁₃ H ₁₆ Cl ₁₂	71.3	1.40×10^{-12}	7.7	11.5
C ₁₃ H ₁₅ Cl ₁₃	73.0	1.31×10^{-12}	8.2	12.2

¹ Assumes 12 hours of sunlight per day.

The major SCCP structures observed in environmental samples, such as in Great Lakes and Arctic air and biota — C₁₀H₁₇Cl₅, C₁₀H₁₆Cl₆, C₁₀H₁₅Cl₇, C₁₁H₁₈Cl₆, C₁₁H₁₇Cl₇, C₁₂H₂₀Cl₆, C₁₂H₁₉Cl₇ — all have estimated atmospheric half-lives greater than 2 days. Furthermore, these structures predominate in SCCP products analyzed by Tomy (1997).

Van Pul et al. (1998) modelled the atmospheric transport of SCCPs and other semivolatile organics. Taking into account wet and dry deposition processes and using a relatively long atmospheric photochemical degradation half-life of 96 hours, they predicted atmospheric half-lives for SCCPs of 23 hours over land and 27 hours over the sea. Unfortunately, the physical property data used by van Pul et al. (1998) for SCCPs are not provided, and their source is unclear. Their study predates the publication of VPs by Drouillard et al. (1998a) as well as other physical properties that apply to individual formula groups.

Based on several lines of evidence, such as their similarity in VP and HLC values to POPs known to undergo long-range transport; their detection in air, sediment and marine mammals in the arctic; the predominance of more volatile chain-lengths in biota; and estimated half-lives greater than two days for many congeners; SCCPs appear to be sufficiently persistent in air for long range transport to occur.

2.3.1.2 Persistence in water

Although SCCPs can undergo slow hydrolysis (Annex E 2010 submission from Costa Rica, citing data from UK Marine SACs Project Chlorinated Paraffins profile, http://www.ukmarinesac.org.uk/activities/water-quality/wq8_35.htm), they are not expected to degrade significantly by abiotic processes such as hydrolysis (IPCS 1996, U.K. Environment Agency 2003a,b). Koh and Thiemann (2001) showed that SCCP mixtures underwent rapid photolysis in acetone–water under ultraviolet light (mercury arc lamp; approximately 254 - 436 nm) with half-lives of 0.7–5.2 hours. The half-life of a 52% chlorine by weight SCCP product in pure water under the same conditions was 12.8 hours. Photoproducts included n-alkanes. While these results suggest that photolysis may be a degradation pathway for some SCCPs, the environmental relevance of this study is questionable for three reasons: 1) the use of an ultraviolet wavelength irradiation source may have produced far shorter half-lives than under natural light conditions; 2) the extent of photodegradation may be limited in waters at depth and/or at northern latitudes, and 3) acetone is a questionable solvent to use in such a study as it is a photo sensitizer.

As part of an Annex E submission (2010), the CPIA submitted the results of a prolonged ready biodegradability test for a 49.75% chlorine by weight SCCP product (Cereclor 50LV) following the OECD 301, Closed Bottle Test method with modifications to prolong the test duration to 56 days. Due to the low water solubility of the SCCP, a solvent (silicone oil) or an emulsifier (polyalkoxylate alkylphenol) were employed to increase the bioavailability of the test substance as allowed in OECD 301. Secondary activated sludge or river water was used as an inoculum. Blanks with the solvent or emulsifier alone were not included. Biodegradation in excess of 60% was only obtained when the SCCP was emulsified using polyalkoxylate alkylphenol, with this percentage exceeded within 28 and 42 days when sludge and river water was used, respectively. The SCCP in silicone oil only achieved a maximum of 38% degradation within 56 days. Therefore, this low chlorinated SCCP passed the ready biodegradable criteria only when emulsified and in the presence of an inoculum obtained from sewage sludge. In the presence of the more environmentally relevant inoculum from river water, it was not readily biodegradable within 28 days, though significant degradation occurred within 42

days in the presence of an emulsifier. Increasing bioavailability through the use of a solvent did not lead to significant biodegradation. Therefore, it can be concluded that this low chlorinated SCCP can be readily degraded if conditions are favourable; however, the relevancy of the use of a surfactant to estimate actual biodegradation in aquatic environments is questionable. It is not known to what extent higher chlorinated SCCPs would be degraded under similar conditions; studies on the biodegradation of SCCPs in sediments suggest that lower chlorinated SCCPs are much more biodegradable than those with higher chlorination.

Based on the available information, SCCPs appear to be hydrolytically stable, and while there is evidence that low chlorinated SCCPs can readily degrade in water under enhanced conditions, the ecological relevance of these test results are unknown. Therefore, there is insufficient information to conclude on the persistence of higher chlorinated SCCPs in water.

2.3.1.3 *Persistence in soil and sediment*

SCCP residues were found in the surficial sediments of the following remote Canadian Arctic lakes (reported in ng/g dry wt., see Section 2.4.5): Yaya Lake (1.6), Hazen Lake (4.5) and Lake DV-09 (17.6) (Tomy et al. 1999, Stern and Evans 2003) (Table 17). Concentration profiles of SCCP residues in Canadian sediments from Lake Winnipeg, Manitoba, and Fox Lake, Yukon, indicated that residues were present in the slices dated at 1947 in the sediments from both of these lakes (Tomy et al. 1999). SCCP residues in sediments were observed from the west basin of Lake Ontario (Canada) dating back to 1949. The highest concentration (800 ng/g dry wt.) was observed in the slice dated at 1971 (Figure 6) (Muir et al. 1999a). A dated sediment core from Lake Thun (Switzerland) covering 1899 to 2004 provides an overview of the historical trend of chlorinated paraffins (Iozza et al. 2008). The maximum SCCP concentration in sediment was 33 ng/g dry weight in 1986.

In the absence of information on loading for any of the years at any of these locations, it is not possible to calculate a discrete half-life value from these data. However, the fact that SCCP residues were detected in sediment cores dating back to the 1940s at these locations is convincing evidence that SCCPs can persist for long periods in sediment. Residues observed in slices dated earlier can be explained by contamination as an artefact of core sampling (personal communication, D. Muir, Environment Canada, 2000). Environment Canada (2008) used first order decay equations in a back calculation method to determine that SCCPs have a half-life in sediments longer than 1 year, which is greater than the criterion for sediments listed in Annex D of the Stockholm Convention. While the back calculation method for determining half lives does not provide a discrete value for the half life of a chemical it can provide an answer as to whether a chemical's half life is significantly greater than a specified timeframe.

Concentration profiles of SCCPs in sediments from Lake Winnipeg, Fox Lake in the Yukon, the west basin of Lake Ontario (see Figure 6) and Lake DV09 (see Figure 8) indicate that SCCP residues were present in the 1940s (Muir et al. 1999a, Tomy et al. 1999). Some transformation of SCCP chain length and formula groups may have occurred over time (i.e., with depth) in these cores, judging by the appearance of greater proportions of shorter chain lengths and lower chlorinated alkanes at shallower depths (Tomy et al. 1999). Tomy et al. (1999) hypothesized that this may be a result of rapid aerobic degradation occurring at these depths. However, these shifts may also be due to changing product formulations at upstream manufacturing and waste treatment sources. In the absence of information on loading for any of the years at any of these locations, it is not possible to calculate specific half-lives from these data. However, the fact that SCCP residues were detected in sediment cores at these dates is convincing evidence that SCCPs can persist for more than 50 years in subsurface anaerobic sediments.

The European risk assessment of SCCPs (EC 2000) concluded that slow biodegradation in the environment may occur, but assumes that SCCPs are not readily biodegradable, because it was not possible to estimate a reliable biodegradation rate with the available data.

Several government assessments and published reviews have concluded that slow biodegradation in the environment may be expected to occur, particularly in the presence of adapted micro-organisms (Government of Canada 1993a,b; Tomy et al. 1998a; EC 2000).

Madeley and Birtley (1980) used 25-day biochemical oxygen demand (BOD) tests to examine the biodegradation of a range of CPs with different carbon chain lengths and chlorine contents. Acclimatized micro-organisms showed a greater ability to degrade CPs than did organisms normally used for treating domestic sewage, and increasing chlorination inhibited biodegradation. An SCCP with 49% chlorine appeared to be rapidly and completely degraded by acclimatized micro-organisms after 25 days. No significant oxygen uptake was observed in tests using the highly chlorinated CPs, which included two SCCPs (60% and 70% chlorine) and one MCCP (58% chlorine).

Fisk et al. (1998a) found that two ¹⁴C-labelled C₁₂ chloro-n-alkanes (56% and 69% chlorine) were degraded at 12°C in aerobic sediments used for a study of the bioavailability of SCCPs to oligochaetes. The biodegradation in sediments was estimated by comparing the amount of toluene-extractable and toluene-non-extractable ¹⁴C after 18 and 32 days of

incubation. It was hypothesized that the toluene-non-extractable ^{14}C fraction represented a polar degradation product of the CPs, which was tightly bound to the sediments and therefore non-extractable using toluene. Half-lives in sediment were 12 ± 3.6 days and 30 ± 2.6 days for the 56% and 69% chlorine products, respectively. No degradation product identification was performed. This study, therefore, would not provide any indication of the mineralization or ultimate degradation half-life of SCCPs.

Allpress and Gowland (1999) isolated a strain of *Rhodococcus* bacteria that could utilize SCCP with 49.5% chlorine as the sole carbon source with the release of chloride ion. They inoculated 100 mL of growth medium with the bacteria and solutions of the CP formulations at 1% v/v. The test lasted 71 days. The greatest release of chloride ion (49%) occurred with the SCCP with 49.5% chlorine, compared with MCCP and LCCP formulations containing 42–47.5% chlorine by weight. The isolate was unable to utilize a more highly chlorinated SCCP (63.5% chlorine) or MCCP (58.5% chlorine) formulation.

A study on the aerobic and anaerobic biodegradation of SCCPs in both freshwater and marine sediments was recently undertaken by Thompson and Noble (2007), as reported in the Draft Updated Risk Assessment of alkanes, C10-14, Chloro (U.K. Environment Agency 2007), an update of the European assessment (EC 2000). Using ^{14}C -labelled n-decane and n-tridecane 65% chlorine by weight products and basing their experiments on the OECD 308 Test Guideline (aerobic and anaerobic transformation in aquatic sediment systems), the mean half-lives (for mineralization [carbon dioxide or methane production]) for a C₁₀₋₁₃, 65% chlorine by weight product were estimated to be 1630 days in freshwater sediments and 450 days in marine sediments under aerobic conditions. Little or no mineralization was noted in anaerobic sediments. The results indicate that SCCPs are persistent in sediments.

Omori et al. (1987) studied the CP dechlorination potential of a series of soil bacterial strains acting on C₁₂H₁₈Cl₈ with 63% chlorine. Although they could not isolate a bacterial strain that could use CP as a sole carbon source, they did find that different strains pre-treated with n-hexadecane had different dechlorination abilities. A mixed culture (four bacterial strains) released 21% of the chlorine after 48 hours. The bacterial strain HK-3, acting alone, released 35% of the chlorine after 48 hours. Omori et al. (1987) suggested that the mechanism for the aerobic degradation was a β -oxidation enzyme system, which would first oxidize the terminal methyl group to produce chlorinated fatty acids and then break down the acids to 2- or 3- chlorinated fatty acids.

Nicholls et al. (2001) were unable to detect SCCPs (<0.1 $\mu\text{g/g}$) in farm soils in the United Kingdom on which several applications of sewage sludge had been applied. However, the study did not specifically follow the fate of SCCPs over time following sludge application.

Based on the available data, though biodegradation may occur in some situations, in general SCCPs appear to be persistent in sediments according to the criterion listed in Annex D of the Stockholm Convention. Insufficient information is available to determine if SCCPs meet the criterion for persistence in soils.

2.3.2 Bioaccumulation

2.3.2.1 Modelled Log KOW and Bioaccumulation Factors

It was calculated that log K_{OW} ranged between 4.8 and 7.6 for all possible SCCP congeners using the equation developed by Sijm and Sinnige (1995). QSAR based models (e.g. EPI KowWin) were used to compare to this empirical model but the predictions (ranging 5.4 – 21.6) are unreliable due to the results exceeding the limits of the training set. However the large amounts of high Kow readings suggest a large potential for bioaccumulation. The few experimental data available also fit with the modelled range. Fisk et al. (1998b) determined the octanol-water partition coefficients of two ^{14}C -labelled short-chain chlorinated paraffins of single carbon chain length (C₁₂). The two compounds used were C₁₂H_{20.1}Cl_{5.9}, 55.9% wt. Cl and C₁₂H_{16.2}Cl_{9.8}, 68.5% wt. Cl. The mean log Kow values determined by a HPLC method were reported to be 6.2 for the 55.9% wt. Cl substance (range of log Kow was 5.0 to 7.1 for the main components of this substance) and 6.6 for the 68.5% wt. Cl substance (range of log Kow was 5.0 to 7.4). Using the empirical Kow data in the Gobas BAF model with an assumption of no metabolism finds that all possible SCCP congeners have a BAF > 5000. As such it is likely they are bioaccumulative.

2.3.2.2 Bioconcentration

Reported bioconcentration factors (BCFs) calculated from laboratory studies for SCCPs have been reviewed in Government of Canada (1993a) and vary dramatically among different species. Relatively low BCF values have been determined in freshwater and marine algae (<1–7.6). BCF values of up to 7816 wet wt. have been measured in rainbow trout (*Oncorhynchus mykiss*) (Madeley and Maddock 1983a,b) and 5785–138 000 wet wt. in the common mussel (*Mytilus edulis*) (Madeley et al. 1983b, Madeley and Thompson 1983, Renberg et al. 1986). Fisk et al. (1999) measured BCFs in Japanese medaka (*Oryzias latipes*) embryos exposed to a range of concentrations of ^{14}C -labelled C₁₀ and C₁₂ SCCPs for 20 days. Highest BCFs (1400–2700) were found for C₁₀H₁₅Cl₇, compared with C₁₂H₂₀Cl₇ (740–1700). These

BCF values were similar to previous measurements for fish, although it should be noted that the test organisms in this case were embryos and therefore accumulating SCCPs solely by respiration.

Information submitted by Japan on the measured BCF test results for C=11, Cl = 7-10 suggested that these chemicals are bioaccumulative (Table 7). In addition the BCF of chlorinated paraffins (C=13-17, Cl = 6-12) were observed to be low (Table 8). Further information submitted by Japan for the POPRC5 review (UNEP/POPS/POPRC.5/INF/23) reported measured BCF test results for carp for a C=13, Cl = 4-9 SCCP. Due to the low concentrations of the Cl = 4, 8 and 9 substitutions and lack of sensitivity of the analysis for such concentrations, steady-state BCFs were only determined for the Cl = 5, 6 and 7 substitutions. BCF measurements were taken at two test concentrations (1 and 10 µg/L); however, it is unclear if these concentrations were for the congeners at each chlorination level or for the mixture itself. The measured concentrations of C=13 at each chlorination level suggest the test concentration was for congeners at each chlorination level. The exposure duration was 62 days. Bioconcentration potential tended to increase with increasing chlorination. BCFs ranged from 1530 to 2830. The test results from Japan indicate that BCFs for some C=13 congeners are lower than BCFs for some congeners of C=11.

Table 7: BCF test results for some SCCP congeners (C=11, Cl=7-10)

		12 days	24 days	38 days	47 days	60 days
Cl=7	1 µg/L	4600	3900	8100	5000	5500
		4900	5400	3300	4000	5200
	0.1 µg/L	2900	3000	3800	3600	2300
		2499	3700	5900	1900	2200
Cl=8	1 µg/L	4900	4700	9200	6000	7200
		5300	6400	4200	5100	7200
	0.1 µg/L	3200	3300	4100	4200	2800
		2500	4400	7000	3200	2700
Cl=9	1 µg/L	4800	5700	11000	7100	8600
		5600	7600	4800	6500	8800
	0.1 µg/L	3600	4100	5600	5300	3700
		2700	5400	9500	4400	3500
Cl=10	1 µg/L	4400	5300	9200	7400	8000
		4600	6700	4300	6200	7700
	0.1 µg/L	4500	3300	3800	4400	3200
		2600	4800	6700	4600	3300

Table 8: BCF test results for some SCCP congeners (C=13-17; Cl = 6-12)

	2 week	4 week	6 week	8 week
0.36ppm	0.9	0.9	1.1	0.9
	0.9	1.1	0.7	1.0
0.036ppm	3.2	4.5	3.0	3.9
	3.6	3.3	3.6	3.4

Laboratory studies of bioaccumulation, biomagnification and biotransformation

SCCPs are readily accumulated from food by fish in laboratory experiments. Dietary accumulation is influenced by carbon chain length and chlorine content (Fisk et al. 1996, 1998b, 2000). SCCPs with greater than 60% chlorine by weight were found to have equilibrium biomagnification factors (BMFs) greater than 1. Compounds with BMFs greater than 1 will theoretically be biomagnified in food webs. Equilibrium BMFs were predicted from the equation $BMF = \alpha \cdot F/k_d$, where α , the assimilation efficiency, was assumed to be 0.5, F is the feeding rate (lipid corrected) and k_d is the depuration rate (Fisk et al. 1998b). Half-lives in fish ranged from 7 to about 53 days in laboratory studies with juvenile rainbow trout, which implies a potential to biomagnify in aquatic food chains (Fisk et al. 1998b). When compared with the half-lives of 40–60 days in rainbow trout for non-metabolizable organochlorine compounds of similar log K_{OW} values of 6 or greater (i.e., PCBs, mirex, hexachlorobenzene), it appeared that SCCPs with lower chlorination were being metabolized by the fish, but biotransformation products were not measured (Fisk et al. 2000). Metabolism was inferred from lower amounts of toluene-extractable ^{14}C residues after 40 days of depuration for some SCCPs. However, the half-lives for two chlorododecanes, $C_{12}H_{16}Cl_{10}$ and $C_{12}H_{20}Cl_6$, were similar to those of recalcitrant organochlorines. These compounds had long half-lives (Table 9) and essentially no biotransformation (biotransformation half-lives >1000 days) based on this method of calculation. The lack of biotransformation was probably related to the particular chlorine substitution pattern on the n-alkanes; however, only the molecular formulas of these SCCPs were known.

