**Stockholm Convention
on Persistent Organic
Pollutants**

Persistent Organic Pollutants Review Committee
Fifth meeting
Geneva, 12–16 October 2009

Revised draft risk profile: short-chained chlorinated paraffins

Note by the Secretariat

At its fifth meeting, the Persistent Organic Pollutants Review Committee considered and revised the draft risk profile on short-chained chlorinated paraffins contained in document UNEP/POPS/POPRC.5/2. The Committee agreed that it would continue its consideration of the draft risk profile at its sixth meeting. The draft risk profile, as amended by the Committee at its fifth meeting, is set out below. It has not been formally edited by the Secretariat

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Annex

SHORT-CHAINED CHLORINATED PARAFFINS

REVISED DRAFT RISK PROFILE

Draft revised by the contact group on
Short-chained chlorinated paraffins
at the fifth meeting of the
Persistent Organic Pollutants Review Committee
of the Stockholm Convention

October 2009

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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 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 relatively persistent in air. SCCPs have been detected in diverse environmental samples (air, sediment, water, wastewater, fish and marine mammals), and in remote areas such as the Arctic, providing evidence of long-range transport.

Available empirical (laboratory and field) and modelled data all indicate that SCCPs can accumulate in biota. Laboratory derived BCFs ranged from 1530 – 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.

Freshwater and marine invertebrates appear particularly sensitive to SCCPs, with a reported chronic NOEC of 5 μg/L for *Daphnia magna* and a chronic NOEC of 7.3 μg/L for the mysid shrimp. Severe liver histopathology was observed in trout, with LOECs ranging from 0.79 to 5.5 μg/g in whole fish tissue.

The International Agency for Research on Cancer considers some SCCPs (average C_{12}, 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 concluded in its risk characterisation that the use of short-chain chlorinated paraffins pose no significant risk for consumers or for man exposed via the environment (CSTEE, 1998). The EU Risk Assessment Report (EC 2000) summarized the effects 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. 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 tolerable daily intake (TDI) for SCCPs of 100 μg/kg-bw per day is given by IPCS (1996).

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 are being released in several countries. The available empirical and modelled data indicate that SCCPs are persistent, bioaccumulative, and toxic, particularly to aquatic organisms, and undergo long-range environmental transport. SCCPs are considered as POPs pursuant to decisions taken under the UNECE Aarhus (POPs) Protocol to the Convention on Long Range Transboundary Air Pollution (LRTAP).

SCCPs are persistent in sediments, and have been measured in sediments in Arctic lakes. SCCPs are also particularly toxic to aquatic invertebrates. Given the key role that invertebrates play in aquatic ecosystems, there is concern relating to potential for effects on sediment-dwelling and other invertebrates. Accumulation by freshwater and marine fish is also of concern, given the effects identified in fish.

Although concentrations in water in remote areas are low, 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. SCCPs are measured in human breast milk both in temperate and Arctic populations. Additionally, simultaneous exposure to SCCP and to the related medium chain chlorinated paraffins (MCCPs) would increase the risks because of similar toxicity profiles of SCCPs and MCCPs.

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 Chained 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

The proposal nominates short-chain chlorinated paraffins (Alkanes, C_{10-13}, 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, C_{10-13}, 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. 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 1-1.
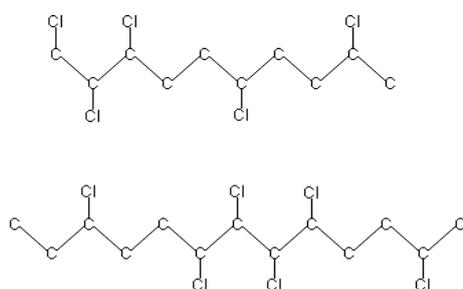
Figure 1-1. Structure of two SCCP compounds ($C_{10}H_{17}Cl_5$ and $C_{13}H_{22}Cl_6$).

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 from several POPRC Parties and observers were also reviewed and any additional information incorporated as appropriate. Some additional information from peer reviewed scientific literature (as of February 1, 2007) is also included, as is additional information identified by Parties and observers during the comment period. Information provided by Parties and observers provided during POPRC 3 has also been incorporated. A more detailed document which served as the basis for this risk profile and a full listing of references for this document can be found in UNEP/POPS/POPRC.3/INF/22.

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 (LRTAP), Aarhus Protocol on Persistent Organic Pollutants. 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.

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. Similar to OSPAR, the Baltic Marine Environment Protection Commission (HELCOM) has included SCCPs on their list of harmful substances (no recommendations have yet been taken).

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.

2.   Summary information relevant to the risk profile

2.1   Physico-chemical properties

Information is available on the physical and 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 x 10^{-7} Pa (Drouillard et al. 1998a, BUA 1992). The vapour pressure of SCCP with 50% chlorine by weight is 0.021 Pa at 40 degree C. (Ref: SRAR-199-ECJRC). Major components of SCCP products with 50-60% chlorine are predicted to have subcooled liquid VPs ranging from 1.4 x 10^{-5} to 0.066 Pa at 25ºC (Tomy et al. 1998a). Henry's Law Constants (HLCs) ranged from 0.7 - 18 Pa•m^3/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 C_{10} and C_{13} chlorinated alkane mixtures ranged from 6.4 - 2370 µg/L (BUA 1992). Water solubility of SCCP containing 59% chlorine content at 20 degree C ranges from 0.15 to 0.47 mg/L (Ref :SRAR-199-ECJRC) . The logarithms of the octanol-water partitioning coefficient (log K_{OW}) were generally greater than five, ranging from 4.48 – 8.69. The log Kow SCCP with chlorine content ranging from 49-71 % ranges from 4.39-5.37 (Ref: SRAR-199-ECJRC). 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

SCCPs are no longer produced in Germany, the latter stopping production in 1995. Prior to 1995, Clariant, Hoechst, and Huels produced SCCPs in Germany. Hoechst produced between 9,300 and 19,300 tonnes/year in Germany between the years 1993 and 1995.

Chlorinated paraffins (CPs) (of various chain lengths) are currently produced in the United States, the EU, Russia, India, China, Japan, Brazil and Slovakia. As noted in the Annex E information submitted by the United States, 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. In 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 (23,000 – 45,000 tonnes), and for CAS # 61788-76-9 (alkanes, chloro; chloroparaffins) in the range of 50 million – 100 million pounds (23,000 – 45,000 tonnes). In 1994, the production and import volume for CAS # 68920-70-7 (alkanes, C_{6-18}, chloro) was in the range of 1 million – 10 million pounds (450 – 23,000 tonnes). Annex E 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.

Between March 1998 and March 2000, approximately 360 tonnes of SCCPs were imported by Australia, according to information submitted by Australia. However, one company had ceased imports of SCCPs by 2002 (NICNAS 2004). In Canada, total reported annual usage of all CPs was approximately 3000 tonnes in 2000 and 2001 (Environment Canada 2003a). In 2002, the Republic of Korea imported approximately 156 tons of SCCPs (CAS No. 85535-84-8) (Comments submitted on April 7, 2008 POPRC SCCPs risk profile). The Canadian sales pattern for SCCPs (as a proportion of total usage of chlorinated paraffins) is similar to the European sales pattern. The Republic of Mauritius does not produce SCCPs (Comments submitted on April 7, 2008 POPRC SCCPs risk profile).

