

Stockholm Convention on Persistent Organic Pollutants

**Persistent Organic Pollutants Review Committee
(POPRC)**

DRAFT RISK PROFILE

for

Short-chained chlorinated paraffins

Draft prepared by:

The ad hoc working group on short-chained chlorinated paraffins

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Draft Risk Profile for Short-chained chlorinated paraffins

Note:

In accordance with the procedure laid down in Article 8 of the Stockholm Convention, this draft was prepared by the Persistent Organic Review Committee (POPRC) during its intersessional work. Parties and observers to the Stockholm Convention are invited to provide technical and substantive comments on this draft. Comments received will be considered by the ad hoc working group and the revised draft will be made available for the fourth meeting of the POPRC (13-17 October 2008 in Geneva). Please submit your comments to the Secretariat of the Stockholm Convention preferably by e-mail no later than **18 May, 2008** to:

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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.

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

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 homologues of SCCPs (average C₁₂, average 60% chlorination) to be possible carcinogens (groups 2B), although questions have been raised regarding the mechanisms for induction of tumours and the relevance for human health of the studies on which this classification was derived.

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.

To prevent SCCPs from continuing to be released to the environment, it is desirable to ensure global action. Based on the available evidence, it is concluded that SCCPs are likely, as result of their long-range environmental transport, to lead to significant adverse environmental 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). Within the nomination, SCCPs are defined as C10-13 and >48% chlorine by weight in section 1, and then as C10-13 and 1-13 chlorine (~16-78% by weight) in section 1.2.

1.1 Chemical Identity of the Proposed Substance

The CAS No. and EINECS No. for SCCPs (Alkanes, C₁₀₋₁₃, chloro) are 85535-84-8 and 287-476-5, respectively. As presented in the proposal for listing, short-chained chlorinated paraffins are chlorinated derivatives of n-alkanes "that have a carbon chain length of between 10 and 13 carbon atoms and the degree of chlorination more than 48% by weight". Chlorination of the n-alkane feedstock yields extremely complex mixtures, owing to the many possible positions for the chlorine atoms, and standard analytical methods do not permit their separation and identification. Examples of two SCCP structures are presented in Figure 1-1. More detailed information on chemical identity of SCCPs can be found in document UNEP/POPPS/POPRC.3/INF/22.

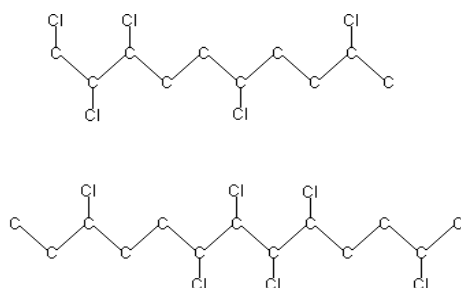


Figure 1-1. Structure of two SCCP compounds (C₁₀H₁₇Cl₅ and C₁₃H₂₂Cl₆).

1.2 Conclusion of the Review Committee Regarding Annex D Information

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

1.3 Data Sources

The risk profile for SCCPs builds on information gathered by the EU in its proposal of SCCPs to the POPRC (UNEP/POPS/POPRC.2/INF/6). The risk profile also incorporates information collected from risk assessment documents prepared by Canada (Environment Canada) and the United Kingdom (DEFRA). Annex E information submissions 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).

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 2.8×10^{-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 1.4×10^{-5} to 0.066 Pa at 25°C (Tomy et al. 1998a). Henry's Law Constants (HLCs) ranged from 0.7 - 18 Pa·m³/mol (Drouillard et al. 1998a), suggesting that SCCPs can remobilise from water to air as a result of environmental partitioning. Measured water solubilities of individual C₁₀₋₁₂ chlorinated alkanes ranged from 400 - 960 µg/L (Drouillard et al. 1998b), while estimated solubilities of C₁₀ and C₁₃ chlorinated alkane mixtures ranged from 6.4 - 2370 µg/L (BUA 1992). 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 Canada (Camford Information Services, 2001) and Germany, the latter stopping production in 1995. Prior to 1995, Clariant, Hoechst, and Huels produced SCCPs in Germany. Hoechst produced between 9,300 – 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, C₆₋₁₈, 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). The Canadian sales pattern for SCCPs (as a proportion of total usage of chlorinated paraffins) is similar to the European sales pattern.

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).

