

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

May, 2007

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 Pollutants Review Committee (POPRC) during its inter-sessional 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 third meeting of the POPRC (19-23 November in Geneva). Please submit your comments to the Secretariat of the Stockholm Convention preferably by e-mail before **July 1, 2007** to:

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**Stockholm Convention POPs Review Committee
SCCPs Intersessional Working Group**

SCCPs – Draft Risk Profile – 2007-05-29

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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 also relatively persistent in air. The Henry's law constant indicates that there may be substantial partitioning from water to air under certain conditions, thus facilitating atmospheric partitioning and transport. SCCPs have been detected in a diverse array of environmental samples (air, sediment, water, wastewater, fish and marine mammals) and in remote areas such as the Arctic (which is additional evidence of long range transport). In addition, Arctic Contamination Potential (ACP) modeling and OECD LRTP screening tools suggests that SCCPs have moderate ACP when emitted to air and have properties similar to known POPs that undergo long range transport.

Bioaccumulation factors (BAFs) of 16 440–25 650 wet weight (wet wt.) in trout from Lake Ontario indicate that SCCPs can bioaccumulate to a high degree in aquatic biota. This is supported by modeling 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. High concentrations of SCCPs in upper trophic level organisms is additional evidence of bioaccumulation. Evidence for the bioaccumulation of SCCPs is further supported by the high concentrations of SCCPs measured in marine mammals and aquatic freshwater biota (e.g., beluga whales, ringed seals and various fish). High concentrations of SCCPs have also been measured in the breast milk of Inuit women in Northern Quebec.

There is evidence that SCCPs are toxic to sensitive aquatic organisms at relatively low concentrations. The most sensitive organism, *Daphnia magna*, has chronic NOECs of 5 µg/L.

The weight of evidence supports the conclusion that SCCPs are persistent, bioaccumulative, inherently toxic to some species, and undergo long range transport to remote areas. Concentrations of SCCPs currently measured in the environment are generally below those associated with effects. However, because of their widespread distribution, persistence and accumulation, they continue to have potential for long-term harmful effects.

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

Concentrations currently measured in the environment are generally below levels that have been

associated with effects in laboratory studies. In some cases, concentrations are approaching those that may be of concern, for example in secondary consumers, and elevated levels have been measured in human breast milk, including in remote communities. Particularly in view of SCCPs persistence, bioaccumulation and their inherent toxicity to a range of organisms, it is considered that SCCPs are likely to cause significant adverse effects as a result of long range transport.

Based on the available evidence, it is thus likely that SCCPs can, as result of long range environmental transport, cause significant adverse effects on human health and/or the environment, such that global action is warranted.

1. INTRODUCTION

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

1.1 Chemical Identity of the Proposed Substance

[Note that an INF Document was drafted regarding the proposed identity of this substance.]

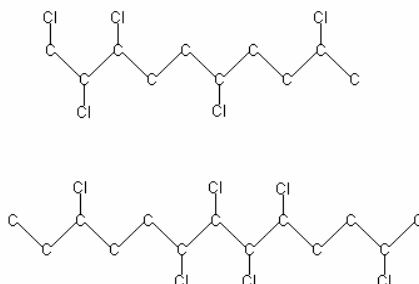
IUPAC Name: Alkanes, C₁₀₋₁₃, chloro
CAS No: 85535-84-8
EINECS No: 287-476-5

Synonyms

chlorinated alkanes (C₁₀₋₁₃)
chloro (50-70%) alkanes (C₁₀₋₁₃)
chloro (60%) alkanes (C₁₀₋₁₃)
chlorinated paraffins (C₁₀₋₁₃)
polychlorinated alkanes (C₁₀₋₁₃)
paraffins chlorinated (C₁₀₋₁₃)

Short chain chlorinated paraffins (SCCPs) are chlorinated derivatives of n-alkanes, having carbon chain lengths ranging from 10 to 13 and 1-13 chlorine atoms (Molecular formula: C_xH_(2x-y+2)Cl_y, where x=10-13 and y=1-13). 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. Thus the commercial mixture would fall under the proposed identity for SCCPs specified here.

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



1.2 Conclusion of the Review Committee Regarding Annex D Information

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

1.3 Data Sources

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

1.4 Status of the Chemical under International Conventions

In August, 2005, the European Community proposed SCCPs to be added to the UNECE Convention on Long Range Transboundary Air Pollution, Protocol on Persistent Organic Pollutants (LRTAP). SCCPs were proposed to meet the criteria of decision 1998/2 of the Executive Body for persistence, potential to cause adverse effects, bioaccumulation and potential for long range transport. At the 24th session of the Executive Body in December 2006, the Parties to the UNECE POPs Protocol agreed that SCCPs should be considered as a POP as defined under the Protocol, and requested that the Task Force continue with the Track B reviews of the substances and explore management strategies for them.

In 1989, as a result of laboratory testing in animals, SCCPs were classified as a group 2B carcinogen by the International Agency (IA) for Research on Cancer (IARC).

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 should be prohibited by the end of 1999. Exemptions will allow 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).

SCCPs have been identified as priority hazardous substances in the field of water policy under the Water Framework Directive (Directive 2000/60/EC of 23 October 2000) and are listed in the draft amendment of Directive 2000/60/EC, which defines water quality standards for European surface waters. Substances listed in this Directive will be subject to cessation or phasing out of discharges, emissions and losses with in an appropriate time table that shall not exceed 20 years (EC, 2005).

The most important uses (metal working fluids and leather fat liquors) in the EU were restricted in directive 2002/45/EC. SCCP in plastics is a major use in Europe that was not covered by directive 2002/45/EC.

2. SUMMARY INFORMATION RELEVANT TO THE RISK PROFILE

2.1 Sources

2.1.1 Production

Total reported annual usage of all chlorinated paraffins (CPs) in Canada (production + imports – exports) 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, rather than the North American sales pattern, which is dominated by the United States (Table 1).

Whether these sales patterns are the same at present is not known. North American demand for total CPs fluctuates depending on the strength of the economy (Camford Information Services 2001). 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: Sales of CPs in the EU and North America during the 1990s

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

¹ OSPAR (2001).

² CPIA (2000).

In addition to production in the US and the EU, it should be noted that there are CP's (of various chain length) producers in Russia, India, Taiwan, China and Japan. In some cases, total CPs are produced in Asia under licence to the European manufacturer. It is unclear to what extent imports from these countries are accounted for in the information provided by industry associations such as Eurochlor and CPIA (see Table 1). There is no production of SCCPs in Canada (Camford Information Services 2001).

Information submitted under Annex E of the Stockholm convention indicated that SCCPs were produced in the former Czechoslovakia (Novaky, Slovakia), though quantities are not known. Japan also noted that there is a possibility of 1% in medium chained chlorinated paraffin production. Germany noted that there has been no production in Germany since 1995. Prior to 1995, Clariant, Hoechst, and Huels produced SCCPs in Germany. Hoechst produced between 9300 - 19300 tonnes/year in Germany between the years 1993 and 1995.

As noted in the Annex E information submitted by the USA, some chlorinated paraffins are on the Toxic Substances Control Act (TSCA) inventory and are subject to the Environmental Protection Agency's (EPA's) TSCA inventory update reporting rule under which production and import information is collected. The CAS numbers used in the United States are not specific to SCCPs, hence the information collected includes other chain-length chlorinated paraffins. For 2002, the production and import volumes reported for CAS# 63449-39-8 (paraffin waxes and hydrocarbon waxes, chloro) were in the range of >50 million – 100 million pounds (>23 million – 45 million kg),

and for CAS # 61788-76-9 (alkanes, chloro; chloroparaffins) in the range of >50 million – 100 million pounds (>23 million – 45 million kg). In 1994, for CAS # 68920-70-7, (alkanes, C₆₋₁₈, chloro) production and import volume in the range of >1 million – 10 million pounds (>0.45 million – 23 million kg) were reported.

Annex E information submitted by Brazil indicates that 150 tons/year of SCCPs are produced in Brazil.

2.1.2 Uses

Nearly all reported usage of SCCPs in Canada was for metalworking applications. Minor uses included use as a flame retardant in plastics and rubber. European use pattern data for SCCPs from the years 1994 and 1998 are given in Table 2.

Table 2: Applications of SCCPs in Europe

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

¹ Data from Euro Chlor (1995).

² Data from OSPAR (2001) from Western Europe.

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

The use of SCCPs in the EU in metalworking (and also in fat liquoring of leather) is now subject to marketing and use restrictions. EU Directive 2002/45/EC, which was adopted in June 2002, restricts the concentration of SCCPs in metalworking and leather fat liquoring preparations to 1% or less. The use of SCCPs in these applications has decreased significantly since the release estimates initially used in the European risk assessment of SCCPs (EC 2000) were obtained (U.K. Environment Agency 2003c).

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

As noted in Annex E submission by Germany, the most important uses (74% of the total) of SCCPs were banned by the EU directive 2002/45/EC. SCCPs have been used as a PCB substitute in gaskets

(e.g. splices, in buildings) and this may be a source when buildings are renovated.

Annex E information submitted by Brazil indicates that 300 tons/year is used in Brazil for the purposes of flame retardant in rubber, car carpet and accessories. It was noted that leather processing and use in paints was not relevant.

Use of SCCPs in Australia decreased by 80% between 2001 to 2003 to approximately 25 tonnes per annum of SCCPs in the metal working industry (NICNAS, 2004).

Plastics and Rubber

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

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

Adhesives/sealants

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

Paints

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

2.1.3 Releases to the environment

There is currently no evidence of any significant natural source of CPs (U.K. Environment Agency 2003b). Anthropogenic releases of CPs into the environment may occur during production, storage, transportation, industrial and consumer usage of CP-containing products, disposal and burning of waste, and land filling of products such as PVC, textiles, painted materials, paint cans and cutting oils (Tomy et al. 1998a). The possible sources of releases to water from manufacturing include

spills, facility wash-down and storm water runoff. CPs 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.