Table 1-1 presents the sales pattern of the EU and North America, the latter being dominated by the United States. Whether these sales patterns are the same at present is not known. Overall, SCCP uses have declined within the EU, in part owing to the phasing out of production and use in Germany (Stolzenberg 1999; OSPAR 2001) and the EU marketing and Use Directive.

Table 1-1. Sales of SCCPs in the EU and North America during the 1990s.

EU[1]			North America[2]		
Year	(tonnes/year)	% of Total CPs Sales	Year	(tonnes/year)	% of Total CPs Sales
1994	13,200				
1997	7,370				
1998	4,080	6.4	1998	7,900	20.6

[1] OSPAR (2001).
[2] CPIA (2000).

2.2.2   Uses and Releases

In Canada in 2003 (Environment Canada 2003a), and in the EU in 1994 (Euro Chlor 1995) and 1998 (OSPAR 2001), the major uses and releases of SCCPs were in metalworking applications. In the EU, 9,380 tonnes/year were used for metalworking in 1994. These amounts were reduced significantly in 1998 (2,018 tonnes/year). Other uses include paints, adhesives and sealants, leather fat liquors, plastics and rubber, flame retardants and textiles and polymeric materials (Table 2). The amounts of SCCPs used in the EU were reduced from 13,208 to 4,075 tonnes/year for all uses in 1994 and 1998, respectively. Since 2002, the use of SCCPs in the EU in metalworking and fat liquoring of leathers has been subject to restrictions under EU Directive 2002/45/EC.

In 1994, 70 tonnes of SCCPs were used in Switzerland and it is estimated that uses have reduced by 80% (Annex E submission). The most widespread use of SCCPs in Switzerland was in joint sealants. In Germany, the most important uses (74% of the total) of SCCPs were banned by the EU directive 2002/45/EC (Annex E submission). 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. Brazil indicates that 300 tonnes/year is used in Brazil for the purposes of flame retardant in rubber, car carpet and accessories (Annex E submission). Use of SCCPs in Australia decreased by 80% between 1998/2000 to 2002 to approximately 25 tonnes/year of SCCPs in the metal working industry (NICNAS 2004). 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 risk profile). The Republic of Mauritius do use SCCPs (Comments submitted on April 7, 2008 POPRC SCCPs risk profile).

Table 2-2 presents the most common uses and releases of SCCPs. When data on SCCPs were not available, data on chlorinated paraffins (CPs) of no specified chain length were presented. There is currently no evidence of any significant natural source of CPs (U.K. Environment Agency 2003a). Anthropogenic releases of SCCPs into the environment may occur during production, storage, transportation, industrial and consumer usage of SCCP-containing products, disposal and burning of waste, and land filling of products (Table 2). The possible sources of releases to water from manufacturing include spills, facility wash-down and storm water runoff. 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 1993a). 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.

Other releases could include use of 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 (CPIA 2002; Environment Canada 2003b).

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).

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.

Table 2-2. Uses and Releases of SCCPs or CPs (various chain lengths)

Application/Use	% by Weight of Final Product	Types of Releases	Amounts Released	Reference
Metalworking lubricants		Loss at production/ formulation site	Controlled CPs losses of 1-2%; 0.06 g/kg CPs consumed; loss of 23 tonnes SCCPs /year in mid-1990s in Europe; default emission factors for CPs are 0.005% to air and 0.25% to wastewater before on-site treatment	EC (2000); KEMI (1991); EU (2003)
		Loss from use	Carry-off from workpieces is 2.5 kg/site/year for small user (100-L capacity) and 2,500 kg/site/year for larger user (95,000-L capacity); annual losses of CPs from cutting fluid are 48%, 75% and 100% for large, medium and small machine shops; 18% loss of SCCPs to wastewater (733 tonnes/year in 1998 in the EU) and 3% disposed in landfill from use in metalworking fluids; 10% discharged to wastewater from use in water-based metalworking fluids; loss of CPs are 18.5% and 31.6% for oil-based and water-based metalworking fluids, respectively. Default emission factors for CPs are 0.02% to air for both types of fluids.	Government of Canada (1993a); EC (2000); U.K. Environment Agency (2003a)
Paints, adhesives and sealants	5-15% CPs (paints) 10-15% CPs (typically for sealant) Up to 20% CPs for some applications (sealant)	Loss at production/for mulation site	Insignificant (paint); Low or zero (sealants); 5% solid waste (sealants)	Zitko and Arsenault (1974); U.K. Environment Agency (2003b)
		Loss from use/application	Waste during application may be disposed in landfill sites; default emission factors for thermosetting resins are 0% to air and 0.1% MCCPs and LCCPs s to wastewater	BRE (1998); U.K. Environment Agency (2003a)
		Loss from leaching	Emission factor of 0.15%/year for MCCPs	U.K. Environment Agency (2003a)
Leather fat liquors	1% SCCPs or less (EU only)			EU Directive 2002/45/EC
Plastics and rubber	10.1-16.8% CPs (conveyor belts) 6.5% CPs (shoe soles) 13% CPs Industrial sheeting)	Loss at production/ formulation site	Default emission factors for plastic additives are 0.1% to air and 0.05% CPs to wastewater; default emission factors for thermosetting resins are 0% to air and 0.05% CPs to wastewater	BRMA (2001); U.K. Environment Agency (2001); BRE (1998)
		Loss from use	Loss during wear and abrasion of products	
		Losses from volatilization	0.05% during lifetime of product	
Flame retardants	1-4% CPs (typically) Up to 15% CPs for some applications 1-10% SCCPs added to rubber			Zitko and Arsenault (1974); BUA (1992)
Textiles and polymeric materials		Loss at production/ formulation site	17% of SCCPs use in 1998	EC (2000)

2.2.3 Overall emissions in Europe and 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 2-3. Behaviour similar to that of MCCPs (U.K. Environment Agency 2003a) 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 2-3. 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.

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; http://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 C_{10-13} species with 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 (19404 kg) of polychlorinated alkanes was reported for on and off site disposal or other releases by U.S. industries subject to reporting, including 1,527 pounds (693 kg) as fugitive air emissions; 1,941 pounds (880 kg) as point source air emissions; and 7 pounds (3.2 kg) as 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).

2.3 Environmental Fate

2.3.1 Persistence

Persistence in Air
Estimated atmospheric half-lives for SCCPs based on reaction with hydroxyl radicals range from 0.81 to 10.5 days, using the default atmospheric hydroxyl radical concentration of 1.5×10^6 molecules/cm^3 during sunlight hours in AOPWIN (v. 1.86) computer program (Meylan and Howard, 1993; Atkinson 1986, 1987). Using a lower hydroxyl radical concentration of 5×10^5 molecules/cm^3, which is generally used as a daily (24-hour) average in relatively unpolluted air in the EU, atmospheric half-lives ranged from 1.2 to 15.7 days. It should be noted that hydroxyl radical reaction rates vary temporally with average daily sunlight, and 5×10^5 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 conditions at high latitudes, may limit the atmospheric oxidation pathway.