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/formulation site	Insignificant (paint); Low or zero (sealants); 5% solid waste (sealants)	Zitko and Arsenaault (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 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 Arsenaault (1974); BUA (1992)

Application/Use	% by Weight of Final Product	Types of Releases	Amounts Released	Reference
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 × 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 C₁₀₋₁₃ species with an average chlorine content of 40-70%). Information submitted by the USA during the comment period indicated that, for the 2005 TRI data release year, a total of 42,779 pounds (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/cm³ 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/cm³, 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/cm³ 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). However, additional information submitted by the Japanese government suggest that one SCCP congener (1-chlorooctadecane; C = 12, Cl = 1) is readily biodegradable. The applicability of these data to higher chlorinated SCCPs is not known; studies on biodegradation in sediments (see section "Persistence in soil and sediment") indicate that biodegradation may be dependent on the SCCP congener tested. 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.

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 ¹⁴C-labelled C₁₂ chloro-n-alkanes (56% and 69% chlorine) were degraded at 12°C in aerobic sediments used for a study of the bioavailability of SCCPs to oligochaetes. 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 ¹⁴C-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 C₁₀₋₁₃, 65% chlorine by weight product were estimated to be 1630 days in freshwater sediments and 450 days in marine sediments under aerobic conditions. Little or no mineralization was noted in anaerobic sediments.

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 C₁₁ and C₁₂.

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 C₁₂H₁₈Cl₈ (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 K_{ow} 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 C₁₂H_{20.1}Cl_{5.9}, 55.9% wt. Cl and C₁₂H_{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 K_{ow} 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.

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, C₁₂H₁₆Cl₁₀ and C₁₂H₂₀Cl₆, 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 C₁₂H₂₀Cl₆ (56% chlorine by weight) and C₁₂H₁₆Cl₁₀ (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 C₁₂H₁₆Cl₁₀ to an average of 6.8 for C₁₂H₂₀Cl₆. 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.

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 (C₁₂) 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 (C₁₃) because of their low water concentrations. These BMF values suggest that SCCPs, especially the chlorinated decanes and dodecanes, are not biomagnifying in the pelagic food web of Lake Ontario. However, a higher BMF for SCCPs in general was determined between lake trout and alewife alone (BMF 0.91) (Muir et al. 2003). Relatively high concentrations of SCCPs in sculpin (*Cottus cognatus*) and diporeia (*Diporeia* sp.) imply that sediments are an important source of SCCPs for bottom feeders (Muir et al. 2002).

Biomagnification of SCCPs in the Lake Ontario food web was examined by Muir et al. (2003). BMFs for the C₁₂ and C₁₃ SCCPs were higher for prey to lake trout compared to C₁₁ and C₁₀ SCCPs. BMFs exceeded 1 for alewife to lake trout for the C₁₂ and C₁₃ SCCPs, as did the BMF for diporeia to sculpin for all chain lengths (C₁₀₋₁₃).

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/m³ in air collected in the high Arctic (Alert, Ellesmere Island), while Borgen et al. (2000) measured SCCPs ranging from 9.0-57 pg/m³ at Mt. Zeppelin, Svalbard, Norway in 1999. SCCPs have also been measured in the sediments of remote Arctic lakes (Tomy et al. 1999; Stern and Evans 2003) far from local sources of contamination, 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 C₁₀ and C₁₁ formula groups (Tomy et al. 2000), some of the more volatile components of SCCP mixtures (Drouillard et al. 1998a), suggesting that these compounds are more likely to be transported long distances. This is in agreement with the results of Reth et al. (2005, 2006), who found an enrichment of C₁₀ SCCPs in biota in the North Sea compared to the Baltic Sea (Reth et al. 2005) and in the Arctic as compared to the Baltic Sea (Reth et al. 2006). Supporting this are modelled results indicating that the atmospheric half-lives for the major SCCP homologues observed in environmental samples, such as the Great Lakes and Arctic air and biota (C₁₀H₁₇Cl₅, C₁₀H₁₆Cl₆, C₁₀H₁₅Cl₇, C₁₁H₁₈Cl₆, C₁₁H₁₇Cl₇, C₁₂H₂₀Cl₆, C₁₂H₁₉Cl₇), are greater than 2 days (Section 2.2.1).

A comparison of vapour pressures (VPs) and Henry Law Constants (HLCs) demonstrate that SCCPs have VPs (2.8×10^{-7} to 0.028 Pa) and HLCs (0.68–18 Pa·m³/mol for C₁₀₋₁₂ 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.

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/m³ 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/m³ 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/m³.

Concentrations of SCCPs in air samples collected at Egbert, Ontario, Canada, in 1990 ranged from 65 to 924 pg/m³ (Tomy 1997; 1998a). Concentrations of SCCPs over Lake Ontario in 1999 and 2000 ranged from 120 to 1,510 pg/m³ (Muir et al. 2001; D.C.G. Muir, unpublished data, 2001).

