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**Stockholm Convention
on Persistent Organic
Pollutants**

Persistent Organic Pollutants Review Committee
Eleventh meeting
Rome, 19–23 October 2015

Report of the Persistent Organic Pollutants Review Committee on the work of its eleventh meeting

Addendum

Risk profile on short-chained chlorinated paraffins

At its eleventh meeting, by its decision POPRC-11/3, the Persistent Organic Pollutants Review Committee adopted a risk profile for short-chained chlorinated paraffins on the basis of the draft contained in the note by the secretariat (UNEP/POPS/POPRC.11/4). The text of the risk profile, as amended, is set out in the annex to the present addendum. It has not been formally edited.

Annex

SHORT-CHAINED CHLORINATED PARAFFINS

RISK PROFILE

Prepared by the intersessional working group on
short-chained chlorinated paraffins
Persistent Organic Pollutants Review Committee

October 2015

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Executive Summary

1. Releases of short-chain chlorinated paraffins (SCCPs) can occur during production, storage, transportation, use and disposal of SCCPs and SCCPs containing products. Facility wash-down and spent metalworking / metal cutting fluids are amongst others sources to aquatic ecosystems. In industrialized areas, electronic waste (e-waste) recycling, as well as in densely populated areas, high emissions to the environment have been reported. 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, reductions have been noted in recent years in some countries, while in others production volumes of CP mixtures including SCCPs increased.

2. SCCPs are not expected to degrade significantly by hydrolysis in water, and degradation studies and dated sediment cores indicate that they persist in sediment longer than 1 year. SCCPs have atmospheric half-lives ranging from 0.81 to 10.5 days, indicating that they are relatively persistent in air. SCCPs have been detected in diverse environmental samples (air, sediment, water, wastewater, fish, birds, terrestrial and marine mammals), and in remote areas such as the Arctic and Antarctic, providing evidence of long range transport.

3. Available empirical (laboratory and field) and modelled data all indicate that SCCPs can accumulate in biota. Laboratory derived BCFs ranged from 1,900 – 138,000, depending on the species and congener tested. Field derived BAFs for lake trout ranged from 16,440 – 26,650 L/kg ww (wet weight) and for marine fish a mean BAF of 125,892 L/kg ww was determined. For marine arthropod shrimps a BAF up to 63,096 L/kg ww was measured. Modelled BAFs were >5,000 for all SCCPs. For some food webs, including in the Arctic, BMFs and TMFs >1 have been observed, indicating biomagnification and trophic transfer potential. High concentrations of SCCPs in upper trophic level organisms, notably in marine mammals and aquatic freshwater biota (e.g., beluga whales, ringed seals and various fish), is additional evidence of bioaccumulation.

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

5. The International Agency for Research on Cancer considers some SCCPs (average C_{12}, average 60% chlorination) to be possible carcinogens (groups 2B), although questions have been raised regarding the mechanisms for induction of tumours and the relevance for human health of the studies on which this classification was derived. In 1998, the Science Committee on Toxicity, Ecotoxicity and the Environment of the EU suggests that the finding of lung tumors in male mice may be of importance for humans, but concluded in its risk characterisation that the use of SCCPs pose no significant risk for consumers or for man exposed via the environment (CSTEE, 1998). The EU Risk Assessment Report (EC 2000) summarized the effects of SCCPs in mammalian species. Rodent studies showed dose-related increases in adenomas and carcinomas in the liver, thyroid, and kidney. It has been concluded that the concern for humans could not be ruled out. Recent investigations have demonstrated that the mechanism for development of kidney tumors does not follow the classic profile of male-rat specific nephropathy; however, the study could not conclude if the mechanism was rat-specific or not. The most recent assessment of the EU in the frame of the REACH process for identification of substances of very high concern concludes that effects on the liver, thyroid, and kidney have been shown to occur in mammalian species exposed to SCCPs. The effects are manifested as organ weight increases and histological changes after exposure for weeks or months, but may turn into carcinomas and adenomas after chronic exposure (EC 2000, ECHA 2008, Serrone 1987).

6. In summary, the increasing regulation of SCCPs has resulted in a decrease in SCCPs currently in use in some countries. However, evidence suggests that significant amounts are still in use and are being released in several countries. The available empirical and modelled data indicate that SCCPs are persistent, bioaccumulative, and toxic, particularly to aquatic organisms, and undergo long range environmental transport. SCCPs are considered as POPs pursuant to decisions taken under the UNECE (United Nations Economic Commission for Europe) Aarhus (POPs) Protocol to the Convention on Long Range Transboundary Air Pollution.

7. SCCPs are persistent in sediments, and have been measured in sediments also in Arctic lakes. SCCPs are particularly toxic to aquatic invertebrates. Given the key role that invertebrates play in aquatic ecosystems, there is concern relating to the measured SCCP-concentrations and to the potential for toxic effects on sediment-dwelling and other invertebrates. The bioaccumulation by freshwater and

marine fish is also of high concern, given the effects identified in fish at low concentrations. For the regional scale measured levels in water can exceed toxicity thresholds for fish.

8. Although concentrations in water in remote areas are low, SCCPs are measured in Arctic biota at levels comparable to known POPs indicating widespread contamination. Notably, SCCPs are present in Arctic terrestrial and marine mammals, which are in turn food for northern indigenous people. SCCPs are measured in human breast milk both in temperate and Arctic populations. Additionally, simultaneous exposure to SCCPs, other chlorinated paraffins with similar modes of action and POPs may increase the risks due to toxic interactions.

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

1. Introduction

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

1.1 Chemical Identity of the Proposed Substance

11. The risk profile focuses on short-chain chlorinated paraffins (Alkanes, C_{10-13}, chloro) with greater than 48% chlorination. The nomination proposal identified the substance as CAS No. 85535-84-8 and EINECS No. 287-476-5 (Alkanes, C_{10-13}, chloro). This CAS # represents the commercial SCCP product that is produced by the chlorination of a single hydrocarbon fraction consisting of n-alkanes that have a carbon chain length distribution consisting of 10, 11, 12 and 13 carbon atoms; however, this CAS # does not specify the degree of chlorination of the SCCP. Please note that there are other CAS #s which contain SCCPs e.g. CAS No 63449-39-8[1]. The Stockholm Convention nomination for listing is directed at SCCP products that contain more than 48% by weight chlorination. Examples of two molecules that can be found within an SCCP product are presented in Figure 1-1.

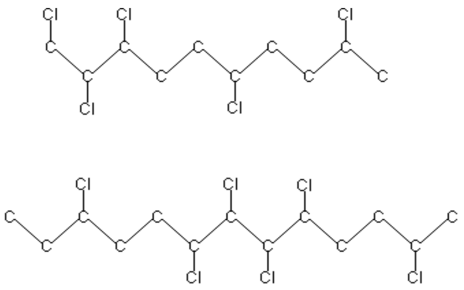

Figure 1-1. Structure of two SCCP compounds ($C_{10}H_{17}Cl_5$ and $C_{13}H_{22}Cl_6$).

1.2 Conclusion of the Review Committee Regarding Annex D Information

12. 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). At the 8th POPRC meeting the Committee agreed to revise the draft risk profile for its 11th meeting (UNEP/POPS/POPRC.8/16/Annex IV)

1.3 Data Sources

13. The risk profile for SCCPs builds on information gathered by the EU in its proposal of SCCPs to the POPRC (UNEP/POPS/POPRC.2/INF/6). The risk profile also incorporates information collected from risk assessment documents prepared by Canada (Environment Canada) and the United Kingdom (DEFRA). Annex E information submissions (2007, 2010 and 2015) from several POPRC Parties and observers were reviewed and any additional information incorporated as appropriate. Information provided by Parties and observers during POPRC 3 and POPRC 5 has also

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[1] List of Toxic Chemicals within the Polychlorinated Alkanes Category and Guidance for Reporting, Section 3, page 9, http://www2.epa.gov/sites/production/files/documents/1999polychloroalkanes.pdf

been incorporated. A detailed but not updated document which served as the basis for the risk profile and a full listing of references for this document can be found in UNEP/POPS/POPRC.5/INF/18.

1.4 Status of the Chemical under International Conventions

14. In August, 2005, the European Community proposed SCCPs to be added to the UNECE Convention on Long Range Transboundary Air Pollution (LRTAP), Aarhus Protocol on Persistent Organic Pollutants. SCCPs met the criteria of decision 1998/2 of the Executive Body for persistence, potential to cause adverse effects, bioaccumulation and potential for long range transport. Thus SCCPs were added to Annexes I and II of the 1998 Aarhus Protocol in December 2009 at the 27th session of the Executive Body. Annex II restricts SCCP uses to fire retardants in rubber used in conveyor belts in the mining industry and in dam sealants, and states that action to eliminate these uses should occur once suitable alternatives are available.

15. In 1995, OSPAR (Oslo/Paris) Commission for the Protection of Marine Environment of the North-East Atlantic adopted a decision on SCCPs (Decision 95/1). OSPAR Decision 95/1 and subsequent EU measures regulate the main uses of SCCPs and sources. In 2006, OSPAR prepared an overview assessment of the implementation of PARCOM (Paris Commission) Decision 95/1 on SCCPs (OSPAR 2006). The assessment was based on national implementation reports received from nine of 15 Contracting Parties which have been requested to submit, in the 2005/2006 meeting cycle, reports on the national measures taken. All reporting Contracting Parties have taken measures to implement PARCOM Decision 95/1. Some Contracting Parties reported a full ban of all or certain uses of SCCPs and reductions of other uses. In general, Contracting Party measures have addressed those uses covered by European Directive 2002/45/EC.

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

2. Summary information relevant to the risk profile

2.1 Physico-chemical properties

17. Information is available on the physical and chemical properties of various SCCP congeners and mixtures (Renberg et al. 1980, Madeley et al 1983a, BUA 1992, Sijm and Sinnige 1995, Drouillard et al. 1998a, Drouillard et al. 1998b, Fisk et al. 1998a). Estimated and measured vapour pressures (VPs) range from 0.028 to 2.8×10^{-7} Pa (Drouillard et al. 1998a, BUA 1992). The vapour pressure of SCCP with 50% chlorine by weight is 0.021 Pa at 40 degree C. (Ref: SRAR-199-ECJRC). Major components of SCCP products with 50-60% chlorine are predicted to have subcooled liquid VPs ranging from 1.4×10^{-5} to 0.066 Pa at 25ºC (Tomy et al. 1998a). Henry's Law Constants (HLCs) ranged from 0.7 - 18 Pa x m^3/mol (Drouillard et al. 1998a), suggesting that SCCPs can remobilise from water to air as a result of environmental partitioning. Measured water solubilities of individual C_{10-12} chlorinated alkanes ranged from 400 - 960 µg/L (Drouillard et al. 1998b), while estimated solubilities of C_{10} and C_{13} chlorinated alkane mixtures ranged from 6.4 - 2370 µg/L (BUA 1992). Water solubility of SCCPs containing 59% chlorine content at 20 degree C ranges from 150 to 470 µg/L (Ref :SRAR-199-ECJRC). The logarithms of the octanol-water partitioning coefficient (log K_{OW}) were generally greater than five, ranging from 4.48 – 8.69. The log K_{OW} SCCPs with chlorine content ranging from 49-71% ranges from 4.39-5.37 (Ref: SRAR-199-ECJRC). Hilger et al. (2011) found that the log K_{OW} value was influenced linearly at a given chlorine content by chain length, while a polynominal effect was observed in dependence on the chlorination degree of an alkane chain. Chlorine substitution pattern influenced markedly the log K_{OW} value. Gawor and Wania (2013) estimated partitioning coefficients for all components of SCCPs based on two QSPR programs and experimental data and displayed the partition behaviour graphically as a function of log K_{OA} and log K_{AW}. For SCCPs (chlorination content 30-70%) the log K_{AW} values are from -6.05 (min) to 1.07 (max) and for log K_{OA} from 4.07 (min) to 12.55 (max).

18. Due to the acknowledged complexity of the mixtures, the chemical analysis of SCCPs is challenging. In the absence of more complete characterizations of the mixtures and suitable individual standards, quantification is usually based on a technical product, introducing major uncertainties if compositions of the sample and the standard do not match (Bayen et al. 2006; Reth et al. 2006 cited in Vorkamp et Riget 2014). Also Sverko et al. (2012) stated that there is a need for a globally concerted effort to standardize methods for SCCP analysis. The most advanced technique in CPs detection is

2-dimensional gas chromatography combined with electron capture detection. The method is able to qualitatively identify groups of CP isomers by carbon chain length and chlorination level. Currently, the most commonly used method of detection and quantification used in the literature is gas chromatography followed by either high or low resolution electron capture negative ion mass spectrometry (GC-ECNI-MS) (Comment submitted on February 27, 2015 POPRC SCCPs risk profile by WCC).

2.2 Sources

2.2.1 Production

19. Chlorinated paraffins (CPs) (of various chain lengths) are currently produced in the Russia, India, China, Japan and Brazil. China is currently the largest volume producer of chlorinated paraffins, with an increasing estimated annual production from 600 kt (metric kilo tonnes) in 2007 (Fiedler 2010) to 1000 kt/year in 2009 (Chen et al. 2011). However according to Annex E (2014) information from China, no specific SCCP production data are available since production is related to several chlorinated paraffin products that do not distinguish SCCPs from other chlorinated paraffins (CPs). Most abundant are CP-42, CP-52 and CP-70 (other are CP-13, CP-30, CP-40, CP-45, CP55 and CP-60). Very limited information is available on SCCP production in some countries.

20. European production estimates for the year 2010 by Euro Chlor was given as 45 kt for all chlorinated alkanes (van Wijk, 2012 in Annex E 2014 submission, Netherlands) and production stopped in 2012 (Comments submitted on February 27, 2015 POPRC SCCPs risk profile by WCC).

21. Sverko et al. (2012) indicated total production estimates for SCCPs in the United States and Europe ranging from 7.5 to 11.3 kt per year. Annex E (2014) information from US indicated 45 kt production volumes in 2007 for SCCP and MCCP. The Chlorinated Paraffins Industry Association (CPIA) submitted Annex E (2010) information on the yearly production of SCCPs in North America from 2000 to 2009. The information collected includes other chain-length chlorinated paraffins. Production was approximately 3.7 kt in 2000, peaked at approximately 4 kt in 2001, and steadily declined to approximately 800 tonnes in 2009. The United States uses several CAS numbers to identify SCCPs. In addition, the United States collects information which includes other chain-length chlorinated paraffins. It is also important to note that since 2012 domestic production and use of SCCPs has been phased-out in the United States. Annex E (2007) information submitted by Brazil indicates that 0.15 kt/year of SCCPs is produced in Brazil.

22. The Republic of Mauritius (Comments submitted on April 7, 2008 POPRC SCCPs risk profile), Australia, Mali, Ecuador, Bulgaria, Costa Rica, Ecuador, Lithuania, Sri Lanka, Canada, Serbia and the Dominican Republic (Annex E 2010 and 2014 submissions) do not currently produce SCCPs. Monaco neither produces nor uses SCCPs (Annex E 2010 submission).

23. Between March 1998 and March 2000, approximately 360 tonnes of SCCPs were imported by Australia, according to information submitted by Australia. However, one company had ceased imports of SCCPs by 2002 (NICNAS 2004). In Canada, total reported annual usage of all CPs was approximately 3,000 tonnes in 2000 and 2001 (Environment Canada 2003a). In 2002, the Republic of Korea imported approximately 156 tonnes of SCCPs (CAS No. 85535-84-8) (Comments submitted on April 7, 2008 POPRC SCCPs risk profile). In 2005 and 2006, Croatia imported 2.3-2.7 and 1.2-1.5 tonnes, respectively, of SCCPs as a component (13±1%) of flame retardant (Annex E 2010 submission). Argentina imported SCCPs in quantities of 40.02 tonnes in 2008 at one location and 53.688 tonnes in 2009 (sum of two locations) (Annex E 2010 submission). The Dominican Republic imported 11.880 tonnes in 2013 (Annex E 2014 submission).