Persistence in water
SCCPs 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. While the results suggest that photolysis may be a degradation pathway for some SCCPs, the environmental relevance of this study is questionable as the use of an ultraviolet irradiation source may have produced far shorter half-lives than under natural light conditions. Also, the extent of photodegradation may be limited in waters at depth and/or at northern latitudes, and as acetone is a questionable solvent to use in such a study as it is a photo sensitizer.

Persistence in soil and sediment

Using 25-day biochemical oxygen demand (BOD) tests, Madeley and Birtley (1980) found that SCCPs (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). Fisk et al. (1998a) found that two ^{14}C-labelled C_{12} 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. Half-lives in sediment were 12 ± 3.6 days and 30 ± 2.6 days for the 56% and 69% chlorine products, respectively.

A study on the aerobic and anaerobic biodegradation of SCCPs in both freshwater and marine sediments was recently undertaken (Thompson and Noble 2007, in U.K. Environment Agency 2007). Using ^{14}C-labelled n-decane and n-tridecane 65% chlorine by weight products and basing their experiments on the OECD 308 Test Guideline, the mean half-lives (for mineralization [carbon dioxide or methane production]) for a C_{10-13}, 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.

Concentration profiles of SCCP residues in 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 dating back to 1949. The highest concentration (800 ng/g dry wt.) was observed in the slice dated at 1971 (Muir et al. 1999a). SCCPs were also measured in a dated sediment core collected in Lake St. Francis (Lac Saint-François), in 1996, downstream of a former CPs manufacturing site. The historical profiles show the presence of relatively low levels of SCCPs compared with Lake Ontario (Muir et al. 1999a; 2002). The highest SCCP concentrations had a median date of 1985 ± 4 years (Turner 1996). The predominant chain length groups in sediments were C_{11} and C_{12}.

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 evidence that SCCPs can persist for more than 50 years in subsurface anaerobic sediments. Environment Canada (2004) used first order decay equations in a back calculation method to determine that SCCPs have a half-life in sediments longer than 1 year. 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. Several government assessments and published reviews have concluded that only slow biodegradation in the environment may be expected to occur, even in the presence of adapted micro-organisms (Government of Canada 1993a,b; Tomy et al. 1998a; EC 2000).

Little information is available on the persistence of SCCPs in soil. A study by Omori et al. (1987) studied the dechlorination potential of a series of soil bacterial strains acting on $C_{12}H_{18}Cl_8$ (63% chlorine). Although they could not isolate a bacterial strain that could use this chemical 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) and a single strain (HK-3) acting alone released 21% and 35% of the chlorine, respectively, after 48 hours. Another study by Nicholls et al. (2001) investigated the presence of SCCPs in farm soils in the United Kingdom on which several applications of sewage sludge had been applied. They were unable to detect SCCPs (<0.1 μg/g). However, the study did not specifically follow the fate of SCCPs over time following sludge application, and therefore the relevance of these results is questionable.

Summary of Persistence

SCCPs meet the criterion for persistence for sediment (Annex D, Stockholm Convention). They also are sufficiently persistent in air for long range transport to occur. While SCCPs appear to be hydrolytically stable, there is insufficient information to conclude on their persistence in water. There is also insufficient information on their persistence in soil. Overall, SCCPs are considered to meet the criteria for persistence in Annex D.

2.3.2 Bioaccumulation

Modelled Log Kow and Bioaccumulation Factors

Sijm and Sinnige (1995) calculated a log K_{ow} range between 4.8 and 7.6 for all possible SCCP congeners. Fisk et al. (1998b) determined the octanol-water partition coefficients for $C_{12}H_{20.1}Cl_{5.9}$, 55.9% wt. Cl and $C_{12}H_{16.2}Cl_{9.8}$, 68.5% wt. Cl. The mean log K_{ow} values was estimated at 6.2 for the 55.9% wt. Cl substance (range of log Kow was 5.0 to 7.1) and 6.6 for the 68.5% wt. Cl substance (range of log K_{ow} was 5.0 to 7.4). Using empirical K_{OW} data and assuming no metabolism, the Gobas BAF model for fish estimated BAF values greater than 5000 for all possible SCCPs.

Bioconcentration

Bioconcentration factors (BCFs) calculated from laboratory studies for SCCPs have been reviewed in Government of Canada (1993b) and were found to 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). No other recent laboratory BCF studies have been identified.

Information submitted by Japan on the measured BCF test results for CPs (C=11, Cl = 7-10) suggested that these chemicals are bioaccumulative. BCF measurements were taken at two test concentrations (0.1 and 1 μg/L) and test organisms were exposed between 12 and 60 days. A trend in bioconcentration potential was not observed. BCFs ranged from 1900 to 11,000.

Another paper submitted by Japan provides information on SCCPs of a C13 chain length. CPs (C=13, Cl=5-7) were tested at two concentrations (0.01 and 0.001 mg/L) and carp (*Cyprinus carpio*) were exposed for 62 days. BCFs of the tested CPs ranged from 1530 to 2830 (UNEP/POPS/POPRC.5/INF/23).

Laboratory studies of bioaccumulation, biomagnification and biotransformation

SCCPs can accumulate in food consumed by fish. 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, implying a potential to biomagnify in aquatic food chains. Depuration half-lives in fish ranged from 7 to about 53 days for juvenile rainbow trout (Fisk et al. 1998b). In another study, Fisk et al. (2000) estimated SCCPs depuration half-lives that ranged from 7.1 to 86.6 days for rainbow trout. The authors observed that SCCPs with lower chlorination were being metabolized by fish but that the half-lives of two higher chlorinated SCCPs, $C_{12}H_{16}Cl_{10}$ and $C_{12}H_{20}Cl_6$, were similar to those of recalcitrant organochlorines, with long biotransformation half-lives >1000 days. Fisk et al. (2000) found that the depuration and biotransformation half-lives for some C_{10-12} SCCPs, particularly the decanes, were similar, suggesting that the depuration is primarily due to biotransformation. Fisk et al. (2000) also showed that the BMFs for 35 chlorinated n-alkanes (combined data for SCCPs and MCCPs) were significantly related to the number of carbon plus chlorine atoms per compound and to log K_{ow}.