Peters et al. (2000) reported a mean SCCP concentration of 99 pg/m³ 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/m³

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

(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 Narbuvooll). 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 $\mu\text{g/kg}$ 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. Catharines 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.

SCCPs have also been measured in sewage sludge. Stevens et al. (2002) found SCCP concentrations ranging from 6.9 to 200 $\mu\text{g/g}$ 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 $\mu\text{g/g}$) 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). 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, because of the similarity of the formula group abundance profile to that of PCA-60, a commercial SCCP that was used as the external standard. 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 $\mu\text{g/L}$ between chain lengths).

2.4.4 Sediments

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

Fluxes for surface slices of sediment core ($\mu\text{g/m}^2$ 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 $\mu\text{g/kg}$ dry weight in sediment 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 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 $\mu\text{g/kg}$ dry wt. The 83 $\mu\text{g/kg}$ 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 in 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).

Příbylová et al. (2006) reported concentrations of SCCPs in 36 sediment samples from 11 Czech rivers and five drainage vents near industrial areas. The concentrations ranged from non-detect to 347.4 ng/g dry weight. Sediment samples from the Czech Republic were analyzed for SCCPs in three locations having different industrial discharges during 2001 and 2002. SCCPs sediment concentrations varied between 24 and 45.78 ng/g dry weight in the Košetice area (reference area), 16.30 to 180.75 ng/g dry weight in the Zlín area (rubber, tanning and textile industries) and 4.58 to 21.57 ng/g dry weight in the Beroun area (cements and machinery industry) (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).

2.4.5 Biota

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

Fish

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. C₁₂ SCCPs predominated in lake trout, whereas C₁₁ was the major SCCP in sculpin and smelt.

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 (C₁₀-C₁₃) ranging between 7 - 206 µg/kg of fish oil and 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 (UNECE-LRTAP POPs Protocol, 2007)

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) found concentrations of SCCPs ranging from 49 to 820 µg/kg lipid for the same two cities.

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).

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. A tolerable daily intake (TDI) for SCCPs of 100µg/kg-bw per day is given by IPCS (1996).

An independent technical peer review on SCCPs submitted under the UNECE-LRTAP POPs Protocol indicated that , aboriginal people living in the Arctic and consuming contaminated animals may be exposed to SCCPs at concentrations greater than the WHO health guideline of 11 µg/kg bw for neoplastic effects (tumor formation) (UNECE-LRTAP POPs Protocol, 2007)

2.5 Hazard Assessment for Endpoints of Concern

2.5.1 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. In a 13-week oral rat study by IRDC (1984), increases in liver and kidney

weight and hypertrophy of the liver and thyroid occurred at doses of 100 mg/kg-bw per day. The No-Observed-Effect Level (NOEL) was 10 mg/kg-bw per day.

The EU Risk Assessment Report (EC 2000) summarized the effect of SCCPs in mammalian species. SCCPs have the potential to cause minimal skin irritation, are of low acute toxicity in animals and are not mutagenic. Rodent studies showed dose related increases in adenomas and carcinomas in the liver, thyroid, and kidney. They concluded that carcinogenicity observations in the liver and thyroid in mice and the benign tumours in the kidney of male rats are not likely to be relevant for human health. Yet consequences for humans could not be ruled out.

The Scientific Committee on Toxicity, Ecotoxicity and the Environment found that the alveolar/bronchiolar carcinomas in male mice should not totally be discounted and that the finding of lung tumours in male mice may be of importance for humans (CSTEE 1998).

The International Agency for Research on Cancer considers there is sufficient evidence for the carcinogenicity (possibly carcinogenic – groups 2B) of a commercial chlorinated paraffin product of average carbon-chain length C₁₂ and average degree of chlorination 60% in experimental animals (IARC, 1990). However, there continues to be contention over the mechanisms of these tumours and whether they are relevant for human health (EC 2000). SCCPs are classified in the EU as a Carcinogen Category 3: R40 – Limited evidence of a carcinogenic effect, and are similarly classified as hazardous in Australia.

There are no data on fertility or developmental effects for humans. 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).