24. Importation quantities for chlorinated paraffins (no specified chain length) were provided by Ecuador (Annex E 2010 submission). From 1990-2010, 8 kt of chlorinated paraffins were imported, with 4.5 kt of this during the period 2005-2010. Mexico also reported importation quantities for chlorinated paraffins (no specific chain length) (Annex E 2010 submission) of 18 kt from 2002-2010.

2.2.2 Uses and Releases

25. In Canada and the EU the major historic uses and releases of SCCPs were in metalworking applications (Environment Canada 2003a, Euro Chlor 1995, OSPAR 2001). Stiehl et al. (2008) anticipated that, the use of SCCPs as flame retardants might increase following the ban of commercial pentaBDE mixtures. In the EU, 9.4 kt/year were used for metalworking in 1994. Petersen (2012) reported an earlier maximum of around 14 kt/year for the period 1978 to 1988. Other uses include paints, adhesives and sealants, leather fat liquors, plastics and rubber, flame retardants and textiles and polymeric materials. The amounts of SCCPs used in the EU were reduced from 13 kt to 4 kt /year for all uses in 1994 and 1998, respectively. Since 2002, the use of SCCPs in the EU in metalworking and

fat liquoring of leathers has been subject to restrictions under EU Directive 2002/45/EC (SCCPs were not allowed in concentrations higher than 1 % in metal working fluids and for fat liquoring of leather). Based on 2004 consumption data, estimated total releases of SCCPs in the EU25 are 4.7-9.5, 7.4-19.6, 0.6-1.8, and 8.7-13.9 tonnes/year to surface water, waste water, air and industrial/urban soil, respectively (Annex E 2010 submission from Germany citing European Chemicals Agency Prioritisation and Annex XIV Background Information for Alkanes, C10-23, chloro). A more recent consumption estimate from Zarogiannis and Nwaogu (2010) in Annex E (2014) information from Netherlands estimated the amount of SCCPs consumed in EU to be 530 tons in 2009. SCCPs are restricted according to the POP Commission Regulation (EU) No. 519/2012 in the EU, which prohibits all uses except for conveyor belts for mining and dam sealants (a total ban is proposed in the future). There is no evidence of any significant natural source of CPs (U.K. Environment Agency 2003a).

26. SCCPs were banned in Norway in 2002. Emissions of SCCPs in Norway are today therefore assumed to be low. Limited amounts of SCCPs may however still be released to the environment from older products and materials that are still in use or from imported articles. Release estimates suggests that releases in Norway were reduced by 73% over the period 1995-2010 from about 1 tonne in 1995 to 0.3 tonnes in 2010 (Norwegian Environment Agency 2014a). In spite strict regulations, SCCPs continues to be detected in environmental- and biota samples in Norway, including in air and biota samples from Svalbard (e.g. Norwegian Environment Agency 2014 a, b, c, 2013).

27. The largest use of SCCPs in the United States was as a component of lubricants and coolants in metal cutting and metal forming operations. The second-largest use was as both a secondary plasticizer and a flame retardant in plastics, especially PVC (US EPA, 2009). Use of SCCPs has been phased-out in the United States. In Japan metal-working industries voluntarily phased out the use of SCCPs by 2007 (Harada et al. 2011).

28. In 1994, 70 tonnes of SCCPs were used in Switzerland and it is estimated that uses have reduced by 80% (Annex E 2007 submission). The most widespread use of SCCPs in Switzerland was in joint sealants. SCCPs have been used as a PCB substitute in gaskets (e.g., splices, in buildings) and this may be a source when buildings are renovated. In Poland, SCCPs are used as a flame retardant in rubber conveyor belts (Annex E 2010 submission). Poland produced 16.4% (approximately 39 tonnes) of the 237.880 tonnes of conveyor belts sold in the EU in 2007, and used 23.062 tonnes of conveyor belts. In Romania, approximately 23,700 kg of SCCPs were used as a plasticizer (Annex E 2010 submission). Brazil indicates that 300 tonnes/year is used in Brazil for the purposes of flame retardant in rubber, car carpet and accessories (Annex E 2007 submission). Use of SCCPs in Australia decreased by 80% between 1998/2000 to 2002 to approximately 25 tonnes/year of SCCPs in the metal working industry (NICNAS 2004). In 2006, in the Republic of Korea, SCCPs were mainly used in lubricant and additive agents. The release pattern and quantitative data are not available (Comments submitted on April 7, 2008 POPRC SCCPs risk profile). The Republic of Mauritius have been reported to use SCCPs (Comments submitted on April 7, 2008 POPRC SCCPs risk profile). Senegal is using SCCPs but quantities are not known (Comments submitted on May 8, 2015 POPRC SCCPs risk profile). Honduras indicates that SCCPs are present in imported products and feedstock, though the quantity is unknown; uses include in PVC production, additive in paints and sealants, the automotive industry, and in electrical cables (Annex E 2010 submission). In Argentina, SCCPs are mainly imported for the plastics industry (Annex E 2010 submission). There is little publically available information regarding the use of SCCPs in wellbore fluid for drilling in oil and gas exploration despite a range of patents indicating their use as such (IPEN, 2015).

29. Anthropogenic releases of SCCPs into the environment may occur during production, storage, transportation, industrial and consumer usage of SCCP-containing products, disposal and burning of waste, and land filling of products. The possible sources of releases to water from manufacturing include spills, facility wash-down and storm water runoff. SCCPs in metalworking/metal cutting fluids may also be released into aquatic environments from drum disposal, carry-off and spent bath use (Government of Canada 1993a). Ecuador notes that the cleaning of metallurgical facilities results in releases to aquatic ecosystems (Annex E 2010 submission). These releases are collected in sewer systems and ultimately end up in the effluents of sewage treatment plants. Information on percentage releases to sewage treatment plants or on removal efficiency is not currently available.

30. For industrialized and densely populated areas high emissions are possible as shown by monitoring data (Chaemfa et al. 2014, Chen et al. 2011). Other releases could include use of gear oil packages, fluids used in hard rock mining and equipment use in other types of mining, fluids and equipment used in oil and gas exploration, manufacture of seamless pipe, metalworking and operation of turbines on ships (CPIA 2002; Environment Canada 2003b).

31. 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).

32. 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. The emission rate depends on the extent of dust control at the facility (De Boer 2010).

33. Recently it was shown that intense e-waste recycling activities are a major source of CPs in the environment (sediment and biota) (Chen et al. 2011, Luo et al. 2015). Also application of sewage sludge to soil or irrigation of wastewater can be a source of SCCP loadings to soil (Zeng et al. 2011b, 2012b).

34. Petersen (2012) reported around 25 kt in building materials as a “stock” of SCCP in buildings and construction. The estimates revealed that sealants and coatings clearly constitute the largest part of the stock, while SCCP in rubber is negligible. The generated waste on a yearly basis was calculated with 1.2 kt SCCP-waste from buildings and construction. The potential for loss during production and transport is expected to be less than that during product use and disposal for chlorinated paraffins (Fiedler 2010).

35. The European Pollutant Release and Transfer Register (E-PRTR)[2] is a web-based register established by Commission Regulation (EC) No 166/2006 which implements the UNECE PRTR Protocol. For the year 2012 17 facilities reported SCCP releases to water (none to air or soil) of 301 kg. The largest share has urban wastewater treatment plants, chemical industry and the energy sector. Data from 2007 suggested a release to water of 290 kg.

36. 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[3]. In 2001-2002, the NPRI reported 1.45 tonnes CPs for disposal to landfill and 1.94 tonnes recycling by recovery of organics from two companies in Ontario. Both of these companies use SCCPs as a formulation component in the manufacture of wires and cables and of paints and coatings, respectively. In 2005, NPRI reported that one company in Ontario disposed 0.023 tonnes of Alkanes 10-13, chloro (CAS# 85535-84-8) off-site and 2.967 tonnes were recycled off-site

37. In the USA, SCCPs are subject to the Toxic Release Inventory (TRI) reporting as part of a broader category of polychlorinated alkanes (all saturated C_{10-13} species with average chlorine content of 40-70%). Information submitted by the USA during the comment period indicated that, for the 2005 TRI data release year, a total of 19,404 kg of polychlorinated alkanes was reported for on and off site disposal or other releases by U.S. industries subject to reporting, including 693 kg as fugitive air emissions; 880 kg as point source air emissions; and 3.2 kg as surface water discharges (TRI release year 2005 data set frozen on 11/15/2006, released to public 03/22/2007, available at www.epa.gov/tri). The 2013 TRI data indicates 192 kg of total releases. As the oil and gas industry are exempt from reporting under the TRI, this may represent an underestimation of releases to air and water (Comments submitted on February 27, 2015 POPRC SCCPs draft risk profile). The use of SCCPs has been phased-out in the United States through a consent decree between the U.S. government and the last manufacturer of the chemical. This phase-out of SCCPs was based on persistence, bioaccumulation and toxicity concerns (Annex E information 2014).

38. Listed POPs can be unintentionally released from rubber or polyurethane foams treated with CPs. Takasuga et al. (2012) detected high levels of PCBs and HCBs in the ppm range as contaminants due to a change in the manufacturing method of CPs.

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[2] http://prtr.ec.europa.eu/PollutantReleases.aspx

[3] http://www.ec.gc.ca/pdb/querysite/query_e.cfm, website accessed August 9, 2007

2.3 Environmental Fate

2.3.1 Persistence

Persistence in Air

39. Estimated atmospheric half-lives for SCCPs based on reaction with hydroxyl radicals range from 0.81 to 10.5 days, using the default atmospheric hydroxyl radical concentration of 1.5×10^6 molecules/cm^3 during sunlight hours in AOPWIN (v. 1.86) computer program (Meylan and Howard, 1993; Atkinson 1986, 1987). Using a lower hydroxyl radical concentration of 5×10^5 molecules/cm^3, which is generally used as a daily (24-hour) average in relatively unpolluted air in the EU, atmospheric half-lives ranged from 1.2 to 15.7 days. It should be noted that hydroxyl radical reaction rates vary temporally with average daily sunlight, and 5×10^5 molecules/cm^3 may not be typical of northern latitudes since hydroxyl radical concentrations decline with latitude. In addition, the high adsorption of CPs to atmospheric particles at low temperatures, typical of conditions at high latitudes, may limit the atmospheric oxidation pathway. Li et al. (2014b) reported based on a newly developed QSAR model (using density functional theory and a OH concentration of 9.7 x10^5 molecules cm^3) that C_{10-11} Cl_{5-8} and $C_{12}Cl_{6-8}$ homologs tend to have a long life time (τ between 3 and 15 days) in air. C_{10-13} (especial C_{13}) with $n_{Cl\ <9}$ have even much longer life times. Monitoring data suggest that C_{10-11} Cl_{5-7} are the dominant group in the gas-phase and C_{11-12} Cl_{6-8} are most abundant in atmospheric deposition samples. C_{13} are seldom detected in air due to their lower volatility (Ma et al. 2014c).

Persistence in water

40. Although SCCPs can undergo slow hydrolysis (Annex E 2010 submission from Costa Rica[4]), they are not expected to degrade significantly by abiotic processes such as hydrolysis (IPCS 1996; U.K. Environment Agency 2003a,b). Koh and Thiemann (2001) showed that SCCP mixtures underwent rapid photolysis in acetone–water under ultraviolet light (mercury arc lamp; approximately 254-436 nm) with half-lives of 0.7–5.2 hours. The half-life of 52% chlorine by weight SCCP product in pure water under the same conditions was 12.8 hours. While the results suggest that photolysis may be a degradation pathway for some SCCPs, the environmental relevance of this study is questionable as the use of an ultraviolet irradiation source may have produced far shorter half-lives than under natural light conditions. Also, the extent of photodegradation may be limited in waters at depth and/or at northern latitudes, and as acetone is a questionable solvent to use in such a study as it is a photo sensitizer.

41. As part of an Annex E submission (2010), the Chlorinated Paraffins Industry Association (CPIA) submitted the results of a prolonged ready biodegradability test for a 49.75% chlorine by weight SCCP product (Cereclor 50LV) following the OECD 301, Closed Bottle Test method with modifications to prolong the test duration to 56 days. Due to the low water solubility of the SCCP, a solvent (silicone oil) or an emulsifier (polyalkoxylate alkylphenol) were employed to increase the bioavailability of the test substance as allowed in OECD 301. Secondary activated sludge or river water was used as an inoculum. Blanks with the solvent or emulsifier alone were not included. Biodegradation in excess of 60% was only obtained when the SCCP was emulsified using polyalkoxylate alkylphenol, with this percentage exceeded within 28 days when river water was used. The SCCP in silicone oil achieved 33% degradation within 56 days. Therefore, this low chlorinated SCCP passed the ready biodegradable criteria only when emulsified and in the presence of an inoculum obtained from river water. In the presence of the inoculum from sludge, it was not readily biodegradable within 28 days, though significant degradation (i.e., 55%) occurred within 42 days in the presence of an emulsifier. The relevance of the use of a surfactant to increase bioavailability for estimation of biodegradation in aquatic environments is unclear, especially in view of conflicting results from biodegradation studies and monitoring results in sediments. It is not known to what extent higher chlorinated SCCPs would be degraded under similar conditions; studies on the biodegradation of SCCPs in sediments suggest that lower chlorinated SCCPs are much more biodegradable than those with higher chlorination.

42. Lu (2013) isolated *Pseudomonas* sp. N35 capable of degrading SCCP in pure culture and sewage sludge microcosms.

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[4] Data from UK Marine SACs Project Chlorinated Paraffins profile, http://www.ukmarinesac.org.uk/activities/water-quality/wq8_35.htm

Persistence in soil and sediment

43. Using 25-day biochemical oxygen demand (BOD) tests, Madeley and Birtley (1980) found that SCCPs (49% chlorine) appeared to be rapidly and completely degraded by acclimatized micro-organisms after 25 days. No significant oxygen uptake was observed in tests using the highly chlorinated CPs, which included two SCCPs (60% and 70% chlorine). Fisk et al. (1998a) found that two ^{14}C-labelled C_{12} chloro-n-alkanes (56% and 69% chlorine) were degraded at 12°C in aerobic sediments used for a study of the bioavailability of SCCPs to oligochaetes. Half-lives in sediment were 12 ± 3.6 days and 30 ± 2.6 days for the 56% and 69% chlorine products, respectively.

44. A study on the aerobic and anaerobic biodegradation of SCCPs in both freshwater and marine sediments was undertaken (Thompson and Noble 2007, in U.K. Environment Agency 2007). Using ^{14}C-labelled n-decane and n-tridecane 65% chlorine by weight products and basing their experiments on the OECD 308 Test Guideline, the mean half-lives (for mineralization [carbon dioxide or methane production]) for a C_{10-13}, 65% chlorine by weight product were estimated to be 1,630 days in freshwater sediments and 450 days in marine sediments under aerobic conditions. Little or no mineralization was noted in anaerobic sediments.

