MASARYK UNIVERSITY

FACULTY OF SCIENCE

RECETOX

TRACKING THE CHEMICAL SIGNAL OF MODERNITY: LINKING POLICY, BEHAVIOUR AND ENVIRONMENT TO UNDERSTAND HUMAN EXPOSURE TO CHEMICAL POLLUTION

HABILITATION THESIS

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ABSTRACT

With the rapid growth of the use of synthetic chemicals and the exceedance of the planetary boundary on novel entities, new tools and multi-disciplinary strategies are needed to understand the major drivers of human exposure to synthetic chemicals of concern.

In this thesis, I summarize and link 20 studies that together provide a multi-faceted approach to improving our understanding of human exposure to chemical pollutants. The articles cover the development and improvement of sampling techniques for human exposure, the integration of multiple data sources to improve our understanding of exposure, and the integration of elements of policy, human behaviour and environmental impacts to provide a framework for human exposure evaluation. Results from original research highlight the development and optimization of sampling tools necessary to collect data in indoor and urban environments, and ensure comparability across studies. Integration of exposure estimation techniques allows us to contextualize and prioritize the broad set of chemical exposure data that is available from indoor and outdoor monitoring. The included studies also incorporate evaluation of environmental policy and regulation; this is critical in evaluating the implementation and effectiveness of policy actions to ensure a science-backed strategy for chemical management, with the ultimate goal of reducing our exposures to hazardous chemicals.

ACKNOWLEDGEMENTS

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ABBREVIATIONS

BDE-209	decabromodiphenyl ether
BFR	brominated flame retardant
c-pentaBDE	commercial pentabromodiphenyl ether mixture
DBDPE	decabromodiphenyl ethane
DDT	dichlorodiphenyltrichloroethane
DDX	DDT and related metabolites/degradation products
DEHP	bis(2-ethylhexyl)phthalate
DPHP	diphenyl phosphate
FR	flame retardant
GAPS	Global Atmospheric Passive Sampling network
HBM4EU	Human Biomonitoring for Europe
HCH	hexachlorocyclohexane
MONET	Monitoring Network
OPE	organophosphate ester
PAH	polycyclic aromatic hydrocarbon
PAS	passive air sampler
PBDE	polybrominated diphenyl ether
PCB	polychlorinated biphenyl
PCP	personal care product
PFAS	per- and polyfluoroalkyl substances
POP	persistent organic pollutant
PUF	polyurethane foam
PVC	polyvinyl chloride
SVOCs	semi-volatile organic compound
TB-117	Technical Bulletin 117
TCIPP	tris(2-chloroisopropyl)phosphate
TDCIPP	tris(1,3-dichloro-2-propyl)phosphate
TPHP	triphenyl phosphate
UNEP	United Nations Environment Program
WHO	World Health Organization
XAD	polystyrene-divinyl benzene-based resins

CONTENTS

1 INTRODUCTION

Human exposure to chemicals has been known to impact health since the 1700s – chimney sweep's cancer (later identified as scrotal cancer) was linked to exposure to chimney soot, in a paradigm-shifting insight by British doctor Percivall Pott (Dronsfield, 2006). While it was not until the 1920s that the implicated chemical was identified as benzo[a]pyrene, a polycyclic aromatic hydrocarbon (PAH) (Dronsfield, 2006), the identification of environmental exposure to chemicals as a trigger for health impacts was one of the first steps in exposure science, linking environmental causes with health outcomes. Over the past centuries, many links between chemicals and health impacts have been established, e.g., ingestion exposure to lead identified in Australian children in 1892 (Needleman, 2009), Minamata disease caused the discharge and biomagnification of mercury in the Japanese area of Minamata Bay in the 1950s (Budnik and Casteleyn, 2019), and the field of human exposure to chemicals continues to grow in complexity.

Synthetic chemicals, a basis for so much development in modern society, pose one of the biggest challenges to exposure science today. Around 350 000 chemicals are being marketed globally (Fenner and Scheringer, 2021) and the diversity and quantity of synthetic chemicals have been increasing at rates surpassing the drivers of global environmental change (Bernhardt et al., 2017). Conventional strategies are challenged by this rapid increase in complexity: the constantly increasing number of chemicals in commerce and under development requires advanced tools and strategies for a comprehensive overview of human exposure to chemicals. The planetary boundary of *Novel Entities* (Steffen et al., 2015) is estimated to have been passed as the increasing rate of production and releases of novel entities exceed society's ability to conduct safety-related assessments and monitoring (Persson et al., 2022).

This thesis presents a collection of 19 peer-reviewed papers published between 2014 and 2023 related to human exposure to chemicals. The papers are grouped according to a focus on the development and improvement of sampling techniques for human exposure (Articles 16-19), integration of multiple data sources to improve our understanding of exposure (Articles 8-15), and synthesis, integrating elements of policy, human behaviour and environmental impacts to provide a framework for human exposure evaluation that can support the challenge of increasing chemical complexity (Articles 1-7).

Currently, I have published 88 papers, including 78 articles and 10 conference proceedings. My contribution to the 19 articles selected for this habilitation thesis is summarized in the tables in the attached commentary document, giving my percentage contribution to different elements of the work.

2 A FRAMEWORK FOR UNDERSTANDING CHEMICAL EXPOSURE

Chemical exposure is typically categorized according to exposure route – for most chemicals this is distributed between inhalation, ingestion (including both diet and non-dietary ingestion), and dermal uptake. However, a broader framework is necessary to understand the relevance of specific exposure pathways for individual substances or groups of chemicals, what factors lead to within-population differences in exposures, and more broadly, identify actions which can be taken to reduce exposure to harmful chemicals.

Human chemical exposure sits at the junction of several key factors. To provide a framework for guiding research on chemical exposures, we consider a framework presenting the "lifecycle" of a chemical exposure, from product use involving direct exposures (also called "near-field" exposures in some exposure models), and indirect exposures from the wider environment (also called "far-field" exposures). A conceptual overview of this framework is presented in Figure 1

Each element of this exposure framework can be influenced by three major types of determinants of exposure:

- Chemical use, influenced by policy and regulations permitting/restricting chemicals, and industrial developments in the synthesis and production of new materials and products;
- Human behaviour, and shifts towards certain lifestyle patterns, professions, dietary patterns and consumer choices;
- Environment, combining the influence of near and far-field exposures on humans.

While there are many overlaps and interlinkages between these factors, this thesis uses these three concepts as themes to highlight key scientific understanding in each area and show my contribution to the study of human exposure to synthetic chemicals, from the development of sampling techniques through to data synthesis.

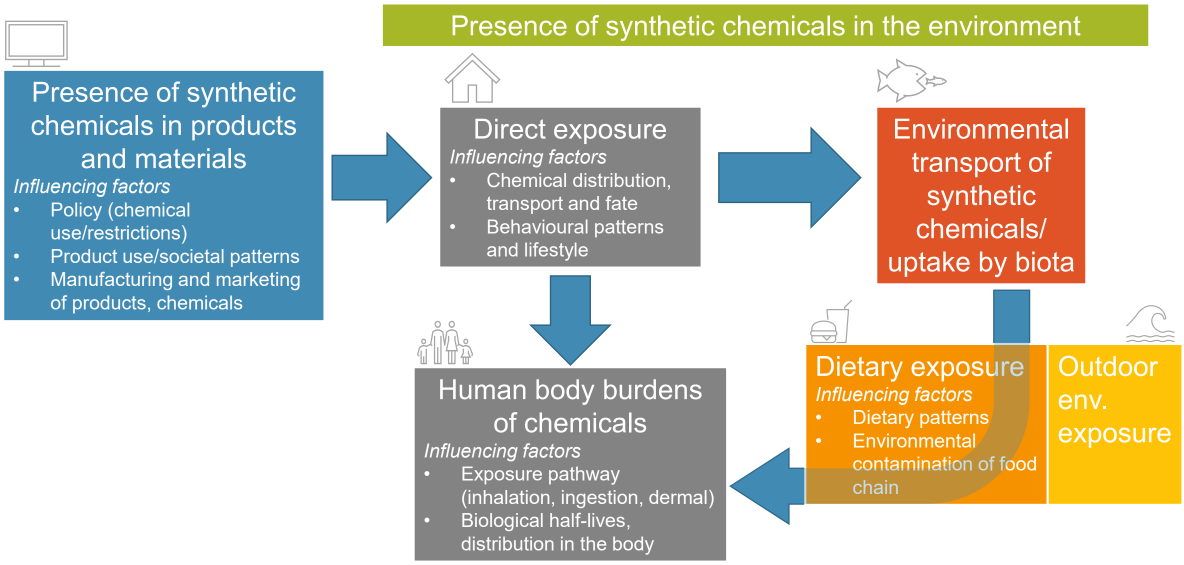


Figure 1 A conceptual representation of key factors driving human exposure to chemicals

2.1 CHEMICAL POLICY AS A DRIVER OF HUMAN EXPOSURE TO CHEMICALS

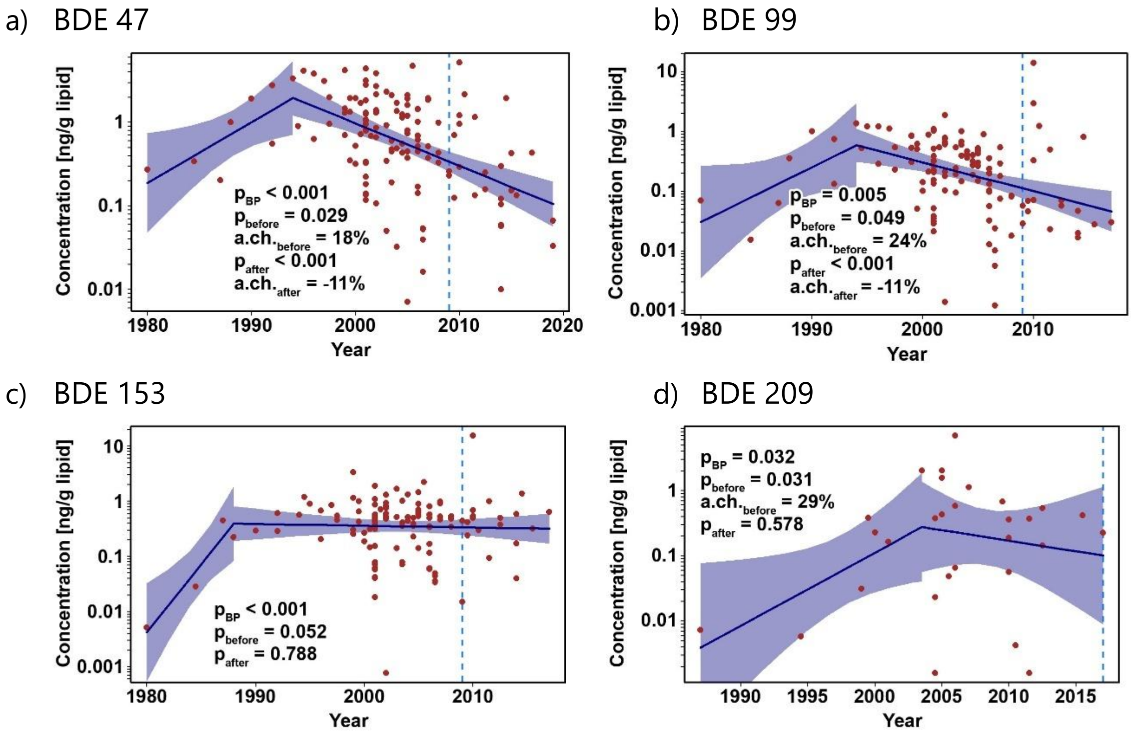
At the fundamental level, for human exposure to a synthetic chemical substance to occur, there must be sufficient use of the chemical in some application that could lead either to its direct exposure to humans, or to its release into the environment and subsequent exposure. In practice, both these aspects often occur simultaneously.

Therefore, a framework for understanding human exposure to synthetic chemicals must address the element of chemical use, through consideration of policy and regulatory aspects that lead to the use or restriction of a chemical, regulatory changes altering chemical use patterns over time, and broad societal level trends impacting exposures.

A clear potential policy impact on chemical exposures is when a restriction of a chemical is introduced; restrictions to reduce risks of hazardous chemicals are introduced with the aim that the use of the restricted substance, and consequently exposure, should decrease over time. We see clear examples of this in environmental monitoring of persistent organic pollutants (POPs), which have been restricted through the Stockholm Convention on POPs (UNEP, 2017), entered into force in 2004 and now with 186 parties. For example, air concentrations of POP chemicals such as hexachlorocyclohexane, polychlorinated biphenyls (PCBs), dichlorodiphenyltrichloroethane (DDT) and polybrominated diphenyl ethers (PBDEs) showed continuous declining trends at Arctic monitoring stations from 1992-2018, reflecting global decreases in chemical stocks and emissions (Wong et al., 2021), and similar responses to restrictions and reductions of POP use have been seen in other air (Kalina et al., 2018; Venier and Hites, 2010), sediments (Bogdal et al., 2008), and biota monitoring (Crimmins et al., 2012; Rigét et al., 2019).

While the time trends generated from routine monitoring are indicative of overall reductions in environmental emissions, and provide a suggestion of human exposure trends, more direct evaluation of the effectiveness of restrictions on reducing human exposures can be achieved through human biomonitoring (Magulova and Priceputu, 2016). Many POPs are persistent, lipophilic compounds which are best monitored through lipid-heavy matrices, e.g., blood serum and breast milk (Vorkamp et al., 2021). Human milk has been selected as a core medium of the Global Monitoring Plan of the Stockholm Convention, and has routinely been monitored through a strategic partnership between the World Health Organization (WHO) and United Nations Environment Programme (UNEP) to track patterns and trends of human exposure to chemicals (Magulova and Priceputu, 2016).

A meta-analysis of human biomonitoring data reporting levels of PBDEs was one of the first analyses to generate alarm regarding PBDE exposures: the analysis found a doubling-time of levels of PBDEs in human tissues of five years, translating to rapidly increasing levels in human populations (Hites, 2004). PBDEs are flame retardants - chemicals added to consumer products and building materials to reduce their flammability. They have been widely used in consumer products and building materials on a global basis (Abbasi et al., 2015), and the concerns raised about PBDEs' environmental persistence and toxicity (Darnerud, 2003), and specifically neurodevelopmental impacts related to children's exposure (Hudson-Hanley et al., 2018), have led to a transition over the past 20 years away from PBDE flame retardants to alternative synthetic chemicals. Our recent meta-analysis of PBDEs identified clear evidence of **a breakpoint in increasing time trends of PBDEs in human tissues, evidence of a positive impact of regulations on reducing human exposure to chemicals** (Figure 2, van der Schyff et al., 2023, Appendix 1). However, this decreasing time trend does not translate universally to all restricted chemicals; interfering factors can moderate the impact of restrictions on human exposures. In the case of PBDEs, we see evidence that differing chemical half-lives in the body delay the impact of regulations (notably seen in the lack of decrease in PBDE 153, which has a longer half-life in the human body), but additionally, we capture the lag-time between restriction of a chemical and the removal of the items containing that chemical from use. Most restrictions on a chemical eliminate the new production and sale of a chemical, but, for practical reasons, typically exempt existing uses. Without the effort to explicitly remove existing uses of a chemical, there will be a substantial lag between the restrictions on new use, and the elimination of chemical use as a source to humans and the environment. In the case of flame-retarded items, the product lifetimes vary widely, from smartphones (2-6 years) to building insulation (30-50 years), suggesting that responses to regulations will be significantly delayed for flame retardants predominantly used in insulation (van der Schyff et al., 2023).


a) BDE 47

b) BDE 99

c) BDE 153

d) BDE 209

Figure 2 Results of breakpoint analysis for PBDEs in human milk for Europe for individual PBDE congeners: BDE-47, BDE-99, BDE-153 and BDE-209. The shaded area indicates 95th percent confidence interval. The dotted blue line indicates when regulation was implemented by the Stockholm Convention (2009: BDE-47, -99, and -153; 2019: BDE-209). From van der Schyff et al. (2023).

The differing impact of regulations restricting production vs. required removal from use are seen in a global evaluation of the impact of policy of the management of PCB stocks. PCBs were widely used in electrical equipment, and as plasticizers and flame retardants from 1930s to 1980s (Breivik et al., 2002). Due to concerns about their persistence and toxicity, restrictions were introduced in many jurisdictions over the 1970s and 1980s, eventually eliminating the intentional production of PCBs (Breivik et al., 2002; Melymuk et al., 2022). However, many existing uses remained, as early laws restricted only production, and did not include a requirement to remove existing PCB stocks from use. PCBs are included in the Stockholm Convention which includes a 2025 deadline for removal all PCBs from use. Therefore, parties to the Stockholm Convention are required to take active measures to remove PCBs from use, which in most regions has resulted in decreasing stocks of PCBs (Melymuk et al., 2022, Appendix 2). However, the USA, notably with the highest historical PCB use per capita, is not a party to the Stockholm Convention, and as such, has not enacted laws requiring removal of PCBs from use. A comparative analysis of the implications of this regulatory difference examined PCB stocks over the past 15 years in Canada and Czechia (parties to the Stockholm Convention) compared with the USA (as a non-party). Our analysis identified that the active removal of PCBs resulted in a >90 % reduction in stocks of PCBs in Canada and Czechia, attributed to the active measures undertaken to ensure compliance with the Stockholm deadlines, while USA has had only a 3 % decline, showing that **without active policy measures to require chemical elimination, lag times between initial restrictions and eliminations of exposures can be very substantial** (Melymuk et al., 2022).

It is not only chemicals policy that directly impacts chemical use and therefore exposure, but other policies that trigger chemical use. The incorporation of flame retardants in products is primarily driven by regulations: the flammability standards to which an item must conform in order to be sold on a given market. Such flammability standards exist for building materials, vehicles, and in some jurisdictions, for upholstered furniture (Stapleton et al., 2012), and we can see evidence of regional differences in these flammability standards on human chemical exposure.

An early decision introduced by the state of California in the 1970s has had widespread implications for human exposure to flame retardants. California introduced a standard called TB-117 which required a high resistance to flame in upholstered furniture (Stapleton et al., 2012). Because of shared markets, this flammability standard indirectly impacted the whole USA, and to some extent, Canada, as furniture manufacturers tend to conform to the highest standards (rather than manufacturing dedicated items for California). The introduction of this flammability standard resulted in the high use of synthetic organic flame retardants in upholstered furniture to meet this standard (Stapleton et al., 2012). For the period of 1970s-2000, this flammability standard was primarily met by the incorporation of a commercial mixture of polybrominated diphenyl ethers (PBDEs), specifically c-pentaBDE, a commercial mixture of tetra, penta- and hexabromodiphenyl ether isomers. The majority of the global use of c-pentaBDE was in the USA, in large part as additives to upholstered furniture (Abbasi et al., 2019).

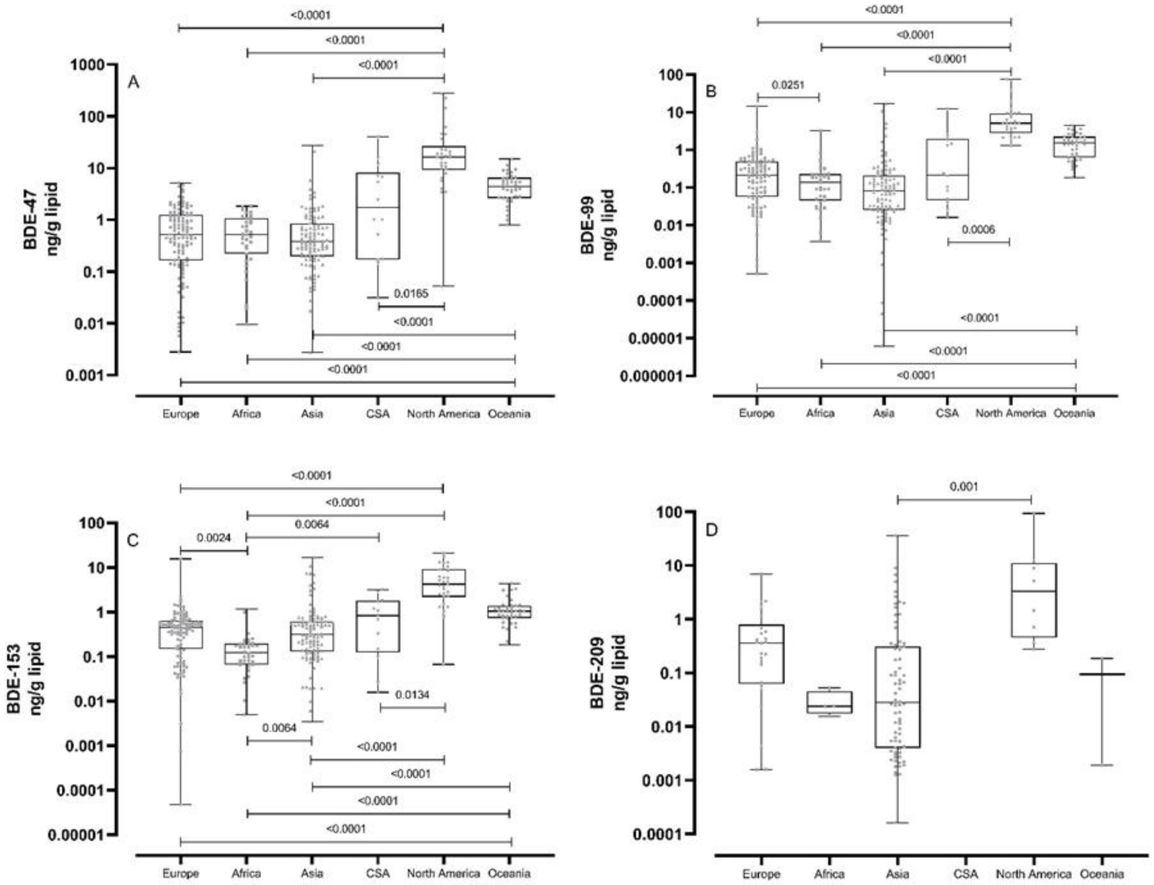


Figure 3 . Box-and-whiskers (horizontal lines are medians, 95% confidence intervals, minima, and maxima) of individual PBDE congeners: A) BDE-47, B) BDE-99, C) BDE-153, and D) BDE-209 concentrations in different regions (CSA= Central- and South America and the Caribbean), in breast milk data collected after year 2000. Non-parametric ANOVA tests (Kruskal-Wallis with Dunn's post-tests) were conducted to determine significant differences. From van der Schyff et al. (2023).

We see clear impacts of this flammability standard and the high use of PBDEs in USA when comparing measurements of PBDEs in environmental and human samples from USA vs. other countries. Our meta-analysis of existing biomonitoring data for PBDEs in human breast milk clearly identified that North American women had levels of PBDEs in their breast milk 1-2 orders of magnitude higher than other regions (Figure 3), attributed to the **higher use of PBDEs in USA, resulting in higher direct and environmental exposures.** Moreover, we noted similar variations in a comparison of American vs. Canadian and Czech homes: concentrations of PBDEs measured in surface dust from USA found levels that were consistently **1-2 orders of magnitude higher than in Czechia**, and similar geographic differences were seen in other indicators of indoor contamination – surface wipes and indoor air (Figure 4, Venier et al., 2016, Appendix 3).

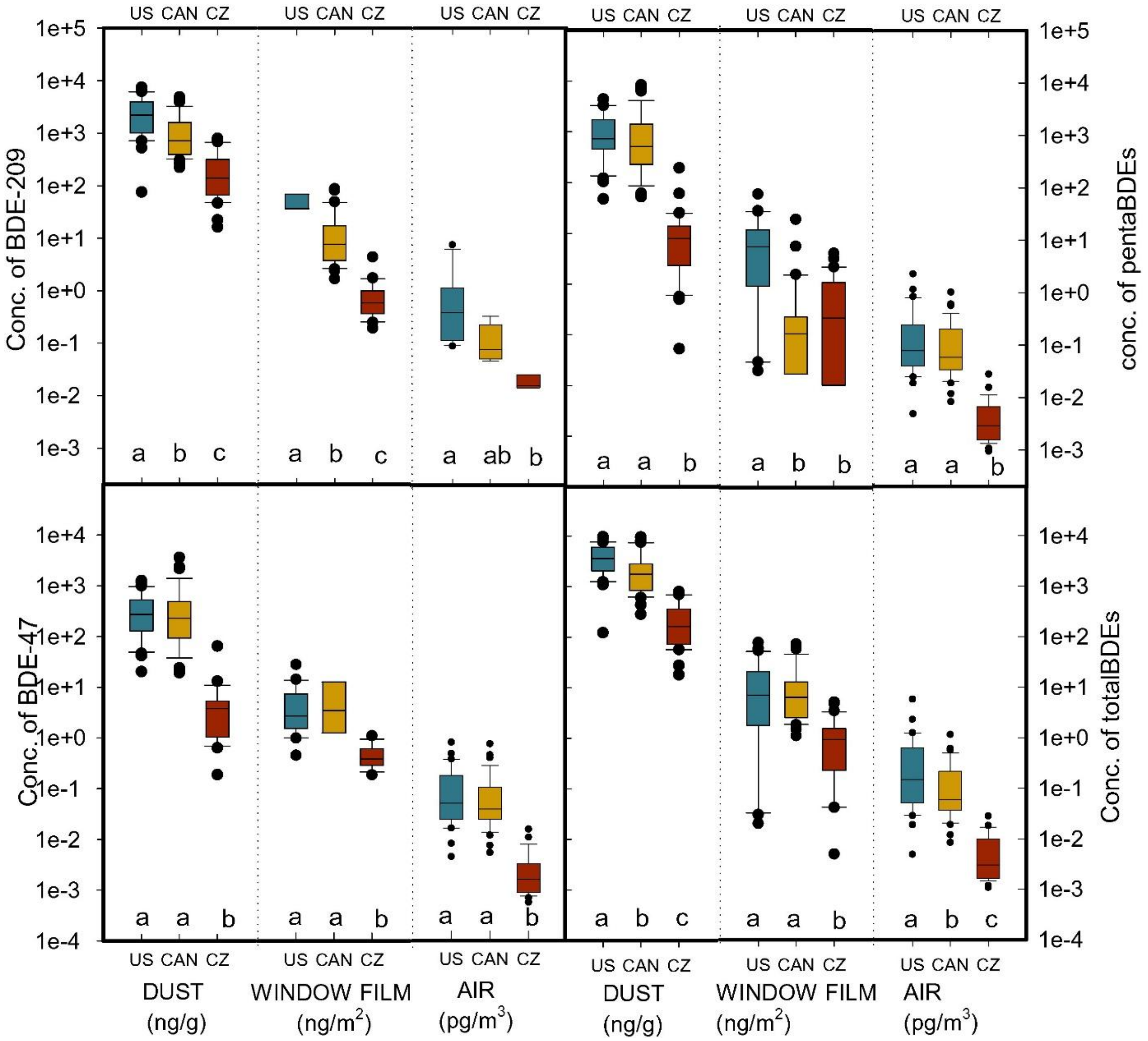


Figure 4 Boxplots of concentrations for selected PBDEs congeners. The plots share the same y-axis concentration scales although the units are different for each matrix: dust is ng/g, window film is ng/m2, and air is pg/m^3. Within each matrix, boxes that share the same letter are not statistically significantly different at a 5% level in an ANOVA analysis using the Tukey's test. From Venier et al. (2016).

The similarity in exposure profiles in many populations further emphasizes the importance of regulatory policy and other broad societal factors in driving exposures. In the recent HBM4EU (Human Biomonitoring for Europe) aligned studies, indicators of exposure to many chemicals were similar across populations from different regions, including widespread presence of many flame retardants and other plastic additives in European children, teenagers, and adults (Govarts et al., 2023). We noted similar homogeneity in a study of flame retardant and plasticizers in the homes and dermal wipes of 51 Canadian women: a consistent similarity in the profiles of both chemical groups was noted in wipes from the hands of 51 unrelated women, **demonstrating the ubiquity of the target**

chemicals in modern environments, regardless of the participants' home contents, activities, or other personal characteristics (Diamond et al., 2021, Appendix 4).

2.2 LIFESTYLE IMPACTS ON CHEMICAL EXPOSURE

While policy and regulations have broader societal implications on chemical exposures, many of the differences in human exposure to chemicals are driven by individual variations in lifestyle and behaviour. For example, while our HBM4EU study found only a 50% difference in population medians of diphenyl phosphate (DPHP; metabolite of TPHP) across countries, the variation within individual populations far exceeds that, covering a 10-fold range in most regions (Figure 5), suggesting the importance of individual factors in controlling exposures.

Variations in the use of personal care products (PCPs) within a population is a clear source of within-population variations in exposures. Use of personal care products has been linked with exposures to synthetic chemicals used as additives in the products or their packaging. This includes exposure to TPHP from nail polish use (Mendelsohn et al., 2016), exposure to synthetic fragrance compounds from the use of perfumes (Nakata et al., 2015), and exposure to UV filters from sunscreen use (Krause et al., 2017). **In Czech teenagers and young adults, we found that higher use of PCPs is associated with higher levels of phthalate ester metabolites in urine** (Stuchlík Fišerová et al., 2022, Appendix 5), **as phthalate esters are used as carriers and plasticizers in PCP formulations and packaging**, and this link has been shown in other analyses of biomonitoring data for both men and women (Fruh et al., 2022; Nassan et al., 2017).

The case of PCPs is also an example of how shifts in personal behaviour can alter chemical impacts. In many markets, including the cosmetics market, there is an increasing share of "green" products and a societal shift towards what are perceived to be "safer" products can also translate into shifts in chemical exposures (Chin et al., 2018). Studies evaluating the impact of a shift to products "free-from" certain chemicals have shown impacts: when consumers intentionally replace products with those with lower levels of parabens, UV filters and biocides, indicators of exposure, in this case urinary metabolites, have been shown to decrease (Harley et al., 2016). However, there remain many elements of uncertainty in the shift towards "safer" products; one important one is the uncertainty associated with various green labels and other marketing elements suggesting green/safe products. The use of key phrases associated with "safer" products (e.g. eco, bio, free-from) can, in some cases, constitute "greenwashing". In a set of 50 products available on the Czech market, we observed no significant difference in concentrations of parabens and triclosan between conventional PCPs and those with a "green"-indicating label, suggesting a disconnect between a societal shift towards interest in safer products, and actual human exposure (van der Schyff et al., 2022, Appendix 6). **Thus, in some cases, shifts in personal**

behaviour may not have the expected or intended impact, and it is crucial that recommended actions to reduce exposure to hazardous chemicals have a clear scientific backing.

Given that indoor dust is a known reservoir of flame retardants (Jílková et al., 2018; Venier et al., 2016; Vykoukalová et al., 2017) removing dust from living spaces is a good way to limit exposure, and has been linked to lower levels of flame retardants in indoor environments (Sugeng et al., 2018).

2.3 ENVIRONMENT AS A DETERMINANT OF CHEMICAL EXPOSURE

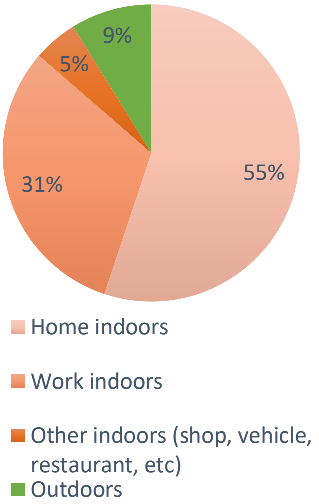


Figure 5 Typical distribution of time for adult, Prague. Data from Schweizer et al. (2007).

Occupation is known to be a key determinant of exposure for many occupations associated with high chemical releases, for example, e-waste processing (Balasch et al., 2022; Julander et al., 2014; Nguyen et al., 2019) and construction (Estill et al., 2020; Jarvholm, 2006; Wingfors et al., 2006). However, with the ubiquity of synthetic materials containing additive chemicals in all indoor environments, it is not only industrial occupations that are associated with chemical exposure, but rather all time spent indoors. In developed countries, we spend an average of 90% of our time in indoor spaces (Figure 6), largely distributed between home and work/school, and consequently, these environments are key determinants of our chemical exposures (Matz et al., 2014; Schweizer et al., 2007).

When spending time indoors, we inhale indoor air, including indoor aerosols, and through hand-to-mouth behaviours, accidentally ingest settled dust particles, which can be a particularly large reservoir of lipophilic compounds (Weschler and Nazaroff, 2010). We additionally have direct contact with consumer products and dust particles, leading to potential dermal exposure (Weschler and Nazaroff, 2012). While initial concerns about indoor environments focussed largely on inhalation exposure, and this is often dominant in occupational exposure settings, non-dietary ingestion and dermal contact are important exposure pathways in residential and non-industrial settings (Lioy et al., 2002; Salthammer et al., 2018). Young children are particularly susceptible to high exposures to house dust given their increased frequency of hand-to-mouth behaviour and mouthing of objects (Moya and Phillips, 2014); ingestion of indoor dust is the dominant pathway for young children’s exposure to some flame retardants (Demirtepe et al., 2019; Johnson-Restrepo and Kannan, 2009). Dermal exposure, although receiving limited attention, also appears to be an important pathway. Comparisons of hand wipe profiles of Canadian women with the chemical profiles on the surfaces of their electronics found similarities in the profiles between the flame retardants in hand-held electronics and their hand surfaces (Diamond et al., 2021), although substantial uncertainty remains regarding dermal penetration. Estimates of the contribution of inhalation, ingestion and dermal exposure pathways in Slovak children found that **dermal contact contributed ~30% of total exposures for PBDEs and halogenated OPEs** (Demirtepe et al., 2019, Appendix 7). There is a large set of studies suggesting that the concentrations of air and dust in indoor spaces are associated with human exposure to a range of chemicals, notably for flame retardants, PCBs and phthalate esters (Coakley et al., 2013; Fromme et al., 2014; Herrick et al., 2011), and thus determinants of indoor environmental quality will also be determinants of human exposure to selected chemicals.

The age of a building is frequently cited as a primary environmental factor impacting chemical levels in indoor air and dust. There are two layers to this age impact. One is related to chemical regulations as discussed in section 2.1. Older buildings can contain elevated residues of chemicals which are now restricted, particularly those that were built at the time of peak use of some legacy POPs of concern, e.g., PCBs and DDT. In some cases, this is related to documented past use of the chemicals indoors. For example, a wood preservative called pentalidol (2% DDT; 5% pentachlorophenol and 0.1% γ-hexachlorocyclohexane (HCH)) was applied in the Baroque theater of Český Krumlov Castle in the 1970s and 1980s to address dry rot, and this resulted in acute health impacts in the 1990s, before remediation removed 5 t of contaminated material (Holt et al., 2017, Appendix 8). However, **even after this removal exceedances of exposure guidance values remained for DDT and HCHs, and indoor levels were orders of magnitude above background levels** (Holt et al., 2017), **indicating the persistence of these chemicals indoors, as well as their emission from secondary indoor sources**. But in many cases, surveys of indoor air or dust find an association between levels of legacy POPs in indoor air and dust without documented sources, indicating past general use of the chemicals. For example, **higher levels of OCPs were detected in dust and air from Slovak homes built before restrictions on the chemical use** (Figure 7, Demirtepe et al., 2019, Appendix 7). This is in line with what is known about the past use of both PCBs and DDT, as uses were very diverse, but were not well-documented.

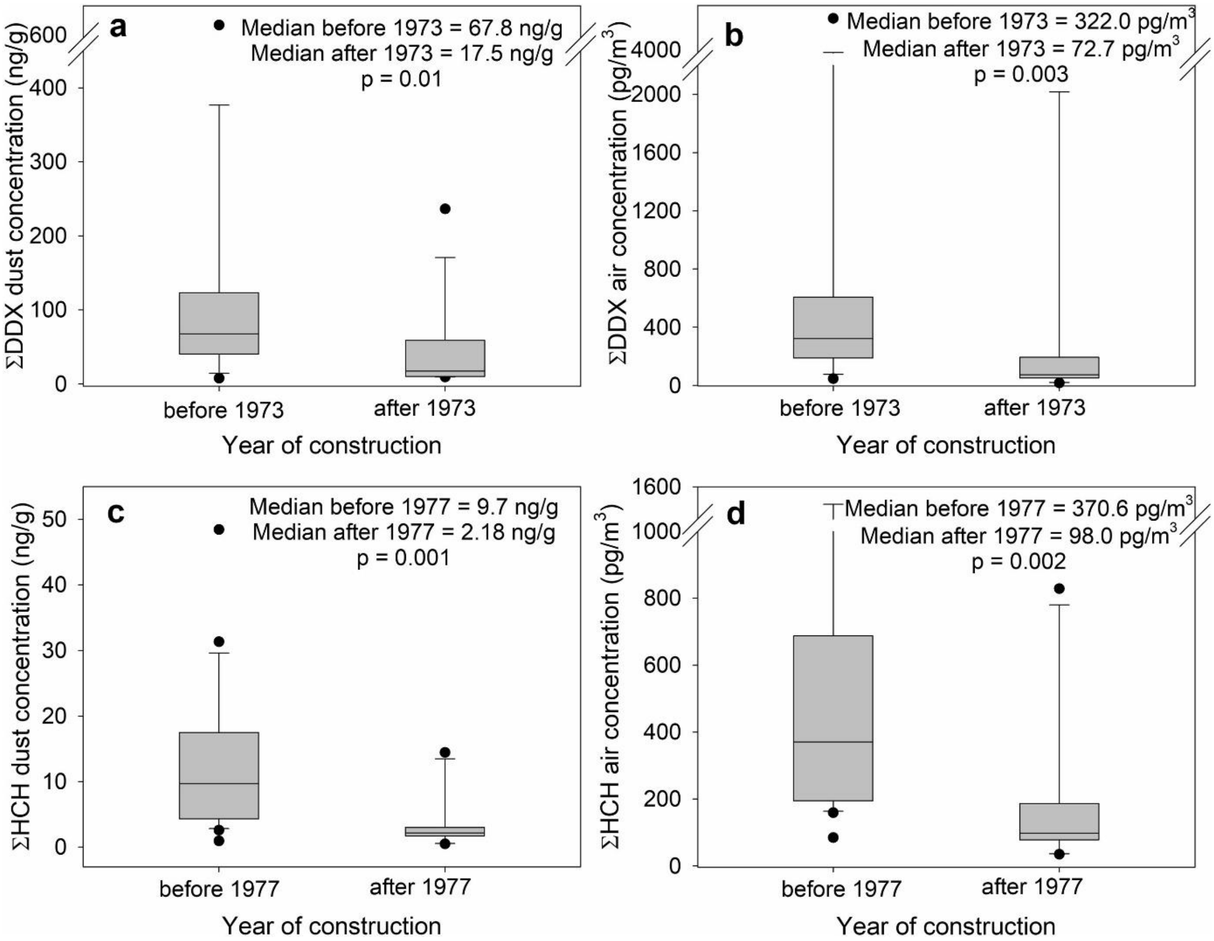


Figure 6 Differences in indoor dust and air concentrations of ∑DDX (a and b) and ∑HCH (c and d) measured in homes built before (n = 18 for ∑DDX, n = 22 for ∑HCH) and after (n = 14 for ∑DDX, n = 10 for ∑HCH) OCPs controls were implemented in Czechoslovakia. Excerpted from Demirtepe et al. (2019).

The properties of indoor environments are ideal for the long-term persistence of chemical residues. The potential for degradation is much lower than outdoors due to lower solar radiation, lower levels of atmospheric oxidants, high surface area-to-volume ratios, small variations in temperature, and low air exchange (Abbatt and Wang, 2020). As a result, chemical half-lives indoors can be substantially longer than what is estimated for outdoor environments, and these are poorly characterized (Abbatt and Wang, 2020).

However, on top of differences seen due to regulations, in newer buildings, we observe the impact of a transition to indoor environments based heavily on synthetic polymer-based materials. The past 70 years have seen a substantial increase in the amount of synthetic materials used in indoor spaces (Weschler 2009), and in conjunction, an increase in the amount of chemical additives found in these materials. Typical building materials and consumer products today have a heavy reliance on plastics (e.g., building materials), and consequently, higher use of chemical additives such as plasticizers, flame retardants, stabilizers and antioxidants in indoor spaces. Many of these chemicals are semi-volatile, and the exposure pathways are then multifaceted, as these semi-volatile organic compounds (SVOCs) are distributed through multiple phases in indoor environments, leading to exposure through air, dust and surface contact (Weschler and Nazaroff, 2008). This wide range of synthetic chemicals used as additives in building materials and furnishings is now consistently detected in most indoor spaces, the most notable being phthalate esters, organophosphate flame retardants, chlorinated paraffins, and siloxanes (Lucattini et al., 2018).

Building materials and consumer products have very large ranges in the presence of additive chemicals. In an examination of the chemical content of 126 Czech consumer products and building materials, we identified **order of magnitude ranges even within individual product groups.** For example, levels of perfluoroalkyl acids ranged from 0.0068-34.3 µg/kg across 16 insulation materials (Bečanová et al., 2016, Appendix 9), while BDE-209 ranged from 1.77-626000 µg/kg (Vojta et al., 2017a, Appendix 10). It is clear that the chemical content of consumer products and building materials has a direct impact on indoor environmental levels and human exposures, however the larger variability in product groups creates a challenge in generalizing sources. Through the subsequent introduction of new materials and use of those materials in new buildings, we see an increase in the indoor environmental burdens of chemicals. During the construction phases of a university lecture room, the **levels of flame retardants increased** in steps with the addition of carpet, furnishings, and **most substantially, with turning on of computers,** suggesting the specific contribution of these items to the indoor environmental burden of chemicals (Vojta et al., 2017b, Appendix 11). We see in new indoor environments a **transition from the older brominated flame retardants** (Venier et al., 2016, Appendix 3) **to the replacement chemicals, which today are found at levels 1-3 orders of magnitude higher** than older BFRs (Vykoukalová et al., 2017, Appendix 12).

Ventilation rates are another key factor impacting chemical levels indoors: for most synthetic chemicals, higher ventilation rates will lead to reductions in indoor chemical levels, as the primary chemical sources are in the indoor space. In urban areas, the impact of ventilation as a source to outdoors is substantial enough to one of the main factors driving outdoor air concentrations of synthetic organic chemicals (Björklund et al., 2012; Melymuk et al., 2012). In Czech homes, **lower levels of novel flame retardants were observed in summer compared with winter**, attributed to higher summer air exchange rates due to open windows (Melymuk et al., 2016, Appendix 13).