Table 9: Half-lives and estimated biotransformation rates of SCCPs in rainbow trout (from Fisk et al. 2000)

SCCP	Depuration half-life (days) ¹	Biotransformation half-life (days) ²	% depuration associated with biotransformation	Reference
C ₁₀ H ₁₈ Cl ₄	8.3 ± 1.5	10	81	Fisk et al. (1998b)
C ₁₀ H ₁₇ Cl ₅	7.8 ± 1.3	9	85	Fisk et al. (1998b)
C ₁₀ H ₁₇ Cl ₅	7.1 ± 1.2	8	87	Fisk et al. (1998b)
C ₁₀ H ₁₆ Cl ₆	10.2 ± 0.9	12	84	Fisk et al. (1998b)
C ₁₀ H ₁₆ Cl ₆	10.0 ± 0.9	12	84	Fisk et al. (1998b)
C ₁₀ H ₁₆ Cl ₆	20.4 ± 5.4	30	68	Fisk et al. (1998b)
C ₁₀ H _{15.3} Cl _{6.7} (low)	43.3 ± 5.7	124	35	Fisk et al. (2000)
C ₁₀ H _{15.3} Cl _{6.7} (high)	25.7 ± 1.7	41	63	Fisk et al. (2000)
C ₁₀ H ₁₅ Cl ₇	14.7 ± 1.6	19	79	Fisk et al. (1998b)
C ₁₀ H ₁₅ Cl ₇	8.6 ± 0.5	10	88	Fisk et al. (1998b)
C ₁₀ H ₁₄ Cl ₈	30.1 ± 5.2	50	61	Fisk et al. (1998b)
C ₁₀ H ₁₄ Cl ₈	13.9 ± 1.1	17	82	Fisk et al. (1998b)
C ₁₁ H ₂₀ Cl ₄	10.8 ± 1.0	14	80	Fisk et al. (1998b)
C ₁₁ H ₁₉ Cl ₅	9.0 ± 1.4	11	86	Fisk et al. (1998b)
C ₁₁ H ₁₈ Cl ₆	16.9 ± 2.5	22	76	Fisk et al. (1998b)
C ₁₁ H ₁₆ Cl ₈	36.5 ± 9.6	69	53	Fisk et al. (1998b)
C ₁₂ H _{19.3} Cl _{6.7} (low)	38.5 ± 4.3	81	48	Fisk et al. (1996)
C ₁₂ H _{19.3} Cl _{6.7} (high)	77.0 ± 8.6	>1000	0	Fisk et al. (1996)
C ₁₂ H _{16.2} Cl _{9.8} (low)	86.6 ± 10.8	>1000	0	Fisk et al. (1996)
C ₁₂ H _{16.2} Cl _{9.8} (high)	77.0 ± 8.6	>1000	8	Fisk et al. (1996)

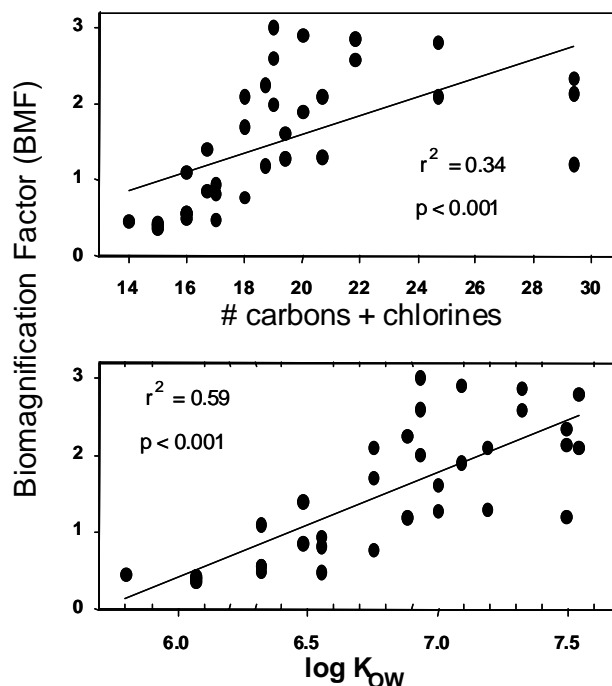
¹ Measured depuration rates (± standard error) converted to half-lives using the equation $t_{1/2}$ (days) = 0.693/rate (day⁻¹).

² Biotransformation rate = measured depuration rate – minimum depuration rate derived from a polynomial relationship of half-life with log K_{OW} developed by Fisk et al. (2000). Biotransformation rate converted to a half-life using the equation $t_{1/2}$ (days) = 0.693/rate (day⁻¹).

Table 9 compares the measured depuration half-lives and estimated biotransformation half-lives of C_{10–12} SCCPs. The similar depuration and biotransformation half-lives for some SCCPs, particularly the decanes, suggest that the depuration is primarily due to biotransformation.

Fisk et al. (2000) showed that BMFs for 35 chlorinated n-alkanes (combined data for SCCPs and MCCPs; some were single homologues and others were homologue mixtures of single carbon chains) were significantly related to the number of carbon plus chlorine atoms per compound and to log K_{OW} (Figure 2). Biotransformation half-lives in trout ranged from 8 to >1000 days (Table 9).

Figure 2: Relationship of BMF for SCCPs and MCCPs and carbon + chlorine number and log K_{OW} in rainbow trout for 35 chlorinated n-alkane single chain length products (from Fisk et al. 2000)



Bengtsson and Baumann-Ofstad (1982) studied the uptake and elimination of two commercial SCCPs — a 49% chlorine product and a 71% chlorine product — in bleak (*Alburnus alburnus*). The SCCPs were administered in food at concentrations ranging from 590 to 5800 µg/g food. There was a 91-day uptake period followed by a 316-day depuration period. Although the uptake efficiency of the 71% chlorine product was low (6%, or half that of the 49% chlorine product), a remarkably high retention was observed. This formulation remained in the fish tissues at a steady level until the experiment was terminated after the 316-day elimination period.

Fisk et al. (1998a) studied the bioaccumulation and depuration of sediment-sorbed ¹⁴C-labelled C₁₂H₂₀Cl₆ (56% chlorine by weight) and C₁₂H₁₆Cl₁₀ (69% chlorine by weight) by oligochaetes (*Lumbriculus variegatus*). The experiment consisted of a 14- or 21-day uptake period followed by a 42-day depuration period. Organic carbon normalized biota–sediment accumulation factors (BSAF) calculated from the rates of uptake and depuration ranged from 1.9 for C₁₂H₁₆Cl₁₀ to an average of 6.8 for C₁₂H₂₀Cl₆. Half-lives of the two SCCPs were similar, but uptake of the more highly chlorinated dodecane was significantly slower than that of the less chlorinated dodecane. There was evidence for biotransformation of the SCCPs by *Lumbriculus*; however, the degradation products may also have been formed in the sediments and accumulated by the oligochaetes.

Results on uptake of SCCPs by rainbow trout from food have been obtained as part of a toxicity investigation (Cooley et al., 2001). In the study, juvenile trout were exposed to one of four short-chain chlorinated paraffins daily via food for either 21 or 85 days. The food used in the test had a lipid content of 14%. The whole-fish tissue concentrations ranged from 0.1 to 5.5 mg/kg. Further experimental details of this study are report in Section 1.9.2. Fish from several of the high exposure concentrations fed erratically during the test which means that the actual relative exposure of these fish may be lower than indicated by the concentration in food.

2.3.2.3 Field studies of bioaccumulation and biomagnification

Bioaccumulation factors (BAFs) for SCCP chain length groups in western Lake Ontario lake trout (*Salvelinus namaycush*) were calculated based on concentrations in whole fish (wet weight or lipid weight) and dissolved water concentrations (Muir et al. 2001). BAFs ranged from 88 000 to 137 600 in lake trout on a lipid weight basis, or from 16 440 to 25 650 on a wet weight basis (Table 10). Chlorinated dodecanes (C₁₂) were the most prominent SCCPs in lake water and fish. The highest BAFs were for the chlorinated tridecanes (C₁₃) because of their low water concentrations. BMFs for the SCCPs based on an alewife (*Alosa pseudoharengus*)/smelt (*Osmerus mordax*) diet ranged from 0.33 to 0.94 and were highest for the tridecanes. The higher BMF for SCCPs in general was determined between lake trout and alewife alone (BMF 0.91) (Muir et al. 2003). These BMF values suggest that SCCPs, especially the chlorinated decanes and dodecanes, are not biomagnifying in the pelagic food web of Lake Ontario. By contrast, PCBs have been shown to biomagnify in the same food web. However, higher BMFs were determined between individual

forage species and lake trout (Table 11, Muir et al. 2003). Relatively high concentrations of SCCPs in sculpin (*Cottus cognatus*) and diporeia (*Diporeia* sp.) imply that sediments are an important source of SCCPs for bottom feeders (Muir et al. 2002).

Table 10: BAFs and BMFs for SCCPs in lake trout and forage fish from western Lake Ontario (from data of Muir et al. 2002)

Homologue	Concentration in water (ng/L) ¹	BAF _{ww} ² in lake trout	BAF _{lw} ³ in lake trout	BMF, lake trout to forage fish ⁴
C ₁₀	0.12	20 690	110 990	0.33
C ₁₁	0.36	25 650	137 590	0.51
C ₁₂	0.73	16 440	88 180	0.74
C ₁₃	0.07	19 950	107 000	0.94
ΣC ₁₀ –C ₁₃	1.27	17 210	105 160	0.58

¹ Water concentrations from large volume (90 L) of water samples collected in October 2000.

² Bioaccumulation factor (wet weight).

³ Bioaccumulation factor (lipid weight). Average lipid content of (whole) lake trout samples = 16% in samples from July 2001.

⁴ Assumes 50% alewife and 50% rainbow smelt diet. BMF = ng/g lw in whole trout ÷ ng/g lw in forage fish.

SCCPs have been found in all components of the food chain in Lake Ontario, including 2630 ng/g wet wt. in carp (refer to Table 18). SCCPs have also been found in high concentrations in marine mammals, including beluga (*Delphinapterus leucas*), ringed seal (*Phoca hispida*) and walrus (*Odobenus rosmarus*) from the Arctic (95-626ng/g wet wt.) (see Table 19).

Jansson et al. (1993) reported concentrations of SCCPs in rabbit, moose, reindeer and osprey from various regions in Sweden to be 2.9, 4.4, 0.14 and 0.53 µg/g lipid weight (lipid wt.), respectively. Thus, some herbivores had higher concentrations than a fish-eating bird (osprey).

Biomagnification of MCCPs and SCCPs in the Lake Ontario food web was examined (Table 11). MCCPs and C₁₄–C₁₇ chain length groups had very low BMFs between the three forage fish species and lake trout compared with the SCCPs. The SCCPs, by comparison, had much higher BMFs, especially for C₁₂ and C₁₃ SCCPs in the same food web (Table 11). With SCCPs, the alewife to lake trout BMF exceeded 1, as did the BMF for diporeia to sculpin.

Table 11: BMFs for MCCPs and SCCPs in the Lake Ontario food web (from Muir et al. 2003)

Group	Alewife–lake trout	Smelt–lake trout	Sculpin–lake trout	Diporeia–sculpin
SCCPs	0.91	0.43	0.27	1.4
C ₁₀	0.43	0.27	0.17	1.3
C ₁₁	0.94	0.35	0.22	1.6
C ₁₂	1.1	0.56	0.37	1.2
C ₁₃	1.5	0.68	0.35	1.1
MCCPs	0.09	0.05	0.05	5.2
C ₁₄	0.27	0.16	0.10	2.7
C ₁₅	0.04	0.02	0.03	9.56
C ₁₆	0.01	0.01	0.01	14.5
C ₁₇	0.00	0.00	0.00	11.5

¹ BMF = lipid weight concentration in predator / lipid weight concentration in prey.

Bioaccumulation factors were calculated in Lake Ontario for SCCP isomers that were detected in water and organism samples (Houde et al. 2008). Mean BAFs for SCCPs congeners of differing chlorine contents in lake trout from Lake Ontario varied between 4.0x10⁴ and 1.0x10⁷ for C₁₀, 1.3 x10⁴ and 3.2 x10⁶ for C₁₁, 1.0x10⁵ to 2.5x10⁵ for C₁₂, and 2.5x10⁵ for C₁₃ homologues. Total SCCPs had a BAF ranging from 2.7x10 to 4.1x10 (Table 12). The highest biomagnification factor was observed for sculpin-*Diporeia* in Lake Ontario and Lake Michigan (BMF = 3.6) (Table 13). The BMFs exceeded 1 for lake trout-rainbow smelt and Sculpin-*Diporeia* in both lakes, and exceed 1 for lake trout-alewife in Lake Ontario. Trophic magnification factors (TMFs) were reported to be 0.97 and 1.2 for Lake Ontario and Lake Michigan, respectively. A TMF above 1 indicates that certain SCCP isomers have the potential to biomagnify in aquatic food web (Houde et al. 2008).

Table 12: Lipid normalized bioaccumulation factors (BAFs) for SCCPs in the Lake Ontario in 2002 and 2004.

Lake Ontario	Water-plankton	Water-alewife	Water-sculpin	Water-rainbow smelt	Water-lake trout
C10	5.0x10 ⁵	2.5x10 ⁶	1.0x10 ⁷	5.0x10 ⁶	2.5x10 ⁶
C11	1.6x10 ⁵	4.0x10 ⁵	2.5x10 ⁶	1.6x10 ⁶	1.0x10 ⁶
C12	5.0x10 ⁴	1.0x10 ⁵	2.5x10 ⁵	1.6x10 ⁵	1.6x10 ⁵
C13	2.5 x10 ⁵	4.0 x10 ⁵	6.3 x10 ⁵	6.3 x10 ⁵	2.5x10 ⁵
SCCPs	2.5x10 ⁵	1.0x10 ⁶	4.0x10 ⁶	2.0x10 ⁶	1.3x10 ⁶

Table 13: Lipid normalized biomagnification factors (BMFs) for SCCPs in the Lake Ontario and Lake Michigan food webs in 2002 and 2004.

	Lake Ontario				Lake Michigan			
	Lake trout-Alewife	Lake trout-Rainbow smelt	Lake trout-Sculpin	Sculpin-Diporeia	Lake trout-Alewife	Lake trout-Rainbow smelt	Lake trout-Sculpin	Sculpin-Diporeia
C10	1.3±1.1	0.63±0.30	0.50±0.39	2.8±2.5	0.98±0.53	3.3±2.7	1.3±0.96	6.6±11
C11	1.6±2.7	0.84±0.74	0.44±0.16	4.7±5.9	0.60±0.34	2.4±1.8	0.80±0.40	2.8±1.8
C12	4.3±11	1.5±6.7	0.66±5.6	4.9±5.5	0.60±0.15	2.2±0.5	1.0±0.16	2.5±1.1
C13	3.5±4.0	1.5±1.6	0.58±0.65	1.8±0.55	0.47±0.020	1.1±0.064	0.70±0.03	1.4±0.61
SCCPs	2.5±2.7	1.04±0.81	0.54±0.24	3.6±4.2	0.68±0.37	2.4±1.8	0.97±0.55	3.6±5.9

2.3.2.4 Bioaccumulation Summary

BAFs for SCCP homologue groups in western Lake Ontario trout range from 16 440 to 25 650 wet wt. (Table 10). Very high bioconcentration factors (BCFs) have been measured for SCCPs in rainbow trout (*Oncorhynchus mykiss*) (up to 7816 wet wt.) (Madeley and Maddock 1983a,b) and in mussels (*Mytilus edulis*) (5785–138 000 wet wt.) (Madeley et al. 1983b, Madeley and Thompson 1983, Renberg et al. 1986), though BCFs less than 5000 have been observed for some organisms (algae) or congeners (C13 SCCPs and carp). Estimated and measured log K_{OW} values for SCCPs range from 4.39 to 8.69. Based on measured Log K_{OW} values in the range of 4.48 to 8.69 the Gobas model predicts a log BAF ranging from 3.47 to 6.2 with the large majority of the predictions having a BAF > 5000.

High SCCP concentrations in marine mammals (ringed seal, beluga whales and walrus) and carp (Tables 20 and 21) show that SCCPs are likely to bioaccumulate in aquatic biota. Bengtsson and Baumann-Ofstad (1982) found evidence of very high retention of a highly chlorinated (71% chlorine by weight) SCCP formulation in bleak during a 316-day elimination period of an uptake/elimination study.

