Final report
8 March 2010

Programområde *Miljögiftssamordning*
Delprogram *Screening*
Överenskommelse nr *2190704*
Dnr *721-1539-07Mm*

Human exposure to chlorinated paraffins via indoor air and dust

Ulrika Fridén, Michael McLachlan, Urs Berger*
Department for Applied Environmental Science (ITM), Stockholm University, 106 91 Stockholm, Sweden

* Contact phone 08-674 7099, e-mail urs.berger@itm.su.se

Background

Chlorinated paraffins (CPs, also called polychlorinated *n*-alkanes or PCAs) are complex mixtures containing thousands of different homologues and isomers. The commercial mixtures are classified according to their carbon chain length into short chain CPs (SCCPs, C10-C13), medium chain CPs (MCCPs, C14-C17) and long chain CPs (LCCPs, C18-C30). CPs have been produced since 1930. Their total world production is estimated to be 300 kt/year currently[1]. In Sweden, the use of CPs was estimated to 100 t/year in 2002. European regulations forbid the use of SCCPs as extreme pressure additives in metal working fluids[2]. However, CPs are also utilised as flame retardants or plasticisers in PVC, rubber, paints, coatings and sealants.

SCCPs have been the subject of a detailed risk assessment, while an evaluation of MCCPs is in progress. Both SCCPs and MCCPs have been found to be persistent and to bioaccumulate. CPs have a low acute toxicity, but SCCPs are classified as toxic to aquatic organisms. Furthermore, carcinogenic effects in rats and mice have been observed[3,4].

Due to the difficulties in the determination of these complex mixtures, information about CP levels in the environment is scarce. Furthermore, a comparison of the existing data is difficult, since different analytical methodologies and – even more problematic – different quantification procedures and standards have been applied[5-7]. Many of these problems have been overcome during the last 5 years, and the possibility to achieve more reliable quantitative CP analysis has increased[5,6].

The ubiquitous presence of CPs in the environment is evident from existing data. CPs have been found in all compartments of the environment as well as in aquatic and terrestrial food webs in rural and remote areas[8]. Recently, the presence of CPs in human milk was shown in samples from Germany[9] and from the United Kingdom[10]. However, the human exposure pathways for CPs have been barely investigated to date. Leakage of persistent organic pollutants from products is known to cause an increase in indoor air concentrations, and in some cases this can make indoor air a significant human exposure pathway[11]. SCCPs and MCCPs have been detected in outdoor air samples from Sweden (6-33 ng/m^3)[12] and from the United Kingdom (<0.18-3.4 ng/m^3 for SCCPs and <0.81-14 ng/m^3 for MCCPs)[13]. However, in indoor air CP concentrations have hardly been investigated[13]. Therefore, the aim of this study was to determine if the presence of a wide range of CP treated goods in households contributes to increased indoor air and dust levels, resulting in these being important exposure pathways.

Materials and methods

Samples

The samples used in this study were obtained from another screening project on brominated flame retardants (BFRs) and perfluorinated compounds in indoor air and dust[14]. The study design and sampling are described in detail elsewhere[14]. Indoor air samples for analysis of CPs in this study were chosen from apartments from 21 different houses from the city of Stockholm. For two houses air samples from three different apartments were chosen, while for the other 19 houses samples from two apartments were analysed. This resulted in a total number of 44 indoor air samples. Dust samples were chosen on the basis of the sampled amount of dust. Only 6 dust samples were available with enough sample material in order to perform CP analysis. Unfortunately, only one of these samples was from an apartment where

also indoor air was analysed. Correlation analysis between paired indoor air and dust samples can therefore not be performed.

Sample preparation

Sample extraction and clean-up is described in detail elsewhere[14]. In short, the surrogate standard dechlorane was added to the sampling medium (polyurethane foam plugs and glass fibre filter for air samples or cellulose filter for dust samples) and extraction was performed twice with dichloromethane in an ultrasonic bath. The extracts were concentrated and the solvent changed to *n*-hexane. Clean-up was performed on an acidic (sulphuric acid) silica column. The volume of the final extract was reduced to approximately 50 µl before instrumental analysis. The same extracts as for analysis of BFRs[14] were used for CP analysis.