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

Data since 1999 reported to Canada's National Pollutant Release Inventory (NPRI) found that very small amounts of CPs (short, medium and long chain) are being released to the Canadian environment by companies that meet the NPRI reporting requirements (Environment Canada, 2005). In 2001-2002, the NPRI found 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 found 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 (C10-13). As described in the USA Annex E submission, a total of 62023 lb (28133 kg) was reported for on and off site disposal or other releases, 4220 lb (1914 kg) of which were releases to air and water (TRI, 2004 data for all US industries in the polychlorinated alkanes category, data release April 12, 2006).

Releases from production

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

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

Releases from formulation of metalworking fluids

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

emission factors for all CPs for the formulation of metalworking fluids are 0.005% to air and 0.25% to wastewater before any on-site treatment, which small blending businesses may not have.

Releases from production of rubber

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

Releases from production of textiles and polymeric materials

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

Releases from production of paints and sealants

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

Releases from use of CP-containing products

Releases from PVC, paints, adhesives and sealants

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

Losses from volatilization

Rubber

SCCPs are reported to be used in rubber, with applications mostly in high-density conveyor belts (EC 2000). Releases from use of SCCPs in polymers such as rubber or PVC may also occur via volatilization or from loss of polymeric material as particles during wear and abrasion of the products. Belts may also be recycled into other products at the end of their life. Volatilization of

0.05% of the MCCPs during the lifetime of the product was assumed, and it is reasonable to assume that this would be the case for SCCPs as well.

Losses from leaching

Paints and varnishes

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

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

Releases from use of metalworking fluids

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

Table 3: Total losses of CPs for a large and small machine shop using oil-based cutting fluids (from EC 2000; U.K. Environment Agency 2001, 2003b)

	Large facility with swarf reprocessing		Small facility with no swarf reprocessing	
Misting/evaporation	2%	to air	2%	to air
Overalls	1%	to water	2%	to water
Leaks	1%	to water ¹	3%	to water ¹
Dragout/swarf	27%	incinerated	81%	incinerated
	3%	to landfill	9%	to landfill
Dragout/workpiece	1%	to water	1%	to water
	2%	chemical waste	2%	chemical waste
Internal reprocessing	1%	to water ¹		
External reprocessing	10%	reused/discarded as waste oil		
Total losses	48%		100%	

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

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

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

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

Releases from leather fat liquid and carbonless copy paper

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

Releases from other uses

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

Releases during disposal

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

The releases and bioavailability of CPs from polymers that are landfilled or from losses of polymeric material as particles during wear and abrasion of flooring, rubber products, etc., are unknown. These releases could be sources of input of CPs to air and soils in urban and industrial areas (U.K. Environment Agency 2001, 2003a,b). Polymer-incorporated CPs could also be released during recycling of plastics, which may involve processes such as chopping, grinding and washing. If released as dust from these operations, the CPs would be adsorbed to particles because of high sorption and octanol-air partition coefficients.

Overall emissions in Europe and estimates for North America

Fractional losses of SCCPs to wastewater and surface waters have been estimated based on EU data (EC 2000) and are summarized in Table 4. Behaviour similar to that of MCCPs (U.K. Environment Agency 2003b) is assumed. Overall most releases of SCCPs are expected to be associated with metal working operations, however there is potential for widespread release in small amounts associated with uses in products (e.g. paints, textiles, rubber).

Table 4: Estimated fractional losses of SCCPs in the EU to wastewaters, surface waters and the terrestrial environment

Application	Release to each compartment		
	Wastewater¹	Surface water²	Terrestrial³
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.

2.2 Environmental Fate

2.2.1. Persistence

Persistence in Air

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

SCCPs were detected in four individual samples of air 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 (see Table 11). The occurrence of SCCPs at a remote site like Alert indicates that the substance is subject to long range transport. The occurrence of long range transport is one line of evidence suggesting that SCCPs are persistent chemicals.

Tomy et al. (2000) reported SCCPs in the blubber of ringed seal from Eureka, southwest Ellesmere Island, beluga whales from northwest Greenland and the Mackenzie Delta and walrus from northwest Greenland at concentrations ranging from 199 to 626 ng/g wet wt. Tomy et al. (2000) also observed that the concentration profiles for the Arctic marine mammals show a predominance of the shorter carbon chain length congeners, i.e., the C₁₀ and C₁₁ formula groups. Drouillard et al. (1998a) showed that these congeners are the more volatile components of SCCP mixtures, which show a trend of decreasing VPs with increasing carbon chain length and degree of chlorination. SCCPs have also been detected in sediments from Hazen Lake and Lake DV09 in the Arctic, at levels of 7 and 17 ng/g dry wt., respectively (Tomy et al. 1998a; Stern and Evans 2003).

Although CPs do not degrade by direct photolysis in air, they theoretically would be subject to attack via hydroxyl radicals in the troposphere (Bunce 1993). Estimated atmospheric half-lives for SCCPs based on reaction with hydroxyl radicals are shown in Table 5 and 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). In the risk assessment methodology used in the EU, a lower hydroxyl radical concentration of 5×10^5 molecules/cm³ is generally used as a daily (24-hour) average, which is typically found in relatively unpolluted air. Use of the latter hydroxyl radical concentration results in atmospheric half-lives ranging from 1.2 to 15.7 days (Table 5). 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 Canadian conditions, may limit the atmospheric oxidation pathway.

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

Table 5: Estimated atmospheric half-lives for SCCPs calculated using the Syracuse Research Corporation AOPWIN computer program

Example structure	Chlorine content (% by weight)	Estimated k_{OH} ($\text{cm}^3/\text{molecule per second}$)	Estimated atmospheric half-life (days)	
			$[OH]^1 = 1.5 \times 10^6$ molecules/ cm^3	$[OH] = 5 \times 10^5$ molecules/ cm^3
$\text{C}_{10}\text{H}_{21}\text{Cl}$	20.1	9.75×10^{-12}	1.1	1.6
$\text{C}_{10}\text{H}_{20}\text{Cl}_2$	33.6	8.16×10^{-12}	1.3	2.0
$\text{C}_{10}\text{H}_{19}\text{Cl}_3$	43.4	6.57×10^{-12}	1.6	2.4
$\text{C}_{10}\text{H}_{18}\text{Cl}_4$	50.7	5.17×10^{-12}	2.1	3.1
$\text{C}_{10}\text{H}_{17}\text{Cl}_5$	56.4	5.22×10^{-12}	2.0	3.1
$\text{C}_{10}\text{H}_{16}\text{Cl}_6$	61.0	3.77×10^{-12}	2.8	4.3
$\text{C}_{10}\text{H}_{15}\text{Cl}_7$	64.8	3.68×10^{-12}	2.9	4.4
$\text{C}_{10}\text{H}_{14}\text{Cl}_8$	68.0	2.63×10^{-12}	4.1	6.1
$\text{C}_{10}\text{H}_{13}\text{Cl}_9$	70.6	1.59×10^{-12}	6.7	10.1
$\text{C}_{10}\text{H}_{12}\text{Cl}_{10}$	72.9	1.02×10^{-12}	10.5	15.7
$\text{C}_{13}\text{H}_{27}\text{Cl}_1$	16.2	13.2×10^{-12}	0.81	1.2
$\text{C}_{13}\text{H}_{26}\text{Cl}_2$	28.0	10.9×10^{-12}	0.98	1.5
$\text{C}_{13}\text{H}_{25}\text{Cl}_3$	37.0	9.35×10^{-12}	1.1	1.7
$\text{C}_{13}\text{H}_{24}\text{Cl}_4$	44.1	7.76×10^{-12}	1.4	2.1
$\text{C}_{13}\text{H}_{23}\text{Cl}_5$	49.8	7.11×10^{-12}	1.5	2.3
$\text{C}_{13}\text{H}_{22}\text{Cl}_6$	54.5	5.94×10^{-12}	1.8	2.7
$\text{C}_{13}\text{H}_{21}\text{Cl}_7$	58.4	4.96×10^{-12}	2.2	3.2
$\text{C}_{13}\text{H}_{20}\text{Cl}_8$	61.7	4.87×10^{-12}	2.2	3.3
$\text{C}_{13}\text{H}_{19}\text{Cl}_9$	64.6	4.39×10^{-12}	2.4	3.7
$\text{C}_{13}\text{H}_{18}\text{Cl}_{10}$	67.1	3.34×10^{-12}	3.2	4.8
$\text{C}_{13}\text{H}_{17}\text{Cl}_{11}$	69.3	2.30×10^{-12}	4.7	7.0
$\text{C}_{13}\text{H}_{16}\text{Cl}_{12}$	71.3	1.40×10^{-12}	7.7	11.5
$\text{C}_{13}\text{H}_{15}\text{Cl}_{13}$	73.0	1.31×10^{-12}	8.2	12.2

¹ Assumes 12 hours of sunlight per day.

The major SCCP structures observed in environmental samples, such as in Great Lakes and Arctic air and biota — $\text{C}_{10}\text{H}_{17}\text{Cl}_5$, $\text{C}_{10}\text{H}_{16}\text{Cl}_6$, $\text{C}_{10}\text{H}_{15}\text{Cl}_7$, $\text{C}_{11}\text{H}_{18}\text{Cl}_6$, $\text{C}_{11}\text{H}_{17}\text{Cl}_7$, $\text{C}_{12}\text{H}_{20}\text{Cl}_6$, $\text{C}_{12}\text{H}_{19}\text{Cl}_7$ — all have estimated atmospheric half-lives greater than 2 days. Furthermore, these structures predominate in SCCP products analyzed by Tomy (1997).

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

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. Furthermore, Koh and Thiemann (2001) showed that SCCP mixtures underwent rapid photolysis in acetone–water under ultraviolet light (mercury arc lamp; >254 nm) with half-lives of 0.7–5.2 hours. The half-life of a 52% chlorine by weight SCCP product in pure water under the same conditions was 12.8 hours. Photoproducts included n-alkanes. The use of a 254-nm wavelength irradiation source in these studies may have produced far shorter half-lives than under natural light conditions. These results suggest that sunlight photolysis may be a significant degradation pathway for some SCCPs.