Tomy (1997) found that SCCPs (around 60–70% chlorine by weight) were present at a concentration of 11–17 µg/kg lipid (mean concentration 13 µg/kg lipid) in human breast milk from Inuit women living on the Hudson Strait in northern Quebec, Canada. Similarly, SCCPs were found at concentrations of 4.5–110 µg/kg lipid in a study in the United Kingdom (Thomas and Jones 2002). These findings are indicative of bioaccumulation through the food chain, especially in northern Quebec, since food would be the major or only source of environmental exposure for the Inuit.

While biomagnifications factors (BMF) are not a criterion considered in the regulations for bioaccumulation, BMFs are important supplemental information. If a substance has a BMF greater than 1, it is likely to have high BCF/BAF values. SCCPs have been shown to have BMF values greater than 1 for some species in Lakes Ontario and Michigan (Muir et al. 2003, Houde et al. 2008) (Tables 11 and 13).

It is therefore concluded that short-chain chlorinated paraffins are bioaccumulative substances according to the criteria in Annex D of the Stockholm Convention.

2.3.3 Potential for Long Range Transport

SCCPs have been detected in air, sediment and mammals in the Arctic (see Section 2.4). Tomy (1997) and Bidleman et al. (2001) detected SCCPs at concentrations ranging from <1 to 8.5 pg/m³ in air collected in the high Arctic (Alert, Ellesmere Island) (see Section 2.4.1), while Borgen et al. (2000) measured SCCPs ranging from 9.0 – 57 pg/m³ at Mt. Zeppelin, Svalbard, Norway in 1999. SCCPs have also been measured in the sediments of remote Arctic lakes (Tomy et al. 1999, Stern and Evans 2003) far from local sources of contamination (see Section 2.4.5), as well as in Arctic biota such as ringed seal, beluga whale, walrus (Tomy et al. 2000), char and seabirds (Reth et al. 2006) (see Section 2.4.6). The concentration profiles for SCCPs in Arctic marine mammals show a predominance of the

shorter carbon chain length congeners, i.e., the C10 and C11 formula groups (Tomy et al. 2000), some of the more volatile components of SCCP mixtures (Drouillard et al. 1998a), suggesting that these compounds are more likely to be transported long distances. This is in agreement with the results of Reth et al. (2005, 2006), who found an enrichment of C10 SCCPs in biota in the North Sea compared to the Baltic Sea (Reth et al. 2005) and in the Arctic as compared to the Baltic Sea (Reth et al. 2006). Supporting this are modelled results indicating that the atmospheric half-lives for the major SCCP homologues observed in environmental samples, such as the Great Lakes and Arctic air and biota (C10H17Cl5, C10H16Cl6, C10H15Cl7, C11H18Cl6, C11H17Cl7, C12H20Cl6, C12H19Cl7), are greater than 2 days (Section 2.2.1). These data provide evidence of the long range transportation and deposition of these contaminants.

In addition, several modelling exercises have been undertaken. OECD (1999) performed fugacity modeling of SCCPs using a Level II fugacity model. Among their input data, they used a half-life in air of 7.2 days and a VP of 0.0213 Pa (at 40°C). In the 100% release to air and 100% release to water scenarios, 0.11% and 0.05%, respectively, of the SCCPs remained in air, and 0.02% and 1.16%, respectively, remained in water at steady state, indicating slight mobility of SCCPs in the environment.

Van de Meent et al. (2000) ran a global-scale multimedia model (similar to GloboPOPs) assuming degradation only in air, water and soil. This method indicates that SCCPs should partition to all three compartments, except for some highly chlorinated compounds such as C13H16Cl12.

The Arctic contamination potential (ACP) of SCCPs and other compounds has been determined. The ACP is the amount of chemical in Arctic surface media divided by the amount of chemical in the global environment estimated from the GloboPOPs model (Wania 2003, Wania and Mackay 2000). The ACP results were generated from the GloboPOPs model for a hypothetical series of chemicals of varying K_{OA} and K_{AW} (Wania 2003) assuming 10 years of emission into air, no atmospheric degradation and emission distribution by population density (refer to Canadian Annex E 2007 submission for diagram and more details). ACP values range from 0 to 5; based on this range, SCCPs have moderate ACP when released to air, similar to tetra- to heptachlorobiphenyls. Only chemicals such as chlorobenzenes and mono/di/trichlorobiphenyls with $\log K_{OA}$ of 3–7 and $\log K_{AW}$ of –1 to –2 have high (>3) ACP. Similar assessment using emissions to water or soil (in which the ACP values in the two-dimensional [K_{OA} vs. K_{AW}] ACP “space” are smaller than for air emissions) show that SCCPs have low ACP when emitted to water.

A comparison of vapour pressures (VPs) and Henry Law Constants (HLCs) demonstrate that SCCPs have VPs (2.8×10^{-7} to 0.028 Pa) and HLCs (0.68–18 Pa·m³/mol for C_{10–12} congeners) that are in the range of VPs and HLCs for some persistent organic pollutants that are known to undergo long-range atmospheric transport under the 1979 UNECE Convention on Long Range Transboundary Air Pollution (e.g., hexachlorocyclohexane [lindane], heptachlor, mirex).¹ In general, the HLC values reported imply that SCCPs can remobilise from water to air or from moist soils to air as a result of environmental partitioning.

Annex E (2007) information submitted by Switzerland for this risk profile outlines a recent study by Wegmann et al. (2007) which used OECD Pov and LRTP Screening Tool to examine the long range transport of several POPs candidate substances including SCCPs. The authors included Monte Carlo calculations to demonstrate the influence of uncertain chemical properties. Although there are considerable uncertainties in the chemical properties, the results indicated that SCCPs have Pov and LRTP properties similar to those of several known POPs.

The monitoring data, modelled data and comparative exercise support the conclusion that SCCPs undergo long range transport.

2.4 Exposure

2.4.1 Atmospheric concentrations

SCCPs were measured in four individual samples of air collected at Alert at the northern tip of Ellesmere Island in the high Arctic (Table 14). Concentrations ranged from non-detectable to 8.5 pg/m³ in gas-phase samples. Blank samples from the air sampling program at Alert also contained low levels of SCCPs, but at concentrations lower than those in air samples. Heptachloroundecane was the major component of the SCCPs, and undecanes (C₁₁) were the predominant alkane group. Tomy (1997) had previously analyzed air samples collected at Egbert, Ontario, in 1990 for SCCPs and found similar profiles, with chlorodecanes (C₁₀) and chloroundecanes (C₁₁) predominating. The higher SCCP concentrations at Egbert (Table 14) are consistent with its location near population centres of southern Ontario.

¹ The VP of lindane is 4.3×10^{-3} Pa (IPCS 1991), the VP of heptachlor is 3.0×10^{-6} Pa (IPCS 1984a) and the VP of mirex is 2.3×10^{-9} Pa (IPCS 1984b). The HLCs of lindane and heptachlor are 0.13 and 0.02 Pa·m³/mol, respectively.

Table 14: SCCP concentrations in Arctic and Ontario air (from Tomy 1997, Tomy et al. 1998a, Bidleman et al. 2001)

Location	Date	Total SCCP concentration (pg/m ³)
Alert, Nunavut ¹	September 14, 1992	5.7
Alert, Nunavut	September 21, 1992	non-detectable
Alert, Nunavut	September 28, 1992	2.1
Alert, Nunavut	December 28, 1992	8.5
Alert, Nunavut ²	July (weeks 25–28) 1994	4.39
Alert, Nunavut	August (weeks 29–32) 1994	7.25
Alert, Nunavut	September (weeks 33–36) 1994	6.14
Egbert, Ontario ³	May 2, 1990	65
Egbert, Ontario	May 26, 1990	500
Egbert, Ontario	June 19, 1990	525
Egbert, Ontario	July 13, 1990	924
Lake Ontario ^{4,5}	July 1999	249, 1510
Lake Ontario ⁵	October 2000	120, 430

¹ Alert data from Tomy (1997). Note: The blank values for the Alert data were about 1 pg/m³. The blank value was not subtracted from the sample values in the table.

² Bidleman et al. (2001).

³ Egbert data from Tomy et al. (1998a) and Tomy (1997).

⁴ Muir et al. (2001). Blank-subtracted results.

⁵ D.C.G. Muir, unpublished data, 2001. Blank-subtracted results from two high-volume air samples collected from the CSS *Limnos* on July 7–9, 1999, and October 30, 2000, over Lake Ontario

In the United Kingdom, Peters et al. (2000) reported a mean SCCP concentration of 99 pg/m³ in air collected from a semi rural site in Lancaster. In comparison to the Canadian values, SCCP concentrations in air were ranked such that Egbert > Lancaster > Alert and most likely reflect the proximity of Egbert and Lancaster to local source areas. A recent paper by Barber et al. (2005) found that concentrations in the U.K. atmosphere ranged between <185 to 3430 pg/m³ (mean of 1130 pg/m³) and were higher than 1997 levels at the same site. Barber et al. (2005) also calculated an average concentration of 600 pg/m³ of SCCPs for the UK atmosphere. The profiles of SCCPs at Alert resemble those measured in Lancaster but are quite different from those at Egbert and over western Lake Ontario. This is illustrated in Figures 3 and 4. In Figure 3, the individual “formula groups” for each carbon chain length within the SCCP mixture are plotted for Egbert and Lancaster (Tomy et al. 1998a, Peters et al. 2000). From this figure, it is clear that Egbert air samples have higher proportions of penta-, hexa- and heptachlorodecanes and undecanes compared with Lancaster, whereas Lancaster air has higher proportions of hexa-, hepta- and octadodecanes and tridecanes than Egbert air. Both locations are north of major urban areas (Lancaster is about 70 km north of the Manchester–Liverpool area, and Egbert is about 50 km north of Toronto), so the reasons for the differences in homologue patterns between Lancaster and Egbert are not clear.

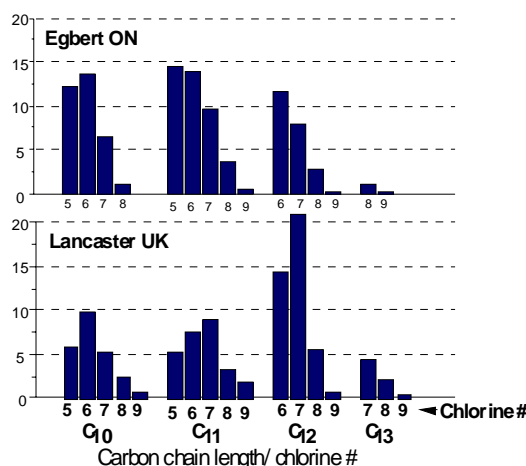
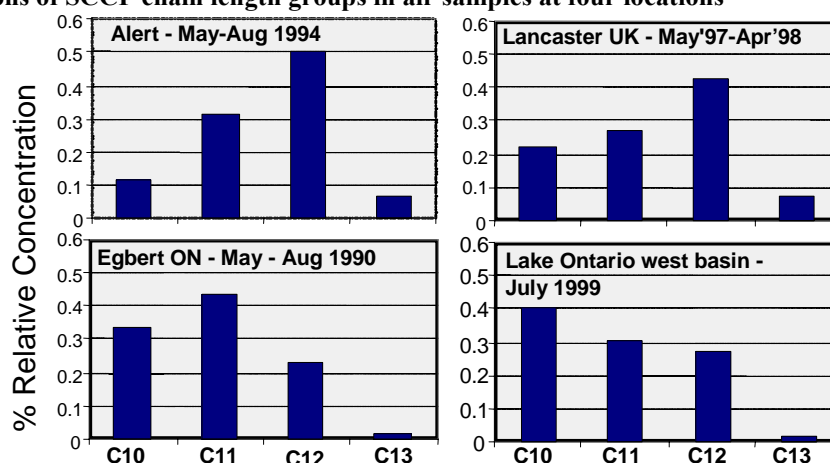
Figure 3: SCCP carbon chain length and formula group abundance (% contribution) in air samples from Egbert, Ontario, in 1990 (Tomy et al. 1998a) and Lancaster, United Kingdom, in 1997–98 (from Peters et al. 2000)

Figure 4: Proportions of SCCP chain length groups in air samples at four locations

In Figure 4, the proportions of SCCP chain length groups are compared at four locations. It can be seen that the profile of SCCP components in Egbert air in summer 1990 resembles that of air over western Lake Ontario collected in July 1999 (Muir et al. 2001), in that both have high proportions of chlorododecanes and chloroundecanes. The profile in Alert air resembles that at Lancaster but differs from that of the southern Ontario sites by having higher proportions of chlorododecanes. These differences appear to be consistent over time. Analysis of air samples collected in October 2000 over western Lake Ontario showed the same profile as in July (i.e., higher chlorododecanes and chloroundecanes), despite much lower air temperatures (D.C.G. Muir, unpublished data). The reasons for these differences are not clear. The Alert site is very remote from urban areas and receives air predominantly from sources in Europe and Asia because of its unique geographic location. However, this air flow from Europe/Asia is more pronounced in winter than in summer months (Halsall et al. 1998). The three other sites are outside of, but within 100 km of, urban areas.

Borgen et al. (2000) measured SCCPs in Arctic air samples taken at Mt. Zeppelin, Svalbard, Norway, during the period March to May 1999, using HR-ECNIMS. Concentrations ranging from 9.0 to 57 pg/m³ were found, which are higher than those found at Alert. Borgen et al. (2002) found much higher SCCP concentrations in air at Bear Island, a small isolated island between Svalbard and mainland Norway. Total SCCP concentrations (air + filter) ranged from 1800 to 10 600 pg/m³. These high air concentrations are unusual and may be related to the transport of persistent organic pollutants to terrestrial and freshwater environments on the island via seabird guano (Borgen et al. 2002).

SFT (2002) measured SCCP concentrations in 3 moss samples from Norway (Valvil, Molde, and Narbuvoll). Samples were taken in forest areas at a minimum distance of 300m from roads and buildings and 10 km from towns. Concentrations of 3 – 100 µg/kg wet weight were measured, suggesting the presence of SCCPs in the atmosphere.

2.4.2 Wastewater treatment effluents

SCCPs were detected in all eight sewage treatment plant final effluents from southern Ontario (Canada) at ng/L concentrations. The highest concentrations were found in samples from treatment plants in industrialized areas, including Hamilton, St. Catharines and Galt, compared with lower concentrations in samples from treatment plants in non-industrial towns, such as Niagara-on-the-Lake and Niagara Falls (Table 15).

Table 15: SCCP concentrations in final effluent of sewage treatment plants in southern Ontario based on samples collected in 1996 (from Muir et al. 2001)

Sewage treatment plant	Concentration (ng/L)				
	C ₁₀	C ₁₁	C ₁₂	C ₁₃	Total
Woodward Avenue, Hamilton, Ontario	128	155	153	11.5	448
Halton Skyway, Burlington, Ontario	38	19	12	<1	69
Stanford, Niagara Falls, Ontario	11	34	36	1	82
Port Dalhousie, St. Catharines, Ontario	19	39	47	5	110
Port Weller, St. Catharines, Ontario	22	27	28	4	81
Niagara-on-the-Lake, Ontario	13	18	27	1	59
Galt, Ontario	82	85	86	11	264
Guelph, Ontario	23	32	34	4	93

¹ Total concentrations (dissolved + particulate).

Reiger and Ballschmiter (1995) reported C₁₀₋₁₃, 62% chlorine SCCP concentrations of 80 ± 12 ng/L in water upstream and 73 ± 10 ng/L in water downstream of a sewage treatment plant in Germany. The concentration of SCCPs in the effluents was 115 ng/L. In the United States, Murray et al. (1988) reported C₁₀₋₁₃, 60% chlorine SCCP concentrations of <150–3300 ng/L in water from an impoundment drainage ditch that received effluent from a CPs production plant in Dover, Ohio.

Iino et al. (2005) measured concentrations of SCCPs in influent and effluent of three MWTPs in Japan in 2002. Concentrations of SCCPs in influent and effluent ranged from 220 to 360 ng/L and 16 to 35 ng/L, respectively. All three influent samples contained C10-C13 with five to eight chlorines for each homologue. None of the three effluent samples showed any detected levels of C12 and C13 homologues suggesting that sewage sludge treatment may be capable of removing C12 and C13 homologues.

2.4.3 Sewage sludge and soils

Nicholls et al. (2001) found total CP (SCCP + MCCP) concentrations in digested sewage sludge ranging from 1.8 to 93.1 µg/g dry wt. Nicholls et al. (2001) did not detect CPs in farm soils on which sewage sludge had been applied (<0.1 µg/g); however, SCCPs/MCCPs were detected in earthworms (<0.1–1.7 µg/g wet wt.) from four of nine farms receiving urban sewage sludge.

Stevens et al. (2002) found SCCP concentrations ranging from 6.9 to 200 µg/g dry wt. in sewage sludge from 14 WWTPs in the United Kingdom. They found that SCCP concentrations were not related to the population equivalent (wastewater volume/population) of WWTPs, whereas other contaminants, such as PCBs, were strongly correlated. However, highest concentrations of SCCPs were in sludge from industrial catchments. A rural catchment with zero industrial effluent had significant levels (590 µg/g) of total SCCPs/MCCPs in sludge (Stevens et al. 2002).