However, many more indoor determinants are chemical-specific. The presence of PVC flooring in homes has been associated with higher levels of phthalates (Bornehag et al., 2005), while the presence of TVs and electronics in general has been associated with higher levels of flame retardants (Harrad et al., 2009; Muenhor and Harrad, 2012; Yang et al., 2020). Reductions in indoor settled dust levels of bis(2-ethylhexyl)phthalate (DEHP), a phthalate ester that has now been restricted in many applications, suggest that stocks and indoor uses of DEHP are being rapidly removed from indoor environments, and **provide evidence**

of the effectiveness of restrictions in reducing individual chemical exposures (Demirtepe et al., 2021, Appendix 14).

While for many chemicals the indoor environment is the dominant exposure determinant, outdoor exposures can be important for certain chemicals and lifestyle patterns. Of the exposure pathways outside, air inhalation is most often the focus, although soil ingestion can be substantial for young children (Özkaynak et al., 2023). Established air monitoring networks exist to track synthetic chemicals in outdoor environments. For SVOCs, these are notably the MONET networks (White et al., 2023) and the GAPS network (Pozo et al., 2006; Saini et al., 2020), both of which rely on passive air sampling to increase the spatial resolution of available data. Passive air sampling (PAS) relies on passive diffusion of chemicals from air to a sorbent (Shoeib and Harner, 2002; Wania and Shunthirasingham, 2020), in the case of MONET and GAPS these sorbents are based on polyurethane foam (Harner et al., 2013; Melymuk et al., 2011), XAD resin (Armitage et al., 2013), or a combination of both (Schuster et al., 2012). They have the advantage in outdoor environments of not requiring electricity, being low-cost and easy to deploy, which has greatly increased the spatial resolution of outdoor air monitoring. However, even with the greater spatial resolution possible because of passive sampling, measurements are limited to point locations, which may not reflect the small-scale variations observed in populated areas (Melymuk et al., 2013, 2012). To supplement these air monitoring networks, data modeling techniques can be incorporated to provide better spatial resolution of air concentration estimates that can be incorporated into exposure assessment (Hoek et al., 2008; Mikeš et al., 2023). Land use regression has proved to be a useful technique to translate coverage of passive air sampling networks into concentration maps providing better spatial resolution. For Czechia, outdoor ambient air concentrations of **PCBs were found to be dependent on soil concentrations and topography**, while **PAHs were related to fuel consumption and industrial sources, as well as topography** (White et al., 2021, Appendix 15); the identification of such relationships allows point passive sampler measurements to be extended to provide broader spatial coverage. The regression relationships generated by this analysis could be used to produce maps highlighting regional variations in outdoor concentrations (Figure 8). Similar techniques have been applied for other air pollutants (Mikeš et al., 2023) and SVOCs in different locations (Melymuk et al., 2013).

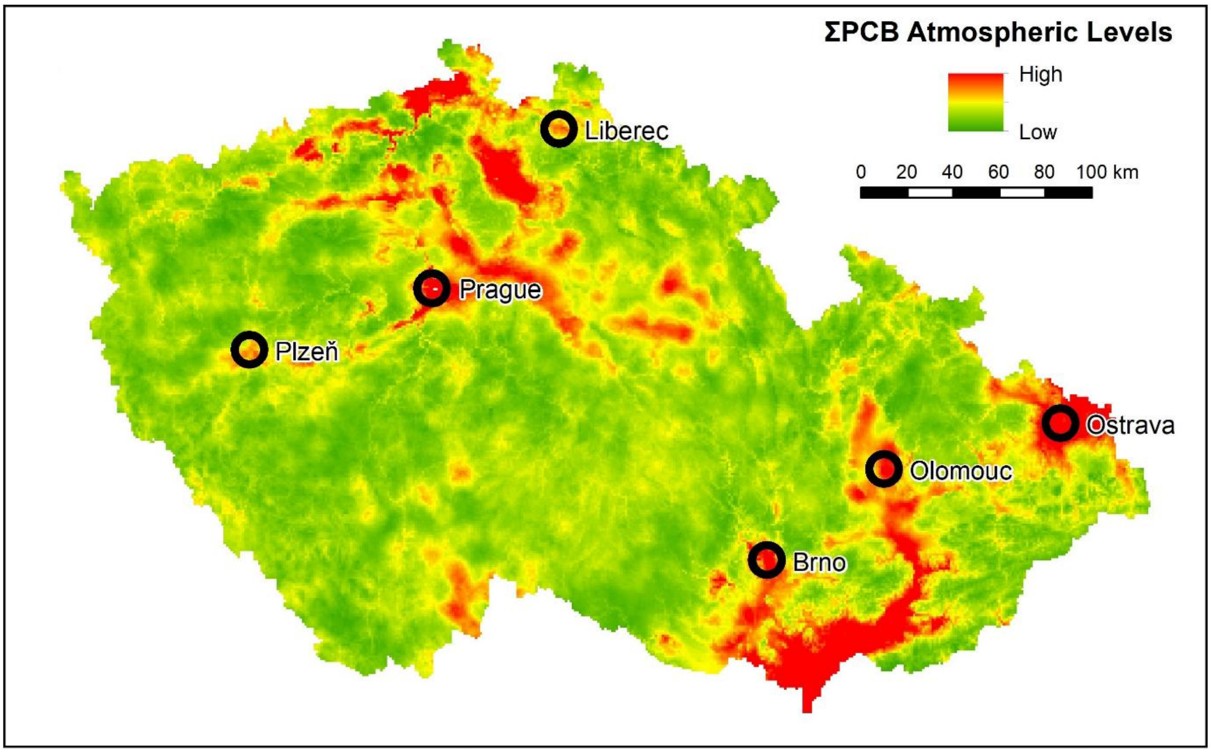


Figure 7 Application of the LUR models across all of Czechia, indicating the estimated gradient of low to high atmospheric levels of Σ_7PCB, (R^2 = 0.62). Cities with population > 100,000 are indicated on the map. From White et al. (2021)

3 TOOLS AND TECHNIQUES FOR UNDERSTANDING CHEMICAL EXPOSURES

3.1 AIR SAMPLING TECHNIQUES

A comprehensive understanding of human exposure requires a diverse set of sampling, analytical, and data-handling tools. One fundamental step in characterizing human exposure pathways is selection of representative and unbiased environmental samples. For characterization of indoor environments, a set of sampling techniques unique to indoor spaces is often used.

Passive air samplers, particularly polyurethane foam-based passive air samplers (PUF-PAS) have been a major innovation in indoor environments, with the significant advantages of being non-intrusive indoors and providing integrated concentrations, typically over one month (Bohlin et al., 2014; Vojta et al., 2024). While some uncertainty remains in the conversion of masses sampled to quantified air concentrations due to uncertainty in air uptake to the sampler (Wania and Shunthirasingham, 2020) indoor environments are actually ideal for PAS, as they are not subject to the large variations in temperature and wind speed, which are known to vary sampling rates outdoors (Bohlin-Nizzetto et al., 2020;

Chaemfa et al., 2009), and lower levels of atmospheric oxidants, which should minimize the within-sampler degradation that has been noted in outdoor PUF-PAS (Melymuk et al., 2017). The applicability of PUF-PAS is broad, however some limitations exist. As a sorbent, PUF is best suited to lipophilic compounds of intermediate volatility – more volatile compounds may be susceptible to non-linear uptake or breakthrough (Melymuk et al., 2014, Appendix 16), while polar compounds are not well-captured by the sorbent. This presents a particular limitation for their application to per- and polyfluoroalkyl substances (PFAS), as **PUF-PAS have not been found to have consistent uptake of many PFAS** over a typical deployment period, which precludes the conversion of masses sorbed to air concentrations (Karásková et al., 2018, Appendix 17).

PUF-PAS were originally developed for use outdoors (Bidleman and Melymuk, 2019; Harner et al., 2006), and this remains the most substantial use (e.g., GAPS and MONET, as mentioned in above). The low cost and ease of use has led to many regional variations in the samplers with slight differences in sampler geometry, sorbent mass and density (Melymuk et al., 2021, Appendix 18). However, the **sampler performance appears substantially robust that such small variations do not lead to large impacts on chemical uptake to the PUF**; variations caused by sampler geometry are within the same ranges as uncertainties due to wind and temperature impacts on the samplers (Bohlin-Nizzetto et al., 2020; Chaemfa et al., 2009; Melymuk et al., 2021).

3.2 DUST SAMPLING TECHNIQUES

Many broad surveys of chemical pollution in indoor spaces use settled indoor dust as an indicator of indoor levels and human exposure (Lioy et al., 2002; Melymuk et al., 2020). Dust has the advantage of being easy to collect, a relatively stable matrix that can be easily transported, and typically has high levels of SVOC chemicals, making it a good screening matrix for a broad set of compounds. In addition, unlike PAS, it presents a reliable matrix for the quantification of indoor levels of PFAS (Karásková et al., 2016), which are of particular interest given the EU universal PFAS restriction proposal of 2023 (ECHA, 2024). However, in order to collect a dust sample representative of the average conditions of an indoor space, care must be taken to collect an aggregated sample: within an indoor space. Substantial variations in the chemical levels in indoor dust exist between different surfaces (Jílková et al., 2018, Appendix 19); **samples from individual surfaces can be strongly influenced by individual products and not representative of the whole room conditions**. For example, in floor dust from classrooms was substantially elevated in two flame retardants, TCIPP and DBDPE, relative to dust on surfaces, whereas in a flat, surface dust was elevated in the same compounds (Figure 9, Jílková et al., 2018). Given the small-scale spatial heterogeneity in dust concentrations, composite or aggregated samples are needed to reflect average exposures; localized samples can show substantial bias.

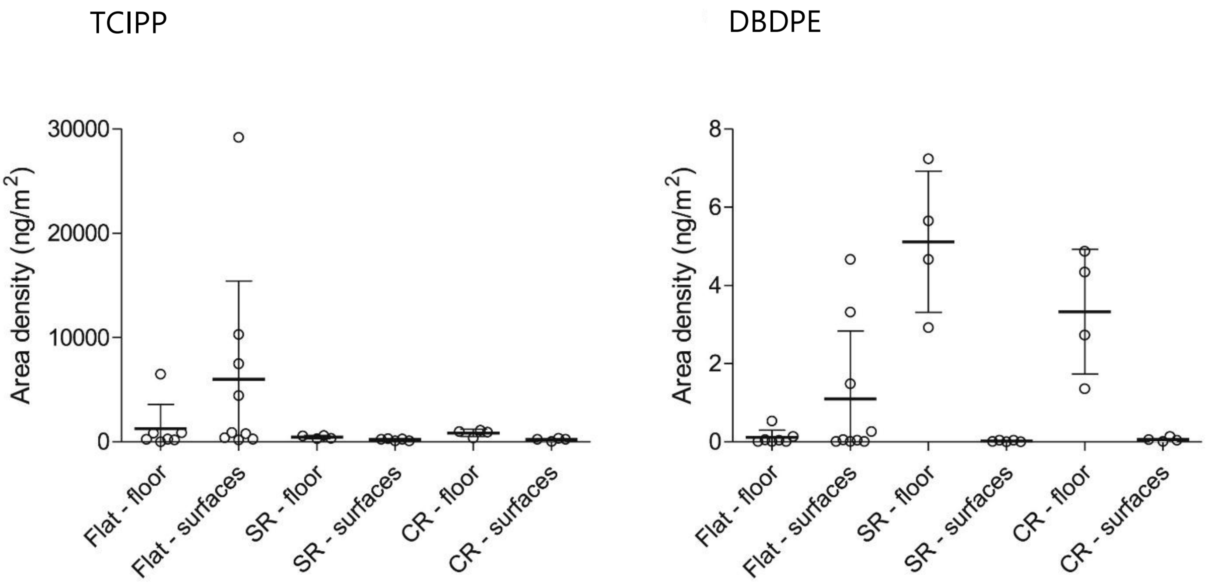


Figure 8 Area density of TCIPP and DBDPE in a flat and a classroom floor dust and wipe samples. Horizontal lines show mean and whiskers show standard deviation. Extracted from Jilkova et al. (2018). SR refers to Seminar room and CR refers to Computer Room.

3.3 ANALYTICAL CONSIDERATIONS

While uncertainties in sampler configurations and implementation can impact data quality and comparability across studies, the greater source of uncertainty is consistently due to aspects of the laboratory analysis. In the global intercomparison of PAS configurations discussed in Section 3.1, while sampler configurations accounted for 50% variations between concentrations, **laboratory analysis and quantification were responsible for up to four orders of magnitude variations in concentrations of some POPs** (Melymuk et al., 2021, Appendix 19). Similarly, for FRs the variations in elements of the analytical method across laboratories lead to substantial variations in reported chemical concentrations and present challenges when comparing results across studies (Melymuk et al., 2018).

Given the diversity of chemicals of interest in indoor environments, interest is growing in non-target and suspect screening analysis of chemicals. While these are starting to be more widely used on indoor dust in particular, a challenge still exists in the comparability and the ability to interpret this data at a sufficient quality necessary to understand exposures (Caballero-Casero et al., 2021; Hollender et al., 2023; Rostkowski et al., 2019). This will be one of the major future challenges as we develop the field of indoor exposure assessment.

Moreover, interpretation of exposure data can rarely be effectively done in the absence of other supporting information, taking the form of questionnaires, chemical use databases, product content, and time-activity patterns. It is crucial that these datasets be given equal consideration as the chemical datasets, as evidence of exposures in the absence of source information has limited value.

4 CONCLUSIONS

Our framework for understanding the drivers of chemical exposure provides a structure for the identification of the most substantial drivers. The goal of exposure assessment is to contribute to an overall risk evaluation and reduction process to identify chemical risks, and provide the data needed to take actions to reduce such risks.

These actions can take many forms, be it regulatory, behavioural or technological, however, the necessary fundament to selecting the correct actions is a sufficient understanding of the sources and key factors driving the exposure. Thus, our evaluation of exposures in the context of regulation, environment and personal behaviours provides a valuable resource for both evaluation exposure, and where needed supporting actions to reduce risks.

Regulatory actions have been shown to have a clear impact on either increasing or decreasing exposures, and ongoing effectiveness evaluation is crucial to support the most effective implementation of chemical policy, and other regulatory actions that impact chemical exposures (Chapter 2.1). Behaviour and lifestyle provide an additional, more individual layer to understanding exposures, with the potential to lead to wide variations in exposures within similar populations (Chapter 2.2). Finally, our environment, particularly the indoor spaces where we spend the majority of our time, is a major driver of exposure to selected chemicals (Chapter 2.3), and the impact of the environment and, in particular, actionable solutions to improve environmental quality to reduce exposures, has not been fully exploited in the current scientific framework.

We have developed a reliable set of sampling and analytical tools that can enable us to explore the links between environment, behaviour, policy and exposure (Chapter 3). Many of these tools have the advantage of broad applicability for chemicals of interest, and ease of use enabling widespread data collection. However, better interlinkages and integrations of techniques are needed to allow more comprehensive determinations and data gap-filling.

5 LIST OF APPENDICES

1. van der Schyff, Veronica, Jiří Kalina, Annalisa Abballe, Anna Laura Iamiceli, Eva Govarts, and Lisa Melymuk. 2023. "Has Regulatory Action Reduced Human Exposure to Flame Retardants?" *Environmental Science & Technology* 57 (48): 19106-24. https://doi.org/10.1021/acs.est.3c02896

2. Melymuk, Lisa, Jonathan Blumenthal, Ondřej Sáňka, Adriana Shu-Yin, Veena Singla, Kateřina Sebková, Kristi Pullen Fedinick, and Miriam L Diamond. 2022. "Persistent Problem: Global Challenges to Managing PCBs." *Environmental Science & Technology* 2022: 9040. https://doi.org/10.1021/acs.est.2c01204

3. Venier, Marta, Ondřej Audy, Šimon Vojta, Jitka Bečanová, Kevin Romanak, Lisa Melymuk, Martina Krátká, Petr Kukucka, Joseph Okeme, Amandeep Saini, Miriam L. Diamond, Jana Klanova. 2016. "Brominated Flame Retardants in Indoor Environment - Comparative Study of Indoor Contamination from Three Countries." *Environment International* 94: 150-60. https://doi.org/10.1016/j.envint.2016.04.029

4. Diamond, Miriam L., Joseph O. Okeme, and Lisa Melymuk. 2021. "Hands as Agents of Chemical Transport in the Indoor Environment." *Environmental Science & Technology Letters* 8 (4): 326-32. https://doi.org/10.1021/acs.estlett.0c01006

5. Stuchlík Fišerová, Petra, Lisa Melymuk, Klára Komprdová, Elena Domínguez-Romero, Martin Scheringer, Jiří Kohoutek, Petra Přibylová, Lenka Andrýsková, Pavel Piler, Holger M. Koch, Martin Zvonař, Marta Esteban-López, Argelia Castaño, Jana Klánová. 2022. "Personal Care Product Use and Lifestyle Affect Phthalate and DINCH Metabolite Levels in Teenagers and Young Adults." *Environmental Research* 213: 113675. https://doi.org/10.1016/j.envres.2022.113675

6. Schyff, Veronica van der, Lenka Suchánková, Katerina Kademoglou, Lisa Melymuk, and Jana Klánová. 2022. "Parabens and Antimicrobial Compounds in Conventional and 'Green' Personal Care Products." *Chemosphere* 297: 1-8. https://doi.org/10.1016/j.Chemosphere.2022.134019

7. Demirtepe, Hale, Lisa Melymuk, Miriam L. Diamond, Lola Bajard, Šimon Vojta, Roman Prokeš, Ondřej Sáňka, Jana Klánová, Ľubica Palkovičová Murínová, Denisa Richterová, Vladimíra Rašplová, Tomáš Trnovec. 2019. "Linking Past Uses of Legacy SVOCs with Today's Indoor Levels and Human Exposure." *Environment International* 127: 653-63. https://doi.org/10.1016/j.envint.2019.04.001

8. Holt, Eva, Ondřej Audy, Petra Booij, Lisa Melymuk, Roman Prokes, and Jana Klánová. 2017. "Organochlorine Pesticides in the Indoor Air of a Theatre and Museum in the Czech Republic: Inhalation Exposure and Cancer Risk." *Science of The Total Environment* 609: 598-606. https://doi.org/10.1016/j.scitotenv.2017.07.203

9. Bečanová, Jitka, Lisa Melymuk, Šimon Vojta, Klára Komprdová, and Jana Klánová. 2016. "Screening for Perfluoroalkyl Acids in Consumer Products, Building Materials and Wastes." *Chemosphere* 164: 322-29. https://doi.org/10.1016/j.Chemosphere.2016.08.112

10. Vojta, Šimon, Jitka Bečanová, Lisa Melymuk, Klára Komprdová, Jiří Kohoutek, Petr Kukučka, and Jana Klánová. 2017. "Screening for Halogenated Flame Retardants in European Consumer Products, Building Materials and Wastes." *Chemosphere* 168: 457-66. https://doi.org/10.1016/j.Chemosphere.2016.11.032

11. Vojta, Šimon, Lisa Melymuk, and Jana Klánová. 2017. "Changes in Flame Retardant and Legacy Contaminant Concentrations in Indoor Air during Building Construction, Furnishing, and Use." *Environmental Science & Technology* 51 (20): 11891-99. https://doi.org/10.1021/acs.est.7b03245

12. Vykoukalová, Martina, Marta Venier, Šimon Vojta, Lisa Melymuk, Jitka Bečanová, Kevin Romanak, Roman Prokeš, Joseph O. Okeme, Amandeep Saini, Miriam L. Diamond, Jana Klanova. 2017. "Organophosphate Esters Flame Retardants in the Indoor Environment." *Environment International* 106: 97-104. https://doi.org/10.1016/j.envint.2017.05.020

13. Melymuk, Lisa, Pernilla Bohlin-Nizzetto, Petr Kukučka, Šimon Vojta, Jiří Kalina, Pavel Cupr, Jana Klánová, Pavel Čupr, and Jana Klánová. 2016. "Seasonality and Indoor/Outdoor Relationships of Flame Retardants and PCBs in Residential Air." *Environmental Pollution* 218: 392-401. https://doi.org/10.1016/j.envpol.2016.07.018

14. Demirtepe, Hale, Lisa Melymuk, Garry Codling, Ľubica Palkovičová Murínová, Denisa Richterová, Vladimíra Rašplová, Tomáš Trnovec, and Jana Klánová. 2021. "Targeted and Suspect Screening of Plasticizers in House Dust to Assess Cumulative Human Exposure Risk." *Science of The Total Environment* 781: 146667. https://doi.org/10.1016/j.scitotenv.2021.146667

15. White, Kevin B., Ondřej Sáňka, Lisa Melymuk, Petra Přibylová, and Jana Klánová. 2021. "Application of Land Use Regression Modelling to Describe Atmospheric Levels of Semivolatile Organic Compounds on a National Scale." *Science of The Total Environment* 793. https://doi.org/10.1016/j.scitotenv.2021.148520

16. Melymuk, Lisa, Pernilla Bohlin-Nizzetto, Ondřej Sáňka, Karla Pozo, and Jana Klánová. 2014. "Current Challenges in Air Sampling of Semivolatile Organic Contaminants: Sampling Artifacts and Their Influence on Data Comparability." *Environmental Science & Technology* 48 (24): 14077-91. https://doi.org/10.1021/es502164r

17. Karásková, Pavlína, Garry P. Codling, Lisa Melymuk, and Jana Klánová. 2018. "A Critical Assessment of Passive Air Samplers for Per- and Polyfluoroalkyl Substances." *Atmospheric Environment* 185: 186-95. https://doi.org/10.1016/j.atmosenv.2018.05.030

18. Melymuk, Lisa, Pernilla Bohlin Nizzetto, Tom Harner, Kevin B. White, Xianyu Wang, Maria Yumiko Tominaga, Jun He, Jun Li, Jianmin Ma, Wan-Li Ma, Beatriz H. Aristizábal, Annekatrin Dreyer, Begoña Jiménez, Juan Muñoz-Arnanz, Mustafa Odabasi, Yetkin Dumanoglu, Baris Yaman, Carola Graf, Andrew Sweetman, Jana Klánová. 2021. "Global Intercomparison of Polyurethane Foam Passive Air Samplers Evaluating Sources of Variability in SVOC Measurements." *Environmental Science and Policy* 125: 1-9. https://doi.org/10.1016/j.envsci.2021.08.003

19. Jílková, Simona, Lisa Melymuk, Šimon Vojta, Martina Vykoukalová, Pernilla Bohlin-Nizzetto, and Jana Klánová. 2018. "Small-Scale Spatial Variability of Flame Retardants in Indoor Dust and Implications for Dust Sampling." *Chemosphere* 206: 132-41. https://doi.org/10.1016/J.CHEMOSPHERE.2018.04.146

6 REFERENCES

Abbasi, G., Li, L., Breivik, K., 2019. Global Historical Stocks and Emissions of PBDEs. Environmental Science & Technology 53, 6330–6340. https://doi.org/10.1021/acs.est.8b07032

Abbasi, G., M. Buser, A., Soehl, A., W. Murray, M., L. Diamond, M., 2015. Stocks and flows of PBDEs in products from use to waste in the U.S. and Canada from 1970 to 2020. Environmental science & technology 49, 1521–1528. https://doi.org/10.1021/es504007v

Abbatt, J.P.D., Wang, C., 2020. The atmospheric chemistry of indoor environments. Environmental Science: Processes and Impacts 22, 25–48. https://doi.org/10.1039/c9em00386j

Armitage, J.M., Hayward, S.J., Wania, F., 2013. Modeling the uptake of neutral organic chemicals on XAD passive air samplers under variable temperatures, external wind speeds and ambient air concentrations (PAS-SIM). Environmental science & technology 47, 13546–54. https://doi.org/10.1021/es402978a

Balasch, A., López, M., Reche, C., Viana, M., Moreno, T., Eljarrat, E., 2022. Exposure of e-waste dismantlers from a formal recycling facility in Spain to inhalable organophosphate and halogenated flame retardants. Chemosphere 294. https://doi.org/10.1016/j.chemosphere.2022.133775

Bečanová, J., Melymuk, L., Vojta, Š., Komprdová, K., Klánová, J., 2016. Screening for perfluoroalkyl acids in consumer products, building materials and wastes. Chemosphere 164, 322–329. https://doi.org/10.1016/j.chemosphere.2016.08.112

Bernhardt, E.S., Rosi, E.J., Gessner, M.O., 2017. Synthetic chemicals as agents of global change. Frontiers in Ecology and the Environment 15, 84–90. https://doi.org/10.1002/fee.1450

Bidleman, T.F., Melymuk, L., 2019. Forty-five Years of Foam: A Retrospective on Air Sampling with Polyurethane Foam. Bulletin of Environmental Contamination and Toxicology 102, 447–449. https://doi.org/10.1007/s00128-019-02591-4

Björklund, J.A., Thuresson, K., Cousins, A.P., Sellström, U., Emenius, G., de Wit, C.A., 2012. Indoor air is a significant source of tri-decabrominated diphenyl ethers to outdoor air via ventilation systems. Environmental science & technology 46, 5876–84. https://doi.org/10.1021/es204122v

Bogdal, C., Schmid, P., Kohler, M., Müller, C.E., Iozza, S., Bucheli, T.D., Scheringer, M., Hungerbühler, K., 2008. Sediment Record and Atmospheric Deposition of Brominated Flame Retardants and Organochlorine Compounds in Lake Thun, Switzerland: Lessons from the Past and Evaluation of the Present. Environmental Science & Technology 42, 6817–6822. https://doi.org/10.1021/es800964z

Bohlin, P., Audy, O., Škrdlíková, L., Kukučka, P., Vojta, Š., Přibylová, P., Prokeš, R., Čupr, P., Klánová, J., 2014. Evaluation and guidelines for using polyurethane foam (PUF) passive air samplers in double-dome chambers to assess semi-volatile organic compounds (SVOCs) in non-industrial indoor environments. Environmental Science: Processes & Impacts 16, 2617–2626. https://doi.org/10.1039/C4EM00305E

Bohlin-Nizzetto, P., Melymuk, L., White, K.B., Kalina, J., Madadi, V.O., Adu-Kumi, S., Prokeš, R., Přibylová, P., Klánová, J., 2020. Field- and model-based calibration of polyurethane foam passive air samplers in different climate regions highlights differences in sampler uptake performance. Atmospheric Environment 238. https://doi.org/10.1016/j.atmosenv.2020.117742

Bornehag, C.G., Lundgren, B., Weschler, C.J., Sigsgaard, T., Hagerhed-Engman, L., Sundell, J., 2005. Phthalates in indoor dust and their association with building characteristics. Environmental Health Perspectives 113, 1399–1404. https://doi.org/10.1289/ehp.7809

Breivik, K., Sweetman, A., Pacyna, J.M., Jones, K.C., 2002. Towards a global historical emission inventory for selected PCB congeners - A mass balance approach: 2. Emissions. Science of the Total Environment 290, 199–224. https://doi.org/10.1016/S0048-9697(01)01076-2

Budnik, L.T., Casteleyn, L., 2019. Mercury pollution in modern times and its socio-medical consequences. Science of The Total Environment 654, 720–734. https://doi.org/10.1016/j.scitotenv.2018.10.408

Caballero-Casero, N., Belova, L., Vervliet, P., Antignac, J.-P., Castaño, A., Debrauwer, L., López, M.E., Huber, C., Klanova, J., Krauss, M., Lommen, A., Mol, H.G.J., Oberacher, H., Pardo, O., Price, E.J., Reinstadler, V., Vitale, C.M., van Nuijs, A.L.N., Covaci, A., 2021. Towards harmonised criteria in quality assurance and quality control of suspect and non-target LC-HRMS analytical workflows for screening of emerging contaminants in human biomonitoring. TrAC Trends in Analytical Chemistry 136, 116201. https://doi.org/10.1016/j.trac.2021.116201

Chaemfa, C., Wild, E., Davison, B., Barber, J.L., Jones, K.C., 2009. A study of aerosol entrapment and the influence of wind speed, chamber design and foam density on polyurethane foam passive air samplers used for persistent organic pollutants. Journal of environmental monitoring 11, 1135–9. https://doi.org/10.1039/B823016A

Chin, J., Jiang, B., Mufidah, I., Persada, S., Noer, B., 2018. The Investigation of Consumers' Behavior Intention in Using Green Skincare Products: A Pro-Environmental Behavior Model Approach. Sustainability 10, 3922. https://doi.org/10.3390/su10113922

Coakley, J.D., Harrad, S.J., Goosey, E.R., Ali, N., Dirtu, A.C., Van den Eede, N., Covaci, A., Douwes, J., Mannetje, A. t., 2013. Concentrations of polybrominated diphenyl ethers in matched samples of indoor dust and breast milk in New Zealand. Environment International 59, 255–261. https://doi.org/10.1016/j.envint.2013.06.020

Crimmins, B.S., Pagano, J.J., Xia, X., Hopke, P.K., Milligan, M.S., Holsen, T.M., 2012. Polybrominated diphenyl ethers (PBDEs): Turning the corner in Great Lakes trout 1980-2009. Environmental Science and Technology 46, 9890–9897. https://doi.org/10.1021/es302415z

Darnerud, P.O., 2003. Toxic effects of brominated flame retardants in man and in wildlife. Environment international 29, 841–53. https://doi.org/10.1016/S0160-4120(03)00107-7

Demirtepe, H., Melymuk, L., Codling, G., Murínová, Ľ.P., Richterová, D., Rašplová, V., Trnovec, T., Klánová, J., 2021. Targeted and suspect screening of plasticizers in house dust to assess cumulative human exposure risk. Science of The Total Environment 781, 146667. https://doi.org/10.1016/j.scitotenv.2021.146667

Demirtepe, H., Melymuk, L., Diamond, M.L., Bajard, L., Vojta, Š., Prokeš, R., Sáňka, O., Klánová, J., Palkovičová Murínová, Ľ., Richterová, D., Rašplová, V., Trnovec, T., 2019. Linking past uses of legacy SVOCs with today's indoor levels and human exposure. Environment International 127, 653–663. https://doi.org/10.1016/j.envint.2019.04.001

Diamond, M.L., Okeme, J.O., Melymuk, L., 2021. Hands as Agents of Chemical Transport in the Indoor Environment. Environmental Science & Technology Letters 8, 326–332. https://doi.org/10.1021/acs.estlett.0c01006

Dronsfield, A., 2006. Percivall Pott, chimney sweeps and cancer. RSC Education. https://edu.rsc.org/feature/percivall-pott-chimney-sweeps-and-cancer/2020205.article

ECHA, 2024. Per- and polyfluoroalkyl substances (PFAS). https://echa.europa.eu/hot-topics/perfluoroalkyl-chemicals-pfas

Estill, C.F., Slone, J., Mayer, A., Chen, I.-C., La Guardia, M.J., 2020. Worker exposure to flame retardants in manufacturing, construction and service industries. Environment International 135, 105349. https://doi.org/10.1016/j.envint.2019.105349

Fenner, K., Scheringer, M., 2021. The Need for Chemical Simplification As a Logical Consequence of Ever-Increasing Chemical Pollution. Environmental Science and Technology 55, 14470–14472. https://doi.org/10.1021/acs.est.1c04903

Fromme, H., Lahrz, T., Kraft, M., Fembacher, L., Mach, C., Dietrich, S., Burkardt, R., Völkel, W., Göen, T., 2014. Organophosphate flame retardants and plasticizers in the air and dust in German daycare centers and human biomonitoring in visiting children (LUPE 3). Environment International 71, 158–163. https://doi.org/10.1016/j.envint.2014.06.016

Fruh, V., Preston, E.V., Quinn, M.R., Hacker, M.R., Wylie, B.J., O'Brien, K., Hauser, R., James-Todd, T., Mahalingaiah, S., 2022. Urinary phthalate metabolite concentrations and personal care product use during pregnancy – Results of a pilot study. Science of The Total Environment 835, 155439. https://doi.org/10.1016/j.scitotenv.2022.155439

Govarts, E., Gilles, L., Rodriguez Martin, L., Santonen, T., Apel, P., Alvito, P., Anastasi, E., Andersen, H.R., Andersson, A.-M., Andryskova, L., Antignac, J.-P., Appenzeller, B., Barbone, F., Barnett-Itzhaki, Z., Barouki, R., Berman, T., Bil, W., Borges, T., Buekers, J., Cañas-Portilla, A., Covaci, A., Csako, Z., Den Hond, E., Dvorakova, D., Fabelova, L., Fletcher, T., Frederiksen, H., Gabriel, C., Ganzleben, C., Göen, T., Halldorsson, T.I., Haug, L.S., Horvat, M., Huuskonen, P., Imboden, M., Jagodic Hudobivnik, M., Janasik, B., Janev Holcer, N., Karakitsios, S., Katsonouri, A., Klanova, J., Kokaraki, V., Kold Jensen, T., Koponen, J., Laeremans, M., Laguzzi, F., Lange, R., Lemke, N., Lignell, S., Lindroos, A.K., Lobo Vicente, J., Luijten, M., Makris, K.C., Mazej, D., Melymuk, L., Meslin, M., Mol, H., Montazeri, P., Murawski, A., Namorado, S., Niemann, L., Nübler, S., Nunes, B., Olafsdottir, K., Palkovicova Murinova, L., Papaioannou, N., Pedraza-Diaz, S., Piler, P., Plichta, V., Poteser, M., Probst-Hensch, N., Rambaud, L., Rauscher-Gabernig, E., Rausova, K., Remy, S., Riou, M., Rosolen, V., Rousselle, C., Rüther, M., Sarigiannis, D., Silva, M.J., Šlejkovec, Z., Snoj Tratnik, J., Stajnko, A., Szigeti, T., Tarazona, J.V., Thomsen, C., Tkalec, Ž., Tolonen, H., Trnovec, T., Uhl, M., Van Nieuwenhuyse, A., Vasco, E., Verheyen, V.J., Viegas, S., Vinggaard, A.M., Vogel, N., Vorkamp, K., Wasowicz, W., Weber, T., Wimmerova, S., Woutersen, M., Zimmermann, P., Zvonar, M., Koch, H., Kolossa-Gehring, M., Esteban López, M., Castaño, A., Stewart, L., Sepai, O., Schoeters, G., 2023. Harmonized human biomonitoring in European children, teenagers and adults: EU-wide exposure data of 11 chemical substance groups from the HBM4EU Aligned Studies (2014–2021). International Journal of Hygiene and Environmental Health 249, 114119. https://doi.org/10.1016/j.ijheh.2023.114119

Harley, K.G., Kogut, K., Madrigal, D.S., Cardenas, M., Vera, I.A., Meza-Alfaro, G., She, J., Gavin, Q., Zahedi, R., Bradman, A., Eskenazi, B., Parra, K.L., 2016. Reducing Phthalate, Paraben, and Phenol Exposure from Personal Care Products in Adolescent Girls: Findings from the HERMOSA Intervention Study. Environmental Health Perspectives 124, 1600–1607. https://doi.org/10.1289/ehp.1510514

Harner, T., Pozo, K., Gouin, T., Macdonald, A.-M., Hung, H., Cainey, J., Peters, A., 2006. Global pilot study for persistent organic pollutants (POPs) using PUF disk passive air samplers. Environmental Pollution 144, 445–452. https://doi.org/10.1016/j.envpol.2005.12.053

Harner, T., Su, K., Genualdi, S., Karpowicz, J., Ahrens, L., Mihele, C., Schuster, J.K., Charland, J.-P., Narayan, J., 2013. Calibration and application of PUF disk passive air samplers for tracking polycyclic aromatic compounds (PACs). Atmospheric Environment 75, 123–128. https://doi.org/10.1016/j.atmosenv.2013.04.012

Harrad, S.J., Abdallah, M.A.-E., Covaci, A., 2009. Causes of variability in concentrations and diastereomer patterns of hexabromocyclododecanes in indoor dust. Environment International 35, 573–579. https://doi.org/10.1016/j.envint.2008.10.005

Herrick, R.F., Meeker, J.D., Altshul, L., 2011. Serum PCB levels and congener profiles among teachers in PCB-containing schools: a pilot study. Environmental Health 10, 56. https://doi.org/10.1186/1476-069X-10-56

Hites, R.A., 2004. Polybrominated diphenyl ethers in the environment and in people: a meta-analysis of concentrations. Environmental science & technology 38, 945–956. https://doi.org/10.1021/es035082g

Hoek, G., Beelen, R., de Hoogh, K., Vienneau, D., Gulliver, J., Fischer, P., Briggs, D.J., 2008. A review of land-use regression models to assess spatial variation of outdoor air pollution. Atmospheric Environment 42, 7561–7578. https://doi.org/10.1016/j.atmosenv.2008.05.057

Hollender, J., Schymanski, E.L., Ahrens, L., Alygizakis, N., Béen, F., Bijlsma, L., Brunner, A.M., Celma, A., Fildier, A., Fu, Q., Gago-Ferrero, P., Gil-Solsona, R., Haglund, P., Hansen, M., Kaserzon, S., Kruve, A., Lamoree, M., Margoum, C., Meijer, J., Merel, S., Rauert, C., Rostkowski, P., Samanipour, S., Schulze, B., Schulze, T., Singh, R.R., Slobodnik, J., Steininger-Mairinger, T., Thomaidis, N.S., Togola, A., Vorkamp, K., Vulliet, E., Zhu, L., Krauss, M., 2023. NORMAN guidance on suspect and non-target screening in environmental monitoring. Environmental Sciences Europe 35, 75. https://doi.org/10.1186/s12302-023-00779-4

Holt, E., Audy, O., Booij, P., Melymuk, L., Prokes, R., Klánová, J., 2017. Organochlorine pesticides in the indoor air of a theatre and museum in the Czech Republic: Inhalation exposure and cancer risk. Science of The Total Environment 609, 598–606. https://doi.org/10.1016/j.scitotenv.2017.07.203

Hudson-Hanley, B., Irvin, V., Flay, B., MacDonald, M., Kile, M.L., 2018. Prenatal PBDE Exposure and Neurodevelopment in Children 7 Years Old or Younger: a Systematic Review and Meta-analysis. Current Epidemiology Reports 5, 46–59. https://doi.org/10.1007/s40471-018-0137-0

Jarvholm, B., 2006. Carcinogens in the Construction Industry. Annals of the New York Academy of Sciences 1076, 421–428. https://doi.org/10.1196/annals.1371.055

Jílková, S., Melymuk, L., Vojta, Š., Vykoukalová, M., Bohlin-Nizzetto, P., Klánová, J., 2018. Small-scale spatial variability of flame retardants in indoor dust and implications for dust sampling. Chemosphere 206, 132–141. https://doi.org/10.1016/J.CHEMOSPHERE.2018.04.146

Johnson-Restrepo, B., Kannan, K., 2009. An assessment of sources and pathways of human exposure to polybrominated diphenyl ethers in the United States. Chemosphere 76, 542–8. https://doi.org/10.1016/j.chemosphere.2009.02.068

Julander, A., Lundgren, L., Skare, L., Grandér, M., Palm, B., Vahter, M., Lidén, C., 2014. Formal recycling of e-waste leads to increased exposure to toxic metals: An occupational exposure study from Sweden. Environment International 73, 243–251. https://doi.org/10.1016/j.envint.2014.07.006

Kalina, J., Scheringer, M., Borůvková, J., Kukucka, P., Přibylová, P., Sáňka, O., Melymuk, L., Vana, M., Klánová, J., 2018. Characterizing spatial diversity of passive sampling sites for measuring levels and trends of semivolatile organic chemicals. Environmental Science & Technology 52, 10599–10608. https://doi.org/10.1021/acs.est.8b03414

Karásková, P., Codling, G.P., Melymuk, L., Klánová, J., 2018. A critical assessment of passive air samplers for per- and polyfluoroalkyl substances. Atmospheric Environment 185, 186–195. https://doi.org/10.1016/j.atmosenv.2018.05.030

Karásková, P., Venier, M., Melymuk, L., Bečanová, J., Vojta, Š., Prokeš, R., Diamond, M.L., Klánová, J., 2016. Perfluorinated alkyl substances (PFASs) in household dust in Central Europe and North America. Environment international 94, 315–324. https://doi.org/10.1016/j.envint.2016.05.031

Krause, M., Andersson, A.-M., Skakkebaek, N.E., Frederiksen, H., 2017. Exposure to UV filters during summer and winter in Danish kindergarten children. Environment International 99, 177–184. https://doi.org/10.1016/j.envint.2016.11.011

Lioy, P.J., Freeman, N.C.G., Millette, J.R., 2002. Dust: A Metric for Use in Residential and Building Exposure Assessment and Source Characterization. Environmental Health Perspectives 110, 969–983. https://doi.org/10.1289/ehp.02110969

Lucattini, L., Poma, G., Covaci, A., de Boer, J., Lamoree, M.H., Leonards, P.E.G., 2018. A review of semi-volatile organic compounds (SVOCs) in the indoor environment: occurrence in consumer products, indoor air and dust. Chemosphere 201, 466–482. https://doi.org/10.1016/j.chemosphere.2018.02.161

Magulova, K., Priceputu, A., 2016. Global monitoring plan for persistent organic pollutants (POPs) under the Stockholm Convention: Triggering, streamlining and catalyzing global POPs monitoring. Environmental Pollution 217, 82–84. https://doi.org/10.1016/j.envpol.2016.01.022

Matz, C., Stieb, D., Davis, K., Egyed, M., Rose, A., Chou, B., Brion, O., 2014. Effects of Age, Season, Gender and Urban-Rural Status on Time-Activity: Canadian Human Activity Pattern

Survey 2 (CHAPS 2). International Journal of Environmental Research and Public Health 11, 2108–2124. https://doi.org/10.3390/ijerph110202108

Melymuk, L., Blumenthal, J., Sáňka, O., Shu-Yin, A., Singla, V., Sebková, K., Fedinick, K.P., Diamond, M.L., 2022. Persistent Problem: Global Challenges to Managing PCBs. Environ. Sci. Technol 2022, 9040. https://doi.org/10.1021/acs.est.2c01204

Melymuk, L., Bohlin-Nizzetto, P., Kukučka, P., Vojta, Š., Kalina, J., Cupr, P., Klánová, J., Čupr, P., Klánová, J., 2016. Seasonality and indoor/outdoor relationships of flame retardants and PCBs in residential air. Environmental Pollution 218, 392–401. https://doi.org/10.1016/j.envpol.2016.07.018

Melymuk, L., Bohlin-Nizzetto, P., Prokeš, R., Kukučka, P., Přibylová, P., Vojta, Š., Kohoutek, J., Lammel, G., Klánová, J., 2017. Uncertainties in monitoring of SVOCs in air caused by within-sampler degradation during active and passive air sampling. Atmospheric Environment 167, 553–565. https://doi.org/10.1016/j.atmosenv.2017.08.038

Melymuk, L., Bohlin-Nizzetto, P., Sáňka, O., Pozo, K., Klánová, J., 2014. Current challenges in air sampling of semivolatile organic contaminants: Sampling artifacts and their influence on data comparability. Environmental science & technology 48, 14077–14091. https://doi.org/10.1021/es502164r

Melymuk, L., Demirtepe, H., Jílková, S.R., 2020. Indoor dust and associated chemical exposures. Current Opinion in Environmental Science & Health 15, 1–6. https://doi.org/10.1016/j.coesh.2020.01.005

Melymuk, L., Diamond, M.L., Riddell, N., Wan, Y., Vojta, Š., Chittim, B., 2018. Challenges in the analysis of novel flame retardants in indoor dust: results of the INTERFLAB 2 interlaboratory evaluation. Environmental Science & Technology 52, 9295–9303. https://doi.org/10.1021/acs.est.8b02715

Melymuk, L., Nizzetto, P.B., Harner, T., White, K.B., Wang, X., Tominaga, M.Y., He, J., Li, J., Ma, J., Ma, W.L., Aristizábal, B.H., Dryer, A., Jiménez, B., Muñoz-Arnanz, J., Odabasi, M., Dumanoglu, Y., Yaman, B., Graf, C., Sweetman, A., Klánová, J., 2021. Global intercomparison of polyurethane foam passive air samplers evaluating sources of variability in SVOC measurements. Environmental Science and Policy 125, 1–9. https://doi.org/10.1016/j.envsci.2021.08.003

Melymuk, L., Robson, M., Helm, P.A., Diamond, M.L., 2013. Application of Land Use Regression to Identify Sources and Assess Spatial Variation in Urban SVOC Concentrations. Environmental Science & Technology 47, 1887–1895. https://doi.org/10.1021/es3043609

Melymuk, L., Robson, M., Helm, P.A., Diamond, M.L., 2012. PCBs, PBDEs, and PAHs in Toronto air: spatial and seasonal trends and implications for contaminant transport.