45. Concentration profiles of SCCP residues in sediments from Lake Winnipeg, Manitoba, and Fox Lake, Yukon, indicated that residues were present in the slices dated at 1947 in the sediments from both of these lakes (Tomy et al. 1999). SCCP residues in sediments were observed from the west basin of Lake Ontario dating back to 1949. The highest concentration (800 ng/g dw) was observed in the slice dated at 1971 (Muir et al. 1999a). SCCPs were also measured in a dated sediment core collected in Lake St. Francis (Lac Saint-François), in 1996, downstream of a former CPs manufacturing site. The historical profiles show the presence of relatively low levels of SCCPs compared with Lake Ontario (Muir et al. 1999a; 2002). The highest SCCP concentrations had a median date of 1985 ± 4 years (Turner 1996). The predominant chain length groups in sediments were C_{11} and C_{12}. Iozza et al. (2008) measured SCCPs levels from dated sediment cores as far as from 1960. However the pattern of homolog groups does show changes over time (Iozza et al. 2008, De Boer 2010, Zeng et al. 2012c, Chen et al. 2011).

46. In the absence of information on loading for any of the years at any of these locations, it is not possible to calculate a discrete half-life value from these data. However, the fact that SCCP residues were detected in sediment cores dating back to the 1940s at these locations is evidence that SCCPs can persist for more than 50 years in subsurface anaerobic sediments. Environment Canada (2004) used first order decay equations in a back calculation method to determine that SCCPs have a half-life in sediments longer than 1 year. While the back calculation method for determining half lives does not provide a discrete value for the half life of a chemical, it can provide an answer as to whether a chemical's half life is significantly greater than a specified timeframe. Several government assessments and published reviews have concluded that only slow biodegradation in the environment may be expected to occur, even in the presence of adapted micro-organisms (Government of Canada 1993a,b; Tomy et al. 1998a; EC 2000).

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

Summary of Persistence

48. SCCPs meet the criterion for persistence for sediment (Annex D, Stockholm Convention). They also are sufficiently persistent in air for long range transport to occur. SCCPs appear to be hydrolytically stable. While there is evidence that low chlorinated SCCPs can readily degrade in water under enhanced conditions, the ecological relevance of the test results is not known. There is insufficient information to conclude on the persistence of higher chlorinated SCCPs in water. There is also insufficient information on their persistence in soil. Overall, SCCPs are considered to meet the Stockholm Convention criteria for persistence.

2.3.2 Bioaccumulation

Modelled Log Kow and Bioaccumulation Factors

49. Sijm and Sinnige (1995) calculated a log K_{ow} range between 4.8 and 7.6 for all possible SCCP congeners. Fisk et al. (1998b) determined the octanol-water partition coefficients for $C_{12}H_{20.1}Cl_{5.9}$, 55.9% wt. Cl and $C_{12}H_{16.2}Cl_{9.8}$, 68.5% wt. Cl. The mean log K_{OW} values was estimated at 6.2 for the 55.9% wt. Cl substance (range of log Kow was 5.0 to 7.1) and 6.6 for the 68.5% wt. Cl substance (range of log K_{ow} was 5.0 to 7.4). Using empirical K_{OW} data and assuming no metabolism, the Gobas BAF model for fish estimated BAF values greater than 5,000 for all possible SCCPs.

Bioconcentration

50. Bioconcentration factors (BCFs) calculated from laboratory studies for SCCPs have been reviewed in Government of Canada (1993b) and were found to vary dramatically among different species. Relatively low BCF values have been determined in freshwater and marine algae (<1–7.6). BCF values of up to 7816 wet wt. have been measured in rainbow trout (*Oncorhynchus mykiss*) (Madeley and Maddock 1983a,b) and 5785–138 000 wet wt. in the common mussel (*Mytilus edulis*) (Madeley et al. 1983b, Madeley and Thompson 1983, Renberg et al. 1986).

51. Information submitted by Japan on the measured BCF test results for CPs (C=11, Cl = 7-10) suggested that these chemicals are bioaccumulative. BCF measurements were taken at two test concentrations (0.1 and 1 µg/L) and test organisms were exposed for 60 days. A correlation between chlorine content of CPs and bioconcentration potential was not observed. BCFs ranged from 1,900 to 11,000 (Information submitted to the POPRC SCCPs risk profile by Japan).

52. Further information submitted by Japan for the POPRC5 review (UNEP/POPS/POPRC.5/INF/23) reported measured BCF test results for carp for a C_{13}, $Cl_{4\text{-}9}$ SCCP. Due to the low concentrations of the Cl = 4, 8 and 9 substitutions and lack of sensitivity of the analysis for such concentrations, steady-state BCFs were only determined for the Cl=5, 6 and 7 substitutions. BCF measurements were taken at two test concentrations (1 and 10 µg/L); however, it is unclear if these concentrations were for the congeners at each chlorination level or for the mixture itself. The measured concentrations of C=13 at each chlorination level suggest the test concentration was for congeners at each chlorination level. The exposure duration was 62 days. Bioconcentration potential tended to increase with increasing chlorination. BCFs ranged from 1,530 to 2,830.

53. SCCP also bioaccumulate in humans. Specially, SCCPs with 5–6 chlorines are predicted to have the highest combined potential for LRT and bioaccumulation in humans (Gawor and Wania 2013), if biotransformation is not considered in the prediction. SCCPs with $Cl_{5–6}$ or higher chlorination are estimated to have the potential to bioaccumulate in humans in the Arctic including biotransformation (estimated half-lives in fish was used as crude estimate of the potential for metabolic elimination along the entire human food chain) (Gawor and Wania 2013).

Laboratory studies of bioaccumulation, biomagnification and biotransformation

54. Dietary accumulation is influenced by carbon chain length and chlorine content (Fisk et al. 1996, 1998b, 2000). SCCPs with greater than 60% chlorine by weight were found to have equilibrium biomagnification factors (BMFs) greater than 1, implying a potential to biomagnify in aquatic food chains. Depuration half-lives in fish ranged from 7 to about 53 days for juvenile rainbow trout (Fisk et al. 1998b). In another study, Fisk et al. (2000) estimated SCCPs depuration half-lives that ranged from 7.1 to 86.6 days for rainbow trout. The authors observed that SCCPs with lower chlorination were being metabolized by fish but that the half-lives of two higher chlorinated SCCPs, $C_{12}H_{16}Cl_{10}$ and $C_{12}H_{20}Cl_6$, were similar to those of recalcitrant organochlorines, with long biotransformation half-lives >1000 days. Fisk et al. (2000) found that the depuration and biotransformation half-lives for some $C_{10–12}$ SCCPs, particularly the decanes, were similar, suggesting that the depuration is primarily due to biotransformation. Fisk et al. (2000) also showed that the calculated BMFs for 35 chlorinated n-alkanes (combined data for SCCPs and MCCPs) were significantly related to the number of carbon plus chlorine atoms per compound and to log K_{ow}.

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

similar (~12-14 days), but uptake of the more highly chlorinated dodecane was significantly slower than that of the less chlorinated dodecane.

Field studies of bioaccumulation and biomagnification

56. Bioaccumulation factors (BAFs) for SCCP chain length groups in western Lake Ontario lake trout (*Salvelinus namaycush*) were calculated based on concentrations in whole fish and dissolved water concentrations (Muir et al. 2001). BAFs ranged from 88,000 to 137,600 in lake trout on a lipid weight basis, and from 16,440 to 25,650 on a wet weight basis. Chlorinated dodecanes (C_{12}) were the most prominent SCCPs in lake water and fish. BMFs for the SCCPs based on an alewife (*Alosa pseudoharengus*)/smelt (*Osmerus mordax*) diet ranged from 0.33 to 0.94 and were highest for the tridecanes (C_{13}) because of their low water concentrations. The higher BMF for SCCPs in general was determined between lake trout and alewife alone (BMF 0.91) (Muir et al. 2003). These BMF values suggest that SCCPs, especially the chlorinated decanes and dodecanes, are not biomagnifying in this studied pelagic 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).

57. Bioaccumulation and biomagnification factors were calculated in Lake Ontario for total SCCP isomers, as well as SCCP chain length groups and homologs (same molecular formula), that were detected in water and organism samples (Houde et al. 2008). In both Lake Ontario and Lake Michigan, SCCPs were found to biomagnify between prey and predators. Mean BAFs in lake trout from Lake Ontario varied between 4.0×10^4 and 1.0×10^7 for C_{10}, 1.3×10^5 and 3.2×10^6 for C_{11}, 1.0×10^5 to 2.5×10^5 for C_{12}, and 2.5×10^5 for C_{13} homologs. The highest BMF for total SCCPs was observed for sculpin-*Diporeia* (BMF = 3.6) in both Lake Ontario and Lake Michigan food webs. The BMFs exceeded 1 for lake trout-rainbow smelt and sculpin-*Diporeia* in both lakes, and exceed 1 for lake trout-alewife in Lake Ontario. TMFs ranged between 0.47 and 1.5 for SCCPs (17 homologs) in the Lake Ontario food web. In Lake Michigan, TMFs for SCCPs ranged from 0.41 to 2.4. Among individual SCCP homologs $C_{10}Cl_8$ and $C_{11}Cl_8$ had the highest TMFs (1.5). A TMF above 1 indicates that certain SCCP isomers have the potential to biomagnify in aquatic food web (Houde et al. 2008).

58. In an aquatic food web (including plankton, fish and turtle) near a municipal sewage treatment plant biomagnification of SCCPs was shown (Zeng et al. 2011a). BAFs for most studied SCCP congeners range from 56,160 to 280,200 g/kg lw (lipid weight) in the selected fish species with a mean TMFs for sum $C_{10\text{-}13}$ of 1.61. Homolog specificTMFs were 1.66 for C_{11}, 1.79 for C_{12}, 1.75 for Cl_7, and 1.78 for Cl_8. The specific TMFs for other homologue groups were not evaluated due to their weak linear relationships (Zeng et al. 2011a).

59. Yuan et al. (2012) showed that SCCPs did not biomagnify in molluscs from coastal waters in the Chinese Bohai Sea based on a significant negative linear relationship between lipid normalized SCCPs and trophic level. A similar trend of higher chlorinated PCDD/Fs and PCBs was observed in food chains of the Chinese Bohai Sea (Wan et al. 2005, Wan et al. 2008 cited in Yuan et al. 2012).

60. Bioaccumulation and trophic magnification was reported for a marine food web including zooplankton, eight benthic invertebrate species (bivalves, conch, shrimp and crab) and eight fish species from Liaodong Bay, North China. Log BAFs for sum of SCCPs (24 congeners, C_{10} to C_{13}, Cl_5 to Cl_{10}) ranged from 4.5 to 5.6 ww (wet weight) for all organism. Zooplankton had the lowest log BAF value, arthropod shrimps had log BAFs of 4.6 (39,810 L/kg ww) and 4.8 (63,096 L/kg ww) and highest values were determined in fish (range 4.7 to 5.6, mean 5.1). BAF values of SCCPs formula groups generally present an increasing tendency with the increase of carbon chain length, chlorine contents and K_{OW} values. TMF values (except for $C_{10}Cl_6$, $C_{11}Cl_7$, $C_{12}Cl_7$ and $C_{13}Cl_7$ that showed no significant correlation between tropic level and lw of sum SCCPs) ranged from 1.45 to 5.65 in this food web. TMFs increased with longer chain length and Cl_8 and Cl_9 homologues exhibited the highest TMFs. TMF of sum SCCPs was determined to be 2.38 in the zooplankton-shrimp-fish food web (Ma et al. 2014b).

61. The BSAFs (lipid normalized Biota Sediment Accumulation Factor) of individual congeners for three bivalve species from the Bohai Sea ranged from 0.14 to 4.03, 0.09 to 2.27, and 0.09 to 1.90, respectively (Ma et al. 2014a). The authors concluded that these values were comparable with those reported for PCBs (0.01–5.0) and HCHs (0.81–2.34), but slightly lower than those reported for DDTs (1.1–27.9) and PBDEs (1.0–11.4) (Yang et al. 2006, Wu et al. 2011, Yang et al. 2006 and Wang et al. 2009, all cited in Ma et al. 2014a).

62. Luo et al. (2015) indicated trophic magnification for migratory but not for resident terrestrial bird species inhabiting an e-waste recycling area possibly due to heterogeneity distribution of SCCPs in the studied environment area.

63. Reported levels in Greenland shark liver (caught in waters around Iceland) were 430 ng/g fat (median) for SCCPs and similar to individual PCB congeners previously reported (Strid et al. 2013). SCCPs were detected in seabirds and seabird eggs from Svalbard at mean concentrations of 3.95 ng/ml (glaucous gull plasma) and 7.83 and 3.23 ng/g wet weight in eggs of kittiwake and common eider, respectively (Norwegian Environment Agency, 2013). Eggs of common eider, European shag and herring gull collected in remote areas of Northern Norway contained SCCP concentrations up to about 5 ng/g wet weight (Huber et al., 2015). Based on the relative trophic levels and log-transformed concentrations for polar cod, eider egg, kittiwake egg, glaucous gull plasma, ringed seal plasma and polar bear plasma from the Norwegian Arctic, Svalbard, a TMF of 2.3 was calculated (Norwegian Environment Agency 2013).

64. Results from some monitoring data show different results with regards to bioavailability. Some data indicate that the C_{13} formula group is less bioavailable to fish than the shorter chain length SCCPs based on pattern of SCCPs in Baltic Sea sediments (high $C_{13}Cl_7$ and $C_{13}Cl_8$ content that is not observed in flounder from the same area). However, in another study the proportions of C_{10}–C_{12} homologs in sediments are reflected in flounder. Similarly in Lake Ontario, the C_{11} and C_{12} homolog pattern is reflected in sculpin while the relatively high proportions of $C_{13}Cl_7$ and $C_{13}Cl_8$ in sediment are not. Also $C_{10}Cl_5$ and $C_{10}Cl_6$ are underrepresented in sculpin compared to sediment due to possibly indications on biotransformation. This transformation might occur in sediment dwelling invertebrates preyed upon by the fish as well as in the fish (De Boer 2010). In contrast, Ma et al. (2014b) identified TMFs >1 for $C_{13}Cl_8$ and $C_{13}Cl_9$ homologs in a zooplankton-invertebrate-fish food web.

Bioaccumulation Summary

65. Available empirical (laboratory and field) and modelled data all indicate that many SCCPs can accumulate in biota. Laboratory derived BCFs ranged from 1,900 – 138,000, depending on the species and congener tested. Field derived BAFs for fish ranged from 16,440 – 398,107 L/kg ww and modelled BAFs were >5,000 for all SCCPs. For some SCCPs compounds biotransformation and limited bioavailability might occur. For some food webs including Arctic organisms, BMFs and TMFs were >1, indicating biomagnification and trophic transfer potential. BAF values of SCCPs generally present an increasing tendency with the increase in carbon chain length, chlorine atoms and K_{OW} values. There are also indications that SCCPs can biomagnify in terrestrial species such as birds. Overall, SCCPs meet the Stockholm Convention criteria on bioaccumulation.

2.3.3 Potential for Long Range Transport

66. Annex E (2007) information submitted by Switzerland outlines a study by Wegmann et al. (2007) which examined the long range transport potential (LRTP) of SCCPs and other POPs candidates using the OECD Pov (overall persistence) and LRTP Screening Tool. The results indicated that SCCPs have Pov and LRTP properties similar to those of several known POPs. The Arctic contamination potential (ACP) of several SCCPs was estimated based on their K_{OA} and K_{AW} values, and compared to the ACP results generated for a hypothetical series of chemicals (Wania 2003). Results suggest that SCCPs have ACPs similar to tetra- to heptachlorobiphenyls. Gawor and Wania (2013) modelled the long range transport potential for all SCCPs congeners showing that SCCPs with $C_{5\text{-}7}$ and $Cl_{6\text{-}9}$ (depending on their log K_{OA}) are transported as “multiple hoppers/multiple hoppers swimmers” and have the potential to accumulate significantly in Arctic surface media. The available modelled and empirical information supports the conclusion that SCCPs undergo long range transport.