Quantification by GC/EI-MS/MS

CPs were separated and detected using gas chromatography coupled to tandem mass spectrometry according to a method described elsewhere[15] with some modifications. This method allows for quantification of the sum of SCCPs and MCCPs in a single chromatographic run. Separation was performed on a HP 5890 Series II gas chromatograph equipped with a fused silica capillary column (DB-5MS, 15m, 0.25 mm i.d., 0.25 µm film). Sample volumes of 2.5 µl were injected in splitless mode at an injector temperature of 275 °C. The temperature program was 90 °C (2 min), then 30 °C/min to 300 °C (5 min) with helium as carrier gas (1.4 ml/min). A TSQ 7000 mass spectrometer was employed in the EI-MS/MS mode with the transfer line temperature set to 275 °C, ion source temperature 200 °C, and manifold temperature 70 °C. Conditions for EI-MS/MS were as follows: scan time 0.05 s, electron energy -70 eV, filament emission current 400 µA, and electron lens voltage 0 V. Two mass transitions for CPs were recorded: precursor ion *m/z* 102 to product ion *m/z* 67 (quantifier ion) and precursor ion *m/z* 91 to product ion *m/z* 53 (qualifier ion), both with a collision energy of 12 V and argon as collision gas (0.4 mTorr).

Recovery experiments were performed using gas chromatography (HP 5890 series II, equipped with a DB5-MS column, 15m, 0.25 mm i.d., 0.25 µm film) coupled to an electron capture detector (ECD). Sample volumes of 1 µl were injected in splitless mode at an injector temperature of 250 °C. The temperature program was 90 °C (2 min), then 25 °C/min to 300 °C (6 min) with helium as carrier gas (1.4 ml/min).

Quantification of CPs was performed using a technical SCCP mixture (C10-13, 55.5% Cl, Ehrenstorfer) as external calibration standard. This mixture was chosen because it best reflected the typical composition of CPs in indoor air samples. Integration in standard, blank and sample chromatograms was performed “from baseline to baseline”, i.e. one area was calculated over the whole elution time of CP homologues and isomers.

Quality control

Detection of CPs was based on the presence of a matching CP pattern in the chromatograms of both the quantifier and qualifier ion transition as well as a ratio of quantifier area to qualifier area that did not deviate more than 30% from the ratio in the external calibration standard. In some cases a matching CP pattern was detected in both chromatograms but the area ratio deviated more than 30% from the standard. In these cases the area (quantifier or qualifier) was used for quantification that resulted in the lower calculated CP concentration, assuming that the other area was influenced by co-eluting impurities.

Field blanks, lab blanks and solvent blanks were regularly taken and analysed along with the samples. None of the blank extracts showed detectable concentrations of CPs. Quantified CP concentrations in samples were therefore not corrected for blank values. Recoveries of the CPs in the extraction and acidic silica clean-up were determined in five replicates for the technical SCCP mixture used as calibration standard. Average recovery ± 1 standard deviation was 90 ± 7%. Final results were corrected for average recovery. One air sample was analysed in duplicate, the two results deviated <14%.

Congener group analysis by GC/ECNI-MS

The CP mixtures in SCCP and MCCP standards (C10-13, 51.5% Cl; C10-13, 55.5% Cl; C10-13, 63% Cl; C14-17, 42% Cl; C14-17, 52% Cl; all purchased from Ehrenstorfer) and twelve indoor air samples were classified according to carbon chain length and degree of chlorination. This was done by GC/MS using electron capture negative ionization (ECNI). Isomers with the same sum formula ($C_nH_{2n+2-x}Cl_x$), i.e. of the same CP congener group, were quantified together as one sum. The two most abundant isotopes of each congener group were monitored as outlined by Reth and Oehme (2004)[16] with some modifications. In total 96 CP *m/z* ratios were selected for monitoring, requiring five injections for each sample and standard, monitoring up to 20 CP *m/z* ratios plus 2 *m/z* ratios for the surrogate standard dechlorane per injection.

Separation was performed on a HP 5890 Series II gas chromatograph equipped with a fused silica capillary column (DB-5MS, 15 m, 0.25 mm i.d., 0.25 µm film). Sample volumes of 2 µl were injected in splitless mode at an injector temperature of 275 °C. The temperature program was 100 °C (2 min), then 10 °C/min to 300 °C (8 min) with helium as carrier gas (1.4 ml/min). A SSQ 7000 quadrupole mass spectrometer was employed in the ECNI-MS mode with ammonia (NH_3) as reagent gas at a pressure of about 9 mbar. The ion source temperature was 150 °C, the emission current 400 µA and the electron energy was 150 eV. Selected *m/z* ratios were monitored (SIM mode) at a scan time of 0.6 sec/cycle. The manifold temperature was set to 70 °C and the transfer line connecting the GC and the MS was held at 280 °C.