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

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 C₁₀₋₁₃ CP and 0.05 mg/L for a 62% chlorine C₁₀₋₁₃ 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 C₁₀, C₁₁ and C₁₂ SCCP compounds (single chain lengths with mixtures of isomers) to Japanese medaka (*Oryzias latipes*) embryos. Lowest-Observed-Effect Concentrations (LOECs) ranged from 55 µg/L for C₁₂H₂₀Cl₇ to 460 µg/L for C₁₀H₁₆Cl₇. Effects in eggs and larvae over the first 20 days after laying included large yolk sac, lethargic or no movement but heart beat present. These effects were observed in C₁₀₋₁₂ of SCCP, and no dose-response in C₁₂, 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 ¹⁴C-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 C₁₀, C₁₁ and C₁₂ 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 C₁₀H₁₅Cl₇ and C₁₁H₁₈Cl₆ (whole fish concentrations of 0.92 and 5.5 µg/g wet wt., respectively), consisting of extensive fibrous lesions and hepatocyte necrosis not seen in controls or lower exposed fish. No thyroid lesions were observed. LOECs for the C_{10–12} SCCPs ranged from 0.79 to 5.5 µg/g in whole fish tissue following dietary exposure to concentrations ranging from 0.84 to 74 µg/g in food.

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 (C₁₂ 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 LC₅₀ (adult survival), and EC₅₀ and EC₁₀ (reproduction) values of 5733 mg/kg, 1230 mg/kg, and 660 mg/kg dry wt. 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 EC₁₀ 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 C₁₂, 64% Cl; included short-chain paraffin fractions C₁₀ 6%, C₁₁ 37%, C₁₂ 32%, C₁₃ 25%). For SCCPs, 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. The sensitivity was similar to that reported by Bezchlebová et al. (2007) for *Folsomia candida*, but lower than that for *Eisenia fetida* and *Enchytraeus albidus*.

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.

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 effected, 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.

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	LOEC or EC _x	Reference
Pelagic organisms (<i>Daphnia magna</i>)	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 <i>Daphnia magna</i>	NA	35.5 mg/kg dry weight	Environment Canada 2004, based on data from Thompson and Madeley 1983a
Soil organisms	Equilibrium partitioning based on 21-day study using <i>Daphnia magna</i>	NA	35.5 mg/kg dry weight	Environment Canada 2004, based on data from Thompson and Madeley 1983a
Microorganisms (bacterial, soil nitrification)	Soil nitrification		EC10 = 570 mg/kg dry wt. nominal	Sverdrup et al. (2006)

3 Synthesis of Information

Total reported annual usage of SCCPs was high in several countries, but there have been notable reductions in recent years, including in Canada, Switzerland and Australia. 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 (BAFs) 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 K_{ow} and bioaccumulation factors which indicate that SCCPs bioaccumulate. In addition, biomagnification factors for some SCCPs have been found to be greater than 1. High concentrations of SCCPs in upper trophic level organisms, notably in marine mammals and aquatic freshwater biota (e.g., beluga whales, ringed seals and fish), are additional evidence of bioaccumulation. SCCPs have also been measured in the breast milk of Inuit women in Northern Quebec, as well as of women in the United Kingdom.

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 homologues of SCCPs (average C₁₂, average 60% chlorination) to be possible carcinogens (groups 2B), although questions have been raised regarding the mechanisms for induction of tumours and the relevance for human health of the studies on which this classification was derived.

Because of the complexities associated with the chemical analysis of chlorinated paraffins, there is a limited number of measurements from the environment, most of them being from the past decade. 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. Concentrations currently measured at those sites are generally below levels that have been associated with effects in laboratory studies. For pelagic, benthic, and soil dwelling organisms, the maximum reported environmental concentrations are approximately 50-200 times lower than the most sensitive toxicity values. However, these margins of exposure do not incorporate any safety factors.

In biota, SCCPs have been measured in North America and Europe. At some urban sites with high exposure, measured concentrations of SCCPs were within the range of concentrations causing effects: concentrations in carp from Hamilton

Harbour and in yellow perch from the Detroit River were similar to those associated in the laboratory with histopathological effects in juvenile rainbow trout. SCCPs were also measured in a range of marine fish species, farther from potential point sources, with concentrations as high as 750 µg/kg in cod liver. In the Arctic SCCPs have been measured in the blubber of marine mammals from several locations, at concentrations of 95 to 626 ng/g.

4 Concluding Statement

In summary, the increasing regulation of SCCPs has resulted in a decrease in SCCPs currently in use. However, evidence suggests that significant amounts are still in use and 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.

To prevent SCCPs from continuing to be released to the environment, it is desirable to ensure global action. Based on the available evidence, it is concluded that SCCPs are likely, as result of their long-range environmental transport, to lead to significant adverse environmental effects, such that global action is warranted.

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