Persistence in soil and sediment

SCCP residues were found in the surficial sediments of the following remote Arctic lakes (reported in ng/g dry wt.): Yaya Lake (1.6), Hazen Lake (4.5) and Lake DV-09 (17.6) (Table 16). 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. 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 (Figure 6).

In the absence of information on loading for any of the years at any of these locations, it is not possible to calculate a discrete half-life value from these data. However, the fact that SCCP residues were detected in sediment cores dating back to the 1940s at these locations is convincing evidence that SCCPs can persist for long periods in sediment. Residues observed in slices dated earlier can be explained by contamination as an artefact of core sampling (Muir 2000). 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, which is the criterion for persistence in sediment in the Persistence and Bioaccumulation Regulations of CEPA 1999. 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.

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

Several government assessments and published reviews have concluded that slow biodegradation in the environment may be expected to occur, particularly in the presence of adapted micro-organisms (Government of Canada 1993a,b; Tomy et al. 1998a; EC 2000). However, no information is currently available on anaerobic biodegradation. There has been relatively little progress in development of information on degradation of SCCPs in sediments since the mid-1990s.

Madeley and Birtley (1980) used 25-day biochemical oxygen demand (BOD) tests to examine the biodegradation of a range of CPs with different carbon chain lengths and chlorine contents. Acclimatized micro-organisms showed a greater ability to degrade CPs than did organisms

normally used for treating domestic sewage, and increasing chlorination inhibited biodegradation. An SCCP with 49% chlorine appeared to be rapidly and completely degraded by acclimatized micro-organisms after 25 days. No significant oxygen uptake was observed in tests using the highly chlorinated CPs, which included two SCCPs (60% and 70% chlorine) and one MCCP (58% chlorine).

Fisk et al. (1998a) found that two ^{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. The biodegradation in sediments was estimated by comparing the amount of toluene-extractable and toluene-non-extractable ^{14}C after 18 and 32 days of incubation. It was hypothesized that the toluene-non-extractable ^{14}C fraction represented a polar degradation product of the CPs, which was tightly bound to the sediments and therefore non-extractable using toluene. Half-lives in sediment were 12 ± 3.6 days and 30 ± 2.6 days for the 56% and 69% chlorine products, respectively. No degradation product identification was performed. This study, therefore, would not provide any indication of the mineralization or ultimate degradation half-life of SCCPs.

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

Omori et al. (1987) studied the CP dechlorination potential of a series of soil bacterial strains acting on $\text{C}_{12}\text{H}_{18}\text{Cl}_8$ with 63% chlorine. Although they could not isolate a bacterial strain that could use CP as a sole carbon source, they did find that different strains pre-treated with n-hexadecane had different dechlorination abilities. A mixed culture (four bacterial strains) released 21% of the chlorine after 48 hours. The bacterial strain HK-3, acting alone, released 35% of the chlorine after 48 hours. Omori et al. (1987) suggested that the mechanism for the aerobic degradation was a β -oxidation enzyme system, which would first oxidize the terminal methyl group to produce chlorinated fatty acids and then break down the acids to 2- or 3- chlorinated fatty acids.

Concentration profiles of SCCPs in sediments from Lake Winnipeg, Fox Lake in the Yukon, the west basin of Lake Ontario (see Figure 6) and Lake DV09 (see Figure 8) indicate that SCCP residues were present in the 1940s (Muir et al. 1999; Tomy et al. 1999). Some transformation of SCCP chain length and formula groups may have occurred over time (i.e., with depth) in these cores, judging by the appearance of greater proportions of shorter chain lengths and lower chlorinated alkanes at shallower depths (Tomy et al. 1999). Tomy et al. (1999) hypothesized that this may be a result of rapid aerobic degradation occurring at these depths. However, these shifts may also be due to changing product formulations at upstream manufacturing and waste treatment sources. In the absence of information on loading for any of the years at any of these locations, it is not possible to calculate specific half-lives from these data. However, the fact that SCCP residues were detected in sediment cores at these dates is convincing evidence that SCCPs can persist for more than 50 years in subsurface anaerobic sediments. Nicholls et al. (2001) were unable to detect SCCPs ($<0.1 \mu\text{g/g}$) in farm soils in the United Kingdom on which several applications of sewage sludge had been applied. However, the study did not specifically follow the fate of SCCPs over time

following sludge application.

2.2.2. Bioaccumulation

2.2.2.1 Modeled Log K_{OW} and Bioaccumulation Factors

It was calculated that log K_{OW} ranged between 4.8 and 7.6 for all possible SCCP congeners using the equation developed by Sijm and Sinnige (1995). QSAR based models (e.g. EPI KowWin) were used to compare to this empirical model but the predictions (ranging 5.4 – 21.6) are unreliable due to the results exceeding the limits of the training set. However the large amounts of high Kow readings suggest a large potential for bioaccumulation. The few experimental data available also fit with the modelled range. Fisk et al. (1998b) determined the octanol-water partition coefficients of two ^{14}C -labelled short-chain chlorinated paraffins of single carbon chain length (C_{12}). The two compounds used were $C_{12}H_{20.1}Cl_{5.9}$, 55.9% wt. Cl and $C_{12}H_{16.2}Cl_{9.8}$, 68.5% wt. Cl. The mean log Kow values determined by a HPLC method were reported to be 6.2 for the 55.9% wt. Cl substance (range of log Kow was 5.0 to 7.1 for the main components of this substance) and 6.6 for the 68.5% wt. Cl substance (range of log Kow was 5.0 to 7.4). Using the empirical Kow data in the Gobas BAF model with an assumption of no metabolism finds that all possible SCCP congeners have a BAF > 5000. As such it is likely they are bioaccumulative.

2.2.2.1 Bioconcentration

Reported bioconcentration factors (BCFs) calculated from laboratory studies for SCCPs have been reviewed in Government of Canada (1993b) and vary dramatically among different species. Relatively low BCF values have been determined in freshwater and marine algae (<1–7.6). BCF values of up to 7816 wet wt. have been measured in rainbow trout (*Oncorhynchus mykiss*) (Madeley and Maddock 1983a,b) and 5785–138 000 wet wt. in the common mussel (*Mytilus edulis*) (Madeley et al. 1983b, Madeley and Thompson 1983d, Renberg et al. 1986). Fisk et al. (1999) measured BCFs in Japanese medaka (*Oryzias latipes*) embryos exposed to a range of concentrations of ^{14}C -labelled C_{10} and C_{12} SCCPs for 20 days. Highest BCFs (1400–2700) were found for $C_{10}H_{15}Cl_7$, compared with $C_{12}H_{20}Cl_7$ (740–1700). These BCF values were similar to previous measurements for fish, although it should be noted that the test organisms in this case were embryos and therefore accumulating SCCPs solely by respiration. No other recent laboratory BCF studies have been identified.

Information submitted by Japan on the measured BCF test results for C=11, Cl = 7-10 suggested that these chemicals are bioaccumulative. In addition the BCF of chlorinated paraffins (C=13-17, Cl = 6-12) were observed to be are low. The test results find that BCFs for some C=13 congeners are lower than BCFs for some congeners of C=11.

Table 6 BCF test results for some SCCP congeners (C=11, Cl=7-10)

		12 days	24 days	38 days	47 days	60 days
Cl=7	1 µg/L	4600	3900	8100	5000	5500
		4900	5400	3300	4000	5200
	0.1 µg/L	2900	3000	3800	3600	2300
		2499	3700	5900	1900	2200
Cl=8	1 µg/L	4900	4700	9200	6000	7200

		5300	6400	4200	5100	7200
	0.1µg/L	3200	3300	4100	4200	2800
		2500	4400	7000	3200	2700
Cl=9	1µg/L	4800	5700	11000	7100	8600
		5600	7600	4800	6500	8800
	0.1µg/L	3600	4100	5600	5300	3700
		2700	5400	9500	4400	3500
Cl=10	1µg/L	4400	5300	9200	7400	8000
		4600	6700	4300	6200	7700
	0.1µg/L	4500	3300	3800	4400	3200
		2600	4800	6700	4600	3300

Table 7 BCF test results for some SCCP congeners (C=13-17; Cl = 6-12)

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

2.2.2.2 Laboratory studies of bioaccumulation, biomagnification and biotransformation

SCCPs are readily accumulated from food by fish in laboratory experiments. Dietary accumulation is influenced by carbon chain length and chlorine content (Fisk et al. 1996, 1998b, 2000). SCCPs with greater than 60% chlorine by weight were found to have equilibrium biomagnification factors (BMFs) greater than 1. Compounds with BMFs greater than 1 will theoretically be biomagnified in food webs. Equilibrium BMFs were predicted from the equation $BMF = \alpha \cdot F/k_d$, where α , the assimilation efficiency, was assumed to be 0.5, F is the feeding rate (lipid corrected) and k_d is the depuration rate (Fisk et al. 1998b). Half-lives in fish ranged from 7 to about 53 days in laboratory studies with juvenile rainbow trout, which implies a potential to biomagnify in aquatic food chains (Fisk et al. 1998b). When compared with the half-lives of 40–60 days in rainbow trout for non-metabolizable organochlorine compounds of similar log K_{OW} values of 6 or greater (i.e., PCBs, mirex, hexachlorobenzene), it appeared that SCCPs with lower chlorination were being metabolized by the fish, but biotransformation products were not measured. Metabolism was inferred from lower amounts of toluene-extractable ^{14}C residues after 40 days of depuration for some SCCPs. However, the half-lives for two chlorododecanes, $C_{12}H_{16}Cl_{10}$ and $C_{12}H_{20}Cl_6$, were similar to those of recalcitrant organochlorines. These compounds had long half-lives (Table 8) and essentially no biotransformation (biotransformation half-life >1000 days) based on this method of calculation. The lack of biotransformation was probably related to the particular chlorine substitution pattern on the n-alkanes; however, only the molecular formulas of these SCCPs were known.

