Chapter 2

Chlorinated paraffins in the environment: a review on their production, fate, levels and trends between 2010 and 2015

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Based on: Chemosphere (2016), vol.155, p.415-428
Abstract
This review provides an update on information regarding the production volumes, regulations, as well as the environmental levels, trends, fate and human exposure to chlorinated paraffin mixtures (CPs). CPs encompasses >10,000 congeners with varying properties and environmental fate. Based on their carbon chain lengths, CPs are divided into short- (SCCPs; C_{10-13}), medium- (MCCPs; C_{14-17}) and long- (LCCPs; C≥18) chain groups. They are high production volume and persistent chemicals, and their cumulative global production already surpasses that of other persistent anthropogenic chemicals. However, international regulations are still curbed by insufficient information on their levels and fate, including bioaccumulation, and toxicity potential. An increasing number of studies since 2010 demonstrate that CPs are detected in almost every compartment in the environment, including remote areas. Consensus on the long-range transport and high bioaccumulation potential (bioconcentration factor >5000 and trophic magnification factor >1) has recently been reached for SCCPs, partially fulfilling criteria under the Stockholm Convention for designation as a persistent organic pollutant. Information on their levels is, however, still sparse for many countries. M/LCCPs have received comparatively little attention in the past, but as replacement chemicals for SCCPs, MCCPs are now considered in an increasing number of studies. The limited data to date suggest MCCPs are widely used. Although data on their bioaccumulation and toxicity are still inconclusive, MCCPs and LCCPs with C_{<20} may also have a bioaccumulation potential. Considering this and their high production volumes, use, and ubiquitous occurrence in the environment, a better understanding on the levels and fate of all CPs is needed.
2.1 Introduction

To assess the impact of a contaminant on the environment, information on its toxicity, fate and levels is essential. For chlorinated paraffins (CPs), knowledge across these areas is relatively scarce, although concerns regarding their high production volumes and their potential persistent, bioaccumulative and toxic characteristics (PBT) are rising [1, 2]. Of the CP groups, short-chain CPs (SCCPs) have the highest toxicity, bioaccumulation and long-range transport potential and are therefore under particular scrutiny [2]. For medium-chain CPs (MCCPs) conclusive data are still limited [1, 3], while recent information on long-chain CPs (LCCPs) is lacking.

The scarcity of data is mainly caused by the complexity of CP mixtures. For example, an SCCP mixture with 60% chlorine by weight already can comprise more than 4,200 congeners [4]. CP analysis, toxicity and environmental fate assessments are, therefore, extremely challenging. To date, no fully validated routine analytical method is available to monitor CPs in environmental samples and only semi-quantitative analysis is possible. Information on the toxicity of S/MCCPs and SCCPs was recently summarised by Lassen et al. [5], the persistent organic pollutant (POP) review committee [2] and Geng et al. [6], and are beyond the scope of this review. Briefly, among all CP groups, SCCPs appears to have the highest toxic potential. While their acute toxicity to animals is low, they are carcinogenic for rodents (No observed adverse effect level (NOAEL) 10 mg kg⁻¹ d⁻¹ for rats), and they are considered highly toxic to aquatic organisms (No observed adverse concentrations and NOAEL in the low µg L⁻¹ range for invertebrates and fish; 2). Data on the underlying toxicity mechanisms of CPs (i.e. biological events involved in the cellular response) are, however, limited and further studies are in high demand [6].

The number of studies on levels, trends and fate of CPs has increased rapidly over the last five years. Particularly in China, numerous studies have appeared since 2010 (>30). The complexity of CPs is also reflected in the classification of CP congeners and congener groups, which in some cases is still unclear [7]. An increasing number of studies are reporting on CP congener groups and/or ‘congener’ levels since 2010. Each individual CP is a congener, while a group of congeners with the same number of chlorine atoms and carbon chain lengths is a congener group [7]. Hence, in this review we distinguish between in four categories: i) groups (i.e. SCCPs, MCCPs, LCCPs), ii) CPs with the same carbon chain length (e.g. C_{10}–SCCPs) or same chlorine content (e.g. Cl₆–SCCPs), iii) congener group (e.g. C_{10}H_{16}Cl₆) and iv) congeners (e.g. 2,3,5,6,8,9-hexachlorodecane).

In 2010, a comprehensive review was published about the production and use of CPs, and their physico-chemical properties, environmental fate, effects, levels and trends [8]. Because of the availability of a considerable amount of new data, the present review aims to provide an update of this information. More specifically, we discuss the current production, applications and regulations as well as their physico-chemical properties, environmental processes and occurrence, including spatial and temporal trends and
reported congener group distributions. In addition, current research and knowledge gaps are highlighted.

2.2 Production, use and regulations

2.2.1 Production
Commercial CP mixtures consist of polychlorinated \( n \)-alkanes produced by radical chlorination of \( n \)-alkane mixtures (via petroleum feedstocks) under forcing conditions, such as ultraviolet (UV) light and/or high temperatures, which have low positional selectivity [9]. Based on the carbon feedstocks, commercial mixtures are divided into three groups: SCCPs (\( C_{10-13} \), MCCPs (\( C_{14-17} \)) and LCCPs (\( C_{>18} \)). LCCPs are further subdivided into three subgroups according to their carbon feedstocks and physical state (\( C_{18-20} \) liquid products, \( C_{20-25} \) liquid products, \( C_{>25} \) solid products). Commercial mixtures are also subcategorised by their content of chlorine on a weight basis: 40-50%, 50-60%, and 60-70% [10].

Although the purity of the feedstocks has increased since the start of CP production in 1930, the final commercial product still contains impurities such as iso-paraffins (ca. 1-2%), aromatic compounds (<0.5%), metals and unreacted \( n \)-alkanes. In addition, stabilisers such as epoxides and/or organotins (usually <0.5%) are added during the chlorination process to inhibit decomposition of CPs by loss of HCl at temperatures \( >300 \) °C [9].