Detection of the CP congener groups was based on the presence of a matching CP pattern in the chromatograms of both the quantifier and qualifier ion as well as a ratio of quantifier area to qualifier area comparable to the ratio in the external calibration standard. The total area in the selected mass chromatogram of each quantifier ion, corresponding to the sum of all isomers within a CP congener group ($C_nH_{2n+2-x}Cl_x$), was then normalised by division by the area of the surrogate standard quantifier in the same injection. For calculation of relative fractions of congener groups the sum of all normalised congener group areas of a given standard or sample was set to 1.

Results

Indoor air levels

Table 1 summarizes the concentrations of sum SCCPs and MCCPs in indoor air samples measured by GC/EI-MS/MS. CPs were detected in 40 out of 44 samples (91%) with a typical method detection limit (MDL) around 10 ng/m^3. The CP pattern in the indoor air extract chromatograms typically resembled the technical mixture of SCCPs used as the external calibration standard both in retention time and chromatographic pattern. This suggests that the CP pattern in indoor air is usually dominated by the relatively volatile short chain paraffins. A more detailed analysis of homologue patterns and chlorination degrees is described below.

Table 1. Total CP concentration (EI-MS/MS measurements) and relative fractions of SCCPs and MCCPs (ECNI-MS measurements) in indoor air from apartments in the city of Stockholm.

House: apartm. No	CP conc. [ng/m^3]	Rel. fraction SCCPs/MCCPs	House: apartm. No	CP conc. [ng/m^3]	Rel. fraction SCCPs/MCCPs
House 2:1	31.6		House 12:1	99.5	0.92 / 0.08
House 2:2	31.7		House 12:2	212	0.63 / 0.37
House 2:3	<10.5		House 13:1	201	
House 3:1	5.9		House 13:2	90.7	
House 3:2	<15.2		House 14:1	70.1	
House 3:4	<4.6		House 14:2	64.7	
House 4:1	55.8		House 15:1	72.6	
House 4:2	42.5		House 15:3	64.4	
House 5:1	<12.1		House 16:2	162	0.66 / 0.34
House 5:2	27.3		House 16:3	120	0.75 / 0.25
House 6:1	29.3		House 17:1	66.7	
House 6:2	109		House 17:2	32.3	
House 7:1	51.6	n.d.	House 19:2	107	
House 7:2	159	0.61 / 0.39	House 19:3	42.4	
House 8:1	23.7		House 20:1	63.8	
House 8:2	16.9		House 20:2	69.5	
House 9:1	46.9		House 21:2	155	0.71 / 0.29
House 9:2	116		House 21:3	89.9	0.74 / 0.26
House 10:1	41.2	0.71 / 0.29	House 22:1	76.7	0.72 / 0.28
House 10:2	53.7	0.59 / 0.41	House 22:2	52.9	0.66 / 0.34
House 11:1	33.2		House 23:1	69.7	
House 11:2	85.9		House 23:2	84.6	

In Figure 1 the results for the indoor air samples are depicted per house with error bars giving the minimum and maximum value of the respective apartments in the house. Considerable differences in CP concentration between houses were found, with the lowest average value of 5.3 ng/m^3 (House 3, two of three apartments below MDL) and the highest average value of 156 ng/m^3 (House 12). These differences between houses could reflect different building materials used in the construction of the house (e.g. joint sealer with or without CPs). However, in many cases quite large differences were also found between apartments of the same house (see error bars). These values could be influenced by CP treated household goods or floor mats in the apartments.