Table 8: Half-lives and estimated biotransformation rates of SCCPs in rainbow trout (from Fisk et al. 2000)

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

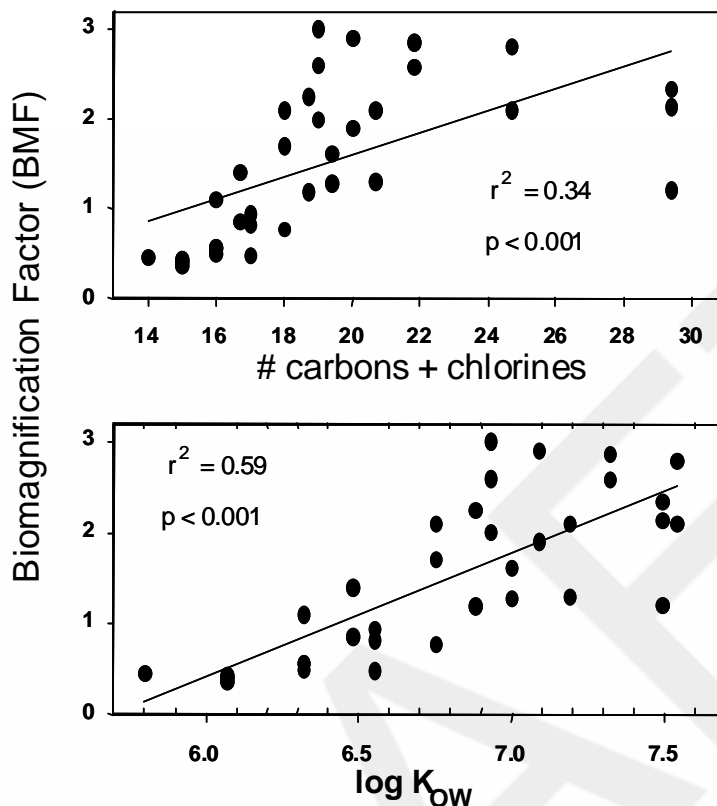
¹ Measured depuration rates (± standard error) converted to half-lives using the equation $t_{1/2}$ (days) = 0.693/rate (day⁻¹).

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

Table 8 compares the measured depuration half-lives and estimated biotransformation half-lives of C_{10–12} SCCPs. The similar depuration and biotransformation half-lives for some SCCPs, particularly the decanes, suggest that the depuration is primarily due to biotransformation.

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

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



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

Fisk et al. (1998a) studied the bioaccumulation and depuration of sediment-sorbed ^{14}C -labelled $\text{C}_{12}\text{H}_{20}\text{Cl}_6$ (56% chlorine by weight) and $\text{C}_{12}\text{H}_{16}\text{Cl}_{10}$ (69% chlorine by weight) by oligochaetes (*Lumbriculus variegatus*). The experiment consisted of a 14- or 21-day uptake period followed by a 42-day depuration period. Organic carbon normalized biota–sediment accumulation factors (BSAF) calculated from the rates of uptake and depuration ranged from 1.9 for $\text{C}_{12}\text{H}_{16}\text{Cl}_{10}$ to 6.8 for $\text{C}_{12}\text{H}_{20}\text{Cl}_6$. Half-lives of the two SCCPs were similar, but uptake of the more highly chlorinated dodecane was significantly slower than that of the less chlorinated dodecane. There was evidence for biotransformation of the SCCPs by *Lumbriculus*; however, the degradation products may also have been formed in the sediments and accumulated by the oligochaetes.

2.2.2.3 Field studies of bioaccumulation

Bioaccumulation factors (BAFs) for SCCP chain length groups in western Lake Ontario lake trout (*Salvelinus namaycush*) were calculated based on concentrations in whole fish (wet weight or lipid

weight) and dissolved water concentrations (Muir et al. 2001). BAFs ranged from 88 000 to 137 600 in lake trout on a lipid weight basis, or from 16 440 to 25 650 on a wet weight basis (Table 9). Chlorinated dodecanes (C₁₂) were the most prominent SCCPs in lake water and fish. The highest BAFs were for the chlorinated tridecanes (C₁₃) because of their low water concentrations. BMFs for the SCCPs based on an alewife (*Alosa pseudoharengus*)/smelt (*Osmerus mordax*) diet ranged from 0.33 to 0.94 and were highest for the tridecanes. These BMF values suggest that SCCPs, especially the chlorinated decanes and dodecanes, are not biomagnifying in the pelagic food web of Lake Ontario. By contrast, PCBs have been shown to biomagnify in the same food web. Relatively high concentrations of SCCPs in sculpin (*Cottus cognatus*) and diporeia (*Diporeia* sp.) imply that sediments are an important source of SCCPs for bottom feeders (Muir et al. 2002).

Table 9: BAFs and BMFs for SCCPs in lake trout and forage fish from western Lake Ontario (from data of Muir et al. 2002)

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

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

² Bioaccumulation factor (wet weight).

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

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

SCCPs have been found in all components of the food chain in Lake Ontario, including 2630 ng/g wet wt. in carp (refer to Table 17). SCCPs have also been found in high concentrations in marine mammals, including beluga (*Delphinapterus leucas*), ringed seal (*Phoca hispida*) and walrus (*Odobenus rosmarus*) from the Arctic (95–626 ng/g wet wt.) (see Table 18).

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

Biomagnification of MCCPs and SCCPs in the Lake Ontario food web was examined. MCCPs and C₁₄–17 chain length groups had very low BMFs between the three forage fish species and lake trout compared with the SCCPs (Table 10).

Table 10: BMFs¹ for MCCPs and SCCPs in the Lake Ontario food web (from Muir et al. 2003)

Group	Alewife–lake trout	Smelt–lake trout	Sculpin–lake trout	Diporeia–sculpin
SCCPs	0.91	0.43	0.27	1.4
C ₁₀	0.43	0.27	0.17	1.3
C ₁₁	0.94	0.35	0.22	1.6
C ₁₂	1.1	0.56	0.37	1.2
C ₁₃	1.5	0.68	0.35	1.1
MCCPs	0.09	0.05	0.05	5.2

Group	Alewife–lake trout	Smelt–lake trout	Sculpin–lake trout	Diporeia–sculpin
C ₁₄	0.27	0.16	0.10	2.7
C ₁₅	0.04	0.02	0.03	9.56
C ₁₆	0.01	0.01	0.01	14.5
C ₁₇	0.00	0.00	0.00	11.5

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

The SCCPs, by comparison, had much higher BMFs, especially for C₁₂ and C₁₃ SCCPs in the same food web (Table 10). With SCCPs, the alewife to lake trout BMF exceeded 1, as did the BMF for diporeia to sculpin.

Bioaccumulation Summary

BAFs for SCCP homologue groups in western Lake Ontario trout range from 16 440 to 25 650 wet wt. (Table 9). Very high bioconcentration factors (BCFs) have been measured for SCCPs in rainbow trout (*Oncorhynchus mykiss*) (up to 7816 wet wt.) (Madeley and Maddock 1983a,b) and in mussels (*Mytilus edulis*) (5785–138 000 wet wt.) (Madeley and Thompson 1983d, Madeley et al. 1983b). Estimated and measured log K_{OW} values for SCCPs range from 4.39 to 8.69. Based on measured Log K_{ow} values in the range of 4.48 to 8.69 the Gobas model predicts a log BAF ranging from 3.47 to 6.2 with the large majority of the predictions having a BAF > 5000.

High SCCP concentrations in marine mammals (ringed seal, beluga whales and walrus) and carp (Tables 17 and 18) show that SCCPs are likely to bioaccumulate in aquatic biota. Bengtsson and Baumann-Ofstad (1982) found evidence of very high retention of a highly chlorinated (71% chlorine by weight) SCCP formulation in bleak during a 316-day elimination period of an uptake/elimination study.

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

While biomagnifications factors (BMF) are not a criterion considered in the regulations for bioaccumulation, BMFs are important supplemental information. If a substance has a BMF greater than 1, it is likely to have high BCF/BAF values. SCCPs have been shown to have BMF values approaching 1 (ranging from 0.33-0.94) between lake trout and forage fish in western Lake Ontario (Environment Canada, 2004) (Table 9).

It is therefore concluded that short-chain chlorinated paraffins are bioaccumulative substances according to the criteria stipulated in the Persistence and Bioaccumulation Regulations of CEPA 1999 (Government of Canada 2000).

2.2.3. Potential for Long Range Transport

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

Van de Meent et al. (2000) ran a global-scale multimedia model (similar to GloboPOPs) assuming no degradation except in air, water and soil. This method shows that SCCPs are multimedia chemicals found in all three compartments, except for some highly chlorinated compounds such as C₁₃H₁₆Cl₁₂.

Arctic accumulation potential

Arctic contamination potential (ACP) is the amount of chemical in Arctic surface media divided by the amount of chemical in the global environment estimated from the GloboPOPs model (Wania 2003; Wania and Mackay 2000). The ACP results were generated from the GloboPOPs model for a hypothetical series of chemicals of varying K_{OA} and K_{AW} assuming 10 years of emission into air, no atmospheric degradation and emission distribution by population density (refer to Canadian Annex E submission for diagram and more details). High ACP values are >3 while low ACP values are <1.0. Results suggest that SCCPs have moderate ACP when released to air, similar to tetra- to heptachlorobiphenyls. Only chemicals such as chlorobenzenes and mono/di/trichlorobiphenyls with log K_{OA} of 3–7 and log K_{AW} of –1 to –2 have high ACP. Similar assessment using emissions to water or soil (in which the ACP values in the two-dimensional [K_{OA} vs. K_{AW}] ACP “space” are smaller than for air emissions) also shows that SCCPs have low ACP when emitted to water.

This data is in agreement with Reth et al. (2005) who found that the SCCP congener pattern measured in the Baltic Sea is similar to that of commercial SCCP mixtures and low molecular weight (C₁₀) congeners increased in the mixtures as samples were taken further away in the North Sea. In addition, lower molecular weight congener mixtures were reported by Tomy et al. (2000) to be present in marine mammals, consistent with long range transport of SCCPs.

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

The EU (2005) dossier prepared for the UNECE LRTAP Convention found that the available data from remote areas (e.g. see section 2.3) clearly show the contamination of the environment and biota by SCCPs.