Information on production, import and use of CPs is limited and difficult to determine, especially on a group level (i.e. S/M/LCCPs). For example, in industrial mixtures from China, distinction between CPs are made based on chlorination degrees rather than on carbon chain lengths, resulting in no official information on the distribution of S/M/LCCPs [2]. Furthermore, at least 40 CAS numbers have been used worldwide for CPs [11]. Based on the limited publically available information, CPs are high production volume chemicals (>1 million metric tonnes yr\(^{-1} \); Figure 2-1), with a cumulative production of >7 million tonnes since the 1930s [12]. To our knowledge, no other persistent anthropogenic chemical has been produced in such quantities. By comparison, the cumulative global production volume of polychlorinated biphenyls (PCBs) is estimated at ca. 1.3 million tonnes [13]. Although production of SCCPs has stopped in the US, Japan, Canada and Europe, M/LCCPs are still produced as alternatives in those regions, resulting in similar or even increasing total CP production volumes per year. Stiehl et al. [14] further suggest that with the ban of pentabromodiphenyl ether, the use and production of CPs as a flame retardant could increase even more. The global rise of CP production volumes comes primarily from China (Figure 2-1), with most recent annual production rates of 1.1 million tonnes in 2013 (15% of the total global CP production), whereas China’s production capacity has reached 1.6 million tonnes/yr (a 14% increase compared to 2012) [15]. Temporal trends in dated sediments cores reflect this rapid increase in CP production and use, showing a marked increase in SCCPs accumulation from the 1950s onwards in China, [16], and a shift to MCCPs in recent
years [17]. In 2014, 100-140 CP producers were identified in China, with the main production provinces located in East China. The most commonly produced mixtures are CP-52 (90% of the total Chinese CP output), CP-42 and CP-70 [18]. Recently, Gao et al. [19] determined that at least CP-52 contains a relatively high proportion of SCCPs (mass fraction 24.9%). CP production volumes also appear to be still increasing in India, although current data are lacking (Figure 2-1). In Europe, the total registered manufacture and import of SCCPs and MCCPs is indicated to be between 1,000-10,000 and 10,000-100,000 tonnes/yr respectively [5]. Other CP producing countries include Thailand, Taiwan, Australia and Japan [2, 20], but data on their production volumes are unavailable and cannot be elucidated from the sparse publically available information.

**Figure 2-1:**
Summary of identified information on CP production volumes between 2007-2013. Missing information is blank; “-“ denotes SCCP production bans [2,15,18,79].
2.2.2 Applications

Because of their varying carbon chain lengths and chlorine content, high chemical stability, flame resistance, viscosity, low vapour pressure, strength at low temperatures and low costs for production, CPs are used for a wide range of applications [10]. For example, CPs are used as coolants and lubricants in metal working fluids and flame retardants. They are also used as plasticisers in polymers, mainly in polyvinyl chloride (PVC) products [5], as well as other plastics, including rubber (e.g. underground mining conveyor belts). Other applications are paints, sealants and adhesives (e.g. double-glazed windows and dam sealants) [11]. CPs have also been reported to be applied for fat liquoring of leather [21]. The US prohibited the use of SCCPs since 2013, while the EU restricted their use since 2011, except for the use as fire retardant in conveyor belts in the mining industry and dam sealants [22]. The Japanese metalworking industry voluntarily phased out SCCPs since 2007 [23]. However, SCCPs are still used at rates of several 100 ktonnes/yr worldwide [21]. In addition, large amounts of plastic products with unknown quantities of CPs and thus possibly SCCPs are imported by Europe from Asia [20]. CPs found in sewage sludge in Sweden represented 4.1% of the national recorded annual CP use, which was approximately ten times higher compared to that of other plasticisers (i.e. 0.04% for phthalates) with similar physico-chemical properties and thus presumably similar emission rates [24]. The authors suggested that considerable amounts of CPs are entering the country as additives in imported products (e.g. textile and plastic). These are not included in national use statistics, which results in much higher actual use than reported. In Europe, the main applications of SCCPs in 2011 included the rubber industry, followed by sealants and adhesives, paints and varnishes [22]. SCCP use for these applications has increased from 638 to 1,254 tonnes/year in 1998 and 713 to 796 tonnes/yr in 2011. MCCPs are mainly used as additives in PVC in Europe (ca. 79-83% of use). Other minor applications in Europe include metal cutting/working fluids (5-9%), paints and sealants (4-5%), rubber and plastics other than PVC (3-4%), leather fat liquors (2-3%), and carbonless copy paper (1-2%) [1]. In India, the use of CPs as additives in PVC is estimated to be 200-340 ktonnes yr⁻¹, based on limited available information from the polymer industry [20].

Overall, it is evident that more public information on the amounts of CPs produced and used is required, including trends over time. This is particularly the case for CP producing regions where production might be high, such as India, or where information of production quantities is entirely lacking (e.g. Taiwan, Thailand and Australia). These data will help prioritise which CPs merit study and will assist in modelling their environmental occurrence, fate and transport. Nonetheless, it can already be concluded that, based on the limited information available, production volumes and use of CPs are in such high quantities that understanding their potential releases into the environment becomes essential.
### Table 2-1 Physico-chemical properties and important classification of CPs.