In comparison to river and lake sediments, concentrations of SCCPs/MCCPs in sewage sludge are much higher, especially from WWTPs serving industrial areas (Table 16). The fate of CPs in biosolids that are then applied to farmland, a common practice in Europe and a growing practice in North America, has not been thoroughly investigated. The work of Stevens et al. (2002) and Nicholls et al. (2001) points to agricultural soils as potentially a major reservoir of CPs (SCCPs and MCCPs) due to sewage sludge application. This source needs further investigation to determine if it is short term due to rapid biodegradation or represents a long-term reservoir for further environmental distribution.

The worst-case SCCP concentration in sewage sludge-amended soils is estimated as follows. According to Ontario Ministry of Environment guidelines (MOE 1998), the maximum allowable rate for sewage biosolid application to agricultural lands is 8 tonnes of solids per hectare per 5 years. Assuming that the biosolids are incorporated into the top 20 cm of the soil and assuming a standard dry soil bulk density of 2500 kg/m³ (EU 2003), the soil mass is 5000 tonnes/ha. Using an SCCP concentration in sewage sludge of 200 mg/kg dry wt. and assuming that SCCP-containing sludge is applied to the land for 10 years and that no or little biodegradation of the SCCPs occurs, this would result in a soil concentration of 3.2 mg/kg dry wt., calculated as follows:

Soil SCCP concentration after 10 years of sludge application

$$= \frac{200 \text{ mg/kg dry wt.} \times 8 \text{ tonnes/ha} \times 10}{5000 \text{ tonnes/ha}}$$

$$= 3.2 \text{ mg/kg dry wt. soil}$$

Table 16: Concentrations of MCCPs in sewage sludges

CP	Location	Country	Concentration (µg/g dry weight)	Reference
C ₁₄₋₁₈ , 52% Cl	Zürich, urban area	Switzerland	30	Schmid and Müller (1985)
C ₁₄₋₁₇ , 52% Cl	Manufacturing site	Ohio, U.S.A.	0.76–50	Murray et al. (1988)
MCCP	Urban and rural WWTPs	U.K.	1800 (30–9700)	Stevens et al. (2002)
SCCP/MCCP	Urban and rural WWTPs	U.K.	1.8–93.1	Nicholls et al. (2001)

SCCPs have been measured in soils. An average SCCP concentration of 60.4 ± 54.9 ng/g dry wt was determined for soils taken near the landfill site in Iqaluit, Nunavut, in the Canadian Arctic (Dick et al. 2010, as provided in the CPIA Annex E 2010 submission). The results indicate that the landfill may act as a local source of SCCP contamination in Iqaluit. In addition, chlorinated paraffins (sum of SCCPs, MCCPs and LCCPs) have been measured in humus samples from alpine regions in Europe with concentrations ranging from 7–199 ng/g dry wt (Iozza et al. 2009, as provided in the IPEN Annex E 2010 submission).

2.4.4 Surface waters

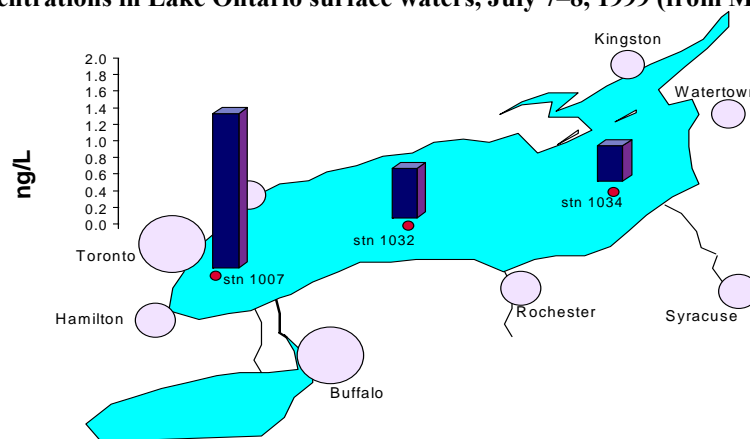
SCCPs and PCBs were measured in large-volume water samples (100 L) collected in mid-Lake Ontario in July 1999 and October 2000 (Table 17; Figure 5). In 1999, the highest dissolved SCCP concentration (1.8 ng/L) was observed in the western basin and probably resulted from sewage treatment plant sources from large urbanized areas, such as Toronto and Hamilton. A similar gradient was observed in October 2000; however, concentrations were lower. For comparison, the concentration of the PCBs in the samples are also provided in Table 17.

Table 17: Concentrations of SCCPs and PCBs in filtered surface waters from western Lake Ontario (from Muir et al. 2001)

Carbon chain group	Concentrations (pg/L)	
	July 1999 ¹	October 2000
C ₁₀	168	74
C ₁₁	490	217
C ₁₂	1000	441
C ₁₃	94	42
ΣSCCP	1750	774
ΣPCBs	620	730

¹ Average of duplicate samples of water at site 1007 in the west basin of Lake Ontario.

Figure 5: SCCP concentrations in Lake Ontario surface waters, July 7–8, 1999 (from Muir et al. 2001)*



* Dots under each bar indicate approximate collection location. Circles indicate relative population sizes of the indicated cities.

The average concentration of total SCCPs in water from Lake Ontario (4 meter depths) between 2000 and 2004 was 1.194 ng/L. The concentrations ranged from 0.770 to 1.935 ng/L, 1.039 to 1.488 ng/L and 0.606 to 1.711 ng/L for 2000, 2002 and 2004, respectively (Houde et al. 2008).

SCCP concentrations of 30 ± 14 ng/L were measured in the Red River in Selkirk, Manitoba, over a 6-month period in 1995 (Tomy 1997). Tomy et al. (1999) attributed the SCCPs in the water to a local source, possibly a metal machining/recycling plant in the town of Selkirk, as suggested by the similarity of the formula group abundance profile to that of the PCA-60 standard used in the study.

SCCPs have been detected in river waters in a range of 300 (LOD) to 1100 ng/L in Spain (Castells et al. 2003; 2004), 15.74 to 59.57 ng/L in the St. Lawrence River, Canada (Moore et al. 2003), < 100 to 1700 ng/L in England and Wales (Nicholls et al. 2001) and 7.6 to 220 ng/L in Japan. (Iino et al. 2005; Takasuga et al. 2003).

SCCPs have also been detected in streams, rivers and lakes in Iqaluit, Nunavut in the Canadian Arctic (Dick et al. 2010, as provided in the CPIA Annex E 2010 submission). Depending on the location, concentrations ranged from below detection to 117 ng/L. The results suggest that local sources of SCCP contamination exist in Iqaluit in addition to atmospheric transport (Dick et al. 2010).

The Ministry of the Environment (2006) in Japan, monitored SCCPs in 6 surface water samples from across the country and did not find any concentrations above the detection limits (which varied from 0.0055 to 0.023 µg/L between chain lengths). A surface water sample was taken in each of four different rivers in Japan in 2002 (Iino et al. 2005). SCCPs concentrations ranged from 7.6 to 31 ng/L.

2.4.5 Sediments

Tomy et al. (1997) measured SCCPs at concentrations around 245 µg/kg dry weight in sediment from the mouth of the Detroit River at Lake Erie and Middle Sister Island in western Lake Erie (sampled August 1995).

SCCPs were detected in all surface sediment samples from harbour areas along Lake Ontario at concentrations ranging from 5.9 to 290 ng/g dry wt. (Table 18). The highest concentrations were found at the most industrialized site (Windermere Basin, Hamilton Harbour), which has well-documented heavy metal, PAH and PCB contamination. Surface sediments from cores at more remote lakes in northern Ontario, Manitoba, and the Canadian Arctic had concentrations ranging from 1.6 to 257 ng/g dry wt. (see Table 19 below).

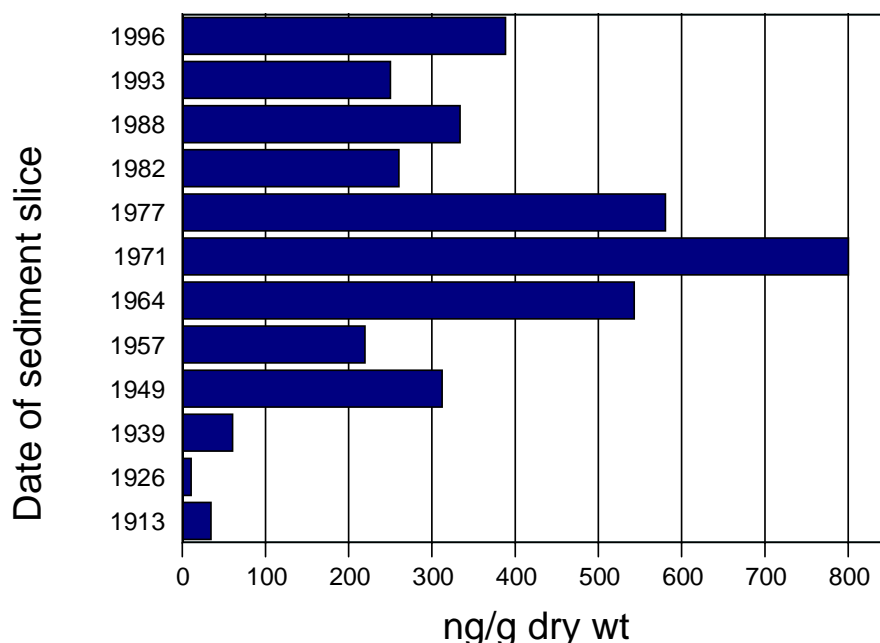
Table 18: SCCP concentrations in surface sediment grab samples collected in 1996 from Lake Ontario harbours (from Muir et al. 2001)

Location	Concentration (ng/g dry wt.)				
	C ₁₀	C ₁₁	C ₁₂	C ₁₃	Total C ₁₀₋₁₃
Toronto Harbour: inner harbour	0.7	4.7	10	12	27
Toronto Harbour: inner harbour duplicate	0.6	4.2	9.1	10.3	24
Toronto Harbour: Humber River mouth	0.9	2.5	2.2	0.3	5.9
Port Credit Harbour	1.4	2.2	3.3	0.4	7.3
Hamilton Harbour, site 1: west harbour	3.3	14	17	7.0	41
Hamilton Harbour, site 1: west harbour duplicate	2.2	9.1	11	4.6	27
Hamilton Harbour, site 2: Windermere Basin	11	56	127	90	290
Hamilton Harbour, site 3: Skyway WWTP discharge	2.9	16	41	21	81

The highest concentration of SCCPs in Lake Ontario sediments measured by Marvin et al. (2003) was 410 ng/g dry wt. from the Niagara basin, in an industrialized area. SCCPs were detected in all 26 samples from Lake Ontario, and the average SCCP concentration was 49 ng/g dry wt., which is much higher than sediment concentrations reported for lakes influenced primarily by atmospheric sources (see Table 19 below). Marvin et al. (2003) found that sediment samples from industrialized areas had higher proportions (17–44%) of the longer C₁₃ SCCPs compared with urban non-industrial areas or lakes affected mainly by atmospheric deposition.

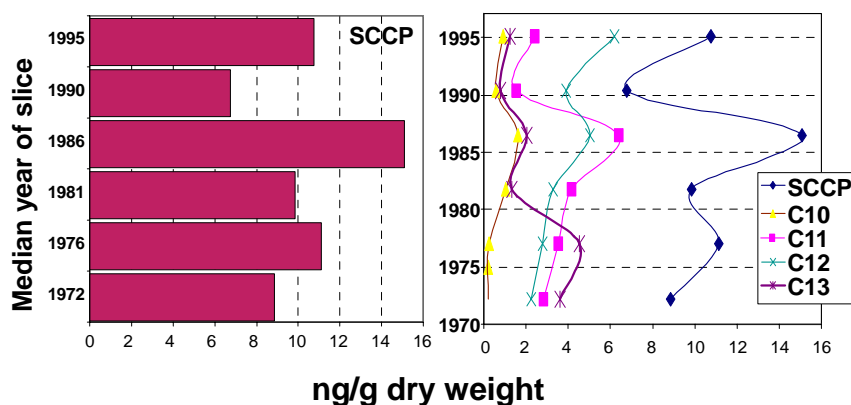
The historical concentration profile of SCCPs in Lake Ontario is shown in Figure 6. SCCP residues are found dating back to 1913, with the maximum residues (800 ng/g dry wt.) occurring in the 1970s. SCCP residues observed in 1996 in the surface sediment layer are approximately 390 ng/g dry wt. SCCPs in pre-industrial sediment slices were attributed to sample contamination due to an artefact of sampling (personal communication, D. Muir, Environment Canada, 2000).

Figure 6: Total C10–13 SCCP concentrations in a dated sediment core from Lake Ontario, west basin, Station 80, sampled in June 1998 (from Muir et al. 1999a)



SCCPs were determined in a dated sediment core collected in Lake St. Francis (Lac Saint-François) (Muir et al. 2002), downstream of Cornwall, Ontario in 1996 (there is a CPs manufacturing plant at Cornwall – CPs are currently not manufactured at this site). The historical profiles of SCCPs in the core are shown in Figure 7. The results show the presence of relatively low levels of SCCPs compared with Lake Ontario (Muir et al. 2002). The highest SCCP concentrations were found in slice 6, which has a median date of 1985 ± 4 years (Turner 1996). The predominant chain length groups in sediments were C₁₁ and C₁₂. These were present at almost equal proportions of total SCCP.

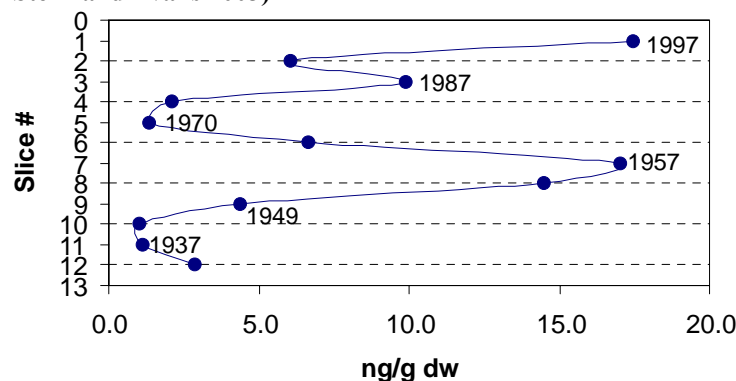
Figure 7: Profiles of SCCPs in a sediment core from Lake St. Francis in the St. Lawrence River downstream of Cornwall, Ontario* (Muir et al. 2002)



* Left panel shows concentrations of total SCCPs, and right panel presents results for chain length groups.

SCCPs have also been detected in sediments from Hazen Lake (northern Ellesmere Island) and Yaya Lake (Mackenzie River delta, Northwest Territories) in the Canadian Arctic (Tomy et al. 1999) and recently in sediments from Lake DV09, a very remote lake on Devon Island, Nunavut, at low ng/g levels (Stern and Evans 2003; Figure 8). Results for Hazen Lake (Tomy et al. 1999) are close to detection limits; however, the analysis also showed that SCCPs are readily detectable above a method detection limit based on sediment samples pre-dating 1900 that were used as blanks.

Figure 8: Concentration profile of SCCPs in a dated, laminated core from Lake DV09 on Devon Island, Nunavut, Canada (from Stern and Evans 2003)



The sediment core from Lake DV09, Devon Island, Nunavut (Figure 8), is laminated (or varved) and therefore has much less mixing than the cores analyzed by Tomy et al. (1999); hence, the SCCP profile is sharper than reported in Tomy et al. (1999). A full geochemical description of this core is given by Lockhart et al. (2000). SCCP concentrations in DV09 surface sediments were higher than in Hazen and Yaya lakes. However, the SCCP stratigraphy in DV09 shows the same trends as for Hazen and Yaya lakes.

Lake DV09 is remote from local human activities. The detection of SCCPs in sediments from this lake, suggests that these compounds reached the lake via atmospheric deposition (Stern and Evans 2003). The downcore profile generally follows the pattern of historical usage of SCCPs (with some delay in movement to the Arctic), according to Stern and Evans (2003). While the authors did not report the use of sediment cores proceeding the manufacture of SCCPs as a method blank, SCCP concentrations were generally well above the concentrations noted at the lowest depths (circa 1931) (Stern and Evans 2003), which just preceded the first large volume use of SCCPs in 1932 (Howard et al. 1975, in Stern and Evans 2003). SCCPs in the 1931 sediments may be trace background contamination from sampling or analytical equipment and supplies (Tomy et al. 1998a), or an artefact of core sampling, as noted with other studies (personal communication, D. Muir, Environment Canada, 2000). Although various dating methods were used to date the sediments (Lockhart et al. 2000), error in estimated date of the 1931 sediments could also explain the SCCP concentrations observed. The detection of SCCPs at this remote site and also in other remote Arctic lakes, especially Yaya Lake in the Mackenzie River delta, where levels of SCCPs were also well above background, illustrates the wide dispersal of SCCPs.

Sediments from streams, rivers and lakes in Iqaluit, Nunavut, in the Canadian Arctic had measurable SCCP concentrations ranging from 5.2 to 138.1 ng/g dry wt (Dick et al. 2010, as provided in the CPIA Annex E 2010 submission). However, the results from this study suggest that local sources of contamination exist in Iqaluit in addition to atmospheric transport of SCCPs (Dick et al. 2010).