2.3 Exposure

Atmospheric concentrations

SCCPs were detected in four individual samples of air collected at Alert at the northern tip of Ellesmere Island in the high Arctic (Table 11). Concentrations ranged from non-detectable to 8.5 pg/m^3 in gas-phase samples. Blank samples from the air sampling program at Alert also contained low levels of SCCPs, but at concentrations lower than those in air samples. Heptachloroundecane was the major component of the SCCPs, and undecanes (C_{11}) were the predominant alkane group. Tomy (1997) had previously analyzed air samples collected at Egbert, Ontario, in 1990 for SCCPs and found similar profiles, with chlorodecanes (C_{10}) and chloroundecanes (C_{11}) predominating. The higher SCCP concentrations at Egbert (Table 11) are consistent with its location near population centers of southern Ontario.

Table 11: SCCP concentrations in Arctic and Ontario air (from Tomy 1997; Tomy et al. 1998a; Bidleman et al. 2001)

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

¹ Alert data from Tomy (1997). Note: The blank values for the Alert data were about 1 pg/m^3 . The blank value was not subtracted from the sample values in the table.

² Bidleman et al. (2001).

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

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

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

Comparison of SCCPs in the atmosphere in the United Kingdom, Canada and Norway

In the United Kingdom, Peters et al. (2000) reported a mean SCCP concentration of 99 pg/m^3 in air collected from a semi rural site in Lancaster. SCCP concentrations in air were ranked such that Egbert > Lancaster > Alert and most likely reflect the proximity of Egbert and Lancaster to local source areas. A recent paper by Barber et al. (2005) found that concentrations in the U.K. atmosphere ranged between <185 to 3430 pg/m^3 (mean of 1130 pg/m^3) and were higher than 1997 levels at the same site. Barber et al. (2005) also calculated an average concentration of 600 pg/m^3 of SCCPs for the UK atmosphere. The profiles of SCCPs at Alert resemble those measured in Lancaster but are quite different from those at Egbert and over western Lake Ontario. This is illustrated in Figures 3 and 4. In Figure 3, the individual “formula groups” for each carbon chain length within the SCCP mixture are plotted for Egbert and Lancaster (Tomy et al. 1998a; Peters et

al. 2000). From this figure, it is clear that Egbert air samples have higher proportions of penta-, hexa- and heptachlorodecanes and undecanes compared with Lancaster, whereas Lancaster air has higher proportions of hexa-, hepta- and octadecanes and tridecanes than Egbert air. Both locations are north of major urban areas (Lancaster is about 70 km north of the Manchester–Liverpool area, and Egbert is about 50 km north of Toronto), so the reasons for the differences in homologue patterns between Lancaster and Egbert are not clear.

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

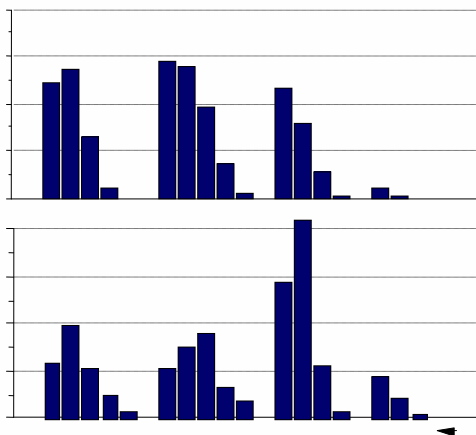
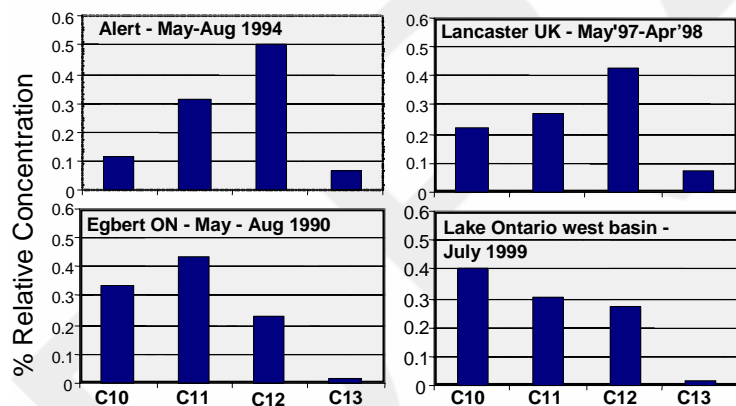


Figure 4: Proportions of SCCP chain length groups in air samples at four locations



In Figure 4, the proportions of SCCP chain length groups are compared at four locations. It can be seen that the profile of SCCP components in Egbert air in summer 1990 resembles that of air over western Lake Ontario collected in July 1999 (Muir et al. 2001), in that both have high proportions of chlorodecanes and chloroundecanes. The profile in Alert air resembles that at Lancaster but differs from that of the southern Ontario sites by having higher proportions of chlorododecanes. These differences appear to be consistent over time. Analysis of air samples collected in October 2000 over western Lake Ontario showed the same profile as in July (i.e., higher chlorodecanes and chloroundecanes), despite much lower air temperatures (D.C.G. Muir, unpublished data). The

reasons for these differences are not clear. The Alert site is very remote from urban areas and receives air predominantly from sources in Europe and Asia because of its unique geographic location. However, this air flow from Europe/Asia is more pronounced in winter than in summer months (Halsall et al. 1998). The three other sites are outside of, but within 100 km of, urban areas.

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

SFT (2002) measured SCCP concentrations in 3 moss samples from Norway (Valvil, Molde, and 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 µg/kg wet weight were measured, suggesting the presence of SCCPs in the atmosphere.

Wastewater treatment effluents

SCCPs were detected in all eight sewage treatment plant final effluents from southern Ontario (Canada) at ng/L concentrations. The highest concentrations were found in samples from treatment plants in industrialized areas, including Hamilton, St. Catharines and Galt, compared with lower concentrations in samples from treatment plants in non-industrial towns, such as Niagara-on-the-Lake and Niagara Falls (Table 12).

Table 12: SCCP concentrations¹ in final effluent of sewage treatment plants in southern Ontario based on samples collected in 1996 (from Muir et al. 2001)

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

¹ Total concentrations (dissolved + particulate).

Reiger and Ballschmiter (1995) reported C₁₀₋₁₃, 62% chlorine SCCP concentrations of 80 ± 12 ng/L in water upstream and 73 ± 10 ng/L in water downstream of a sewage treatment plant in Germany. The concentration of SCCPs in the effluents was 115 ng/L. In the United States, Murray et al.

(1988) reported C₁₀₋₁₃, 60% chlorine SCCP concentrations of <150–3300 ng/L in water from an impoundment drainage ditch that received effluent from a CPs production plant in Dover, Ohio.

Sewage sludge and soils

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

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

In comparison to river and lake sediments, concentrations of SCCPs/MCCPs in sewage sludge are much higher, especially from WWTPs serving industrial areas (Table 13). The fate of CPs in biosolids that are then applied to farmland, a common practice in Europe and a growing practice in North America, has not been thoroughly investigated. The work of Stevens et al. (2002) and Nicholls et al. (2001) points to agricultural soils as potentially a major reservoir of CPs (SCCPs and MCCPs) due to sewage sludge application. This source needs further investigation to determine if it is short term due to rapid biodegradation or represents a long-term reservoir for further environmental distribution.

Table 13: Concentrations of MCCPs in sewage sludges

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

Surface waters

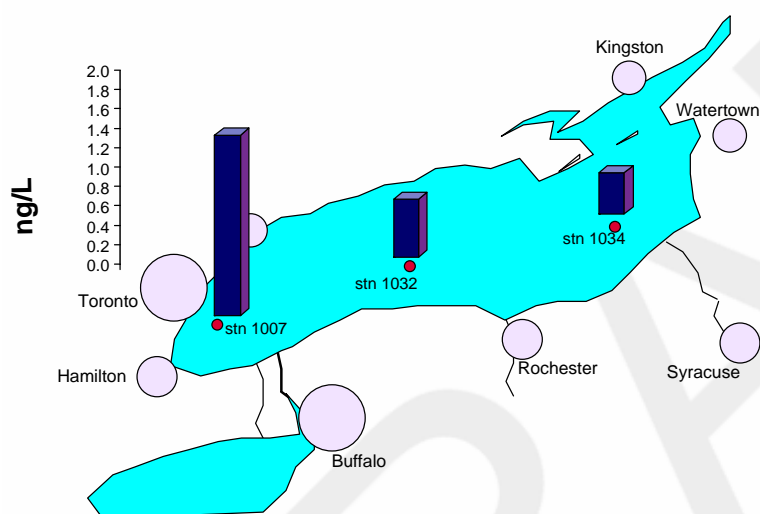
Large-volume water samples (100 L) collected in mid-Lake Ontario in July 1999 and October 2000 showed low levels of SCCPs (Table 14; Figure 5). In 1999, the highest dissolved SCCP concentration (1.8 ng/L) was observed in the western basin and probably resulted from sewage treatment plant sources from large urbanized areas, such as Toronto and Hamilton. A similar gradient was observed in October 2000; however, concentrations were lower. Notwithstanding the high dilution factor that would be involved in Lake Ontario, the fact that these concentrations were observed is an indication that considerable inputs of SCCPs are occurring.

Table 14: Concentrations of SCCPs in filtered surface waters from western Lake Ontario (from Muir et al. 2001)

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

¹ Average of duplicate samples of water at site 1007 in the west basin of Lake Ontario. Collected with XAD-2 resin after filtration through glass fibre filters.

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



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

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 was used as the external standard.

The Ministry of the Environment (2006) in Japan, monitored SCCPs in 6 surface water samples from across the country and did not find any concentrations above the detection limits (which varied from 0.0055 to 0.023 between chain lengths).

Sediments

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

SCCPs were detected in all surface sediment samples from harbour areas along Lake Ontario at concentrations ranging from 5.9 to 290 ng/g dry wt. (Table 15). The highest concentrations were found at the most industrialized site (Windermere Basin, Hamilton Harbour), which has well-documented heavy metal, PAH and PCB contamination. Surface sediments from cores at more remote lakes in northern Ontario, Manitoba, and the Canadian Arctic had concentrations ranging from 1.6 to 257 ng/g dry wt. (see Table 16 below).