67. SCCPs have been detected in air, sediment, fish and mammals in the Arctic as well as in air in the Antarctica (see Section 2.4). Tomy (1997) and Bidleman et al. (2001) detected SCCPs at concentrations ranging from <1 to 8.5 pg/m^3 in air collected in the high Arctic (Alert, Ellesmere Island). In 2011, 12 air samples collected at the Canadian High Arctic station of Alert (from January to August) were analysed for SCCPs. The mean and median concentrations were 913.3 and 684.6 pg/m^3, respectively, with a range of 205.7 to 2876 pg/m^3, dominated by the C_{10} and C_{11} formula groups (Hung et al. in press). This is considerably higher than measurements by Tomy (1997) and Bidleman et al (2001). Borgen et al. (2000) measured SCCPs ranging from 9.0-57 pg/m^3 at Mt. Zeppelin, Svalbard, Norway in 1999. In 2013 mean annual SCCP concentrations at Mt. Zeppelin were reported to 360 pg/m^3, with monthly means ranging from 185.8 to 596.5 pg/m^3 (Norwegian Environment Agency 2014b). The annual average for 2014 was 240 pg/m^3, with a range from 140 to 480 pg/m^3 (Norwegian Environment Agency, 2015). Borgen et al. (2002) found much higher SCCP concentrations in air at Bear Island, a small isolated island between Svalbard and mainland Norway. Total SCCP concentrations ranged from 1,800 to 10,600 pg/m^3. SCCPs have been measured in the sediments of remote Arctic lakes (Tomy et al. 1999; Stern and Evans 2003) far from local sources of contamination, as well as in Arctic biota, such as ringed seal, beluga whale, walrus, (Tomy et al. 2000), char, seabirds (Reth et al. 2006), trout from remote lakes (Basconcillo et al. 2015) and Greenland shark (Strid et al.

2013), polar bear, ringed seal, glaucous gull, kittiwake, eider and polar cod (Norwegian Environment Agency 2013, Climate and Pollution Agency Norway 2012). The concentration profiles for SCCPs in Arctic marine mammals show a predominance of the shorter carbon chain length congeners, i.e., the C_{10} and C_{11} formula groups (Tomy et al. 2000), some of the more volatile components of SCCP mixtures (Drouillard et al. 1998a), suggesting that these compounds are more likely to be transported long distances. This is in agreement with the results of Reth et al. (2005, 2006), who found an enrichment of C_{10} SCCPs in biota in the North Sea compared to the Baltic Sea (Reth et al. 2005) and in the Arctic as compared to the Baltic Sea (Reth et al. 2006). However SCCPs in Greenland shark liver showed a predominance of C_{11} and C_{12} with Cl_6 and Cl_7 substitutes (Strid et al. 2013). Halse et al. (2015) found in soils collected along an latitudinal UK - Norway declining SCCPs concentrations with increasing latitude that indicate that SCCPs are comparatively less prone to long range atmospheric transport according to this study finding.

68. Modelled results indicating that the atmospheric half-lives for the major SCCP homologues observed in environmental samples, such as the Great Lakes and Arctic air and biota ($C_{10}H_{17}Cl_5$, $C_{10}H_{16}Cl_6$, $C_{10}H_{15}Cl_7$, $C_{11}H_{18}Cl_6$, $C_{11}H_{17}Cl_7$, $C_{12}H_{20}Cl_6$, $C_{12}H_{19}Cl_7$), are greater than 2 days (Section 2.2.1). Overall, SCCPs meet the Stockholm Convention criteria on long range transport.

2.4 Exposure

2.4.1 Atmospheric concentrations

69. The complexity of SCCPs and the difficulties of accurately detecting and quantifying these compounds in the environment are described in section 2.1. SCCPs were measured in air in several countries including Canada (cf. section 2.3.3), the United Kingdom (U.K.), Norway (cf. section 2.3.3), Sweden and China.

70. The total atmospheric concentrations for SCCPs (predominately existed in the gas phase during summertime) ranged from 9.6 to 20.8 pg/m^3 in George King Island, Antarctica (Ma et al. 2014c). C_{10} and C_{11} carbon chains with Cl_5 and Cl_6 atoms predominated both in the gas and particle-phase. Absorption to organic matter of aerosols played a more important role on atmospheric partitioning and thus transferring SCCPs in the remote Antarctic area (Ma et al. 2014c).

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

72. Peters et al. (2000) reported a mean SCCP concentration of 99 pg/m^3 in air collected from a semi-rural site in Lancaster, U.K. Barber et al. (2005) found that concentrations in the U.K. atmosphere in 2003 ranged between <185 to 3,430 pg/m^3 (mean of 1,130 pg/m^3) and were higher than 1997 concentrations at the same site. Barber et al. (2005) also calculated an average concentration of 600 pg/m^3 of SCCPs for the UK atmosphere.

73. At an urban site in Beijing SCCP levels range from 1.9 to 332 ng/m^3 (Wang et al. 2012). 17.69 ng/m^3 average SCCP concentration were detected at the Pearl River Delta in South China with an enrichment of shorter-chain and less chlorinated congeners (Wang et al. 2013b). Li et al. (2012) investigated the concentrations and congeners group pattern of SCCPs throughout East Asia. Atmospheric concentrations in China (concentration range from 13.4 to 517 ng/m^3) exceeded those in Japan (0.28 to 14.2 ng/m^3) and South Korea (0.60 to 8.96 ng/m^3), both of which exceed the levels determined for other regions of the world. In China C_{10} components with Cl_5 and Cl_6 predominate the profile whereas for Japan and Korea C_{11} (with respect to total chlorine Cl_6 and Cl_5 were most abundant in Japan, Cl_7 in Korea) were the most abundant group.

74. SFT (2002) measured SCCP concentrations in three moss samples from Norway (Valvil, Molde, and Narbuvoll). Samples were taken in forest areas at a minimum distance of 300m from roads and buildings and 10 km from towns. Concentrations of 3 – 100 µg/kg ww were measured, suggesting deposition of SCCPs from the atmosphere. SCCPs have been monitored in air and deposition in the Swedish Environmental Protection Agency’s monitoring program for air since 2009. The concentration of SCCPs at the two measurement stations, situated on the west- and east coast of Sweden, varied between 0.02-2 ng/m^3 in air and 10-380 ng/m^2 per day in deposition (Sjöberg et al. 2014).

75. The results of the 2009 GAPS retrospective study that included four Arctic sites revealed that SCCPs analysed using high resolution GC/MS were detectable in PUF disk samples at 220 pg/m^3 at Barrow, 81 pg/m^3 at Ny-Alesund, 1000 pg/m^3 at Stórhöfði, and 87 pg/m^3 at Little Fox Lake (Harner et al., 2014).

2.4.2 Wastewater treatment effluents, sewage sludge and soils

76. SCCPs were detected in all eight sewage treatment plant (STP) final effluents sampled from southern Ontario, Canada. Total SCCPs (dissolved and particulate $C_{10\text{-}13}$) ranged from 59 to 448 ng/L. The highest concentrations were found in samples from treatment plants in industrialized areas, including Hamilton and St. Catharine's (Muir et al. 2001).

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

78. Iino et al. (2005) measured concentrations of SCCPs in influent and effluent of three MWTPs (municipal wastewater treatment plants) in Japan in 2002. Concentrations of SCCPs in influent and effluent ranged from 220 to 360 ng/L and 16 to 35 ng/L, respectively. All three influent samples contained C_{10}-$_{13}$ with five to eight chlorines for each homologue. None of the three effluent samples showed any detected levels of C_{12} and C_{13} homologues suggesting that sewage sludge treatment may be capable of removing C_{12} and C_{13} homologues.

79. SCCPs have also been measured in sewage sludge. Stevens et al. (2002) found SCCP concentrations ranging from 6.9 to 200 µg/g dw (dry weight) in sewage sludge from 14 waste water treatment plants in the UK. The highest concentrations of SCCPs were detected in sludge from industrial catchments. However, a rural catchment with zero industrial effluent had significant levels (590 µg/g) of total SCCPs/MCCPs in sludge (Stevens et al. 2002). SCCPs have been monitored on an annual basis in sludge from nine Swedish waste water treatment plants since 2004 with SCCPs levels of 0.8-2 µg/g in 2013. No significant time trends can be seen so far (IVL, 2015). Zeng et al. (2012b) measured SCCPs concentrations in sewage sludge ranging from 0.8 to 52.7 µg/g dw (mean 10.7 µg/g) from 52 wastewater treatment plants in China. Mass balance results from a municipal STP in Beijing indicate that 0.8% and 72.6% of the initial SCCPs mass loading (influent 184 +-19 ng/L) were found in the effluents (27+-6 ng/L) and dewatered sludge 15.6 +- 1.4 µg/g dw whereas 26.6% was lost due to dissipation. The authors concluded that sludge is a huge reservoir for SCCPs during wastewater treatment (Zeng et al. 2012a). Agricultural soils may also be a potentially major reservoir of CPs due to sewage sludge application (Stevens et al. 2002; Nicholls et al. 2001) or wastewater irrigation that lead to high accumulation of SCCPs in farm soil with a concentration range of 159.9 to 1,450 ng/g dw (Zeng et al. 2011b).

80. Gasperi et al. (2012) evaluated the quality of combined sewer overflow in an urban watershed. SCCPs concentrations range from 15 to 50 µg/L indicating that SCCPs in combined sewer overflow may pose a significant environmental risk when compared to EU Environmental Quality Standards together with PAHs (polycyclic aromatic hydrocarbons) and tributyltin compounds.

81. An average SCCP concentration of 60.4 ± 54.9 ng/g dw was determined for soils taken near the landfill site in Iqaluit, Nunavut, in the Canadian Arctic (Dick et al. 2010, as provided in the CPIA Annex E 2010 submission). The results indicate that the landfill may act as a local source of SCCP contamination in Iqaluit.

82. SCCPs were measured in surface soils from a background area in China (Chongming Island) with concentrations of total SCCPs in 27 soil samples between 0.42 to 420 ng/g dw with a predominance of C_{13} and C_{11} congeners and Cl_7 and Cl_8 chlorine congeners in most soil samples (Wang et al. 2013a). In 39 background sites from woodland and grassland surface soil collected from Western Europe (UK-Norway transect). SCCPs were detected with the highest average concentrations (35±100 ng/g soil organic matter followed by endosulfan and pentachlorobenzene). A steep decline in concentrations of SCCPs with increasing latitude indicated that the spatial distribution is caused by the proximity of source regions. SCCP concentrations (means of 50 and 22 ng/g soil organic matter in the UK and Norway, respectively) were clearly higher than the Σ31PCB concentrations in the same samples (5 and 8 ng/g soil organic matter for the UK and Norway, respectively (Halse et al. 2015).

83. Wang et al. (2014) reported SCCP concentration data for urban soils collected from Shanghai. The concentrations of sum SCCPs (75 sampling sites) ranged from below the detection limit to 615 ng/g dw with a median value of 15.7 ng/g.

84. In a smaller scale study, Chen et al. (2013) provided data on SCCPs in soils collected in the Guangzhou suburban area with different land use. Sum SCCP concentration data from 17 samples ranged from 7 to 541 ng/g (mean: 84 ng/g). The dominant congeners in the SCCP profile typically had C_{10} and C_{11} carbon chain lengths and between 6 and 7 chlorine atoms. Gao et al. (2012) measured 56.9

to 171.1 ng/g dw SCCPs in paddy soil and 83.5 to 189.3 ng/g dw in upland soils located in the Liaohe River Basin.

2.4.3 Surface and sea waters

85. SCCPs were detected in surface waters in Ontario and Manitoba, Canada. Low levels of dissolved total (C_{10-13}) SCCPs were measured in western Lake Ontario in 1999 and 2000. Concentrations ranged from 0.168 to 1.75 ng/L in 1999 while concentrations were lower (0.074 to 0.77 ng/L) in 2000 (Muir et al. 2001). The average concentration of total SCCPs in water from Lake Ontario (4 metre depths) between 2000 and 2004 was 1.194 ng/L. The concentrations ranged from 0.770 to 1.935 ng/L, 1.039 to 1.488 ng/L and 0.606 to 1.711 ng/L for 2000, 2002 and 2004, respectively (Houde et al. 2008). SCCP concentrations of 30 ± 14 ng/L were measured in the Red River in Selkirk, Manitoba, over a 6-month period in 1995 (Tomy 1997). Tomy et al. (1999) attributed the SCCPs in the water to a local source, possibly a metal machining/recycling plant in the town of Selkirk, as suggested by the similarity of the formula group abundance profile to that of the PCA-60 standard used in the study.

86. The Ministry of the Environment (2006) in Japan monitored SCCPs in six surface water samples from across the country in 2004 (Japanese fiscal year) and did not find any concentrations above the detection limits (which varied from 0.0055 to 0.023 µg/L between chain lengths). A surface water sample was taken in each of two rivers (four sites) in Japan in 2002. SCCPs concentrations ranged from 7.6 to 31 ng/L (Iino et al. 2005; Takasuga et al. 2003). Zeng et al. (2011a) measured sum SCCPs in lake water that receives effluents discharged from the STP in Beijing, China at a concentration range of 162 to 176 ng/L.

87. SCCPs have been detected in river waters in a range of 300 (LOD, limit of detection) to 1,100 ng/L in Spain (Castells et al. 2003; 2004), 15.74 to 59.57 ng/L in the St. Lawrence River, Canada (Moore et al. 2003), <100 to 1,700 ng/L in England and Wales (Nicholls et al. 2001).

88. In seawater samples from the Liaodong Bay in North China SCCPs levels ranged from 4.1 to 13.1 ng/L (Ma et al. 2014b). SCCPs have also been detected in streams, rivers and lakes in Iqaluit, Nunavut in the Canadian Arctic. Depending on the location, concentrations ranged from below detection to 117 ng/L. The results suggest that local sources of SCCP contamination exist in Iqaluit in addition to atmospheric transport (Dick et al. 2010).

2.4.4 Sediments

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

90. Fluxes for surface slices (0-1 cm depth) of sediment core (µg/m^2 per year) of SCCPs to various Canadian lake sediments have been measured (Muir et al. 1999a; Tomy et al. 1999). The highest fluxes are observed in lake sediments near urban areas (western Lake Ontario and the south basin of Lake Winnipeg). The lowest fluxes are observed in more remote lakes, including Lake Superior, which are influenced mainly by atmospheric inputs. These results suggest that the most elevated SCCP residues observed in lake sediments are mainly derived from urban areas.

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

92. In the Canadian Arctic, total SCCP concentrations in sediment cores (0-1 cm depth) ranged from 1.6 to 17.6 ng/g dw in three remote lakes (Tomy et al. 1998a, Stern and Evans 2003, Stern et al. 2005). Sediments from streams, rivers and lakes in Iqaluit, Nunavut in the Canadian Arctic had measurable SCCP concentrations ranging from 5.2 to 138.1 ng/g dw (Dick et al. 2010). The results suggest that local sources of contamination exist in Iqaluit in addition to atmospheric transport of SCCPs (Dick et al. 2010).