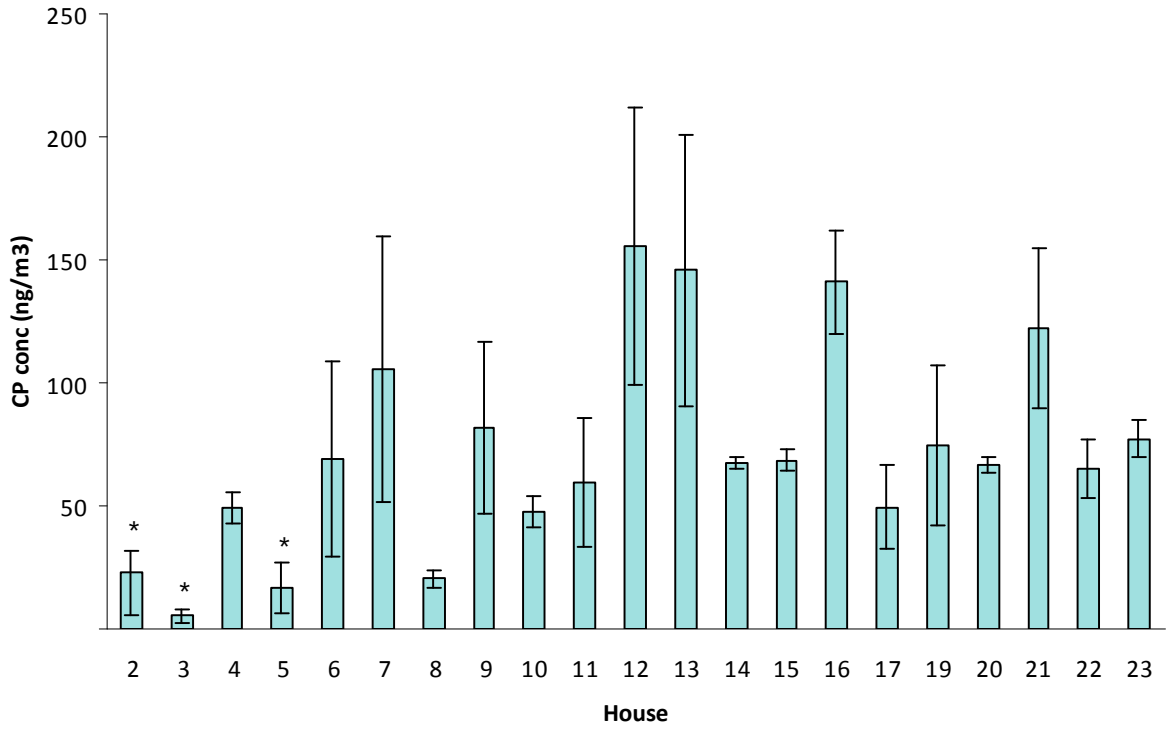

Figure 1. Mean total CP concentration in indoor air samples per house. Error bars represent minimum and maximum values. An asterisk (*) indicates that one or more apartments had a CP concentration below the MDL (see also Table 1). In these cases half the MDL value was used for the calculations.

To our best knowledge, this is the first report on indoor air concentrations of CPs. Barber et al. analysed CPs in indoor air samples in Lancaster, UK, by passive sampling techniques[13]. Unfortunately, the passive samplers were not calibrated, thus it was not possible to calculate air concentrations. Per sample concentrations of the sum of the CPs ranged from 670 ng (residential house) to 36 μg (mechanical workshop).

Few reports on ambient air levels exist[12,13,17-19]. Concentrations of total CPs (usually sum of SCCPs and MCCPs) ranged from <60 pg/m^3 in remote Arctic regions[18] to <1-15 ng/m^3 in UK air in Hazelrigg (semirural)[13] and 6-33 ng/m^3 in the city of Stockholm (Rosenlundsgatan)[12]. The overall mean CP concentration in the 44 indoor air samples from this study was 69 ng/m^3 (median 64 ng/m^3, range <5-212 ng/m^3). This is considerably higher than all measurements that have been conducted in ambient air so far. Differences in analytical methodologies and quantification procedures as well as use of calibration standards add some uncertainty to this comparison. However, this uncertainty does not cover the whole extent of the concentration difference between indoor air and literature data on ambient air. Indoor air may therefore represent an important exposure pathway of CPs to humans.

The median concentration of the sum of 10 polybrominated diphenyl ether congeners (ΣPBDE) in the same 44 indoor air samples was 63 pg/m^3 [14]. This is a factor of 1000 lower than the median of sum CPs found in this study. Despite the restriction in use of SCCPs in Europe[2], these compounds are probably today the most abundant halogenated organic contaminants in indoor air. Concentrations, sources and possible effects of CPs on human health should be further investigated.

Congener group patterns in indoor air

ECNI is a relatively soft ionization technique for CPs that primarily produces $[M\text{-}Cl]^-$ and $[M\text{-}HCl]^-$ negative ions, corresponding to the loss of one chlorine or hydrochloric acid from the molecular ion $[M]^-$. GC/ECNI-MS thus allows one to obtain information about the congener and homologue patterns and the chlorine content by analyzing CP congener groups separately. However, due to the great number of existing CP compounds, some congener groups with different numbers of carbons and chlorines formes ions with the same nominal *m/z* values. These congener groups could only partly be separated by the gas chromatographic method and the low resolution mass spectrometer used in this study. For these congener groups an elevated uncertainty in the quantification of their relative fractions in the samples is expected.