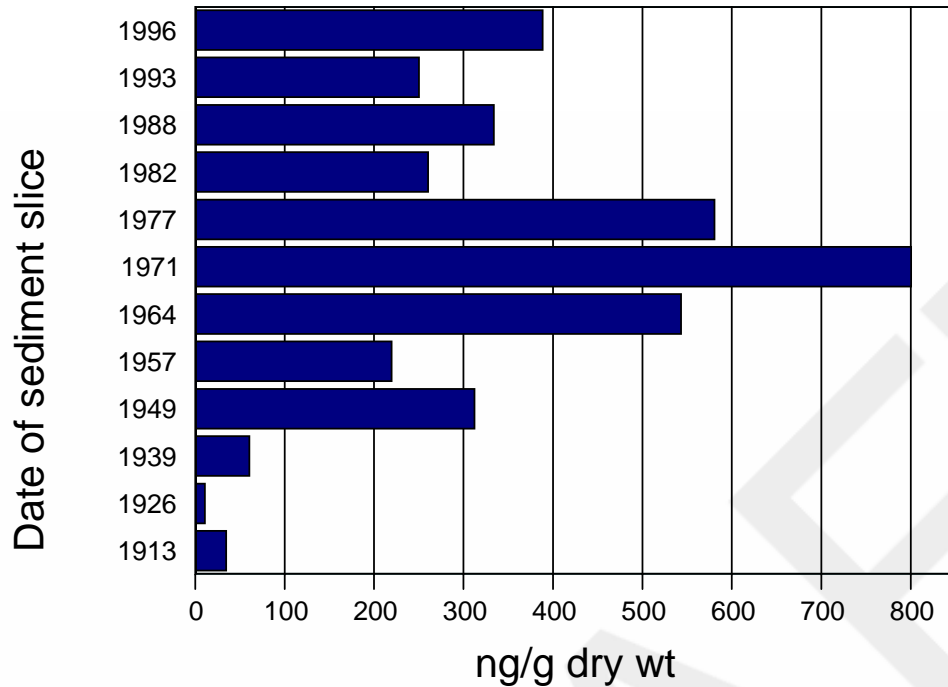
Table 15: SCCP concentrations in surface sediment grab samples collected in 1996 from Lake Ontario harbours (from Muir et al. 2001)

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

The highest concentration of SCCPs in Lake Ontario sediments measured by Marvin et al. (2003) was 410 ng/g dry wt. from the Niagara basin, in an industrialized area. SCCPs were detected in all 26 samples from Lake Ontario, and the average SCCP concentration was 49 ng/g dry wt., which is much higher than sediment concentrations reported for lakes influenced primarily by atmospheric sources (see Table 16 below). Marvin et al. (2003) found that sediment samples from industrialized areas had higher proportions (17–44%) of the longer C₁₃ SCCPs compared with urban non-industrial areas or lakes affected mainly by atmospheric deposition.

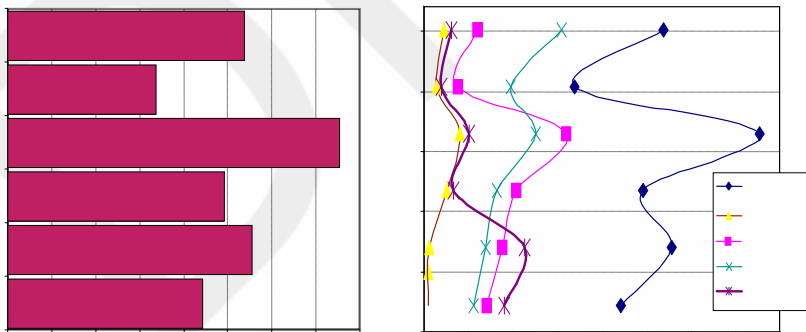
The historical concentration profile of SCCPs in Lake Ontario is shown in Figure 6. SCCP residues are found dating back to 1913, with the maximum residues (800 ng/g dry wt.) occurring in the 1970s. SCCP residues observed in 1996 in the surface sediment layer are approximately 390 ng/g dry wt. SCCPs in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

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



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

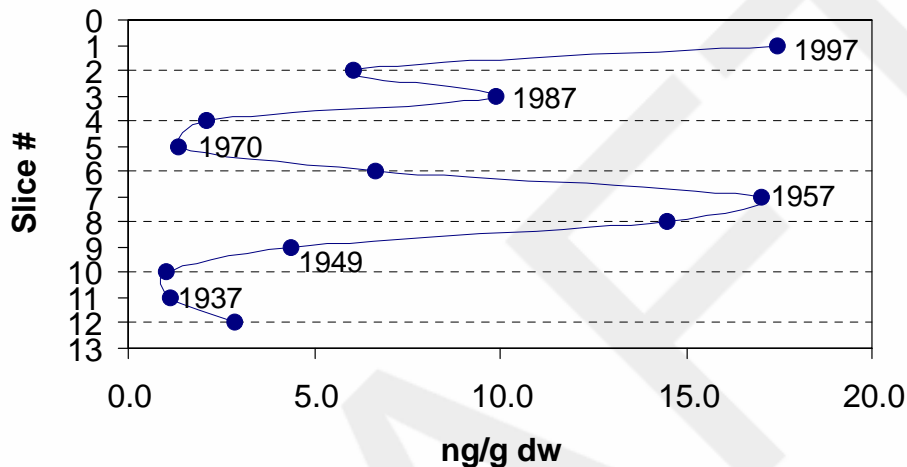
Figure 7: Profiles of SCCPs in a sediment core from Lake St. Francis in the St. Lawrence River downstream of Cornwall, Ontario*



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

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

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



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

The detection of SCCPs in sediments from Lake DV09, which is a remote Arctic lake and thus unlikely to be affected by local sources of contamination, is very good evidence for long-range transport and deposition in the Arctic. The downcore profile clearly shows that SCCP concentrations are well above concentrations in the lowest depth slices, dated to prior to manufacture of SCCPs. These “premanufacture” sediments are good indicators of the relative amount of sampling and laboratory contamination, especially in the case of laminated sediments, where little or no physical mixing occurred. The detection of SCCPs at this remote site and also in other remote Arctic lakes, especially Yaya Lake in the Mackenzie River delta, where levels of SCCPs were also well above background, illustrates the wide dispersal of SCCPs.

Fluxes ($\mu\text{g}/\text{m}^2$ per year) of SCCPs to various Canadian lake sediments are plotted in Figure 9. These fluxes are for surface slices of each sediment core and represent recent inputs (last 5–10 years). The highest fluxes are observed in lake sediments near urban areas (western Lake Ontario and the south basin of Lake Winnipeg). The lowest fluxes are observed in more remote lakes, including Lake Superior, which are influenced mainly by atmospheric inputs. These results suggest that the most elevated SCCP residues observed in lake sediments are mainly derived from urban areas. The sources contributing to SCCP residues observed in Fox Lake, Yukon, are of uncertain origin.

Elevated fluxes of SCCPs to these lakes (Table 16) are higher than for PCBs, which range from about 0.1 to 0.5 $\mu\text{g}/\text{m}^2$ per year at these latitudes (Muir et al. 1996).

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

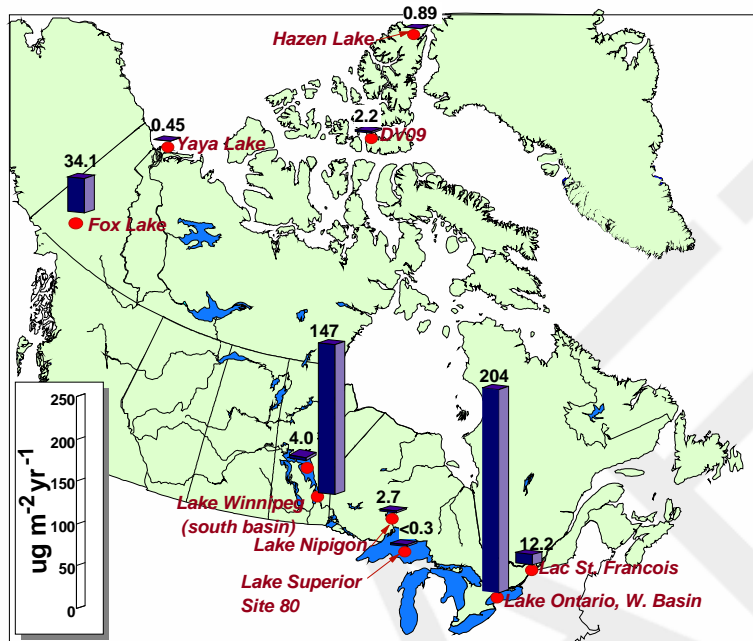


Table 16: Locations, concentrations and fluxes of SCCPs in three Arctic lake sediment cores (Tomy et al. 1999; Stern and Evans 2003)

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

¹ South basin of Lake Winnipeg.

² North basin of Lake Winnipeg.

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

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

to be widely distributed in the U.K. environment (Nicholls et al. 2001).

Pribylova et al. (2006) reported concentrations of SCCPs in 36 sediment samples from 11 Czech rivers and 5 drainage vents near industrial areas. Concentrations ranging from non-detect to 347.41 ng/g dry weight were measured. The Ministry of the Environment (2006) in Japan, has monitored SCCPs in 6 bottom sediment samples from across the country and did not find any concentrations above the detection limits (which varied from 0.34 to 3.0 ng g⁻¹ among carbon lengths).

Biota

Fish

SCCPs were analyzed in lake trout and forage fish samples collected in Lake Ontario in 2001 (Muir et al. 2002) and in lake trout samples collected in 1996 (Muir et al. 2001). C₁₂ SCCPs predominated in lake trout, whereas C₁₁ was the major SCCP in sculpin and smelt (Table 17). SCCP concentrations (lipid wt.) were about 2-fold lower in the samples from 2001 compared with concentrations in lake trout of similar age collected in 1996 (Muir et al. 2001). Further studies with additional sampling times are needed to determine if this represents a consistent temporal trend.