<table>
<thead>
<tr>
<th></th>
<th>SCCPs</th>
<th>MCCPs</th>
<th>LCCPs C18–20 liquid</th>
<th>LCCPs C&gt;20 liquid</th>
<th>LCCPs C&gt;20 solid</th>
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</thead>
<tbody>
<tr>
<td>Molecular formula</td>
<td>$\text{C}<em>x\text{H}</em>{(2x-y+2)}\text{Cl}_y$ where $x = 10$ to $13$ and $y = 3$ to $x$</td>
<td>$\text{C}<em>x\text{H}</em>{(2x-y+2)}\text{Cl}_y$ where $x = 14$ to $17$ and $y = 3$ to $x$</td>
<td>$\text{C}<em>x\text{H}</em>{(2x-y+2)}\text{Cl}_y$ where $x = 18$ to $20$ and $y = 3$ to $x$</td>
<td>$\text{C}<em>x\text{H}</em>{(2x-y+2)}\text{Cl}_y$ where $x &gt; 20$ and $y = 3$ to $x$</td>
<td></td>
</tr>
<tr>
<td>CAS No.</td>
<td>85535-84-8</td>
<td>85535-85-9</td>
<td>various</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Log Kow</td>
<td>$4.71-6.93$ (30-70% Cl)$^c$</td>
<td>$5.47 - 8.21$ (32-68% Cl)</td>
<td>$7.34 - 7.57$ (34-54% Cl)</td>
<td>$7.46 - 12.83$ (42 - 49% Cl)</td>
<td>NA</td>
</tr>
<tr>
<td>Sw (μg/L)</td>
<td>$6.4 - 2370$ (48-71% Cl)</td>
<td>$9.6 \times 10^{-2} - 50$ (37-56% Cl)</td>
<td>$0.017 - 6.1$ (34-54% Cl)</td>
<td>$1.6 \times 10^{-6} - 6.6$ (41.9 - 50% Cl)</td>
<td>$1.6 \times 10^{-11} - 5.9$ (70-71% Cl)</td>
</tr>
<tr>
<td>Vp (Pa)$^b$</td>
<td>$2.8 \times 10^{-2} - 0.028$ (48-71% Cl)</td>
<td>$4.5 \times 10^{-8} - 2.3 \times 10^{-3}$ (42-58% Cl)</td>
<td>$2 \times 10^{-5} - 5 \times 10^{-3}$ (40-52% Cl)</td>
<td>$3 \times 10^{-15} - 2.7 \times 10^{-3}$ (40-54% Cl)</td>
<td>$1 \times 10^{-23} - 3 \times 10^{-14}$ (70% Cl)</td>
</tr>
<tr>
<td>HLC (Pa x m$^3$/mol)</td>
<td>$0.68 - 17$ (48-56% Cl)</td>
<td>$0.014 - 51$ (37-56% Cl)</td>
<td>$0.021 - 55$ (34-54% Cl)</td>
<td>$0.003$ (50% Cl)</td>
<td>$3.6 \times 10^{-7} - 5.6 \times 10^{-6}$ (70-72% Cl)</td>
</tr>
<tr>
<td>Log Koa</td>
<td>$4.86 - 13.71$ (30-70% Cl)$^d$</td>
<td>$8.81 - 12.96$ (32-68% Cl)$^e$</td>
<td>$9.21 - 12.12$ (34-54% Cl)</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>Log Koc</td>
<td>$4.10 - 5.44$</td>
<td>$5.0 - 6.23$</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>Classifications</td>
<td>SCCPs C18–20 liquid</td>
<td>MCCPs</td>
<td>LCCPs C18–20 liquid</td>
<td>LCCPs C&gt;20 liquid</td>
<td>LCCPs C&gt;20 solid</td>
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</tr>
<tr>
<td>UNEP [2]</td>
<td>≥48% Cl: POP candidates, B and LRTP</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>UNEP [26]</td>
<td>LRTP: significant adverse human health &amp; environmental effects</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Environment Canada [27]</td>
<td>P, B, toxic to the environment &amp; posing a threat to human health; Track 1 priority toxic substances</td>
<td>P, B, toxic to the environment &amp; posing a threat to human health</td>
<td>P, B, toxic to the environment &amp; posing a threat to human health</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>US EPA [29, 36]</td>
<td>40 – 70% Cl: P, very B, toxic to aquatic environment</td>
<td>May be very P &amp; very B, may present an unreasonable risk following acute and chronic exposures to aquatic organisms</td>
<td>May be very P &amp; very B, may present an unreasonable risk following acute and chronic exposures to aquatic organisms</td>
<td>May be very P &amp; very B, may present an unreasonable risk following acute and chronic exposures to aquatic organisms</td>
<td>May be very P &amp; very B, may present an unreasonable risk following acute and chronic exposures to aquatic organisms</td>
</tr>
<tr>
<td>EU REACH [11, 37, 38]</td>
<td>Very P &amp; very B, T (NOEC&lt;0.01 mg L⁻¹), Substances of very high concern Carcinogenic and very toxic to the aquatic environment</td>
<td>Very P &amp; B candidates Carcinogenic and very toxic to the aquatic environment, may cause harm to breast-fed children</td>
<td>Very P</td>
<td>Very P</td>
<td>Very P</td>
</tr>
</tbody>
</table>

B bioaccumulative (BCF≥5000), BCF bioconcentration factor, CLP classification, labelling and packaging regulation, EPA Environmental Protection Agency, HLC Henry Laws Constant, Log KOA log octanol-air coefficient, Log KOC log octanol-carbon coefficient, Log KOW log octanol-water coefficient, LRTP Long-range transport potential, NA not available, NOEC No observed effect concentration, UNEP United Nations Environment Program, Sw Water solubility, POP persistent organic pollutant, P persistent, Vp Vapour pressure. a From Environment Canada Canada [27], unless stated otherwise; b Vapour pressure values not given at a consistent temperature; c subcooled liquid vapour pressure data, from [10]; d Estimated from ratio of Kow / HLC (unitless); e Data from Gawor and Wania [33].
2.2.3 International classifications and regulations

PBT assessments on M/LCCPs are still ongoing in Europe and the US (Table 2-1). At present, SCCPs, and in some cases MCCPs and a subset of LCCPs (C_{18-20}), are classified by several regulatory agents as environmentally hazardous substances, especially to the aquatic environment (Table 2-1). Regarding their hazard for humans, SCCPs are listed as suspected carcinogenic substances (Carcinogenic Category 3) [25], while MCCPs may cause harm to breast-fed children (Lact H362) [1]. SCPs and MCCPs are furthermore both on the EU candidate list of endocrine disruptors.

Recently, SCCPs have been recommended by the Chemical Review Committee (CRC) to be included in Annex III of the Rotterdam Convention as industrial chemicals [26]. They are also currently candidates for classification as persistent organic pollutants (POPs) under the Stockholm Convention. In the latest adapted risk profile document, it was concluded that “SCCPs are likely, as a result of their long-range environmental transport, to lead to significant adverse human health and environmental effects such that global action is warranted” [2].

SCCPs and products containing >1 wt% SCCPs are also included on the list of the European Water Framework Directive, the US EPA Toxic Release Inventory list [2], the prohibited substances list in Canada, and the Pollutant Release and Transfer Register Law in Japan [15]. The ban on production, sale and use of SCCPs in Europe, Canada and the US include products containing >1 wt% SCCPs [15]. However, in Europe’s mining industry, SCCP containing products, including products with long life spans, are still permitted [22], and it is not expected that products (i.e. paints, adhesives, sealants), in which SCCPs have previously been used in concentrations well above the 1%, would now be produced with an intentional content of SCCPs below 1% [5].

To our knowledge, no regulations on M/LCCPs exist as yet, although Canada has proposed their virtual elimination on M/LCCPs [27] and Oslo and Paris Conventions for the protection of the marine environment of the Northeast Atlantic (OSPAR) proposals exist to take action on LCCPs [28]. Furthermore, based on their concern stated in the M/LCCPs draft risk assessment, the US EPA has recently requested that manufacturers discontinue their production by May 2016 [29].