Science of the Total Environment 429, 272–80. https://doi.org/10.1016/j.scitotenv.2012.04.022

Melymuk, L., Robson, M., Helm, P.A., Diamond, M.L., 2011. Evaluation of passive air sampler calibrations: Selection of sampling rates and implications for the measurement of persistent organic pollutants in air. Atmospheric Environment 45, 1867–1875. https://doi.org/10.1016/j.atmosenv.2011.01.011

Mendelsohn, E., Hagopian, A., Hoffman, K., Butt, C.M., Lorenzo, A., Congleton, J., Webster, T.F., Stapleton, H.M., 2016. Nail polish as a source of exposure to triphenyl phosphate. Environment International 86, 45–51. https://doi.org/10.1016/j.envint.2015.10.005

Mikeš, O., Sáňka, O., Rafajová, A., Vlaanderen, J., Chen, J., Hoek, G., Klánová, J., Čupr, P., 2023. Development of historic monthly land use regression models of SO2, NOx and suspended particulate matter for birth cohort ELSPAC. Atmospheric Environment 301, 119688. https://doi.org/10.1016/j.atmosenv.2023.119688

Moya, J., Phillips, L., 2014. A review of soil and dust ingestion studies for children. Journal of exposure science & environmental epidemiology 24, 545–554. https://doi.org/10.1038/jes.2014.17

Muenhor, D., Harrad, S.J., 2012. Within-room and within-building temporal and spatial variations in concentrations of polybrominated diphenyl ethers (PBDEs) in indoor dust. Environment international 47, 23–7. https://doi.org/10.1016/j.envint.2012.06.001

Nakata, H., Hinosaka, M., Yanagimoto, H., 2015. Macrocyclic-, polycyclic-, and nitro musks in cosmetics, household commodities and indoor dusts collected from Japan: Implications for their human exposure. Ecotoxicology and Environmental Safety 111, 248–255. https://doi.org/10.1016/j.ecoenv.2014.09.032

Nassan, F.L., Coull, B.A., Gaskins, A.J., Williams, M.A., Skakkebaek, N.E., Ford, J.B., Ye, X., Calafat, A.M., Braun, J.M., Hauser, R., 2017. Personal care product use in men and urinary concentrations of select phthalate metabolites and parabens: Results from the environment and reproductive health (EARTH) study. Environmental Health Perspectives 125, 1–10. https://doi.org/10.1289/EHP1374

Needleman, H., 2009. Low Level Lead Exposure: History and Discovery. Annals of Epidemiology 19, 235–238. https://doi.org/10.1016/j.annepidem.2009.01.022

Nguyen, L.V., Diamond, M.L., Venier, M., Stubbings, W.A., Romanak, K., Bajard, L., Melymuk, L., Jantunen, L.M., Arrandale, V.H., 2019. Exposure of Canadian electronic waste dismantlers to flame retardants. Environment International 129, 95–104. https://doi.org/10.1016/j.envint.2019.04.056

Özkaynak, H., Cohen, J., Hubbard, H., Thomas, K., Phillips, L., Tulve, N., 2023. Advancing Methodologies Used in Trace Element-Based Mass Balance Studies to Separately Estimate Soil and Dust Ingestion Rates for Children. Environment International 178, 107983. https://doi.org/10.1016/j.envint.2023.107983

Persson, L., Carney Almroth, B.M., Collins, C.D., Cornell, S., de Wit, C.A., Diamond, M.L., Fantke, P., Hassellöv, M., MacLeod, M., Ryberg, M.W., Søgaard Jørgensen, P., Villarrubia-Gómez, P., Wang, Z., Hauschild, M.Z., 2022. Outside the Safe Operating Space of the Planetary Boundary for Novel Entities. Environmental Science & Technology 56, 1510–1521. https://doi.org/10.1021/acs.est.1c04158

Pozo, K., Harner, T., Wania, F., Muir, D.C.G., Jones, K.C., Barrie, L.A., 2006. Toward a global network for persistent organic pollutants in air: results from the GAPS study. Environmental science & technology 40, 4867–73. https://doi.org/10.1021/es060447t

Rigét, F., Bignert, A., Braune, B., Dam, M., Dietz, R., Evans, M., Green, N., Gunnlaugsdóttir, H., Hoydal, K.S., Kucklick, J., Letcher, R., Muir, D., Schuur, S., Sonne, C., Stern, G., Tomy, G., Vorkamp, K., Wilson, S., 2019. Temporal trends of persistent organic pollutants in Arctic marine and freshwater biota. Science of The Total Environment 649, 99–110. https://doi.org/10.1016/j.scitotenv.2018.08.268

Rostkowski, P., Haglund, P., Aalizadeh, R., Alygizakis, N., Thomaidis, N., Arandes, J.B., Nizzetto, P.B., Booij, P., Budzinski, H., Brunswick, P., Covaci, A., Gallampois, C., Grosse, S., Hindle, R., Ipolyi, I., Jobst, K., Kaserzon, S.L., Leonards, P., Lestremau, F., Letzel, T., Magnér, J., Matsukami, H., Moschet, C., Oswald, P., Plassmann, M., Slobodnik, J., Yang, C., 2019. The strength in numbers: comprehensive characterization of house dust using complementary mass spectrometric techniques. Analytical and Bioanalytical Chemistry 411, 1957–1977. https://doi.org/10.1007/s00216-019-01615-6

Saini, A., Harner, T., Chinnadhurai, S., Schuster, J.K., Yates, A., Sweetman, A., Aristizabal-Zuluaga, B.H., Jiménez, B., Manzano, C.A., Gaga, E.O., Stevenson, G., Falandysz, J., Ma, J., Miglioranza, K.S.B., Kannan, K., Tominaga, M., Jariyasopit, N., Rojas, N.Y., Amador-Muñoz, O., Sinha, R., Alani, R., Suresh, R., Nishino, T., Shoeib, T., 2020. GAPS-megacities: A new global platform for investigating persistent organic pollutants and chemicals of emerging concern in urban air. Environmental Pollution 267, 115416. https://doi.org/10.1016/j.envpol.2020.115416

Salthammer, T., Zhang, Y., Mo, J., Koch, H.M., Weschler, C.J., 2018. Assessing Human Exposure to Organic Pollutants in the Indoor Environment. Angewandte Chemie - International Edition 57. https://doi.org/10.1002/anie.201711023

Schuster, J.K., Gioia, R., Harner, T., Lee, S.C., Breivik, K., Jones, K.C., 2012. Assessment of sorbent impregnated PUF disks (SIPs) for long-term sampling of legacy POPs. Journal of environmental monitoring 14, 71–8. https://doi.org/10.1039/C1EM10697J

Schweizer, C., Edwards, R.D., Bayer-Oglesby, L., Gauderman, W.J., Ilacqua, V., Jantunen, M.J., Lai, H.K., Nieuwenhuijsen, M.J., Künzli, N., 2007. Indoor time-microenvironment-activity patterns in seven regions of Europe. Journal of exposure science & environmental epidemiology 17, 170–81.

Shoeib, M., Harner, T., 2002. Characterization and comparison of three passive air samplers for persistent organic pollutants. Environmental science & technology 36, 4142–51. https://doi.org/10.1021/es020635t

Stapleton, H.M., Sharma, S., Getzinger, G., Ferguson, P.L., Gabriel, M., Webster, T.F., Blum, A., 2012. Novel and high volume use flame retardants in US couches reflective of the 2005 PentaBDE phase out. Environmental Science & Technology 46, 13432–13439. https://doi.org/10.1021/es303471d

Steffen, W., Richardson, K., Rockström, J., Cornell, S., Fetzer, I., Bennett, E., Biggs, R., Carpenter, S.R., de Wit, C.A., Folke, C., Mace, G., Persson, L.M., Veerabhadran, R., Reyers, B., Sörlin, S., 2015. Planetary Boundaries: Guiding human development on a changing planet. Science 347. https://doi.org/10.1126/science.1259855

Stuchlík Fišerová, P., Melymuk, L., Komprdová, K., Domínguez-Romero, E., Scheringer, M., Kohoutek, J., Přibylová, P., Andrýsková, L., Piler, P., Koch, H.M., Zvonař, M., Esteban-López, M., Castaño, A., Klánová, J., 2022. Personal care product use and lifestyle affect phthalate and DINCH metabolite levels in teenagers and young adults. Environmental Research 213, 113675. https://doi.org/10.1016/j.envres.2022.113675

Sugeng, E.J., de Cock, M., Leonards, P.E.G., van de Bor, M., 2018. Electronics, interior decoration and cleaning patterns affect flame retardant levels in the dust from Dutch residences. Science of the Total Environment 645, 1144–1152. https://doi.org/10.1016/j.scitotenv.2018.07.127

UNEP, 2017. Stockholm Convention on Persistent Organic Pollutants (POPs). https://www.pops.int/

van der Schyff, V., Kalina, J., Abballe, A., Iamiceli, A.L., Govarts, E., Melymuk, L., 2023. Has Regulatory Action Reduced Human Exposure to Flame Retardants? Environ. Sci. Technol. 57, 19106–19124. https://doi.org/10.1021/acs.est.3c02896

van der Schyff, V., Suchánková, L., Kademoglou, K., Melymuk, L., Klánová, J., 2022. Parabens and antimicrobial compounds in conventional and “green” personal care products. Chemosphere 297, 1–8. https://doi.org/10.1016/j.chemosphere.2022.134019 van der Veen, I., de Boer, J., 2012. Phosphorus flame retardants: properties, production, environmental occurrence, toxicity and analysis. Chemosphere 88, 1119–53. https://doi.org/10.1016/j.chemosphere.2012.03.067

Venier, M., Audy, O., Vojta, Š., Bečanová, J., Romanak, K., Melymuk, L., Krátká, M., Kukučka, P., Okeme, J.O., Saini, A., Diamond, M.L., Klánová, J., 2016. Brominated flame retardants in indoor environment - comparative study of indoor contamination from three countries. Environment international 94, 150–160. https://doi.org/10.1016/j.envint.2016.04.029

Venier, M., Hites, R.A., 2010. Time trend analysis of atmospheric POPs concentrations in the Great Lakes region since 1990. Environmental science & technology 44, 8050–5. https://doi.org/10.1021/es101656u

Vojta, Š., Bečanová, J., Melymuk, L., Komprdová, K., Kohoutek, J., Kukučka, P., Klánová, J., 2017a. Screening for halogenated flame retardants in European consumer products, building materials and wastes. Chemosphere 168, 457–466. https://doi.org/10.1016/j.chemosphere.2016.11.032

Vojta, Š., Melymuk, L., Klánová, J., 2017b. Changes in Flame Retardant and Legacy Contaminant Concentrations in Indoor Air during Building Construction, Furnishing, and Use. Environmental Science & Technology 51, 11891–11899. https://doi.org/10.1021/acs.est.7b03245

Vojta, Š., Melymuk, L., Prokeš, R., Klánová, J., 2024. Assessment of the impact of sampler housing on indoor passive air sampler measurements of SVOCs. Journal of Environmental Exposure Assessment 3, 6. https://doi.org/10.20517/jeea.2023.39

Vorkamp, K., Castaño, A., Antignac, J.-P., Boada, L.D., Cequier, E., Covaci, A., Esteban López, M., Haug, L.S., Kasper-Sonnenberg, M., Koch, H.M., Pérez Luzardo, O., Osīte, A., Rambaud, L., Pinorini, M.-T., Sabbioni, G., Thomsen, C., 2021. Biomarkers, matrices and analytical methods targeting human exposure to chemicals selected for a European human biomonitoring initiative. Environment International 146, 106082. https://doi.org/10.1016/j.envint.2020.106082

Vykoukalová, M., Venier, M., Vojta, Š., Melymuk, L., Bečanová, J., Romanak, K., Prokeš, R., Okeme, J.O., Saini, A., Diamond, M.L., Klánová, J., 2017. Organophosphate esters flame retardants in the indoor environment. Environment International 106, 97–104. https://doi.org/10.1016/j.envint.2017.05.020

Wania, F., Shunthirasingham, C., 2020. Passive air sampling for semi-volatile organic chemicals. Environmental Science: Processes & Impacts 22, 1925–2002. https://doi.org/10.1039/D0EM00194E

Weschler, C.J., Nazaroff, W.W., 2012. SVOC exposure indoors: fresh look at dermal pathways. Indoor air 22, 356–377.

Weschler, C.J., Nazaroff, W.W., 2010. SVOC partitioning between the gas phase and settled dust indoors. Atmospheric Environment 44, 3609–3620.

Weschler, C.J., Nazaroff, W.W., 2008. Semivolatile organic compounds in indoor environments. Atmospheric Environment 42, 9018–9040.

White, K.B., Kalina, J., Scheringer, M., Přibylová, P., Kukučka, P., Kohoutek, J., Prokeš, R., Klánová, J., 2023. Spatial and Temporal Trends of Persistent Organic Pollutants across Europe after 15 Years of MONET Passive Air Sampling. Environ. Sci. Technol. 57, 11583–11594. https://doi.org/10.1021/acs.est.3c00796

White, K.B., Sáňka, O., Melymuk, L., Přibylová, P., Klánová, J., 2021. Application of land use regression modelling to describe atmospheric levels of semivolatile organic compounds on a national scale. Science of the Total Environment 793. https://doi.org/10.1016/j.scitotenv.2021.148520

Wingfors, H., Seldén, A.I., Nilsson, C., Haglund, P., 2006. Identification of markers for PCB exposure in plasma from Swedish construction workers removing old elastic sealants. The Annals of occupational hygiene 50, 65–73. https://doi.org/10.1093/annhyg/mei063

Wong, F., Hung, H., Dryfhout-Clark, H., Aas, W., Bohlin-Nizzetto, P., Breivik, K., Mastromonaco, M.N., Lundén, E.B., Ólafsdóttir, K., Sigurðsson, Á., Vorkamp, K., Bossi, R., Skov, H., Hakola, H., Barresi, E., Sverko, E., Fellin, P., Li, H., Vlasenko, A., Zapevalov, M., Samsonov, D., Wilson, S., 2021. Time trends of persistent organic pollutants (POPs) and Chemicals of Emerging Arctic Concern (CEAC) in Arctic air from 25 years of monitoring. Science of the Total Environment 775. https://doi.org/10.1016/j.scitotenv.2021.145109

Yang, C., Jílková, S.R., Melymuk, L., Harris, S.A., Jantunen, L.M., Pertili, J., Winn, L., Diamond, M.L., 2020. Are We Exposed to Halogenated Flame Retardants from both Primary and Secondary Sources? Environmental Science & Technology Letters 7, 585–593. https://doi.org/10.1021/acs.estlett.0c00268

APPENDIX 1

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Has Regulatory Action Reduced Human Exposure to Flame Retardants?

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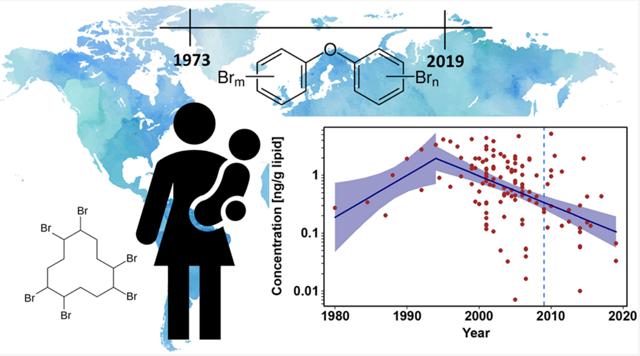

ABSTRACT: Flame retardant (FR) exposure has been linked to several environmental and human health effects. Because of this, the production and use of several FRs are regulated globally. We reviewed the available records of polybrominated diphenyl ethers (PBDEs) and hexabromocyclododecanes (HBCDDs) in human breast milk from literature to evaluate the efficacy of regulation to reduce the exposure of FRs to humans. Two-hundred and seven studies were used for analyses to determine the spatial and temporal trends of FR exposure. North America consistently had the highest concentrations of PBDEs, while Asia and Oceania dominated HBCDD exposure. BDE-49 and -99 indicated decreasing temporal trends in most regions. BDE-153, with a longer half-life than the aforementioned isomers, typically exhibited a plateau in breast milk levels. No conclusive trend could be established for HBCDD, and insufficient information was available to determine a temporal trend for BDE-209. Breakpoint analyses indicated a significant decrease in BDE-47 and -99 in Europe around the time that regulation has been implemented, suggesting a positive effect of regulation on FR exposure. However, very few studies have been conducted globally (specifically in North America) after 2013, during the time when the most recent regulations have been implemented. This meta-analysis provides insight into global trends in human exposure to PBDEs and HBCDD, but the remaining uncertainty highlights the need for ongoing evaluation and monitoring, even after a compound group is regulated.

KEYWORDS: *flame retardant, polybrominated diphenyl ether, hexabromocyclododecane, breast milk, biomonitoring, temporal trends, effectiveness evaluation*

INTRODUCTION

Flame retardants are added to a wide range of consumer products and materials in order to reduce ignition or flammability of a material or fulfill fire safety requirements.[1] However, past efforts to reduce flammability through the addition of synthetic organic flame retardants have led to negative impacts on human and environmental health due to exposure to harmful chemicals.[2]

Polybrominated diphenyl ethers (PBDEs) and hexabromocyclododecanes (HBCDDs) were among the dominant FRs used for decades.[3,4] The three technical mixtures of PBDEs (penta-, octa-, and deca-BDE) had multiple uses, including polyurethane foam, electrical and electronic equipment, building materials, and vehicle parts.[5] Technical HBCDD (a mixture of the stereoisomers, α-, β-, and γ-HBCDD, with γ-HBCDD being the most abundant) was primarily used in electronics, textiles, and especially in expanded (EPS) and extruded polystyrene (XPS) applied as construction and packing materials.[6,7]

PBDEs and HBCDDs are known to be persistent, bioaccumulative, and subject to long-range transport in the environment and are ubiquitous across environmental systems.[8,9] PBDEs have been reported in human blood,[10−12] adipose tissues,[13−15] and milk[16−19] since the early 1990s, and evidence of human exposure to HBCDDs arose shortly thereafter.[20−23]

PBDE exposure has been associated with numerous adverse health outcomes, including alterations to thyroid function, reproductive systems, and breast cancer,[24−26] with strong evidence for neurodevelopmental impacts, including lower IQ and ADHD.[27,28] Elevated levels of PBDEs in breast milk have specifically been associated with neurodevelopmental effects and alterations to the gut microbiome in young children.[29,30] Similar adverse effects are associated with elevated HBCDD

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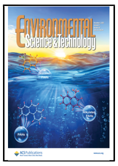

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Table 1. Countries Used in This Study Grouped According to the United Nations Geoscheme

region	countries included			PBDEs	HBCDDs
Africa	Congo	South Africa	South Africa	7 studies[79−85]	5 studies[79−83]
	Cote d'Ivoire	Mauritius	Tanzania		
	Djibouti	Morocco	Togo		
	Egypt	Niger	Tunisia		
	Ethiopia	Nigeria	Uganda		
	Ghana	Senegal	Zambia		
	Kenya				
Asia	China	Japan	Syria	53 studies[18,81,86−136]	15 studies[81,89,95,132−135,137−144]
	Georgia	Macao	Taiwan		
	India	Philippines	Tajikistan		
	Indonesia	Russia[a]	Vietnam		
	Israel	South Korea			
Central and South America and Caribbean	Antigua and Barbuda	Chile	Peru	3 studies[73,81,145]	2 studies[73,81]
	Barbados	Haiti	Suriname		
	Brazil	Jamaica	Uruguay		
Europe	Belgium	Greece	Romania	61 studies[1,19,29,30,59,81,146−200]	25 studies[23,59,76,81,146−151,153−159,167,185,189,193,201−204]
	Bulgaria	Hungary	Russia[a]		
	Croatia	Ireland	Slovakia		
	Cyprus	Italy	Spain		
	Czechia	Lithuania	Sweden		
	Denmark	Luxembourg	Switzerland		
	Faroe Islands	Moldova	Turkey		
	Finland	Netherlands	UK		
	France	Norway	Ukraine		
	Germany	Poland			
North America	Canada	Mexico	USA	25 studies[17,51,70,77,81,205−222]	5 studies[51,77,78,81,83]
Oceania	Australia	Kiribati	Tonga	7 studies[223−228]	2 studies[81,226]
	Fiji	New Zealand	Tuvalu		

[a] General samples from Russia were included within the European category. When a study specified a geographic region that was within the Asian part of Russia, it was included in Asia.

concentrations, including endocrine disruption, specifically thyroid, neurobehavioral, and developmental disorders.[26,31]

In response to concerns regarding the environmental and human health impacts of certain FRs, actions were taken to reduce production.[32] In 2004, the European Union stated that "in order to protect health and the environment the placing on the market and the use of pentaBDE and octaBDE and the placing on the market of articles containing one or both of these substances should be prohibited",[33] and in the same year these mixtures were voluntarily withdrawn from the U.S. marketplace by their manufacturers.[34] The lower brominated PBDE congeners, tetra- and penta-BDE (main components of commercial penta-BDE[4][35]) and hexa- and hepta-BDEs (main components of commercial octa-BDE[4])[36] were listed in the Stockholm Convention on Persistent Organic Pollutants (POPs) in 2009, requiring parties to eliminate the production and use of the compounds. Deca-BDE, the fully brominated PBDE molecule and main component of the decaBDE commercial product,[37] was similarly listed in 2017.[38] In 2008, HBCDDs were recognized as substances of very high concern (SVHC) in the EU due to environmental and human health risks[39,40] and were added to the Stockholm Convention in 2013.[41] In specific cases, individual countries were permitted continued production of HBCDDs until 2024.[7]

The evaluation of temporal patterns of a chemical's concentration in a predetermined medium is an effective tool to determine the efficiency of policy in mitigating chemical exposure.[42] Recent studies have identified declines in components of the penta- and octa-BDE technical mixtures,[9] in air (1993−2018),[43] soil (1998−2008),[44] sediment (2002−2012),[45] sewage sludge (2004−2010),[46] and fish (1980−2009),[47] while BDE-209 has been stable or increasing in many matrices.[46,48,49] In humans, declines in levels of less brominated PBDEs and a more recent plateau have been identified in several countries, however, this is not uniform across regions or matrices:[9,22,50,51] there is a lack of understanding of how generalizable these regional trends are. For HBCDDs, there is even less evidence of a global trend. Although time trends of HBCDD concentrations in multiple environmental matrices[52−54] have been determined, analysis of temporal patterns of HBCDDs has been limited to regional scales. No consensus or clear global time trend of HBCDD concentrations in humans has been identified,[9] with some studies reporting an increase of HBCDDs,[54−56] others a decrease[57] or no trend.[58,59]

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Studying the effectiveness of policies concerning FRs through time trend analysis of biomonitoring data presents inherent complexities. Different FRs, each with distinct chemical structures, are incorporated into diverse applications such as furniture, electronics, and building insulation, which are associated with different emission and exposure routes.[4] After restriction, continued presence of FR-containing products further complicates this, as different product types have very different replacement rates (e.g., smartphones, 2−6 years[60] vs building insulation, 30−50 yrs[61]). The environmental persistence of compounds is generally longer indoors[62,63] and indoor levels are sustained until active removal of sources.[64,65] Exposure to FRs is further influenced by regional factors like building and cleaning practices and dietary patterns.[66] Moreover, the differing persistence of FRs within the body complicates our understanding of exposure,[67] with some FRs possessing longer half-lives in human tissues[68] and partitioning within body tissues varying by compound/congener.[69,70] Concentrations of POPs in human tissues typically reflect long-term exposures, for example, variations of PCB concentrations in breast milk levels can be explained by differences in early life exposures rather than current dietary exposures.[71] Despite the above-mentioned complexities, our insight into the effectiveness of restrictions and the remaining risks to human populations from legacy FRs can be improved by multistudy analyses to understand and interpret the global time patterns of PBDEs and HBCDDs in humans.

In this analysis, we first review available records of PBDE and HBCDD in human milk to determine time patterns of global human exposure to FRs. Second, we evaluate the impact of regulations that were introduced over the past 20 years on exposure to these legacy FRs. Finally, we investigate the regional differences in exposure to FRs in relation to use. We supplement this with a review of past studies evaluating temporal patterns of PBDEs and HBCDDs in human matrices, to provide a comprehensive review of trends in global exposure.

■ METHODS

Rationalization of Study Matrix. It is impossible to select a single biological matrix that encompasses the global population, as well as all target compounds. Due to the lipophilic nature of PBDEs and HBCDDs, they are best evaluated through matrices such as blood serum or breast/maternal milk.[72] Breast milk has a high lipid content and can be collected noninvasively, making it a reliable and accessible matrix for assessing body burdens of PBDEs and HBCDDs, as well as many other POPs. The Stockholm Convention, in cooperation with the World Health Organization, has identified human milk as a core matrix of its Global Monitoring Plan and supported with routine quantification in pooled milk samples.[70,73]

In addition to its importance in routine monitoring programs, maternal milk is an ideal matrix for meta-analyses of biomonitoring data because of the relative homogeneity of the study population: all female, with an age range generally spanning 18−45 years. Moreover, breast milk is not only an indicator of human exposure but also represents a direct exposure route to infants, and breast milk is typically the most important exposure pathway of young children to POPs, including PBDEs.[74,75]

Search Strategy and Selection Criteria−Human Milk Meta-Analysis. A literature search of peer-reviewed studies and reports produced by regulatory bodies (e.g., UNEP, German Federal Environment Agency) was conducted using ISI Web of Science and Google Scholar. The search was not limited by years or language of publication. The search was initially conducted in March 2020 and updated in September 2022. The following search terms were used.

For HBCDDs. All fields: [hexabromocyclododecane* OR HBCD*] AND [(human milk) or (breast milk)]. This search produced 168 results, which were evaluated for their appropriateness. The criteria for inclusion were that the studies provided lipid-standardized HBCDD levels in human milk (either isomer-specific HBCDD or $\sum$HBCDD) and included basic information on the study population (country of residence, sampling year). Of the initial 168 studies identified, 49 met the criteria and were used for further analysis (Figure S1, data sources listed in Table 1).

For PBDEs. All fields: [polybrominated diphenyl ether* OR PBDE*] AND [(human milk) or (breast milk)]. This search identified 1204 results, which were then evaluated for their appropriateness. Only studies reporting individual PBDE congeners were included, and four congeners were selected as indicators due to their prevalence in literature: BDE-47, BDE-99, BDE-153, and BDE-209. Studies also had to include lipid-standardized concentrations for at least one of these congeners and include information on the study population (country of residence, sampling year). Of the initial 1204 studies, 158 met the criteria and were used for further analysis (Figure S1, data sources listed in Table 1).

All available data (either primary data reporting individual concentrations or all summary statistics) were extracted from the articles to a spreadsheet database. Data reported from pooled samples were treated as mean values.

Data Set Standardization. Statistical evaluation was carried out by R software (version R 4.1.2). The data sources were separated by compound, sampling location, and/or date of sample collection. These records were aggregated by country and date of sampling, characterized either by mean or median value, minimum, 5th, 10th, 25th, 75th, 90th, and 95th quantile and maximum or any combination of these descriptive statistics. If all primary data/summary statistics within an aggregate record were taken during one year, the aggregated record was assigned to that year. In the other cases, the aggregated record was assigned to the middle point between the years of the oldest and the newest primary data/summary statistic.

For HBCDDs, 41 aggregated records included both α-HBCDD and the sum of α, β, and γ-isomers. These 41 records were used for estimating the contribution of α-HBCDD to the sums. This contribution was 94.0%, showing clearly that α-HBCDD dominates over the β and γ isomers. This 94.0% was then used to extrapolate α-HBCDD from records where only $\sum$HBCDDs were reported for the original data set, resulting in 260 aggregated records for α-HBCDD.

Primary FR data from five locations[76−78] were used to establish that α-HBCDD and PBDE concentrations in breast milk have a general log-normal distribution of primary data within an aggregate data record. On the basis of the log-normal distribution assumption, the maximum likelihood estimation was used to apply a log-normal distribution for each aggregated sample and thus estimate its median value in cases where only other summary statistics were reported. In a few cases, the maximum likelihood estimate was not directly applicable since the only descriptive statistic characterizing the aggregated sample was an arithmetic mean. In such cases, a median standard deviation based on the rest of the samples was used (0.62 ng/g lipid weight; lw for α-HBCDD, 0.32 ng/g lw for BDE-47, 0.23 ng/g lw for BDE-99, 0.35 ng/g lw for BDE 153, and 50.15 ng/g lw for BDE-209) to derive quantiles describing the expected log-normal distribution. Finally, since there are no theoretical minimal and maximal values for the log-normal statistical distribution, the min and max values were considered as $(1/n)$th and $(1-1/n)$th quantiles for aggregated samples with known n; for aggregated samples with an unknown number of primary samples, $n = 40$ was used as it was the median value of studies where n was specified (Figure S2).

Temporal Pattern Meta-Analysis. Data were grouped regionally following the United Nations geoscheme (Table 1). With the use of the aggregate data by region and sampling year, a weighted Theil-Sen trend analysis was conducted, assigning each aggregated sample a weight in the range of 0.1 to 1.0 for 10 to 100 primary samples and 1.0 for more than 100 primary samples. A weighted Mann-Kendall test was then used for assessing the trends' significance.

Breakpoint Analysis. An additional type of temporal pattern analysis was applied to European and Asian data sets, as these continents had the most complete records for both PBDEs and HBCDDs and are of particular interest given the early introduction of FR regulations. To evaluate whether this early introduction of restrictions led to a change in the FR time patterns over time, breakpoint analysis was applied to find a time point when the slope of the trend breaks,[229] i.e., suggesting a change in the rate of change of a given FR concentration in human milk. The breakpoint analysis identifies the breakpoint by aiming for normally distributed residuals of both linear trends before and after the breakpoint, which indicates an optimal fit. The method searches for all possible breakpoints (in this case in increments of whole years only) and selects the breakpoint with the smallest sum of squares of residuals. Only significant results according to the difference of halving times before and after the breakpoint are considered.[229]

Analysis of Geographic Patterns. Additional comparative statistical analyses were conducted using Graphpad Prism 8.0.2 using all studies after the year 2000. Data were grouped according to geographic region (Table 1). The concentrations from the different regions were compared using Kruskall-Wallis nonparametric ANOVA tests, and individual regions were compared with all others using Dunn's multiple comparison tests. For geographic patterns, significance was set at $p < 0.05$.

Limitations. The quality of the analytical work performed in individual studies was not evaluated, in favor of allowing for a greater breadth of data to be incorporated in the meta-analysis. All studies were published in peer-reviewed journals or as reports available from reputable national/international organizations, leading to the assumption of an acceptable level of data quality. Some reports (notably the UNEP/WHO data included in the Stockholm Convention GMP reports) do not include analytical information, although data are produced by recognized, accredited laboratories, and exclusion of this data would lead to a substantial loss in geographic coverage. Additionally, older studies reflecting very early analyses of PBDEs and HBCDDs may have more generous allowances in terms of QA/QC; however, it was important for the temporal analyses that these could be included. However, a consequence of this is that not all studies will meet the most stringent QA/QC standards.

While breastfeeding mothers present a relatively homogeneous population with respect to age and sex, some additional factors can impact breast milk concentrations of FRs, and these could not be incorporated into our meta-analysis, primarily because of inconsistent data across studies.

Representativeness. Data from breastmilk only reflects the chemical burden in the breast-feeding female population. The prevalence of breastfeeding mothers also varies by socioeconomic status and cultural background.[230]

Parity. Some studies have shown that parity is related to differences in levels of persistent compounds in breast milk. Primiparous mothers have been shown to have higher HBCDD concentrations than multiparous mothers,[132] while studies focused on PBDEs have not found a relationship.[84,187,207,231] As information on parity was not consistently recorded for all data sources, we did not include this confounder in our analyses and used all available data, regardless of parity.

Duration of Lactation and Sample Collection. The timing of breast milk collection within the lactation period varied widely, from 1 week to 10 months after birth, although the most typical was 3−8 weeks after birth, following the guidance of the WHO/UNEP breast milk surveys.[232] Whether this impacts levels of PBDEs and HBCDDs in milk is unclear. Some studies have reported variability[166,207] or significant decreases[233] in PBDE levels in breast milk up to a year postpartum; however, Harrad and Abdallah[151] reported no change in HBCDDs in milk over 12 months of lactation.

Maternal Age. This is often identified as a determinant of breast milk levels; however, this is directly related to the understanding of the persistence of these FRs in the body and temporal changes in exposures within a country.[234] Older maternal age has been associated with higher PBDE levels in some studies in breast milk[129,204] and sera,[235] while others have found no association[187] or an inverse association of lower levels in breast milk from older mothers.[84,207] Similarly for HBCDDs, Fujii et al. identified age-dependency of γ-HBCDD in milk,[132] but not other HBCDD isomers, while Drage et al.[235] found no age-dependency in HBCDDs in sera.

Selection of Breast Milk as Biomonitoring Matrix. While breast milk has the advantage of being noninvasive and widely monitored, BDE-209 preferentially partitions to serum lipids rather than milk lipids,[69] leading to proportionally lower levels in breast milk compared with exposures. However, while milk:serum partitioning can vary by congener/compound,[69] the relative geographic patterns and temporal trends (1227 samples from 1973 to 2019) should be appropriately captured by either matrix, as milk and serum concentrations are typically well-correlated.[215] The limited number of spatial and temporal studies conducted on certain compounds, such as BDE-209 and HBCDD in human breast milk is also a limitation for this study.

Uncertainties Regarding Partitioning and Half-Lives of PBDEs in Human Breast Milk. Very few studies have been conducted on the human biological distributions and half-lives of PBDE and HBCDDs. The existing evidence suggests higher persistence of BDE-153 in the human body[68,236,237] and decreased partitioning to milk for higher molecular weight FRs.[69] As a result, comparisons of concentrations across congeners would not necessarily reflect exposure trends; however, the bulk of our analysis relies on trends built individually for each congener and thus should not be impacted by the uncertainty in partitioning and half-lives.

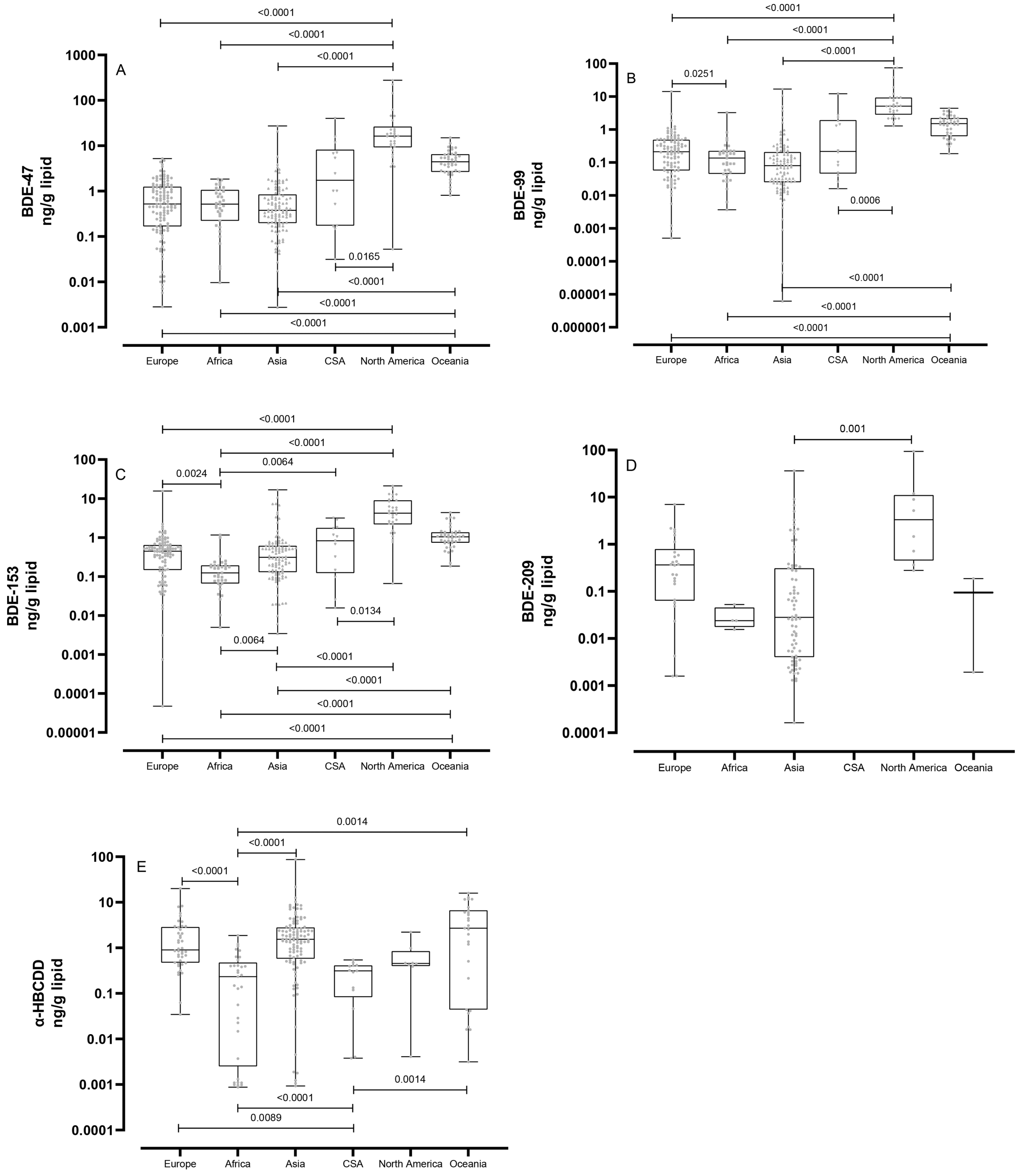

Figure 1. Box-and-whiskers (horizontal lines are medians, 95% confidence intervals, minima, and maxima) of (A) BDE-47, (B) BDE-99, (C) BDE-153, (D) BDE-209, and (E) *α*-HBCDD concentrations in different regions (CSA = Central and South America and the Caribbean), in breast milk data collected after year 2000. Nonparametric ANOVA tests (Kruskal–Wallis with Dunn's post-tests) were conducted to determine significant differences.

Temporal Patterns: Search Strategy and Selection Criteria. We supplemented our meta-analysis of temporal patterns of PBDE and HBCDD exposure with a comprehensive review of published time trends. A literature search was conducted using ISI Web of Science and Google Scholar, initially in March 2020, updated in September 2022. Search terms for the temporal trend studies were a combination of chemical-related terms (polybrominated diphenyl ether*, PBDE*, hexabromocyclododecane*, HBCD*), matrix-related (human, blood, serum, plasma, milk), and trend-related (time-trend* or temporal*).