Bengtsson and Baumann-Ofstad (1982) found that, although the uptake efficiency (91-day uptake period) of a SCCP composed of 71% chlorine by weight was low (6%), it had a remarkably high retention in bleak (*Alburnus alburnus*). This formulation remained in the fish tissues at a steady level until the experiment was terminated after the 316-day elimination period. Similar observations were reported by Fisk et al. (1998a) in oligochaetes (*Lumbriculus variegatus*) for $C_{12}H_{20}Cl_6$ (56% chlorine by weight) and $C_{12}H_{16}Cl_{10}$ (69% chlorine by weight). Organic carbon normalized biota-sediment accumulation factors (BSAF) calculated from the rates of uptake and depuration ranged from 1.9 for $C_{12}H_{16}Cl_{10}$ to an average of 6.8 for $C_{12}H_{20}Cl_6$. Half-lives of the two SCCPs were similar (~12-14 days), but uptake of the more highly chlorinated dodecane was significantly slower than that of the less chlorinated dodecane.

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.

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 and dissolved water concentrations (Muir et al. 2001). BAFs ranged from 88 000 to 137 600 in lake trout on a lipid weight basis, and from 16 440 to 25 650 on a wet weight basis. Chlorinated dodecanes (C_{12}) were the most prominent SCCPs in lake water and fish. 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 (C_{13}) because of their low water concentrations. 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. 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).

Bioaccumulation and biomagnification factors were calculated in Lake Ontario for total SCCP isomers, as well as SCCP chain length groups and homologs (same molecular formula), that were detected in water and organism samples (Houde et al. 2008). Mean BAFs in lake trout from Lake Ontario varied between $4.0x10^4$ and $1.0x10^7$ for C10, $1.3 x10^5$ and $3.2 x10^6$ for C11, $1.0x10^5$ to $2.5x10^5$ for C12, and $2.5x10^5$ for C13 homologs, on a lipid-normalized basis. The highest BMF for total SCCPs was observed for sculpin-*Diporeia* (BMF = 3.6) in both Lake Ontario and Lake Michigan food webs. 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) for total SCCPs were reported to be 0.97 and 1.2 for Lake Ontario and Lake Michigan, respectively. Among individual SCCP homologs $C_{10}Cl_8$ and $C_{11}Cl_8$ had the highest TMFs (1.5). A TMF above 1 indicates that certain SCCP isomers have the potential to biomagnify in aquatic food web (Houde et al. 2008).

Bioaccumulation Summary
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. Based on these data, SCCPs are considered bioaccumulative according to the criteria listed 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/m^3 in air collected in the high Arctic (Alert, Ellesmere Island), while Borgen et al. (2000) measured SCCPs ranging from 9.0-57 pg/m^3 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, as well as in Arctic biota, such as ringed seal, beluga whale, walrus (Tomy et al. 2000), char and seabirds (Reth et al. 2006). The concentration profiles for SCCPs in Arctic marine mammals show a predominance of the shorter carbon chain length congeners, i.e., the C_{10} and C_{11} 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 C_{10} 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 ($C_{10}H_{17}Cl_5$, $C_{10}H_{16}Cl_6$, $C_{10}H_{15}Cl_7$, $C_{11}H_{18}Cl_6$, $C_{11}H_{17}Cl_7$, $C_{12}H_{20}Cl_6$, $C_{12}H_{19}Cl_7$), are greater than 2 days (Section 2.2.1).

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·m^3/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 (e.g., hexachlorocyclohexane [lindane], heptachlor, mirex).[1] As well, modelling exercises have been undertaken. Annex E information submitted by Switzerland outlines a study by Wegmann et al. (2007) which examined the long range transport of SCCPs and other POPs candidates using the OECD Pov and LRTP Screening Tool. The results indicated that SCCPs have Pov and LRTP properties similar to those of several known POPs. The Arctic contamination potential (ACP) of several SCCPs was estimated based on their K_{OA} and K_{AW} values, and compared to the ACP results generated for a hypothetical series of chemicals (Wania 2003). Results suggest that SCCPs have ACPs similar to tetra- to heptachlorobiphenyls.

The available information supports the conclusion that SCCPs undergo long range transport.

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[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·m^3/mol, respectively.

2.4 Exposure

2.4.1 Atmospheric concentrations

SCCPs were measured in air in several countries including Canada, the United Kingdom (U.K.) and Norway. SCCPs were detected in four air samples collected at Alert at the northern tip of Ellesmere Island in the high Arctic. Concentrations ranged from <1 to 8.5 pg/m^3 in gas-phase samples (Tomy 1997, Bidleman et al. 2001). Borgen et al. (2000) measured SCCPs in Arctic air samples taken at Mt. Zeppelin, Svalbard, Norway, in 1999. Concentrations ranging from 9.0 to 57 pg/m^3 were detected. 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 ranged from 1800 to 10 600 pg/m^3.

Concentrations of SCCPs in air samples collected at Egbert, Ontario, Canada, in 1990 ranged from 65 to 924 pg/m^3 (Tomy 1997; 1998a). Concentrations of SCCPs over Lake Ontario in 1999 and 2000 ranged from 120 to 1,510 pg/m^3 (Muir et al. 2001; D.C.G. Muir, unpublished data, 2001).

Peters et al. (2000) reported a mean SCCP concentration of 99 pg/m^3 in air collected from a semi rural site in Lancaster, U.K. Barber et al. (2005) found that concentrations in the U.K. atmosphere in 2003 ranged between <185 to 3430 pg/m^3 (mean of 1130 pg/m^3) and were higher than 1997 concentrations at the same site. Barber et al. (2005) also calculated an average concentration of 600 pg/m^3 of SCCPs for the UK atmosphere.

SFT (2002) measured SCCP concentrations in three 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 deposition of SCCPs from the atmosphere.

2.4.2 Wastewater treatment effluents, sewage sludge and soils

SCCPs were detected in all eight sewage treatment plant final effluents sampled from southern Ontario, Canada. Total SCCPs (dissolved and particulate C$_{10-13}$) ranged from 59 to 448 ng/L. The highest concentrations were found in samples from treatment plants in industrialized areas, including Hamilton, St. Catharine's and Galt.

Reiger and Ballschmiter (1995) reported C$_{10-13}$, 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 C$_{10-13}$, 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.

SCCPs have also been measured in sewage sludge. Stevens et al. (2002) found SCCP concentrations ranging from 6.9 to 200 µg/g dry wt. in sewage sludge from 14 waste water treatment plants in the United Kingdom. Highest concentrations of SCCPs were in sludge from industrial catchments. However, a rural catchment with zero industrial effluent had significant levels (590 µg/g) of total SCCPs/MCCPs in sludge (Stevens et al. 2002). Agricultural soils may also be a potentially major reservoir of CPs due to sewage sludge application (Stevens et al. 2002; Nicholls et al. 2001).

2.4.3 Surface waters

SCCPs were detected in surface waters in Ontario and Manitoba, Canada. Low levels of dissolved total (C$_{10-13}$) SCCPs were measured in western Lake Ontario in 1999 and 2000. Concentrations ranged from 0.168 to 1.75 ng/L in 1999 while concentrations were lower (0.074 to 0.77 ng/L) in 2000 (Muir et al. 2001). The average concentration of total SCCPs in water from Lake Ontario (4 metre 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. The Ministry of the Environment (2006) in Japan monitored SCCPs in six 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 two rivers (four sites) in Japan in 2002. SCCPs concentrations ranged from 7.6 to 31 ng/L (Iino et al. 2005; Takasuga et al. 2003).