In Table 1 the relative fractions of sum SCCPs and sum MCCPs in the 12 indoor air samples analysed by GC/ECNI-MS are given. The higher abundance of SCCPs compared to MCCPs in all investigated samples confirmed the patterns seen in the GC/EI-MS/MS chromatograms based on retention time and elution window (see above). This finding is also in accordance with an expected preferential partitioning of the smaller and more volatile CPs to air. More detailed classifications of congener groups in a SCCP and MCCP standard as well as in the samples house 12:1 and house 16:3 are shown in Figure 2. These two samples are representative for all of the analysed indoor air samples. Figure 2 shows that the shortest SCCPs, i.e. homologues with 10 and 11 carbons, dominated the CP levels in the air samples. Furthermore, within a group of compounds of a given carbon chain length, the congener groups with lower degree of chlorination were relatively enriched compared to the corresponding standard (see e.g. C10 congener groups in house 12:1 compared to the SCCP standard or C15 congener groups in house 16:2 compared to the MCCP standard). This is in contrast to literature data on food from Japan[20], where e.g. within the C10 compounds, the congener group with 7 or 8 chlorine atoms dominated throughout all food categories apart from (the less important) vegetables and eggs, while in indoor air the congener groups with 5 or 6 chlorines were prevalent (Figure 2). This pattern difference could be used for elucidating human exposure sources. However, to date differences in pharmacokinetics of different congener groups are not yet well understood and human monitoring data is scarce[9,10], hampering conclusions on sources based on congener group patterns.

Only limited literature data on congener group patterns in air samples is available. Peters et al. (2000) published congener group patterns for SCCPs in ambient UK air[17]. Interestingly, they found the C12 congener groups on an average at higher abundances compared to C10 and C11, while in this study C10 and C11 congener groups were more abundant than C12 (Figure 2). Furthermore, Barber et al. (2005) found higher levels of sum MCCPs compared to sum SCCPs in three out of four ambient air samples collected at Hazelrigg, UK[13], while in this study, SCCPs were consistently more abundant than MCCPs (Table 1). This could imply different sources of CPs to UK ambient air as compared to Stockholm indoor air. Alternatively, environmental partitioning may alter the congener group profile between air samples collected near to sources and samples from rural or remote sites. More monitoring including the elucidation of congener group patterns is needed in order to understand the environmental sources and fate of the complex CP mixtures.