For the temporal trend analysis of PBDEs, 268 data sources were identified, and the data were further examined to identify only studies that reported any of the four indicator PBDEs (−47, −99, −153, or −209), reflected the general population (not occupational exposure), reported basic biomonitoring parameters (e.g., geographic region, matrix, year of sample collection, number of samples), and had at least two time points with harmonized analyses (e.g., by the same laboratory). This resulted in 24 studies which were included in the overview of temporal trends (Table S2). For the temporal trend analysis of HBCDDs, the literature search identified 88 data sources, and the inclusion criteria (reported either *α*- or

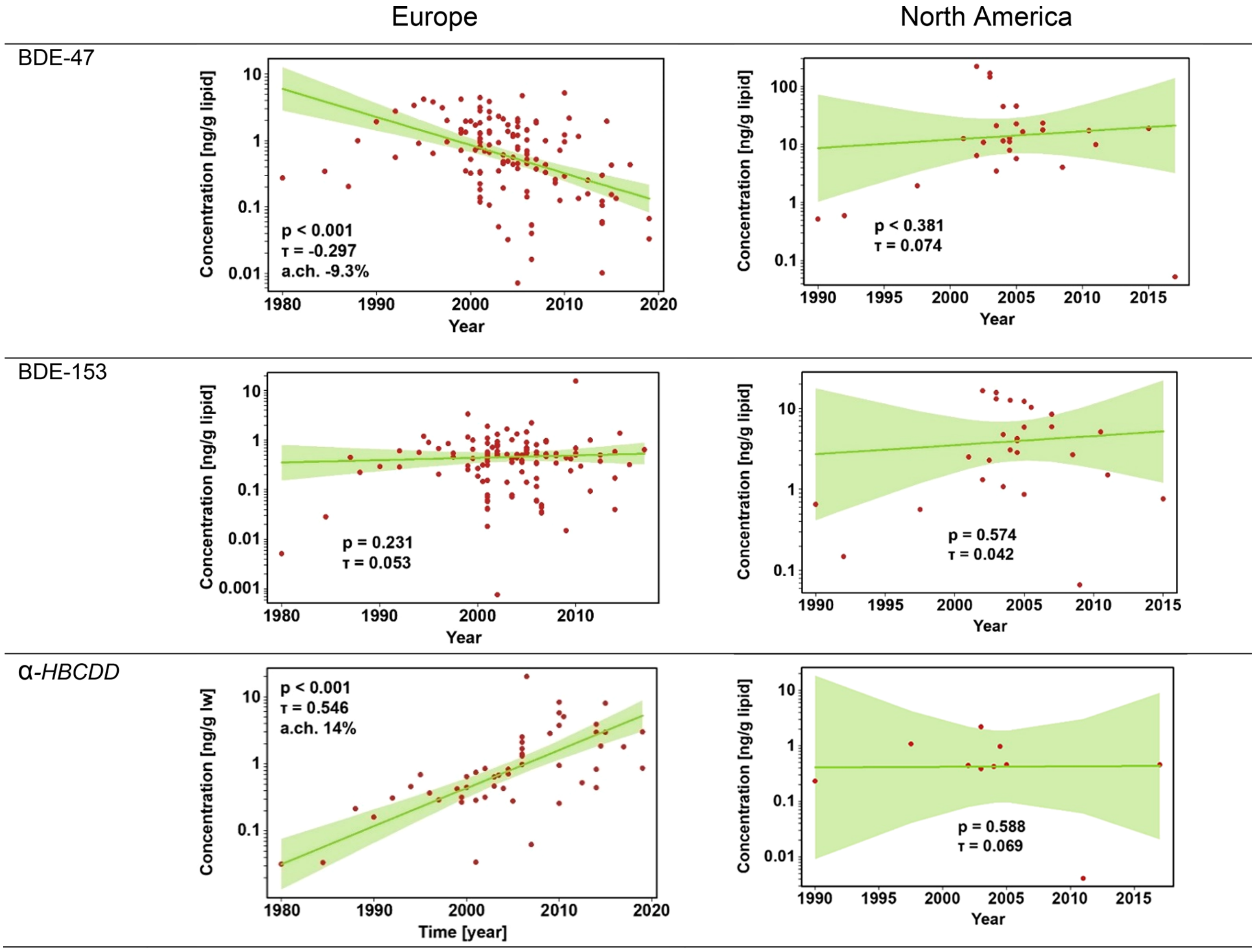

Figure 2. Weighted temporal trends of BDE-47, BDE-153, and α-HBCDD concentrations (ng/g lipid weight, lw) in breast milk from Europe and North America from literature. Shaded area indicates 95% confidence interval.

∑HBCDD, not occupational exposure, reported basic biomonitoring parameters) led to the inclusion of 20 studies (Table S3). The trends from these studies were extracted, using the interpretation of the study authors to determine whether a trend is classified as increasing or decreasing over time, or whether no time trend is apparent.

RESULTS

Spatial Patterns. Available data were not equally distributed across all geographic regions. Most studies were conducted in Europe and Asia, while Central and South America, North America, and Oceania had only limited data (Table 1). Very few studies on human biomonitoring of HBCDDs have been conducted in North America, which is surprising, considering that the region is known to have had stringent flame retardant regulations and historically high use of BFRs.[3]

Oceania had the highest median α-HBCDD concentration in breast milk (2.7 ng/g lipid), followed by Asia (1.5 ng/g lipid) and Europe (0.9 ng/g lipid) (Figure 1). The elevated concentrations of α-HBCDD in breast milk from Oceania were unexpected but may reflect a common market with many products from Asian manufacturers and Oceania implementing HBCDD regulations years later than Europe.[238] Breast milk from Asia had significantly higher concentrations of α-HBCDD than milk from Africa, and Europe had significantly higher concentrations than Central and South America (Figure 1).

The concentrations of BDE-47, -99, and -153 in breast milk from North America were significantly higher than those from Europe, Africa, Asia, and Central- and South America: BDE-47, and -99 concentrations in North America were 39 and 65 times higher, respectively, than concentrations in Asia (Figure 1, Table S1). For the higher brominated compounds, North America had 50 and 138 times higher concentrations than Africa for BDE-153 and -209, respectively (Figure 1, Table S1). Europe had significantly higher concentrations of BDE-99 than Africa, and Africa had significantly lower concentrations of BDE-153 than all the other regions. For BDE-209, there were fewer data records which limited the comparison. The fact that BDE-209 is notoriously difficult to quantify due to high molecular mass and chemical instability likely contributes to the lack of data on levels in breast milk.[239,240] For BDE-209, concentrations in milk from Asia were substantially lower than in North America, and no other regions had significant differences (Figure 1).

Temporal Patterns. Africa, Oceania, and Central and South America had limited or no data on BDE-209 concentrations in breast milk, and temporal patterns could not be fully evaluated (Table 1; Figures S5, S7, and S8). The most prominent differences in trends are seen between Europe

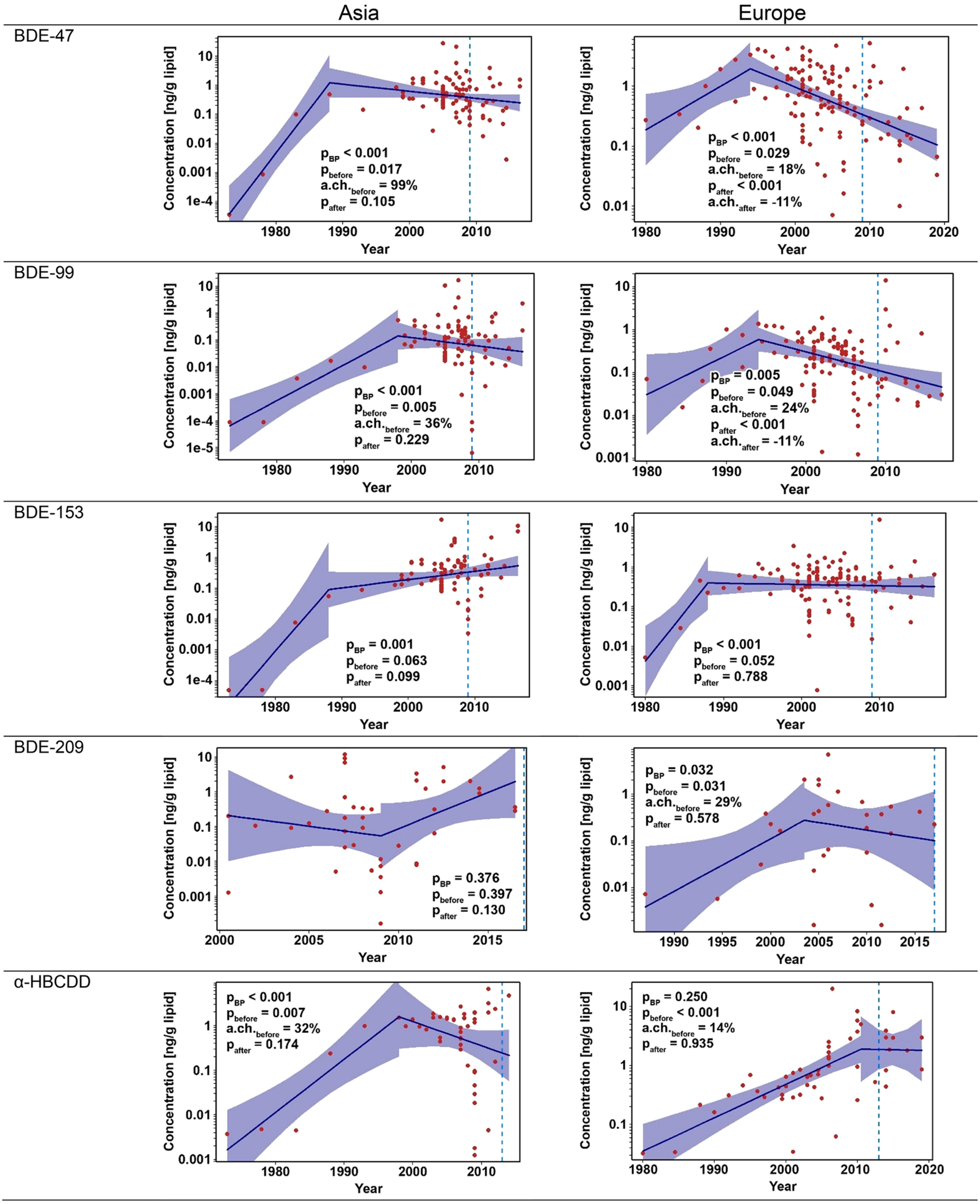

Figure 3. Results of breakpoint analysis for PBDEs in human milk for Asia and Europe for BDE-47, BDE-99, BDE-153, BDE-209, and *α*-HBCDD. The shaded area indicates the 95th percent confidence interval. The dotted blue line indicates when regulation was implemented by the Stockholm Convention (2009: BDE-47, -99, and -153; 2013: HBCDD; 2019: BDE-209). Please note that the Stockholm Convention date does not directly indicate the introduction of restrictions in each country; these may be earlier due to national/regional initiatives, or later, as Stockholm Convention parties enact regulations to implement the Convention. Thus, they are only indicative of the general timing of global restrictions.

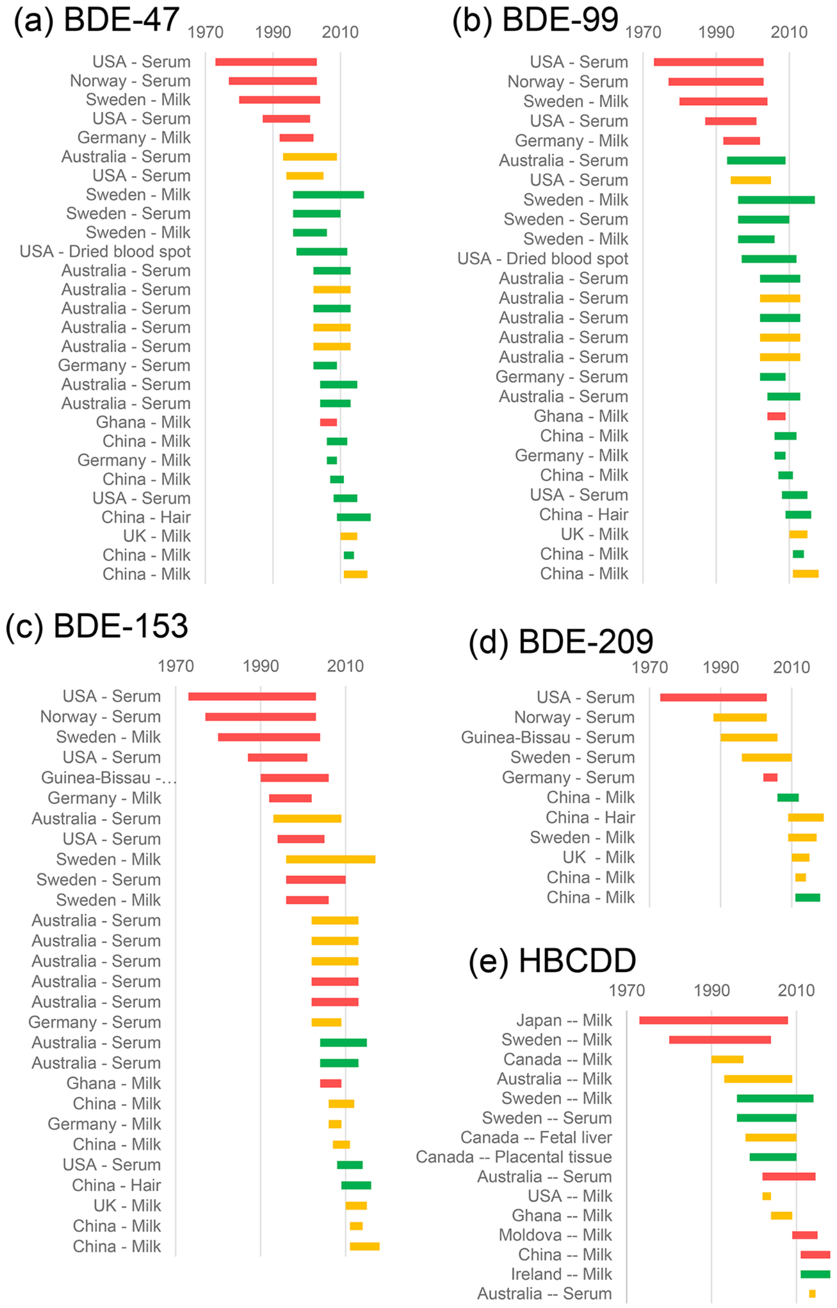

Figure 4. Time trends reported in human matrices in literature for (a) BDE-47, (b) BDE-99, (c) BDE-153, (d) BDE-209, and (e) HBCDD. Red bars indicate an increasing trend, yellow bars indicate no trend or a plateau, and green bars indicate a decreasing trend. Trends are classified based on the interpretations of the authors of each study. References for all studies can be found in Tables S2 and S3.

and North America for BDE-47, -153, and α-HBCDD. The temporal trends of BDE-99 and -209 are depicted elsewhere (Figures S3 and S4).

In Europe, BDE-47 and -99 decreased significantly (p = 0.0001; annual change of −9.3% and −10.1% respectively), while BDE-153 and -209 had no change over time (Figures 2 and S3). α-HBCDD concentrations increased significantly in the European population ($p < 0.0001$; 13.9% per annum) (Figure 2).

No significant changes ($p < 0.05$) were observed for any compound in North America (Figures 2 and S4), although we note that North American data was generally very sparse, limiting the ability to distinguish temporal trends. Notably, North America was the region with the least available α-HBCDD data points in breastmilk. The western hemisphere regions were the only regions where BDE-47 and -99 did not show a decreasing trend (Figures 2 and S4). The temporal patterns of all other geographical regions are presented in the Figures S5−S8.

Africa had a decrease in BDE-47 (p = 0.022; −7.8% per annum). Africa was the only continent where a significant decrease of α-HBCDD was observed ($p < 0.0001$; −33.6% per annum) (Figure S5). A similar pattern to Europe was seen in Asia. BDE-47 decreased (p = 0.002; −9.7 per annum), while BDE-153 and BDE-209 both increased, with BDE-209 increasing by 12.8% per annum (p = 0.05). α-HBCDD concentrations increased at a rate of 7.9% per annum (p = 0.006) (Figure S6).

In Central and South America and the Caribbean, BDE-153 increased significantly ($p = 0.08$) with an annual change of 14.8% (Figure S7). Oceania was the only region with a consistent decrease in all PBDE congeners. BDE-47 and -99 decreased significantly with an annual change of −13.1% and −15.5%, respectively. BDE-153 decreased at a rate of −6% per annum ($p = 0.054$) (Figure S8).

Breakpoint Analysis. Only Asia and Europe had sufficient data for breakpoint analyses (Figure 3). Breakpoints were calculated for other regions to identify the timing of concentration peaks (Figures S9−S12), but the trends are of limited value due to scarce data and are not discussed in detail.

In Europe, an increase in α-HBCDD of 13.9% per annum was observed from 1980 to 2010. In 2010, there was a change (breakpoint) in the temporal trends of α-HBCDD (Figure 3). The post-2010 decrease is not statistically significant due to limited data collected since 2010; thus, we cannot determine if recent concentrations are stable or declining. Although the breakpoint is not statistically significant, it represents the best fit for the available data and suggests a shift in exposure post-2010. In Asia, the α-HBCDD increase was even sharper (+31.6% per year; $p = 0.007$ Figure 3) and the breakpoint was identified earlier (1998), suggesting Asian concentrations reached a plateau at this point. Like Europe, the modeled decrease since the breakpoint in Asia is not statistically significant.

In both Asian and European breast milk data, the breakpoint in concentrations for BDE-47 and 99 was substantially earlier than for HBCDDs, close to 1990 for BDE-47 and between 1995 and 2000 for BDE-99. In all cases, concentrations decreased after the breakpoint for both congeners, but this postbreakpoint decrease is only significant for BDE-47 and BDE-99 in Europe ($p > 0.0001$).

However, there is a clear contrast between the breakpoints and before/after trends for BDE-153. The breakpoint for BDE-153 reflects only a change in the rate of increase of BDE-153 (Asia) or plateau (Europe) with no evidence of declining breast milk levels. Temporal patterns for BDE-209 in Asia were not significant, indicating no clear time trends. The European breakpoint for BDE-209 indicated a shift from the significant increase before 2004 to a current plateau or declining phase.

Comparison with Other Reported Time Trends. For the PBDEs, clear differences in the time trends by congener and by study timing are seen. For BDE-47 and -99 (Figure 4), there is a clear shift from early increasing time trends to more recent plateaus or decreasing time trends. The same pattern of early increasing trends and more recent reports of decreasing time trends is visible for BDE-153 (Figure 4), but the first decreases are not reported until much more recently. Most time trends for BDE-153 indicate a plateau. Relatively few studies report time trend analysis for HBCDD and BDE-209. For BDE-209 and HBCDD, no discernible time trend can be derived from literature for human matrices; trends were variable and not generalizable by region or duration of the time trend (Figure 4).

■ DISCUSSION AND IMPLICATIONS

Temporal Patterns. The analysis of published studies suggests that BDE-47 and -99 have a global decreasing trend. Decreasing temporal patterns were found in Europe, Asia, and Oceania. BDE-47 and -99 have human elimination half-lives of approximately 0.37−3 and 0.77−8 years, respectively.[68,236,237] In the time since the penta- and octa-PBDEs were included in the Stockholm Convention in 2004, the population would have been exposed to lower concentrations of the compounds, and the existing compounds would have been eliminated from their bodies. The United States is a signatory to the Stockholm Convention but has yet to ratify or implement the convention in national legislation.[241] The North American region was the only region with a continuous increase in BDE-47 and -99 (Figure 2).

While BDE-153 was also added to the Stockholm Convention in 2004, it does not share the same decreasing trends as the lower brominated congeners. All regions exhibited either a plateau or an increase in BDE-153 concentrations. Beyond the reduction in exposure due to the introduction of chemical restrictions, congener-specific differences in metabolism and storage in the body also affect breast milk trends differently. The human elimination half-life of BDE-153 is between 3.5 and 11.7 years.[68,236,237] When considering worst case scenario, a half-life of up to 11.7 years would result in a 4-fold reduction (in the absence of continued exposure) of concentrations measured around 23 years ago, at the time of the implementation of the Stockholm Convention. Therefore, a full elimination would not be expected by the time of writing. This, coupled with continued exposure from existing products, could explain the lack of a decrease of BDE-153 in breast milk (Figures 2 and 4).

Global restrictions on penta- and octa-BDE technical mixtures, which are dominated by lower brominated congeners were generally between 2004 and 2013,[33,242,243] whereas deca-BDE/BDE-209 was restricted only in 2017.[244] Conclusions on BDE-209 are limited because of the lack of data and likely because the temporal changes are not yet significant enough to be identified in the generally short-time trend analyses that have been performed (Figure 4). The short half-life of BDE-209 in the body (e.g., 15 days in blood[245]), combined with the lack of a visible decline in BDE-209 levels in milk, suggests ongoing consistent BDE-209 exposures, despite recent restrictions in production, particularly in Asia, where the temporal trend (Figure S6) and the inverse breakpoint (Figure 3) indicated increasing concentrations in BDE-209 in breast milk. This agrees with our understanding of the recent high use of BDE-209 in consumer products and building materials on a global scale and the lag time between chemical restrictions and product replacements: emission of BDE-209 from in-use and waste stocks is estimated to continue until 2050.[3]

It is concerning how few studies investigated HBCDDs in residents from the Americas ($n = 7$), Africa ($n = 5$), and Oceania ($n = 2$) (Table 1). A similar problem was observed with PBDEs, where Africa ($n = 7$), Oceania ($n = 7$), and Central- and South America ($n = 3$) had limited studies, while Asia and Europe had more studies ($n = 53$ and 61, respectively) (Table 1). Without the proper information on the state of contamination in the Americas, particularly the highly developed and industrialized North America ($n = 25$ studies on PBDE), it is impossible to determine a global perspective on the state of human exposure.

Breakpoint analyses are useful tools to determine whether the accumulation trend of a compound has increased, plateaued, or decreased over time, highlighting the approximate time when the change occurred. In Asia and Europe, the broad time trends of FR concentrations in breast milk are closely tied to the timing of chemical restrictions (Figure 3). The European trend indicated a breakpoint in ~2010 for α-HBCDD (Figure 3), which coincides with the increase in restrictions in Europe (identified as SVHC in 2009 and listed in Annex XIV of REACH in 2011)[40] and provides evidence that the restrictions impacted HBCDD use and thus exposures in Europe. After 2010, it is unclear whether α-HBCDDs are in a plateau phase or whether we begin seeing evidence of a decrease in Europe and Asia (Figure 3), but it suggests ongoing human exposure at levels close to the European peak.

It is important to highlight that in Figure 3, none of the breakpoints observed for Europe or Asia align with the timing of the Stockholm Convention's implementation (represented by the blue dotted line) but rather occurred earlier. This observation suggests that regional restrictions likely had a more pronounced impact than global restrictions. However, it should be acknowledged that the number of studies conducted after the implementation of the Stockholm Convention is limited compared to those conducted before its implementation, making it challenging to precisely assess the Convention's effectiveness, especially concerning HBCDD and deca-BDE.

Spatial Patterns. The strong contrast between PBDE concentrations in North America, particularly the USA, and most other regions is directly related to differences in flammability standards and PBDE use. BFRs have been quantified at higher concentrations in North America than in Europe in multiple matrices, including human tissue,[26] house dust,[246] and bird eggs.[247] The USA has historically had higher concentrations of flame retardants in its consumer products compared with other regions due to stricter flammability standards.[248−250] The fact that PBDE congeners still show an increasing time trend in North America (Figure 2) is likely linked to the large past use of PBDEs, combined with the later introduction of regulations. The United States has not ratified the Stockholm Convention and does not have any federal regulations on FR in existing uses, although 13 states have internal, state-wide concentration limits on FRs in selected products.[251] Significant New Use Rules (SNURs), implemented by the United States Environmental Protection Agency in 2012, aim to ensure that any new uses of specific flame retardant chemicals undergo a thorough review and approval process prior to manufacturing or processing. The US EPA implemented SNURs for Penta and OctaBDE in 2006, ensuring no new use or manufacturing of these compounds.[252] All manufacture, import, processing and distribution of decaBDE was under the US TSCA in 2021; however, significant exemptions remain, e.g., in motor vehicle parts until 2036.[253] Large numbers of products containing PBDEs are likely still in use or circulation, which leads to continuous exposure and a higher body burden.

Surprisingly, breast milk from the Oceania region was also significantly higher than most other global regions, save North America (Figure 1). While most of the world reduced PBDE use in 2004, whether through regulation voluntary action, Australia only began implementing regulation on PBDE use, manufacturing, and import in 2007.[238] Even though the import of PBDEs is banned, no regulation exists for the import of products that potentially could contain PBDEs, such as automobile parts, textiles, or electronic products.[254]

BDE-47 and -99, both primary components of penta-BDE, displayed similar spatial patterns (Figure 1), attributed to patterns in the use of technical penta-BDE formula worldwide.

General Observations. A substantial lag-time exists between cessation of production and cessation of use of FRs because of the long half-lives of the compounds and the lifespan of products that they are used in. Significant reductions in production have a slower effect on use and emissions because of the large stock of PBDE-containing materials in use. Abbasi et al.[3] estimated that the peak in PBDE use occurred in 2003; however, thousands of tons of PBDEs will remain in use in consumer products for decades. Plastic, textile, and electronic products containing FRs are still in use, to say nothing of buildings' thermal insulation containing EPS or XPS, which accounts for more than 97% of the global HBCDD volume used.[255] The ongoing human exposure to HBCDDs will be further mediated by HBCDD exposure through the renovation and demolition of buildings. Demolition activity can release significant amounts of building material-associated chemicals and demolition waste, including EPS or XPS panels, which are estimated to stay in place for ∼50 years before renovation takes place.[61] Thus, direct exposures to HBCDDs in indoor spaces and environmental release will continue for many decades,[256] effectively slowing the decreasing concentration levels through constant primary exposures. These can either contribute directly to either occupational or local population exposures, as well as increase the burden of secondary environmental exposures through landfill disposal and subsequent leaching of HBCDDs from the products.[31]

Furthermore, as we move toward a circular economy, there is significant potential for FRs to be incorporated into new consumer products made from recycled materials.[257,258] Abbasi et al.[3] estimated that 45000 t of PBDEs may reappear in new products made from recycled materials, such as plastics, food contact materials,[259,260] and children's toys.[261,262] Due to the persistence of FRs, all environmental releases can contribute to secondary FR contamination in the surrounding air, soil, and water sources and lead to human exposure via dietary sources.[263]

The lack of recent biomonitoring studies on PBDEs and HBCDDs limits the evaluation of current population exposures and the impact of regulation on time trends. Of the human biomonitoring studies (Table 1 and Figure 4), less than 10% of the records cover the period after 2013, limiting our ability to evaluate the effectiveness of legislation on the global exposure trend of FRs. This may be due to the perception that once chemicals have been regulated, the problem has been dealt with, and it can be difficult to maintain interest and/or financial support for chemicals that are perceived to have been already addressed.[110]

Regulation has a quantifiable effect on the concentrations of FRs in human breast milk. Regions such as Asia and Europe where earlier regional regulations, prior to the Stockholm Convention, have been implemented had the clearest decline of most of the compounds that were considered in this study. Australia, where regulations were implemented later still shows elevated concentrations in human matrices, as does North America, which had the historically highest FR use globally. On top of use and regulation, the chemical characteristics of specific PBDE congeners also affect the response to regulation and can impact the time needed to evaluate the efficacy of policy actions. Although PBDEs and HBCDD are regulated through a ban on production, regulation will have different outcomes for different individual compounds based on their dominant use, the volume of historical use, and biological half-lives. However, it is encouraging to see that existing regulation and policy, when implemented early and comprehensively, had a positive impact on the decreasing human body burden of lower brominated flame retardants.

■ ASSOCIATED CONTENT

Supporting Information

The Supporting Information is available free of charge at https://pubs.acs.org/doi/10.1021/acs.est.3c02896.

Figures showing temporal trend and breakpoint analysis for all flame retardants from all regions and tables detailing references used in literature review and meta-analysis (PDF)

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Notes

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■ REFERENCES

(1) Wemken, N.; Drage, D. S.; Cellarius, C.; Cleere, K.; Morrison, J. J.; Daly, S.; Abdallah, M. A. E.; Tlustos, C.; Harrad, S.; Coggins, M. A. Emerging and Legacy Brominated Flame Retardants in the Breast Milk of First Time Irish Mothers Suggest Positive Response to Restrictions on Use of HBCDD and Penta- and Octa-BDE Formulations. *Environ. Res.* **2020**, *180*, No. 108805.

(2) Montalbano, A. M.; Albano, G. D.; Anzalone, G.; Moscato, M.; Gagliardo, R.; Di Sano, C.; Bonanno, A.; Ruggieri, S.; Cibella, F.; Profita, M. Cytotoxic and Genotoxic Effects of the Flame Retardants (PBDE-47, PBDE-99 and PBDE-209) in Human Bronchial Epithelial Cells. *Chemosphere* **2020**, *245*, No. 125600.

(3) Abbasi, G.; Li, L.; Breivik, K. Global Historical Stocks and Emissions of PBDEs. *Environ. Sci. Technol.* **2019**, *53* (11), 6330−6340.

(4) De Wit, C. A. An Overview of Brominated Flame Retardants in the Environment. *Chemosphere* **2002**, *46* (5), 583−624.

(5) Abbasi, G.; Buser, A. M.; Soehl, A.; Murray, M. W.; Diamond, M. L. Stocks and Flows of PBDEs in Products from Use to Waste in the U.S. and Canada from 1970 to 2020. *Environ. Sci. Technol.* **2015**, *49* (3), 1521−1528.

(6) Covaci, A.; Gerecke, A. C.; Law, R. J.; Voorspoels, S.; Kohler, M.; Heeb, N. V.; Leslie, H.; Allchin, C. R.; de Boer, J. Hexabromocyclododecanes (HBCDs) in the Environment and Humans: A Review. *Environ. Sci. Technol.* **2006**, *40* (12), 3679−3688.

(7) Koch, C.; Schmidt-Kötters, T.; Rupp, R.; Sures, B. Review of Hexabromocyclododecane (HBCD) with a Focus on Legislation and Recent Publications Concerning Toxicokinetics and -Dynamics. *Environ. Pollut.* **2015**, *199*, 26−34.

(8) de Wit, C. A.; Herzke, D.; Vorkamp, K. Brominated Flame Retardants in the Arctic Environment - Trends and New Candidates. *Sci. Total Environ.* **2010**, *408* (15), 2885−2918.

(9) Law, R. J.; Covaci, A.; Harrad, S. J.; Herzke, D.; Abdallah, M. A.-E.; Fernie, K.; Toms, L.-M. L.; Takigami, H. Levels and Trends of PBDEs and HBCDs in the Global Environment: Status at the End of 2012. *Environ. Int.* **2014**, *65*, 147−158.

(10) Sjodin, A; Hagmar, L; Klasson-Wehler, E; Kronholm-Diab, K; Jakobsson, E; Bergman, A Flame Retardant Exposure: Polybrominated Diphenyl Ethers in Blood from Swedish Workers. *Environ. Health Perspect* **1999**, *107* (8), 643−648.

(11) Klasson-Wehler, E.; Hovander, L.; Bergman, A. New Organohalogens in Human Plasma - Identification and Quantification. *Organohalogen Compd.* **1997**, *33*, 420−425.

(12) Schroter-Kermani, C.; Helm, D.; Herrmann, T.; Papke, O. The German Environmental Specimen Bank-Application in Trend Monitoring of Polybrominated Diphenyl Ethers in Human Blood. *Organohalogen Compd.* **2000**, *47*, 49−52.

(13) Stanley, J. S.; Cramer, P. H.; Thornburg, K. R.; Remmers, J. C.; Breen, J. J.; Schwemberger, J. Mass Spectral Confirmation of Chlorinated and Brominated Diphenylethers in Human Adipose Tissues. *Chemosphere* **1991**, *23* (8−10), 1185−1195.

(14) Haglund, P. S.; Zook, D. R.; Buser, H. R.; Hu, J. Identification and Quantification of Polybrominated Diphenyl Ethers and Methoxy-Polybrominated Diphenyl Ethers in Baltic Biota. *Environ. Sci. Technol.* **1997**, *31* (11), 3281−3287.

(15) Meneses, M.; Wingfors, H.; Schuhmacher, M.; Domingo, J.L.; Lindstrom, G.; Bavel, B.v. Polybrominated Diphenyl Ethers Detected in Human Adipose Tissue from Spain. *Chemosphere* **1999**, *39* (13), 2271−2278.

(16) Norén, K.; Meironyté, D. Certain Organochlorine and Organobromine Contaminants in Swedish Human Milk in Perspective of Past 20−30 Years. *Chemosphere* **2000**, *40* (9−11), 1111−1123.

(17) Siddiqi, M. A.; Laessig, R. H.; Reed, K. D. Polybrominated Diphenyl Ethers (PBDEs): New Pollutants-Old Diseases. *Clin Med. Res.* **2003**, *1*, 281−290.

(18) Inoue, K.; Harada, K.; Takenaka, K.; Uehara, S.; Kono, M.; Shimizu, T.; Takasuga, T.; Senthilkumar, K.; Yamashita, F.; Koizumi, A. Levels and Concentration Ratios of Polychlorinated Biphenyls and Polybrominated Diphenyl Ethers in Serum and Breast Milk in Japanese Mothers. *Environ. Health Perspect* **2006**, *114* (8), 1179−1185.

(19) Lind, Y.; Darnerud, P. O.; Atuma, S.; Aune, M.; Becker, W.; Bjerselius, R.; Cnattingius, S.; Glynn, A. Polybrominated Diphenyl Ethers in Breast Milk from Uppsala County, Sweden. *Environ. Res.* **2003**, *93* (2), 186−194.

(20) Bjermo, H.; Aune, M.; Cantillana, T.; Glynn, A.; Lind, P. M.; Ridefelt, P.; Darnerud, P. O. Serum Levels of Brominated Flame Retardants (BFRs: PBDE, HBCD) and Influence of Dietary Factors in a Population-Based Study on Swedish Adults. *Chemosphere* **2017**, *167*, 485−491.

(21) Barghi, M.; Shin, E. su; Choi, S. D.; Dahmardeh Behrooz, R.; Chang, Y. S. HBCD and TBBPA in Human Scalp Hair: Evidence of Internal Exposure. *Chemosphere* **2018**, *207*, 70−77.

(22) Drage, D. S.; Mueller, J. F.; Hobson, P.; Harden, F. A.; Toms, L. M. L. Demographic and Temporal Trends of Hexabromocyclododecanes (HBCDD) in an Australian Population. *Environ. Res.* **2017**, *152*, 192−198.

(23) Inthavong, C.; Hommet, F.; Bordet, F.; Rigourd, V.; Guérin, T.; Dragacci, S. Simultaneous Liquid Chromatography−Tandem Mass Spectrometry Analysis of Brominated Flame Retardants (Tetrabromobisphenol A and Hexabromocyclododecane Diastereoisomers) in French Breast Milk. *Chemosphere* **2017**, *186*, 762−769.

(24) Dorman, D. C.; Chiu, W.; Hales, B. F.; Hauser, R.; Johnson, K. J.; Mantus, E.; Martel, S.; Robinson, K. A.; Rooney, A. A.; Rudel, R.; Sathyanarayana, S.; Schantz, S. L.; Waters, K. M. Polybrominated Diphenyl Ether (PBDE) Neurotoxicity: A Systematic Review and Meta-Analysis of Animal Evidence. *Journal of Toxicology and Environmental Health, Part B* **2018**, *21* (4), 269−289.

(25) Wan, M. L. Y.; Co, V. A.; El-Nezami, H. Endocrine Disrupting Chemicals and Breast Cancer: A Systematic Review of Epidemiological Studies. *Crit Rev. Food Sci. Nutr* **2022**, *62* (24), 6549−6576.

(26) Lyche, J. L.; Rosseland, C.; Berge, G.; Polder, A. Human Health Risk Associated with Brominated Flame-Retardants (BFRs). *Environment International*. Elsevier Ltd, 2015; pp 170−180, DOI: 10.1016/j.envint.2014.09.006.

(27) Lam, J.; Lanphear, B. P.; Bellinger, D.; Axelrad, D. A.; McPartland, J.; Sutton, P.; Davidson, L.; Daniels, N.; Sen, S.; Woodruff, T. J. Developmental PBDE Exposure and IQ/ADHD in Childhood: A Systematic Review and Meta-Analysis. *Environ. Health Perspect* **2017**, *125* (8), DOI: 10.1289/EHP1632.

(28) Moore, S.; Paalanen, L.; Melymuk, L.; Katsonouri, A.; Kolossa-Gehring, M.; Tolonen, H. The Association between ADHD and Environmental Chemicals—A Scoping Review. *Int. J. Environ. Res. Public Health* **2022**, *19* (5), 2849.

(29) Iszatt, N.; Janssen, S.; Lenters, V.; Dahl, C.; Stigum, H.; Knight, R.; Mandal, S.; Peddada, S.; González, A.; Midtvedt, T.; Eggesbø, M. Environmental Toxicants in Breast Milk of Norwegian Mothers and Gut Bacteria Composition and Metabolites in Their Infants at 1 Month. *Microbiome* **2019**, *7*, 34.

(30) Gascon, M.; Fort, M.; Martinez, D.; Carsin, A.-E.; Forns, J.; Grimalt, J. O.; Santa Marina, L.; Lertxundi, N.; Sunyer, J.; Vrijheid, M. Polybrominated Diphenyl Ethers (PBDEs) in Breast Milk and Neuropsychological Development in Infants. *Environ. Health Perspect* **2012**, *120* (12), 1760−1765.

(31) Feiteiro, J.; Mariana, M.; Cairrão, E. Health Toxicity Effects of Brominated Flame Retardants: From Environmental to Human Exposure. *Environ. Pollut.* **2021**, *285*, No. 117475.

(32) Brown, P.; Cordner, A. Lessons Learned from Flame Retardant Use and Regulation Could Enhance Future Control of Potentially Hazardous Chemicals. *Health Aff* **2011**, *30* (5), 906−914.

(33) European Commission. Directive 2003/11/EC of the European Parliament and of the Council of February 6 2003 Amending for the 24th Time Council Directive 76/669/EEC Relating to Restrictions on the Marketing and Use of Certain Dangerous Substances and Preparations (Pentabromodip. *Off J. Eur. Union OJ. L* **2003**, *16* (1), 45−46.

(34) ATSDR. Toxic Substances Portal. ToxFAQs. https://wwwn.cdc.gov/TSP/ToxFAQs/ToxFAQsDetails.aspx?faqid=1462&toxid=183 (accessed 2023-01-28).

(35) Stockholm Convention; TetraBDE and pentaBDE. https://chm.pops.int/Implementation/Alternatives/AlternativestoPOPs/ChemicalslistedinAnnexA/TetraBDEandpentaBDE/tabid/5868/Default.aspx (accessed 2023-07-24).

(36) Stockholm Convention: HexaBDE and heptaBDE. https://chm.pops.int/Implementation/Alternatives/AlternativestoPOPs/ChemicalslistedinAnnexA/HexaBDEandheptaBDE/tabid/5862/Default.aspx (accessed 2023-07-24).

(37) Stockholm Convention: Decabromodiphenyl ether (commercial mixture, c-decaBDE). https://chm.pops.int/Implementation/Alternatives/AlternativestoPOPs/ChemicalslistedinAnnexA/cdecaBDE/tabid/5985/Default.aspx (accessed 2023-07-24).

(38) UNEP. Stockholm Convention: United Nations Environment Programme (UNEP). www.pops.int (accessed 2023-02-13).

(39) Risk Assessment: Hexabromocyclododecane, Final report, May 2008. https://echa.europa.eu/documents/10162/661bff17-dc0a-4475-9758-40bdd6198f82. (accessed 2023-01-28).

(40) Kemmlein, S.; Herzke, D.; Law, R. J. Brominated Flame Retardants in the European Chemicals Policy of REACH-Regulation and Determination in Materials. *Journal of Chromatography A* **2009**, *1216* (3), 320−333.

(41) UNEP. Stockholm Convention Global Monitoring Plan for Persistent Organic Pollutants: First Global Monitoring Report; Geneva, 2009. https://chm.pops.int/implementation/globalmonitoringplan/monitoringreports/tabid/525/default.aspx. (accessed 2023-03-30).

(42) Sharma, B. M.; Kalina, J.; Whaley, P.; Scheringer, M. Towards Guidelines for Time-Trend Reviews Examining Temporal Variability in Human Biomonitoring Data of Pollutants. *Environ. Int.* **2021**, *151*, No. 106437.

(43) Wong, F.; Hung, H.; Dryfhout-Clark, H.; Aas, W.; Bohlin-Nizzetto, P.; Breivik, K.; Mastromonaco, M. N.; Lundén, E. B.; Ólafsdóttir, K.; Sigurosson, Á.; Vorkamp, K.; Bossi, R.; Skov, H.; Hakola, H.; Barresi, E.; Sverko, E.; Fellin, P.; Li, H.; Vlasenko, A.; Zapevalov, M.; Samsonov, D.; Wilson, S. Time Trends of Persistent Organic Pollutants (POPs) and Chemicals of Emerging Arctic Concern (CEAC) in Arctic Air from 25 Years of Monitoring. *Sci. Total Environ.* **2021**, *775*, No. 145109.

(44) Schuster, J. K.; Gioia, R.; Moeckel, C.; Agarwal, T.; Bucheli, T. D.; Breivik, K.; Steinnes, E.; Jones, K. C. Has the Burden and Distribution of PCBs and PBDEs Changed in European Background Soils between 1998 and 2008? Implications for Sources and Processes. *Environ. Sci. Technol.* **2011**, *45* (17), 7291−7297.

(45) Sutton, R.; Sedlak, M. D.; Yee, D.; Davis, J. A.; Crane, D.; Grace, R.; Arsem, N. Declines in Polybrominated Diphenyl Ether Contamination of San Francisco Bay Following Production Phase-Outs and Bans. *Environ. Sci. Technol.* **2015**, *49* (2), 777−784.

(46) Olofsson, U.; Bignert, A.; Haglund, P. Time-Trends of Metals and Organic Contaminants in Sewage Sludge. *Water Res.* **2012**, *46* (15), 4841−4851.

(47) Crimmins, B. S.; Pagano, J. J.; Xia, X.; Hopke, P. K.; Milligan, M. S.; Holsen, T. M. Polybrominated Diphenyl Ethers (PBDEs): Turning the Corner in Great Lakes Trout 1980−2009. *Environ. Sci. Technol.* **2012**, *46* (18), 9890−9897.

(48) Hale, R. C.; La Guardia, M. J.; Harvey, E.; Chen, D.; Mainor, T. M.; Luellen, D. R.; Hundal, L. S. Polybrominated Diphenyl Ethers in U.S. Sewage Sludges and Biosolids: Temporal and Geographical Trends and Uptake by Corn Following Land Application. *Environ. Sci. Technol.* **2012**, *46* (4), 2055−2063.

(49) Leslie, H. A.; Brandsma, S. H.; Barber, J. L.; Gabrielsen, G. W.; Bersuder, P.; Barry, J.; Shore, R. F.; Walker, L. A.; de Boer, J. Decabromodiphenylether Trends in the European Environment: Bird Eggs, Sewage Sludge and Surficial Sediments. *Sci. Total Environ.* **2021**, *774*, No. 145174.

(50) Turyk, M. E.; Anderson, H. A.; Steenport, D.; Buelow, C.; Imm, P.; Knobeloch, L. Longitudinal Biomonitoring for Polybrominated Diphenyl Ethers (PBDEs) in Residents of the Great Lakes Basin. *Chemosphere* **2010**, *81* (4), 517−522.

(51) Schecter, A.; Papke, O.; Tung, K. C.; Joseph, J.; Harris, T R.; Dahlgren, J. Polybrominated Diphenyl Ether Flame Retardants in the U.S. Population: Current Levels, Temporal Trends, and Comparison with Dioxins, Dibenzofurans, and Polychlorinated Biphenyls. *J. Occup Environ. Med.* **2005**, *47* (3), 199−211.