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).

2.4.4 Sediments

SCCPs were detected in sediments around the Great Lakes in Canada, Germany, Czech Republic, Switzerland, Spain, France, Norway, Japan and the United Kingdom.   They have also been detected in Arctic sediment.

Fluxes for surface slices (0-1 cm depth) of sediment core (µg/m^2 per year) of SCCPs to various Canadian lake sediments have been measured (Muir et al., 1999a; Tomy et al., 1999). 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.

Tomy et al. (1997) measured SCCPs at concentrations around 245 µg/kg dry weight in sediment grab samples from the mouth of the Detroit River at Lake Erie and Middle Sister Island in western Lake Erie, in 1995. SCCPs were also detected in all surface sediment samples from harbour areas along Lake Ontario at concentrations ranging from 5.9 to 290 ng/g dry wt. (Muir et al. 2001). The highest concentrations were found at the most industrialized site (Windermere Basin, Hamilton Harbour). Similarly, Marvin et al. (2003) reported a SCCPs concentration of 410 ng/g dry wt. in Lake Ontario sediments near an industrialized area.

In the Canadian Arctic, total SCCP concentrations in sediment cores (0-1 cm depth) ranged from 1.6 to 17.6 ng/g dry wt. in three remote lakes (Tomy et al. 1998a, Stern and Evans 2003).

Ballschmiter (1994) found SCCPs in sediments in Germany at concentrations ranging from <5 to 83 µg/kg dry wt. The 83 µg/kg dry wt. sample was from the Rhine River (U.K. Environment Agency 2003b).

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

Přibylová 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 180.75ng/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) (Štejnarová et al. 2005). The Ministry of the Environment (2006) in Japan monitored SCCPs in six 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 three 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.

2.4.5　Biota

SCCPs were detected in biota in Canada, United Kingdom, Chile, Greece, Germany, Iceland, France, Sweden, United States, Norway, Japan and the North Sea. They have also been detected in Arctic biota.

Freshwater organisms
Muir et al. (2001, 2002) measured SCCPs in fish collected in Lake Ontario in 1996 and 2001. Concentrations ranged from 7.01 to 2,630 ng/g wet weight. The highest concentration was measured in carp collected at Hamilton harbour. C_{12} SCCPs predominated in lake trout, whereas C_{11} was the major SCCP in sculpin and smelt.

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).

Concentrations of SCCPs in lake trout, carp and food web samples from Lake Ontario and Lake Michigan were determined from samples collected between 1999 and 2004 (Houde et al. 2008). 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.

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. Lahaniatis et al. (2000) reported mean values for SCCPs of individual chain length (C_{10}-C_{13}) 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. 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.

The Ministry of the Environment (2006) in Japan monitored SCCPs in six 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).

Marine mammals
SCCPs have been found at concentrations ranging from 95 to 626 ng/g wet wt. in the blubber of marine mammals, including beluga (*Delphinapterus leucas*), ringed seal (*Phoca hispida*) and walrus (*Odobenus rosmarus*) from several locations in the Arctic (Tomy et al. 1998b; 2000).

SCCPs have been detected in belugas from the St. Lawrence River at an average concentration of 785 ng/g wet weight (Tomy et al. 1998b; 2000). 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 lengths in 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 (Tomy et al. 1999). More recent measurements of SCCPs in blubber of 5 species of marine mammals from west Greenland showed total SCCPs concentrations ranging from 10 ng/g wet wt in ringed seals to 282 ng/g ww in beluga (Johansen et al. 2004).

Jansson et al. (1993) reported a SCCP concentration of 130 ng/g wet wt. in ringed seal blubber from Svalbard. 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).

Terrestrial wildlife

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 (Grismsö, 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 earthworms ranging from <0.1 to 0.7 µg/g dry weight in the United Kingdom in the summer of 1998. Campbell and McConnell (1980) determined levels of C_{10-20} CPs in liver of birds and seabird eggs in the United Kingdom. The C_{10-20} levels were likely to be dominated by contributions from the SCCPs and MCCPs. Concentrations of C_{10-20} CPs ranged from 0.1 to 1.2 µg/g wet weight in liver of birds and from <0.05 to >6 µg/g.

2.4.6   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.

SCCPs were also detected in human breast milk samples from the United Kingdom (Thomas and Jones 2002). 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. The estimated mean level of SCCPs was 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). In a follow-up study, Thomas et al. (2003; 2006) found concentrations of SCCPs ranging from 49 to 820 µg/kg lipid (median 180 µg/kg lipid) for the same two UK cities. Twenty-five human milk-fat samples were analyzed. SCCPs were detected in all but four samples.

SCCPs have also been detected in food. Thomas and Jones (2002) detected SCCPs in a sample of cow's milk from Lancaster and butter samples from various regions of Europe (i.e., Denmark, Wales, Normandy, Bavaria, Ireland and southern and northern Italy). SCCPs were also found in butter samples from Denmark at 1.2 µg/kg and Ireland at 2.7 µg/kg. In a market basket survey of 234 ready-to-eat foods representing approximately 5000 food types in American diets, "Chlorowax 500C" was detected once, in enriched white bread, at a concentration of 0.13 µg/g (KAN-DO Office and Pesticides Team, 1995). Lahaniatis et al. (2000) reported mean values for SCCPs of individual chain length (C_{10}-C_{13}) 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 lower 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.

Upper-bound estimates of intake of SCCP for the general Canadian population were calculated by Health Canada (2003). Virtually the entire 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.

2.5   Hazard Assessment for Endpoints of Concern

2.5.1   Mammalian Toxicity

Wyatt et al. (1993) exposed male rats by gavage for 14 days to two SCCPs (58% and 56% chlorine). Both absolute and relative liver weights were significantly increased in a dose-related manner for the 58% chlorine SCCPs at 100 mg/kg-bw per day and higher. For the 56% chlorine SCCPs, relative liver weight was significantly increased in a dose-related manner at 50 mg/kg-bw per day and higher.

Several subchronic studies were conducted. In a 13-week oral rat study (IRDC, unpublished, 1984; Serrone et al. 1987), rats were given short-chain chlorinated paraffins via the diet or via gavage (in separate studies) at doses of 10, 100, or 625 mg/kg/day. Dose-dependent increases in absolute and relative liver and kidney weights were observed as from doses of 100 mg/kg/day. While the original interpretation considered these effects as adaptive, more recent interpretations consider them as adverse effects; at this dose, microscopic changes in liver, kidney and thyroid were also observed. A NOAEL of 10 mg/kg/day can be derived (draft support document for identification of SCCP as a substance of very high concern under REACH, October 2008).

The US NTP also conducted studies on SCCPs in both rats and mice. 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 NTP also conducted a 13-week and a two year lifetime study 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.