Table 17: Concentrations of SCCPs in lake trout, carp and food web samples from Lake Ontario, including comparison with results from samples collected in 1996 (from Muir et al. 2001, 2002)

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

¹ sd = standard deviation.

Carp (*Cyprinus carpio*) collected from Hamilton Harbour and lake trout (*Salvelinus namaycush*) collected from two locations in western Lake Ontario (Port Credit [northwest] and Niagara-on-the-Lake [southwest]) in 1996 were analyzed for SCCPs. SCCPs were detected in all samples of carp and lake trout from Lake Ontario (Table 17). The higher concentrations observed in carp are probably due to higher exposure of fish to SCCPs in Hamilton Harbour. Table 15 (above) shows a higher concentration of SCCPs in sediment at Hamilton Harbour than was found at Port Credit Harbour, which would imply that water concentrations of SCCPs in Hamilton Harbour may also have been higher. Alternatively, the higher concentrations of SCCPs in carp from Hamilton Harbour could simply be as a result of bioconcentration. Reth et al. (2005) measured SCCP concentrations ranging between 19 and 286 ng/g wet weight in fish liver (North Sea dab, cod, and

flounder) from the North and Baltic Seas.

SFT (2002) measured concentrations of SCCPs in blue mussel and cod livers from Norway. SCCPs were present in all samples with concentrations ranging from 14-130 µg/kg wet weight in mussel and 23-750 µg/kg in cod liver. The Ministry of the Environment (2006) in Japan, has monitored SCCPs in 6 aquatic wildlife samples from across the country and did not find any concentrations above the detection limits (which varied from 0.2 to 1.5 among carbon lengths). Lahaniatis et al. (2000) reported mean values for individual chain length (C10-C13) SCCPs 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.

Marine mammals

Tomy et al. (2000) reported levels of SCCPs in the blubber of ringed seal from Eureka, southwest Ellesmere Island, beluga whales from northwest Greenland, the Mackenzie Delta and the St. Lawrence River estuary, and walrus from northwest Greenland (Table 18). The data in Table 18 show that concentrations of SCCPs in the St. Lawrence beluga were approximately 4 times higher than concentrations in beluga from Greenland and the Mackenzie Delta. The elevated levels of SCCPs in belugas from the St. Lawrence River are consistent with the findings of elevated levels of other organochlorines by Muir et al. (1996), who suggested that this was a food chain effect attributed to local source contamination. The data also show that in many cases SCCP concentrations in Arctic biota are lower than those of other persistent organochlorines, however a few samples showed higher SCCPs.

Table 18: Concentrations of C₁₀₋₁₃ SCCPs and other persistent organic pollutants in blubber of marine mammals from the Arctic and the St. Lawrence River estuary (from Tomy et al. 1998b, 2000)

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

Species	Location	Gender	Year	N	Lipid (%)	Concentrations ¹ (ng/g wet wt.)			
						ΣDDT	ΣPCB	Toxaphene	SCCPs
Walrus	NW Greenland	M	1978	2	83 ± 0.2	33 ± 9.2	160 ± 63	275 ± 74	426 ± 91

¹ Arithmetic means ± standard deviations.

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

There are few other published data on SCCPs in marine mammals for comparison, and differences in analytical methodology make comparisons problematic. Jansson et al. (1993) reported an SCCP concentration of 130 ng/g wet wt. in ringed seal blubber from Svalbard, which is quite similar to levels found by Tomy et al. (2000). It should be noted, however, that the substances measured in Jansson’s study were CPs of unspecified chain length with 6–16 chlorine atoms per molecule and so could have also included MCCPs and LCCPs.

Terrestrial wildlife

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

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

Campbell and McConnell (1980) determined levels of C_{10–20} CPs in birds (Table 19) and seabird eggs in the United Kingdom (Table 20), as well as in water and sediments. The C_{10–20} levels are likely to be dominated by contributions from the SCCPs and MCCPs. The method of analysis used was TLC with argentation. The results of Campbell and McConnell (1980) are generally regarded as valid, because similar concentrations in various media in the United Kingdom have been reported using GC-MS methods (e.g., CEFAS 1999; Nicholls et al. 2001). Nevertheless, they must be regarded as semi quantitative.

Table 19: Concentrations of CPs in birds from the United Kingdom (from Campbell and McConnell 1980)¹

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

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

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

Table 20: Concentrations of CPs in seabird eggs¹ (from Campbell and McConnell 1980)²

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

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

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

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

Human breast milk and food

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

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

Thomas and Jones (2002) also determined the levels of SCCPs in a sample of cow's milk from Lancaster and single butter samples from various regions of Europe (Denmark, Wales, Normandy, Bavaria, Ireland and southern and northern Italy). SCCPs were not detected in the cow's milk sample (detection limit <1.2 µg/kg lipid) but were found in the butter samples from Denmark at 1.2 µg/kg and Ireland at 2.7 µg/kg. The detection limit for the butter samples ranged between 0.72 and 1.1 µg/kg.

A follow up study by Thomas et al. (2003) used more sensitive analytical procedures to analyze SCCPs in breast milk samples (20 from London, 5 from Lancaster). They found concentrations ranging between 49 and 820 µg/kg lipid.

In a market basket survey (KAN-DO Office and Pesticides Team, 1995) of 234 ready-to-eat foods, which represented approximately 5000 food types in American diets, "Chlorowax 500C" was detected once, in enriched white bread, at a concentration of 0.13 µg/g. Food items were screened by gas or liquid chromatography using ion-selective detectors. Findings were confirmed by unspecified analysis.

2.4 Hazard Assessment for Endpoints of Concern

Environmental Effects

Microorganisms

Hildebrecht (1972) concluded that a C₁₀₋₁₃ CP (59% chlorine) and Exchlor 5C (composition unknown) did not affect oxygen utilization by sewage sludge bacteria (species not reported) at concentrations between 1 and 200 mg/L. Birtley et al. (1980) reported that there was no indication that a C₁₀₋₁₃ CP (49% chlorine) was toxic to four strains of *Salmonella typhimurium* at concentrations as high as 2500 µg/plate. Madeley et al. (1983c) found that a short-chain polychlorinated alkane (58% chlorine) caused significant inhibition (>10%) of gas production by anaerobic microorganisms at concentrations of 3.2, 5.6 and 10%. Effects were observed only for the first 3–4 days of the experiments; by day 10, gas production had returned to normal levels.

The following study is described in EC (2000) and U.K. Environment Agency (2003a): Koh and Thiemann (2001) investigated the toxicity of two SCCPs to bioluminescent bacteria *Vibrio fischeri*. The endpoint was the concentration of the SCCP solution that would cause <20% inhibition of the light emission of the bacteria at 585 nm. The long-term (24-hour) test found a No-Observed-Effect Concentration (NOEC)/EC₂₀ value of 0.1 mg/L for the 56% chlorine by weight C₁₀₋₁₃ CP and a NOEC/EC₂₀ of 0.05 mg/L for the 62% chlorine by weight C₁₀₋₁₃ CP.

Pelagic aquatic organisms

There have been only a limited number of studies on the aquatic toxicity of SCCPs that have been published since the reviews of Tomy et al. (1998a) and the risk assessment of SCCPs by the EU (EC 2000). A summary of these studies is given in Table 21, and they are described further below.

Table 21: Recent studies of the aquatic toxicity of SCCPs

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

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

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

Fisk et al. (1999) studied the toxicity of four C₁₀, C₁₁ and C₁₂ SCCP compounds (single chain lengths with mixtures of isomers) to Japanese medaka (*Oryzias latipes*) embryos. Lowest-Observed-Effect Concentrations (LOECs) ranged from 55 µg/L for C₁₂H₂₀Cl₇ to 460 µg/L for C₁₀H₁₆Cl₇. Effects in eggs and larvae over the first 20 days after laying included oil globule migration away from the head and thinning of blood vessels between the head and the oil globule. Toxicity was independent of carbon chain length and chlorine content. The mechanism of toxicity to the embryos was suggested to be narcosis. It should be noted that this study did not meet some of

the criteria specified in the OECD 201 test guideline for a fish early life stage test, including the following points: there were only 10 eggs per test concentration as opposed to 60 stipulated by OECD, the test took place in sealed vials and the study does not describe dissolved oxygen levels, and the study was carried out for approximately 3 days post-hatch, as opposed to the OECD guideline of 30 days post-hatch.

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

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 (Table 21). The daily feeding rate was 1.5% of the mean body weight, and two exposure concentrations for each substance were used. At these feeding rates, none of the compounds was found to have any negative effect on the growth or liver somatic index of juvenile rainbow trout. Concentrations in the trout were much less than the LOECs reported for SCCPs in the Japanese medaka study described above (Fisk et al. 1999).

Cooley et al. (2001) examined the behaviour and liver and thyroid histology of juvenile rainbow trout of the same four C₁₀, C₁₁ and C₁₂ SCCP compounds as in Fisk et al. (1999) via dietary exposure. The exposed trout showed responses indicative of a narcotic mode of action, such as delayed or absent startle response and reduced feeding. Severe liver histopathologies were observed in trout exposed to 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. While a reduced feeding rate was observed in some fish at medium exposure, the fish weight and Liver Somatic Index (LSI) remained the same. These concentrations are within the range of SCCP concentrations seen in carp from Hamilton Harbour (Muir et al. 2001) and in yellow perch from the Detroit River (Tomy et al. 1997) but considerably higher than concentrations found in Lake Ontario fish. Nevertheless, this suggests that histological effects resulting from exposure to SCCPs may be occurring in fish in areas of relatively high exposure, such as Areas of Concern in the Great Lakes.

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

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

Thompson and Shillabeer (1983) carried out a further study on mussels *Mytilus edulis* using a 58% chlorinated SCCP using only two exposure concentrations. Groups of 30 mussels were exposed to concentrations of 2.3 µg/L or 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. Thompson and Madley (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.