(52) Law, R. J.; Barry, J.; Barber, J. L.; Bersuder, P.; Deaville, R.; Reid, R. J.; Brownlow, A.; Penrose, R.; Barnett, J.; Loveridge, J.; Smith, B.; Jepson, P. D. Contaminants in Cetaceans from UK Waters: Status as Assessed within the Cetacean Strandings Investigation Programme from 1990 to 2008. *Mar. Pollut. Bull.* **2012**, *64* (7), 1485−1494.

(53) Munschy, C.; Marchand, P.; Venisseau, A.; Veyrand, B.; Zendong, Z. Levels and Trends of the Emerging Contaminants HBCDs (Hexabromocyclododecanes) and PFCs (Perfluorinated Compounds) in Marine Shellfish along French Coasts. *Chemosphere* **2013**, *91* (2), 233−240.

(54) Smythe, T. A.; Loseto, L. L.; Bignert, A.; Rosenberg, B.; Budakowski, W.; Halldorson, T.; Pleskach, K.; Tomy, G. T. Temporal Trends of Brominated and Fluorinated Contaminants in Canadian Arctic Beluga Delphinapterus Leucas. *Arct Sci.* **2018**, *4* (3), 388−404.

(55) Johansson, A.-K.; Sellström, U.; Lindberg, P.; Bignert, A.; de Wit, C. A. Temporal Trends of Polybrominated Diphenyl Ethers and Hexabromocyclododecane in Swedish Peregrine Falcon (Falco Peregrinus Peregrinus) Eggs. *Environ. Int.* **2011**, *37* (4), 678−686.

(56) Vorkamp, K.; Falk, K.; Møller, S.; Rigét, F. F.; Sørensen, P. B. Regulated and Unregulated Halogenated Flame Retardants in Peregrine Falcon Eggs from Greenland. *Environ. Sci. Technol.* **2018**, *52* (2), 474−483.

(57) Rüdel, H.; Müller, J.; Nowak, J.; Ricking, M.; Klein, R.; Kotthoff, M. Hexabromocyclododecane Diastereomers in Fish and Suspended Particulate Matter from Selected European Waters—Trend Monitoring and Environmental Quality Standard Compliance. *Environmental Science and Pollution Research* **2017**, *24* (22), 18048−18062.

(58) Aznar-Alemany, Ò.; Sala, B.; Jobst, K. J.; Reiner, E. J.; Borrell, A.; Aguilar, À.; Eljarrat, E. Temporal Trends of Halogenated and Organophosphate Contaminants in Striped Dolphins from the Mediterranean Sea. *Sci. Total Environ.* **2021**, *753*, No. 142205.

(59) Tao, F.; Abou-Elwafa Abdallah, M.; Ashworth, D. C.; Douglas, P.; Toledano, M. B.; Harrad, S. Emerging and Legacy Flame Retardants in UK Human Milk and Food Suggest Slow Response to Restrictions on Use of PBDEs and HBCDD. *Environ. Int.* **2017**, *105*, 95−104.

(60) Magnier, L.; Mugge, R. Replaced Too Soon? An Exploration of Western European Consumers' Replacement of Electronic Products. *Resour Conserv Recycl* **2022**, *185*, No. 106448.

(61) Charbonnet, J. A.; Weber, R.; Blum, A. Flammability Standards for Furniture, Building Insulation and Electronics: Benefit and Risk. *Emerg. Contam.* **2020**, *6*, 432−441.

(62) Abbatt, J. P. D.; Wang, C. The Atmospheric Chemistry of Indoor Environments. *Environ. Sci.: Processes Impacts* **2020**, *22*, 25−48.

(63) Melymuk, L.; Demirtepe, H.; Jílková, S. R. Indoor Dust and Associated Chemical Exposures. *Current Opinion in Environmental Science and Health* **2020**, *15*, 1−6.

(64) Rodgers, K. M.; Bennett, D.; Moran, R.; Knox, K.; Stoiber, T.; Gill, R.; Young, T. M.; Blum, A.; Dodson, R. E. Do Flame Retardant Concentrations Change in Dust after Older Upholstered Furniture Is Replaced? *Environ. Int.* **2021**, *153*, No. 106513.

(65) Bennett, D. H.; Moran, R. E.; Wu, X. M.; Tulve, N. S.; Clifton, M. S.; Colon, M.; Weathers, W.; Sjodin, A.; Jones, R.; Hertz-Picciotto, I. Polybrominated Diphenyl Ether (PBDE) Concentrations and Resulting Exposure in Homes in California: Relationships among Passive Air, Surface Wipe and Dust Concentrations, and Temporal Variability. *Indoor Air* **2015**, *25* (2), 220−229.

(66) van der Schyff, V.; Kalina, J.; Govarts, E.; Gilles, L.; Schoeters, G.; Castaño, A.; Esteban-López, M.; Kohoutek, J.; Kukučka, P.; Covaci, A.; Koppen, G.; Andrýsková, L.; Piler, P.; Klánová, J.; Jensen, T. K.; Rambaud, L.; Riou, M.; Lamoree, M.; Kolossa-Gehring, M.; Vogel, N.; Weber, T.; Göen, T.; Gabriel, C.; Sarigiannis, D. A.; Sakhi, A. K.; Haug, L. S.; Murinova, L. P.; Fabelova, L.; Tratnik, J. S.; Mazej, D.; Melymuk, L. Exposure to Flame Retardants in European Children — Results from the HBM4 EU Aligned Studies. *International Journal of Hygiene and Environmental Health* **2023**, *247*, No. 114070.

(67) Shin, H. M.; McKone, T. E.; Tulve, N. S.; Clifton, M. S.; Bennett, D. H. Indoor Residence Times of Semivolatile Organic Compounds: Model Estimation and Field Evaluation. *Environ. Sci. Technol.* **2013**, *47* (2), 859−867.

(68) Geyer, H. J.; Schramm, K.-W.; Darnerud, O.; Aune, M.; Feicht, E. A.; Fried, K. W.; Henkelmann, B.; Lenoir, D.; Schmid, P.; Mcdonald, T. A. Terminal Elimination Half-Lives of the Brominated Flame Retardants TBBPA, HBCD, and Lower Brominated PBDEs in Humans. *Organohalogen Compd.* **2004**, *66*, 3820−3825.

(69) t'Mannetje, A.; Coakley, J.; Mueller, J. F.; Harden, F.; Toms, L.-M.; Douwes, J. Partitioning of Persistent Organic Pollutants (POPs) between Human Serum and Breast Milk: A Literature Review. *Chemosphere* **2012**, *89* (8), 911−918.

(70) Schecter, A.; Colacino, J.; Sjödin, A.; Needham, L.; Birnbaum, L. Partitioning of Polybrominated Diphenyl Ethers (PBDEs) in Serum and Milk from the Same Mothers. *Chemosphere* **2010**, *78* (10), 1279−1284.

(71) Komprda, J.; Komprdová, K.; Domínguez-Romero, E.; Mikeš, O.; Řiháčková, K.; Čupr, P.; Černá, M.; Scheringer, M. Dynamics of PCB Exposure in the Past 50 Years and Recent High Concentrations in Human Breast Milk: Analysis of Influencing Factors Using a Physiologically Based Pharmacokinetic Model. *Sci. Total Environ.* **2019**, *690*, 388−399.

(72) Hajeb, P.; Castaño, A.; Cequier, E.; Covaci, A.; López, M. E.; Antuña, A. G.; Haug, L. S.; Henríquez-Hernández, L. A.; Melymuk, L.; Pérez Luzardo, O.; Thomsen, C.; Vorkamp, K. Critical Review of Analytical Methods for the Determination of Flame Retardants in Human Matrices. *Anal. Chim. Acta* **2022**, *1193*, No. 338828.

(73) UNEP. Global Monitoring Plan for Persistent Organic Pollutants under the Stockholm Convention Article 16 on Effectiveness Evaluation - Third Regional Monitoring Report - Latin America and the Caribbean; 2021. https://chm.pops.int/implementation/globalmonitoringplan/monitoringreports/tabid/525/default.aspx. (accessed 2023-03-30).

(74) Johnson-Restrepo, B.; Kannan, K. An Assessment of Sources and Pathways of Human Exposure to Polybrominated Diphenyl Ethers in the United States. *Chemosphere* **2009**, *76* (4), 542−548.

(75) Sahlström, L. M. O.; Sellström, U.; de Wit, C. A.; Lignell, S.; Darnerud, P. O. Brominated Flame Retardants in Matched Serum Samples from Swedish First-Time Mothers and Their Toddlers. *Environ. Sci. Technol.* **2014**, *48* (13), 7584−7592.

(76) Eljarrat, E.; Guerra, P.; Martínez, E.; Farré, M.; Alvarez, J. G.; López-Teijón, M.; Barceló, D. Hexabromocyclododecane in Human Breast Milk: Levels and Enantiomeric Patterns. *Environ. Sci. Technol.* **2009**, *43* (6), 1940−1946.

(77) Ryan, J. J.; Rawn, D. F. K. The Brominated Flame Retardants, PBDEs and HBCD, in Canadian Human Milk Samples Collected from 1992 to 2005; Concentrations and Trends. *Environ. Int.* **2014**, *70*, 1−8.

(78) Carignan, C. C.; Abdallah, M. A. E.; Wu, N.; Heiger-Bernays, W.; McClean, M. D.; Harrad, S.; Webster, T. F. Predictors of Tetrabromobisphenol-A (TBBP-A) and Hexabromocyclododecanes (HBCD) in Milk from Boston Mothers. *Environ. Sci. Technol.* **2012**, *46* (21), 12146−12153.

(79) Asante, K. A.; Adu-Kumi, S.; Nakahiro, K.; Takahashi, S.; Isobe, T.; Sudaryanto, A.; Devanathan, G.; Clarke, E.; Ansa-Asare, O. D.; Dapaah-Siakwan, S.; Tanabe, S. Human Exposure to PCBs, PBDEs and HBCDs in Ghana: Temporal Variation, Sources of Exposure and Estimation of Daily Intakes by Infants. *Environ. Int.* **2011**, *37* (5), 921−928.

(80) Darnerud, P. O.; Aune, M.; Larsson, L.; Lignell, S.; Mutshatshi, T.; Okonkwo, J.; Botha, B.; Agyei, N. Levels of Brominated Flame Retardants and Other Pesistent Organic Pollutants in Breast Milk Samples from Limpopo Province, South Africa. *Sci. Total Environ.* **2011**, *409* (19), 4048−4053.

(81) Hůlek, R.; Borůvková, J.; Gregor, J.; Kalina, J.; Bednářová, Z.; Šebková, K.; Melkes, O.; Šalko, M.; Novák, R.; Jarkovský, J.; Dušek, L.; Klánová, J. *Global Monitoring Plan of the Stockholm Convention on Persistent Organic Pollutants: visualisation and on-line analysis of global levels of chemicals in air, water, breast milk and blood [online]*. Masaryk University, 2014, http://visualization.pops-gmp.org/2014/ (accessed 2022-07-14).

(82) Müller, M. H. B.; Polder, A.; Brynildsrud, O. B.; Lie, E.; Løken, K. B.; Manyilizu, W. B.; Mdegela, R. H.; Mokiti, F.; Murtadha, M.; Nonga, H. E.; Skaare, J. U.; Lyche, J. L. Brominated Flame Retardants

(BFRs) in Breast Milk and Associated Health Risks to Nursing Infants in Northern Tanzania. *Environ. Int.* **2016**, *89–90*, 38–47.

(83) UNEP. Global Monitoring Plan for Persistent Organic Pollutants under the Stockholm Convention Article 16 on Effectiveness Evaluation - Third Regional Monitoring Report - Africa Region; 2021. https://chm.pops.int/implementation/globalmonitoringplan/monitoringreports/tabid/525/default.aspx. (accessed 2023-03-30).

(84) Hassine, S. B.; Ameur, W. B.; Gandoura, N.; Driss, M. R. Determination of Chlorinated Pesticides, Polychlorinated Biphenyls, and Polybrominated Diphenyl Ethers in Human Milk from Bizerte (Tunisia) in 2010. *Chemosphere* **2012**, *89* (4), 369–377.

(85) Matovu, H.; Sillanpää, M.; Ssebugere, P. Polybrominated Diphenyl Ethers in Mothers' Breast Milk and Associated Health Risk to Nursing Infants in Uganda. *Science of The Total Environment* **2019**, *692*, 1106–1115.

(86) Horng, C.; Chao, H.; Chang, C.; Govindasamy, A.; Tzeng, T.; Shieh, P. Polybrominated Diphenyl Ethers in Human Milk in Taiwan. *Asian J. Chem.* **2010**, *22* (4), 2869–2878.

(87) Zhang, J. G.; Sun, X. W.; Ai, H. Levels and Congener Profiles of Polybrominated Diphenyl Ethers (PBDEs) in Primipara Breast Milk from Shenzhen and Exposure Risk for Breast-Fed Infants. *Journal of Environmental Monitoring* **2012**, *14*, 893–900.

(88) Sudaryanto, A.; Kajiwara, N.; Takahashi, S.; Muawanah; Tanabe, S. Geographical Distribution and Accumulation Features of PBDEs in Human Breast Milk from Indonesia. *Environ. Pollut.* **2008**, *151* (1), 130–138.

(89) UNEP. Global monitoring plan for Persistent Organic Pollutants. Third regional monitoring report Asia-Pacific region. http://chm.pops.int/implementation/globalmonitoringplan/monitoringreports/tabid/525/default.aspx (accessed 2023-03-30).

(90) Zhu, L.; Ma, B.; Li, J.; Wu, Y.; Gong, J. Distribution of Polybrominated Diphenyl Ethers in Breast Milk from North China: Implication of Exposure Pathways. *Chemosphere* **2009**, *74*, 1429–1434.

(91) Zhang, X.; Zhang, K.; Yang, D.; Ma, L.; Lei, B.; Zhang, X.; Zhou, J.; Fang, X.; Yu, Y. Polybrominated Biphenyl Ethers in Breast Milk and Infant Formula from Shanghai, China: Temporal Trends, Daily Intake, and Risk Assessment. *Sci. Total Environ.* **2014**, *497*, 508–515.

(92) Yin, S.; Guo, F.; Aamir, M.; Liu, Y.; Tang, M.; Liu, W. Multicenter Biomonitoring of Polybrominated Diphenyl Ethers (PBDEs) in Colostrum from China: Body Burden Profile and Risk Assessment. *Environ. Res.* **2019**, *179*, No. 108828.

(93) Yang, L.; Lu, Y.; Wang, L.; Chang, F.; Zhang, J.; Liu, Y. Levels and Profiles of Polybrominated Diphenyl Ethers in Breast Milk During Different Nursing Durations. *Bull. Environ. Contam. Toxicol.* **2016**, *97*, 510–516.

(94) Wang, Y.-F.; Wang, S.-L.; Chen, F.-A.; Chao, H. A.; Tsou, T.-C.; Shy, C.-G.; Päpke, O.; Kuo, Y.-M.; Chao, H.-R. Associations of Polybrominated Diphenyl Ethers (PBDEs) in Breast Milk and Dietary Habits and Demographic Factors in Taiwan. *Food Chem. Toxicol.* **2008**, *46*, 1925–1932.

(95) Tue, N. M.; Sudaryanto, A.; Minh, T. B.; Isobe, T.; Takahashi, S.; Viet, P. H.; Tanabe, S. Accumulation of Polychlorinated Biphenyls and Brominated Flame Retardants in Breast Milk from Women Living in Vietnamese E-Waste Recycling Sites. *Science of the Total Environment, The* **2010**, *408*, 2155–2162.

(96) Tsydenova, O. V.; Sudaryanto, A.; Kajiwara, N.; Kunisue, T.; Batoev, V. B.; Tanabe, S. Organohalogen Compounds in Human Breast Milk from Republic of Buryatia, Russia. *Environ. Pollut.* **2007**, *146*, 225–232.

(97) Sun, S.; Zhao, J.; Leng, J.; Wang, P.; Wang, Y.; Fukatsu, H.; Liu, D.; Liu, X.; Kayama, F. Levels of Dioxins and Polybrominated Diphenyl Ethers in Human Milk from Three Regions of Northern China and Potential Dietary Risk Factors. *Chemosphere* **2010**, *80*, 1151–1159.

(98) Sudaryanto, A.; Kajiwara, N.; Tsydenova, O. V.; Isobe, T.; Yu, H.; Takahashi, S.; Tanabe, S. Levels and Congener Specific Profiles of PBDEs in Human Breast Milk from China: Implication on Exposure Sources and Pathways. *Chemosphere* **2008**, *73* (10), 1661–1668.

(99) Shy, C.-G.; Huang, H.-L.; Chao, H.-R.; Chang-Chien, G.-P. Cord Blood Levels of Thyroid Hormones and IGF-1 Weakly Correlate with Breast Milk Levels of PBDEs in Taiwan. *Int. J. Hyg Environ. Health* **2012**, *215*, 345–351.

(100) Shi, Z.; Wang, Y.; Niu, P.; Wang, J.; Sun, Z.; Zhang, S.; Wu, Y. Concurrent Extraction, Clean-up, and Analysis of Polybrominated Diphenyl Ethers, Hexabromocyclododecane Isomers, and Tetrabromobisphenol A in Human Milk and Serum. *J. Sep Sci.* **2013**, *36* (20), 3402–3410.

(101) Shi, Z.; Jiao, Y.; Hu, Y.; Sun, Z.; Zhou, X.; Feng, J.; Li, J.; Wu, Y. Levels of Tetrabromobisphenol A, Hexabromocyclododecanes and Polybrominated Diphenyl Ethers in Human Milk from the General Population in Beijing, China. *Sci. Total Environ.* **2013**, *452–453*, 10–18.

(102) Shi, Z.; Zhang, L.; Zhao, Y.; Sun, Z.; Zhou, X.; Li, J.; Wu, Y. A National Survey of Tetrabromobisphenol-A, Hexabromocyclododecane and Decabrominated Diphenyl Ether in Human Milk from China: Occurrence and Exposure Assessment. *Sci. Total Environ.* **2017**, *599–600*, 237–245.

(103) Shen, H.; Ding, G.; Wu, Y.; Pan, G.; Zhou, X.; Han, J.; Li, J.; Wen, S. Polychlorinated Dibenzo-p-Dioxins/Furans (PCDD/Fs), Polychlorinated Biphenyls (PCBs), and Polybrominated Diphenyl Ethers (PBDEs) in Breast Milk from Zhejiang, China. *Environ. Int.* **2012**, *42*, 84–90.

(104) Ohta, S.; Ishizuka, D.; Nishimura, H.; Nakao, T.; Aozasa, O.; Shimidzu, Y.; Ochiai, F.; Kida, T.; Nishi, M.; Miyata, H. Comparison of Polybrominated Diphenyl Ethers in Fish, Vegetables, and Meats and Levels in Human Milk of Nursing Women in Japan. *Chemosphere* **2002**, *46*, 689–696.

(105) Ma, S.; Yu, Z.; Zhang, X.; Ren, G.; Peng, P.; Sheng, G.; Fu, J. Levels and Congener Profiles of Polybrominated Diphenyl Ethers (PBDEs) in Breast Milk from Shanghai: Implication for Exposure Route of Higher Brominated BDEs. *Environ. Int.* **2012**, *42*, 72–77.

(106) Lin, Y.; Feng, C.; Xu, Q.; Lu, D.; Qiu, X.; Jin, Y.; Wang, G.; Wang, D.; She, J.; Zhou, Z. A Validated Method for Rapid Determination of Dibenzo-p-Dioxins/Furans (PCDD/Fs), Polybrominated Diphenyl Ethers (PBDEs) and Polychlorinated Biphenyls (PCBs) in Human Milk: Focus on Utility of Tandem Solid Phase Extraction (SPE) Cleanup. *Anal Bioanal Chem.* **2016**, *408*, 4897–4906.

(107) Li, X.; Tian, Y.; Zhang, Y.; Ben, Y.; Lv, Q. Accumulation of Polybrominated Diphenyl Ethers in Breast Milk of Women from an E-Waste Recycling Center in China. *Journal of Environmental Sciences* **2017**, *52*, 305–313.

(108) Li, J.; Yu, H.; Zhao, Y.; Zhang, G.; Wu, Y. Levels of Polybrominated Diphenyl Ethers (PBDEs) in Breast Milk from Beijing, China. *Chemosphere* **2008**, *73*, 182–186.

(109) Li, J.; Ma, W.; Zhao, Y.; Jin, Y.; Xiao, X.; Ge, W.; Shi, H.; Zhang, Y. Lactational Exposure of Polybrominated Diphenyl Ethers and Its Association with Infant Developmental Measurements. *J. Hazard Mater.* **2020**, *388*, No. 122031.

(110) Leung, A. O. W.; Chan, J. K. Y.; Xing, G. H.; Xu, Y.; Wu, S. C.; Wong, C. K. C.; Leung, C. K. M.; Wong, M. H. Body Burdens of Polybrominated Diphenyl Ethers in Childbearing-Aged Women at an Intensive Electronic-Waste Recycling Site in China. *Environmental Science and Pollution Research* **2010**, *17*, 1300–1313.

(111) Lee, S.; Kim, S.; Kim, E.; Lee, I.-S.; Choi, G.; Kim, H.-J.; Park, J.; Jae Lee, J.; Choi, S.; Young Kim, S.; Kim, S.; Kim, S.; Choi, K.; Moon, H.-B. Polybrominated Diphenyl Ethers (PBDEs) in Breast Milk of Korea in 2011: Current Contamination, Time Course Variation, Influencing Factors and Health Risks. *Environ. Res.* **2013**, *126*, 76–83.

(112) Koh, T.-W.; Chih-Cheng Chen, S.; Chang-Chien, G.-P.; Lin, D.-Y.; Chen, F.-A.; Chao, H.-R. Breast-Milk Levels of Polybrominated Diphenyl Ether Flame Retardants in Relation to Women's Age and Pre-Pregnant Body Mass Index. *Int. J. Hyg Environ. Health* **2010**, *213*, 59–65.

(113) Kim, U.-J.; Lee, I.-S.; Kim, H. S.; Oh, J.-E. Monitoring of PBDEs Concentration in Umbilical Cord Blood and Breast Milk from Korean Population and Estimating the Effects of Various Parameters on Accumulation in Humans. *Chemosphere* **2011**, *85*, 487−493.

(114) Kim, T. H.; Bang, D. Y.; Lim, H. J.; Jin Won, A.; Ahn, M. Y.; Patra, N.; Chung, K. K.; Kwack, S. J.; Park, K. L.; Han, S. Y.; Choi, W. S.; Han, J. Y.; Lee, B. M.; Oh, J. E.; Yoon, J. H.; Lee, J.; Kim, H. S. Comparisons of Polybrominated Diphenyl Ethers Levels in Paired South Korean Cord Blood, Maternal Blood, and Breast Milk Samples. *Chemosphere* **2012**, *87* (1), 97−104.

(115) KAWASHIRO, Y.; FUKATA, H.; OMORI-INOUE, M.; KUBONOYA, K.; JOTAKI, T.; TAKIGAM, H.; SAKAI, S.-i.; MORI, C. Perinatal Exposure to Brominated Flame Retardants and Polychlorinated Biphenyls in Japan. *Endocr J.* **2008**, *55* (6), 1071−1084.

(116) Kao, C.-C.; Chen, C.-C.; Avelino, J. L.; Cortez, M. P.; Tayo, L. L.; Lin, Y.-H.; Tsai, M.-H.; Lin, C.-W.; Hsu, Y.-C.; Hsieh, L.-T.; Lin, C.; Wang, L.-C.; Yu, K.-L. J.; Chao, H.-R. Infants' Neurodevelopmental Effects of PM2.5 and Persistent Organohalogen Pollutants Exposure in Southern Taiwan. *Aerosol Air Qual Res.* **2019**, *19*, 2793−2803.

(117) Kang, C. S.; Lee, J.-H.; Kim, S.-K.; Lee, K.-T.; Lee, J. S.; Park, S.; Yun, H.; Kannan, K.; Yoo, Y. W.; Ha, Y.; Lee, S. W. Polybrominated Diphenyl Ethers and Synthetic Musks in Umbilical Cord Serum, Maternal Serum, and Breast Milk from Seoul, South Korea. *Chemosphere* **2010**, *80*, 116−122.

(118) Jin, J.; Wang, Y.; Yang, C.; Hu, J.; Liu, W.; Cui, J.; Tang, X. Polybrominated Diphenyl Ethers in the Serum and Breast Milk of the Resident Population from Production Area, China. *Environ. Int.* **2009**, *35*, 1048−1052.

(119) Hedley, A. J.; Hui, L. L.; Kypke, K.; Malisch, R.; van Leeuwen, F. X. R.; Moy, G.; Wong, T. W.; Nelson, E. A. S. Residues of Persistent Organic Pollutants (POPs) in Human Milk in Hong Kong. *Chemosphere* **2010**, *79* (3), 259−265.

(120) Haraguchi, K.; Koizumi, A.; Inoue, K.; Harada, K. H.; Hitomi, T.; Minata, M.; Tanabe, M.; Kato, Y.; Nishimura, E.; Yamamoto, Y.; Watanabe, T.; Takenaka, K.; Uehara, S.; Yang, H. R.; Kim, M. Y.; Moon, C. S.; Kim, H. S.; Wang, P.; Liu, A.; Hung, N. N. Levels and Regional Trends of Persistent Organochlorines and Polybrominated Diphenyl Ethers in Asian Breast Milk Demonstrate POPs Signatures Unique to Individual Countries. *Environ. Int.* **2009**, *35* (7), 1072−1079.

(121) Fujii, Y.; Ito, Y.; Harada, K. H.; Hitomi, T.; Koizumi, A.; Haraguchi, K. Regional Variation and Possible Sources of Brominated Contaminants in Breast Milk from Japan. *Environ. Pollut.* **2012**, *162*, 269−274.

(122) Eslami, B.; Koizumi, A.; Ohta, S.; Inoue, K.; Aozasa, O.; Harada, K.; Yoshinaga, T.; Date, C.; Fujii, S.; Fujimine, Y.; Hachiya, N.; Hirosawa, I.; Koda, S.; Kusaka, Y.; Murata, K.; Nakatsuka, H.; Omae, K.; Saito, N.; Shimbo, S.; Takenaka, K.; Takeshita, T.; Todoriki, H.; Wada, Y.; Watanabe, T.; Ikeda, M. Large-Scale Evaluation of the Current Level of Polybrominated Diphenyl Ethers (PBDEs) in Breast Milk from 13 Regions of Japan. *Chemosphere* **2006**, *63* (4), 554−561.

(123) Cui, C.; Tian, Y.; Zhang, L.; Gao, Y.; Jin, J.; Wang, P.; Ding, W.; Wang, X.; Shi, R.; Wang, Y. Polybrominated Diphenyl Ethers Exposure in Breast Milk in Shanghai, China: Levels, Influencing Factors and Potential Health Risk for Infants. *Science of The Total Environment* **2012**, *433*, 331−335.

(124) Chen, Z. J.; Liu, H. Y.; Cheng, Z.; Man, Y. B.; Zhang, K. S.; Wei, W.; Du, J.; Wong, M. H.; Wang, H. S. Polybrominated Diphenyl Ethers (PBDEs) in Human Samples of Mother-Newborn Pairs in South China and Their Placental Transfer Characteristics. *Environ. Int.* **2014**, *73*, 77−84.

(125) Chen, T.; Huang, M.; Li, J.; Li, J.; Shi, Z. Polybrominated Diphenyl Ethers and Novel Brominated Flame Retardants in Human Milk from the General Population in Beijing, China: Occurrence, Temporal Trends, Nursing Infants' Exposure and Risk Assessment. *Science of The Total Environment* **2019**, *689*, 278−286.

(126) Chen, M.-W.; Castillo, B. A. A.; Lin, D.-Y.; Chao, H.-R.; Tayo, L. L.; Gou, Y.-Y.; Chen, F.-A.; Huang, K.-L. Levels of PCDD/Fs, PBDEs, and PBDD/Fs in Breast Milk from Southern Taiwan. *Bull. Environ. Contam. Toxicol.* **2018**, *100*, 369−375.

(127) Chao, H. R.; Wang, S. L.; Lee, W. J.; Wang, Y. F.; Päpke, O. Levels of Polybrominated Diphenyl Ethers (PBDEs) in Breast Milk from Central Taiwan and Their Relation to Infant Birth Outcome and Maternal Menstruation Effects. *Environ. Int.* **2007**, *33* (2), 239−245.

(128) Chao, H.-R.; Tsou, T.-C.; Huang, H.-L.; Chang-Chien, G.-P. Levels of Breast Milk PBDEs From Southern Taiwan and Their Potential Impact on Neurodevelopment. *Pediatr. Res.* **2011**, *70* (6), 596−600.

(129) Chao, H. A.; Chen, S. C. C.; Chang, C. M.; Koh, T. W.; Chang-Chien, G. P.; Ouyang, E.; Lin, S. L.; Shy, C. G.; Chen, F. A.; Chao, H. R. Concentrations of Polybrominated Diphenyl Ethers in Breast Milk Correlated to Maternal Age, Education Level, and Occupational Exposure. *J. Hazard Mater.* **2010**, *175* (1−3), 492−500.

(130) Bi, X.; Qu, W.; Sheng, G.; Zhang, W.; Mai, B.; Chen, D.; Yu, L.; Fu, J. Polybrominated Diphenyl Ethers in South China Maternal and Fetal Blood and Breast Milk. *Environ. Pollut.* **2006**, *144* (3), 1024−1030.

(131) Akutsu, K.; Kitagawa, M.; Nakazawa, H.; Makino, T.; Iwazaki, K.; Oda, H.; Hori, S. Time-Trend (1973−2000) of Polybrominated Diphenyl Ethers in Japanese Mother's Milk. *Chemosphere* **2003**, *53* (6), 645−654.

(132) Fujii, Y.; Kato, Y.; Masuda, N.; Harada, K. H.; Koizumi, A.; Haraguchi, K. Contamination Trends and Factors Affecting the Transfer of Hexabromocyclododecane Diastereomers, Tetrabromobisphenol A, and 2,4,6-Tribromophenol to Breast Milk in Japan. *Environ. Pollut.* **2018**, *237*, 936−943.

(133) Devanathan, G.; Subramanian, A.; Sudaryanto, A.; Takahashi, S.; Isobe, T.; Tanabe, S. Brominated Flame Retardants and Polychlorinated Biphenyls in Human Breast Milk from Several Locations in India: Potential Contaminant Sources in a Municipal Dumping Site. *Environ. Int.* **2012**, *39* (1), 87−95.

(134) Malarvannan, G.; Isobe, T.; Covaci, A.; Prudente, M.; Tanabe, S. Accumulation of Brominated Flame Retardants and Polychlorinated Biphenyls in Human Breast Milk and Scalp Hair from the Philippines: Levels, Distribution and Profiles. *Sci. Total Environ.* **2013**, *442*, 366−379.

(135) Malarvannan, G.; Kunisue, T.; Isobe, T.; Sudaryanto, A.; Takahashi, S.; Prudente, M.; Subramanian, A.; Tanabe, S. Organohalogen Compounds in Human Breast Milk from Mothers Living in Payatas and Malate, the Philippines: Levels, Accumulation Kinetics and Infant Health Risk. *Environ. Pollut.* **2009**, *157* (6), 1924−1932.

(136) Tsai, M. H.; Chao, H. R.; Hsu, W. L.; Tsai, C. C.; Lin, C. W.; Chen, C. H. Analysis of Polybrominated Diphenyl Ethers and Lipid Composition in Human Breast Milk and Their Correlation with Infant Neurodevelopment. *Int. J. Environ. Res. Public Health* **2021**, *18* (21), 11501.

(137) Huang, M.; Li, J.; Xiao, Z.; Shi, Z. Tetrabromobisphenol A and Hexabromocyclododecane Isomers in Breast Milk from the General Population in Beijing, China: Contamination Levels, Temporal Trends, Nursing Infant's Daily Intake, and Risk Assessment. *Chemosphere* **2020**, *244*, No. 125524.

(138) Kakimoto, K.; Akutsu, K.; Konishi, Y.; Tanaka, Y. Time Trend of Hexabromocyclododecane in the Breast Milk of Japanese Women. *Chemosphere* **2008**, *71* (6), 1110−1114.

(139) Kakimoto, K.; Akutsu, K.; Konishi, Y.; Toriba, A.; Hayakawa, K. Hexabromocyclododecane Levels in Foodstuff and Human Breast Milk in Osaka, Japan. *Interdisciplinary Studies on Environmental Chemistry* **2012**, *6*, 253−259, DOI: 10.1021/bk-2016-1244.ch006.

(140) Lu, S.; Tan, Z.; Jiang, Y.; Wu, D.; Zhang, J.; Zhou, J.; Lin, X. Hexabromocyclododecanes in Breast Milk from Residents in Shenzhen, China: Implications for Infant Exposure. *Sci. Total Environ.* **2018**, *622−623*, 1090−1097.

(141) Shi, Z.; Wang, Y.; Niu, P.; Wang, J.; Sun, Z.; Zhang, S.; Wu, Y. Concurrent Extraction, Clean-up, and Analysis of Polybrominated Diphenyl Ethers, Hexabromocyclododecane Isomers, and Tetrabro-

mobisphenol A in Human Milk and Serum. *J. Sep. Sci.* **2013**, *36*, 3402−3410.

(142) Shi, Z.; Zhang, L.; Zhao, Y.; Sun, Z.; Zhou, X.; Li, J.; Wu, Y. A National Survey of Tetrabromobisphenol-A, Hexabromocyclododecane and Decabrominated Diphenyl Ether in Human Milk from China: Occurrence and Exposure Assessment. *Sci. Total Environ.* **2017**, *599−600*, 237−245.

(143) Shi, Z. X.; Wu, Y. N.; Li, J. G.; Zhao, Y. F.; Feng, J. F. Dietary Exposure Assessment of Chinese Adults and Nursing Infants to Tetrabromobisphenol-A and Hexabromocyclododecanes: Occurrence Measurements in Foods and Human Milk. *Environ. Sci. Technol.* **2009**, *43* (12), 4314−4319.

(144) Shi, Z.; Jiao, Y.; Hu, Y.; Sun, Z.; Zhou, X.; Feng, J.; Li, J.; Wu, Y. Levels of Tetrabromobisphenol A, Hexabromocyclododecanes and Polybrominated Diphenyl Ethers in Human Milk from the General Population in Beijing, China. *Sci. Total Environ.* **2013**, *452−453*, 10−18.

(145) Souza, M. C. O.; Devóz, P. P.; Ximenez, J. P. B.; Bocato, M. Z.; Rocha, B. A.; Barbosa, F. Potential Health Risk to Brazilian Infants by Polybrominated Diphenyl Ethers Exposure via Breast Milk Intake. *Int. J. Environ. Res. Public Health* **2022**, *19* (17), 11138.

(146) Antignac, J. P.; Main, K. M.; Virtanen, H. E.; Boquien, C. Y.; Marchand, P.; Venisseau, A.; Guiffard, I.; Bichon, E.; Wohlfahrt-Veje, C.; Legrand, A.; Boscher, C.; Skakkebæk, N. E.; Toppari, J.; Le Bizec, B. Country-Specific Chemical Signatures of Persistent Organic Pollutants (POPs) in Breast Milk of French, Danish and Finnish Women. *Environ. Pollut.* **2016**, *218*, 728−738.

(147) Aune, M.; Barregård, L.; Claesson, A.; Darnerud, P. O. *Resultatrapport till Miljöövervakningen: Organiska Miljögifter i Bröstmjölk Från Göteborg 2001*; Livsmedelsverket: 2002.

(148) Darnerud, P. O.; Lignell, S.; Aune, M.; Isaksson, M.; Cantillana, T.; Redeby, J.; Glynn, A. Time Trends of Polybrominated Diphenylether (PBDE) Congeners in Serum of Swedish Mothers and Comparisons to Breast Milk Data. *Environ. Res.* **2015**, *138*, 352−360.

(149) Eggesbø, M.; Thomsen, C.; Jørgensen, J. V.; Becher, G.; Øyvind Odland, J.; Longnecker, M. P. Associations between Brominated Flame Retardants in Human Milk and Thyroid-Stimulating Hormone (TSH) in Neonates. *Environ. Res.* **2011**, *111* (6), 737−743.

(150) Glynn, A.; Lignell, S.; Darnerud, P. O.; Aune, M.; Halldin Ankarberg, E.; Bergdahl, I. A.; Barregård, L.; Bensryd, I. Regional Differences in Levels of Chlorinated and Brominated Pollutants in Mother's Milk from Primiparous Women in Sweden. *Environ. Int.* **2011**, *37* (1), 71−79.

(151) Harrad, S.; Abdallah, M. A. E. Concentrations of Polybrominated Diphenyl Ethers, Hexabromocyclododecanes and Tetrabromobisphenol-A in Breast Milk from United Kingdom Women Do Not Decrease over Twelve Months of Lactation. *Environ. Sci. Technol.* **2015**, *49* (23), 13899−13903.

(152) Hoopmann, M.; Albrecht, U. V.; Gierden, E.; Huppmann, R.; Suchenwirth, R. Time Trends and Individual Characteristics Associated with Polybrominated Diphenyl Ethers in Breast Milk Samples 2006−2009 in Lower Saxony, Germany. *Int. J. Hyg Environ. Health* **2012**, *215* (3), 352−359.

(153) Lignell, S.; Aune, M.; Darnerud, P. O.; Cnattingius, S.; Glynn, A. Persistent Organochlorine and Organobromine Compounds in Mother's Milk from Sweden 1996−2006: Compound-Specific Temporal Trends. *Environ. Res.* **2009**, *109* (6), 760−767.

(154) Polder, A.; Thomsen, C.; Lindström, G.; Løken, K. B.; Skaare, J. U. Levels and Temporal Trends of Chlorinated Pesticides, Polychlorinated Biphenyls and Brominated Flame Retardants in Individual Human Breast Milk Samples from Northern and Southern Norway. *Chemosphere* **2008**, *73* (1), 14−23.

(155) Polder, A.; Gabrielsen, G. W.; Odland, J.; Savinova, T. N.; Tkachev, A.; Løken, K. B.; Skaare, J. U. Spatial and Temporal Changes of Chlorinated Pesticides, PCBs, Dioxins (PCDDs/PCDFs) and Brominated Flame Retardants in Human Breast Milk from Northern Russia. *Sci. Total Environ.* **2008**, *391* (1), 41−54.

(156) Pratt, I.; Anderson, W.; Crowley, D.; Daly, S.; Evans, R.; Fernandes, A.; Fitzgerald, M.; Geary, M.; Keane, D.; Morrison, J. J.; Reilly, A.; Tlustos, C. Brominated and Fluorinated Organic Pollutants in the Breast Milk of First-Time Irish Mothers: Is There a Relationship to Levels in Food? *Food Addit Contam Part A Chem. Anal Control Expo Risk Assess* **2013**, *30* (10), 1788−1798.

(157) Thomsen, C.; Liane, V. H.; Becher, G. Automated Solid-Phase Extraction for the Determination of Polybrominated Diphenyl Ethers and Polychlorinated Biphenyls in Serum-Application on Archived Norwegian Samples from 1977 to 2003. *J. Chromatogr B Analyt Technol. Biomed Life Sci.* **2007**, *846* (1−2), 252−263.

(158) UNEP. Global monitoring report for Persistent Organic Pollutants. Third regional monitoring report Western Europe and Others Group (WEOG) region. https://chm.pops.int/implementation/globalmonitoringplan/monitoringreports/tabid/525/default.aspx (accessed 2023-03-30).

(159) UNEP. Global monitoring report for Persistent Organic Pollutants. Third regional monitoring report Central and Eastern European region. 2021. http://chm.pops.int/implementation/globalmonitoringplan/monitoringreports/tabid/525/default.aspx (accessed 2023-03-30).

(160) Čechová, E.; Vojta, Š.; Kukučka, P.; Kočan, A.; Trnovec, T.; Murínová, L. P.; de Cock, M.; van de Bor, M.; Askevold, J.; Eggesbø, M.; Scheringer, M. Legacy and Alternative Halogenated Flame Retardants in Human Milk in Europe: Implications for Children's Health. *Environ. Int.* **2017**, *108*, 137−145.

(161) Fürst, P. Dioxins, Polychlorinated Biphenyls and Other Organohalogen Compounds in Human Milk. *Mol. Nutr Food Res.* **2006**, *50* (10), 922−933.

(162) Strandman, T.; Koistinen, J.; Vartiainen, T. Polybrominated Flame Retardants (PBDEs) in Placenta and Human Milk. *Organohalogen Compd.* **2000**, *47*, 61−64.

(163) *AMAP Assessment 2002 : The Influence of Global Change on Contaminant Pathways to within, and from the Arctic*; Arctic Monitoring and Assessment Programme, 2003.

(164) Zeilmaker, M.; Moermond, C.; Brandon, E.; Hoogerhuis, P.; Razenberg, L.; Janssen, M. *Persistent Organic Pollutants in Human Milk* RIVM Letter Report 2020−0025; Rijksinstituut voor Volksgezondheid en Milieu RIVM, 2020, DOI: 10.21945/RIVM-2020-0025.

(165) Weber, H.; Heseker, H. Analysis of Polybrominated Diphenyl Ethers in Breast Milk of German Mothers - Results of a Pilot Study. *Fresenius Environmental Bulletin and Advances in Food Sciences* **2004**, *13* (4), 356−360.

(166) Thomsen, C.; Haug, L. S.; Stigum, H.; Frøshaug, M.; Broadwell, S. L.; Becher, G. Changes in Concentrations of Perfluorinated Compounds, Polybrominated Diphenyl Ethers, and Polychlorinated Biphenyls in Norwegian Breast-Milk during Twelve Months of Lactation. *Environ. Sci. Technol.* **2010**, *44*, 9550−9556.

(167) Sahlström, L. M. O.; Sellström, U.; de Wit, C. A.; Lignell, S.; Darnerud, P. O. Estimated Intakes of Brominated Flame Retardants via Diet and Dust Compared to Internal Concentrations in a Swedish Mother-Toddler Cohort. *Int. J. Hyg Environ. Health* **2015**, *218* (4), 422−432.

(168) Raab, U.; Preiss, U.; Albrecht, M.; Shahin, N.; Parlar, H.; Fromme, H. Concentrations of Polybrominated Diphenyl Ethers, Organochlorine Compounds and Nitro Musks in Mother's Milk from Germany (Bavaria). *Chemosphere* **2008**, *72* (1), 87−94.