2.3 Environmental processes and exposure pathways

Key parameters for assessing the environmental fate of a compound are the uptake, resistance to degradation, bioaccumulation and long-range transport potential [2]. These parameters depend on the environmental conditions as well as on the physico-chemical properties of the compound. The properties of CPs, however, vary among congeners and are at present difficult to determine [7]. Available data are usually provided by a specific combination of carbon-chain length distribution and chlorine content (%), typically with large variability and therefore difficult to interpret. While the knowledge on how chlorine content and carbon chain length affects these properties and hence parameters is still limited and needs further validation [30-32], new data were recently provided for SCCPs [33].
2.3.1 Physico-chemical properties

CPs are also known as polychlorinated \( n \)-alkanes. CPs, in which paraffins means “not enough affinity” [L. par(um), not enough, + afins, affinity], are in general unreactive compounds. At room temperature, none of the C-H or C-C bonds are attacked by a strong acid (e.g. \( \text{H}_2\text{SO}_4 \)) or oxidizing agents (e.g. potassium permanganate). However, at elevated temperatures (i.e. 300 °C) rapid HCl emission occurs [9]. Properties of CPs relevant to the scope of this review are summarized in Table 2-1 on a (sub)group level. Many of the reported physico-chemical properties are derived via modelling rather than empirically. All CPs are in general poorly water soluble, semi-volatile and have high octanol-water (\( K_{\text{OW}} \)) and high octanol–air partition coefficients (\( K_{\text{OA}} \)).

Some recent studies investigated the influence of chlorine content and substitution, as well as carbon chain length on the physico-chemical properties of SCCPs [33-35]. For mixtures with a chlorine content of 45-55%, no significant differences in \( K_{\text{OW}} \) values were observed, but when chlorine content increased from 55% to 70% the log \( K_{\text{OW}} \) value increased according to a second order polynomial function [31, 32, 34]. At a given chlorine content, a positive linear relationship was found between carbon chain length and \( K_{\text{OW}} \) [10, 34, 35]. Furthermore, SCCP congeners that contain areas along the carbon chain where chlorine atoms are more concentrated have higher \( K_{\text{OW}} \) values compared to SCCPs that have more evenly distributed chlorine atoms over the carbon chain length [34]. This highlights the influence of the chlorine arrangement in CPs on their properties. Other effects of chlorine substitution on other physico-chemical properties have been extensively described in Glüge et al. [35].

2.3.2 Releases to the environment

Unintentional production of CPs has not been reported and it appears that CPs are exclusively anthropogenic compounds. Releases occur during production, storage, transportation, use, recycling and disposal of CPs, as well as CP containing products [7]. Even in countries where SCCPs are banned, they are still being released into the environment from old materials that remain in use or have been disposed [22], or via imported products. Underground mining conveyor belts in Europe are, for instance, products that still contain SCCPs. They are considered as major source of the total SCCP load released to the European waste stream (55%; 2,000 tonnes in 2011), followed by sealants (20%) and paints 14%) [22]. Olofsson et al. [24] suggested that CPs could presumably evaporate at low rates from (imported) plastic materials, partition to dust, and reach the sewage system after cleaning of indoor environments. Although the emission rate is expected to be low, the stock of plastic materials in society is large and the resulting total emitted amounts may be substantial.

A particular important pathway of CPs to the environment is the release of sludge and effluents from sewage treatment plants (STPs). Zeng et al. [39] reported total SCCP removal efficiencies of 83% from the aqueous phase in a Chinese STP. They found relatively high mass fractions of the initial \( \Sigma \text{SCCPs} \) load (73%) as well as high
concentrations (0.80−53 μg/g dw) [40] in the final dewatered sludge, implying the need for consideration of CPs during sludge management. Although only 0.8% of the initial SCCPs mass was found in the effluents [39], effluent discharge from STPs was found to be a significant source of SCCPs contamination to waste water irrigated farmlands [41], river sediments [19] and receiving waters [42]. Especially lower chlorinated SCCPs (e.g. Cl5-6) may have a potential to pass more easily through the STP treatment [42], as relatively higher abundance of Cl5-6–SCCPs was found in effluent and lake water compared to that in the influent and sewage sludge [39]. Gao et al. [19] showed that industrial wastewater discharges in north-eastern China (Liaho River Basin) result in SCCP contamination of the environment, and estimated an average discharge of SCCPs of approximately 74 mg/tonne industrial wastewater.

Waste is another source of CPs to the environment. Luo et al. [43] showed high SCCP concentrations in birds foraging within e-waste recycling areas in China (620-17,000 ng/g lw), and Lassen et al. [5] suggested that waste consisting of rubber, sealants, adhesives, paints and textiles is of main concern for the potential of SCCPs entering the environment in Denmark.

Other CP release pathways include leaching, runoff or volatilization from landfills, facility wash-downs and burning of waste [2]. Recently, high CP concentrations were found in effluent laundry water in Lithuania with up to 53 and 170 mg/L for SCCPs and MCCPs, respectively [44]. Another potential source is the renovation and demolition of buildings, as CPs have frequently been used in building materials, i.e. double-glazed windows, sealants and adhesives [5].

Little is known about the pathway and quantities of M/LCCPs entering the environment [22], although in recent years environmental contamination by MCCPs has raised more and more public concern [29]. Clearly, the numerous release options show the need for understanding the occurrence, fate and effects of these contaminants in the environment.

2.4 Environmental accumulation

CPs can transfer between air, water, sediment, soil and organisms, and have the potential to accumulate in both abiotic and biotic phases. Although CPs have a very low vapour pressure at room temperature, S/MCCPs can be volatilised from products and during their diverse applications [45]. SCCPs with a shorter carbon chain length and lower chlorine content are more prone to volatilization and migration compared to longer chain CPs, which have greater affinity for particulates in water and air [7]. Recently, Li et al. [46] modelled the OH radical degradation of some SCCPs in the atmosphere and showed that C10-12Cl5-8–SCCPs tend to have long atmospheric lifetimes, which is consistent with the SCCP congener group pattern in air (Section Composition profiles2.5.5). With atmospheric half-lives of 0.8-10.5 days for SCCPs [2] and 1-2 days for MCCPs [25], at least some S/MCCPs are persistent enough in air to undergo atmospheric long-range transport, and evidence to date show that SCCPs have a long-range transport potential (LRTP) [2, 37]. In addition, with their high KOA values,
sorption of MCCPs to aerosols can be expected to increase the half-life in air [47]. Indeed Ma et al. [48] showed that sorption to aerosol organic matter had a significant effect on the transfer and partitioning of S/MCCPs to remote places (Antarctica) and that long-range atmospheric transport was a major input route to this region.

As hydrophobic compounds, CPs are less likely to be present in water at high concentrations. However, SCCPs have been detected in marine waters near point sources (e.g. Bohai Sea, China) at concentrations of 4.1-13 ng/L [49]. CPs are not expected to degrade significantly via hydrolysis in water [2, 25, 38], although it appears that under enhanced conditions (pH 9.0), some chlorine atoms (0.24%) of SCCPs are released [50]. As surface sea water usually has a pH of 8 [51], SCCPs that are present in water are persistent enough for long-range oceanic transport to occur.