Based on these studies, the International Agency for Research on Cancer determined in 1990 that there is sufficient evidence for the carcinogenicity (possibly carcinogenic – groups 2B) of the commercial chlorinated paraffin product tested, which is described as an average carbon-chain length C_{12} and average degree of chlorination 60% (IARC, 1990). Several mechanistic studies were conducted to understand the mechanisms of these tumours and whether they are relevant for human health (EC 2000).

The Scientific Committee on Toxicity, Ecotoxicity and the Environment reviewed a draft of the EU risk assessment for SCCPs. The CSTEE concluded that:

> *"The liver and thyroid are target organs in repeated dose studies with rats and mice. The liver damage is associated with peroxisome proliferation, whereas the thyroid effects are correlated to altered thyroid hormone status and glucuronyl transferase induction. Humans would be much less sensitive to peroxisome proliferation and thyroid hormone perturbation than rats and mice."*

> *In rats, neoplastic effects are seen in the liver, the thyroid (females) and the kidneys (males) as well as mononuclear cell leukaemia. There was poor survival in this experiment. In order to elucidate the mechanism underlying the renal tumorigenic effect, there were indications that a 58% chlorinated paraffin caused hyaline droplet formation in male rats, but this could not be confirmed by immunocytochemical techniques.*

CSTEE also found that the alveolar/bronchiolar carcinomas in male mice should not totally be discounted but acknowledged that while "the control animals in this experiment did not show any evidence of lung tumours, whereas the historical control incidence was 5.8%."(CSTEE 1998). The final EU risk assessment (EC 2000) 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 rodent studies showed dose related increases in adenomas and carcinomas in the liver, thyroid, and kidney. They determined 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. SCCPs are classified in the EU as a Carcinogen Category 3: R40 – Limited evidence of a carcinogenic effect, and are similarly classified as hazardous in Australia.

In conclusion, effects on the liver, thyroid, and kidney have been shown to occur in mammalian species exposed to short-chain chlorinated paraffins. The effects are manifested as organ weight increases and histological changes after exposure for weeks or months, but may turn into carcinomas and adenomas after chronic exposure. An overall NOAEL of 10 mg/kg/day can be derived from the 13 weeks studies. No NOAEL can be obtained from the chronic studies.

No changes in reproductive organs were observed in a 13-week study with rats and mice dosed with 5000 and 2000 mg SCCP/kg/day. Developmental effects were observed in rats at concentrations causing severe maternal toxicity (2000 mg/kg/day) but not at lower doses (EC 2000).

There are no fertility studies conducted with short-chain chlorinated paraffins, and there is thus a data gap when it comes to potential effects on pups, e.g., during lactation. However, the structural analogue medium-chain chlorinated paraffins (C14-C17 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, in EU-RAR on MCCP, 2008). 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. Read-across of this effect from MCCPs to SCCPs was discussed by the EU classification and labelling group. It was agreed that, in view of remaining uncertainties in the mechanisms of toxicity of these chlorinated paraffins, read-across was not justified. This leaves a potential data gap for SCCPs in relation to this endpoint.

To summarise the potential toxicological effects of short-chain chlorinated paraffins on (e.g., marine) mammals, it 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.

2.5.2   Ecotoxicity

Microorganisms

A number of tests, utilizing various species, endpoints, and SCCPs have investigated the toxicity of SCCPs to microorganisms (Hildebrecht 1972, Birtley et al. 1980, Madeley et al.1983c, Koh and Thiemann 2001, Sverdrup et al. 2006). The lowest NOEC/EC20 values were 0.1 mg/L for a 56% chlorine C_{10-13} CP and 0.05 mg/L for a 62% chlorine C_{10-13} CP with *Vibrio fischeri* (Koh and Thiemann 2001). In soils, the lowest effect concentration was noted by Sverdrup et al. (2006), who determined an EC10 of 570 mg/kg dry wt. for a 60% chlorine SCCP.

Pelagic aquatic organisms

There are 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 provided below.

The lowest toxic effect level identified for a pelagic freshwater aquatic species is 8.9 µg/L based on a 21-day chronic LOEC for *Daphnia magna* (Thompson and Madeley 1983a). The effect was for mortality of the offspring. The NOEC is 5 µg/L. Other effects on *Daphnia* have been reported at similar concentrations. In a 14-day static renewal study, 50% mortality was observed after 5 days at 10 µg/L (Thompson and Madeley 1983a).

The most sensitive measurement endpoint identified for a marine species is 7.3 µg/L based on a 28-day chronic NOEC for the mysid shrimp (*Mysidopsis bahia*) (Thompson and Madeley 1983b). Thompson and Madeley (1983c) reported a NOEC of 12.1 µg/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. Thompson and Shillabeer (1983) exposed groups of 30 mussels (*Mytilus edulis*) to 58% chlorinated SCCP (2.3 µg/L and 9.3 µg/L) in a flow-through sea water system for 12-hours. No mortalities were observed but growth was reduced in the 9.3 µg/L exposure.

Fisk et al. (1999) studied the toxicity of four C_{10}, C_{11} and C_{12} SCCP compounds (single chain lengths with mixtures of isomers) to Japanese medaka (*Oryzias latipes*) embryos. While there are uncertainties in the study results, notably associated with the controls, Lowest-Observed-Effect Concentrations (LOECs) ranged from 55 µg/L for $C_{12}H_{20}Cl_7$ to 460 µg/L for $C_{10}H_{16}Cl_7$. Effects in eggs and larvae over the first 20 days after laying included large yolk sac, lethargic or no movement but heart beat present. These effects were observed in C_{10-12} of SCCP, and no dose-response in C_{12}, in which 9.6 µg/L of NOEC was observed. Toxicity was independent of carbon chain length and chlorine content. The mechanism of toxicity to the embryos was suggested to be narcosis.

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. The daily feeding rate was 1.5% of the mean body weight. None of the compounds was found to have any negative effect on the growth or liver somatic index of juvenile rainbow trout.

Cooley et al. (2001) examined the behaviour of juvenile rainbow trout and their liver and thyroid histology following exposure to the same four C_{10}, C_{11} and C_{12} SCCP compounds as in Fisk et al. (1999) via dietary exposure. 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 $C_{10}H_{15}Cl_7$ and $C_{11}H_{18}Cl_6$, consisting of extensive fibrous lesions and hepatocyte necrosis not seen in controls or lower exposed fish. Effects for the $C_{10–12}$ SCCPs were observed at concentrations ranging from 0.79 to 5.5 µg/g in whole fish tissue following dietary exposure to concentrations ranging from 13 to 74 µg/g in food**.** No thyroid lesions were observed. It is not clear from these studies if the effects seen were a result of a direct toxic effect of the short-chain chlorinated paraffin or occurred as result of the reduced feeding seen in many of the exposed fish (although this in itself could be considered an effect of the short-chain chlorinated paraffin). Thus, although it is clear that adverse effects were seen, it is not certain that they are directly related to the toxicity of short-chain chlorinated paraffins.

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 the chlorination pattern.

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 µg/L) was used to estimate the toxicity to benthic organisms, since a valid measurement endpoint was not available for a sediment-dwelling invertebrate. The $LOEC_{benthic}$ was estimated to be 35.5 mg/kg dry wt. (Environment Canada 2004).