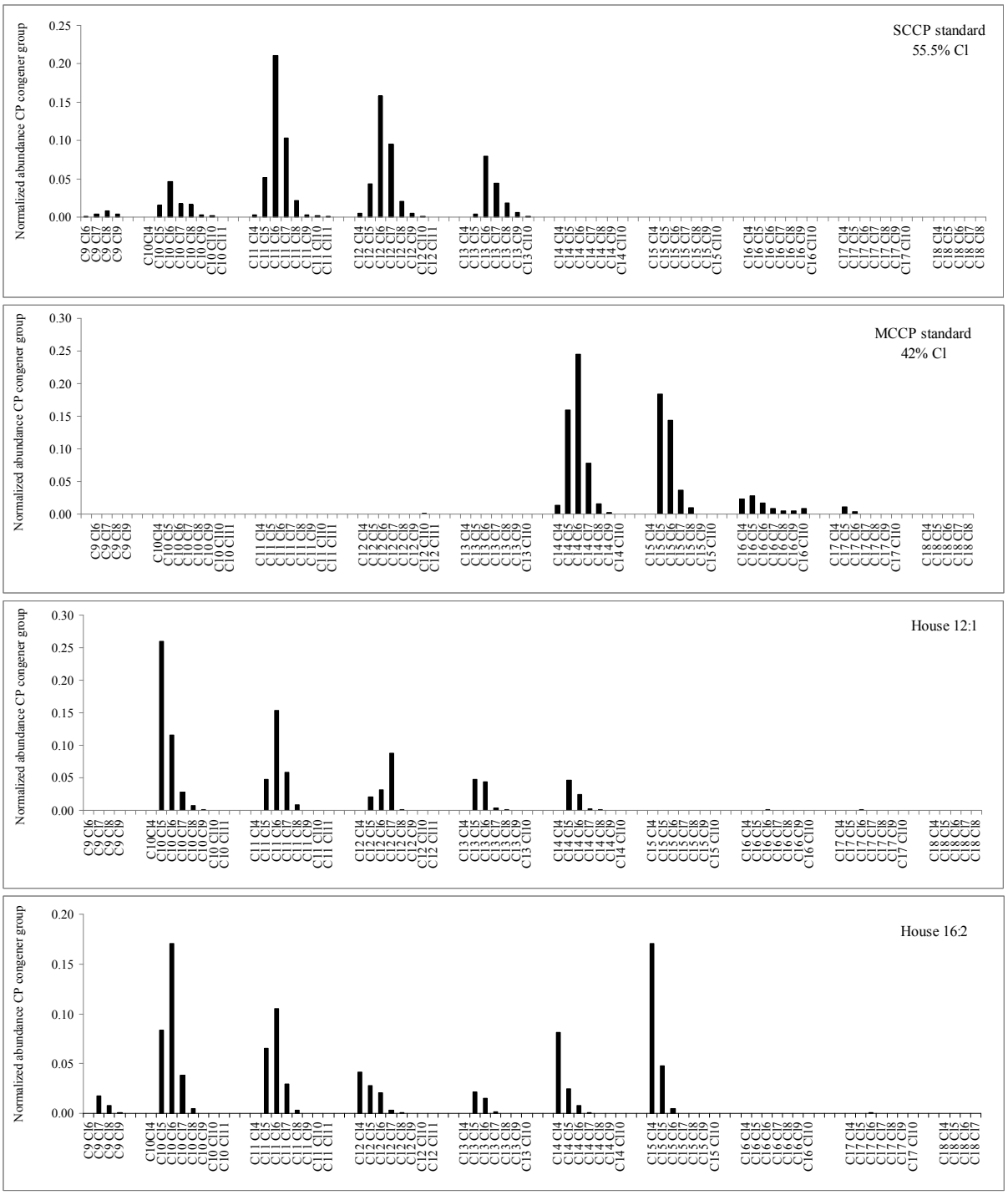

Figure 2. Relative congener group fractions obtained by GC/ECNI-MS measurements in a SCCP and MCCP standard as well as in the indoor air samples from houses 12:1 and 16:2.

Dust samples

In Table 2 the measured concentrations of CPs in the dust samples are listed. The levels were too low for congener group analysis by ECNI-MS. However, compared to the air samples and calibration standard, the CP pattern in the dust extract chromatograms obtained by EI-MS/MS detection showed higher retention times and a larger width of the elution window, reflecting the presence of longer chain chlorinated paraffins such as MCCPs. The technical SCCP

calibration standard was therefore not the optimal choice for quantification of the dust samples, even though Zencak et al. showed that the response of CPs in the EI-MS/MS method is neither influenced by degree of chlorination nor by carbon chain length[15]. Additionally, the CP concentrations in several dust samples were close to the MDL. Taken together, this led to higher uncertainty in quantitative results for dust samples compared to indoor air samples. The dust concentrations have therefore to be considered semi-quantitative. However, they give a picture of the order of magnitude and variability of CP concentrations in dust from apartments.

Table 2. Total CP concentration in dust samples from apartments in the city of Stockholm.

House:apartm. No	CP concentration [µg/g]	House:apartm. No	CP concentration [µg/g]
House 9:3	7.0	House 22:4	17.9
House 10:1	7.6	House 37:3	7.4
House 16:4	17.8	House 38:1	3.2

CPs were detected in all six samples at levels between 3 and 18 µg/g. The variability between single dust samples (factor 6) is much smaller than between indoor air samples (factor >40). However, the number of analysed samples has also to be taken into account. The only data on CPs in house dust found in the literature is from Hamburg, Germany, from the years 1998 to 2000 [21]. The 95 percentile of SCCP concentrations in dust from 65 apartments was 180 µg/g and thus 10 times higher than the highest level in this study. However, the comparison is hampered by different analytical and quantification techniques as well as uncertainties in analysis as describe above.

The median concentration of ΣPBDE in dust samples from Stockholm apartments from the same sampling program as this study was 1.4 µg/g [14] as compared to 7.5 µg/g for the CPs. This difference is much less than for the indoor air samples (see above), suggesting that the CPs show a significantly higher tendency to partition to the gas phase in indoor environments compared to PBDEs. However, the partitioning between dust and air for both PBDEs and CPs is greatly dependant on the size of the molecule and particularly on the degree of halogenation. The PBDE concentration in dust is thus dominated by the nona- and decaBDE congeners[14]. Two perfluorinated compounds, i.e. perfluorooctane sulfonate (PFOS) and perfluorooctanoate (PFOA), have also been investigated in the same dust samples[14]. Their median values were comparatively low (19 and 78 ng/g for PFOS and PFOA, respectively).