(169) Ozcan, S.; Tor, A.; Aydin, M. E. Levels of Organohalogenated Pollutants in Human Milk Samples from Konya City, Turkey. *Clean (Weinheim)* **2011**, *39* (10), 978−983.

(170) Meironyté Guvenius, D.; Aronsson, A.; Ekman-Ordeberg, G.; Bergman, Å.; Norén, K. Human Prenatal and Postnatal Exposure to Polybrominated Diphenyl Ethers, Polychlorinated Biphenyls, Polychlorobiphenylols, and Pentachlorophenol. *Environ. Health Perspect* **2003**, *111* (9), 1235−1241.

(171) Main, K. M.; Kiviranta, H.; Virtanen, H. E.; Sundqvist, E.; Tuomisto, J. T.; Tuomisto, J.; Vartiainen, T.; Skakkebæk, N. E.; Toppari, J. Flame Retardants in Placenta and Breast Milk and

Cryptorchidism in Newborn Boys. *Environ. Health Perspect* **2007**, *115* (10), 1519−1526.

(172) Lignell, S.; Aune, M.; Darnerud, P. O.; Stridsberg, M.; Hanberg, A.; Larsson, S. C; Glynn, A. Maternal Body Burdens of PCDD/Fs and PBDEs Are Associated with Maternal Serum Levels of Thyroid Hormones in Early Pregnancy: A Cross-Sectional Study. *Environmental Health* **2016**, *15*, 1−11.

(173) Li, Z.-M.; Albrecht, M.; Fromme, H.; Schramm, K.-W.; De Angelis, M. Persistent Organic Pollutants in Human Breast Milk and Associations with Maternal Thyroid Hormone Homeostasis. *Environ. Sci. Technol.* **2020**, *54*, 1111−1119.

(174) Lacorte, S.; Ikonomou, M. G. Occurrence and Congener Specific Profiles of Polybrominated Diphenyl Ethers and Their Hydroxylated and Methoxylated Derivatives in Breast Milk from Catalonia. *Chemosphere* **2009**, *74* (3), 412−420.

(175) Kazda, R.; Hajšlová, J.; Poustka, J.; Čajka, T. Determination of Polybrominated Diphenyl Ethers in Human Milk Samples in the Czech Republic: Comparative Study of Negative Chemical Ionisation Mass Spectrometry and Time-of-Flight High-Resolution Mass Spectrometry. *Anal. Chim. Acta* **2004**, *520* (1−2), 237−243.

(176) Kalantzi, O. I.; Martin, F. L.; Thomas, G. O.; Alcock, R. E.; Tang, H. R.; Drury, S. C.; Carmichael, P. L.; Nicholson, J. K.; Jones, K. C. Different Levels of Polybrominated Diphenyl Ethers (PBDEs) and Chlorinated Compounds in Breast Milk from Two U.K. Regions. *Environ. Health Perspect* **2004**, *112* (10), 1085−1091.

(177) Jaraczewska, K.; Lulek, J.; Covaci, A.; Voorspoels, S.; Kaluba-Skotarczak, A.; Drews, K.; Schepens, P. Distribution of Polychlorinated Biphenyls, Organochlorine Pesticides and Polybrominated Diphenyl Ethers in Human Umbilical Cord Serum, Maternal Serum and Milk from Wielkopolska Region, Poland. *Sci. Total Environ.* **2006**, *372* (1), 20−31.

(178) Jakobsson, K., Athanasiadou, M., Christiansson, A., Athanassiadis, I., Bergman, Å, Hagmar, L. *Polybrominated diphenyl ethers (PBDEs) in serum from Swedish men 1988−2002 : A longitudinal study*. Lunds universitet, 2005, https://www.diva-portal.org/smash/record.jsf?pid=diva2%3A657721&dswid=-8348.

(179) Ingelido, A. M.; Ballard, T.; Dellatte, E.; di Domenico, A.; Ferri, F.; Fulgenzi, A. R.; Herrmann, T.; Iacovella, N.; Miniero, R.; Päpke, O.; Porpora, M. G.; Felip, E. De. Polychlorinated Biphenyls (PCBs) and Polybrominated Diphenyl Ethers (PBDEs) in Milk from Italian Women Living in Rome and Venice. *Chemosphere* **2007**, *67* (9), S301−S306.

(180) Hernik, A.; Struciński, P.; Buckley, B. T.; Góralczyk, K.; Czaja, K.; Korcz, W.; Matuszak, M.; Łyczewska, M.; Minorczyk, M.; Liszewska, M.; Ludwicki, J. K. Relationship between Paired Cord Blood and Milk POPs Levels as a Tool for Assessing Perinatal Exposure, a Pilot Study. *Human and Ecological Risk Assessment: An International Journal* **2016**, *22* (7), 1456−1468, DOI: 10.1080/10807039.2016.1185688.

(181) Hernik, A.; Goralczyk, K.; Strucinski, P.; Czaja, K.; Kucharska, A.; Korcz, W.; Snopczynski, T.; Ludwicki, J. K. Polybrominated Diphenyl Ethers, Polychlorinated Biphenyls and Organochlorine Pesticides in Human Milok as Markers of Environmental Exposure to These Compounds. *Ann. Agric. Environ. Med.* **2011**, *18* (1), 113−118.

(182) Gómara, B.; Herrero, L.; Pacepavicius, G.; Ohta, S.; Alaee, M.; González, M. J. Occurrence of Co-Planar Polybrominated/Chlorinated Biphenyls (PXBs), Polybrominated Diphenyl Ethers (PBDEs) and Polychlorinated Biphenyls (PCBs) in Breast Milk of Women from Spain. *Chemosphere* **2011**, *83* (6), 799−805.

(183) Gómara, B.; Herrero, L.; Ramos, J. J.; Mateo, J. R.; Fernandez, M. A.; Garcia, J. F.; Gonzalez, M. J. Distribution of Polybrominated Diphenyl Ethers in Human Umbilical Cord Serum, Paternal Serum, Maternal Serum, Placentas, and Breast Milk from Madrid Population, Spain. *Environ. Sci. Technol.* **2007**, *41* (20), 6961−6968.

(184) Fängström, B.; Strid, A.; Grandjean, P.; Weihe, P.; Bergman, Å. Environmental Health: A Global Access Science Source A Retrospective Study of PBDEs and PCBs in Human Milk from the Faroe Islands. *Environmental health* **2005**, *4*, 1−9.

(185) Fängström, B.; Athanassiadis, I.; Odsjö, T.; Norøn, K.; Bergman, Š. Temporal Trends of Polybrominated Diphenyl Ethers and Hexabromocyclododecane in Milk from Stockholm Mothers, 1980−2004. *Mol. Nutr. Food Res.* **2008**, *52*, 187−193.

(186) Erdoğrul, Ö.; Covaci, A.; Kurtul, N.; Schepens, P. Levels of Organohalogenated Persistent Pollutants in Human Milk from Kahramanmaraş Region, Turkey. *Environ. Int.* **2004**, *30* (5), 659−666.

(187) Dimitriadou, L.; Malarvannan, G.; Covaci, A.; Iossifidou, E.; Tzafettas, J.; Zournatzi-Koiou, V.; Kalantzi, O. I. Levels and Profiles of Brominated and Chlorinated Contaminants in Human Breast Milk from Thessaloniki, Greece. *Sci. Total Environ.* **2016**, *539*, 350−358.

(188) Croes, K.; Colles, A.; Koppen, G.; Govarts, E.; Bruckers, L.; Van de Mieroop, E.; Nelen, V.; Covaci, A.; Dirtu, A. C.; Thomsen, C.; Haug, L. S.; Becher, G.; Mampaey, M.; Schoeters, G.; Van Larebeke, N.; Baeyens, W. Persistent Organic Pollutants (POPs) in Human Milk: A Biomonitoring Study in Rural Areas of Flanders (Belgium). *Chemosphere* **2012**, *89* (8), 988−994.

(189) Colles, A.; Koppen, G.; Hanot, V.; Nelen, V.; Dewolf, M. C.; Noël, E.; Malisch, R.; Kotz, A.; Kypke, K.; Biot, P.; Vinkx, C.; Schoeters, G. Fourth WHO-Coordinated Survey of Human Milk for Persistent Organic Pollutants (POPs): Belgian Results. *Chemosphere* **2008**, *73* (6), 907−914.

(190) Chovancová, J.; Čonka, K.; Kočan, A.; Sejáková, Z. S. PCDD, PCDF, PCB and PBDE Concentrations in Breast Milk of Mothers Residing in Selected Areas of Slovakia. *Chemosphere* **2011**, *83* (10), 1383−1390.

(191) Bramwell, L.; Fernandes, A.; Rose, M.; Harrad, S.; Pless-Mulloli, T. PBDEs and PBBs in Human Serum and Breast Milk from Cohabiting UK Couples. *Chemosphere* **2014**, *116*, 67−74.

(192) Bordajandi, L. R.; Abad, E.; González, M. J. Occurrence of PCBs, PCDD/Fs, PBDEs and DDTs in Spanish Breast Milk: Enantiomeric Fraction of Chiral PCBs. *Chemosphere* **2008**, *70* (4), 567−575.

(193) Björklund, J. A.; Sellström, U.; de Wit, C. A.; Aune, M.; Lignell, S.; Darnerud, P. O. Comparisons of Polybrominated Diphenyl Ether and Hexabromocyclododecane Concentrations in Dust Collected with Two Sampling Methods and Matched Breast Milk Samples. *Indoor Air* **2012**, *22* (4), 279−288.

(194) Bergman, Å.; Hovander, L.; Sundström, M.; Athanassiadis, I.; Athanasiadou, M.; Sällsten, G.; Bignert, A.; Nyberg, E. *Insamling Och Kemisk Analys Av Miljöföroreningar i Svensk Modersmjölk; Resultat från 2008−2010. Rapport till Naturvårdsverket 2010−11−24* **2008**, 20.

(195) Athanasiadou, M.; Bergman, Å. *Insamling Av Bröstmjölksprover Från Stockholm, Göteborg, Lund Och Umeå Samt Analyser Av Insamlade Bröstmjölksprover Resultat Från 2007 Års Arbete*; Department of Environmental Chemistry, Stockholm University: 2008.

(196) Alivernini, S.; Battistelli, C. L.; Turrio-Baldassarri, L. Human Milk as a Vector and an Indicator of Exposure to PCBs and PBDEs: Temporal Trend of Samples Collected in Rome. *Bull. Environ. Contam. Toxicol.* **2011**, *87*, 21−25.

(197) Abdallah, M. A. E.; Harrad, S. Polybrominated Diphenyl Ethers in UK Human Milk: Implications for Infant Exposure and Relationship to External Exposure. *Environ. Int.* **2014**, *63*, 130−136.

(198) Desalegn, A. A.; Collet, B.; Iszatt, N.; Stigum, H.; Jensen, T. K.; Jonker, L.; Besselink, H.; van der Burg, B.; Eggesbø, M. Aryl Hydrocarbon Receptor Activity in Human Breast Milk and Cryptorchidism: A Case-Control Study within the Prospective Norwegian HUMIS Cohort. *Environ. Res.* **2022**, *214*, No. 113861.

(199) Fromme, H.; Fuchs, V.; Albrecht, M.; Aschenbrenner, B.; Röhl, C.; Janitzki, N.; Herber-Jonat, S.; Wöckner, M.; Völkel, W.; Flemmer, A. W.; Schober, W. Polychlorinated Dioxins and Dibenzofurans (PCDD/F), Polybrominated Dioxins and Dibenzofurans (PBDD/F), Polychlorinated Biphenyls (PCB), Polybrominated Diphenyl Ethers (PBDE), and per- and Polyfluoroalkyl Substances (PFAS) in German Breast Milk Samples (LUPE 8). *Sci. Total Environ.* **2022**, *825*, No. 154066.

(200) Rovira, J.; Martínez, M. Á.; Mari, M.; Cunha, S. C.; Fernandes, J. O.; Marmelo, I.; Marques, A.; Haug, L. S.; Thomsen, C.; Nadal, M.; Domingo, J. L.; Schuhmacher, M. Mixture of Environmental

Pollutants in Breast Milk from a Spanish Cohort of Nursing Mothers. *Environ. Int.* **2022**, *166*, No. 107375.

(201) Roosens, L.; D'Hollander, W.; Bervoets, L.; Reynders, H.; Van Campenhout, K.; Cornelis, C.; Van Den Heuvel, R.; Koppen, G.; Covaci, A. Brominated Flame Retardants and Perfluorinated Chemicals, Two Groups of Persistent Contaminants in Belgian Human Blood and Milk. *Environ. Pollut.* **2010**, *158* (8), 2546−2552.

(202) Stoffmonographie Für 1,2,5,6,9,10-Hexabromcyclododecan (HBCDD) - HBM-Werte Für HBCDD Im Fettanteil Der Muttermilch Oder Des Blutplasmas. *Bundesgesundheitsblatt Gesundheitsforschung Gesundheitsschutz.* **2015**, *58*, 889−907, .

(203) Lankova, D.; Lacina, O.; Pulkrabova, J.; Hajslova, J. The Determination of Perfluoroalkyl Substances, Brominated Flame Retardants and Their Metabolites in Human Breast Milk and Infant Formula. *Talanta* **2013**, *117*, 318−325.

(204) Thomsen, C.; Stigum, H.; Frøshaug, M.; Broadwell, S. L.; Becher, G.; Eggesbø, M. Determinants of Brominated Flame Retardants in Breast Milk from a Large Scale Norwegian Study. *Environ. Int.* **2010**, *36* (1), 68−74.

(205) Butryn, D. M.; Gross, M. S.; Chi, L. H.; Schecter, A.; Olson, J. R.; Aga, D. S. One-Shot" Analysis of Polybrominated Diphenyl Ethers and Their Hydroxylated and Methoxylated Analogs in Human Breast Milk and Serum Using Gas Chromatography-Tandem Mass Spectrometry. *Anal. Chim. Acta* **2015**, *892*, 140−147.

(206) Butryn, D. M.; Chi, L. H.; Gross, M. S.; McGarrigle, B.; Schecter, A.; Olson, J. R.; Aga, D. S. Retention of Polybrominated Diphenyl Ethers and Hydroxylated Metabolites in Paired Human Serum and Milk in Relation to CYP2B6 Genotype. *J. Hazard Mater.* **2020**, *386*, No. 121904.

(207) Daniels, J. L.; Pan, I. J.; Jones, R.; Anderson, S.; Patterson, D. G.; Needham, L. L.; Sjödin, A. Individual Characteristics Associated with PBDE Levels in U.S. Human Milk Samples. *Environ. Health Perspect* **2010**, *118* (1), 155−160.

(208) Dunn, R. L.; Huwe, J. K.; Carey, G. B. Biomonitoring Polybrominated Diphenyl Ethers in Human Milk as a Function of Environment, Dietary Intake, and Demographics in New Hampshire. *Chemosphere* **2010**, *80* (10), 1175−1182.

(209) Guo, W.; Holden, A.; Smith, S. C.; Gephart, R.; Petreas, M.; Park, J. S. PBDE Levels in Breast Milk Are Decreasing in California. *Chemosphere* **2016**, *150*, 505−513.

(210) Hartle, J. C.; Cohen, R. S.; Sakamoto, P.; Barr, D. B.; Carmichael, S. L. Chemical Contaminants in Raw and Pasteurized Human Milk. *Journal of Human Lactation* **2018**, *34* (2), 340−349.

(211) Hoffman, K.; Adgent, M.; Goldman, B. D.; Sjödin, A.; Daniels, J. L. Lactational Exposure to Polybrominated Diphenyl Ethers and Its Relation to Social and Emotional Development among Toddlers. *Environ. Health Perspect* **2012**, *120* (10), 1438−1442.

(212) Hoffman, K.; Mendez, M.; Siega-Riz, A. M.; Herring, A. H.; Sjödin, A.; Daniels, J. L. Lactational Exposure to Polybrominated Diphenyl Ethers and Its Relation to Early Childhood Anthropometric Measurements. *Environ. Health Perspect* **2016**, *124* (10), 1656−1661.

(213) Johnson-Restrepo, B.; Addink, R.; Wong, C.; Arcaro, K.; Kannan, K. Polybrominated Diphenyl Ethers and Organochlorine Pesticides in Human Breast Milk from Massachusetts, USA. *Journal of Environmental Monitoring* **2007**, *9*, 1205−1212.

(214) LaKind, J. S.; Berlin, C. M.; Sjödin, A.; Turner, W.; Wang, R. Y.; Needham, L. L.; Paul, I. M.; Stokes, J. L.; Naiman, D. Q.; Patterson, D. G. Do Human Milk Concentrations of Persistent Organic Chemicals Really Decline during Lactation? Chemical Concentrations during Lactation and Milk/Serum Partitioning. *Environ. Health Perspect* **2009**, *117* (10), 1625−1631.

(215) Marchitti, S. A.; Fenton, S. E.; Mendola, P.; Kenneke, J. F.; Hines, E. P. Polybrominated Diphenyl Ethers in Human Milk and Serum from the U.S. EPA MAMA Study: Modeled Predictions of Infant Exposure and Considerations for Risk Assessment. *Environ. Health Perspect* **2017**, *125* (4), 706−713.

(216) Schecter, A.; Pavuk, M.; Päpke, O.; Ryan, J. J.; Birnbaum, L.; Rosen, R. Polybrominated Diphenyl Ethers (PBDEs) in U.S. Mothers' Milk. *Environ. Health Perspect* **2003**, *111* (14), 1723−1729.

(217) She, J.; Holden, A.; Sharp, M.; Tanner, M.; Williams-Derry, C.; Hooper, K. Polybrominated Diphenyl Ethers (PBDEs) and Polychlorinated Biphenyls (PCBs) in Breast Milk from the Pacific Northwest. *Chemosphere* **2007**, *67* (9), S307.

(218) Sjödin, A.; Jones, R. S.; Focant, J. F.; Lapeza, C.; Wang, R. Y.; McGahee, E. E.; Zhang, Y.; Turner, W. E.; Slazyk, B.; Needham, L. L.; Patterson, D. G. Retrospective Time-Trend Study of Polybrominated Diphenyl Ether and Polybrominated and Polychlorinated Biphenyl Levels in Human Serum from the United States. *Environ. Health Perspect* **2004**, *112* (6), 654−658.

(219) Zhou, S. N.; Buchar, A.; Siddique, S.; Takser, L.; Abdelouahab, N.; Zhu, J. Measurements of Selected Brominated Flame Retardants in Nursing Women: Implications for Human Exposure. *Environ. Sci. Technol.* **2014**, *48*, 8873−8880.

(220) Wu, N.; Herrmann, T.; Paepke, O.; Tickner, J.; Hale, R.; Harvey, E.; La Guardia, M.; McClean, M. D.; Webster, T. F. Human Exposure to PBDEs: Associations of PBDE Body Burdens with Food Consumption and House Dust Concentrations. *Environ. Sci. Technol.* **2007**, *41* (5), 1584−1589.

(221) Park, J.-S.; She, J.; Holden, A.; Sharp, M.; Gephart, R.; Souders-Mason, G.; Zhang, V.; Chow, J.; Leslie, B.; Hooper, K. High Postnatal Exposures to Polybrominated Diphenyl Ethers (PBDEs) and Polychlorinated Biphenyls (PCBs) via Breast Milk in California: Does BDE-209 Transfer to Breast Milk? *Environ. Sci. Technol.* **2011**, *45*, 4579−4585.

(222) Martínez, C.; Martínez Arroyo, A.; Barrientos Alemán, D.; Gavilán García, A.; Caba, M.; Calderón Garcidueñas, A. L.; Mora, A.; Zenteno, E. Persistent Organic Compounds in Human Milk and Evaluation of the Effectiveness of the Stockholm Convention in Mexico. *Environmental Advances* **2022**, *8*, No. 100190.

(223) Mannetje, A.; Coakley, J.; Bridgen, P.; Brooks, C.; Harrad, S.; Smith, A. H.; Pearce, N.; Douwes, J. Current Concentrations, Temporal Trends and Determinants of Persistent Organic Pollutants in Breast Milk of New Zealand Women. *Sci. Total Environ.* **2013**, *458−460*, 399−407.

(224) Coakley, J. D.; Harrad, S. J.; Goosey, E.; Ali, N.; Dirtu, A. C.; Van den Eede, N.; Covaci, A.; Douwes, J.; Mannetje, A. Concentrations of Polybrominated Diphenyl Ethers in Matched Samples of Indoor Dust and Breast Milk in New Zealand. *Environ. Int.* **2013**, *59*, 255−261.

(225) Chen, Y.; Wang, X.; Li, Y.; Toms, L. M. L.; Gallen, M.; Hearn, L.; Aylward, L. L.; McLachlan, M. S.; Sly, P. D.; Mueller, J. F. Persistent Organic Pollutants in Matched Breast Milk and Infant Faeces Samples. *Chemosphere* **2015**, *118* (1), 309−314.

(226) Toms, L. M. L.; Guerra, P.; Eljarrat, E.; Barceló, D.; Harden, F. A.; Hobson, P.; Sjodin, A.; Ryan, E.; Mueller, J. F. Brominated Flame Retardants in the Australian Population: 1993−2009. *Chemosphere* **2012**, *89* (4), 398−403.

(227) Toms, L. M. L.; Hearn, L.; Kennedy, K.; Harden, F.; Bartkow, M.; Temme, C.; Mueller, J. F. Concentrations of Polybrominated Diphenyl Ethers (PBDEs) in Matched Samples of Human Milk, Dust and Indoor Air. *Environ. Int.* **2009**, *35* (6), 864−869.

(228) Toms, L. M. L.; Harden, F. A.; Symons, R. K.; Burniston, D.; Fürst, P.; Müller, J. F. Polybrominated Diphenyl Ethers (PBDEs) in Human Milk from Australia. *Chemosphere* **2007**, *68* (5), 797−803.

(229) Hites, R. A. Break Point Analyses of Human or Environmental Temporal Trends of POPs. *Science of The Total Environment* **2019**, *664*, 518−521.

(230) Heck, K. E.; Braveman, P.; Cubbin, C.; Chávez, G. F.; Kiely, J. L. Socioeconomic Status and Breastfeeding Initiation among California Mothers. *Public Health Reports* **2006**, *121* (1), 51−59.

(231) Chao, H. A.; Chen, S. C. C.; Chang, C. M.; Koh, T. W.; Chang-Chien, G. P.; Ouyang, E.; Lin, S. L.; Shy, C. G.; Chen, F. A.; Chao, H. R. Concentrations of Polybrominated Diphenyl Ethers in Breast Milk Correlated to Maternal Age, Education Level, and Occupational Exposure. *J. Hazard Mater.* **2010**, *175* (1−3), 492−500.

(232) Malisch, K.; Malisch, R.; Fiedler, H. Guidelines for Organization, Sampling and Analysis of Human Milk on Persistent Organic Pollutants. *Chemicals Branch Division of Technology, Industry*

and Economics (DTIE) United Nations Environment Programme (UNEP) **2017**, 39.

(233) Hooper, K.; She, J.; Sharp, M.; Chow, J.; Jewell, N.; Gephart, R.; Holden, A. Depuration of Polybrominated Diphenyl Ethers (PBDEs) and Polychlorinated Biphenyls (PCBs) in Breast Milk from California First-Time Mothers (Primiparae). *Environ. Health Perspect* **2007**, *115* (9), 1271−1275.

(234) Li, L.; Hoang, C.; Arnot, J. A.; Wania, F. Clarifying Temporal Trend Variability in Human Biomonitoring of Polybrominated Diphenyl Ethers through Mechanistic Modeling. *Environ. Sci. Technol.* **2020**, *54*, 166−175.

(235) Drage, D. S.; Harden, F. A.; Jeffery, T.; Mueller, J. F.; Hobson, P.; Toms, L. M. L. Human Biomonitoring in Australian Children: Brominated Flame Retardants Decrease from 2006 to 2015. *Environ. Int.* **2019**, *122*, 363−368.

(236) Wong, F.; Cousins, I. T.; MacLeod, M. Bounding Uncertainties in Intrinsic Human Elimination Half-Lives and Intake of Polybrominated Diphenyl Ethers in the North American Population. *Environ. Int.* **2013**, *59*, 168−174.

(237) Trudel, D.; Scheringer, M.; von Goetz, N.; Hungerbuhler, K. Total Consumer Exposure to Polybrominated Diphenyl Ethers in North America and Europe. *Environ. Sci. Technol.* **2011**, *45*, 2391−2397.

(238) McGrath, T. J.; Morrison, P. D.; Ball, A. S.; Clarke, B. O. Concentrations of Legacy and Novel Brominated Flame Retardants in Indoor Dust in Melbourne, Australia: An Assessment of Human Exposure. *Environ. Int.* **2018**, *113*, 191−201.

(239) Pietroń, W. J.; Małagocki, P. Quantification of Polybrominated Diphenyl Ethers (PBDEs) in Food. A Review. *Talanta* **2017**, *167*, 411−427.

(240) Stapleton, H. M. Instrumental Methods and Challenges in Quantifying Polybrominated Diphenyl Ethers in Environmental Extracts: A Review. *Anal. Bioanal. Chem.* **2006**, *386*, 807−817.

(241) United States Department of State. Stockholm Convention on Persistent Organic Pollutants. https://www.state.gov/key-topics-office-of-environmental-quality-and-transboundary-issues/stockholm-convention-on-persistent-organic-pollutants/ (accessed 2022-12-05).

(242) Undeman, E.; Johansson, J. Polybrominated diphenyl ethers (PBDEs) in the Baltic Sea − Sources, transport routes and trends. *Helcom Baltic Sea Environment Proceedings* **2020**, *172*, 1−22.

(243) The 16 New POPs. In *Stockholm Convention on Persistent Organic Pollutants (POPs), UN Environment*; Stockholm Convention Secretariat United Nations Environment, 2017; pp 1−25.

(244) UNEP. Eighth Meeting of the Conference of the Parties to the Stockholm Convention. http://chm.pops.int/theconvention/conferenceoftheparties/meetings/cop8/tabid/5309/default.aspx (accessed 2023-01-06).

(245) Thuresson, K.; Höglund, P.; Hagmar, L.; Sjödin, A.; Bergman, Å.; Jakobsson, K. Apparent Half-Lives of Hepta- to Decabrominated Diphenyl Ethers in Human Serum as Determined in Occupationally Exposed Workers. *Environ. Health Perspect* **2006**, *114* (2), 176−181.

(246) Kalachova, K.; Hradkova, P.; Lankova, D.; Hajslova, J.; Pulkrabova, J. Occurrence of Brominated Flame Retardants in Household and Car Dust from the Czech Republic. *Sci. Total Environ.* **2012**, *441*, 182−193.

(247) Guerra, P.; Alaee, M.; Jiménez, B.; Pacepavicius, G.; Marvin, C.; MacInnis, G.; Eljarrat, E.; Barceló, D.; Champoux, L.; Fernie, K. Emerging and Historical Brominated Flame Retardants in Peregrine Falcon (Falco Peregrinus) Eggs from Canada and Spain. *Environ. Int.* **2012**, *40* (1), 179−186.

(248) Dodson, R. E.; Van Den Eede, N.; Covaci, A.; Perovich, L. J.; Brody, J. G.; Rudel, R. A. Urinary Biomonitoring of Phosphate Flame Retardants: Levels in California Adults and Recommendations for Future Studies. *Environ. Sci. Technol.* **2014**, *48* (23), 13625−13633.

(249) Wu, X.; Bennett, D. H.; Moran, R. E.; Sjödin, A.; Jones, R. S.; Tancredi, D. J.; Tulve, N. S.; Clifton, M. S.; Colón, M.; Weathers, W.; Hertz-Picciotto, I. Polybrominated Diphenyl Ether Serum Concentrations in a Californian Population of Children, Their Parents, and Older Adults: An Exposure Assessment Study. *Environ. Health* **2015**, *14* (1), DOI: 10.1186/s12940-015-0002-2.

(250) Parry, E.; Zota, A. R.; Park, J. S.; Woodruff, T. J. Polybrominated Diphenyl Ethers (PBDEs) and Hydroxylated PBDE Metabolites (OH-PBDEs): A Six-Year Temporal Trend in Northern California Pregnant Women. *Chemosphere* **2018**, *195*, 777−783.

(251) Sharkey, M.; Harrad, S.; Abou-Elwafa Abdallah, M.; Drage, D. S.; Berresheim, H. Phasing-out of Legacy Brominated Flame Retardants: The UNEP Stockholm Convention and Other Legislative Action Worldwide. *Environ. Int.* **2020**, *144*, No. 106041.

(252) Environmental Protection Agency. Significant New Use and Test Rules: Certain Polybrominated Diphenylethers. *Federal Register* **2012**, *77* (63), 19862−19899.

(253) Environmental Protection Agency. Decabromodiphenyl Ether (DecaBDE); Regulation of Persistent, Bioaccumulative, and Toxic Chemicals Under TSCA Section 6(h). *Federal Register* **2021**, *86* (3), 880−894.

(254) National Industrial Chemicals Notification and Assessment Scheme. *Interim Public Health Risk Assessment of Certain PBDE Congeners* **2007**, 62.

(255) Li, L.; Wania, F. Elucidating the Variability in the Hexabromocyclododecane Diastereomer Profile in the Global Environment. *Environ. Sci. Technol.* **2018**, *52* (18), 10532−10542.

(256) Ali, N.; Dirtu, A. C.; Eede, N. V. d.; Goosey, E.; Harrad, S.; Neels, H.; Mannetje, A.; Coakley, J.; Douwes, J.; Covaci, A. Occurrence of Alternative Flame Retardants in Indoor Dust from New Zealand: Indoor Sources and Human Exposure Assessment. *Chemosphere* **2012**, *88* (11), 1276−1282.

(257) Hennebert, P. Concentrations of Brominated Flame Retardants in Plastics of Electrical and Electronic Equipment, Vehicles, Construction, Textiles and Non-Food Packaging: A Review of Occurrence and Management. *Detritus* **2020**, *12*, 34−50.

(258) Drage, D. S.; Sharkey, M.; Abdallah, M. A.-E.; Berresheim, H.; Harrad, S. Brominated Flame Retardants in Irish Waste Polymers: Concentrations, Legislative Compliance, and Treatment Options. *Sci. Total Environ.* **2018**, *625*, 1535−1543.

(259) Samsonek, J.; Puype, F. Occurrence of Brominated Flame Retardants in Black Thermo Cups and Selected Kitchen Utensils Purchased on the European Market. *Food Additives & Contaminants: Part A* **2013**, *30* (11), 1976−1986.

(260) Kuang, J.; Abdallah, M. A.-E.; Harrad, S. Brominated Flame Retardants in Black Plastic Kitchen Utensils: Concentrations and Human Exposure Implications. *Sci. Total Environ.* **2018**, *610−611*, 1138−1146.

(261) Ionas, A. C.; Dirtu, A. C.; Anthonissen, T.; Neels, H.; Covaci, A. Downsides of the Recycling Process: Harmful Organic Chemicals in Children's Toys. *Environ. Int.* **2014**, *65*, 54−62.

(262) Fatunsin, O. T.; Oluseyi, T. O.; Drage, D.; Abdallah, M. A.-E.; Turner, A.; Harrad, S. Children's Exposure to Hazardous Brominated Flame Retardants in Plastic Toys. *Sci. Total Environ.* **2020**, *720*, No. 137623.

(263) Zhang, B.; Lu, S.; Huang, M.; Zhou, M.; Zhou, Z.; Zheng, H.; Jiang, Y.; Bai, X.; Zhang, T. Urinary Metabolites of Organophosphate Flame Retardants in 0−5-Year-Old Children: Potential Exposure Risk for Inpatients and Home-Stay Infants. *Environ. Pollut.* **2018**, *243*, 318−325.

APPENDIX 2

Melymuk, Lisa, Jonathan Blumenthal, Ondřej Sáňka, Adriana Shu-Yin, Veena Singla, Kateřina Sebková, Kristi Pullen Fedinick, and Miriam L Diamond. 2022. "Persistent Problem: Global Challenges to Managing PCBs." *Environmental Science & Technology* 2022: 9040. https://doi.org/10.1021/acs.est.2c01204

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Persistent Problem: Global Challenges to Managing PCBs

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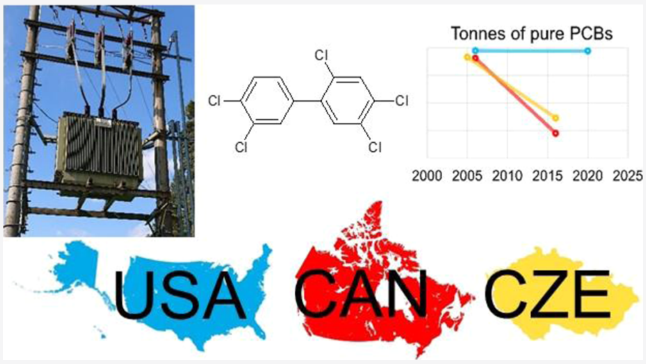
ABSTRACT: Polychlorinated biphenyls (PCBs), "famous" as persistent organic pollutants (POPs), have been managed nationally since the 1970s and globally under the Stockholm Convention on POPs since 2004, requiring environmentally sound management (ESM) of PCBs by 2028. At most, 30% of countries are on track to achieve ESM by 2028. Globally over 10 million tonnes of PCB-containing materials remain, mostly in countries lacking the ability to manage PCB waste. Canada (Ontario) and Czechia, both parties to the Stockholm Convention, are close to achieving the 2028 goal, having reduced their stocks of pure PCBs by 99% in the past 10 years. In contrast, the USA, not a party to the Stockholm Convention, continues to have a substantial but poorly inventoried stock of PCBs and only ~3% decrease in mass of PCBs since 2006. PCB management, which depends on Stockholm Convention support and national compliance, portends major challenges for POP management. The failure to manage global PCB stocks >30 years after the end of production highlights the urgent need to prioritize reducing production and use of newer, more widely distributed POPs such as chlorinated paraffins and per- and polyfluorinated alkyl substances, as these management challenges are unlikely to be resolved in the coming decades.



KEYWORDS: *polychlorinated biphenyls, Stockholm Convention, chemicals management, persistent organic pollutants, PCB stocks, environmentally sound management, Canada, Czechia, USA*

■ INTRODUCTION

Polychlorinated biphenyls (PCBs) are the epitome of a persistent organic pollutant (POP) because of their persistence, bioaccumulative potential, and toxicity. Owing to their environmental mobility and persistence, they are distributed globally, from the high Arctic and Antarctic to the Mariana Trench in the deep Pacific Ocean.[1] PCBs pose risks to ecosystems as they potently bioaccumulate through the food web to reach levels of concern among top trophic level animals. In utero exposures are associated with neurodevelopmental toxicity, manifesting as learning, behavioral, or intellectual impairment in children.[2−6] PCB exposures are also associated with impaired immunological function, auditory deficits, and central nervous system disorders such as Parkinson-like symptoms.[7−9]

PCBs were introduced for use in dielectric fluids to reduce the risk of explosion in capacitors and transformers and saw widespread use as plasticizers and flame retardants in products such as building materials and paints.[10] Breivik et al.[11] estimated that more than 1.3 million tonnes of pure PCBs were manufactured between 1930 and 1993 in at least 10 countries, primarily in the USA, followed by West Germany, the USSR, and France (Figure S1). However, PCBs were widely exported from manufacturing countries, resulting in use in at least 114 countries.[11] The ~1.3 million tonnes of pure PCBs, through dilution for use and subsequent poor management, expanded to 17 million tonnes of PCB-contaminated materials and waste (Figure 1), with an estimated 20−35% of PCBs already released in the environment.[17] Monsanto, in the USA, produced more than 50% of global PCBs and recognized the toxicity of PCBs shortly after the start of large-scale commercial production in the 1930s.[13] According to Monsanto's documentation, the company argued that PCBs were of minimal risk in closed systems such as capacitors and transformers, but the company intended to use PCBs in a very wide array of products that would result in environmental release and human exposure.[12] Clear evidence of their widespread environmental distribution came in the 1960s and 1970s, first in Baltic Sea biota,[13] along with indications of their toxicity.[14]

Global production of PCBs decreased with the introduction of restrictions in the 1970s in Western countries (Figures 1 and

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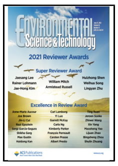

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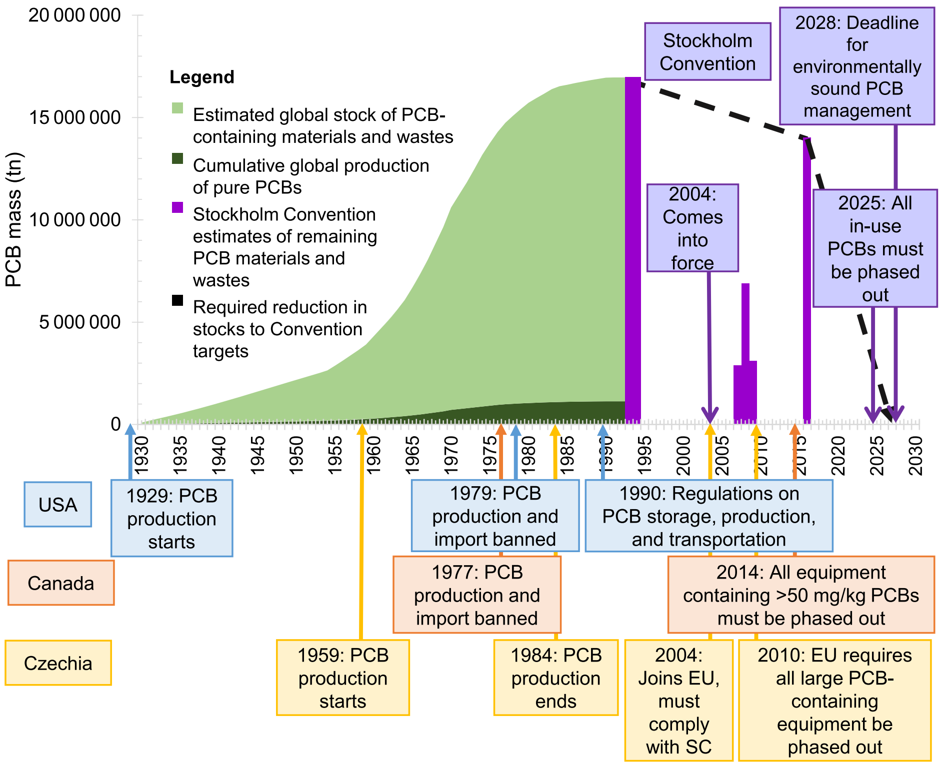

Figure 1. Timeline of major policy actions on PCBs in the USA, Canada, and Czechia and in the Stockholm Convention. Rate of PCB production and associated increase in stocks is estimated based on the study by Breivik et al.[11] The Stockholm Convention estimates of remaining stocks are obtained from refs 17−181920. The total stock of PCB-containing materials and waste (up to 20 million t) reflects how pure PCBs are diluted to create this mass and how mismanagement spreads pure PCBs to create a larger contaminated mass. The variation in the estimated stock over 1994−2016 reflects how reported global inventories changed over time due to uncertainties in reported PCB stocks. Additional details on the timeline of policies impacting PCBs are given in Figure S2.

S2). These were followed by restrictions implemented through international agreements, notably the Stockholm Convention on POPs, which entered into force in 2004 and currently has 185 parties (184 countries plus the European Union, chm.pops.int). The Stockholm Convention bans the production of PCBs and aims to phase out in-use PCBs by 2025 and ensure environmentally sound management (ESM) of materials with >0.005% (50 mg/kg) PCB content by 2028. ESM largely constitutes chemical destruction by high-temperature combustion methods where the PCB content of the waste is destroyed, with exceptions for low PCB content wastes with large volumes, in which case specially engineered landfills or permanent storage in underground mines/rock formations can serve as reasonable substitutes (Table S1).[15] Each party implements the Stockholm Convention through the enactment of binding legislation. Parties have devoted considerable efforts over decades to eliminating PCB stocks and controlling further primary and major secondary releases. Much has been accomplished. However, as of 2016, UNEP identified that only 17% of PCB-containing materials have been eliminated, at the rate of about 200,000 t/y since 2000. Addressing the remaining 83% would require the elimination of ~1 million tonnes of PCB-containing oils and contaminated equipment per year to reach the 2028 target.[16]

Today, despite restrictions, primary PCB emissions continue from on-going use in products and materials: closed applications (e.g., transformers, capacitors, electric motors, and light ballasts), partially open applications (e.g., hydraulic fluid, heat transfer fluid, switches, electrical cables, and vacuum pumps), and open applications (e.g., paints, sealants, inks, lubricants, flame retardants, insulation, dyes, and pesticides) (see Table S2).[21,22] This diversity of uses, combined with their poor documentation, creates a global challenge for managing primary PCB sources.

Although the greatest mass of PCB use has been in closed applications, open applications of PCBs have received increasing attention, particularly in relation to sensitive indoor environments such as schools.[23−25] Open applications have been shown to result in direct exposure, particularly in schools, and secondary exposure from emissions to the surrounding environment; open applications were estimated to be the primary contributors to global emissions up to 1980.[29] Building materials have received the most attention, notably joint sealants and paints, but >15 types of open applications have been identified by UNEP,[26] and previously undocumented open source uses (e.g., floor waxes,[27] book bindings[28]) continue to be identified. Open applications were the first to receive international regulatory attention through OECD restriction on open applications in 1973.[30] While reports frequently state that approximately 21−26% of PCBs were used in open applications,[26,29] this is a rough global average, and the type and amount of open PCB use varied substantially by region: in Japan, the majority of open use was in carbonless copy paper, while in Western Europe, it was in building sealants.[26,29] Open applications of PCBs present a unique challenge as they are not typically included in national inventories and are frequently not even recognized as PCB wastes.[26] Countries that have estimated stocks of PCBs in open applications (e.g., Germany, Finland, Norway, Sweden, and Switzerland) have identified amounts from hundreds to thousands of tonnes.[26] For example, Germany had an estimated 12,000 t remaining in open applications as of 2013, contributing 7−12 t of PCBs to the environment annually.[31]

Global PCB contamination has many facets, from the legacy use of PCBs in industrial and consumer products, to unintentional production and releases from industries and combustion[32,33] and to emissions of unintentionally produced nonlegacy PCB congeners from modern materials such as paints and cabinetry.[34,35] Here, we focus on the challenge of managing primary legacy PCB stocks through a review and analysis of current PCB management status globally. We assess progress toward ESM of PCBs through a two-part analysis: (1) challenges of PCB management on a global scale and (2) a detailed case study comparison of the current status of PCB management in three economically developed countries (Canada, Czechia, and USA) with differing histories of Stockholm Convention participation, PCB production and use, and management capabilities.