93. A dated sediment core from Lake Thun (Switzerland) covering the last 120 years (1899 to 2004) was analyzed to obtain an overview of the historical trend of chlorinated paraffins (Iozza et al. 2008). A 5 ng/g dw concentration was found for the deeper, pre-industrial, sediment slice. The concentrations remained below 10 ng/g dw from 1960 to the 1980s and increased to a maximum of 33 ng/g dw in 1986 to become more or less stable till 2000. In 2004 surface sediments had SCCP

levels of 21 ng/g dw. Data from sediments from Norwegian urban fjords were in the range of 60 to 100 ng/g dw (Norwegian Environment Agency 2014d).

94. Concentrations of SCCPs ranging from 1,250 to 2,090 ng/g dw were measured in marine sediment samples collected from the coastal area of Barcelona (Spain) and near of a submarine emissary coming from a waste water treatment plant located at the mouth of the Besòs River (Barcelona) (Castells et al. 2008). Zeng et al. (2012c) found SCCPs in all surface sediment samples for the East China Sea with a concentration range of 5.8 to 64.8 ng/g dw. The spatial trend showed a decrease with distance from the coast although the highest value was found far away from the land. The C_{10} homolog group was the most abundant carbon chain group. Vertical profiles of sediment cores showed that SCCP concentrations decreased from the surface to the depth of 36 cm and then slightly increased again in depth, which showed a significant positive correlation with TOC (total organic carbon) and chlorine contents. Zhao et al. (2013) measured SCCPs in the 37 surface sediments from the Yangtze River Delta and along the Zhejiang-Fujian coastline in the range from 9 to 37.2 ng/g dw. TOC markedly affected the migrations of SCCPs. River input from the Yangtze river was attributed with the major portion of sedimentary SCCPs in the coastal area, but only a minor fraction was transported to the offshore areas (Zeng et al. 2012a). In a recent study from the Bohai Sea levels on SCCPs ranged from 97.4 to 1,756.7 ng/g dw in 15 surface sediments (average of 650.7 ng/g dw, Ma et al. 2014a). In 10 marine sediments from the Liaodong Bay in China sum SCCPs of 65 to 541 ng/g dw (mean 299 ng/g) were measured (Ma et al. 2014b).

95. SCCPs were detected in the Besòs River in Spain (250 to 3,040 ng/g dw) (Parera et al. 2004), and in various rivers in Germany, France and Norway (2 to 75 ng/g dw) (Hütting 2006). Ballschmiter (1994) found SCCPs in sediments in Germany at concentrations ranging from <5 to 83 µg/kg dw. The 83 µg/kg dw sample was from the Rhine River (U.K. Environment Agency 2003b).

96. A study of SCCPs and MCCPs from the United Kingdom included 20 aquatic and eight agricultural sites (Nicholls et al. 2001). Nicholls et al. (2001) selected surface sediments from three locations, ranging from 1 to 100 m, from 200 to 300 m and from 1–2 km downstream of municipal sewage treatment effluents. SCCPs and MCCPs were judged to be widely distributed in the United Kingdom environment. At sites where the concentration of SCCPs was determined separately from MCCPs, concentrations ranged from 0.6 to 10.3 mg/kg dw (Nicholls et al. 2001).

97. Přibylová et al. (2006) reported concentrations of SCCPs in 36 sediment samples from 11 Czech rivers and five drainage vents near industrial areas. The concentrations ranged from non-detect to 347.4 ng/g dw. Sediment samples from the Czech Republic were analyzed for SCCPs in three locations having different industrial discharges during 2001 and 2002. SCCPs sediment concentrations varied between 24 and 45.78 ng/g dw in the Košetice area (reference area), 16.30 to 180.75 ng/g dw in the Zlín area (rubber, tanning and textile industries) and 4.58 to 21.57 ng/g dw in the Beroun area (cements and machinery industry) (Štejnarová et al. 2005).

98. The Ministry of the Environment (2006) in Japan monitored SCCPs in six bottom sediment samples from across the country in 2004 (Japanese fiscal year) and did not find any concentrations above the detection limits (which varied from 0.34 to 3.0 ng/g among carbon lengths). However six sediment samples were taken in three rivers in Japan in 2003 (Iino et al. 2005). Concentrations of SCCPs ranged from 4.9 to 484.4 ng/g dw. Five of the six samples had concentrations greater than 196.6 ng/g dw.

99. Spatial and temporal trends of SCCPs were investigated in sediments from the Pearl River Delta in South China (Chen et al. 2011). The concentrations varied from 320 to 6,600 ng/g dw. Elevated levels were measured in pond sediments near an e-waste recycling area (2,800 ng/g dw) and in river sediment in highly industrialized area (1,200 ng/g dw). An increased abundance of short chain and low chlorinated congeners was observed in the low industrial activity areas which could be due to preferred transportation of short chain and dechlorination of higher chlorinated congeners. Sediment cores showed highest concentrations on the upper segments (670 to 1,100 ng/g dw at 0 to 32 cm) compared to deeper layers (290 – 680 ng/g dw at 36 to 68 cm) indicating an increase in consumption in recent years.

100. SCCPs were detected in 22 sediment samples from the Liaohe River in northern China with concentrations between 39.8 to 480.3 ng/g dw. Levels increased as the Liaohe River flowed through industrial areas. A preliminary sediment inventory indicated approximately 30.82 tonnes of SCCPs residual in the sediments of the Liaohe River (Gao et al. 2012).

101. From the Chinese Bohai and Yellow Seas 117 surficial sediment samples and two sediment cores were collected to study the spatial and temporal trends of SCCPs at a large scale in the Chinese marine environment. Sum SCCPs in the surficial sediments were in the range of 14.5−85.2 ng/g dw

with an average level of 38.4 ng/g dw. Spatial distribution showed a decreasing trend with the distance from the coast to the open waters. The concentrations of total SCCPs in the two sediment cores ranged from 11.6 to 94.7 ng/g dw and 14.7 to 195.6 ng/g dw, respectively, with a sharp rise from the early 1950s to present. Sources of SCCPs were mainly from river outflows via ocean current and partly from atmospheric depositions (Zeng et al. 2013).

102. Monitoring data for SCCPs from 2002 to 2012 with sampling sites distributed across Norway indicated a mean of all national data of 33 ng/g dw of marine sediment, 36 ng/g dw fresh water sediment, 31 ng/g dw for sludge, 19 ng/L for STP water and 5 ng/L marine and fresh water (Climate and Pollution Agency, Norway 2012).

103. In summary the geographical coverage of environmental levels and trend information on SCCPs increased in recent years covering now also East Asia. Proximity to industrial and urban areas explains much of the spatial variation in sediments. Highest concentrations (in the mg/kg range) were found in the Bohai sea and in marine and freshwater sediments in Spain. Background sites such as lakes in the Canadian Arctic had detectable concentrations, typically in the lower ng/g range for total SCCPs. Dated sediment cores suggested a reduction in SCCPs loadings in Lake Thun and East China Sea but an increasing trend in the Pearl River Delta.

2.4.5 Biota

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

105. Strid et al. (2013) reported levels in Greenland shark liver that showed a large variation from detection limit to 5,200 ng/g lw with a median of 430 ng/g lw.

106. SCCP concentrations in Arctic char from Ellasjøen ranged from 11–27 ng/g ww in liver (Evenset et al. 2004 cited in De Boer 2010). Two species of seabirds, little auks and kittiwakes collected at Bjørnøya had a similar range of SCCPs in liver and muscle as the char liver. There were relatively similar proportions of C_{10}, C_{11}, C_{12}, and C_{13} SCCP chain length groups in char, and the two seabirds possibly reflecting the fact that seabird guano is considered one source of SCCPs to Ellasjøen (Evenset et al. 2007, De Boer 2010). Reth et al. (2006) reported SCCP levels in Arctic char and seabirds (little auk and kittiwake) collected at Bear Island (European Arctic) as well as in cod from Iceland and Norway. SCCP concentrations were between 5 and 88 ng/g ww.

107. Dick et al. (2010) measured SCCPs in anadromous and landlocked Arctic char (*Salvelinus alpinus*) and ninespine stickleback (*Pungitius pungitius*) from various locations near Iqaluit, Nunavut, Canada. SCCP concentrations ranged from non-detectable to 96 ng/g dw and 10.7 to 13.8 ng/g de in Arctic char and stickleback, respectively.

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

109. SCCPs were also detected in fish and invertebrates in the Great Lakes and mean levels ranged from 130 to 500 ng/g ww (Muir et al. 2003). Levels of SCCPs ranged from 118 to 1,250 ng/g ww in carp and 447 to 5,333 ng/g in trout from Lake Ontario (Bennie et al. 2000).

110. Concentrations of SCCPs in lake trout, carp and food web samples from Lake Ontario and Lake Michigan were determined from samples collected between 1999 and 2004 (Houde et al. 2008). SCCP predominated in organisms from Lake Michigan with the highest mean concentrations found in lake trout (*Salvelinus namaycush*, 123 ng/g ww). The concentration of SCCPs ranged from 7.5 to 123 and 1.02 to 34 ng/g ww for Lake Michigan and Lake Ontario, respectively. The lowest mean concentration of the SCCPs (7.5 and 2.4 ng/g ww) was detected in *Mysis* in Lake Michigan and Lake Ontario, respectively.

111. Further studies on concentrations of SCCPs in lake trout from Lake Ontario were conducted by Ismail et al. (2009). Time trends in tissue concentrations were determined from 1979-2004 using stored whole fish homogenate from a long-term monitoring program. Concentrations of SCCPs ranged from 17 to 91 ng/g ww. Concentrations showed an increasing but no significant trend from 1979 to 1988 followed by a significant decrease to 2004 (Ismail et al. 2009).

112. Basoncillo et al. (2015) measured SCCPs in top predatory fish from nine water bodies in Canada, spanning remote and industrialized/urbanized areas. The study included the remote Lake Kusawa, which only receives SCCPs from long-range transport. The SCCP concentrations in lake

trout were relatively low (mean of 2 ng/g wet weight), but similar to the concentrations in lake trout of the Great Lakes, an urbanized and industrialized region. . When comparing Lake Ontario lake trout samples from years 2001 and 2011, concentrations of sum SCCPs decreased by a factor of 6.6, though samples were from different studies. However C_{11}, C_{12} and C_{13} SCCPs with a 5 to 7 chlorine atoms increase significantly in 2011 indicating a shift towards atmospheric transport as the most important source (Basoncillo et al. 2015). SCCPs were measured in perch and Arctic char from Swedish lakes in 2007 and 2010. The SCCP concentrations ranged between 0.6 to 30 ng/g ww. There was no consistent spatial variation in concentration of SCCPs and no significant differences between the year 2007 and 2010 (Nyberg et al. 2012).

113. 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 ww in mussel and 23-750 µg/kg in cod liver. Norwegian Environment Agency (2014d) reported high SCCP concentrations (>2,000 ng/g lw) in hering gull blood from Oslofjord. Lahaniatis et al. (2000) reported mean values for SCCPs of individual chain length (C_{10}-C_{13}) ranging between 6 – 135 µg/kg in fish (sprat, redfish, herring, halibut, sardine, and trout) from a variety of sites in England, Norway, Chile, Greece, Germany, Iceland, France, USA, and the North Sea. Reth et al. (2005) measured SCCP concentrations ranging between 19 and 286 ng/g ww in fish liver (North Sea dab, cod, and flounder) from the North and Baltic Seas. Huber et al (2015) measured SCCPs in Norwegian seabird eggs. SCCP concentrations ranged from 2 – 4.8 ng/g ww.

114. The Ministry of the Environment (2006) in Japan monitored SCCPs in five aquatic wildlife samples from across the country in 2004 (Japanese fiscal year) and did not find any concentrations above the detection limits (0.2 to 1.5 ng/g ww).

115. Yuan et al. (2012) reported sum SCCPs in 91 molluscs samples from coastal waters in the Chinese Bohai Sea with a concentration range of 64.9 to 5,510 ng/g dw. Li et al. (2014a) reviewed POPs level in biota in the Bohai Sea and found that concentrations of SCCPs was the highest, followed by DDTs, HCHs, PCBs, PFOS, HBCDs.

116. In an aquatic ecosystem receiving effluents from a STP concentrations of SCCPs in muscle of leather catfish, common carp, crucian carp, Chinese softshell turtle and java tilapia were 1.7, 1.4, 3.5, 1.3 and 1 µg/g dw, respectively. SCCP concentrations were dependent on the lipid contents (Zeng et al. 2011a).

117. SCCP levels in a marine food web from Liaodong Bay, North China ranged from 86 to 4,400 ng/g ww (lowest and highest values detected in zooplankton and Chinese scallop) with an enrichment of $C_{10}Cl_5$ and $C_{11}Cl_5$ formula groups in most aquatic organisms (Ma et al. 2014b).

<u>Marine mammals</u>

118. SCCPs have been found at concentrations ranging from 95 to 770 ng/g ww in the blubber of marine mammals, including beluga (*Delphinapterus leucas*), ringed seal (*Phoca hispida*), walrus (*Odobenus rosmarus*) and narwhal (*Monodon Monoceros*) from several locations in the Arctic (Tomy et al. 1998b; 1999, 2000).

119. Measurements of SCCPs in blubber of 5 species of marine mammals from west Greenland showed total SCCPs concentrations ranging from 10 ng/g ww in ringed seals to 282 ng/g ww in beluga (Johansen et al. 2004).

120. SCCPs have also been measured in archived beluga whale samples from Hendrickson Island and Pangnirtung in the Canadian Arctic (Canadian Department of Fisheries and Oceans report submitted with the CPIA Annex E 2010 submission). Samples of blubber (1993 to 2008 and 1982 to 2008 for Hendrickson Island and Pangnirtung, respectively), were analyzed for total SCCPs. Blubber from 10-11 or 3-10 individuals from Hendrickson Island and Pangnirtung, respectively, were sampled for each year. Temporal trends of SCCPs have also been investigated using beluga blubber from two sites in the Canadian Arctic. Blubber samples collected between the early 1980's and mid-2000's from the southern Beaufort Sea (Hendrickson Island) and Cumberland Sound (Pangnirtung) beluga whale stocks indicated that SCCPs reached the highest levels in the late 1990s and declined thereafter, reaching near detection limits by 2007 (Tomy and Loseto, 2013; Tomy et al., 2010). The geometric mean SCCP concentration for whales from Hendrickson Island for samples analyzed thus far ranged from 3.96 to 340 ng/g lipid. Concentrations were relatively constant up until 2005 but were considerably lower in 2006 (3.96 ng/g lipid). The geometric mean of the SCCP concentrations for whales from Pangnirtung ranged from 2.95-330 ng/g lipid, with the highest concentration recorded in 1992. Concentrations in 2005 and 2006 were 16.5 and 2.95 ng/g lipid, respectively.

121. SCCPs have been detected in belugas from the St. Lawrence River at an average concentration of 785 ng/g ww (Tomy et al. 1998b; 2000). 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.

Terrestrial wildlife

122. Very limited information is available on SCCP concentrations in tissues of terrestrial wildlife. In Sweden, Jansson et al. (1993) reported CP concentrations (unspecified chain length) in rabbit (Revingeshed, Skåne), moose (Grimsö, Västmanland), reindeer (Ottsjö, Jämtland) and osprey (from various regions in Sweden) to be 2.9, 4.4, 0.14 and 0.53 µg/g lw, respectively. CEFAS (1999) reported the concentrations of SCCPs in earthworms ranging from <0.1 to 0.7 µg/g dw in the United Kingdom in the summer of 1998.