Buryskova 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 chlorinated pattern and significant induction of the biomarker phase II detoxification enzyme glutathione S-transferase was observed at 0.5 mg/L.

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. A study was conducted using the midge *Chironomus tentans* (EG&G Bionomics 1983), but exposure was via water only. The following equation was used to calculate the LOEC for benthic invertebrates:

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

where:

- f_{oc} is 0.02, based on the mean organic carbon content for surficial sediment samples from Lake Ontario, expressed on a dry weight basis (Kemp et al. 1977);
- K_{OC} is the organic carbon–water partition coefficient, based on a measured value of 199 500 L/kg for a C₁₀ and C₁₃ CP with ~55% by weight chlorine content (Thompson et al. 1998); and
- $\text{LOEC}_{\text{pelagic}}$ is 8.9 µg/L, based on a 21-day chronic study for *Daphnia magna* (Thompson and Madeley 1983a).

Therefore:

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

Soil-dwelling organisms

A very recently published study (Bezchlebova, 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, EC₅₀ and EC₁₀ values of 5733 mg/kg, 1230 mg/kg, and 660 mg/kg respectively. A PNEC of 5.28 mg/kg was estimated for the soil environment based on their data. The authors compared their results with those of Sverdrup, et al (2006), who investigated the effects of SCCPs (60% chlorine content) on one soil organism (earthworm), 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. This led to an estimated PNEC of 57 mg/kg.

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

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

where:

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

Therefore:

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

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

Birds

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

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

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

when compared with controls.

Mammals

The following is a brief summary of the relevant information reviewed in the EU Risk Assessment Report (EC, 2000) and the draft Health Canada Assessment Report (2003). Health Canada (2003) reviewed mammalian toxicity studies for SCCPs that have been published since the Government of Canada (1993a) risk assessment. However, most of the studies were available only in abstract or summary form, with no quantitative data or statistical analyses presented. These studies are related to effects of SCCPs on the liver, kidney and thyroid in rats. Only one complete report was identified (Wyatt et al. 1993).

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

In a 13-week oral (gavage) rat study 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 the next lowest dose of 10 mg/kg-bw per day. Health Canada (2003) used this NOEL as the Critical Toxicity Value (CTV) for the risk assessment of SCCPs.

The results from available acute studies and skin irritation studies indicate that effects are not dependant upon the chain length and degree of chlorination. SCCPs are of low acute toxicity in animals and are not mutagenic. No evidence is available for human carcinogenicity due to SCCPs. Rodent studies showed dose related increases in adenomas and carcinomas in the liver, thyroid, and kidney. There continues to be contention over the mechanisms of these tumors and whether they are relevant for human health.

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/kg/day of an SCCP. In addition, developmental effects were observed in rats at 2000 mg/kg/day but not at lower doses.

Overall, SCCPs are of low toxicity with the principal toxicological issue being for general non-specific toxicity following repeated exposure. NOAELs for general toxicity of 100 and 1000 mg/kg/day were identified in rats and mice respectively. SCCPs were found to fulfill the criteria of decision 1998/2 of the Executive body (UNECE, 1998) in regard to adverse effects on human health and the environment.

Upper-bound estimates of intake of SCCP for the general Canadian population and the assumptions upon which they are based were calculated by Health Canada (2003). For each age group in the Canadian population, virtually all of the estimated intake 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 (i.e., those being introduced to solid foods). According to Health Canada (2003) the upper-bounding estimates of daily intake of SCCP approach or exceed the TDI for these compounds, which, on the basis of available information, is likely also protective for carcinogenicity.

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).

Summary of the environmental toxicology of SCCPs

The most sensitive toxicity endpoints for SCCPs is summarized in Table 22. The most sensitive aquatic species appears to be *Daphnia*, with chronic NOECs of 5 µg/L. It therefore appears that SCCPs can harm sensitive aquatic organisms at relatively low concentrations (i.e. below threshold criteria of 1 mg/L used to categorize substances on Canada’s Domestic Substances List).

Table 22: Overview of the most sensitive long-term environmental LOEC/NOEC data for SCCPs

Species/endpoint	Effect	NOEC	LOEC	Reference
<i>Daphnia magna</i>	Mortality of offspring, 21 days	5 µg/L	8.9 µg/L	Thompson and Madeley (1983a)
Benthic organisms	Equilibrium partitioning using <i>Daphnia</i> LOEC	NA	88.8 mg/kg dry weight	Thompson and Madeley (1983a)
Japanese medaka embryos, early life stage effects	20-day study, oil globule migration away from head, thinning of blood vessels	9.6 µg/L	55 µg/L	Fisk et al. (1999)
Soil organisms	Equilibrium partitioning using <i>Daphnia</i> LOEC	NA	35.5 mg/kg dry weight (equilibrium partitioning) 570-5733 mg/kg (experimental)	Thompson and Madeley (1983a) Bezchlebova et al., (2007); Sverdup et al. (2006)
Mallard reproduction	Reduced eggshell thickness	168 mg/kg food	954 mg/kg food	EC (2000)
Rat histology	Hypertrophy of the liver and thyroid, increases in liver and kidney weight	10 mg/kg-bw per day	100 mg/kg-bw per day	IRDC (1984)

ASSESSMENT OF POTENTIAL TO CAUSE ECOLOGICAL HARM

It is acknowledged that when risks for persistent and bioaccumulative substances (such as SCCPs) are determined using standard methods, the risks may be underestimated. Table 23 provides exposure and toxicity values associated with margins of exposures for each identified class of

receptors (e.g., pelagic organisms, benthic organisms) but does not incorporate application factors for uncertainties, nor for the conservative approaches that could be considered for persistent and bioaccumulative substances. An EEV (estimated exposure value) and CTV (critical toxicity value) was selected based on available empirical data. The maximum reported value was used as the EEV. CTVs typically represent the lowest chronic ecotoxicity value from an available and acceptable data set. Current exposures are below effects values however they may be approaching in some cases.

Table 23: List of Estimated Exposure Values (EEV) and Critical Toxicity Values (CTV),for SCCPs

	EEV	Sample	CTV
Organism			
Pelagic	44.8 ng/L ¹	STP, Hamilton, ON	8900 ng/L ²
Benthic	0.41 mg/kg ³	Lake Ontario sediment	35.5 mg/kg ⁴
Soil Dwelling	0.64 mg/kg ⁵	UK sewage after 10 years soil app.	35.5 mg/kg ⁴ 570 – 5733 mg/kg ⁶
Secondary Consumer	2.63 mg/kg ⁷	Carp from Hamilton Harbour, Lake Ontario	1000 mg/kg ⁸

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

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

³ Measured in surface sediments from Lake Ontario (Marvin et al. 2003)

⁴ LOEC for *Daphnia magna* using equilibrium partitioning approach (Environment Canada, 2004)

⁵ Calculation of soil amended with sewage sludge for 10 years application (Environment Canada, 2004)

⁶ Experimental data for soil organisms and microorganisms reported by Bezchlebova et al., (2007) and Sverdup et al. (2006)

⁷ Carp from Hamilton Harbour in Lake Ontario (Environment Canada, 2004)

⁸ 13 week rat oral gavage study LOAEL (IRDC, 1984) adapted to otter (Environment Canada, 2004)

3. SYNTHESIS OF INFORMATION

SSCPs are considered to be persistent, bioaccumulative, inherently toxic to some species, and to undergo long range transport to remote areas.

Total reported annual usage of CPs was high in several countries but several have had notable reductions in recent years. For example, use in Canada was approximately 3000 tonnes in 2000 and 2001, in Switzerland 70 tonnes were used in 1994 and likely reduced by 80% now, and Australia reduced by 80% between 2001 and 2003. Releases can occur during production, storage, transportation, and use of SCCPs. Facility wash down and spent metalworking / metal cutting fluids are sources to aquatic ecosystems. Although data are limited, the major sources of release of SCCPs

are likely the formulation and manufacturing of products containing SCCPs, such as polyvinyl chloride (PVC) plastics, and use in metalworking fluids.

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

Bioaccumulation factors (BAFs) of 16 440–25 650 wet weight (wet wt.) in trout from Lake Ontario indicate that SCCPs can bioaccumulate to a high degree in aquatic biota. This is supported by modeling 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. High concentrations of SCCPs in upper trophic level organisms is additional evidence of bioaccumulation. The bioaccumulation of SCCPs is generally supported by the high concentrations of SCCPs measured in marine mammals and aquatic freshwater biota (e.g. beluga whales, ringed seals and fish). High concentrations of SCCPs have also been measured in the breast milk of Inuit women in Northern Quebec.

There is evidence that SCCPs are toxic to sensitive aquatic organisms at relatively low concentrations – below thresholds for inherent toxicity (iT) used to categorize substances on Canada's Domestic Substances List, as well as the OECD, and the USEPA. The most sensitive organism, *Daphnia magna*, has chronic NOECs of 5 µg/L

Concentrations currently measured in the environment are generally below levels that have been associated with effects in laboratory studies. In some cases, concentrations are approaching those that may be of concern, for example in secondary consumers, and elevated levels have been measured in human breast milk, including in remote communities. Particularly in view of SCCPs' persistence, bioaccumulation and their inherent toxicity to a range of organisms, it is considered that SCCPs are likely to cause significant adverse effects as a result of long range transport.

4. CONCLUDING STATEMENT

In summary, the increasing regulation of SCCPs have resulted in a decrease in SCCPs currently in use. However evidence suggests that significant amounts are still in use and being released in several countries. The available empirical and modeled data strongly indicate that SCCPs are persistent, bioaccumulative, and toxic to aquatic organisms at low concentrations. SCCPs have characteristics similar to known POPs that undergo long range environmental transport. SCCPs are considered as POPs pursuant to decisions taken under the UNECE POPs Protocol to the Convention on Long Range Transboundary Air Pollution (LRTAP). Concentrations in biota and sediment from

remote Arctic locations also suggest long range transport of SCCPs is occurring via air or ocean currents.

Based on the available evidence, it is thus likely that SCCPs can, as result of long range environmental transport, cause significant adverse effects on human health and/or the environment, such that global action is warranted.

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