The motivation for this analysis is two-fold. First, it is critical to understand the scale of the future threat posed by PCBs to human and ecosystem health. After countries enacted controls in the 1970s and early 1980s, concentrations in air, water, and relatively short-lived biota dropped rapidly.[36] However, decreases have slowed in recent years.[37,38] Today, 40 years after major production ceased, PCBs may cause the demise of over 50% of the world's killer whale populations.[39] As a neurotoxicant, PCBs contribute to the significant global burden of disease attributable to widespread human exposure to hazardous pollutants.[40] Second, the analysis of successes and failures in managing PCBs provides a clear cautionary lesson on the long-term impacts of producing and widely using persistent compounds and the inability of even wealthy countries to manage and eliminate their on-going use.

■ METHODS

Global PCB Management. Historical PCB consumption was adapted from Breivik et al.[11] to reflect the total PCB mass used (see Text S1). The status of PCB use and management of all UN-registered countries was classified based on the most recent information (Table S3). In most cases, this information was the most recent Stockholm Convention status document (National Implementation Plan or Conference of the Parties update) and the responses to a 2018 questionnaire given to Stockholm Convention parties on PCB management. Where available, other reports were also used, particularly Global Environment Facility (GEF) project reports (see references in Table S3). Based on this information, we placed countries into eight categories according to their PCB management, ranging from no existing PCB management plan or inventory, to full ESM. We also included countries that are not parties to the Stockholm Convention.

Case Study: Canada. Data from Ontario, containing nearly 40% of the Canadian population,[41] were used as an indicator of Canadian performance. Details of the Ontario PCB inventory are provided in the Supporting Information (Text S2). Briefly, the PCB stock in Ontario was estimated by combining data from the Canadian federal "ePCB" database and a provincial-level PCB Waste database maintained by the Ontario Ministry of the Environment, Conservation and Parks. The federal ePCB reporting system lists locations of PCB holdings in-use and in-storage with concentrations >50 mg/kg. Data as of December 31, 2016 for closed sources were used. The Ontario database separately lists PCB waste storage sites in Ontario; data used were as of 2013. Combined, the databases reported a total of 270 unique sites with PCBs in use, stored, or classified as wastes not yet subject to ESM.

Any values given in units of volume were converted to mass based on assumptions of the density of the PCB-containing material. The masses of PCB-containing materials were then converted to estimates of pure PCB mass assuming average concentrations per category of application.[42] Askarel fluid was assumed to have 600,000 mg/kg PCBs (range 400,000−800,000 mg/kg) and mineral oil 250 mg/kg PCBs (range 50−500 mg/kg).[43,44] Concentrations in waste categories were estimated based on their classification as either low- or high-level waste. Assumptions are detailed in Table S4.

We also compared these 2013−2016 data with an inventory of closed sources from 2006 before the enactment of revised Canadian Federal PCB regulations in 2008. This comparison was restricted to the Toronto (largest city in Canada) area because of data availability.[42]

Case Study: Czechia. Data for Czechia were compiled from the current inventory of PCB-containing products and materials, maintained by the CENIA—the Czech Environmental Information Agency. Since the itemized database is not publicly available, our inventory relied on the totals of individual application categories, which have been reported to the Stockholm Convention[45] and the European Union.[46]

Where available, exact reported masses of PCB-contaminated material were used, and when missing, the mass was estimated based on the number of items and the median mass of PCB-containing fluid per item in the category (Table S5).

The masses of PCB-containing fluids were converted to estimates of pure PCB mass using assumptions specific to Czech/EU regulations (Table S5). Czechia complies with the European Council Directive 96/59/EC, which required elimination of all materials containing >500 mg/kg PCBs by 2010; thus, all remaining large PCB equipment (>5 dm^3) should be <500 ppm PCBs. However, there is some ambiguity in the reported information; thus, we have used a 10,000 ppm upper threshold, based on Czech reporting to UNEP,[45] to account for a worst-case scenario of instances of noncompliance.[47]

To evaluate progress since the ratification of the Stockholm Convention, we compared the most recent inventory with totals from the initial Czech National Implementation Plan reflecting 2002−2004.[48] As the original inventory preceded EU legislation, we assumed higher concentrations of PCBs in materials using the values of 20,000 mg/kg (range 10,000−30,000 mg/kg).[47]

Case Study: USA. To estimate the current stock of PCBs in use and waste in USA, we utilized publicly available information from the US Environmental Protection Agency (US EPA), specifically the PCB transformer registration database[49] (Table S6, as of Jan 2020) and the PCB Cleanup and Disposal Program[50] (up to 2020). The US EPA tracks transformers and regulated PCB waste pursuant to regulations under the Toxic Substances Control Act (TSCA). Owners of PCB transformers must register details on transformer location, ownership, and mass in the PCB transformer registration database. PCB transformers that are removed from use may be optionally deregistered from the database. We extracted information on the number, location, and mass of transformers to estimate the stock of PCBs currently held in in-use and stored transformers. Incomplete records were assigned

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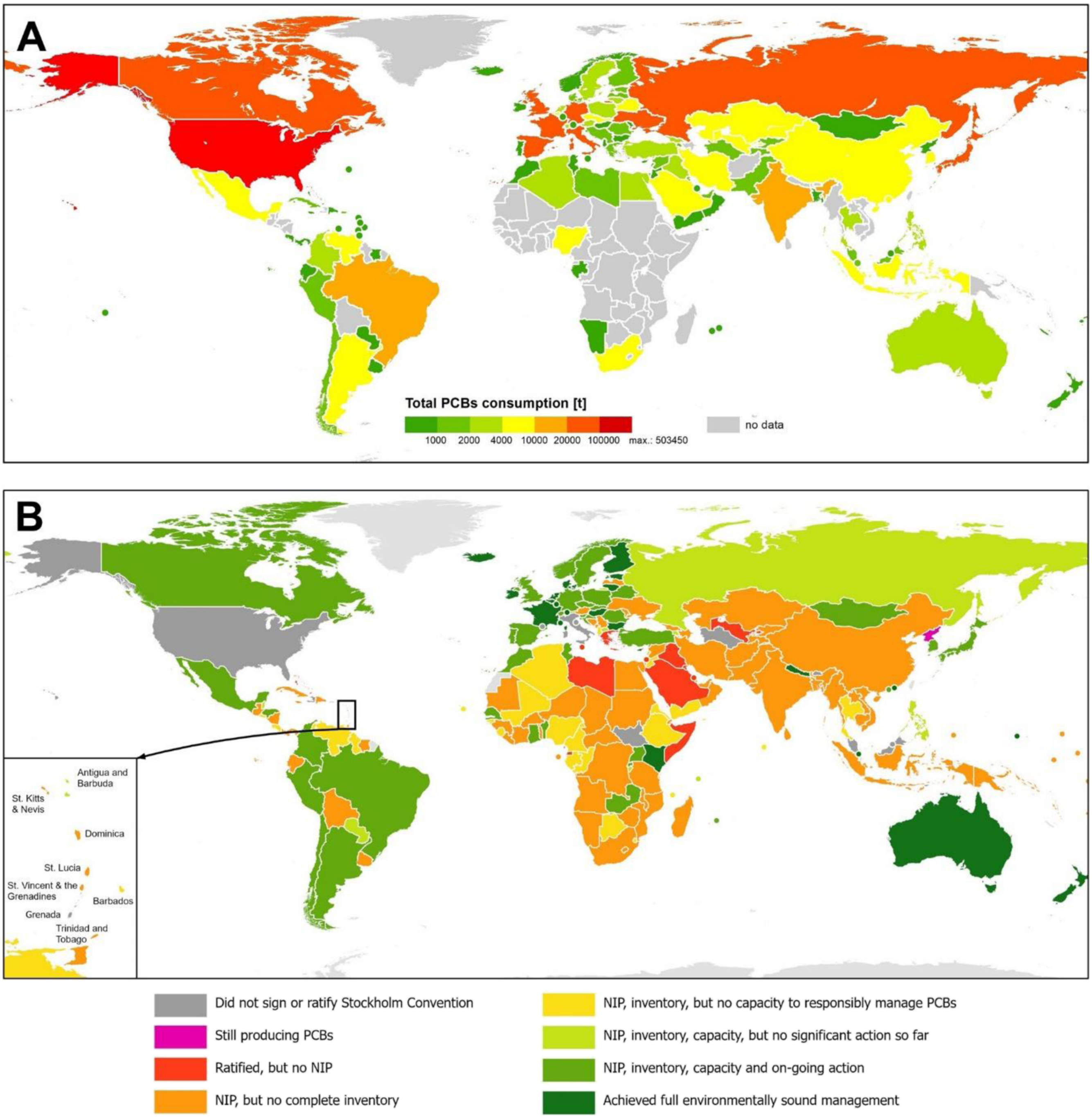

Figure 2. Global PCB use and management. (A) Total PCB consumption by country throughout the period 1930−2000 based on data from Breivik et al.[11] extrapolated to total PCB mass consumption according to Text S1. Figure S6 presents the same data presented as per capita consumption. (B) Current status of PCB management according to the latest reported status for each country, compiled from Stockholm Convention reporting and other sources. Sources used are given in Table S3. NIP indicates Stockholm Convention National Implementation Plan. A color-blind accessible version of this figure can be found in Figure S5.

the median value of the complete records (median two transformers per site; median mass 782 kg/transformer). A transformer was considered to be 30% fluid by mass;[51] this factor was used to convert total transformer masses to masses of PCB-containing fluids. Fluid masses were converted to estimates of pure PCB mass using assumptions about the concentrations in a typical transformer, as for the Canadian inventory. We assumed that before Jan 1, 2000, all transformers reported were askarel transformers, and after that date, all were mineral oil, based on the assumption that transformer owners would be aware of high-level PCB content and report in compliance with the TSCA regulations. However, PCB-contaminated mineral oil may be less thoroughly documented, leading to delayed reporting.[52] Average concentrations used for askarel transformers were 600,000 mg/kg PCBs (range 400,000−800,000 mg/kg) and those for mineral oil transformers were 1000 mg/kg PCBs (range 500−5000 mg/kg).[43,44,52] To evaluate the completeness of the transformer registration database, we compared the total number and mass of transformers reported to the PCB Cleanup and Disposal

Program with the total number of transformers in the registration database and the mass of deregistered transformers from 1998 to 2018.

To evaluate changes in the USA stock over time, we compared the current remaining stock of transformers to records of PCB transformer deregistrations to calculate the number of transformers and mass of PCBs that would have remained in use/stored in 2006.

Uncertainty Analyses. Monte Carlo analysis was performed for the Canadian PCB databases, which identified that concentration levels (i.e., mg/kg PCBs per equipment/waste category), particularly the concentration selected for the high-level equipment/wastes, had the greatest impact on the estimate of the total stock (Text S3, Figures S3, S4). Density and other assumptions made to complete missing data had negligible impact. Therefore, we addressed the uncertainties in the conversion of database entries to pure PCB masses through the inclusion of upper and lower concentration thresholds for individual PCB equipment and waste categories according to the regulations and database thresholds for each country, as described above.

■ RESULTS

Global Management of PCB Stocks. Global use of PCBs varies widely, and the current PCB management capacities of countries also vary widely (Figure 2B, Table S3). One country (Democratic People's Republic of Korea) continues to produce PCBs.[17] 185 parties (184 countries plus the European Union) have ratified the Stockholm Convention, while 13 countries have not, notably the USA, Italy, Malaysia, Haiti, Israel, and Turkmenistan; all other nonratifying countries have <1 million population or are recently established (e.g., South Sudan). We note that while Italy has not ratified the Stockholm Convention, the European Union, of which Italy is a member, is a party to the Convention, and the EU has stricter regulations on PCB management than the Stockholm Convention. Of the 184 ratifying parties, 10 have not submitted any implementation plan. Greece and Malta, as EU member states, should also follow EU PCB management regulations, despite not yet submitting documentation to the Stockholm Convention.

For the 174 Parties that have submitted reports, our analysis highlights that 72 national PCB inventories (42%) are partial or preliminary. Many inventories are limited to transformers and/or only to the public electricity sector, which may capture only half of the uses of PCBs (considering 48% of the PCBs produced were used in transformers[22]). An additional 23 countries (13%) reported complete PCB inventories but no capacity to achieve ESM, while 11 countries had inventories and capacity to manage PCBs but had made no significant progress toward ESM. The number of countries achieving or progressing toward ESM was small; 34 countries (18%) are progressing toward ESM through removal from use and destruction of PCB materials. Only 23 countries (13%) have achieved ESM of PCBs. With three exceptions (Nepal, Kenya, and Micronesia), all of the 23 countries that have achieved ESM are classified as "very high" in the UNDP Human Development Index, or "high income" under by the World Bank (Table S3). Only three countries classified as "low development"/"low income" are currently making substantial progress toward ESM of PCBs: Benin, Rwanda, and Uganda.

National reporting to the Stockholm Convention contains a wealth of information about PCB management, but the quality and quantity of information provided by individual countries vary widely. A questionnaire from the Convention was distributed to 182 parties to evaluate progress toward ESM of PCBs. Fewer than 60 parties provided responses.[53] Based on these responses and additional Stockholm Convention information sources, estimates of current PCB stocks could be determined for 52 countries (see references in Table S3). Many countries lacked recent information, with some countries not submitting documentation since 2004. Therefore, our analysis is uncertain for two reasons. First, we may be presenting a worst-case scenario since countries may have made unreported progress toward PCB elimination. Second, and conversely, many countries reported incomplete inventories (e.g., only inventories of transformers owned by a national electricity provider), and thus most recent reports under-record true stocks.

Many countries are challenged by weak institutions, corruption, and mismanagement, making tracking PCB stocks and limiting their misuse extremely difficult. There are multiple reports of transformers being improperly recycled. For example, Sri Lanka identified that PCB-containing transformers were transferred to informal recyclers and that spilled PCB-containing oils were cleaned up with sawdust of which some were disposed of through burning.[54] Nauru reported that a transformer confirmed to contain PCBs was slated to be shipped to Australia but was instead collected by a scrap metal recycling company with an undetermined fate.[55] In other instances, transformer owners were reported to have actively drained and disposed of PCB contents to avoid responsibility for PCB materials. For example, in Malawi, numerous pieces of equipment suspected to contain PCBs had their contents poured directly onto the ground before they could be tested.[56] The Dominican Republic reported that owners of transformer shops, to avoid PCB disposal regulations, diluted the PCB concentration in the fluids by continually removing them and adding more mineral oils and sold the removed PCB oils to illegal foundries.[57] Ghana reported the use of PCB oils to create beauty creams and to lubricate domestic sewing machines,[58] while in Montenegro, factory workers were reported to have used PCB oils for handwashing and to heat homes.[59]

Several initiatives, such as those funded by the Global Environment Fund (GEF) and implemented by UNEP, have made progress in addressing some of the challenges low-income countries face in managing PCBs. For example, a project, harmonizing efforts in Southern Africa to centralize dismantling, draining, and accumulation of PCB oils/equipment for disposal,[60] was among more than 40 GEF-funded projects on ESM of PCBs. Together, these projects have eliminated 23,000 tonnes of PCBs.[17] However, even these specific projects can be hindered by unreliable national reports, delays in laboratory analysis of suspected PCB materials, and, most crucially, incomplete inventories. The technological and financial capacities required to eliminate PCBs are not available in many regions. GEF-funded projects on PCBs have received ~$450 million USD to support the elimination of 88,000 tonnes of PCB-containing materials and waste (23,000 tonnes eliminated and 65,000 tonnes planned), averaging USD 5,000 per tonne of PCB waste eliminated. It is noted that these project costs include items not directly related to elimination (i.e., capacity building and education).[17] Currently, the cost burden of managing PCBs lies with national governments or international agencies (e.g., national environment agencies,

Figure 3. Inventories of PCBs for USA, Czechia, and Ontario, Canada. (A−C) Estimated stocks as of mid 2010s by (A) number of items, (B) bulk mass of PCB-containing materials, and (C) pure PCBs, with error bars indicating the uncertainty of the estimates of mass. (D−F) Change over time in PCB-contaminated items, bulk mass of PCBs, and pure PCBs. (G,H) Distribution of current stocks of pure PCBs in Ontario and Czechia according to item categories. USA is not shown because only transformers are included in the inventory.

UNEP, and GEF). Producer financial responsibility to date has only been in the form of legal settlements. Funding PCB elimination is clearly a problem, with the estimated need to eliminate 1 million tonnes of PCB-containing materials and waste per year to achieve Stockholm Convention compliance, mostly in countries with minimal financial and/or technical capacity. Yet despite the costs of elimination, there is also a clear public health cost of inaction: toxic chemicals, including PCBs, are neurotoxicants contributing to the "pandemic of developmental neurotoxicity," placing a significant burden on societal resources.[6,10]

At most, 30% of countries are on track to achieve ESM by 2028, and the lack of capacities for PCB management is a barrier to achieving this goal. With the specific examples of Canada, Czechia, and the USA, we demonstrate successes and barriers to ESM even in high-income/highly developed countries.

Case Studies. *Inventory Results: Canada.* Based on the combined provincial and federal databases as of 2013 and 2016, we identified a total of 12,200 tonnes of material in Ontario with PCB content >50 mg/kg. These materials were estimated to contain 32 t of pure PCBs (Figure 3, Table S11) with a range of 21−44 t. The PCB sites were widely distributed across heavily populated Southern Ontario and within or close to urban areas. The stock of PCB materials in Ontario was dominated by PCB-contaminated soil/gravel held at two sites, making up 97% of the bulk mass of PCB materials but <10% of the pure PCBs, which was largely held in transformers (77%) and capacitors (12%) (Figure 3G).

According to a 2006 PCB inventory,[42] Toronto had 455 sites containing 850 t of PCB-containing equipment and materials, equivalent to 424 t (range 282−565 t) of pure PCBs. A large proportion of this was located in Toronto's central business district in electrical transformers in large office towers.[42] In 2016, Toronto had only 7 sites holding 31 separate PCB items, all located outside of the city center, of which 3 sites had PCBs remaining in use (all others were stored for disposal), constituting only ~40 g of pure PCBs. The remaining four sites stored 2.9 t of bulk PCB material, equivalent to 0.8 t (range 0.6−1.0 t) of pure PCBs. Thus, >99% of the 2006 stock of PCBs in Toronto was removed in the 10 years between inventories and after 2008 legislation mandating ESM, clearly showing progress toward elimination of PCB stocks in Toronto (Figure 3F). This agrees with Canada's reporting to the Stockholm Convention[61] which states that 0.02% of the 2008 PCB stock remains today.

Inventory Results: Czechia. Based on the most recent Czech records, we estimated a total stock of 6092 tonnes of material with PCB content >50 mg/kg in 2016. This bulk mass was estimated to contain 2.84 t (0.30−62 t) of pure PCBs (Table S11). Most of the mass of PCB materials (74% of bulk material and 79% of pure PCBs) was in transformers held by a large electricity production and distribution company responsible for more than 14,000 pieces of PCB-containing materials, including close to 10,000 transformers (Figure 3A). However, despite the large number of transformers, Czechia has prioritized the removal of equipment containing high concentrations of PCBs in compliance with European Union and Stockholm Convention regulations. This has resulted in a relatively low level of pure PCBs (~2 tonnes) in these ~10,000 transformers.

In 2005, 25,000 items contained ~3000 t of PCB-contaminated fluid/materials,[48] equivalent to an estimated 460 tonnes of pure PCBs. An inventory in the intervening years (from 2009) reported 9193 t of known PCB materials and 3228 t of possible PCB materials.[62] Later inventories (2009, 2016) were more comprehensive, which resulted in a higher reported bulk mass of PCBs (Figure 3E). The prioritized removal of high concentration items has led to the large decrease of pure PCBs since the 2005 inventory was compiled (Figure 3F).[48]

Inventory Results: USA. The USA inventory was based only on the US EPA transformer registration database[49] (Table S6) and, consequently, was incomplete in two respects. First, it contained only records of transformers—no inventories exist in the USA for any other PCB-containing materials. Data from the PCB Cleanup and Disposal program[50] (Tables S7 and S8) indicate the disposal of millions of kilograms of large low- and high-voltage capacitors and bulk waste from 1998 to 2018, none of which has been included in any inventory. Second, the transformer registration database likely does not include all PCB transformers. From 1998 to 2018, the registration database listed 20,130 total transformers, but the PCB disposal program data indicated that over 180,000 transformers were disposed of over the same period, strongly suggesting that the registration database did not, and likely still does not, include all PCB-containing transformers. Therefore, our stock estimates in Table S11 are a clear underestimate of closed-source PCBs in USA.

As of 2020, the USA transformer database contained records of 11,577 transformers, estimated to contain 13,755 tonnes of PCB material with 776 (517−1040) tonnes of pure PCBs. To provide a temporal comparison similar to Canada and Czechia, the database was re-evaluated considering the stock of transformers existing in 2006. In 2006, the USA had 14,457 transformers containing 47,500 tonnes of PCB material and 770 tonnes of pure PCBs. This suggests a 20% reduction in the number of transformers and a 71% reduction in the mass of bulk PCB material, but only a 3% reduction in pure PCBs over ~15 years (Figure 3D−F).

■ DISCUSSION

Canada. PCBs were never manufactured in Canada, but an estimated 40,000 t of PCBs were imported up to 1980,[44] leading to the second-highest per capita use in the world (1.2 kg/person), behind only the USA (Figure S5). New uses of PCBs were banned in 1977. Canada signed the Stockholm Convention in 2001 and enacted legislation to comply with the Convention in 2008. The purpose of the 2008 Canadian PCB regulations was to accelerate the elimination of PCBs in concentrations greater than 50 mg/kg by 2025 by stipulating end-of-use deadlines, especially for PCBs near sensitive sites (schools, daycares, hospitals, etc.).[63]

Of the three profiled countries, Canada has most successfully managed PCBs. The mass of pure PCBs in Toronto, Canada's largest city, decreased by 3 orders of magnitude within 10 years, indicating that regulations were successful in phasing out PCBs in Toronto. PCB material has been removed from all transformers in large skyscrapers built in the 1960s and 1970s in the downtown core and from sensitive sites such as schools.[42] The enforcement of PCB regulations in Canada includes compliance strategies and environmental officers that inspect PCB facilities. For example, in 2015, inspections were conducted at 44 companies that were set to remove and destroy their PCB equipment, finding 89% compliance.[64] Some inventory data indicated a lack of compliance in a small number of cases; three entries in Ontario databases reported concentrations over 500 mg/kg for equipment that should have been removed by 2009.

Canada's database has only limited inclusion of open sources of PCBs, although these were used in Ontario, for example, as joint sealants in buildings constructed from the 1950s to 1970s.[65] Past inventories estimated that the contributions of PCB-containing building sealants were low relative to the total amount of PCBs held in transformers,[42,65] but with the prioritized removal of closed PCB equipment and little attention given to open sources, their relative importance may now be greater. In countries that consider open sources in their PCB inventories, they typically account for more than a third of the remaining PCB bulk mass, for example, 37% in Germany[66] and 39% in Switzerland.[67]

Committing to national and international PCB agreements has helped expedite Canada's progress on phasing out PCBs. The 2008 regulation was proposed to meet the targets and commitments of the Stockholm Convention.[68] If the current trend continues across Canada, we suspect that Canada will be well on track to meet the 2025 target set by the Stockholm Convention.

Czechia. At the time of PCB production, Czechia was Czechoslovakia, and PCBs were produced by the Chemko Strážske factory in the east of the country (now Slovakia). Chemko produced 21,481 tonnes of PCBs from 1959 to 1984, of which 11,613 tonnes were used in Czechoslovakia, and the rest was exported to other Eastern Bloc countries, primarily East Germany.[47] Per capita use in Czechoslovakia was 580 g/person, the 20th highest in the world (Figure S6). It is estimated that 7000−8500 t of pure PCBs were used in the area that is now Czechia.[48] Most PCB use was by three state companies manufacturing PCB-containing paints and coatings, electrical capacitors, and electrical equipment for transport and industry.[46] These materials were then distributed to over 200 other companies/state agencies. PCB manufacturing ceased in 1984, and use restrictions were introduced in stages over the 1990s as Czechia moved toward compliance with EU regulations with the goal of eventually entering the EU. These late restrictions are the reason why the breast milk of Czech and Slovak mothers contains, on average, the highest concentration of PCBs among industrialized countries.[69,70]

The Czech PCB stockpile and waste management plan covering 2003 to 2013 dictates that all PCB materials exceeding 50 ppm be managed according to ESM provisions of the Stockholm Convention. Czechia, as a member of the EU, additionally follows the EU PCB regulations, which are stricter than those of the Stockholm Convention for large equipment, requiring that equipment with PCB volumes >5 dm^3 was decontaminated or disposed of by 2010.

As with Canada, open sources were not considered in the Czech inventory. PCB-containing building sealants had limited use in Czechia and other former Eastern Bloc countries, unlike in Western Europe and North America, due to their higher cost. The major open use of PCBs was in paints, particularly on bridges and in military applications, constituting approximately 21% of total use.[45,47] PCBs released from paints have caused significant environmental contamination, including of Elbe River sediments with up to 6 mg/kg PCBs due to renovations to a railway bridge that had PCB-containing paint.[71]

The biggest challenge in Czechia is not the documented inventory of PCBs but rather the numerous abandoned industrial/contaminated sites. This number is larger than typical in Western countries due to the country's transition from a communist economy in the early 1990s, dissolution of state-owned companies, and subsequent bankruptcies and abandonment of these sites. Numerous facilities now remain without responsible ownership and contain abandoned industrial infrastructure, with possible PCB contamination, as well as contaminated soils and other materials. The database of contaminated sites maintained by the Czech Ministry of Environment listed 387 sites with PCB contamination as of 2016; however, 88% remain only as suspected contamination, still lacking a proper site survey.[45]

Since the early 2000s, the Czech PCB inventory appears to have grown because of the "discovery" of many of these sites and reporting from companies that had not initially disclosed their stocks. The 2002−2005 database reported 25,000 contaminated or potentially contaminated items, with ~3000 t of bulk PCB fluid/materials.[48] By 2016, there were slightly fewer PCB-containing items (21,300) but a much higher bulk mass of PCBs (6093 t) compared to 2005. This discrepancy arose because of the inclusion of small PCB items that were not included in the original inventory. Most importantly, the mass of pure PCBs has dramatically decreased: only 1% (0.1−17%) of the 2005 stock of pure PCBs was estimated to remain in 2016 (Figure 3). This large decrease in pure PCBs is attributed to progress toward compliance with EU regulations, with priority given to ESM of high-level and large-volume PCB equipment.

Czechia has sufficient capacity for ESM of PCBs (e.g., annual hazardous/POP waste incineration capacity of >21,000 t[46]); however, the country requires action from both private entities and the state, in case of the abandoned stock, and a significant effort to remove PCBs from use if it is to achieve ESM by 2028.

USA. The USA was the world's largest producer and consumer of PCB products with an estimated use of 500,000 tonnes of PCBs, or 1.9 kg per capita, the highest in the world (Figures 2A and S5). The USA passed regulations in 1979 under the TSCA to prohibit the manufacture, processing, distribution in commerce, and new use of PCBs. However, the USA has not ratified the Stockholm Convention and does not have national legislation that sets deadlines for PCB elimination.[72] This lack of national legislation is also reflected in its fragmented and incomplete PCB data. While it is clear from the PCB Cleanup and Disposal program[50] that the USA has removed a large stock of PCBs from use, the incomplete current inventories prohibit the evaluation of the remaining burden of PCBs in the USA. Our analysis strongly suggested that the transformer registration database is missing a substantial number of transformers, and no inventory exists for other PCB materials (capacitors, ballasts, other electrical equipment, and contaminated soil). In Canada, these materials accounted for ~25% of the total mass of pure PCBs and 99% of the mass of bulk PCB materials; we expect similar proportions in the USA. The US transformer database requires responsible parties to self-report information, with limited enforcement, and the concentration of PCBs is not reported. The incomplete or erroneous information greatly limits the accuracy of the inventory.

Our assessment of the US records as incomplete contrasts with a recent UNEP report suggesting that US PCB records are more comprehensive than those for other countries.[17] While this may be the case in comparison to developing countries, we argue that the fragmented and inconsistent nature of reporting, limited mainly to transformers, coupled with the highest global PCB manufacturing and use, presents the USA as a worst-case scenario for PCB management in countries with a capacity to do so.

Even based on incomplete information, the stock of PCBs in the USA remains large compared to Canada and Czechia. While the USA has removed 100,000 t of PCBs from use, the impact of this removal on total PCB stocks is highly uncertain because of poor record-keeping. Moreover, the USA did not show a significant decrease in the mass of pure PCBs between 2005 and 2020 as was seen for Canada and Czechia (Figure 3F), which have prioritized removal of high-level PCB materials (e.g., askarel transformers). Further, PCBs removed from use in the USA are legally allowed to be disposed of by methods not considered ESM by the Stockholm Convention, such as landfilling. This is a major concern as landfills may act as secondary PCB sources by contaminating surrounding ecosystems with resulting ongoing human and environmental exposures.[73] The environmental and societal burden due to the ongoing use and non-ESM disposal of PCBs in the USA is a clear concern, given the country's history as the world's largest producer and user of PCBs.

The per capita mass of pure PCBs in the USA inventory (2.40 g/person for 2020) is comparable to Ontario, Canada (2.41 g/person for 2013−2016), while Czechia is lower, at 0.27 g/person for 2016. However, of all three countries, Canada has the most complete inventory, which includes large masses of contaminated soil and gravel that are not included in inventories in either Czechia or USA. The per capita stock in the USA would be significantly higher if the inventory included the additional categories of capacitors, other electrical equipment, and contaminated waste materials that have been significant portions of the Canadian stock (Figure 3G). It is estimated that USA has at least 26 million cubic meters of soils contaminated with PCBs[17] and 350 "Superfund" sites with reported PCB contamination.

IMPLICATIONS AND LESSONS LEARNED FROM PCB FAILURES

In 2016, UNEP completed an assessment of global efforts to eliminate PCBs, with an update in 2019.[17,53] The report highlighted similar challenges as identified here: incomplete inventories of PCBs in many regions and large uncertainties in current stocks and extent of ESM. The report identified that "the majority of countries (with some notable exceptions) are currently not on track to achieve the ESM of PCBs by 2028." Our analysis of the progress, available infrastructure, and the challenges within individual countries in managing PCBs confirm the implausibility of achieving this goal without a rapid change in actions. UNEP also reported that the PCBs eliminated so far were likely "low-hanging fruit," and further elimination will present more logistical and technical problems, challenging our ability to achieve the rates of elimination needed to reach Stockholm Convention goals.[17]

One substantial challenge is PCBs remaining in open applications, which are poorly documented even in countries that report ESM. To date, Sweden has given the most comprehensive attention to open applications through a program of identification and decontamination of buildings with PCBs in building materials.[74] Also, a small number of other countries have developed general estimates and addressed contaminated buildings on a case-by-case basis. The difficulty in managing open applications of PCBs is the lack of documentation of use combined with their importance to human exposure[25,75] as well as the lack of coherent strategies to manage PCB-containing/PCB-contaminated building materials without increasing emissions of PCBs.

In addition to the challenge of ESM for the remaining global stock of PCBs, we are faced with the challenge that some large fraction of PCBs is no longer "manageable." These "unmanageable" stocks have been released to the environment, landfilled without documentation, or, in arguably worst cases, are "lost" due to a lack of labelling and documentation and have entered the commercial sector as oils without identified PCB content. The longer the use and improper storage of PCBs persists, the greater the potential for environmental releases and human exposures, particularly considering the aging infrastructure that houses PCBs.

Strong regulation combined with financial and technological capacity and enforcement, as demonstrated in Canada, can successfully reduce stocks of PCBs and advance ESM. Many countries face substantial challenges due to historical structures (e.g., as in Czechia), and even with sufficient financial and technological capacity, their progress toward achieving the Stockholm Convention deadlines is hindered by legacies of poor record keeping, environmental practices, and site ownership. However, most troubling is the prevalent lack of capacity to manage PCB stocks. The Stockholm Convention plays an important role in education and capacity building toward PCB elimination, motivating countries, particularly those with the ability to implement the Convention, to inventory and dispose of PCBs. Nonetheless, the inventory quality is generally poor in many countries, compounding the challenge due to the lack of resources and competing national pressures.

The USA is absent from the Stockholm Convention and lacks effective federal policies to remove and safely dispose of PCBs despite being the largest producer, user, and likely holder of the largest stock of PCBs. This is a clear challenge to the global goal of achieving ESM of PCBs because, by their properties of persistence and long-range transport, PCBs are a global threat.

The global distribution of PCB use and stocks is not uniform. The USA, Canada, Soviet Union, Japan, and Western European countries dominated the use of PCBs (Figure 2A). In contrast, today that legacy is shifting globally due to transboundary transport of PCB-containing equipment and wastes. While such movement of hazardous substances is under the purview of the Basel and Rotterdam Conventions, compliance remains a challenge. This global shift is particularly problematic considering the lack of PCB management capacities in lower income/lower development countries. Global inequality is a major challenge in the implementation of the Stockholm Convention objective of achieving ESM of PCBs and POPs in general.

The Stockholm Convention set a deadline to phase out PCBs some 40 years after cessation of production and more than 50 years after many highly developed countries banned their manufacturing, import, and new use. Yet, this deadline appears unachievable due to the resources (financial and technical) and the political will required to address the problem. As the effort required to eliminate PCBs has been seriously underestimated,[17] we question the feasibility of removing other newer POPs that have entered widespread use. In the context of POPs, it could be argued that PCBs are one of the simpler problems: most use was in large, closed items, global trade in PCB-containing materials was limited, and most PCBs were held by large industries which were compelled to inventory and report them. This is a sharp contrast to two classes of POPs which have received recent attention—chlorinated paraffins and per- and polyfluoroalkyl substances, such as PFOS and PFOA. The major use of these chemicals is fundamentally different from PCBs; they are primarily used in open sources and held by millions of individuals, which stymies efforts to inventory and manage removal, with major consequences for the environmental and human health of future generations.[76] Chlorinated paraffins are a clear case of regrettable substitution as the short-chained chlorinated paraffins replaced PCBs in many open applications[26] and, since 2018, are also restricted under the Stockholm Convention. This highlights the critical need and urgency to curtail production and use of chemicals with POP characteristics as complex management challenges will not soon be solved, and the consequences are likely to fall disproportionately on lower-income/lower-development countries.

ASSOCIATED CONTENT

Supporting Information

The Supporting Information is available free of charge at https://pubs.acs.org/doi/10.1021/acs.est.2c01204.

Additional methods details, sensitivity analysis, and additional figures and tables (PDF)

Complete data sources for global analysis (XLSX)

Complete data sources for USA analysis (XLSX)

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Notes

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REFERENCES

(1) Jamieson, A. J.; Malkocs, T.; Piertney, S. B.; Fujii, T.; Zhang, Z. Bioaccumulation of Persistent Organic Pollutants in the Deepest Ocean Fauna. *Nat. Ecol. Evol.* **2017**, *1*, 0051.

(2) Roze, E.; Meijer, L.; Bakker, A.; Van Braeckel, K. N. J. A.; Sauer, P. J. J.; Bos, A. F. Prenatal Exposure to Organohalogens, Including Brominated Flame Retardants, Influences Motor, Cognitive, and Behavioral Performance at School Age. *Environ. Health Perspect.* **2009**, *117*, 1953−1958.

(3) Berghuis, S. A.; Bos, A. F.; Sauer, P. J. J.; Roze, E. Developmental Neurotoxicity of Persistent Organic Pollutants: An Update on Childhood Outcome. *Arch. Toxicol.* **2015**, *89*, 687−709.

(4) Boucher, O.; Jacobson, S. W.; Plusquellec, P.; Dewailly, É.; Ayotte, P.; Forget-Dubois, N.; Jacobson, J. L.; Muckle, G. Prenatal Methylmercury, Postnatal Lead Exposure, and Evidence of Attention Deficit/Hyperactivity Disorder among Inuit Children in Arctic Québec. *Environ. Health Perspect.* **2012**, *120*, 1456−1461.

(5) Bennett, D.; Bellinger, D. C.; Birnbaum, L. S.; Bradman, A.; Chen, A.; Cory-Slechta, D. A.; Engel, S. M.; Fallin, M. D.; Halladay, A.; Hauser, R.; Hertz-Picciotto, I.; Kwiatkowski, C. F.; Lanphear, B. P.; Marquez, E.; Marty, M.; McPartland, J.; Newschaffer, C. J.; Payne-Sturges, D.; Patisaul, H. B.; Perera, F. P.; Ritz, B.; Sass, J.; Schantz, S. L.; Webster, T. F.; Whyatt, R. M.; Woodruff, T. J.; Zoeller, R. T.; Anderko, L.; Campbell, C.; Conry, J. A.; DeNicola, N.; Gould, R. M.; Hirtz, D.; Huffling, K.; Landrigan, P. J.; Lavin, A.; Miller, M.; Mitchell, M. A.; Rubin, L.; Schettler, T.; Tran, H. L.; Acosta, A.; Brody, C.; Miller, E.; Miller, P.; Swanson, M.; Witherspoon, N. O. Project TENDR: Targeting Environmental Neuro-Developmental Risks The TENDR Consensus Statement. *Environ. Health Perspect.* **2016**, *124*, 118−122.

(6) Grandjean, P.; Landrigan, P. Developmental Neurotoxicity of Industrial Chemicals. *Lancet* **2006**, *368*, 2167−2178.

(7) Jusko, T. A.; Sisto, R.; Iosif, A.-M.; Moleti, A.; Wimmerová, S.; Lancz, K.; Tihányi, J.; Šovčiková, E.; Drobná, B.; Palkovičová, L. u.; Jurečková, D.; Thevenet-Morrison, K.; Verner, M.-A.; Sonneborn, D.; Hertz-Picciotto, I.; Trnovec, T. Prenatal and Postnatal Serum PCB Concentrations and Cochlear Function in Children at 45 Months of Age. *Environ. Health Perspect.* **2014**, *122*, 1246−1252.

(8) Poon, E.; Bandara, S. B.; Allen, J. B.; Sadowski, R. N.; Schantz, S. L. Developmental PCB Exposure Increases Susceptibility to Audiogenic Seizures in Adulthood. *Neurotoxicology* **2015**, *46*, 117−124.

(9) Jones, D. C.; Miller, G. W. The Effects of Environmental Neurotoxicants on the Dopaminergic System: A Possible Role in Drug Addiction. *Biochem. Pharmacol.* **2008**, *76*, 569−581.

(10) Diamond, M. L. Toxic Chemicals as Enablers and Poisoners of the Technosphere. *Anthr. Rev.* **2017**, *4*, 72−80.

(11) Breivik, K.; Sweetman, A. J.; Pacyna, J. M.; Jones, K. C. Towards a Global Historical Emission Inventory for Selected PCB Congeners–a Mass Balance Approach. 1. Global Production and Consumption. *Sci. Total Environ.* **2002**, *290*, 181−198.

(12) Markowitz, G.; Rosner, D. Monsanto, PCBs, and the Creation of a "World-wide Ecological Problem". *J Public Health Pol.* **2018**, *39*, 463.

(13) Jensen, S. The PCB Story. *Ambio* **1972**, *1*, 123−131.

(14) Koppe, J. G.; Keys, J. PCBs and the Precautionary Principle. *The Precautionary Principle in the 20th Century: Late Lessons from Early Warnings*; Earthscan Publications, 2002.

(15) UNEP. *Listing of the Basel Convention Technical Guidelines for Environmentally Sound Management (ESM) of Persistent Organic Pollutants (POPs) Wastes*, 2005.

(16) UNEP. Towards Elimination of PCBs. https://www.unep.org/explore-topics/chemicals-waste/what-we-do/persistent-organic-pollutants/toward-elimination-pcb (accessed on May 21, 2021).

(17) UNEP. *Consolidated Assessment of Efforts Made towards the Elimination of Polychlorinated Biphenyls*: Geneva, 2016.

(18) Petrlík, J. When the Polluter Does Not Pay. *PEN Mag* **2010**, *1*, 12−13.

(19) UNEP. *PCBs Elimination Network (PEN)—2028—Environmentally Sound Management of PCBs*: Geneva, 2011.

(20) Warmuth, A.; Ohno, K. The PCBs Elimination Network: The Information Exchange Platform Created for the Risk Reduction of Polychlorinated Biphenyls (PCBs). *J. Epidemiol. Community Health* **2013**, *67*, 4.

(21) UNEP. *Guidelines for the Identification of PCBs and Materials Containing PCBs*; United Nations Environment Programme, 1999; p 40.

(22) Wagner, U. K. Inventories of PCBs: An Expert's Point of View. *PEN Magazine*, 2010; No. *1*.

(23) Herrick, R. F.; Stewart, J. H.; Allen, J. G. Review of PCBs in US Schools: A Brief History, an Estimate of the Number of Impacted Schools, and an Approach for Evaluating Indoor Air Samples. *Environ. Sci. Pollut. Res.* **2016**, *23*, 1975−1985.

(24) Thomas, K.; Xue, J.; Williams, R. W.; Jones, P. D.; Whitaker, D. *Polychlorinated Biphenyls (PCBs) in School Buildings: Sources*; Environmental Levels, and Exposures, 2012.

(25) Marek, R. F.; Thorne, P. S.; Herkert, N. J.; Awad, A. M.; Hornbuckle, K. C. Airborne PCBs and OH-PCBs Inside and Outside Urban and Rural U.S. Schools. *Environ. Sci. Technol.* **2017**, *51*, 7853−7860.

(26) UNEP. *Consolidated Guidance on PCB in Open Applications*, 2019.

(27) Rudel, R. A.; Seryak, L. M.; Brody, J. G. PCB-Containing Wood Floor Finish Is a Likely Source of Elevated PCBs in Residents' Blood, Household Air and Dust: A Case Study of Exposure. *Environ. Health* **2008**, *7*, 2.

(28) Parker, J. S.; Mayer-Blackwell, K. Open Book, Open Source: PCB Usage in Mass-Market Paperback Book Adhesives. *Environ. Sci. Technol. Lett.* **2019**, *6*, 565.

(29) Breivik, K.; Sweetman, A.; Pacyna, J. M.; Jones, K. C. Towards a Global Historical Emission Inventory for Selected PCB Congeners - A Mass Balance Approach: 2. Emissions. *Sci. Total Environ.* **2002**, *290*, 199−224.