Human exposure assessment

To assess the potential importance of indoor air and dust as vectors of human exposure, the exposure via inhalation and dust ingestion was estimated for people living in the buildings in Stockholm and compared with published data for the other major vector of exposure, food ingestion (see Tables 3 and 4). It was assumed that people spend 100 % of their time in an environment with an indoor air / dust contamination equal to the levels measured in this study.

The estimates of exposure via food ingestion were taken from the one published study of dietary exposure to CPs[20]. It is derived from a market basket study in Japan and the samples were analyzed for SCCPs. The unknown difference in overall contamination level between Japan and Sweden, as well the different dietary habits and sources of the food in Japan than in Sweden, adds uncertainty to this comparison. Of particular importance for exposure to lipophilic contaminants are fish, shellfish, and meat. For many lipophilic contaminants,

concentrations in fish and shellfish are higher, and hence the relative amounts of these foodstuffs in the diet can strongly influence dietary exposure. However, the fresh weight normalized SCCP concentrations measured in shellfish, fish, and meat in the Japanese study were within a factor of 2 of each other, so these variations in the relative consumption of fish / shellfish and meat would have only a small impact on the dietary exposure to CPs.

A second factor that complicates the comparison of the dietary exposure data with the results of this study is the fact that the diet samples were analyzed for SCCPs only, while the air and dust samples were analyzed for SCCPs and MCCPs. However, for the air samples the congener group specific analyses showed that most of the CPs were SCCPs (Table 1). Thus, the comparison of the exposure vectors was made for the SCCPs, neglecting the small contribution of the MCCPs to the air samples. For the dust samples, on the other hand, the chromatograms indicated a significant contribution of MCCPs (see above). Hence, by assuming them all to be SCCPs, the assessment may have overestimated the contribution of dust ingestion to SCCP exposure.

The exposure assessment was conducted for both an adult and a toddler (Tables 3 and 4, respectively). The median and the 95 percentile of dust ingestion rates, SCCP concentrations in dust and air, and dietary exposure were used. The dust ingestion rates were taken from the USEPA exposure factor handbooks[22,23].

Table 3. Estimated exposure of a 25 year old to SCCPs via inhalation, dust ingestion, and diet

Exposure Vector	Exposure Factor	Median Concentration	Median Exposure	95%ile Concentration	95%ile Exposure
Inhalation	15 m^3 d^{-1}	75 ng m^{-3}	1.1 μg d^{-1}	200 ng m^{-3}	3 μg d^{-1}
Dust Ingestion	0.004 g d^{-1} (0.055)[§]	7.5 μg g^{-1}	0.03 μg d^{-1}	17.9 μg g^{-1}	0.98 μg d^{-1}[‡]
Diet*			6 μg d^{-1}		12 μg d^{-1}

[§]median and (in brackets) 95 percentile of dust ingestion rates[22]
[‡]a high estimate of exposure via dust ingestion obtained by multiplying the 95 percentile of the dust ingestion rate by the 95 percentile of the measured CP concentrations in dust
*from Iino et al. (2005)[20], based on a 60 kg individual, 25 years of age

Table 4. Estimated exposure of a toddler to SCCPs via inhalation, dust ingestion, and diet

Exposure Vector	Exposure Factor	Median Concentration	Median Exposure	95%ile Concentration	95%ile Exposure
Inhalation	6.8 m^3 d^{-1}	75 ng m^{-3}	0.51 μg d^{-1}	200 ng m^{-3}	1.4 μg d^{-1}
Dust Ingestion	0.1 g d^{-1} (0.2)[§]	7.5 μg g^{-1}	0.75 μg d^{-1}	17.9 μg g^{-1}	3.6 μg d^{-1}[‡]
Diet*			3.6 μg d^{-1}		6.8 μg d^{-1}

[§]median and (in brackets) 95 percentile of dust ingestion rates[23]
[‡]a high estimate of exposure via dust ingestion obtained by multiplying the 95 percentile of the dust ingestion rate by the 95 percentile of the measured CP concentrations in dust
*from Iino et al. (2005)[20], based on a 10 kg individual, 1 year of age

For an adult, the diet accounts for ~85% of the median exposure and inhalation 15%, while the contribution from dust ingestion is negligible. However, the 95 percentile of the inhalation exposure is 50% of the median dietary exposure, and the high end estimate of exposure via dust ingestion is 16% of the median dietary exposure. This suggests that the diet is the

dominant exposure pathway for the SCCPs, but that the contribution from indoor air and dust may be significant for individuals with high exposure via these pathways.