123. Campbell and McConnell (1980) determined levels of $C_{10–20}$ CPs in liver of birds and seabird eggs in the United Kingdom. Concentrations of $C_{10–20}$ CPs ranged from 0.1 to 1.2 µg/g ww in liver of birds and from <0.05 to >0.6 µg/g in eggs. Morales et al. (2012) found SCCP levels in gull eggs (*Laurus michahellis* and *Larus audouinii*) from the Ebro delta Natural Park in the range of 4.5 and 6.4 ng/g ww, respectively. In both gull species several POPs were detected with decreasing concentrations: OC-pesticides>PBDEs>SCCPs>non-ortho PCBs>PCDDs>PCDFs. Higher concentrations were reported recently from muscle of seven bird species collected from an e-waste recycling region in China. SCCPs in two migratory and four residential bird species samples ranged from 19 to 340 ng/g ww (620 to 17,000 ng/g lw) with highest detected levels in goldfinch followed by red-flanked blue tail. Levels in migratory species (red-flanked blue tail and grey-backed thrush) were 71 ng/g and 28 ng/g (mean) ww (Luo et al. 2015).

124. SCCP and MCCP were detected recently in all 79 Norwegian Arctic species samples indicating a widespread exposure to these chemicals in the marine Arctic. The concentrations in polar bear, ringed seal and glaucous gull plasma had an average SCCP content ranging from 4 ng/ml plasma in polar bear and glaucous gull, 5 ng/ml in ringed seal, 7.8 ng/g ww in kittiwake eggs and 3.2 ng/g ww in eider duck eggs, indicating mother to off-spring transfer. SCCP levels of 10 ng/g ww, 20.6 ng/g lw in Atlantic cod liver and 2.3 ng/g ww, 134 ng/g lw in polar cod (pooled sample) where measured (Norwegian Environment Agency, 2013). SCCPs have been detected in the majority of polar bear samples collected 2012 from Hudson Bay, Arctic Canada, with a mean concentration of 493 ± 343 pg/g lw, and might be on the rise (Letcher, 2013).

Plants

125. Chlorinated paraffins (sum of SCCPs, MCCPs and LCCPs) have been detected in spruce needles sampled from Alpine regions of Europe, ranging in concentration from 26-460 ng/g dw No trend could be ascertained between concentration and altitude (Iozza et al. 2009). Levels in pine needles in urban areas around Beijing ranged from 3.0 to 40.8 µg/g lw (Wang et al. 2015).

2.4.6 Human exposure

126. 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.

127. SCCPs were also detected in human breast milk samples from the United Kingdom (Thomas and Jones 2002). SCCPs were found at concentrations of 4.6–110 µg/kg lipid in five out of eight samples from Lancaster and at concentrations of 4.5–43 µg/kg lipid in seven out of 14 samples from London. The estimated mean level of SCCPs was 20 ± 30 µg/kg lipid (based on the positive findings alone) or 12 ± 23 µg/kg lipid (assuming that not detected = half the detection limit). In a follow-up study, Thomas et al. (2006) found concentrations of SCCPs ranging from 49 to 820 µg/kg lipid (median 180 µg/kg lipid) with no differences found between individuals living in urban and rural areas. Twenty-five human milk-fat samples were analyzed. SCCPs were detected in all but four samples. In a study measuring SCCPs and MCCPs in the breast milk of women from Sweden that included pooled samples from 1996-2010, mean levels for SCCPs and MCCPs were reported of 107 and 14 ng/g fat weight, respectively, with no clear time trends, indicating that levels are not declining (Darnerud et al. 2012).

128. SCCPs have also been detected a limited number of plasma samples from pregnant women living in Bodø, Norway and in Taimyr, Russia. In 13 out of the 20 samples analyzed the SCCP levels were below the limit of detection (Climate and Pollution Agency Norway 2012).

129. SCCPs have also been detected in food. Thomas and Jones (2002) detected SCCPs in a sample of cow's milk from Lancaster and butter samples from various regions of Europe (i.e., Denmark, Wales, Normandy, Bavaria, Ireland and southern and northern Italy). SCCPs were also found in butter samples from Denmark at 1.2 µg/kg and Ireland at 2.7 µg/kg. In a market basket survey of 234 ready-to-eat foods representing approximately 5000 food types in American diets, "Chlorowax 500C" was detected once, in enriched white bread, at a concentration of 0.13 µg/g (KAN-DO Office and Pesticides Team, 1995). Lahaniatis et al. (2000) reported mean values for SCCPs of individual chain length (C_{10}-C_{13}) ranging between 7 - 206 µg/kg of fish oil from various origins.

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

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

132. Recent data from Asia showed that whereas in Japan SCCP amounts in food increased only slightly within a decade, an increase of more than 2 orders of magnitude has been identified in Beijing, China. In 1993 the range of SCCPs in dietary samples was between non detectable and 36 ng/kg bw/day whereas 390 – 1,000 ng/kg bw/day were found in 2009. The 95 %percentile was estimated to be 1,200 ng /kg bw/day, whereas it was 72 ng/kg bw/day in Japan in the 1990ies and 81 ng/kg bw/day in 2009 (Harada et al. 2011).

133. Although food is suspected to be the main route of exposure to SCCPs inhalation and dermal uptake may also be important exposure pathways. In France SCCPs have been detected as second most abundant group of compounds in indoor air with a mean concentration of 45 µg/g dust (Bonvallot et al. 2010). Studies from Belgium report 2.08 µg/g median and 12.8 µg/g maximum concentrations in house dust (D'Hollander et al. 2010). In Sweden, Stockholm SCCPs were measured in indoor air (Friden et al. Stockholm University 2010). An exposure assessment was conducted for an adult and a toddler. The exposure of a 25 year old to SCCPs via inhalation, dust ingestion and diet had been estimated to be 3 µg/d, 0.98 µg/day and 12 µg/day (95 %ile Exposure) respectively with diet as major source (~85%) of SCCP exposure. Whereas the exposure of a toddler to SCCPs via inhalation, dust ingestion and diet had been estimated to be 1.4 µg/day, 3.6 µg/day and 6.8 µg/day with a considerable contribution of SSCP exposure via dust ingestion (~30 %). It was concluded by the authors that these data rather present a broadly distributed background contamination than high exposure scenarios. Data from house dust are also available from Germany. Median (and Maximum) concentrations of SCCPs were 6 µg/g (2,050 µg/g) in 2013 (Hilger et al. 2013).

134. Further exposure might occur through consumer articles. The Swedish Chemicals Agency have found SCCPs in a number of articles mainly in soft plastic items made of PVC (i.a. soft plastic bears, beauty cases, exercise mats made of PVC plastic, stickers for wall decoration, fancy dress costumes, plastic toy fishes). Further information about articles found to contain SCCPs is available in Rapex[5]. It has also been reported that hand blenders used for food preparation in Sweden, were found to leak chlorinated paraffins under normal use. The presence of CPs in household appliances that contaminate food during preparation is an unexpected exposure pathway and needs to be addressed (Strid et al. 2014)

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[5] http://ec.europa.eu/consumers/archive/safety/rapex/

2.5 Hazard Assessment for Endpoints of Concern

2.5.1 Mammalian Toxicity

135. The toxicity of SCCPs has been assessed by IPCS, 1986; IARC, 1990, EC 2000, 2005, 2008, Canada 2008, 2013. SCCPs reveal low acute toxicity and may cause skin and eye irritation by repeated exposure.

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

137. Several subchronic studies were conducted. In a 13-week oral rat study (IRDC, unpublished, 1984; Serrone et al. 1987), rats were given short-chain chlorinated paraffins via the diet or via gavage (in separate studies) at doses of 10, 100, or 625 mg/kg/day. Dose-dependent increases in absolute and relative liver and kidney weights were observed from doses of 100 mg/kg/day. While the original interpretation considered these effects as adaptive, more recent interpretations consider them as adverse effects; at this dose, morphological changes in liver, kidney and thyroid were also observed. The US NTP also conducted studies on SCCPs in both rats and mice. In a 13-weeks study where rats were dosed with SCCPs by gavage, a dose-related increase in relative liver weight was observed starting from the lowest dose of 313 mg/kg/day. According to NTP there was clear evidence for carcinogenicity due to increased incidences of hepatocellular neoplasms (primarily neoplastic nodules) in male and female rats, of adenomas or adenocarcinomas (combined) of the kidney tubular cells in male rats, and of follicular cell adenomas or carcinomas (combined) of the thyroid gland in female rats. Also mononuclear leukemia in male rats had been related to the exposure of SCCPs (NTP, 1986). The NTP also conducted a 13-week and a two year lifetime study on mice (NTP, 1986). In the 13 weeks study, a significantly increased relative liver weight was observed at doses of 250 mg/kg/day and higher. In the 2 years carcinogenicity study, doses of 125 and 250 mg/kg/day produced clinical signs of intoxication (decreased activity, prominent backbones, abnormal breathing) at both dose levels and survival was decreased in top dose females. Other effects included dose-related increases in hepatocellular carcinomas and adenomas, and in thyroid follicular cell carcinomas and adenomas in females.

138. Based on these studies, the International Agency for Research on Cancer determined in 1990 that there is sufficient evidence for the carcinogenicity (possibly carcinogenic – groups 2B) of the commercial chlorinated paraffin product tested, which is described as an average carbon-chain length C_{12} and average degree of chlorination 60% (IARC 1990). In the 13th report on carcinogens SCCPs[6] (are classified as reasonably anticipated to be human carcinogens. Several mechanistic studies were conducted to understand the mechanisms of these tumours and whether they are relevant for human health (EC 2000). SCCPs are included in the State of California Office of Environmental Health Hazard Assessment proposition 65 list of chemicals known to the state to cause cancer and they were listed in 1999.

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

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

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[6] Chlorinated Paraffins (C12, 60% Chlorine) CAS No. 108171-26-2

The Scientific Committee on Toxicity, Ecotoxicity and the Environment (CSTEE) who was invited to review the EU Risk Assessment Report (RAR) conclusions on SCCPs stated that the alveolar/bronchiolar carcinomas in male mice should not totally be discounted but acknowledged that "the control animals in this experiment did not show any evidence of lung tumours, whereas the historical control incidence was 5.8%. A dose-related trend was seen with the lowest effective dose of 125 mg/kg/day. The underlying mechanism for this finding is not known, although it should be noted that the chlorinated paraffins are not genotoxic. It was the view of CSTEE that the finding of lung tumours in male mice might be of importance for humans (CSTEE 1998). The final EU risk assessment (EC 2000) noted that although there was an increase in alveolar/bronchiolar carcinomas in mice, the results were within historical control ranges and the controls had a greater incidence of adenomas of the lung than the treated animals. The EU concluded that rodent studies showed dose related increases in adenomas and carcinomas in the liver, thyroid, and kidney. They determined that there was insufficient evidence to conclude that the carcinogenicity observations in the liver and thyroid in mice and the benign tumours in the kidney of male rats were a male rat specific event and consequently the concern for humans could not be ruled out. SCCPs are classified in the EU as a Carcinogen Category 3: R40 – Limited evidence of a carcinogenic effect, and are similarly classified as hazardous in Australia. According to the globalized harmonised system (GHS) which is implemented by the CLP regulation (Commission Regulation (EC) No 1272/2008) in the European Union SSCPs would be classified with Carc Cat 2, H 351: Suspected of causing cancer.

140. The understanding of biological effects of peroxisome proliferation has increased. Peroxisome proliferator –activated receptors (PPAR) play important roles in the regulation of a variety of biological processes, such as adipocyte proliferation and differentiation, glucose homeostasis, intracellular trafficking of lipids and their metabolism, inflammatory responses, vascular functions and embryonic and fetal development (Lau et al. 2010). Therefore further effects of peroxisome proliferation induction might have to be considered as well as the variability in humans including foetuses concerning the amount of the expression of the PPAR receptor.

141. A recent in vitro study with human metabolically competent hepatoma cells (Hep G2) studied the effects of SCCPs on different endpoints. Environmental relevant concentrations of 1µg/l and 10 µg/l lead to significant reduction in cell viability, perturbation of energy production, protein biosynthesis, fatty acid metabolism, ammonia recycling and (Geng et al., 2015).

142. A recent study on the induction of renal tumours in male rats by SCCPs (Warnasuriya et al. 2010, submitted by IPEN as an Annex E 2010 submission) indicates that the mechanism by which renal carcinogenic effects is induced by SCCPs is complex and does not follow the classic profile of male-rat specific alpha-2-urinary globulin (α2u) nephropathy, i.e., accumulation of renal α2u and increase in regenerative cell proliferation. Though SCCPs were found to bind to α2u, exposure to SCCPs resulted in a down-regulation of α2u in the liver leading to no observed accumulation of renal α2u or increase in renal cell proliferation. However, the little α2u that was expressed in the liver appeared to accumulate in the kidney; this, plus the binding of SCCPs to the α2u indicates that α2u nephropathy can not be ruled out. It is hypothesized that peroxisome proliferation in the liver may be responsible for the suppression of α2u expression. As peroxisome proliferation is dose-dependent with SCCPs, lower SCCP doses may result in less α2u expression, thus leading to greater α2u nephropathy and an inverse dose response in renal tumour incidence, as was observed in one study (NTP 1986). Further study is needed to determine the exact mechanism of renal tumour induction by SCCPs before it can be concluded that it is male-rat specific.

143. The most recent assessment of the EU derived a NOAEL of 10 mg/kg/day for subchronic exposure and concluded that the potential toxicological effects of SCCPs on mammals are on the liver, the thyroid hormone system, and the kidneys, e.g., by causing hepatic enzyme induction and thyroid hyperactivity, which in the long-term can lead to carcinogenicity in these organs. Based on the available database, an overall NOAEL of 10 mg/kg/day has been deduced. It has been stated that this NOAEL does not cover chronic exposure situations. No NOAEL can be obtained from the chronic studies (ECHA 2008).

144. One epidemiological registry-based case-control study of cancer of the liver and biliary tract in autoworkers exposed to SCCPs has been identified (Bardin et al. 2005). The case-control study was nested in a cohort study of autoworkers exposed to metalworking fluids. Exposure to specific metalworking fluid components and additives was evaluated, and any exposure to chlorinated paraffins (type not specified) was associated with elevated risk of biliary-tract cancer, based on a small number of cases. No increased risk was found for liver cancer; however, the study included only one exposed worker with liver cancer (NTP, 2014).

145. No changes in reproductive organs were observed in a 13-week study with rats and mice dosed with 5,000 and 2,000 mg SCCP/kg/day. Developmental effects were observed in rats at concentrations causing severe maternal toxicity (2,000 mg/kg/day) but not at lower doses (EC 2000).

146. There are no fertility studies conducted with short-chain chlorinated paraffins, and there is thus a data gap when it comes to potential effects on pups, e.g., during lactation. However, the structural analogue medium-chain chlorinated paraffins (C_{14}-C_{17} 52% chlorination) has been shown to exert a very specific inhibitory effect on the blood clotting system in rats, which is manifested at the sensitive life-stages at and after birth as severe haemorrhaging, leading to mortality both in pups and the dams (IRDC, 1985) (CXR Biosciences Ltd., 2006, in EU-RAR on MCCP, 2008). Pup mortality was observed at 74 mg/kg/day, giving an overall NOAEL of 47 mg/kg/day for the pups. The NOAEL for the dams was 100 mg/kg/day. Read-across of this effect from MCCPs to SCCPs was discussed by the EU classification and labelling group. The EU classification and labelling group agreed that, in view of remaining uncertainties in the mechanisms of toxicity of these chlorinated paraffins, read-across was not justified. This leaves a potential data gap for SCCPs in relation to this endpoint.