Soil-dwelling organisms
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 LC_{50} (adult survival), and EC_{50} and EC_{10} (reproduction) values of 5733 mg/kg, 1230 mg/kg, and 660 mg/kg dry wt. respectively, after a 28 day exposure. Bezchlebová et al. (2007) also reported reproduction EC_{50}s of 2849 mg/kw 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. Sverdrup et al. (2006) investigated the effects of SCCPs (60% chlorine content) on earthworms, soil nitrifying bacteria, and red clover. These authors found the nitrifying bacteria to be the most sensitive, with an EC_{10} value of 570 mg/kg dry wt.

A study by Sochová et al. (2007) used a free-living soil nematode, *Caenorhabditis elegans*, as a toxicity probe for seven pollutants, including SCCPs (labelled as C_{12}, 64% Cl; included short-chain paraffin fractions C_{10} 6%, C_{11} 37%, C_{12} 32%, C_{13} 25%). The 48h-LC_{50} was 8833 mg/kg dw and 0.5 mg/L for exposure in soil and an aquatic medium, respectively. For the aquatic medium, no toxicity was observed at 24 hrs but it was one of the most toxic substances after 48 hours. The greater effect with the longer exposure time is ascribed to more toxicant being taken up with time.

Birds
EC (2000) describes a study in which Mallard ducks, exposed to dietary concentrations of $C_{10–12}$ SCCPs (58% chlorine), were investigated for reproductive effects. The study was conducted over 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 which were not fed the test substance were observed for 14 days. The mean measured concentrations were 29, 168 and 954 mg/kg in diet. 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. Although this decrease was significant, the mean eggshell thickness 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. 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. Since slight effects on reproduction have been seen at 954 mg/kg in diet, the NOAEL is considered as 168 mg/kg in food.

In a study by Ueberschär et al. (2007), hens from 24 to 32 weeks old were fed technical SCCPs (C_{10-13}, 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 77 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.

Mammalian wildlife
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 critical toxicity value (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):

CTV = 100 mg/kg-bw per day × 8 kg-bw/0.8 kg wet wt. per day
   = 1000 mg/kg food wet wt.

Summary of the ecotoxicology of SCCPs
The most sensitive toxicity endpoints for SCCPs are summarized in Table 2-4.

Table 2-4: Overview of the most sensitive long-term ecotoxicity LOEC, NOEC or EC_x data for SCCPs.

Species/endpoint	Effect	NOEC or NOAEL	LOEC or EC_x	Reference
Pelagic organisms (*Daphnia magna*)	Mortality of offspring, 21 days	5 µg/L	8.9 µg/L	Thompson and Madeley (1983a)
Benthic organisms	Equilibrium partitioning based on 21-day study using *Daphnia magna*	NA	35.5 mg/kg dry weight	Environment Canada 2004, based on data from Thompson and Madeley 1983a
Fish (Japanese medaka embryos, early life stage effects)	20-day study, large yolk sac, lethargic or no movement but heart beat present	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 (*Mysidopsis bahia*)	28-day chronic NOEC	7.3 µg/L		Thompson and Madeley 1983b
Folsomia candida (collembola) – soil organisms	Reproduction, 28 day exposure	NA	EC50= 1230 mg/kg dw	Bezchlebová et al. 2007
Microorganisms (bacterial, soil nitrification)	Soil nitrification		EC10 = 570 mg/kg dry wt. nominal	Sverdrup et al. (2006)
Bird – mallard duck	Eggshell thinning	168 mg/kg in diet	954 mg/kg in diet	EC 2000
Mammals - rats	Hypertrophy of the liver and thyroid, increases in liver and kidney weight		100 mg/kg bw/day	IRDC (1984)
Mammals	Hepatic enzyme induction, thyroid hormone system hyperactivity, carcinogenicity, and pup mortality	10 mg/kg/day (and possibly lower at chronic exposure situations)	50 mg/kg/day (and possibly lower at chronic exposure situations)	EU SCCP Risk Assessment Report (EC 2000), CXR Biosciences Ltd., 2006
Mammals – adult otter	Hypertrophy of the liver and thyroid, increases in liver and kidney weight		1000 mg/kg-food wet wt.	Calculated based on IRDC (1984)

3.   Synthesis of Information

Total reported annual usage of SCCPs was high in several countries, but there have been notable reductions in recent years. 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.

Some homologues and isomers of SCCPs are persistent, bioaccumulative and toxic to certain species, and undergo long-range transport to remote areas.

SCCPs are not expected to degrade significantly by hydrolysis in water, and dated sediment cores indicate that they persist in sediment for 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. The Henry's Law Constant indicates that SCCPs will tend to partition from water to air under some conditions, thus facilitating atmospheric transport. Model data (OECD LRTP Screening Tool) indicate that SCCPs have properties similar to known POPs that undergo long range transport. Concentrations measured in biota and sediment from remote Arctic locations also confirm that long-range transport of SCCPs is occurring.

Bioaccumulation factors of 16 440 – 25 650 wet weight 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 Kow and bioaccumulation factors which indicate that SCCPs bioaccumulate. In addition, biomagnification factors for some SCCPs have been found to be greater than 1.

SCCPs have so far been found in Arctic whales, seals, walruses (Tomy et al, 2000), 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 wellrecognised 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 are presented in table 3-1 and 3-2, respectively.

Table 3-1. Comparison by ratio of mean concentrations of SCCPs and POPs in Arctic species. The mean SCCPs 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]/[SCCPs]	[sumDDT/SCCPs]	[Toxaphene/SCCPs]
Beluga whale	19-24	11-18	15
Ringed seals	2.3	1.3	0.9
Walrus	0.4	0.1	0.6

Table 3-2. Comparison of concentrations of SCCPs and POPs (ng/g lipid weight) in Arctic Kittiwake (Reth et al, 2006; AMAP 2004).

Species	[SCCPs]	[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.

Hazard assessments for SCCPs and MCCPs (medium chain chlorinated paraffins) have shown that these structurally very similar substances may also have 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. 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.

Freshwater and marine invertebrates appear particularly sensitive to SCCPs, with a reported chronic NOEC of 5 µg/L for *Daphnia magna* and a chronic NOEC of 7.3 µg/L for the mysid shrimp. Severe liver histopathology was observed in trout, with LOECs ranging from 0.79 to 5.5 µg/g in whole fish tissue.

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.

Table 3-3 presents exposure values for organisms at sites close to likely sources of release, at more distant sites influenced by regional sources, and sites remote from known or significant sources. Also listed are toxicity values relevant to the type of exposure reported – for example, expressed as exposure to the ambient environment, as doses in food, or as body burden.

Table 3-3: Exposures of organisms/receptors at sites close to likely sources of release (Local), more distant sites influenced by regional sources (Regional) and sites distant from known or significant sources (Remote), and relevant toxicity values.