While persistency of CPs in soil remains largely unknown, degradation studies and dated sediment cores indicate that SCCPs have half-lives in sediment of longer than 1 yr [2]. Even in the presence of adapted micro-organisms, only slow biodegradation of SCCPs in the environment is expected [10], which is, based on the relative similarity in physico-chemical properties, likely also the case for M/LCCPs. There are indications for dechlorination of higher chlorinated SCCPs in sediment and soil cores over decennia. For instance, concentrations of Cl_{9-10}–SCCPs were found to decrease with increasing marine sediment core depth in China [17]. Similarly, Zeng et al. [16] reported an increasing contribution of C_{10, Cl_{5-6}–SCCPs} in older layers of the same sediment core and correspondingly decreasing contribution of C_{11,12, Cl_{7-8}–SCCPs}, respectively, to the sum concentration of SCCP. These results imply possible microbial mediated transformation of SCCPs in sediment, but more detailed studies are needed. Even in activated sewage sludge, CPs are relatively persistent. Zeng et al. [39] found high accumulation of SCCPs in sludge possibly resulting from long-term cycling in the STP, indicating that at least some SCCPs are resistant to biodegradation/biotransformation. Mass balance results by Zeng et al. [39] indicated that 22%, 22%, 28%, and 37% of the initial SCCP mass loading of C_{10}, C_{11}, C_{12}, and C_{13}, respectively, remained unidentified in the sludge nor effluent due to biodegradation/biotransformation. As the mass fraction loss for congener group with increasing chlorine atoms were 18%, 44%, 28%, 19%, 15%, and 15%, higher chlorinated SCCPs were thought to be more resistant to biodegradation/biotransformation. Recently, Lu [50] discovered a newly isolated bacterial strain (Pseudomonas sp. N35) that is capable of degrading SCCPs in sewage sludge (73% within 30 days). After treatment, the relative abundance of Cl_{8,10}–SCCPs increased, while that of the lower chlorinated SCCPs (Cl_{5,7}) decreased. The authors suggested that this is because lower chlorinated SCCPs are more bioavailable to microorganisms, making them subject to rapid metabolic degradation, which results in higher dechlorination compared to that of higher chlorinated SCCPs. However, congener group specific biodegradation studies are sparse and more work is required to understand the potential for biodegradation and biotransformation of CP mixtures.
2.4.1 Bioaccumulation

While the bioaccumulation potential of $C_{\leq 20}$-LCCPs is considered to be low [28], the potential of $C_{18-20}$–LCCPs and MCCPs is unknown as current data, especially relating to bioconcentration factors (BCF) and persistency in organisms, are inconclusive [3, 52]. This is mainly because MCCPs with the highest measured (or predicted) BCFs are also those that are most readily biotransformed. The relationship between physico-chemical properties and biological processes in organisms (i.e. biotransformation) is very complex and warrants further investigation.

For SCCPs, available modelled, laboratory and field results all indicate their potential for bioaccumulation (Log $K_{\text{OW}} > 5$ and BCF > 5000; Figure 2-2). An increasing number of studies are focusing on bioaccumulation per SCCP congener group. For example, Ma et al. [49] reported SCCP congener group lipid-normalized log BAFs (4.1-6.7) and log BSAFs (0.1-7.3) for various aquatic organisms.

An important aspect in the bioaccumulation potential is the resistance of the compound to metabolic transformation or degradation in the organism, usually assessed by toxicokinetic and/or metabolism studies. Like for other POPs (i.e. PCBs), some CP congeners may, for example be enzymatically biotransformed into either more or less toxic congeners. While knowledge on CP biotransformation remains scarce, some studies exist that investigated tissue distribution and SCCP elimination. In hens that were exposed to SCCPs by oral administration for six weeks [53], SCCPs were mainly found in the fatty tissues including abdominal fat, and one one-third of the $\Sigma$SCCPs was found in the egg yolk, urine and faeces. Half-lives of SCCPs in blood and tissues (i.e. fat, liver, kidney and leg) were estimated to 16.5–28.3 days [53]. In rats that were exposed to SCCPs by an oral single dose [54], 28% and 3.5% of the $\Sigma$SCCPs were excreted through faeces and urine without metabolism, respectively. In juvenile rainbow trout (*Oncorhynchus mykiss*), biotransformation of SCCPs was reported with depuration half-lives between 7 and 37 days and positive correlations between half-lives and $K_{\text{OW}}$, carbon chain length and chlorine content were found [55, 56]. SCCP losses via the lactation process have also been reported [57].
To date, it is unclear whether chlorine content or carbon chain length is the most contributing factor for bioaccumulation potential. SCCPs with longer chain lengths and higher chlorination degree appear to have a higher bioaccumulation potential and/or a lower elimination potential than other SCCPs [2] and are found to be more prevalent in biota [49]. For example, Ma et al. [31] found lower percentages of Cl₅–SCCPs in bivalves compared to the associated sediment while SCCP carbon chain length profiles were similar between the two matrices, suggesting higher elimination potential and/or higher excretion potential of Cl₅–SCCPs than other SCCPs. Similarly, a significant increase of bioaccumulation factors (BAFs) with increasing chlorine numbers was observed by [43]. Although BAFs seems to slightly increase with increasing number of carbon atoms no significant relationship was found [31, 42]. In contrast, Fisk et al. [55, 56] showed that bioaccumulation of SCCPs in fish is more influenced by carbon chain length than chlorination degree. In rats, blood and urine analysed during the elimination stage were mainly dominated by Cl₅–SCCPs, while Cl₈–10–SCCPs dominated in the faeces [54].

SCCP also have a potential to biomagnify in the food chain. A trophic magnification factor (TMF) >1 indicative of biomagnification in aquatic food webs have been reported for SCCPs in the marine food web in China (2.38) [49], Arctic marine food web (2.3) [58], freshwater food webs near a municipal STP (1.61) [42] and in Lake Michigan (1.2) [59]. The biomagnification potential of SCCPs is further supported by the high concentrations of SCCPs reported in upper trophic level organisms, both in marine and freshwater (Section 2.5.4). Only two studies reported on TMF values for MCCPs before 2010, which were inconclusive [3], and none for LCCPs. Recently, Herzke et al. [58] reported a TMF of 2.0 for MCCPs in an Arctic food web, supporting the notion that MCCPs also have the potential to biomagnify in the aquatic food chain.
2.4.2 Human exposure pathways