147. A risk assessment for Chlorinated Paraffins has been performed and concluded that the margin between the upper-bound estimate of exposure for the age group with greatest exposure to SCCPs and the Tumorigenic Dose (TD05) was inadequate in view of the uncertainty concerning mode of induction of tumours (Environment Canada 2013).

148. To summarise the potential toxicological effects of SCCPs they may affect the liver, the thyroid hormone system, and the kidneys, e.g., by causing hepatic enzyme induction and thyroid hyperactivity, which in the long-term can lead to carcinogenicity in these organs. SCCPs are also classified as suspecting of causing cancer (EU Cat 2; IARC Cat 2B) and are listed as endocrine disrupters of category 1 human health according to the former preliminary criteria for priorisation of potential endocrine disrupting substances (EU, 2015).

2.5.2 Ecotoxicity

Microorganisms

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

Pelagic aquatic organisms

150. There are 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).The lowest toxic effect level identified for a pelagic freshwater aquatic species is 8.9 µg/L based on a 21-day chronic LOEC for *Daphnia magna* (Thompson and Madeley 1983a). The effect was for mortality of the offspring. The NOEC is 5 µg/L. Other effects on Daphnia have been reported at similar concentrations. In a 14-day static renewal study, 50% mortality was observed after 6 days at 10 µg/L (Thompson and Madeley 1983a). Freshwater food webs are dominated by invertebrates, such as adults and larvae of insects, many of them carnivorous (Klecka 2014).

151. The most sensitive measurement endpoint identified for a marine species is larger than 7.3 µg/L based on a 28-day chronic NOEC for the mysid shrimp (*Mysidopsis bahia*) (Thompson and Madeley 1983b). Thompson and Madeley (1983c) reported a NOEC of 12.1 µg/L in a 10-day study with marine algae *Skeletonema costatum.* The toxic effects were transient with no effects seen at any concentration after 7 days. Thompson and Shillabeer (1983) exposed groups of 30 mussels (*Mytilus edulis*) to 58% chlorinated SCCPs (2.3 µg/L and 9.3 µg/L) in a flow-through sea water system for 12-hours. No mortalities were observed but growth was reduced in the 9.3 µg/L exposure.

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

153. Fisk et al. (1996, 2000) studied the accumulation of several ^{14}C-labelled SCCPs (56–69% chlorine by weight) by juvenile rainbow trout (initial weights 2–7 g) during a 40-day exposure period. The daily feeding rate was 1.5% of the mean body weight. None of the compounds was found to have any negative effect on the growth or liver somatic index of juvenile rainbow trout.

154. Cooley et al. (2001) examined the behaviour of juvenile rainbow trout and their liver and thyroid histology following exposure to the same four C_{10}, C_{11} and C_{12} SCCP compounds as in Fisk et al. (1999) via dietary exposure. Trout showed responses indicative of a narcotic mode of action, such as delayed or absent startle response and reduced feeding. Severe liver histopathologies were observed in trout exposed to $C_{10}H_{15}Cl_7$ and $C_{11}H_{18}Cl_6$, consisting of extensive fibrous lesions and hepatocyte necrosis not seen in controls or lower exposed fish. Effects for the $C_{10–12}$ SCCPs were observed at concentrations ranging from 0.79 to 5.5 µg/g in whole fish tissue following dietary exposure to concentrations ranging from 13 to 74 µg/g in food**.** No thyroid lesions were observed. Burýšková et al. (2006) observed developmental malformations and reduced embryo growth in *Xenopus laevis* frog at 5 mg/L and higher concentrations of a commercial mixture of SCCPs (C_{12} 56% chlorine). The results were not related to the chlorination pattern.

Benthic organisms

155. The benthic (macro)invertebrate fauna plays a key role in mineralization of organic matter and serve as food for economically important fish and shellfish species in most aquatic environment (Olomukoro and Ebehiremhen, 2015). An equilibrium partitioning approach (Di Toro et al. 1991) using the most sensitive chronic measurement endpoint identified for a pelagic freshwater invertebrate aquatic species (8.9 µg/L) was used to estimate the toxicity to benthic organisms, since a valid measurement endpoint was not available for a sediment-dwelling invertebrate. The $LOEC_{benthic}$ was estimated to be 35.5 mg/kg dw (Environment Canada 2004).

Soil-dwelling organisms

156. Bezchlebová et al. (2007) investigated the effects of SCCPs (64% chlorine content) on five species of soil organisms (collembola, earthworms, nematodes) and on soil microorganisms (for carbon transformation). *Folsomia candida* (collembola) was identified as the most sensitive organism with LC_{50} (adult survival), and EC_{50} and EC_{10} (reproduction) values of 5733 mg/kg, 1230 mg/kg, and 660 mg/kg dry wt. respectively, after a 28 day exposure. Bezchlebová et al. (2007) also reported reproduction EC_{50}s of 2849 mg/kw dw (28 day exposure), 6027 mg/kg dw (42 day exposure) and 7809 mg/kg dw (28 day exposure) for *Eisenia fetida, Enchytraeus albidus* and *Enchytraeus crypticus*, respectively. Sverdrup et al. (2006) investigated the effects of SCCPs (60% chlorine content) on earthworms, soil nitrifying bacteria, and red clover. These authors found the nitrifying bacteria to be the most sensitive, with an EC_{10} value of 570 mg/kg dry wt.

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

Birds

158. EC (2000) describes a study in which Mallard ducks, exposed to dietary concentrations of C_{10-12} SCCPs (58% chlorine), were investigated for reproductive effects. The study was conducted over a 22-week feeding study, including a 9-week pre-egg-laying period without photostimulation, a 3-week pre-egg-laying period with photostimulation and a 10-week egg-laying period with photostimulation. Birds were induced (by photoperiod manipulation) to lay eggs. Eggs were collected over a 10-week period, and the young which were not fed the test substance were observed for 14 days. The mean measured concentrations were 29, 168 and 954 mg/kg in diet. The lowest level seen to cause slight effects in this study was 954 mg/kg food, which caused a slight, but statistically significant, decrease (by 0.020 mm) in mean eggshell thickness. Although this decrease was significant, the mean eggshell thickness was still in the range of normal values given in the OECD guidelines (0.35–0.39 mm), and no increase in cracked eggs was seen at this dose. No significant difference in the number of eggs laid, number of cracked eggs or mean egg weight was seen in any treatment group when compared with controls. Since slight effects on reproduction have been seen at 954 mg/kg in diet, the NOAEL is considered as 168 mg/kg in food.

159. In a study by Ueberschär et al. (2007), hens from 24 to 32 weeks old were fed technical SCCPs (C_{10-13}, 60% Cl) in increasing concentrations of up to 100 mg/kg feed. No significant effects were noted on the health, relative organ weights or performance (laying intensity, egg weight, feed

consumption) of the hens. Relative organ weights were not significantly affected, except for the pancreas of hens fed the 77 mg/kgww diet, which was decreased. Less than 1% of the chlorinated paraffins ingested were incorporated into the body, while about 1.5% were eliminated with the egg yolk and 30% were excreted with urine and faeces.

Summary of the ecotoxicology of SCCPs

160. The most sensitive toxicity endpoints for SCCPs are summarized in Table 2.5.2-1 that demonstrates that SCCPs are very toxic to aquatic organisms. Moreover it should be noted that for suspected endocrine disrupting chemicals standard toxicity tests can fail to detected low-dose effects and non-monotonic dose response (Vandenberg et al. 2012).

Table 2.5.2-1: Overview of the most sensitive long-term ecotoxicity LOEC, NOEC or EC$_x$ data for SCCPs.

Species/endpoint	Effect	NOEC or NOAEL	LOEC or EC$_x$	Reference
Pelagic organisms (*Daphnia magna*)	Mortality of offspring, 21 days	5 µg/L	8.9 µg/L	Thompson and Madeley (1983a)
Benthic organisms	Equilibrium partitioning based on 21-day study using *Daphnia magna*	NA	35.5 mg/kg dry weight	Environment Canada 2004, based on data from Thompson and Madeley 1983a
Fish (Japanese medaka embryos, early life stage effects)	20-day study, large yolk sac, lethargic or no movement but heart beat present	9.6 µg/L	55 µg/L	Fisk et al. (1999)
Juvenile rainbow trout	Severe liver histopathologies; extensive fibrous lesions and hepatocyte necrosis, 21 days of exposure	-	0.79 to 5.5 µg/g ww (whole fish); food concentration is 13 to 74 µg/g ww	Cooley et al. (2001)
Mysid shrimp (*Mysidopsis bahia*)	28-day chronic NOEC	7.3 µg/L		Thompson and Madeley 1983b
Folsomia candida (collembola) – soil organisms	Reproduction, 28 day exposure	625 mg/kg dw	EC50= 1,230 mg/kg dw	Bezchlebová et al. 2007
Microorganisms (bacterial, soil nitrification)	Soil nitrification	300 mg/kg dw nominal	EC10 = 570 mg/kg dw nominal	Sverdrup et al. (2006)
Bird – mallard duck	Eggshell thinning	168 mg/kg in diet	954 mg/kg in diet	EC 2000
Mammals - rats	Hypertrophy of the liver and thyroid, increases in liver and kidney weight	-	100 mg/kg bw/day	IRDC (1984)
Mammals	Hepatic enzyme induction, thyroid hormone system hyperactivity, carcinogenicity, and pup mortality	10 mg/kg/day (possibly lower at chronic exposure situations	50 mg/kg/day (and possibly lower at chronic exposure situations)	EU SCCP Risk Assessment Report (EC 2000), ECHA (2008) CXR Biosciences Ltd., 2006
Mammals – adult otter	Hypertrophy of the liver and thyroid, increases in liver and kidney weight from rat studies		1,000 mg/kg-food ww	Calculated based on IRDC (1984), Environment Canada, 2008

2.6 Toxicological interactions involving multiple chemicals

161. The environmental hazards of chemicals are still most often assessed substance-by-substance, neglecting mixture effects. This may underestimate the risk/ hazard as the typical exposure is toward multicomponent chemical "cocktails" (e.g. Backhaus et al. 2012, Kortenkamp et al. 2009, Vighi and Villa 2011, EU 2012). The Arcticfunctions as a sink for a wide range of pollutants, among them many known and potential POPs (e.g. AMAP 2009a). Moreover, across the different environmental matrixes more than 80 different organic pollutants have been detected. As demonstrated by Vighi and Villa (2011), combined effects of POPs present in remote areas may be of extremely high concern for the biological community, particularly for top predators and may contribute to the developmental disorders for baby polar previously reported in literature (Vos et al. 2000, Wiig et al. 1998).

162. There are no studies available on the reprotoxicity of SCCPs. In the EU MCCPs are classified as reprotoxic:"May cause harm to breast-fed children" H362and similar reprotoxic effects cannot be ruled out for SCCPs (ECHA 2008).

163. In the environment SCCPs may also be found along with other types of chlorinated paraffins; the long-chained chlorinated paraffins (LCCPs) and medium-chain chlorinated paraffins (MCCPs) (Environment Canada, 2008, 2013, Brooke and Crookes, 2011, Norwegian Environment Agency, 2013). The three groups are themselves not single substances but rather multi-constituent substances of variable composition and potentially different properties. Recognizing varying levels of bioavailability among the CCPs, it is assumed that co-exposure to some level of MCCPs and LCCPs may increase the risk for any adverse or potential adverse effects observed with SCCPs. More specifically, for predators and humans exposed via food, the available data suggests that all three types of chlorinated paraffins may make a significant contribution to the predicted toxicity (Brooke and Crookes 2011).

164. SCCPs have shown to affect the thyroid and are included in the database for priorisation of potential endocrine disrupters based on thyroid tumours, and thyroid hypertrophy in several studies (EU 2015). In the last decades human thyroid cancer incidence has continuously and sharply increased all over the world (Pellegriti et al. 2013). According to the authors the reasons for the increase are still unclear but some environmental carcinogens may have specifically affected the thyroid.

165. Epidemiologic studies in heavy polluted areas have demonstrated that long term exposure with organochlorine compounds leads to thyroid and metabolic disorders (Langer 2010). A recent Meta analysis provides quantitative evidence supporting the conclusion that exposure to organochlorine pollutants is associated with an increased risk of incidence of diabetes (Tang et al. 2014). Organohalogen contaminants also may well contribute to the onset of obesity in more highly exposed populations (Hansen et al. 2014).

166. The general public, and in particular Arctic indigenous peoples, are exposed to SCCPs in addition to a wide array of other contaminants, including already regulated dioxin-like compounds such as PCBs, adding to the overall body burden. Several of these contaminants (including PCBs), already exceed guideline levels in Arctic aboriginal peoples, and new evidence indicates that POPs and heavy metals such as mercury and lead can affect the health of people and especially children at lower levels of exposure than previously thought (AMAP, 2009a). It has been demonstrated that the Arctic indigenous population is affected due to exposure to a variety of POPs and that the developing fetus, children, women of reproductive age and elderly are most at risk (AMAP, 2014). Lastly, it should also be recognized that, in addition to other chemicals, other stressors such as starvation episodes and reproductive and/ or developmental status as well as climate change induced effects may act alone or in concert to add to the severity of and risk for adverse effects, in the alpine regions and particularly in the Arctic (Sagerup et al. 2010, UNEP/AMAP Expert Group 2010, Letcher et al. 2010, Schröder et al. 2014). With view to the current changes in climate, it very important to protect the habitat diversity in the remote regions from additional stresses like organic pollutants.

3. Synthesis of Information

167. Total reported annual production and usage of SCCPs has been high globally. Although there have been notable reductions in recent years in some countries, in others production volumes of CP mixtures containing SCCPs have increased. Releases can occur during production, storage, transportation, use and disposal of SCCPs and SCCP containing products. Facility wash-down and spent metalworking/metal cutting fluids are amongst others sources to aquatic ecosystems. In industrialized, e-waste recycling areas, as well as in densely populated areas, high emissions to the environment are possible (Yuan et al. 2010, Chen et al. 2011, Luo et al. 2015, Gao et al. 2012). Although data are limited, the major sources of release of SCCPs have likely been the formulation and manufacturing of products containing SCCPs, such as polyvinyl chloride (PVC) plastics, and use in metalworking fluids.

168. SCCPs are not expected to degrade significantly by hydrolysis in water, and biodegradation studies and dated sediment cores indicate that they persist in sediment for much 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. Model data (OECD LRTP Screening Tool) indicate that SCCPs have properties similar to known POPs that undergo long range transport. Recently it was shown by modelled data that several SCCPs homologs have a high Arctic pollution potential. Concentrations measured in air, biota and sediment from remote Arctic and Antarctic locations also confirm that long range transport of SCCPs is occurring.

169. Bioaccumulation factors of 16,440 to 25,650 L/kg ww in trout from Lake Ontario as well as BAFs from 50,119 to 398,107 L/kg ww from eight fish species from the Liaodong Bay, indicate that SCCPs can bioaccumulate to a high degree in aquatic biota. In crustaceans (shrimp) that are especially sensitive to toxic effects from SCCPs field measured BAFs are up to 39,810 L/kg ww and 63,096 L/kg ww. The high bioaccumulation potential is further supported by modelling data for log Kow and bioaccumulation factors. BSAF values in bivlaves from the Bohai Sea have been calculated that are comparable to other POPs. In addition, biomagnification and trophic magnification factors for some SCCPs have been found to be greater than 1 in aquatic food webs (for some congeners TMFs <1 are reported). There is also indication for terrestrial biomagnification in birds, though data are limited.