Receptor	Exposure concentration	Sample	Relevant toxicity value	Notes
Local				
Fish-eating mammals (otter)	2.63 mg/kg ww	Carp from Hamilton Harbour, Lake Ontario	1000 mg/kg food ww	– Assuming concentration in carp representative of food consumed. Measured SCCPs in carp collected in Lake Ontario in 1996 and 2001 (Muir et al. 2001; 2002) – See Section 2.5.2 for calculation of relevant toxicity value.
Pelagic invertebrates	44.8 ng/L	STP outfall, Hamilton, ON	8,900 ng/L	– Pelagic invertebrate exposed to STP effluent. 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) – 21-day chronic LOEC value for *Daphnia magna* (Thompson and Madeley, 1983a)
Pelagic fish	2.63 mg/kg ww	Carp from Hamilton Harbour, Lake Ontario	0.79 mg/kg ww	– Comparing concentrations measured in carp in Hamilton Harbour with LOEC for observed severe liver histopathologies, extensive fibrous lesions and hepatocyte necrosis of rainbow trout (Cooley et al., 2001)
Regional				
Pelagic invertebrates	1.19 ng/L	Water, mid-Lake Ontario	8,900 ng/L	– Assuming exposure of pelagic invertebrates to Lake Ontario water. – 21-day chronic LOEC value for *Daphnia magna* (Thompson and Madeley, 1983a)
Pelagic fish	0.123 µg/g ww	Lake trout from mid-Lake Ontario	0.79 mg/kg ww	– Comparing concentrations measured in lake trout in Lake Ontario with LOEC for severe liver histopathologies, extensive fibrous lesions and hepatocyte necrosis of rainbow trout (Cooley et al., 2001)

Receptor	Exposure concentration	Sample	Relevant toxicity value	Notes
Benthic invertebrates	0.41 mg/kg dw	Lake Ontario sediment	35.5 mg/kg dw	– Assuming benthic invertebrate in Lake Ontario. Sediment concentration measured in surface sediments from Lake Ontario (Marvin et al. 2003) – LOEC calculated using the LOEC for *Daphnia magna* using equilibrium partitioning approach (Environment Canada, 2004)
Benthic fish	0.037 mg/kg ww	North Sea dab	0.79 mg/kg ww	– Assuming benthic fish in North Sea. – LOEC for observed severe liver histopathologies, extensive fibrous lesions and hepatocyte necrosis of rainbow trout (Cooley et al., 2001)
Remote				
Benthic invertebrate	0.0176 mg/kg dw	Surface sediment Lake DV09 Canadian Arctic	35.5 mg/kg dw	– Calculated using the LOEC for *Daphnia magna* using equilibrium partitioning approach (Environment Canada, 2004)
Arctic ringed seal	0.52 mg/kg ww	Ringed seal blubber	100 mg/kg ww.	– Results for seals from Tomy et al. 1999

Table 3-4 presents scenarios for exposure of humans to SCCPs. The first two scenarios assume consumption of traditional food diet. Exposure was calculated using SCCP concentrations measured in the Arctic in in ringed seal blubber, beluga whale blubber and walrus blubber (from Tomy et al 1999 and Muir et al. 2004 NCP Synopsis report), and using dietary intake from Kuhnlein (1995) and Kuhnlein et al. (1995). The scenarios for a breast-fed child assumes intake of 750 mL milk/day and 3% lipid content (Van Oostdam et al. 1999).

Table 3-4: Scenarios for exposure of humans, and relevant toxicity values

Receptor	Exposure	Sample	Relevant toxicity value	Comments
Humans (75 kg)	0.15-0.37 µg/kg body wt/d	Estimated dietary exposure Baffin Island male Inuit	125,000 ug/kg body wt/d	LOEL, 2 year rat study (NTP, 1986)
Humans (75 kg)	0.15-0.37 µg/kg body wt/d	Estimated dietary exposure Baffin Island male Inuit	11 ug/kg body wt/d	The 11 ug/kg body wt/d was derived using a safety factor of 1000 applied to a value of 11 mg/kg body weight, based on multistage modelling of tumours with the highest incidence in the carcinogenesis bioassay of male mice that resulted in 5% increase in tumour incidence (IPCS, EHC 181, 1996)

Breast fed child (5 kg)	0.0585 ug/kg body wt/d	Human milk: 13 ug/kg lipid wt	125,000 ug/kg body wt/d	LOEL, 2 year rat study (NTP, 1986)
Breast fed child (5 kg)	0.0585 ug/kg body wt/dd	Human milk: 13 ug/kg lipid wt	11 ug/kg body wt/d	The 11 ug/kg body wt/d was derived using a safety factor of 1000 applied to a value of 11 mg/kg body weight, based on multistage modelling of tumours with the highest incidence in the carcinogenesis bioassay of male mice that resulted in 5% increase in tumour incidence (IPCS, EHC 181, 1996)

In addition, elevated levels of SCCPs in human breast milk in remote communities have been reported.

The International Agency for Research on Cancer considers SCCPs (average C_{12}, 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. 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. 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.

4. Concluding statement

SCCPs satisfy the Annex D screening criteria as per the decision made at POPRC2. In the Risk Profile, additional information, including both modelled and empirical data, is presented and evaluated which elaborates on Annex D information, and addresses the elements of Annex E, including those on exposure in local areas and as result of long range transport.

Evidence indicates that large amounts of SCCPs are still in use and are being released in several countries.

The available empirical and modelled data indicate that SCCPs are persistent, bioaccumulative, and toxic, particularly to aquatic organisms, and undergo long-range environmental transport.

SCCPs have been measured in sediments in Arctic lakes. SCCPs are particularly toxic to aquatic invertebrates. Given the key role that invertebrates play in aquatic ecosystems, there is concern relating to potential for effects of SCCPs on sediment-dwelling and other invertebrates.

SCCPs accumulate in freshwater and marine fish and marine mammals, having been measured in biota in local, regional and remote regions, the latter including measurements from predators such as seals and belugas in the Arctic. Laboratory studies have shown adverse effects on fish at low concentrations. In mammals, SCCPs may affect the liver, the thyroid hormone system, and the kidneys.

Although concentrations in remote areas are low, SCCPs have been 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. SCCPs are measured in human breast milk both in temperate and Arctic populations.

SCCPs have been measured in Arctic biota at concentrations similar to other known POPs. Simultaneous exposure to POPs may be of concern.

Currently, concentrations of SCCPs in remote areas are lower than known concentrations of concern. Available monitoring data do not show that the environmental levels are increasing in concentrations at this time. Given demonstrated long range transport and ability to accumulate, there is potential for increases should releases continue or increase.

SCCPs are considered as POPs pursuant to decisions taken under the UNECE Aarhus (POPs) Protocol to the Convention on Long Range Transboundary Air Pollution (LRTAP).

[Based on the available evidence, it is concluded that…]
[Based on available information, there is inadequate evidence to support the conclusion 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.

[At the same time, given the properties of SCCPs, Parties are encouraged to undertake necessary actions to ensure that the manufacture, processing and use of SCCPs are adequately managed such that low environmental levels are maintained.]

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