Fluxes ($\mu\text{g}/\text{m}^2$ per year) of SCCPs to various Canadian lake sediments are plotted in Figure 9. These fluxes are for surface slices of each sediment core and represent relatively recent inputs. The highest fluxes are observed in lake sediments near urban areas (western Lake Ontario and the south basin of Lake Winnipeg). The lowest fluxes are observed in more remote lakes, including Lake Superior, which are influenced mainly by atmospheric inputs. These results suggest that the most elevated SCCP residues observed in lake sediments are mainly derived from urban areas. The sources contributing to SCCP residues observed in Fox Lake, Yukon, are of uncertain origin. Elevated fluxes of SCCPs to these lakes (Table 19) are higher than for PCBs, which range from about 0.1 to 0.5 $\mu\text{g}/\text{m}^2$ per year at these latitudes (Muir et al. 1996).

Figure 9: Fluxes ($\mu\text{g}/\text{m}^2$ per year) of SCCPs to lake sediments in Canada (from Muir et al. 1999a, Tomy et al. 1999)

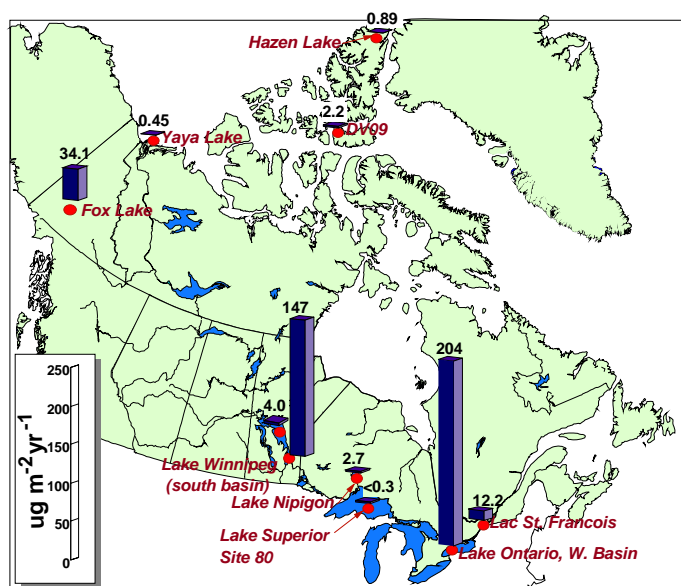


Table 19: Locations, concentrations and fluxes of SCCPs in Canadian lake sediment cores (Tomy et al. 1999, Stern and Evans 2003)

Lake	Latitude/longitude	Surface concentration (ng/g dry wt.)	Sedimentation rate (g/m^2 per year)	Focusing factor	Flux ($\mu\text{g}/\text{m}^2$ per year)
Yaya	69°10'N, 134°39'W	1.6	476	1.6	0.454
DV09	75°34'N, 89°19'W	17.6	304	2.4	2.21
Hazen	81°45'N, 71°30'W	4.5	278	4.5	0.893
S. Winnipeg ¹	50°23'N, 96°22'W	176	1000	1.2	147
N. Winnipeg ²	52°28'N, 98°20'W	8	645	1.3	4.00
Nipigon	49°25'N, 85°30'W	18	411	2.8	2.66
Fox	61°14'N, 135°28'W	257	126	0.9	341

¹ South basin of Lake Winnipeg.

² North basin of Lake Winnipeg.

Ballschmiter (1994) found SCCPs in sediments in Germany at concentrations ranging from <5 to 83 $\mu\text{g}/\text{kg}$ dry wt. The 83 $\mu\text{g}/\text{kg}$ dry wt. sample was from the Rhine River. The method of analysis was GC-MS using NCI and is reasonably specific for SCCPs (U.K. Environment Agency 2003a).

A dated sediment core from Lake Thun covering the last 120 years (1899 to 2004) was analyzed to obtain an overview of the historical trend of chlorinated paraffins (Iozza et al. 2008). A 5 ng/g dry weight concentration was found for the deeper, pre-industrial, sediment slice. The concentrations rose slowly from the 1950s to the 1970s, increased from the 1980s, and became more or less stable from the 1990s to present. The highest concentration of 58 ng/g dry weight was measured in 2000. The authors demonstrated that the chlorinated paraffin concentrations were consistent with global production. SCCPs and MCCPs were detected in all analyzed sediment slices. Comparison of the time profiles of total CPs, SCCPs, and MCCPs revealed that the rapid increase of total CP concentrations in the 1980s was mainly caused by SCCPs. The maximum SCCP concentration in sediment was 33 ng/g dry weight in 1986.

Concentrations of SCCPs ranging from 1,250 to 2,090 ng/g dry weight were measured in marine sediment samples collected from the coastal area of Barcelona (Spain) and near of a submarine emissary coming from a waste water treatment plant located at the mouth of the Besòs River (Barcelona) (Castells et al. 2008). SCCPs were detected in the Besòs River in Spain (250 to 3,040 ng/g dry weight) (Parera et al. 2004), and in various rivers in Germany, France and Norway (21 to 75 ng/g dry weight) (Hütting 2006).

A recent study of SCCPs and MCCPs in the U.K. environment included 20 aquatic and eight agricultural sites (Nicholls et al. 2001). Nicholls et al. (2001) selected surface sediments from three locations, ranging from 1 to 100 m, from 200 to 300 m and from 1–2 km downstream of municipal sewage treatment effluents. Analysis was by GC-LRMS in negative ion mode with ion trap set at low voltage to reduce fragmentation of the SCCPs/MCCPs. Both SCCPs and

MCCPs were judged to be widely distributed in the U.K. environment. At sites where the concentration of SCCPs alone was determined, concentrations ranged from 0.6 to 10.3 mg/kg dry wt. (Nicholls et al. 2001).

Příbylová et al. (2006) reported concentrations of SCCPs in 36 sediment samples from 11 Czech rivers and five drainage vents near industrial areas. The concentrations ranged from non-detect to 347.4 ng/g dry weight. Sediment samples from the Czech Republic were analyzed for SCCPs in three locations having different industrial discharges during 2001 and 2002. SCCPs sediment concentrations varied between 24 and 45.78 ng/g dry weight in the Košetice area (reference area), 16.30 to 180.75 ng/g dry weight in the Zlín area (rubber, tanning and textile industries) and 4.58 to 21.57 ng/g dry weight in the Beroun area (cements and machinery industry) (Štejnárova et al. 2005). The Ministry of the Environment (2006) in Japan, has monitored SCCPs in 6 bottom sediment samples from across the country and did not find any concentrations above the detection limits (which varied from 0.34 to 3.0 ng/g among carbon lengths).

Six sediment samples were taken in two rivers in Japan in 2003 (Iino et al. 2005). Concentrations of SCCPs ranged from 4.9 to 484.4 ng/g dry weight. Five of the six samples had concentrations greater than 196.6 ng/g dry weight.

2.4.6 Biota

2.4.6.1 Freshwater organisms

SCCPs were analyzed in lake trout and forage fish samples collected in Lake Ontario in 2001 (Muir et al. 2002) and in lake trout samples collected in 1996 (Muir et al. 2001). C₁₂ SCCPs predominated in lake trout, whereas C₁₁ was the major SCCP in sculpin and smelt (Table 20). SCCP concentrations (lipid wt.) were about 2-fold lower in the samples from 2001 compared with concentrations in lake trout of similar age collected in 1996 (Muir et al. 2001). Further studies with additional sampling times are needed to determine if this represents a consistent temporal trend.

Table 20: Concentrations of SCCPs in lake trout, carp and food web samples from Lake Ontario, including comparison with results from samples collected in 1996 (from Muir et al. 2001, 2002)

Species	Year		N	% lipid	Concentrations (ng/g wet wt.)				
					SCCPs	C ₁₀	C ₁₁	C ₁₂	C ₁₃
Lake trout	2001	mean	6	16	21.9	2.5	9.2	11.9	1.4
		sd ¹		3	22.9	2.3	8.4	10.9	1.3
Rainbow smelt	2001	mean	2	5	21.5	3.24	9.27	8.18	0.81
Slimy sculpin	2001	mean	2	5	27.0	4.04	11.99	9.75	1.18
Alewife	2001	mean	2	3	7.01	1.14	2.53	3.04	0.30
<i>Diporeia</i>	2001		1	2.9	10.6	1.78	4.14	4.09	0.55
Lake trout (Niagara-on-the Lake)	1996	mean	5	21	58.8	3.1	16.4	30.1	9.3
		sd		5	50.8	2.6	14.2	26.0	8.0
Lake trout (Port Credit, Mississauga)	1996	mean	5	26	67.8	3.5	18.9	34.7	10.7
		sd		4	52.5	2.7	14.6	26.9	8.3
Carp (Hamilton Harbour)	1996	mean	3	–	2630	14.2	355	1090	1170

¹ sd = standard deviation.

Carp (*Cyprinus carpio*) collected from Hamilton Harbour and lake trout (*Salvelinus namaycush*) collected from two locations in western Lake Ontario (Port Credit [northwest] and Niagara-on-the-Lake [southwest]) in 1996 were analyzed for SCCPs. SCCPs were detected in all samples of carp and lake trout from Lake Ontario (Table 20). The higher concentrations observed in carp are probably due to higher exposure of fish to SCCPs in Hamilton Harbour. Table 18 (above) shows a higher concentration of SCCPs in sediment at Hamilton Harbour than was found at Port Credit Harbour, which would imply that water concentrations of SCCPs in Hamilton Harbour may also have been higher, and/or carp (being bottom feeders) could have been exposed to higher sediment concentrations in Hamilton Harbour. Alternatively, the higher concentrations of SCCPs in carp from Hamilton Harbour could simply be as a result of bioconcentration. Reth et al. (2005) measured SCCP concentrations ranging between 19 and 286 ng/g wet weight in fish liver (North Sea dab, cod, and flounder) from the North and Baltic Seas.

SCCPs were also detected in fish and invertebrates in the Great Lakes and mean levels ranged from 130 to 500 ng/g wet wt. (Muir et al. 2003). Levels of SCCPs ranged from 118 to 1250 ng/g wet wt. in carp and 447 to 5333 ng/g in trout from Lake Ontario (Bennie et al. 2000).

Houde et al. (2008) investigated concentrations in water and biota in samples collected between 1999 and 2004 from Lake Ontario and northern Lake Michigan (Table 21). Net plankton (primarily zooplankton >110 µm), *Mysis* (*Mysis relicta*), slimy sculpin (*Cottus cognatus*), alewife (*Alosa pseudoharengus*), rainbow smelt (*Osmerus mordax*), and lake trout (*Salvelinus namaycush*) were collected in northern Lake Michigan (near Charlevoix, MI) and in western

Lake Ontario in July 2001. Additional plankton samples were collected in Lake Ontario in July 2004. *Diporeia* were collected in northern Lake Michigan in 1999 (near Manistique, MI) and in Lake Ontario in July 2001 from surficial sediments using an epibenthic sled.

Table 21. Concentrations of SCCPs in food web samples from Lake Ontario and Lake Michigan from samples collected between 1999 and 2004 (Houde et al. 2008).

Collected between 1999 and 2004 (Houde et al. 2006).						
Species	n	Concentrations (ng/g wet wt.)				
		C10	C11	C12	C13	SCCPs
Lake Michigan						
Plankton	3	4.9	8.6	8.3	1.3	23
Diporeia	3	12	5.5	5.7	0.26	24
Mysis	3	2.9	2.4	2.1	0.20	7.5
Alewife	2	9.7	12	15	0.62	37
Rainbow smelt	2	5.4	5.6	5.2	0.36	17
Sculpin	2	24	23	21	1.1	69
Lake Trout	7	51	30	40	1.6	123
Lake Ontario						
Plankton	3	0.21	0.42	0.34	0.05	1.02
Diporeia	3	0.99	2.3	2.3	0.31	5.9
Mysis	3	0.40	0.93	0.91	0.12	2.4
Alewife	2	0.90	1.7	1.9	0.17	4.6
Rainbow smelt	2	3	8.4	7.03	0.68	19
Sculpin	2	3.8	11	8.6	1.04	25
Lake Trout	7	3.4	13	16	1.9	34

SCCP predominated in organisms from Lake Michigan with the highest mean concentrations found in lake trout [*Salvelinus namaycush*, 123 ng/g wet weight (w.w.)]. Concentration of SCCPs throughout the food web was higher in Lake Michigan and Lake Ontario. The concentration of SCCPs ranged from 7.5 to 123 and 1.02 to 34 ng/g wet wt. for Lake Michigan and Lake Ontario, respectively. The lowest mean concentration of the SCCPs (7.5 and 2.4 ng/g w.w.) was detected in *Mysis* in Lake Michigan and Lake Ontario, respectively.

Further studies on concentrations of SCCPs in lake trout from Lake Ontario were conducted by Ismail et al. (2009) (CPIA Annex E 2010 submission). Time trends in tissue concentrations were determined from 1979-2004 using stored whole fish homogenate from a long-term monitoring program. Concentrations of SCCPs ranged from 17 to 91 ng/g wet wt. Concentrations showed an increasing but non-significant trend from 1979 to 1988 followed by a significant decrease to 2004 (Ismail et al. 2009). As noted by Ismail et al. (2009), these results are comparable to those of Houde et al. (2008).

Dick et al. (2010) (as provided with CPIA Annex E 2010 submission) measured SCCPs in anadromous and landlocked Arctic char (*Salvelinus alpinus*) and ninespine stickleback (*Pungitius pungitius*) from various locations near Iqaluit, Nunavut, Canada. SCCP concentrations ranged from non-detectable to 96 ng/g dry wt and 10.7 to 13.8 ng/g dry wt in Arctic char and stickleback, respectively.

SFT (2002) measured concentrations of SCCPs in blue mussel and cod livers from Norway. SCCPs were present in all samples with concentrations ranging from 14-130 µg/kg wet weight in mussel and 23-750 µg/kg in cod liver. The Ministry of the Environment (2006) in Japan, has monitored SCCPs in 6 aquatic wildlife samples from across the country and did not find any concentrations above the detection limits (which varied from 0.2 to 1.5 ng/g wet wt. among carbon lengths). Lahaniatis et al. (2000) reported mean values for individual chain length (C₁₀-C₁₃) SCCPs ranging between 6 – 135 µg/kg in fish (sprat, redfish, herring, halibut, sardine, and trout) from a variety of sites in England, Norway, Chile, Greece, Germany, Iceland, France, USA, and the North Sea.

2.4.6.2 Marine mammals

Tomy et al. (2000) reported levels of SCCPs in the blubber of ringed seal from Eureka, southwest Ellesmere Island, beluga whales from northwest Greenland, the Mackenzie Delta and the St. Lawrence River estuary, and walrus from northwest Greenland (Table 22). The data in Table 22 show that concentrations of SCCPs in the St. Lawrence beluga were approximately 4 times higher than concentrations in beluga from Greenland and the Mackenzie Delta. The elevated levels of SCCPs in belugas from the St. Lawrence River are consistent with the findings of elevated levels of other organochlorines by Muir et al. (1996), who suggested that this was a food chain effect attributed to local source

contamination. The data also show that in many cases SCCP concentrations in Arctic biota are lower than those of other persistent organochlorines, however a few samples showed higher SCCPs.

Table 22: Concentrations of C₁₀₋₁₃ SCCPs and other persistent organic pollutants in blubber of marine mammals from the Arctic and the St. Lawrence River estuary (from Tomy et al. 1998b, 2000)

Species	Location	Gender	Year	N	Lipid (%)	Concentrations ¹ (ng/g wet wt.)			
						ΣDDT	ΣPCB	Toxaphene	SCCPs
Beluga	St. Lawrence River estuary	M + F	1988	5	83.2 ± 7.3	79 800 ± 50 846	71 500 ± 35 200	–	785 ± 362
Beluga	Mackenzie Delta	M	1995	17	90.9 ± 1.1	3390 ± 1090	4670 ± 1470	5320 ± 1900	626 ± 499
Beluga	Sanikiluaq, Hudson Bay	M	1994	10	94.8 ± 0.7	14 740 ± 6850	7910 ± 1580	15 400 ± 8160	323 ± 76
Beluga	Pangnirtung, Cumberland Sound	M	1994	31	90.5 ± 3.2	4530 ± 1840	3770 ± 1390	9270 ± 2780	460 ± 306
Beluga	Kimmirut	M + F	1994	6	93.2 ± 1.6	5330 ± 3330	6016 ± 3220	11 820 ± 7970	142 ± 49
Beluga	NW Greenland (Sassat/Nuussuaq)	M + F	1989	4	88.3 ± 3.9	2220 ± 584	3640 ± 847	3050 ± 396	199 ± 64
Ringed seal	Pangnirtung, Cumberland Sound	M + F	1993	6	94.6 ± 1.2	855 ± 1122	780 ± 732	196 ± 81	95 ± 33
Ringed seal	SW Ellesmere Island, Eureka	M + F	1994	6	90.3 ± 1.8	660 ± 240	1170 ± 320	481 ± 89	526 ± 175
Walrus	NW Greenland	M	1978	2	83 ± 0.2	33 ± 9.2	160 ± 63	275 ± 74	426 ± 91

¹ Arithmetic means ± standard deviations.