Evidence to date suggests that non-occupational adult human exposure to S/MCCPs occurs mainly via the diet (ca. 85% relative contribution) and inhalation of indoor air (ca. 15%), while exposure via dust is thought to be negligible [60]. For toddlers, however, dust ingestion may be a more crucial exposure pathway (ca. 15%). Worst case scenario indoor exposure estimates for toddlers in Sweden (0.49 μg/kg bw/d) [61] were much higher than for adults (0.06 μg/kg bw/d), but still below the tolerable daily intake (TDI), which is 6 μg/kg bw/d for MCCPs according to Environment Canada [27] and 100 μg/kg bw/d for S/MCCPs according to IPCS [62]. Although not investigated to date, MCCP exposure through soil ingestion may also be a significant exposure pathway for toddlers. For example, children with a weight of 15 kg ingest between 0.1 and 10 g/d [62]. Considering the maximum reported MCCP concentration in soil (1,530 ng/g dw) [45], associated intake could amount up to 10 μg/kg bw/d, which would exceed the TDI [27]. While SCCP dietary exposure and intakes reported to date were below the TDI [23, 63], Harada et al. [23] found >2 order of magnitude increase of SCCPs in food in China over a decade (36 and 390 - 1,000 ng/kg bw/d in 1993 and 2009, respectively). In Japan, SCCP levels remained relatively stable (72 and 81 ng/kg bw/d in 1990 and 2009, respectively). A substantial source of SCCPs in food in China was suggested to be cooking oil, with estimated SCCP dietary intakes of <0.78 to 38 μg/d [63]. Considering the trophic magnification for S/MCCPs (TMF>1), seafood may also be a substantial source for human exposure. Indeed, a human exposure model predicted that consumption of fish could contribute a high proportion (80-100%) to SCCP intake in Nordic countries [64]. Other potential sources for human exposure to MCCPs include metal working fluids (MWFs) in occupational scenarios [65]. In Europe, MCCPs are used as additives in two types of MWFs: water-based fluids that have MCCP contents up to 20%, and oil-based fluids, which have much higher MCCP contents (up to 100%). Mirroring their difference in MCCP contents, estimated dermal exposure for oil-based fluids (520 mg/d as typical exposure and 35,000 mg/d as reasonable worst-case exposure) was higher compared to water-based fluids (30 and 520 mg/d, respectively).

2.5 Environmental occurrence

Comparing measured CP concentrations between studies and media is difficult as concentrations are determined by different instrumental techniques, quantification procedures, and calibration standards that may add substantial uncertainty. All studies reporting on environmental CP levels published since 2010 are summarized in Table S1-1, and clearly show that CPs are detected in different compartments in the environment as well as remote areas, indicating the widespread distribution of these compounds in the environment. Studies reported mainly on SCCPs, and only few quantified MCCPs or total CPs (ΣCPs). Although data are limited, MCCP concentrations in most matrices (i.e. sediment, soil and biota) were usually higher than those of SCCPs within the same study (Table S1-1), with the general exception of air.
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2.5.1 Air

Our comparison of studies reporting sum of SCCPs ($\Sigma$SCCPs) atmospheric concentrations from different countries suggests a relatively large difference between continents and between remote and rural areas (Figure 2-3, Table S1-1). S/MCCPs were found, albeit at relatively low levels (in the range of reported means of 0.0045-0.91 ng/m$^3$), in remote areas such as the Norwegian and Canadian Arctic, as well as in Antarctica. By far the highest $\Sigma$SCCPs concentrations, measured via passive sampling (PAS) in outdoor air have to date been observed in China, with 520 ng/m$^3$ in the Southeast Coastal region and other sites near heavy industry in the Midwest China [66], areas that are known to have high CP production and usage rates in China [45]. Relatively high $\Sigma$SCCPs concentrations (average concentration 16 ng/m$^3$ via PAS) were also found at industrial sites in Europe (Lancaster, UK) around 2000 (Figure 2-3) [67], when the production and use of SCCPs was still permitted. Five times lower concentrations were reported more recently (average concentration 2.9 ng/m$^3$ by PAS) [68]. $\Sigma$SCCPs concentrations decrease in China from more to less industrialized areas [45] as well as from urban to rural areas [66]. This is consistent with findings from Japan and South Korea [66] as well as India [20], where relatively high levels were found in the main industrial areas. These studies indicate that releases from industrial regions are a primary source of SCCPs to the atmosphere [66]. In contrast, no clear spatial pattern was observed in Pakistan [20] and both SCPP and MCCP concentrations were found to be relatively low in some industrial sites but moderately high at agricultural sites. The authors suggested that this could due to the more random distribution of local sources (e.g. polymer and paint manufacturing) in Pakistan, where data on CP production and use are currently unavailable. Such local sources may play an important role in the level of these compounds in the atmosphere and the environment [20].

CPs have also been reported at relatively high concentrations (average $\Sigma$/MCCPs concentration 69 ng/m$^3$) in indoor air (Figure 2-3, Table S1-1) from Stockholm, Sweden [61] and Melbourne, Australia (median $\Sigma$SCCPs concentration 60 ng/m$^3$) [69]. In Melbourne, outdoor concentrations were similar to those reported from China (100 ng/m$^3$); the reason for this is unclear but warrants further investigation. MCCPs are usually present at much lower concentrations in air than SCCPs (Table S1-1), except at sites in close proximity to point sources such as industries and waste water treatment plants [45], which is in line with their environmental fate.

Log $K_{OM}$ generally correlates linearly with inverse temperature and therefore increasing mass of CPs partition to particulate matter with colder temperatures. Indeed, SCCPs were mainly found in the particle phase during winter in China, whereas mainly in the gas phase during summer in China and Antarctica [48, 70]. Accordingly, seasonal variation was reported for gas phase $\Sigma$SCCPs concentrations in China [48, 66, 70] with higher values in summer (110-330 ng/m$^3$) than winter (0.95-27 ng/m$^3$ [70]. In some areas, however, such as Japan, South Korea and two sites in Qingdao and Shanghai in China, no significant variation between spring and fall were found, suggesting that these
sites may be influenced by sustained sources of SCCPs [66].

2.5.2 Snow, marine and freshwater
As CPs are hydrophobic compounds with low aqueous solubility, their concentrations in the aqueous phase are usually below detection limits (Table S1-1) [79], unless the water is close to point sources. For example, relatively high SCCP levels were found in lake water from China that received STP effluent discharges (160-180 ng/L) [42] and in rivers from an industrial area in the UK (<100-1700 ng/L) [80]. Relatively high concentrations were also reported in urban snow (ΣCPs 330-3200 ng/L), reflecting the ability of snow to accumulate pollutants emitted from human activity and to act as a temporary storage of contaminants [81]. As a result, concentration peaks in surface runoff and in receiving waters may occur during the snowmelt period. Decreasing ΣSCCPs concentrations from the coast to the open ocean were observed in China [49], suggesting that riverine input can be an important source of SCCPs. Due to their strong sorption to organic matter, the main fraction of CPs in water are typically sorbed to suspended matter rather than freely dissolved.