170. SCCPs have been detected in remote areas such as the Arctic (notably in air, sediment and biota) and Antarctic (in air up to 20.8 pg/m^3). SCCPs found in Arctic blubber of beluga whales, ringed seals and walruses had mean SCCPs concentrations of 0.2, 0.5, and 0.4 µg/g lw, respectively (Tomy et al. 2000). Table 3-1 and 3-2 show that these concentrations are similar to concentrations of well-recognised POPs such as sum HCHs, PCB, DDT, sum PBDEs and toxaphene (AMAP 2004, Strid et al. 2013, Letcher et al. 2010). For air, higher concentrations of SCCPs were found than these POPs. More detailed comparisons of POP concentrations for the Arctic marine mammals, fish and for the birds are presented in table 3-1 and 3-2, respectively. SCCPs have been measured in the blubber of marine mammals from several locations in the Arctic at concentrations of 95 to 626 ng/g. In addition SCCPs have been detected in polar bear, ringed seals and sea birds from the Arctic indicating widespread contamination.

Table 3-1: Comparison by ratio of mean concentrations of SCCPs and POPs in Arctic species. (Tomy et al, 2000; AMAP 2004).

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

Table 3-2: Comparison of concentrations of SCCPs and POPs (ng/g lw) in Arctic Kittiwake, Kittiwake egg and Greenland shark (Reth et al, 2006; AMAP 2004, Strid et al. 2013, Letcher, 2010).

Species	[SCCPs]	[sumDDT]	[sumPCB]	[sum PBDE]	[sum HCHs]
Kittiwake	110-880 (n=2)	500-1,900	10,000-21,000	-	-
Kittiwake egg	100 (n=12)	806–1562	7,254–7,938	-	20 – 30
Common eider egg	17 (n=12)	-	Egg yolk: 262	Egg yolk: 2	-
Greenland shark liver	5,200 (n=15) 770 ng/g ww	7,195	990-10,000	9.9-200	53
Polar cod	10.3 ng/g ww (n=10)	-	-	2 [1]	-

171. Data on human exposure in the Arctic is limited, and so far the Arctic Monitoring Programme has not included measurements of SCCPs in human samples (AMAP 2014, 2009b). SCCPs have however been detected in the breast milk of Inuit women in Northern Quebec. In addition SCCPs have been detected in breast milk from women in the United Kingdom and Sweden. From the annual monitoring programme in Sweden no clear trend can be observed. The mean concentration of the mean annual values from 1998-2010 is 107 µg/kg lipid, concentrations range from 45 -157 µg/kg lipid (Darnerud 2012); whereas in the UK mean concentrations in 2002 were 180 µg/kg lipid (range: 49-820) (Thomas and Jones 2006). In the only study available on SCCP levels in breast milk from humans living in the Arctic Tomy (1997) reported concentrations ranging from 11-17 ng/g lipid (mean of 13 ng/g lipid) in breast milk of Arctic women living in northern Quebec, Canada (table 3-3).Indigenous people have higher body burdens than e.g. Russian population groups and might therefore also have higher body burdens of SCCPs (AMAP 2014).

Table 3-3: Comparison of mean concentrations of SCCPs and specific POPs in breast milk samples

Substance	Mean ng/g lipid in breast milk (range)	Population, sampling time
SCCPs	13 (11-17)	Inuit, Canada (Tomy 1997)
PBDE (Σ47,99,100, 153, 183)	4.72	Finnish Arctic 2005 (AMAP 2014)
Oxychlordane	4.9 (2.0-10)	Arkhangelsk (Russian) 2003 (AMAP 2014)
	5.2	Murmansk (Russian) 2000-2002 (AMAP 2014)
	41 (2-1070)	Chukotko (Indigenous) 2001-2002 (AMAP 2014)
Toxaphene	5	Arkhangelsk (Russian) 2003 (AMAP 2014)
	7.3	Murmansk (Russian) 2000-2002 (AMAP 2014)
	9 (0.5 -112)	Chukotko (Indigenous) 2001-2002 (AMAP 2014)

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

173. SCCPs have been measured in surface and seawater, sediments, soils, sewage sludge and air with most data available from Europe, North America and East Asia. While concentrations decline in some regions a recent increase in environmental matrices was observed in others. No temporal trend can be established for remote regions. Though recent advances in the analysis of SCCPs have been reported measurements of SCCPs remain challenging.

174. Table 3-4 presents exposure values for organisms at sites close to likely sources of release, at more distant sites influenced by regional sources, and sites remote from known or significant sources. Also listed are toxicity values relevant to the type of exposure reported – for example, expressed as exposure to the ambient environment, as doses in food, or as body burden. Several factors, including timing of exposure, transgenerational impacts, potential for endocrine disruption as well as mixture effects are not reflected in these studies and can cause effects at much lower levels, which is not captured in these tables. Also other endpoints may be relevant, such as behavioural effects, which have not been investigated in the studies listed. According to Environment Canada for persistent and bioaccumulative substances the risks may be underestimated using standard risk assessment methods (Environment Canada, 2008).

Table 3-4: Exposures of organisms/receptors at sites close to likely sources of release (Local), more distant sites influenced by regional sources (Regional) and sites distant from known or significant sources (Remote), and relevant toxicity values.

Receptor	Exposure concentration	Sample	Relevant toxicity value	Notes
Local				
Fish-eating mammals (otter)	2.63 mg/kg ww	Carp from Hamilton Harbour, Lake Ontario ng/g ww	1,000 mg/kg food ww Estimated No Exposure Value: 100 mg/kg food ww (AF of 100)	– Assuming concentration in carp representative of food consumed. Measured SCCPs in carp collected in Lake Ontario in 1996 and 2001 (Muir et al. 2001; 2002) – See Section 2.5.2 for calculation of relevant toxicity value. - Reference for exposure concentration and AF (Assessment Factor): Environment Canada (2008)
Pelagic invertebrates	176 ng/L	Lake water receiving STP effluents, Beijing, China	8,900 ng/L Estimated No Exposure Value: 890 ng/L (AF of 10)	– Pelagic invertebrate exposed to lake water (Zeng et al. 2011a). – 21-day chronic LOEC value for *Daphnia magna* (Thompson and Madeley 1983a) - Reference for AF: Environment Canada 2008
Pelagic fish	2.63 mg/kg ww	Carp from Hamilton Harbour, Lake Ontario	0.79 mg/kg ww	– Comparing concentrations measured in carp in Hamilton Harbour. – LOEC for observed severe liver histopathologies, extensive fibrous lesions and hepatocyte necrosis of rainbow trout (Cooley et al. 2001)

Receptor	**Exposure concentration**	**Sample**	**Relevant toxicity value**	**Notes**
Regional				
Pelagic invertebrates	1.19 ng/L	Water, mid-Lake Ontario	8,900 ng/L Estimated No Exposure Value: see above	– Assuming exposure of pelagic invertebrates to Lake Ontario water (Houde et al. 2008). – 21-day chronic LOEC value for *Daphnia magna* (Thompson and Madeley 1983a)
Marine invertebrates	13.1 ng/L (coastal mud shrimp 502 ng/g ww)	Seawater from Liaodong Bay, China (Ma et al. 2014b)	5 µg/L PNEC: 100 ng/L, (AF of 50)	- 21d- NOEC *Daphnia magna* 5 µg/L; Predicted no effect concentration (PNEC) for marine water of 0.1 µg/L, AF of 50 (according to U.K. Environment Agency 2007)
Pelagic fish	2.9 mg/kg ww	Japanese seabass from Liaodong Bay, China	0.79 mg/kg ww	– Comparing concentrations measured in *Lateolabrax japonicus* (Ma et al. 2014b) with LOEC for severe liver histopathologies, extensive fibrous lesions and hepatocyte necrosis of rainbow trout (Cooley et al. 2001)
Benthic invertebrates	1.8 mg/kg dw	Bohai sea surface sediment	35.5 mg/kg dw Estimated No Exposure Value: 3.55 (AF of 10)	–Max. sediment concentration measured in surface sediments from the Bohai Sea (Ma et al. 2014a) – LOEC calculated using the LOEC for *Daphnia magna* using equilibrium partitioning approach (Environment Canada, 2004) - Reference for the AF: Environment Canada (2008)
Benthic fish	0.69 mg/kg ww	Flathead fish from Liaodong Bay, China	0.79 mg/kg ww	– Max. concentration in *Platycephalus indicus* from the Liaodong Bay, China (Ma et al. 2014b). – LOEC for observed severe liver histopathologies, extensive fibrous lesions and hepatocyte necrosis of rainbow trout (Cooley et al. 2001)
Remote				
Benthic invertebrate	0.0176 mg/kg dw	Surface sediment Lake DV09 Canadian Arctic	35.5 mg/kg dw Estimated No Exposure Value: see above	– Calculated using the LOEC for *Daphnia magna* using equilibrium partitioning approach (Environment Canada, 2004)
Arctic ringed seal	0.52 mg/kg ww	Ringed seal blubber	100 mg/ kg ww $PNEC_{oral}$ 5.5 mg/kg food	– Results for seals from Tomy et al. 1999 - $PNEC_{oral}$ from U.K. Environment Agency. (2007) based on a NOEC of 166 mg/kg food of a mallard duck reproduction study and an AF of 30.
Ringed seal from East Greenland	0.52 mg/kg ww	Ringed seal blubber	100 mg/ kg ww $PNEC_{oral}$ 5.5 mg/kg food	- Results from Tomy et al. 1999 - $PNEC_{oral}$ from U.K. Environment Agency. (2007) based on a NOEC of 166 mg/kg food of a mallard duck reproduction study and an AF of 30.

175. Table 3-5 presents scenarios for exposure of humans to SCCPs. The first two scenarios assume consumption of a traditional food diet. Exposure was calculated using SCCP concentrations measured in the Arctic in ringed seal blubber, beluga whale blubber and walrus blubber (from Tomy et al 1999 and Muir et al. 2004 NCP, 2013), and using dietary intake from Kuhnlein (1995a) and Kuhnlein et al. (1995b). The scenarios for a breast-fed child assumes intake of 750 mL milk/day and 3% lipid content (Van Oostdam et al. 1999). In addition, elevated levels of SCCPs in human breast milk in remote communities have been reported. The upper bound estimation in the Canadian population of SCCPs intake was exceeding the TDI for long term effects (Environment Canada 2008, 2013). Especially long term exposure of children is of concern, by an estimated intake value of 24.6 µg/kg bw per day. Although if there is low confidence in the upper-bounding estimates, they were basis to conclude that SCCPs constitute or may constitute a danger in Canada to human life or health (Environment Canada, 2008). Further it has to be taken into consideration that children are vulnerable towards chemicals exposure due to development and growth processes.

Table 3-5: Scenarios for exposure of humans, and relevant toxicity values

Receptor	Exposure	Sample	Relevant toxicity value	Comments
Humans (75 kg)	0.15-0.37 µg/kg body ww/d	Estimated dietary exposure Baffin Island male Inuit	125,000 ug/kg body ww/d	LOEL, 2 year rat study (NTP, 1986)
Humans (75 kg)	0.15-0.37 µg/kg body ww/d	Estimated dietary exposure Baffin Island male Inuit	100 µg/kg bw/d	TDI, non-neoplastic effects, 13 weeks study in rats; AF: 100 (IPCS, 1992)
Humans (75 kg)	0.15-0.37 µg/kg body ww/d	Estimated dietary exposure Baffin Island male Inuit	11 ug/kg body ww/d	The 11 ug/kg bw/d was derived using a safety factor of 1000 applied to a value of 11 mg/kg bw, based on multistage modelling of tumours with the highest incidence in the carcinogenesis bioassay of male mice that resulted in 5% increase in tumour incidence (IPCS, EHC 181, 1996)
Breast fed child (5 kg)	0.0585 ug/kg body ww/d	Human milk: 13 ug/kg lipid ww	125,000 ug/kg body ww/d	LOEL, 2 year rat study (NTP, 1986)
Breast fed child (5 kg)	0.0585 ug/kg body ww/d	Human milk: 13 ug/kg lipid ww	100 µg/kg bw/d	TDI, non-neoplastic effects, 13 weeks study in rats; AF: 100 (IPCS, 1992)
Breast fed child (5 kg)	0.0585 ug/kg body ww/d	Human milk: 13 ug/kg lipid ww	11 ug/kg body ww/d	The 11 ug/kg bw/d was derived using a safety factor of 1000 applied to a value of 11 mg/kg bw, based on multistage modelling of tumours with the highest incidence in the carcinogenesis bioassay of male mice that resulted in 5% increase in tumour incidence (IPCS, EHC 181, 1996)

176. The International Agency for Research on Cancer considers SCCPs (average C_{12}, average 60% chlorination) to be possible carcinogens (groups 2B), although questions have been raised regarding the mechanisms for induction of tumours and the relevance for human health of the studies on which this classification was derived. NTP has listed SCCPs (C12, 60% chlorine) in the 13th report on carcinogens as reasonably anticipated to be human carcinogens. The EU Risk assessment (EC 2000) concluded that the concern for humans could not be ruled out. A tolerable daily intake (TDI) for SCCPs of 100 µg/kg-bw per day was developed by IPCS (1996); for neoplastic effects a dose of 11 µg/kd bw/day should not be exceeded (IPCS 1996). The most recent assessment of the EU stated that no NOAEL can be obtained from the chronic studies for mammals. Environment Canada comes to the conclusion that the upper bound estimation of SCCPs intake is exceeding the TDI for long term effects and assumes that the margin of safety is not sufficient. Also it is mentioned that some justification for considering a somewhat lower value as TDI exists, to take into account potential progression of the lesions in longer-term studies (Environment Canada 2008, 2013). Further it has to be considered that Arctic indigenous people are exposed to various POPs, several of those as well classified carcinogens.

4. Concluding statement

177. SCCPs have been used worldwide in a wide range of applications such as plasticisers in plastics, in metalworking fluids, flame retardants and additives in paints. Their wide industrial applications probably have provided the major source of environmental contamination. Environmental releases occur during production, service-life and disposal of SCCPs and SCCPs containing products.

178. SCCPs are atmospherically, via water or migratory species transported to Antarctic, Arctic and subarctic regions far from local sources. Several SCCP homologues are persistent in the environment and investigations on food webs and food chains confirm several SCCPs accumulate in invertebrates, freshwater and marine fish at high levels. Laboratory studies have shown adverse effects on fish at low concentrations. Some SCCPs accumulate and biomagnify in Arctic biota including mammalian species. In mammals, SCCPs affect the liver, the thyroid hormone system, and the kidneys.

179. SCCPs have been measured in sediments also in Arctic lakes. SCCPs are particularly toxic to aquatic invertebrates. Given the key role that invertebrates play in aquatic ecosystems, there is

therefore high concern relating to potential for effects of SCCPs on sediment-dwelling and other invertebrates.

180. Although concentrations in remote areas are low, SCCPs have been measured in Arctic biota at levels comparable to already listed POPs. Notably, SCCPs are present in Arctic biota, which are in turn food for northern indigenous people. SCCPs are present in human breast milk both in temperate and Arctic populations.

181. SCCPs have been shown to affect the thyroid, liver and kidneys, which in the long-term can lead to carcinogenicity in these organs as it was seen in animal experiments with laboratory rodents. SCCPs are suspected to cause cancer in humans and disrupt endocrine function also. Upper bound exposure concentrations exceed the respective acceptable exposure levels.

182. SCCPs have been measured in Arctic biota at concentrations similar to other known POPs. Simultaneous exposure to POPs may be of concern. No temporal trend of SCCPs can currently be established for remote areas. Given demonstrated long range transport and ability to accumulate, there is potential for increases in environmental levels should releases continue or increase.

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

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