Tomy et al. (2000) also observed that the concentration profiles for the Arctic marine mammals show a predominance of the shorter carbon chain length congeners (i.e., the C₁₀ and C₁₁ formula groups). This is significant, because Drouillard et al. (1998a) showed that these congeners are the more volatile components of SCCP mixtures, which show a trend of decreasing VPs with increasing carbon chain length and degree of chlorination. These results suggest that these compounds enter this region by long-range atmospheric transport. Tomy et al. (2000) concluded that “Although only a few samples have been analyzed in this study, it is clear that SCCPs are present in Arctic food webs and are being transported to these remote regions either in the atmosphere or ocean currents.” In contrast, the formula group abundance profile for the belugas from the St. Lawrence River estuary shows a shift towards the less volatile components — i.e., higher carbon chain length inherent to commercial formulations. The higher proportions of the less volatile components in the concentration profile suggest that local sources of SCCPs, possibly from the Great Lakes or the industrialized regions of the lower St. Lawrence River, are the most important sources of input of SCCPs to this area. SCCPs have also been detected at concentrations ranging from 100 to 770 ng/g wet weight in beluga and narwhal in Canada and Greenland (UNECE-LRTAP POPs Protocol, 2007)

There are few other published data on SCCPs in marine mammals for comparison, and differences in analytical methodology make comparisons problematic. Jansson et al. (1993) reported an SCCP concentration of 130 ng/g wet wt. in ringed seal blubber from Svalbard, which is quite similar to levels found by Tomy et al. (2000). It should be noted, however, that the substances measured in Jansson’s study were CPs of unspecified chain length with 6–16 chlorine atoms per molecule and so could have also included MCCPs and LCCPs. Ringed seals from Eureka had higher concentrations (mean 527 ng/g ww) than seals from Pangnirtung (mean for both sexes was 95 ng/g) (Muir et al. 1999b; Tomy et al. 2000). In beluga whale samples collected between 1987 and 1991, SCCPs ranged from 6620 to 85600 ng/g wet wt. in blubber and 544 to 38500 ng/g wet wt. in liver samples (Bennie et al. 2000).

SCCPs have also been measured in archived Beluga whale samples from Hendrickson Island and Pangnirtung in the Canadian Arctic (Canadian Department of Fisheries and Oceans report submitted with the CPIA Annex E 2010 submission). Samples of blubber (1993 to 2008 and 1982 to 2008 for Hendrickson Island and Pangnirtung, respectively) were analyzed for total SCCPs (note that samples from some years have yet to be analyzed). Blubber from 10-11 or 3-10 individuals from Hendrickson Island and Pangnirtung, respectively, were sampled for each year. The geometric mean SCCP concentration for whales from Hendrickson Island for samples analyzed thus far ranged from 3.96 to 340 ng/g lipid. Concentrations were relatively constant up until 2005 but were considerably lower in 2006 (3.96 ng/g lipid). Samples from 2007 and 2008 have yet to be analyzed. The geometric mean of the SCCP

concentration for whales from Pangnirtung ranged from 2.95-330 ng/g lipid, with the highest concentration recorded in 1992. Concentrations in 2005 and 2006 were 16.5 and 2.95 ng/g lipid, respectively. Concentrations for 2007 and 2008 have yet to be analyzed. Analysis of the 2007 and 2008 samples for both locations is required to determine if the observed decrease in SCCPs continues or is an artefact.

2.4.6.3 Terrestrial wildlife

To date, very limited information is available on SCCP concentrations in tissues of terrestrial wildlife. In Sweden, Jansson et al. (1993) reported CP concentrations (unspecified chain length) in rabbit (Revingeshed, Skåne), moose (Grimsö, Västmanland), reindeer (Ottsjö, Jaämtland) and osprey (from various regions in Sweden) to be 2.9, 4.4, 0.14 and 0.53 µg/g lipid wt., respectively.

CEFAS (1999) reported the concentrations of SCCPs in sewage sludge, soil, and earthworms associated with uses of chlorinated paraffins in the United Kingdom in the summer of 1998. Concentrations in earthworms ranged between <0.1 to 0.7 mg/kg dry weight.

Campbell and McConnell (1980) determined levels of C₁₀₋₂₀ CPs in birds (Table 23) and seabird eggs in the United Kingdom (Table 24), as well as in water and sediments. The C₁₀₋₂₀ levels are likely to be dominated by contributions from the SCCPs and MCCPs. The method of analysis used was TLC with argentation. The results of Campbell and McConnell (1980) are generally regarded as valid, because similar concentrations in various media in the United Kingdom have been reported using GC-MS methods (e.g., CEFAS 1999; Nicholls et al. 2001). Nevertheless, they must be regarded as semi quantitative.

Table 23: Concentrations of CPs in birds from the United Kingdom (from Campbell and McConnell 1980)¹

Species	Organ	Concentration ² (µg/kg wet wt.)	
		C ₁₀₋₂₀	C ₂₀₋₃₀
Grey heron (<i>Ardea cinerea</i>)	Liver	100–1200	ND–1500
Guillemot (common murre) (<i>Uria aalge</i>)	Liver	100–1100	ND
Herring gull (<i>Larus argentatus</i>)	Liver	200–900	100–500

¹ Table taken from U.K. Environment Agency (2001).

² ND = Not detected (detection limit = 100 µg/kg wet wt.).

Table 24: Concentrations of CPs in seabird eggs¹ (from Campbell and McConnell 1980)²

Concentration (µg/kg)	Number of eggs containing CPs	
	C ₁₀₋₂₀	C ₂₀₋₃₀
Not detected (<50)	7	17
50	3	3
100	3	3
200	5	—
300	1	—
400	2	—
600	1	—
>600 (=2000 µg/kg)	1	—

¹ Species included were great cormorant (*Phalacrocorax carbo*); northern gannet (*Morus bassanus*); great skua (*Catharacta skua*); guillemot (common murre) (*Uria aalge*); black-legged kittiwake (*Rissa tridactyla*); Atlantic puffin (*Fratercula arctica*); Manx shearwater (*Puffinus puffinus*); razorbill (*Alca torda*) and shag (*Phalacrocorax aristotelis*).

² Table taken from U.K. Environment Agency (2001).

Reth et al. (2006) quantified SCCPs in liver from Arctic Char and seabirds (little auk and kittiwake) collected at Bear Island (European Arctic) as well as in cod from Norway. Concentrations between 5 and 88 ng/g wet weight were measured.

2.4.6.4 Plants

Chlorinated paraffins (sum of SCCPs, MCCPs and LCCPs) have been detected in spruce needles sampled from Alpine regions of Europe, ranging in concentration from 26–460 ng/g dry wt. No trend could be ascertained between concentration and altitude (Iozza et al. 2009, as provided in the IPEN Annex E 2010 submission).

2.4.7 Human breast milk and food

Tomy (1997) found that SCCPs (around 60–70% chlorine by weight) were present at a concentration of 11–17 µg/kg lipid (mean concentration 13 µg/kg lipid) in human breast milk from Inuit women living on the Hudson Strait in northern Quebec, Canada.

A recent study has found SCCPs to be present in human breast milk samples from the United Kingdom (Thomas and Jones 2002). In all, 22 breast milk samples were analyzed (eight from Lancaster and 14 from London, randomly chosen from a limited number of samples collected for a different study). SCCPs were found at concentrations of 4.6–110 µg/kg lipid in five out of eight samples from Lancaster and at concentrations of 4.5–43 µg/kg lipid in seven out of 14 samples from London. No SCCPs were found in the remaining samples (the detection limit of the method used varied with sample size but was in the range 1.6–15 µg/kg lipid). Although not calculated in the original paper, it is possible to estimate that the mean level found in breast milk was around 20 ± 30 µg/kg lipid (based on the positive findings alone) or 12 ± 23 µg/kg lipid (assuming that not detected = half the detection limit).

Thomas and Jones (2002) also determined the levels of SCCPs in a sample of cow's milk from Lancaster and single butter samples from various regions of Europe (Denmark, Wales, Normandy, Bavaria, Ireland and southern and northern Italy). SCCPs were not detected in the cow's milk sample (detection limit <1.2 µg/kg lipid) but were found in the butter samples from Denmark at 1.2 µg/kg and Ireland at 2.7 µg/kg. The detection limit for the butter samples ranged between 0.72 and 1.1 µg/kg. In a market basket survey (KAN-DO Office and Pesticides Team, 1995) of 234 ready-to-eat foods, which represented approximately 5000 food types in American diets, "Chlorowax 500C" was detected once, in enriched white bread, at a concentration of 0.13 µg/g. Food items were screened by gas or liquid chromatography using ion-selective detectors. Findings were confirmed by unspecified analysis. Lahaniatis et al. (2000) reported mean values for SCCPs of individual chain length (C₁₀–C₁₃) ranging between 7 - 206 µg/kg of fish oil from various origins.

Concentrations of SCCPs in eleven food categories were determined in Japan (Iino et al. 2005). SCCPs were detected in grain crops (2.5 µg/kg), seeds and potatoes (1.4 µg/kg), sugar, sweets and snacks, seasoning and beverages (2.4 µg/kg), fats (e.g., margarine, oils, etc.; 140 µg/kg), beans, green vegetables, other vegetables, mushrooms and seaweeds (1.7 µg/kg), fruit (1.5 µg/kg), fish (16 µg/kg), shellfish (18 µg/kg), meats (7 µg/kg), eggs (2 µg/kg) and milk (0.75 µg/kg). Total daily intakes (TDIs) of SCCPs for people in Japan were calculated for different age groups of people based on the distributed food consumption and body weight survey data. Due to less body weights, generally, the younger a person is, the higher the person's total daily intake is. The 95th percentile of the total daily intake for 1-year-old female is 0.68 µg/kg/day. Iino et al. (2005) concluded that food was the main exposure pathway of SCCPs to humans, but did not pose any health risk in Japan in 2003.

A follow up study by Thomas et al. (2003; 2006) used more sensitive analytical procedures to analyze SCCPs in breast milk samples (20 from London, 5 from Lancaster). They found concentrations ranging between 49 and 820 µg/kg lipid (median 180 µg/kg lipid). Twenty-five human milk-fat samples were analyzed. SCCPs were detected in all but four samples.

Upper-bound estimates of intake of SCCP for the general Canadian population were calculated by Health Canada (2003). Virtually all of the estimated intake of SCCP for each age group in the Canadian population is from food. The upper-bound estimated intake of breast-fed infants was 1.7 µg/kg-bw per day, and that of formula-fed infants was 0.01 µg/kg-bw per day. For the remaining age groups, intakes ranged from 5.1 µg/kg bw per day for adults over 60 years of age to 26.0 µg/kg-bw per day for infants who were not formula fed. A tolerable daily intake (TDI) for SCCPs of 100 µg/kg-bw per day is given by IPCS (1996).

An independent technical peer review on SCCPs submitted under the UNECE-LRTAP POPs Protocol indicated that aboriginal people living in the Arctic and consuming contaminated animals may be exposed to SCCPs at concentrations greater than the WHO health guideline of 11 µg/kg bw for neoplastic effects (tumor formation) (UNECE-LRTAP POPs Protocol, 2007). However, whether this is the case or not has not been assessed. The only direct measurements of humans in the Arctic are 3 samples of breast milk from the early 1990s analysed by Tomy (1997), which showed concentrations at the low end of the range reported more recently in human milk from the UK (Thomas et al. 2006).

2.5 Hazard Assessment for Endpoints of Concern

2.5.1 Mammalian Toxicity

The following is a brief summary of the relevant information reviewed in the EU Risk Assessment Report (EC, 2000) and the draft Health Canada Assessment Report (2003). Health Canada (2003) reviewed mammalian toxicity studies for SCCPs that have been published since the Government of Canada (1993b) risk assessment.

Wyatt et al. (1993) exposed male rats (Alpk:APfSD strain) by gavage for 14 days to two SCCPs (Chlorowax 500C, C₁₀₋₁₃, 58% chlorine; or Cereclor 56L, C₁₀₋₁₃, 56% chlorine). Doses were 0, 10, 50, 100, 250, 500 and 1000 mg/kg-bw per day. For the 58% chlorine product, absolute and relative liver weights were significantly increased in a dose-related manner, beginning at 100 mg/kg-bw per day. For the 56% chlorine SCCP, the pattern for absolute liver weight was irregular; however, relative liver weight was increased in a dose-related manner, which was significant at 50 mg/kg-bw per day and higher. Peroxisomal fatty acid β -oxidation activity, as indicated by palmitoyl coenzyme A oxidation, was significantly increased at and above 250 mg/kg-bw per day.

In a 13-week oral (gavage) rat study (IRDC 1984; Serrone et al. 1987), increases in liver and kidney weight and hypertrophy of the liver and thyroid occurred at doses of 100 mg/kg-bw per day. The No-Observed-Adverse-Effect Level (NOAEL) was the next lowest dose of 10 mg/kg-bw per day. Health Canada (2003) used this NOEL as the Critical Toxicity Value (CTV) for the risk assessment of SCCPs.

In a 13-weeks study where rats were dosed with SCCPs by gavage, a dose-related increase in relative liver weight was observed starting from the lowest dose of 313 mg/kg/day (NTP, 1986). The US NTP also conducted two long-term studies (13 weeks and 2 years) on mice (NTP, 1986). In the 13 weeks study, a significantly increased relative liver weight was observed at doses of 250 mg/kg/day and higher. In the 2 years carcinogenicity study, doses of 125 and 250 mg/kg/day produced clinical signs of intoxication (decreased activity, prominent backbones, abnormal breathing) at both dose levels and survival was decreased in top dose females. Other effects included dose-related increases in hepatocellular carcinomas and adenomas, and in thyroid follicular cell carcinomas and adenomas.

The EU Risk Assessment Report (EC, 2000) summarized the effect of SCCPs in mammalian species. The results from the available acute and skin irritation studies indicate that effects are not dependant upon the chain length and degree of chlorination. SCCPs have the potential to cause minimal skin irritation, are of low acute toxicity in animals and are not mutagenic. No evidence is available for human carcinogenicity due to SCCPs. Rodent studies showed dose related increases in adenomas and carcinomas in the liver, thyroid, and kidney. They concluded that these was insufficient evidence to conclude that the carcinogenicity observations in the liver and thyroid in mice and the benign tumours in the kidney of male rats were a male rat specific event and consequently the concern for humans could not be ruled out. There are no data on fertility or developmental effects for humans. However, no changes in reproductive organs were observed in a 13 week study with rats and mice dosed with 5000 and 2000 mg/kg/day of an SCCP, though developmental effects were observed in rats at concentrations causing severe maternal toxicity (2000 mg/kg/day) but not at lower doses. Further details on the specific studies can be found in EC (2000).

The Scientific Committee on Toxicity, Ecotoxicity and the Environment found that the alveolar/bronchiolar carcinomas in male mice should not totally be discounted and that the finding of lung tumours in male mice may be of importance for humans (CSTEE 1998). However, the EU risk assessment (EC 2000) also noted that although there was an increase in alveolar/bronchiolar carcinomas in mice, the results were within historical control ranges and the controls had a greater incidence of adenomas of the lung than the treated animals. The EU concluded that there was no significance for human health that could be read into this pattern of results.

A recent study on the induction of renal tumours in male rats by SCCPs (Warnasuriya et al. 2010, submitted by IPEN as an Annex E 2010 submission) indicates that the mechanism by which renal tumours are induced by SCCPs is complex and does not follow the classic profile of male-rat specific alpha-2-urinary globulin (α 2u) nephropathy, i.e., accumulation of renal α 2u and increase in regenerative cell proliferation. Though SCCP was found to bind to α 2u, exposure to SCCP resulted in a down-regulation of α 2u in the liver leading to no observed accumulation of renal α 2u or increase in renal cell proliferation. However, the little α 2u that was expressed in the liver appeared to accumulate in the kidney; this, plus the binding of SCCP to the α 2u indicates that α 2u nephropathy can not be ruled out. It is hypothesized that peroxisome proliferation in the liver may be responsible for the suppression of α 2u expression. As peroxisome proliferation is dose-dependent with SCCPs, lower SCCP doses may result in less α 2u expression, thus leading to greater α 2u nephropathy and an inverse dose response in renal tumour incidence, as was observed in one study (NTP 1986). Further study is needed to determine the exact mechanism of renal tumour induction by SCCPs before it can be concluded that it is male-rat specific.

According to EC (2005), overall, SCCPs are of low toxicity with the principal toxicological issue being for general non-specific toxicity following repeated exposure, with NOAELs for general toxicity of 100 and 1000 mg/kg/day in rats and mice, respectively.

The International Agency for Research on Cancer (IARC) considers there is sufficient evidence for the carcinogenicity (possibly carcinogenic – groups 2B) of a commercial chlorinated paraffin product of average carbon-chain length C_{12} and average degree of chlorination 60% in experimental animals (IARC 1990). The IARC classification is based primarily on a two-year carcinogenesis bioassay of chlorinated paraffins (C_{12} , 60% CL) by the U.S. National Toxicology Program (NTP), which showed evidence of carcinogenicity in rats and mice (NTP 1986). In that study, chlorinated paraffins caused increased incidence of hepatocellular neoplasms in male and female rats, adenomas of the kidney tubular cell in male rats, and follicular cell adenomas of the thyroid gland in female rats. In male and female mice, chlorinated paraffins caused increased incidences of hepatocellular neoplasms and female mice also showed increased incidences of adenomas of thyroid gland follicular cells. However, there continues to be contention over the mechanisms of these tumours and whether they are relevant for human health (EC 2000). SCCPs are classified as hazardous substances in the EU in accordance with Directive 67/548/EEC. They are classified as Carcinogen Category 3: R40 - limited evidence of a carcinogenic effect, and similarly classified as hazardous in Australia.