2.5.3 Sediment and soil
Since 2010, most studies on CP levels in sediment and soil originate from China (Table 2-2), and reported concentrations vary widely. Compared to other organohalogen compounds such as PCBs, endosulfan and polybrominated diphenyl ethers (PBDEs), average concentrations in the same sediment samples were by far the highest for ΣSCCPs [82]. Further, ΣMCCPs concentrations in sediment or soil were much higher compared to SCCPs, which is in line with production trends in China. Maximum ΣSCCPs concentration reported from background soil samples in Europe [76] were similar to those in soil from industrialized areas in China. Similar to observations in water and air, the spatial distribution of SCCP concentrations showed a decreasing trend with increasing distance from the Chinese coast to the sea [16, 40, 49], and from urban to rural areas [83]. Riverine discharges, ocean current and atmospheric deposition play an important role on ΣSCCPs concentrations in the Chinese marine environment [73], and a band-type distribution of SCCP migration along the Chinese coast was observed [74].

Due to the relatively high K_{OC} of SCCPs (Table 2-1), their distribution in sediments is expected to be closely related to the organic carbon content and indeed significant correlations between SCCPs and total organic carbon (TOC) were found for surface marine sediments [16, 31, 40, 49, 74]. In sediments close to sources though, SCCP concentrations are governed more by local inputs compared to TOC [16, 42]. No significant relationships between SCCP concentration and TOC in sediment may therefore indicate local emission of SCCPs and continuing discharge [19].
Figure 2-3:
Table 2-2 Concentrations of sum SCCPs and MCCPs in sediment and soil reported since 2010.

<table>
<thead>
<tr>
<th>Sample type</th>
<th>Concentration range</th>
<th>Location, country</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pond, river, tributary, estuary &amp; core sediment</td>
<td>320-6,600</td>
<td>Pearl River Delta, China</td>
<td>[17]</td>
</tr>
<tr>
<td></td>
<td>ΣMCCP: 880-38,000</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lake sediment</td>
<td>1,100-8,700</td>
<td>Gaobeidan Lake, China</td>
<td>[42]</td>
</tr>
<tr>
<td>Marine sediment</td>
<td>97-1800 (650)</td>
<td>Bohai sea, China</td>
<td>[31]</td>
</tr>
<tr>
<td>Marine sediment</td>
<td>65-540 (300)</td>
<td>Bohai sea, China</td>
<td>[49]</td>
</tr>
<tr>
<td>Lake &amp; river sediment</td>
<td>40-480 (210)</td>
<td>Liaohe River basin, China</td>
<td>[71]</td>
</tr>
<tr>
<td>Marine sediment</td>
<td>0.40-69</td>
<td>Firth of Clyde, Scotland</td>
<td>[72]</td>
</tr>
<tr>
<td></td>
<td>5.6-380</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Marine sediment</td>
<td>15.7-85 (38)</td>
<td>Bohai &amp; Yellow sea, China</td>
<td>[16]</td>
</tr>
<tr>
<td>Marine sediment</td>
<td>5.8-65 (26)</td>
<td>East China Sea, China</td>
<td>[73]</td>
</tr>
<tr>
<td>Marine sediment</td>
<td>9.0-37 (24)</td>
<td>East China Sea, China</td>
<td>[74]</td>
</tr>
<tr>
<td>Soil, wastewater irrigated</td>
<td>160-1,500 (NA)</td>
<td>Liangshui River, China</td>
<td>[41]</td>
</tr>
<tr>
<td>Urban Soil</td>
<td>&lt;1.7-620 (39)</td>
<td>Shanghai, China</td>
<td>[75]</td>
</tr>
<tr>
<td></td>
<td>ΣMCCP: 2.0-190 (16)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Woodland &amp; grassland soil, background</td>
<td>&lt;0.60-570 (35)</td>
<td>Norway</td>
<td>[76]</td>
</tr>
<tr>
<td>Woodland &amp; grassland soil, background</td>
<td>&lt;0.60-490 (22)</td>
<td>UK</td>
<td></td>
</tr>
<tr>
<td>Woodland, vegetable field, paddy and background soil</td>
<td>7.0 -540 (84)c</td>
<td>Guangzhou, China</td>
<td>[77]</td>
</tr>
<tr>
<td>Soil, background</td>
<td>0.42-420 (62)</td>
<td>Chongming Island, China</td>
<td>[78]</td>
</tr>
<tr>
<td>Roadside soil</td>
<td>31-420 (200)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Woodland soil</td>
<td>0.42-14 (7.3)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Farmland soil</td>
<td>1.2-210 (25)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>River Soil</td>
<td>1.9-240 (18)c</td>
<td>Pearl River Delta, China</td>
<td>[45]</td>
</tr>
<tr>
<td></td>
<td>ΣMCCP: 2.1-1,500 (59)c</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Paddy soil</td>
<td>57-170 (NA)</td>
<td>Liaohe river basin, China</td>
<td>[19]</td>
</tr>
<tr>
<td>Upland soil</td>
<td>84-190 (NA)</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

NA not available; a Concentration range of total SCCPs (ΣSCCP), unless stated differently (e.g. ΣMCCPs); b Concentrations obtained by a different instrumental technique, gas chromatography coupled to a flame ionisation detector (GC-FID); c Lack of information regarding if the concentration were on a dry weight or wet weight basis.
2.5.4 Biota

Since 2010, $\Sigma$S/MCCPs concentrations in biota have been reported mainly from China with a focus on aquatic species. In general, the intra and interspecies variability of CP concentrations among these studies is high, resulting in mean concentrations with large standard deviations (Figure 2-4). For example, a difference of three orders of magnitude was found in $\Sigma$SCCPs concentrations between polar bears from Svalbard in the Norwegian artic (200-600 ng/g lw) and Hudson Bay, Canadian Arctic (0.15-0.84 ng/g lw).

Among the different species investigated, relatively high concentrations were reported in sharks ($\Sigma$SCCPs 55-5200 ng/g lw), and marine mammals (i.e. dolphins and porpoises; $\Sigma$SCCPs 570-24,000 and $\Sigma$MCCPs 670-56,000 ng/g lw), indicating that, similar to POPs, animals high in the food chain and with long life spans have the potential to accumulate high concentrations of CPs. Furthermore, $\Sigma$S/MCCPs concentrations observed in near shore/estuarine marine mammal species (i.e. humpback dolphins) were 2-3 times higher compared to those with habitats further offshore [84]. These authors also found a significant trend of increasing $\Sigma$S/MCCPs concentrations in both humpback dolphin and porpoise species between 2004-2014, including a shift from SCCPs towards MCCPs, mirroring the production trends in China.