The relative contributions of diet and inhalation were similar for the toddler compared to the adult. However, the contribution via dust ingestion was more important, accounting for 15% of the median exposure. The high end estimate of exposure via dust ingestion actually equaled the median dietary exposure. This indicates that there may be toddlers for which dust ingestion is the dominant vector of exposure to SCCPs.

Due to the few available data and the assumptions made in the exposure estimates, there is considerable uncertainty in the evaluation of the relative importance of the exposure pathways. Nevertheless, one can conclude that SCCP exposure via sources in the indoor environment is not negligible, but at the same time suggest that the indoor environment is not the only major source of human exposure.

The relatively low variability in the indoor air concentrations indicates that there is not a sub-group of highly exposed individuals as a result of strong CP sources in the indoor environment. Rather, the results suggest the presence of a relatively low but broadly distributed "background" contamination. With the aim of reducing exposure, it could be useful to try to identify the source(s) of this contamination. However, costly exposure management measures should be based on analyses of food, indoor air, and outdoor air measured with the same method in the same laboratory; the great challenges in quantifying this complex mixture means that the results produced by different methods / laboratories can differ by more than an order of magnitude.

Acknowledgments

We would like to thank Cynthia A. de Wit, Kaj Thuresson, Justina Björklund and Karin Syversen for their work with sampling and for letting us analyse their extracts. The Swedish Environmental Protection Agency is acknowledged for funding (project no 721-1539-07).

References

1 World Health Organization *Environ. Health Crit. 181 - Chlorinated Paraffins*; WHO: Genf, 1996
2 European Community *Off. J. Eur. Communities* **2002**, *L177*, 1-21
3 United Kingdom Report: *Risk assessment of alkanes, C14-17, chloro.* **2002**; Environment Agency: Oxfordshire, 1-318
4 European Commission Report: *EUR 190010: European Union risk assessment report: alkanes, C10-13, chloro.* **2000**; Office for official publications of the European Communities: Luxembourg, 1-166
5 Zencak, Z.; Borgen, A., et al. *J. Chromatogr. A* **2005**, *1067*, 295-301
6 Reth, M.; Zencak, Z., et al. *J. Chromatogr. A* **2005**, *1081*, 225-231
7 Tomy, G. T.; Stern, G. A., et al. *Anal. Chem.* **1997**, *69*, 2762-2771
8 Muir, D. C. G.; Stern, G. A., et al. In *The handbook of environmental chemistry*; Paasivirta, J., Ed.; Springer-Verlag: Heidelberg, 2000; Vol. 3 Part K, pp 203-236
9 Reth, M.; Kypke, K., et al. *Organohalogen Compd.* **2005**, *67*, 1671-1673
10 Thomas, G. O.; Farrar, D. G., et al. *Environ. Int.* **2006**, *32*, 34-40
11 Harrad, S.; Diamond, M. *Atmos. Environ.* **2006**, *40*, 1187-1188

12 Järnberg, U.; Fridén, U., et al. Report: *Screening av klorparaffiner i den Svenska miljön.* **2005**; Naturvårdsverket: Stockholm
13 Barber, J. L.; Sweetman, A. J., et al. *Environ. Sci. Technol.* **2005**, *39*, 4407-4415
14 Stockholms Stad Miljöförvaltningen *Brominated Flame Retardants and Perfluorinated Compounds in Air and Dust from Indoor Environments in Stockholm*, Stockholm, 2008, ISSN 1653-9168
15 Zencak, Z.; Reth, M., et al. *Anal. Chem.* **2004**, *76*, 1957-1962
16 Reth, M.; Oehme, M. *Anal. Bioanal. Chem*. **2004**, *378*, 1741-1747
17 Peters, A.J.; Tomy, G.T., et al. *Atmos. Environ.* **2000**, *34*, 3085-3090
18 Borgen, A.R.; Schlabach, M., et al. *Organohalogen Compd.* **2000**, *47*, 272-275
19 Borgen, A.R.; Schlabach, M., et al. *Organohalogen Compd.* **2002**, *59*, 303-306
20 Iino, F.; Takasuga, T., et al. *Environ. Sci. Technol.* **2005**, *39*, 859-866
21 Kersten, W.; Reich, T. *Gefahrstoffe – Reinhaltung der Luft* **2003**, *63*, 85-91
22 U.S. Environmental Protection Agency. Exposure Factors Handbook **1997**; EPA/600/P-95/002; EPA: Washington
23 U.S. Environmental Protection Agency. Child-specific Exposure Factors Handbook **2002**; EPA/600/P-00/002B; National Center for Environmental Assessment EPA: Washington