There is no reproductive toxicity studies conducted with SCCPs. However, the structural analogue MCCPs (medium-chain length paraffin; C_{14} - C_{17} 52% chlorination) has been shown to exert a very specific inhibitory effect on the blood clotting system in rats, which is manifested at the sensitive life-stages at and after birth as severe haemorrhaging, leading to mortality both in pups and the dams (IRDC, 1985) (CXR Biosciences Ltd., 2006). Pup mortality was observed at 74 mg/kg/day, giving an overall NOAEL of 47 mg/kg/day for the pups. The NOAEL for the dams was 100 mg/kg/day. Given the very similar physio-chemical properties and toxicity profiles of SCCPs and MCCPs, SCCPs may also exert toxicity during the reproductive cycle by affecting the blood clotting system, especially in newborn mammals.

To summarise the potential toxicological effects of SCCPs on (probably mainly marine) mammals, SCCPs may affect the liver, the thyroid hormone system, and the kidneys (e.g., by causing hepatic enzyme induction and thyroid hyperactivity, which in the long-term can lead to carcinogenicity in these organs). In addition, given the very similar physico-chemical properties and toxicity profiles of SCCPs and MCCPs, SCCPs may affect the survival of pups via effects on the hematopoietic system similar to MCCPs. Based on the available database, an overall NOAEL of 10 mg/kg/day is deduced from a 13 week rat study. It is not clear whether this NOAEL also covers chronic exposure situations (IRDC, 1984).

2.5.2 Ecotoxicity

2.5.2.1 Microorganisms

Hildebrecht (1972) concluded that a C_{10-13} CP (59% chlorine) and Exchlor 5C (composition unknown) did not affect oxygen utilization by sewage sludge bacteria (species not reported) at concentrations between 1 and 200 mg/L. Birtley et al. (1980) reported that there was no indication that a C_{10-13} CP (49% chlorine) was toxic to four strains of *Salmonella typhimurium* at concentrations as high as 2500 µg/plate. Madeley et al. (1983c) found that a short-chain polychlorinated alkane (58% chlorine) caused significant inhibition (>10%) of gas production by anaerobic microorganisms at concentrations of 3.2, 5.6 and 10%. Effects were observed only for the first 3–4 days of the experiments; by day 10, gas production had returned to normal levels.

The following study is described in EC (2000) and U.K. Environment Agency (2003a): Koh and Thiemann (2001) investigated the toxicity of two SCCPs to bioluminescent bacteria *Vibrio fischeri*. The endpoint was the concentration of the SCCP solution that would cause <20% inhibition of the light emission of the bacteria at 585 nm. The long-term (24-hour) test found a No-Observed-Effect Concentration (NOEC)/EC₂₀ value of 0.1 mg/L for the 56% chlorine by weight C_{10-13} CP and a NOEC/EC₂₀ of 0.05 mg/L for the 62% chlorine by weight C_{10-13} CP.

Sverdrup et al. (2006) investigated the effects of SCCPs (60% Cl) on soil nitrifying bacteria and found them to be the most sensitive, with an EC₁₀ value of 570 mg/kg.

Bezhlebová et al. (2007) investigated the effects of SCCPs (64% CL) on the soil respiration rate (as CO₂ production) of an indigenous microbial community in a natural soil. The authors determined SCCP exposure concentrations of 5,000 and 10,000 mg/kg as NOEC and LOEC values, respectively.

2.5.2.2 Pelagic aquatic organisms

There have been only a limited number of studies on the aquatic toxicity of SCCPs that have been published since the reviews of Tomy et al. (1998a) and the risk assessment of SCCPs by the EU (EC 2000). A summary of these studies is given in Table 25, and they are described further below.

Table 25: Recent studies of the aquatic toxicity of SCCPs

Species	CP	Exposure	Concentration	Notes	Reference
Japanese medaka embryo ¹ (<i>Oryzias latipes</i>)	C ₁₀ H _{15.5} Cl _{6.5}	20-day static test	5.9–9600 µg/L	LOEC = 460 µg/L NOEC = 62 µg/L	Fisk et al. (1999)
Japanese medaka	C ₁₀ H _{15.3} Cl _{6.7} ¹⁴ C-labelled	20-day static test	4.7–7700 µg/L	LOEC = 370 µg/L NOEC = 50 µg/L	Fisk et al. (1999)
Japanese medaka	C ₁₁ H _{18.4} Cl _{5.6}	20-day static test	5.4–8900 µg/L	LOEC = 420 µg/L NOEC = 57 µg/L	Fisk et al. (1999)
Japanese medaka	C ₁₂ H _{19.5} Cl _{6.5} ¹⁴ C-labelled	20-day static test	0.7–270 µg/L	LOEC = 55 µg/L NOEC = 9.6 µg/L	Fisk et al. (1999)
Rainbow trout (<i>Oncorhynchus mykiss</i>)	¹⁴ C ₁₀ H _{15.3} Cl _{6.7} ¹⁴ C ₁₂ H ₂₀ Cl ₆ ¹⁴ C ₁₂ H ₁₆ Cl ₁₀	40 days	0.021–15 µg/g food	No negative effects on mortality, growth or liver somatic index observed	Fisk et al. (1996, 2000)
Rainbow trout ²	C ₁₀ H _{15.5} Cl _{6.5}	21–85 days dietary	0.87–62 µg/g in food	85-day NOEC = 0.10 µg/g in whole fish 21-day LOEC = 0.84 µg/g in whole fish	Cooley et al. (2001)
Rainbow trout ²	C ₁₀ H _{15.3} Cl _{6.7} ¹⁴ C-labelled	21–85 days dietary	0.84–74 µg/g in food	85-day NOEC = 0.099 µg/g in whole fish 21-day LOEC = 0.92 µg/g in whole fish	Cooley et al. (2001)
Rainbow trout ²	C ₁₁ H _{18.4} Cl _{5.6}	21–85 days dietary	0.18–14 µg/g in food	85-day NOEC = 0.10 µg/g in whole fish 21-day LOEC = 5.5 µg/g in whole fish	Cooley et al. (2001)
Rainbow trout ²	C ₁₂ H _{19.5} Cl _{6.5} ¹⁴ C-labelled	21–85 days dietary	1.9–58 µg/g in food	85-day NOEC = 0.14 µg/g in whole fish 21-day LOEC = 0.79 µg/g in whole fish	Cooley et al. (2001)

¹ Lowest-Observed-Effect Concentration (LOEC) and NOEC in medaka are based on any effect in egg or larvae within the first 20 days after laying.

² LOEC and NOEC in rainbow trout are based on feeding behaviour, response to disturbance and histopathological lesions in the liver of group exposed to intermediate concentrations.

Fisk et al. (1999) studied the toxicity of four C₁₀, C₁₁ and C₁₂ SCCP compounds (single chain lengths with mixtures of isomers) to Japanese medaka (*Oryzias latipes*) embryos. Lowest-Observed-Effect Concentrations (LOECs) ranged from 55 µg/L for C₁₂H₂₀Cl₇ to 460 µg/L for C₁₀H₁₆Cl₇. Effects in eggs and larvae over the first 20 days after laying included oil globule migration from the head and thinning of blood vessels between the head and the oil globule. Toxicity was independent of carbon chain length and chlorine content. The mechanism of toxicity to the embryos was suggested to be narcosis. It should be noted that this study did not meet some of the criteria specified in the OECD 201 test guideline for a fish early life stage test, including the following points: there were only 10 eggs per test concentration as opposed to 60 stipulated by OECD, the test took place in sealed vials and the study does not describe dissolved oxygen levels, and the study was carried out for approximately 3 days post-hatch, as opposed to the OECD guideline of 30 days post-hatch.

Hill and Maddock (1983a) found that the hatchability and survival of larvae of the sheephead minnow (*Cyprinodon variegatus*) was unaffected by 28-day exposure to concentrations of SCCPs (58% chlorinated) ranging between 2.4 and 54.8 µg/L. They also observed increases in growth over the acetone controls. In a similar 32-day study with sheephead minnow larvae and SCCP concentrations ranging between 36.2 and 620.5 µg/L the larvae from the highest exposure group were significantly smaller than the acetone group, however larvae from lower exposures were larger than the control. No effects on survival or hatchability were observed (Hill and Maddock 1983b).

Fisk et al. (1996, 2000) studied the accumulation of several ^{14}C -labelled SCCPs (56–69% chlorine by weight) by juvenile rainbow trout (initial weights 2–7 g) during a 40-day exposure period (Table 25). The daily feeding rate was 1.5% of the mean body weight, and two exposure concentrations for each substance were used. At these feeding rates, none of the compounds was found to have any negative effect on the growth or liver somatic index of juvenile rainbow trout. Concentrations in the trout were much less than the LOECs reported for SCCPs in the Japanese medaka study described above (Fisk et al. 1999).

Cooley et al. (2001) examined the behaviour and liver and thyroid histology of juvenile rainbow trout of the same four C_{10} , C_{11} and C_{12} SCCP compounds as in Fisk et al. (1999) via dietary exposure. The exposed trout showed responses indicative of a narcotic mode of action, such as delayed or absent startle response and reduced feeding. Severe liver histopathologies were observed in trout exposed to $\text{C}_{10}\text{H}_{15}\text{Cl}_7$ and $\text{C}_{11}\text{H}_{18}\text{Cl}_6$ (whole fish concentrations of 0.92 and 5.5 $\mu\text{g/g}$ wet wt., respectively), consisting of extensive fibrous lesions and hepatocyte necrosis not seen in controls or lower exposed fish. No thyroid lesions were observed. LOECs for the C_{10-12} SCCPs ranged from 0.79 to 5.5 $\mu\text{g/g}$ in whole fish tissue following dietary exposure to concentrations ranging from 13 to 74 $\mu\text{g/g}$ in food. While a reduced feeding rate was observed in some fish at medium exposure, the fish weight and Liver Somatic Index (LSI) remained the same. These concentrations are within the range of SCCP concentrations seen in carp from Hamilton Harbour (Muir et al. 2001) and in yellow perch from the Detroit River (Tomy et al. 1997) but considerably higher than concentrations found in Lake Ontario fish. Nevertheless, this suggests that histological effects resulting from exposure to SCCPs may be occurring in fish in areas of relatively high exposure, such as Areas of Concern in the Great Lakes.

The lowest toxic effect level identified for a pelagic freshwater aquatic species is 8.9 $\mu\text{g/L}$, which is the 21-day chronic LOEC for *Daphnia magna* (Thompson and Madeley 1983a). The effect was for mortality of the offspring. The NOEC is 5 $\mu\text{g/L}$.

Effects on *Daphnia* and mysid shrimp have been reported at similar concentrations in two other studies. In a 14-day static renewal study using daphnids, 50% mortality was observed after 5 days at 10 $\mu\text{g/L}$ (Thompson and Madeley 1983a). The most sensitive measurement endpoint identified for a marine species is 7.3 $\mu\text{g/L}$, which is the 28-day chronic NOEC for the mysid shrimp (*Mysidopsis bahia*) (Thompson and Madeley 1983b).

Thompson and Shillabeer (1983) carried out a further study on mussels (*Mytilus edulis*) using a 58% chlorinated SCCP using only two exposure concentrations. Groups of 30 mussels were exposed to concentrations of 2.3 $\mu\text{g/L}$ or 9.3 $\mu\text{g/L}$ in a flow through sea water system for 12-hours. No mortalities were observed but growth was reduced in the 9.3 $\mu\text{g/L}$ exposure. Thompson and Madley (1983c) reported a NOEC of 12.1 $\mu\text{g/L}$ in a 10-day study with marine algae *Skeletonema costatum*. The toxic effects were transient with no effects seen at any concentration after 7 days.

Burýšková et al. (2006) observed developmental malformations and reduced embryo growth in *Xenopus laevis* frog at 5 mg/L and higher concentrations of a commercial mixture of SCCPs (C_{12} 56% chlorine). The results were not related to chlorinated pattern and significant induction of the biomarker phase II detoxification enzyme glutathione S-transferase was observed at 0.5 mg/L.

2.5.2.3 Benthic organisms

An equilibrium partitioning approach (Di Toro et al. 1991) using the most sensitive chronic measurement endpoint identified for a pelagic freshwater invertebrate aquatic species (8.9 $\mu\text{g/L}$) was used to estimate the toxicity to benthic organisms, since a valid measurement endpoint was not available for a sediment-dwelling invertebrate. A study was conducted using the midge *Chironomus tentans* (EG&G Bionomics 1983), but exposure was via water only. The following equation was used to calculate the LOEC for benthic invertebrates:

$$\text{LOEC}_{\text{benthic}} = f_{\text{oc}} \cdot K_{\text{OC}} \cdot \text{LOEC}_{\text{pelagic}}$$

where:

- f_{oc} is 0.02, based on the mean organic carbon content for surficial sediment samples from Lake Ontario, expressed on a dry weight basis (Kemp et al. 1977);
- K_{OC} is the organic carbon–water partition coefficient, based on a measured value of 199 500 L/kg for a C_{10} and C_{13} CP with ~55% by weight chlorine content (Thompson et al. 1998); and
- $\text{LOEC}_{\text{pelagic}}$ is 8.9 $\mu\text{g/L}$, based on a 21-day chronic study for *Daphnia magna* (Thompson and Madeley 1983a).

Therefore:

$$\begin{aligned} \text{LOEC}_{\text{benthic}} &= 0.02 \times 199\,500 \text{ L/kg} \times 8.9 \mu\text{g/L} \\ &= 35\,511 \mu\text{g/kg dry wt.} \\ &= 35.5 \text{ mg/kg dry wt.} \end{aligned}$$

2.5.2.4 Soil-dwelling organisms

A very recently published study (Bezchlebová et al. 2007) investigated the effects of SCCPs (64% chlorine content) on five species of soil organisms (collembola, earthworms, nematodes) and on soil microorganisms (for carbon transformation). *Folsomia candida* (collembola) was identified as the most sensitive organism, with LC₅₀ for adult survival, and EC₅₀ and EC₁₀ for reproduction values of 5733 mg/kg, 1230 mg/kg, and 660 mg/kg, respectively, after a 28 day exposure. Bezchlebová et al. (2007) also reported reproduction EC50s of 2849 mg/kg dw (28 day exposure), 6027 mg/kg dw (42 day exposure) and 7809 mg/kg dw (28 day exposure) for *Eisenia fetida*, *Enchytraeus albidus* and *Enchytraeus Crypticus*, respectively.

A study used a “free-living” soil nematode, *Caenorhabditis elegans*, as a toxicity probe for seven pollutants, including SCCPs (labelled as C₁₂, 64% Cl; included short-chain paraffin fractions C₁₀ 6%, C₁₁ 37%, C₁₂ 32%, C₁₃ 25%) (Sochová et al. 2007). Tests were done for 24- and 48-hours and, for the SCCPs, using soil, aquatic and agar toxicity tests. In aquatic tests, no toxicity was observed at the shorter duration but it was one of the most toxic after 48 hours (LC₅₀ and LC₁₀ of 0.5 and 0.3 mg/L, respectively). The greater effect with the longer exposure time is ascribed to more toxicant being taken up with time. The 48h-LC50 was 8833 mg/kg dw.

As a comparison, an equilibrium partitioning approach (Di Toro et al. 1991) using the most sensitive measurement endpoint identified for a pelagic freshwater invertebrate species (8.9 µg/L) was used to estimate the toxicity to soil-dwelling organisms:

$$\text{LOEC}_{\text{soil}} = f_{\text{oc}} \cdot K_{\text{OC}} \cdot \text{LOEC}_{\text{pelagic}}$$

where:

- f_{oc} is 0.02, a standard value given in Mackay (1991); and
- K_{OC} and $\text{LOEC}_{\text{pelagic}}$ values are given in Section 6.1.3.

Therefore:

$$\begin{aligned} \text{LOEC}_{\text{soil}} &= 0.02 \times 199\,500 \text{ L/kg} \times 8.9 \text{ µg/L} \\ &= 35\,511 \text{ µg/kg dry wt.} \\ &= 35.5 \text{ mg/kg dry wt.} \end{aligned}$$

The $\text{LOEC}_{\text{soil}}$ for SCCPs is therefore calculated to be 35.5 mg/kg dry wt.

2.5.2.5 Birds

The EU (EC 2000) reviewed the following avian reproduction study (no reference given), carried out with a C₁₀₋₁₂ CP (58% chlorine), and deemed it to be of good quality. The study was carried out to GLP and was based on the Mallard Reproduction Test (August 1982) of the EPA Environmental Effects Test Guidelines (U.S. EPA 1982).

The study was a 22-week feeding study, including a 9-week pre-egg-laying period without photostimulation, a 3-week pre-egg-laying period with photostimulation and a 10-week egg-laying period with photostimulation. Birds were induced (by photoperiod manipulation) to lay eggs. Eggs were collected over a 10-week period, and the young were observed for 14 days (note that the young were not fed with the test substance). Mortality of adults, egg production, cracked eggs, eggshell thickness, viability, hatchability and effects on young birds were all compared with controls.

The mean measured concentrations were 29, 168 and 954 mg/kg in diet. Twenty pairs of adults were used at each concentration and as control. The lowest level seen to cause slight effects in this study was 954 mg/kg food, which caused a slight, but statistically significant, decrease (by 0.020 mm) in mean eggshell thickness. EC (2000) considered the biological significance of the decrease in eggshell thickness to be questionable, since the mean eggshell thickness in the 954 mg/kg group (0.355 mm) was still in the range of normal values given in the OECD guidelines (0.35-0.39 mm), and no increase in cracked eggs was seen at this dose (EC 2000). No significant difference in the number of eggs laid, number of cracked eggs or mean egg weight was seen in any treatment group when compared with controls.