In studies that investigated CP concentrations in biota along with other compounds, concentrations of $\Sigma$S/MCCPs are at similar levels as perfluoralkyl substances and cyclic volatile methyl siloxanes [85], or typically present at the highest concentrations than other compounds (i.e. dichlorodiphenyldichloroethanes, hexachlorocyclohexanes and PCBs). For example, in different organisms from the Bohai sea in China [82] and the Barents Sea, including high trophic shark species from Iceland [86]. These observations suggest that substantial loads of CPs have entered the marine environment, and that the increasing production volumes may be of concern regarding exposure of marine organisms and accumulation through the food web.

As expected for lipophilic compounds, significant correlations between CP concentrations and the tissue or organism lipid content are frequently reported [31, 49, 87]. However, the influence of other biological factors (e.g. age, weight, sex) is still unclear. Saborido Basconcillo et al. [87] recently investigated some of these aspects and observed significant, but negative correlations between lipid normalized concentrations of SCCPs and length as well as weight of freshwater fish. In contrast, log transformed wet weight and lipid weight based MCCP concentrations showed no significant correlations with length, weight, age, or trophic level. Similarly, Zeng et al. [84] did not found significant gender-associated concentration differences of S/MCCPs in marine mammal species (p > 0.05).
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2.5.5 Composition profiles

Comparing composition profiles (i.e. congener group patterns) obtained by different analytical methods should be done with caution [7]. We emphasize that the current reported CP data provide information mainly on the higher chlorinated paraffins (>Cl<sub>5</sub>), as almost all reported CP data are obtained by gas chromatography coupled to a mass spectrometer in electron capture negative ion mass spectrometer (GC-ECNI-MS), a method in which lower chlorinated CPs (<Cl<sub>5</sub>) are undetected [7].

Our comparison of the relative abundance of different CP carbon chain lengths for Cl<sub>5</sub>–SCCPs show that since 2010 the C<sub>10</sub>–CPs group usually dominates in environmental matrices (Figure 2-5). Patterns found in the environment could reflect those in commercial CP mixtures [89]. Prior to 2010, ambient air and biota from Europe and North America [88, 89] was dominated by C<sub>12</sub>–CPs, reflecting the various commercial CP mixtures from Germany and England (Figure 2-5). Since the ban on SCCP production in these regions and the associated use of other SCCP technical mixtures and/or SCCP containing products, different congener group distributions are found. For example, Fridén et al. [61] found that C<sub>10</sub>–CPs and C<sub>11</sub>–CPs predominated in indoor air samples in Sweden, which is similar to the distribution of Chinese commercial mixtures. They
also found that Cl₄–SCCPs and Cl₅–SCCPs predominated, in contrast to the technical standards and reported congener group patterns in other environmental matrices before 2010, which were typically dominated by Cl₆–CPs [90-92]. Consistent with this, SCCP congener group profiles identified in environmental matrices from China reflect the two major industrial CP products manufactured in China, CP-52 and CP-42, which are dominated by C₁₀–Cl₅–SCCPs [19]. Concurrent with increasing production volumes in China, a shift in the most dominant SCCP carbon chain lengths was also noted in food from China, from C₁₂ and C₁₃ in 1993 to C₁₀ and C₁₁ in 2009 [23]. Spatial differences in congener group distributions can also be expected due to differences in their fate, particularly regarding transport and persistency (Section 2.3.2) [17, 45]. Shorter chain (C₁₀ and C₁₁) and lower chlorinated (Cl₅–8) SCCPs are more volatile and water soluble than other CPs and tend to have long atmospheric lifetimes [46]. These congener group may also more easily migrate to sites distant from emission sources, including remote regions or marine environments by long-range atmospheric or waterborne transport.

Differences in congener group profiles between biota and non-biological matrices and commercial mixtures is likely due to bioaccumulation processes, dependent on both the compounds properties (e.g. availability, Section 2.3.2) and the behaviour and physiology of the species (e.g. feeding and metabolism, Section 2.4.1). As an example, Zeng et al. [42] found that CPs with longer carbon chains are more prone to absorption onto suspended particulate matter and thus less available for uptake by organisms. Another example, the congener group distribution of a bottom feeder (i.e. crucian carp), was similar to that of sediments in the same area, whereas the congener group distribution of other fish reflected the surrounding lake water [42]. However, in biota that are clearly exposed to a local CP sources (e.g. STPs) CP congener group profiles are often similar to that of commercial mixtures. This was also observed in studies published before 2010 [93].

While information is sparse, the congener group patterns of MCCPs appear identical across the globe, i.e. South-China [45, 75], India and Pakistan [20], as well as Canada, Japan and the Baltic sea [7], with C₁₄ and lower chlorinated (Cl₆–7) as the most prevalent MCCPs present in environmental matrices.
2.6 Conclusions

A substantial amount of information on occurrence and fate of SCCPs became available during the last five years, especially from China. In addition, more studies are reporting on congener group-specific distributions. Clearly, reported $\Sigma$SCCP concentrations in China, currently by far the world largest CP producer, exceed that of other countries. However, data of countries near China or other CP producing countries (e.g. Russia, Taiwan, Australia and other Asian/Oceanic countries) are limited or lacking entirely, and further studies are recommended. Studies on environmental levels of MCCPs are even more sparse, despite limited data suggesting that MCCPs are generally present at the highest concentration in the environment. Data on LCCPs levels since 2010 are absent. Considering the high production volumes, more studies on the environmental levels, including spatial and temporal trends of M/LCCPs are needed for a better understanding on the fate of these compounds. Data suggest that at least some CPs are bioaccumulative and subject to long-range atmospheric and oceanic transport, facilitating regulations and classifications. Enough data are now available to conclude that SCCPs fulfil the criteria under the UNEP Stockholm Convention for designation as a POP. In case of SCCPs, more studies are advised particularly on the toxicity and bioaccumulation potential as well as long-range transport on a congener group level. For M/LCCPs, studies on environmental behaviour and fate, as well as their persistency (i.e. degradation), bioaccumulation potential are sparse or lacking entirely and in high demand.

Acknowledgements

Boris van Zanten and Cesar Alexander Villa are kindly acknowledged for input on graphing the data.
Figure 2-5:
Relative abundance of each SCCP homologue by carbon chain length in commercial CP mixtures (graph A) derived from a) Oehme et al. [96] and b) Wang et al. [45], as well as most and second most dominant SCCP group by chain length (graph B and C) and most and second-most dominant SCCP group by chlorine atoms (graph D and E) in various matrices reported since 2010. Please note that Chinese commercial mixtures also exist of M/LCCPs (not shown).
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