In a study by Ueberschär et al. (2007), hens from 24 to 32 weeks old were fed technical SCCPs (C₁₀₋₁₃, 60% Cl) in increasing concentrations of up to 100 mg/kg feed. No significant effects were noted on the health, relative organ weights or performance (laying intensity, egg weight, feed consumption) of the hens. Relative organ weights were not significantly affected, except for the pancreas of hens fed the 100 mg/kg wet wt. diet, which was decreased. Less than 1% of the chlorinated paraffins ingested were incorporated into the body, while about 1.5% were eliminated with the egg yolk and 30% were excreted with urine and faeces.

2.5.2.6 Mammals

To determine the toxicity to mammals, the otter was used as a key species. Using the 13-week oral (gavage) rat study by IRDC (1984) as the most sensitive mammals tested the LOAEL was 100 mg/kg-bw per day, based on increases in liver and kidney weight and hypertrophy of the liver and thyroid. This value is selected as the CTV for daily food intake of SCCPs. This CTV is supported by other chronic studies. NTP (1986) observed a significant increase in relative liver weight at doses of 250 mg/kg/day and higher in a 13 week study. In a 2 year carcinogenicity study, doses of 125 and 250 mg/kg/day produced clinical signs of intoxication (decreased activity, prominent backbones, abnormal breathing) at both dose levels and survival was decreased in top dose females (NTP 1986).

Interspecies scaling using data for a typical adult otter will be used to extrapolate to a food concentration for this species. This calculation will involve the use of a typical adult body weight (i.e., 8 kg) and average daily food ingestion rate (0.8 kg wet wt. per day) of an otter (*Lutra canadensis*) (CCME 1998):

$$\begin{aligned}\text{CTV} &= 100 \text{ mg/kg-bw per day} \times 8 \text{ kg-bw}/0.8 \text{ kg wet wt. per day} \\ &= 1000 \text{ mg/kg food wet wt.}\end{aligned}$$

2.5.2.7 Summary of the ecotoxicology of SCCPs

The most sensitive toxicity endpoints for SCCPs is summarized in Table 26. The most sensitive aquatic species appears to be *Daphnia*, with a chronic NOEC of 5 µg/L. It therefore appears that SCCPs can harm sensitive aquatic organisms at relatively low concentrations.

Table 26: Overview of the most sensitive long-term ecotoxicity LOEC/NOEC or EC_x data for SCCPs

Species/endpoint	Effect	NOEC	LOEC or EC _x	Reference
Pelagic (<i>Daphnia magna</i>)	Mortality of offspring, 21 days	5 µg/L	8.9 µg/L	Thompson and Madeley (1983a)
Benthic organisms	Equilibrium partitioning using <i>Daphnia</i> LOEC	NA	35.5 mg/kg dry weight	Environment Canada 2008, based on data from Thompson and Madeley 1983a
Fish (Japanese medaka embryos, early life stage effects)	20-day study, oil globule migration away from head, thinning of blood vessels	9.6 µg/L	55 µg/L	Fisk et al. (1999)
Juvenile rainbow trout	Severe liver histopathologies; extensive fibrous lesions and hepatocyte necrosis, 21 days of exposure		0.79 to 5.5 µg/g ww (whole fish); food concentration is 13 to 74 µg/g ww.	Cooley et al. (2001)
Mysid shrimp (<i>Mysidopsis bahia</i>)	28-day chronic NOEC	7.3 µg/L		Thompson and Madeley 1983b
<i>Folsomia candida</i> (collembola) – soil organisms	Reproduction, 28 day exposure	NA	EC ₅₀ = 1230 mg/kg dw	Bezchlebová et al. 2007
Microorganisms	Soil nitrification (bacterial)		EC ₁₀ = 570 mg/kg dry weight nominal (experimental)	Sverdup et al. (2006)
Mammals - rats	Hypertrophy of the liver and thyroid, increases in liver and kidney weight		100 mg/kg bw/day	IRDC (1984)
Mammals – adult otter	Hypertrophy of the liver and thyroid, increases in liver and kidney weight		1000 mg/kg-food wet wt.	IRDC (1984)

3 Synthesis of Information

SCCPs are persistent, bioaccumulative, toxic to some species, and to undergo long range transport to remote areas.

Total reported annual usage of SCCPs was high in several countries but several have had notable reductions in recent years. For example, use in Canada was approximately 3000 tonnes in 2000 and 2001 (Environment Canada 2003a), in

Switzerland 70 tonnes were used in 1994, which is likely to be reduced by 80% now (Annex E 2007 submission), and in Western Europe, usage was reduced from 13 208 tonnes in 1994 (Euro Chlor 1995) to 4075 tonnes in 1998 (OSPAR 2001). As well, in Australia one of the two companies importing SCCPs in 1998/2000 had ceased importing by 2002. Furthermore, the use of SCCPs in Australia decreased by 80% during this period in the metal working industry (NICNAC 2004). Releases can occur during production, storage, transportation, and use of SCCPs. Facility wash down and spent metalworking / metal cutting fluids are sources to aquatic ecosystems. Although data are limited, the major sources of release of SCCPs are likely the formulation and manufacturing of products containing SCCPs, such as polyvinyl chloride (PVC) plastics, and use in metalworking fluids.

SCCPs are not expected to degrade significantly by hydrolysis in water, and biodegradation studies and dated sediment cores indicate that they persist in sediment longer than 1 year. SCCPs have atmospheric half lives ranging from 0.81 to 10.5 days indicating that they are also relatively persistent in air. SCCPs also have vapour pressures in the range of known persistent organic pollutants that undergo long range atmospheric transport. In general, the HLC values reported imply that SCCPs atmospheric transport is facilitated by remobilisation from water to air due to environmental partitioning. SCCPs have been detected in a diverse array of environmental samples (air, sediment, water, wastewater, fish and marine mammals) and in remote areas such as the Arctic which is additional evidence of long range transport. In addition Arctic Contamination Potential (ACP) modelling and OECD LRTP Screening Tool suggests that SCCPs have moderate ACP when emitted to air and have properties similar to known POPs that are known to undergo long range transport.

Bioaccumulation factors (BAFs) of 16 440–25 650 wet weight (wet wt.) in trout from Lake Ontario indicate that SCCPs can bioaccumulate to a high degree in aquatic biota. This is supported by modelling data for log K_{ow} and bioaccumulation factors which indicate that SCCPs bioaccumulate. In addition, biomagnification factors for some SCCPs have been found to be greater than 1. High concentrations of SCCPs in upper trophic level organisms, notably in marine mammals and aquatic freshwater biota, is additional evidence of bioaccumulation.

SCCPs have so far been found in arctic whales, seals, walrus (Tomy et al, 2000; Canadian Department of Fisheries and Oceans report submitted with the CPIA Annex E 2010 submission), and two species of arctic birds (Little auk, *Alle alle* and Kittiwake, *Rissa tridactyla*) (Reth et al, 2006). Thus, SCCPs have been measured in animal species living in “remote areas”. The following also shows that these concentrations are similar to concentrations of well-recognised POPs such as PCB, DDT and toxaphene (AMAP 2004). More detailed comparisons of POP concentrations for the arctic marine mammals and for the bird Kittiwake (*Rissa tridactyla*) are presented in table 27 and 28, respectively.

Table 27. Comparison by ratio of mean concentrations of SCCP and POPs in arctic species. The mean SCCP concentrations in the different species were 0.2, 0.5, and 0.4 µg/g blubber in beluga whales, ringed seals, and walrus, respectively. (Tomy et al, 2000; AMAP 2004).

Species	[sumPCB]/[SCCP]	[sumDDT/SCCP]	[Toxaphene/SCCP]
Beluga whale	19-24	11-18	15
Ringed seals	2.3	1.3	0.9
Walrus	0.4	0.1	0.6

Table 28. Comparison of concentrations of SCCP and POPs (ng/g lipid weight) in arctic Kittiwake (Reth et al, 2006; AMAP 2004).

Species	[SCCP]	[sumDDT]	[sumPCB]
Kittiwake	110-880 (n=2)	500-1900	10000-21000

SCCPs have also been measured in the breast milk of Inuit women in Northern Quebec, as well as of women in the United Kingdom.

The hazard assessments for SCCPs and MCCPs (medium chain chlorinated paraffins) have shown that these structurally very similar substances also have very similar hazard profiles. Both substances have a similar potency (i.e., NOAELs of the same order of magnitude) (see EU RARs on SCCPs and MCCPs); target organs in mammals include the liver, kidney, and thyroid. Reth et al, (2006) have measured MCCPs in two arctic bird species at concentrations somewhat exceeding the ones of SCCPs. As noted in Annex E, the risk profile can include “consideration of toxicological interactions involving multiple chemicals”, which in this case, would be to consider the combined risk from exposure to both SCCPs and MCCPs. Therefore, the present risk profile could underestimate the risks from SCCPs in the presence of MCCPs.

There is evidence that SCCPs are toxic. The most sensitive organism, *Daphnia magna*, has a chronic NOEC of 5 µg/L. Japanese medaka was also very sensitive to SCCPs. The NOEL was reported to be 9.6 µg/L.

SCCPs have now been measured in various environmental samples (air, sediment, water, wastewater, fish and marine mammals) and in remote areas such as the Arctic (notably in sediment and biota). Data on concentrations in water and sediment are mostly available for Europe and North America, in areas close to potential sources. In biota, SCCPs have been measured in North America and Europe. In the Arctic, SCCPs have been measured in the blubber of marine mammals from several locations, at concentrations of 95 to 626 ng/g wet wt. SCCPs were also measured in fish near Iqaluit at concentrations up to 96 ng/g dry wt.; however, Dick et al. (2010) believe that local sources of SCCPs contribute to these concentrations in addition to long range transport.

Table 29 provides a PEC (Predicted Environmental Concentration), a CTV (critical toxicity value) and a Predicted No-Effect Concentration (PNEC) based on available empirical data for each class of receptors (e.g., pelagic organisms, benthic organisms). The maximum reported value was used as the PEC for each medium. A CTV typically represents the most sensitive chronic toxicity value and does not incorporate application factors ("safety factors") to account for uncertainties, nor for conservative approaches that could be considered for persistent and bioaccumulative substances. Some jurisdictions use NOECs to determine the CTV, however given the paucity of this type of information for SCCP, LOECs were used instead. Application factors were applied to estimate PNECs. A value of 10 was used for extrapolating from a LOEC/LOAEL to a NOEC/NOAEL, and a value of 100 for laboratory to field and intra- and interspecies variations. Thus, a total value of 1000 was used to derive the PNECs. It should be noted that risk may be underestimated using standard risk quotient methods because persistent chemicals may take a long time to reach maximum steady state concentrations in environmental compartments and in tissues of laboratory organisms. Moreover, because food consumption is usually the primary route of exposure to POPs and PBT substances in the field, PNECs may underestimate effect thresholds if the food pathway is not considered in key toxicity studies. Notwithstanding these uncertainties, risk quotients were used to illustrate potential risks. Risk quotients (RQ) were derived to estimate risk for the different classes of receptors.

RQs presented in Table 3-3 reflect more exposure near local regional sources. Current exposure is likely to be less in remote regions.

Table 29: List of Estimated Exposure Values (EEV), Critical Toxicity Values (CTV) and RQs for SCCPs.

Receptor	EEV	Sample	CTV	PNEC	RQ
Pelagic	44.8 ng/L ¹	STP, Hamilton, ON	8,900 ng/L ²	8.9	5
Pelagic	2.63 µg/g ww ³	Lake Ontario	0.79 µg/g ww ⁴	0.00079	3329
Benthic	0.41 mg/kg ⁵	Lake Ontario sediment	35.5 mg/kg ⁶	0.0355	11.5
Soil dwelling	3.2 mg/kg ⁷	UK sewage after 10 years soil application	1230 mg/kg ⁸	1.23	2.6
Microorganisms (bacterial, soil nitrification)	3.2 mg/kg ⁷	UK sewage after 10 years soil application	570 mg/kg ⁹	0.57	5.6
Mammals	2.63 mg/kg ³	Carp from Hamilton Harbour, Lake Ontario	1000 mg/kg food wet wt. ¹⁰	1	2.63

¹. A dilution factor of 10 applied to the value for final effluent of sewage treatment plant in Hamilton Ontario (448 ng/L) (Environment Canada, 2005)

². 21-day chronic LOEC value for *Daphnia magna* (Thompson and Madeley, 1983a)

³. Measured SCCPs in carp collected in Lake Ontario in 1996 and 2001 (Muir et al. 2001; 2002)

⁴. Severe liver histopathologies; extensive fibrous lesions and hepatocyte necrosis of rainbow trout (Cooley et al. (2001)

⁵. Measured in surface sediments from Lake Ontario (Marvin et al. 2003)

⁶. Calculated using the LOEC for *Daphnia magna* using equilibrium partitioning approach (Environment Canada, 2008)

⁷. Estimated in Section 2.4.3.

⁸. Experimental data for soil organisms and microorganisms reported by Bezchlebova et al., (2007)

⁹. Soil nitrification (Sverdrup et al. 2006)

¹⁰. See Section 2.5.2.6 for calculation.

These RQs show that all species could be at risk from exposure to SCCPs. In addition, elevated levels of SCCPs in human breast milk in remote communities have been reported.

The International Agency for Research on Cancer considers some SCCPs (average C₁₂, average 60% chlorination) to be possible carcinogens (groups 2B), although questions have been raised regarding the mechanisms for induction of tumours and the relevance for human health of the studies on which this classification was derived. The Science Committee on Toxicity, Ecotoxicity and the Environment suggests that the finding of lung tumours in male mice may be of importance for humans, but this information would not alter the conclusion of its risk characterisation that the use of short-chain chlorinated paraffins poses no significant risk for consumers or for man exposed via the environment (CSTEE, 1998). The EU Risk Assessment Report (EC 2000) summarized the effect of SCCPs in mammalian species. Rodent studies showed dose related increases in adenomas and carcinomas in the liver, thyroid, and kidney. They concluded that there was insufficient evidence to conclude that the carcinogenicity observations in the liver and thyroid in mice and the benign tumours in the kidney of male rats were a male rat specific event and consequently the concern for humans could not be ruled out. Recent investigations have demonstrated that the mechanism for development of kidney tumours does not follow the classic profile of male-rat specific nephropathy; however, the study could not conclude if the mechanism was rat-specific or not. The EU risk assessment (EC 2000) also noted that although there was an increase in alveolar/bronchiolar carcinomas in mice the results were within historical control ranges and the controls had a greater incidence of adenomas of the lung than the treated animals. The EU concluded that there was no significance for human health that could be read into this pattern of results. An independent technical peer review on SCCPs submitted under the UNECE-LRTAP POPs Protocol indicated that aboriginal people living in the Arctic and consuming contaminated animals may be exposed to SCCPs at concentrations greater than the WHO health guideline of 11 µg/kg bw for neoplastic effects (tumor formation) (UNECE-LRTAP POPs Protocol, 2007). A tolerable daily intake (TDI) for SCCPs of 100 µg/kg-bw per day is given by IPCS (1996). Although expert groups have different opinions as to the interpretation of these data, evidence of toxicity and exposure suggests that humans could be at risk.

SCCPs are persistent and bioaccumulative, and thus concentrations in the environment and biota are expected to increase with continued release to the environment. Standard risk assessment methods comparing effect levels to environmental concentrations may underestimate the risk of persistent and bioaccumulative substances, such as SCCPs. Persistent substances can take decades to reach a maximum steady state concentration in the environment, resulting in an underestimation of the potential exposure to these compounds if steady-state has not been achieved and releases into the environment continue. Similarly, it can take a long time for persistent and bioaccumulative substances to reach a maximum steady state concentration within an organism; this is supported by the observations of Sochová et al. (2007) who noted an increase toxicity of SCCPs with longer exposure duration with nematodes. The durations of standard toxicity tests may be insufficient to achieve the maximum tissue concentration, resulting in an underestimation of the effect threshold.

4 Concluding Statement

In summary, the increasing regulation of SCCPs has resulted in a decrease in SCCPs currently in use. However evidence suggests that significant amounts are still in use and being released in several countries. The available empirical and modelled data strongly indicate that SCCPs are persistent, bioaccumulative, and toxic to aquatic organisms at low concentrations. In mammals, SCCP may affect the liver, the thyroid hormone system, and the kidneys, e.g., by causing hepatic enzyme induction and thyroid hyperactivity. SCCPs have characteristics similar to known POPs that undergo long range environmental transport. SCCPs are considered as POPs pursuant to decisions taken under the UNECE POPs Protocol to the Convention on Long Range Transboundary Air Pollution (LRTAP). Concentrations in biota and sediment from remote Arctic locations also suggest long range transport of SCCPs is occurring via air or ocean currents. SCCPs are present in Arctic marine mammals, which are in turn food for northern indigenous people. SCCPs are measured in human breast milk both in temperate and Arctic populations. Simultaneous exposure to the analogue MCCP would increase the risks because of similar toxicity profiles of SCCP and MCCP.

[Based on the available evidence, it is concluded that SCCPs are likely, as a result of their long-range environmental transport, to lead to significant adverse environmental and human health effects, such that global action is warranted.]

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