(30) OECD. *Decision of the Council on Protection of the Environment by Control of Polychlorinated Biphenyls—OECD/LEGAL/0108—C(73)1(Final)*; Organisation for Economic Co-operation and Development, 1973.

(31) Weber, R.; Herold, C.; Hollert, H.; Kamphues, J.; Ungemach, L.; Blepp, M.; Ballschmiter, K. Life Cycle of PCBs and Contamination of the Environment and of Food Products from Animal Origin. *Environ. Sci. Pollut. Res.* **2018**, *25*, 16325−16343.

(32) Cui, S.; Qi, H.; Liu, L.-Y.; Song, W.-W.; Ma, W.-L.; Jia, H.-L.; Ding, Y.-S.; Li, Y.-F. Emission of Unintentionally Produced Polychlorinated Biphenyls (UP-PCBs) in China: Has This Become the Major Source of PCBs in Chinese Air? *Atmos. Environ.* **2013**, *67*, 73−79.

(33) Fan, Y.; Ren, M.; Zhang, H.; Geng, N.; Li, Y.; Zhang, N.; Zhao, L.; Gao, Y.; Chen, J. Levels and Fingerprints of Chlorinated Aromatic Hydrocarbons in Fly Ashes from the Typical Industrial Thermal Processes: Implication for the Co-Formation Mechanism. *Chemosphere* **2019**, *224*, 298−305.

(34) Hu, D.; Hornbuckle, K. C. Inadvertent Polychlorinated Biphenyls in Commercial Paint Pigments. *Environ. Sci. Technol.* **2010**, *44*, 2822−2827.

(35) Herkert, N. J.; Jahnke, J. C.; Hornbuckle, K. C. Emissions of Tetrachlorobiphenyls (PCBs 47, 51, and 68) from Polymer Resin on Kitchen Cabinets as a Non-Aroclor Source to Residential Air. *Environ. Sci. Technol.* **2018**, *52*, 5154.

(36) Schneider, A. R.; Stapleton, H. M.; Cornwell, J.; Baker, J. E. Recent Declines in PAH, PCB, and Toxaphene Levels in the Northern Great Lakes as Determined from High Resolution Sediment Cores. *Environ. Sci. Technol.* **2001**, *35*, 3809−3815.

(37) Hung, H.; Katsoyiannis, A. A.; Brorström-Lundén, E.; Olafsdottir, K.; Aas, W.; Breivik, K.; Bohlin-Nizzetto, P.; Sigurdsson, A.; Hakola, H.; Bossi, R.; Skov, H.; Sverko, E.; Barresi, E.; Fellin, P.; Wilson, S. Temporal Trends of Persistent Organic Pollutants (POPs) in Arctic Air: 20 Years of Monitoring under the Arctic Monitoring and Assessment Programme (AMAP). *Environ. Pollut.* **2016**, *217*, 52−61.

(38) Salamova, A.; Pagano, J. J.; Holsen, T. M.; Hites, R. A. Post-1990 Temporal Trends of PCBs and Organochlorine Pesticides in the Atmosphere and in Fish from Lakes Erie, Michigan, and Superior. *Environ. Sci. Technol.* **2013**, *47*, 9109−9114.

(39) Desforges, J.-P.; Hall, A.; McConnell, B.; Rosing-Asvid, A.; Barber, J. L.; Brownlow, A.; De Guise, S.; Eulaers, I.; Jepson, P. D.; Letcher, R. J.; Levin, M.; Ross, P. S.; Samarra, F.; Víkingson, G.; Sonne, C.; Dietz, R. Predicting Global Killer Whale Population Collapse from PCB Pollution. *Science* **2018**, *361*, 1373−1376.

(40) Bellinger, D. C. Applying Methods of the Global Burden of Diseases, Injuries, and Risk Factors Study to Developmental Neurotoxicants: A Commentary. *Environ. Health* **2018**, *17*, 1−6.

(41) Statistics Canada. Population estimates, quarterly. https://www150.statcan.gc.ca/t1/tbl1/en/tv.action?pid=1710000901 (accessed on Feb 16, 2022).

(42) Diamond, M. L.; Melymuk, L.; Csiszar, S. A.; Robson, M. Estimation of PCB Stocks, Emissions, and Urban Fate: Will Our Policies Reduce Concentrations and Exposure? *Environ. Sci. Technol.* **2010**, *44*, 2777−2783.

(43) US EPA. *PCB Inspection Manual*: Washington D.C., 2004.

(44) Canadian Council of Ministers of the Environment. *PCB Transformer Decontamination, CCME EPC-HW-105E*: Ottawa, 1995; p 35.

(45) Bláha, K.; Borůvková, J.; Budňáková, M.; Čupr, P.; Eyblová, D.; Gadas, P.; Gruntorád, J.; Hlavatý, E.; Holoubek, I.; Hůlek, R.; Jarolím, O.; Klánová, J.; Kolář, J.; Krech, L.; Krejčová, V.; Prokeš, R.; Přibylová, P.; Ptáčková, Ž.; Puklová, V.; Straková, H.; Šebková, K.; Špringar, Z.; Štěpánek, P.; Udatný, M.; Vávra, T.; Veselý, I.; Vytopilová, M. *Updated National Implementation Plan for Implementation of Stockholm Convention on Persistent Organic Pollutants in the Czech Republic for the Period 2018-2023*: Prague, 2017.

(46) Ministry of the Environment of the Czech Republic. *Questionnaire on PCB Wastes*: Prague, 2016.

(47) *Electronic Reporting System of the Stockholm Convention (Third Reporting)—Czech Republic*; Stockholm Convention, reported by Czech Republic: Geneva, 2018.

(48) Holoubek, I.; Adamec, V.; Bartoš, M.; Bláha, K.; Bláha, L.; Budňáková, M.; Černá, M.; Čupr, P.; Demnerová, K.; Drápal, J.; Hajšlová, J.; Hanzálková, M.; Holoubková, I.; Hrabětová, S.; Jech, L.; Klánová, J.; Kohoutek, J.; Kužílek, V.; Machálek, P.; Matějů, V.; Matoušek, J.; Matoušek, M.; Mestřík, V.; Novák, J.; Ocelka, T.; Pekárek, V.; Petrlík, J.; Petira, O.; Provazník, K.; Punčochář, M.; Rieder, M.; Ruprich, J.; Sáňka, M.; Tomaniová, M.; Vácha, R.; Volka, K.; Zbíral, J. *The National Implementation Plan for Implementation of the Stockholm Convention in the Czech Republic*: Brno, 2006.

(49) U.S. Environmental Protection Agency (EPA). Registering Transformers Containing Polychlorinated Biphenyls (PCBs). https://www.epa.gov/pcbs/registering-transformers-containing-polychlorinated-biphenyls-pcbs (accessed on Nov 1, 2021).

(50) US Environmental Protection Agency (EPA). Managing Remediation Waste From Polychlorinated Biphenyls (PCBs) Cleanups. https://www.epa.gov/pcbs/managing-remediation-waste-polychlorinated-biphenyls-pcbs-cleanups (accessed on Nov 1, 2021).

(51) Shanahan, C. E.; Spak, S. N.; Martinez, A.; Hornbuckle, K. C. Inventory of PCBs in Chicago and Opportunities for Reduction in Airborne Emissions and Human Exposure. *Environ. Sci. Technol.* **2015**, *49*, 13878−13888.

(52) Gimlin, P. *Environmental Protection Specialist*. Personal Communication [September 2020]; US EPA: Washington, DC, 2020.

(53) UNEP. *Report on Progress towards the Elimination of Polychlorinated Biphenyls*. UNEP/POPS/COP.9/INF/10, 2019.

(54) Sri Lanka Ministry of Environment. *National Implementation Plan under the Stockholm Convention on POPs for Sri Lanka*; Persistent Organic Pollutants (POPs) Project, Ministry of Environment: Colombo, Sri Lanka, 2004.

(55) Republic of Nauru. *National Implementation Plan for Persistent Organic Pollutants*; POPs, 2012.

(56) Environmental Affairs Department. *National Implementation Plan (NIP) for the Management of Persistent Organic Pollutants*; Malawi Ministry of Mines, Natural Resources, and Environment: Lilongwe, Malawi, 2005.

(57) Undersecretariat for Environmental Management. *National Implementation Plan of the Stockholm Convention in the Dominican*

Republic; Ministry of the Environment and Natural Resources (SEMARENA), United Nations Development Programme (UNDP): Santo Domingo, Dominican Republic, 2008.

(58) Environmental Protection Agency. *National Implementation Plan of the Stockholm Convention on Persistent Organic Pollutants*: Accra: Ghana, 2007.

(59) Ministry of Sustainable Development and Tourism. *National Implementation Plan for the Stockholm Convention 2014−2021*: Podgorica, Montenegro, 2013.

(60) Global Environment Facility. Disposal of PCB Oils Contained in Transformers and Disposal of Capacitors Containing PCB in Southern Africa. https://www.thegef.org/project/disposal-pcb-oils-contained-transformers-and-disposal-capacitors-containing-pcb-southern (accessed on May 20, 2021).

(61) Environment and Climate Change Canada. Questionnaire for the review of progress towards the elimination of PCB in accordance with paragraph (h) of part II of Annex A. http://www.pops.int/Countries/CountryProfiles/tabid/4501/Default.aspx (accessed on May 20, 2021).

(62) Manhart, J. *POPs and PCBs Management in the Czech Republic*; Ministry of the Environment, Environmental Damage Department: Prague, 2009.

(63) PCB Regulations. *PCB Regulations (SOR/2008-273)*, 2018.

(64) Treasury Board of Canada Secretariat. *Chemicals Management Plan*, 2022.

(65) Robson, M.; Melymuk, L.; Csiszar, S. A.; Giang, A.; Diamond, M. L.; Helm, P. A. Continuing Sources of PCBs: The Significance of Building Sealants. *Environ. Int.* **2010**, *36*, 506−513.

(66) Federal Ministry for the Environment Nature Conservation Building and Nuclear Safety. Questionnaire for the review of progress towards the elimination of PCB in accordance with paragraph (h) of part II of Annex A. http://www.pops.int/Countries/CountryProfiles/tabid/4501/Default.aspx (accessed on May 20, 2021).

(67) Federal Office for the Environment. Questionnaire for the review of progress towards the elimination of PCB in accordance with paragraph (h) of part II of Annex A. http://www.pops.int/Countries/CountryProfiles/tabid/4501/Default.aspx (accessed on May 20, 2021).

(68) Canada Gazette. *Canada Gazette*: Ottawa November, 2006; pp 1−65.

(69) van den Berg, M.; Kypke, K.; Kotz, A.; Tritscher, A.; Lee, S. Y.; Magulova, K.; Fiedler, H.; Malisch, R. WHO/UNEP global surveys of PCDDs, PCDFs, PCBs and DDTs in human milk and benefit-risk evaluation of breastfeeding. *Arch. Toxicol.* **2017**, *91*, 83−96.

(70) Komprda, J.; Komprdová, K.; Domínguez-Romero, E.; Mikeš, O.; Řiháčková, K.; Čupr, P.; Černá, M.; Scheringer, M. Dynamics of PCB Exposure in the Past 50 Years and Recent High Concentrations in Human Breast Milk: Analysis of Influencing Factors Using a Physiologically Based Pharmacokinetic Model. *Sci. Total Environ.* **2019**, *690*, 388−399.

(71) ELSA. *PCB in Der Elbe, Eigenschaften, Vorkommen Und Trends Sowie Ursachen Und Folgen Der Erhöhten Freisetzung Im Jahr 2015*, 2016.

(72) U.S. EPA. *PART 761—Polychlorinated Biphenyls (PCBs) Manufacturing, Processing, Distribution in Commerce and Use Prohibitions*; TSCA, 2018.

(73) Weber, R.; Watson, A.; Forter, M.; Oliaei, F. Review Article: Persistent organic pollutants and landfills - a review of past experiences and future challenges. *Waste Manag. Res.* **2011**, *29*, 107−121.

(74) Bernevi Rex, G. *Inventory and Clearance of PCBs in Buildings and Facilities*. REPORT 688; Swedish Environmental Protection Agency: Stockholm, 2019.

(75) Herrick, R. F.; McClean, M. D.; Meeker, J. D.; Baxter, L. K.; Weymouth, G. A. An Unrecognized Source of PCB Contamination in Schools and Other Buildings. *Environ. Health Perspect.* **2004**, *112*, 1051−1053.

(76) Harrad, S.; Diamond, M. New Directions: Exposure to Polybrominated Diphenyl Ethers (PBDEs) and Polychlorinated Biphenyls (PCBs): Current and Future Scenarios. *Atmos. Environ.* **2006**, *40*, 1187−1188.

APPENDIX 3

Venier, Marta, Ondřej Audy, Šimon Vojta, Jitka Bečanová, Kevin Romanak, Lisa Melymuk, Martina Krátká, Petr Kukucka, Joseph Okeme, Amandeep Saini, Miriam L. Diamond, Jana Klanova. 2016. "Brominated Flame Retardants in Indoor Environment - Comparative Study of Indoor Contamination from Three Countries." *Environment International* 94: 150-60. https://doi.org/10.1016/j.envint.2016.04.029

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Brominated flame retardants in the indoor environment — Comparative study of indoor contamination from three countries

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ABSTRACT

Concentrations of more than 20 brominated flame retardants (FRs), including polybrominated diphenyl ethers (PBDEs) and emerging FRs, were measured in air, dust and window wipes from 63 homes in Canada, the Czech Republic and the United States in the spring and summer of 2013. Among the PBDEs, the highest concentrations were generally BDE-209 in all three matrices, followed by Penta-BDEs. Among alternative FRs, EHTBB and BEHTBP were detected at the highest concentrations. DBDPE was also a major alternative FR detected in dust and air. Bromobenzenes were detected at lower levels than PBDEs and other alternative FRs; among the bromobenzenes, HBB and PBEB were the most abundant compounds. In general, FR levels were highest in the US and lowest in the Czech Republic — a geographic trend that reflects the flame retardants' market. No statistically significant differences were detected between bedroom and living room FR concentrations in the same house ($n = 10$), suggesting that sources of FRs are widespread indoors and mixing between rooms. The concentrations of FRs in air, dust, and window film were significantly correlated, especially for PBDEs. We found a significant relationship between the concentrations in dust and window film and in the gas phase for FRs with log K_{OA} values <14, suggesting that equilibrium was reached for these but not compounds with log K_{OA} values >14. This hypothesis was confirmed by a large discrepancy between values predicted using a partitioning model and the measured values for FRs with log K_{OA} values >14.

1. Introduction

Because the indoor environment is an important human exposure route for semi-volatile organic compounds (SVOCs), such as brominated flame retardants (BFRs), knowledge of their levels and partitioning between different indoor matrices is essential for evaluating their impact on human health. Most studies on BFRs in the indoor environment have focused on their presence in dust because of its putative contribution to exposure [e.g., Harrad et al. (2008b); Wilford et al. (2004)]. Although, some studies have documented air concentrations indoors (Abdallah et al., 2008; Dodson et al., 2015), significantly fewer studies have dealt with accumulation of BFRs on surfaces such as window films (Bennett et al., 2015; Butt et al., 2004; Cetin and Odabasi, 2011). Of these studies, only Bennett et al. (2015) compared indoor film concentrations with those of other indoor matrices, i.e. air and dust.

Indoor air can be sampled using passive or active techniques, and each has its advantages and disadvantages. Passive samplers are easy to deploy and are unobtrusive, which is important in an indoor setting. These samplers do not require electricity, but they need to be deployed for several weeks, providing an integrated measurement over this time period. In comparison, active samplers are bulky and noisy, which is particularly problematic indoors. They require trained personnel to be deployed, but they can be left at the site for shorter periods. The most common passive air sampling design uses a polyurethane foam (PUF) disk enclosed in a stainless steel bowl (Shoeib and Harner, 2002). With knowledge of sampling rates, one can calculate time-integrated air concentrations for compounds mainly present in the gas phase (Bohlin et al., 2014a, 2014b; Saini et al., 2015). Recent studies have shown that these samplers can also provide reliable results for higher molecular weight compounds that are found mainly in the particulate phase (Bohlin et al., 2014a; Harner et al., 2013; Harrad and Abdallah, 2008a; Peverly et al., 2015).

Indoor dust is a complicated, heterogeneous matrix for which different sampling approaches have been used. The most common technique is to collect floor dust, although in some circumstances undisturbed settled dust on other surfaces can be used (Björklund et al., 2012; de Wit et al., 2012; Lioy et al., 2002). Comparisons between different studies

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can be confounded by differences in sampling methods, within room spatial variability, and by the presence in the room of specific products (e.g. electronics) (Harrad et al., 2009; Muenhor and Harrad, 2012). Recently, Li et al. (2015) found a strong positive correlation between the power consumption of electronics and PBDE levels in air and dust in a large room, which they attributed to heat generated from in-use electronics promoting the release of these compounds. The least common medium studied in the indoor setting is the film formed on hard surfaces by condensation and deposition of gas-phase chemicals and particles respectively (Diamond et al., 2000; Gingrich et al., 2001). Windows are convenient to use for sampling the film because the glass is inert, because of the lack of contamination that could arise from other surfaces, and because the film can be removed quantitatively from this surface. The most common approach to windows film sampling employs pre-cleaned wipes (Butt et al., 2004).

With the control and subsequent decline and cessation of production of polybrominated diphenyl ethers (PBDEs) in the last decade in many countries, other brominated flame retardants (BFRs) have risen in importance. Even though some of these alternative flame retardants have been produced for a long time, most of them have only came to the attention of the public and the global scientific community in the past few years. While levels of PBDEs in the environment are generally stable or decreasing (Crimmins et al., 2012; Ma et al., 2013), concentrations of “new” brominated flame retardants, notably 2-ethylhexyl tetrabromobenzoate (EHTBB) and *bis*(2-ethylhexyl)tetrabromophthalate (BEHTBP), are increasing as more products containing these compounds are introduced to the market to replace products containing PBDEs (Dodson et al., 2012; Ma et al., 2012; Stapleton et al., 2011). In spite of this change in the flame retardant market, data regarding the presence of these replacements of alternative compounds in the environment are limited.

In this paper, we report the concentrations of newer and legacy flame retardants in the indoor environments in three different countries (United States, Canada, and the Czech Republic). We collected air, dust, and windows films in 63 private homes, and we measured about 20 brominated flame retardants and Dechlorane Plus, a chlorinated flame retardant. In this paper, we compare concentrations in these three countries, and we put them in the context of their usage in North America and Europe. We also look at differences between rooms in the same home to elucidate possible sources. Finally, we evaluate how these compounds partition between phases (air, dust, and window film) and evaluate which sampling media provide the most comprehensive characterization of indoor levels.

2. Materials and methods

2.1. Study population

Samples were collected in three different locations: Bloomington, Indiana, United States, Toronto, Canada, and Brno, Czech Republic in May–August 2013. Air, dust, and window film samples were collected from a total of 63 houses and apartments: 20 homes each from the Czech Republic and the U.S. and 23 from Canada. At least one room was sampled in each home (i.e. the main bedroom), and a second room was sampled in 10 houses per country (i.e. the living room). Participation in the campaign was voluntary and did not include any compensation.

On day 1, passive samplers were deployed, and selected windows were cleaned with Kimwipes moistened with 2-propanol until no dirt was visible on the Kimwipes. Participants were asked not to vacuum the room where the sampler was located until completion of the campaign, if at all possible. Participants were interviewed by a field technician to gather information about the house and the household (e.g. electronic equipment and furniture in the sampled rooms, number of occupants, and cleaning and ventilation habits).

2.2. Sample collection

Before sampling, all matrices (PUF disks, nylon vacuum socks, and Kimwipes) were pre-cleaned by Soxhlet extraction (8 h in acetone, then 8 h in toluene), dried, wrapped in aluminum foil, and transported to the site. PUF disks for passive air sampling were exposed to indoor air for 28 days using a single (U.S. and Canada) or double-bowl shaped housing (Czech Republic) (see Fig. S1). Sampling rates for each sampler configuration were calculated in a separate experiment by simultaneously deploying single bowl and double bowl samplers (see Supporting Information for details and Fig. S2). For this study, we used a sampling rate of 1.6 m^3/day for the double bowl sampler and 2.9 m^3/day for the single bowl sampler. These values are consistent with previously reported sampling rates indoors (Zhang et al., 2011). Window film samples were collected after 28 days using pre-cleaned Kimwipes moistened with 2-propanol. Windows were wiped with a succession of Kimwipes until no dirt was visible on the Kimwipes, and all Kimwipes from one window were composited. The sampled area averaged at 0.32 m^2 for Canada, 0.93 m^2 for the U.S., and 1.8 m^2 for the Czech Republic. Floor dust samples from each room were taken using pre-cleaned polyester socks inserted on a vacuum cleaner hose attachment, by vacuuming the largest possible area and recording it. All collected samples were wrapped in clean aluminum foil, sealed, labeled, and subsequently stored at −20 °C until analysis. Pre-cleaned PUF, Kimwipes, and polyester socks, which had been exposed by unsealing the aluminum foil wrap during sample retrievals, were treated as field blanks.

2.3. Target compounds

In this paper, we have focused on the following compounds: polybrominated diphenyl ethers (congeners 28, 47, 66, 85, 99, 100, 153, 154, 183, and 209), hexabromobenzene (HBB), *p*-tetrabromoxylene (*p*-TBX), pentabromobenzene (PBBz), 2-ethylhexyl-2,3,4,5-tetrabromobenzoate (EHTBB), *bis*(2-ethylhexyl)tetrabromophthalate (BEHTBP), 1,2-*bis*(2,4,6-tribromophenoxy)ethane (BTBPE), decabromodiphenylethane (DBDPE), and Dechlorane Plus (*syn* and *anti*). All the analytical standards [except BDE-118, which was purchased from AccuStandard (AccuStandard, New Haven, CT)], were purchased from Wellington Laboratories, Guelph, Canada.

2.4. Sample analysis

The U.S. samples were analyzed in Bloomington, Indiana, U.S. (Indiana University, IU), and the Canadian and Czech samples were analyzed in Brno, Czech Republic (RECETOX). Details of the analytical protocols are given in the SI and are summarized here. Before extraction, all samples were spiked with known amounts of recovery standards. Socks with dust were weighed, the dust was sieved to <500 μm, approximately 100 mg were weighed, and the excess dust was stored in an aluminum foil packet for future use. The sock was rinsed with solvent (30 mL hexane in acetone, 1:1), and the solvent was combined with weighed dust. Dust was sonicated in 30 mL of acetone in hexane 1:1 (*v*:*v*); left to settle for 30 min, and the supernatant was decanted. The procedure was repeated 2 additional times with 10 mL of solvent, and the extracts were combined. At IU, the extract was rotary evaporated to 2 mL and then fractionated on a silica column (3.5% water deactivated) using 25 mL of hexane, 25 mL hexane in dichloromethane 1:1 (*v*:*v*), and 25 mL of dichloromethane in acetone 3:7 (*v*:*v*) as eluting solvents. At RECETOX, the volume of the combined extracts were reduced under a N_2 stream and separated by weight to two aliquots. The first aliquot was 70% of the extract, and it was treated with sulfuric acid-modified silica. The remaining 30% of the extract was cleaned using a standard non-modified silica column. PUF and Kimwipes samples were Soxhlet extracted with 400 mL of acetone in hexane 1:1 (*v*:*v*) for 24 h at IU (Peverly et al., 2015) and with 250 mL of dichloromethane using automated warm Soxhlet extraction (Büchi B-811, Switzerland) at RECETOX. Extracts were fractionated on silica columns as per the dust procedure and eluted with 25 mL of hexane, and 25 mL hexane: dichloromethane 1:1 (*v*:*v*) at IU and with 20 mL of dichloromethane at RECETOX. Extracts were analyzed using GC/MS at IU and GC/HRMS at RECETOX, using previously published methods (Lohmann et al., 2013; Ma et al., 2013). More analytical details are reported in the Supporting Information.

2.4.1. QA/QC

Several measures were taken to ensure data comparability between the different laboratories and the accuracy and reliability of the measurements (see Fig. S3 and Table S1 in the Supporting Information and the related detailed discussion). For the non-PBDE halogenated flame retardants, both the U.S. and Czech laboratories participated in an interlaboratory comparison study [called INTERFLAB Phase I (Melymuk et al. (2015)]. PBDE data from the Czech Republic and Canadian homes were obtained using an isotope dilution method, and the results were recovery corrected. Results were not recovery corrected at IU. Average surrogate recoveries were mostly within the 50–150% range. Solvent blanks were used to evaluate contamination from the laboratory procedures. Blank levels were low, and no correction was necessary. Three field blanks per matrix per country were collected and analyzed. The concentrations of the target compounds in each sample were then compared to the average concentrations in the field blanks (on a country and matrix specific basis — see Table S2 in the Supporting Information for more details) and treated as follows: If the blank level was <10% of the measured level, there was no correction. If the blank level was 10–35% of the measured level, the blank level was subtracted from the measured level. If the blank level was >35% of the measured level, the value was reported as “non-detect”

2.5. Data analysis

Basic and descriptive statistics were calculated using Minitab, and Microsoft Excel software. Plots were generated using SigmaPlot 13 (Systat Software Inc.).

3. Results and discussion

A summary of the results for air, dust, and window films at the three locations is reported in Tables 1–3, respectively. Boxplots for selected compounds are shown in Figs. 1–3.

3.1. PBDEs

BDE-209 was generally the most abundant PBDE congener, followed by the sum of the congeners representing the Penta-BDE commercial mixture (the sum of congeners 28, 47, 66, 85, 99, 100, 153, and 154) — see Fig. 1. The only exceptions to this trend were window film samples from the Czech Republic (CZ), where the levels of BDE-209 were lower than those of Penta-BDEs.

The flame retardant concentrations in the U.S. samples were significantly higher than those measured in samples from the Czech Republic, with Canada (CAN) typically being in between (see Fig. 1). We observed the geographic trend US > CAN > CZ for BDE-209 in dust and window film and for total PBDEs in dust and air ($p < 0.05$). However, the trend was US ≈ CAN > CZ for Penta-BDE in air and dust and for total PBDEs in window film. These geographic trends can be linked to flame retardant use patterns in different regions of the world. North America was the largest global user of both Penta-BDE and Deca-BDE, whereas Europe's usage was far lower (Abbasi et al., 2015). The fact that DecaBDE has been produced in the largest volumes of any commercial PBDEs accounts for the high relative contribution of BDE-209 in all the samples (Earnshaw et al., 2015).

BDE-209 was the most abundant PBDE congener in dust and window film, and BDE-47 and Penta-BDE congeners were the most abundant in air. This distribution is consistent with differences in their vapor pressures and log K_{OA}; the vapor pressure of BDE-209 is 6.2×10^{-10} Pa at 25 ° C, but that of BDE-47 is 3.2×10^{-5} Pa at 25 °C and their log K_{OA} values are 10.686 and 18.423, respectively (Episuite (2012).

An overview of available literature data for indoor studies is given in the Supporting Information (see Table S3). In general, for all matrices, the flame retardant concentrations for the U.S. and Canadian samples

Table 1

Summary results for air samples (pg/m^3) including mean with standard error, median, minimum, maximum, and number of detects. The ANOVA results (calculated on log transformed data) are given as *a*–*c* letters in each line; concentrations for compounds that share a letter are not significantly different from one another at $p < 0.05$.

	Air (pg/m^3)														
	US					CAN					CZ				
	Mean ± SE	Median	Range	N		Mean ± SE	Median	Range	N		Mean ± SE	Median	Range	N	
BDE-28	8.5 ± 5.1	2.2	ND–145	28	b	18 ± 5.4	7.6	1.0–166	34	a	1.2 ± 0.3	0.58	0.10–9.2	29	c
BDE-47	128 ± 32	52	4.5–818	30	a	99 ± 27	39	5.4–759	34	a	3.0 ± 0.7	1.6	0.56–16	29	b
BDE-100	19 ± 10	4.2	ND–272	29	a	4.7 ± 1.0	1.8	0.38–22	34	b	0.19 ± 0.04	0.11	0.05–1.0	29	c
BDE-99	95 ± 51	15	ND–1291	26	a	15 ± 3.3	5.3	1.3–73	34	b	0.46 ± 0.07	0.29	0.16–1.4	29	c
BDE-154	7.6 ± 4.7	0.62	ND–122	27	a	0.80 ± 0.22	0.36	ND–4.4	22	b	0.045 ± 0.003	0.045	ND–0.056	9	c
BDE-153	11 ± 6	5.0	ND–71	12	a	0.60 ± 0.17	0.26	0.06–5.7	34	b	0.072 ± 0.008	0.069	ND–0.10	6	c
BDE-183	2.8 ± 0.7	2.5	ND–5	4	a	1.3 ± 0.32	1.3	ND–1.6	2	b	0.12 ± 0.01	0.12	ND–0.23	28	c
BDE-209	836 ± 475	257	ND–5461	11	a	83 ± 31	49	ND–217	6	ab	11 ± 2	9.4	ND–15	3	b
Penta-BDE	254 ± 85	79	4.9–2291	30	a	140 ± 36	59	8.3–1024	34	ab	5.0 ± 1.1	2.9	0.94–28	29	b
TotBDE	561 ± 202	148	4.9–5756	30	a	155 ± 40	60	8.3–1149	34	b	6.3 ± 1.2	3.0	1.1–28	29	b
pTBX	1.8 ± 0.3	1.9	ND–2.4	6	a	0.66 ± 0.19	0.38	ND–3.4	19	b	0.31 ± 0.03	0.29	ND–0.71	26	b
PBBZ	3.8 ± 1.1	2.5	ND–12	11	b	8.8 ± 2.5	4.8	1.3–74	34	a	4.9 ± 0.8	3.4	0.76–17	29	ab
PBEB	9.0 ± 4.4	1.9	ND–92	21	a	4.0 ± 0.30	3.9	0.95–7.4	34	a	0.66 ± 0.1	0.49	ND–1.4	8	b
HBB	9.6 ± 2.7	4.0	ND–58	26	a	9.9 ± 2.2	5.8	0.85–60	34	a	7.2 ± 1.7	4.6	ND–31	19	a
Tot bromobenzenes	17.6 ± 4.1	9.1	ND–98	28	b	23 ± 3.5	16	5.65–91	34	a	10 ± 1.9	7.6	0.8–41	29	b
EHTBB	23 ± 6	9.2	1.3–142	30	ab	32 ± 10	12	3.1–291	34	a	6.6 ± 0.7	5.5	ND–15	28	b
BEHTBP	16 ± 7	6.0	ND–109	17	a	6.8 ± 2.2	3.1	ND–43	22	ab	3.0 ± 0.3	3.2	ND–5.4	15	b
EHTBB + BEHTBP	32 ± 9	14	1.3–212	30	ab	37 ± 11	14	3.1–293	34	a	8.2 ± 0.8	7.2	ND–17	28	b
BTBPE	0.18 ± 0.03	0.083	ND–0.74	27	c	1.3 ± 0.39	1.0	ND–2.4	4	a	0.49 ± 0.12	0.38	ND–1.4	9	b
syn-DP	0.70 ± 0.26	0.37	ND–4.0	14	b	28 ± 4.9	23	ND–76	22	a					
anti-DP	8.2 ± 3.9	4.1	ND–23	5	b	43 ± 13	25	ND–243	18	ab	65 ±	65	ND–65	1	a
DPsum	3.4 ± 1.8	0.37	ND–27	15	b	61 ± 15	38	ND–316	23	a	65 ±	65	ND–65	1	a
DBDPE	42 ± 28	42	ND–71	2	a	13 ± 2.6	9.2	ND–74	29	b					
f_{anti}	0.87 ± 0.01	0.86	0.85–0.91	4	a	0.50 ± 0.03	0.46	0.3–0.77	17	b					
f_{EHTBB}	0.64 ± 0.05	0.63	0.16–0.95	17	b	0.80 ± 0.04	0.83	0.3–0.99	22	b	0.69 ± 0.05	0.72	0.27–0.89	15	ab

Table 2
Summary results for dust (ng/g) including mean with standard error, median, minimum, maximum, and number of detects. The ANOVA results (calculated on log transformed data) are given as *a–c* letters in each line; concentrations for compounds that share a letter are not significantly different from one another at $p < 0.05$.

	DUST (ng/g)														
	US					CAN					CZ				
	Mean ± SE	Median	Range	N		Mean ± SE	Median	Range	N		Mean ± SE	Median	Range	N	
BDE-28	12 ± 2.2	6.9	0.38–50	30	a	9.0 ± 3.0	3.9	0.21–93	35	b	0.34 ± 0.08	0.23	ND–1.2	18	c
BDE-47	365 ± 60	271	20–1260	30	a	480 ± 127	230	19–3580	35	a	6.3 ± 2.4	3.8	ND–65	26	b
BDE-100	86 ± 12	65	3.2–270	30	a	115 ± 32	42	3.9–901	35	a	1.3 ± 0.59	0.53	ND–17	28	b
BDE-99	634 ± 142	336	20–2800	30	a	534 ± 139	221	23–3830	35	a	7.0 ± 3.1	2.5	ND–84	28	b
BDE-154	35 ± 5.4	30	ND-128	28	a	46 ± 12	18	1.8–344	35	a	0.75 ± 0.29	0.33	ND–6.0	23	b
BDE-153	48 ± 7.6	32	1.7–178	30	a	61 ± 16	25	2.6–479	35	a	1.3 ± 0.41	0.77	ND–8.9	26	b
BDE-183	12 ± 1.5	11	ND–37	29	a	31 ± 8.4	13	2.3–255	35	a	3.4 ± 0.68	1.9	ND–15	24	b
BDE-209	2780 ± 365	2220	75–7450	30	a	1210 ± 206	713	223–4860	35	b	223 ± 39	139	16–788	30	c
Penta-BDE	1210 ± 220	735	47–4410	30	a	1310 ± 338	514	53–8360	35	a	16 ± 6.5	8.0	0.05–193	30	b
TotBDE	4000 ± 446	3650	122–9730	30	a	2550 ± 397	1770	284–9610	35	b	241 ± 40	163	18–797	30	c
pTBX	0.32 ± 0.11	0.21	ND–1.3	11	a	0.39 ± 0.12	0.50	ND–0.53	3	a					
PBBZ	2.6 ± 0.68	1.6	ND–15	22	a	3.3 ± 0.92	3.2	ND–5.6	4	a	0.44 ± 0.10	0.29	ND–2.8	28	b
PBEB	1.4 ± 0.40	0.60	ND–8.4	26	a	1.2 ± 0.12	1.2	ND–1.7	8	a					
HBB	13 ± 2.4	7.2	0.92–46	30	a	6.4 ± 1.3	6.1	ND–11	6	a	2.1 ± 0.45	1.4	ND–6.4	13	b
Tot bromobenzenes	16 ± 2.3	13	1.2–48	30	a	3.9 ± 1.0	1.9	ND–13	16	b	1.4 ± 0.37	0.7	ND–9.2	28	c
EHTBB	918 ± 520	240	ND–15,400	29	b	2410 ± 613	966	121–15,300	35	a	17 ± 5.4	7.8	ND–150	28	c
BEHTBP	2540 ± 900	624	112–22,800	30	a	2650 ± 1170	431	69–34,500	35	a	60 ± 13	42	ND–373	29	b
EHTBB + BEHTBP	3423 ± 1065	797	155–23,648	30	a	5058 ± 1427	1570	194–36,047	35	a	77 ± 14	55	ND–395	29	b
BTBPE	22 ± 7.7	8.5	ND–204	29	a	27 ± 6.8	12	ND–157	31	a	5.8 ± 1.1	3.9	ND–29	28	b
syn-P	5.0 ± 0.80	3.3	ND–16	29	b	8.8 ± 3.1	4.6	ND–99	31	b	24 ± 10	15	ND–62	5	a
anti-DP	45 ± 15	14	ND–311	28	a	35 ± 18	15	ND–634	34	a	45 ± 15	20	ND–215	14	a
DPsum	48 ± 15	18	ND–322	29	a	43 ± 21	22	ND–732	34	a	53 ± 19	20	ND–277	14	a
DBDPE	367 ± 119	148	ND–3140	29	a	95 ± 60	15	ND–2060	34	b	20 ± 7	4.7	ND–114	23	c
f_{anti}	0.81 ± 0.02	0.8161	0.57–0.97	28	a	0.71 ± 0.03	0.71	0.35–0.92	31	b	0.81 ± 0.01	0.81	0.78–0.86	5	ab
$\mathrm{f}_{\text{EHTBB}}$	0.30 ± 0.036	0.28	0.0075–0.81	29	b	0.6 ± 0.041	0.71	0.044–0.91	35	a	0.24 ± 0.043	0.14	0.054–0.68	28	b

are within the same range of values previously reported, but the concentrations for the Czech Republic are lower than those found in the literature for other European countries. This anomaly may be a result of an over-representation of the United Kingdom (UK) in the European measurements. In the UK, the levels for FRs have been historically higher than the rest of Europe because of stricter flammability standards compared to continental or central Europe (Harrad et al. (2008b). An alternative explanation is that many of these UK measurements were obtained about 10 years ago, when the flame retardant market was still heavily dominated by PBDEs.

3.2. EHTBB and BEHTBP

EHTBB and BEHTBP are the two main brominated components of two commercial mixtures, Firemaster 550 and Firemaster BZ-54, which are now considered major replacements of the withdrawn

Table 3
Summary results for window film (ng/m^2) including mean with standard error, median, minimum, maximum, and number of detects. The ANOVA results (calculated on log transformed data) are given as *a–c* letters in each line; concentrations for compounds that share a letter are not significantly different from one another at $p < 0.05$.

	Window film (ng/m^2)														
	US					CAN					CZ				
	Mean ± SE	Median	Range	N		Mean ± SE	Median	Range	N		Mean ± SE	Median	Range	N	
BDE-28	0.14 ± 0.04	0.091	ND–0.75	21	a	0.13 ± 0.029	0.1	ND–0.65	27	a	0.0053 ± 0.0006	0.0049	ND–0.011	14	b
BDE-47	5.3 ± 1.4	2.7	ND–28	21	a	5.8 ± 3.5	3.5	ND–13	3	a	0.46 ± 0.069	0.39	ND–1.1	14	b
BDE-100	1.8 ± 0.5	0.71	ND–7.6	20	a	0.69 ± 0.38	0.69	ND–1.1	2	ab	0.14 ± 0.025	0.10	ND–0.4	14	b
BDE-99	5.1 ± 1.0	3.0	ND–20	20	a	2.5 ± 1.2	2.5	ND–3.7	2	ab	0.74 ± 0.13	0.52	ND–2.0	14	b
BDE-154	0.33 ± 0.06	0.23	ND–1.3	22	a	0.23 ± 0.05	0.27	ND–0.35	6	a	0.058 ± 0.01	0.042	ND–0.17	19	b
BDE-153	0.74 ± 0.27	0.30	ND–4.1	19	a	0.33 ± 0.06	0.30	ND–0.55	7	a	0.077 ± 0.02	0.044	ND–0.27	23	b
BDE-183	0.35 ± 0.08	0.28	ND–1.1	13	a	0.50 ± 0.14	0.17	ND–3.9	33	a	0.014 ± 0.001	0.013	0.003–0.028	29	b
BDE-209	45 ± 9	53	ND–54	3	a	12 ± 2.9	5.5	ND–69	33	b	0.61 ± 0.14	0.40	ND–3.1	21	c
Penta-BDE	9.9 ± 2.4	6.1	ND–60	29	a	1.1 ± 0.7	0.14	ND–19	28	b	0.98 ± 0.22	0.86	ND–4.1	23	b
TotBDE	14 ± 4	7.0	0.02–78	30	a	13 ± 3	6.5	1.1–73	34	a	1.2 ± 0.24	0.98	0.005–5.1	29	b
pTBX	0.053 ± 0.026	0.044	ND–0.12	4											
PBBZ	0.031 ± 0.006	0.027	ND–0.074	10	b	0.098 ± 0.017	0.067	0.015–0.27	25	a	0.0051 ± 0.0008	0.0042	0.001–0.02	28	c
PBEB	0.028 ± 0.006	0.016	ND–0.1	22	b								—		
HBB	0.60 ± 0.23	0.12	ND–2.9	17	a	0.58 ± 0.27	0.24	ND–8.1	29	a	0.036 ± 0.003	0.038	ND–0.056	19	b
Tot Bromobenzenes	0.48 ± 0.17	0.11	ND–2.9	24	a	0.57 ± 0.27	0.27	ND–8.1	32	a	0.0285 ± 0.004	0.032	0.001–0.069	29	b
EHTBB	6.0 ± 1.6	2.0	ND–34	28	a	5.0 ± 1.3	2.4	0.59–35	34	a	0.047 ± 0.007	0.041	ND–0.099	14	b
BEHTBP	13 ± 3	6.4	ND–46	18	a	2.0 ± 0.9	0.83	ND–20	22	b	0.30 ± 0.07	0.11	ND–1.2	26	c
EHTBB + BEHTBP	18 ± 3	11	ND–68	29	a	6.3 ± 1.8	3.2	0.61–54	34	b	0.31 ± 0.07	0.11	ND–1.3	26	c
BTBPE	0.50 ± 0.22	0.16	ND–3.7	19	a	0.56 ± 0.22	0.15	ND–5.7	28	a	0.032 ± 0.009	0.017	ND–0.23	28	b
*syn*P	0.92 ± 0.40	0.40	ND–6.5	16	c	12 ± 5	6.7	ND–28	5	a	2.8 ± 1.5	0.97	ND–12	7	b
*anti*DP	1.9 ± 0.7	1.1	ND–8.8	12	b	9.4 ± 2.5	7.2	ND–21	7	a	4.5 ± 2.5	0.99	ND–22	10	b
DPsum	2.2 ± 0.9	1.5	ND–15	17	b	16 ± 5.0	8.4	ND–48	8	a	6.5 ± 3.6	1.4	ND–34	10	ab
DBDPE	9.7 ± 3.4	5.4	ND–56	15	a	1.6 ± 0.8	0.60	ND–6.8	8	b	0.15 ± 0.06	0.068	ND–0.84	13	c
f_{anti}	0.65 ± 0.06	0.7108	0.21–0.859	11	a	0.462 ± 0.0	0.45	0.39–0.57	4	a	0.488 ± 0.07	0.45	0.31–0.84	7	ab
$\mathrm{f}_{\text{EHTBB}}$	0.32 ± 0.04	0.32	0.026–0.86	25	b	0.74 ± 0.03	0.77	0.28–0.95	22	a	0.14 ± 0.02	0.15	0.026–0.25	13	c

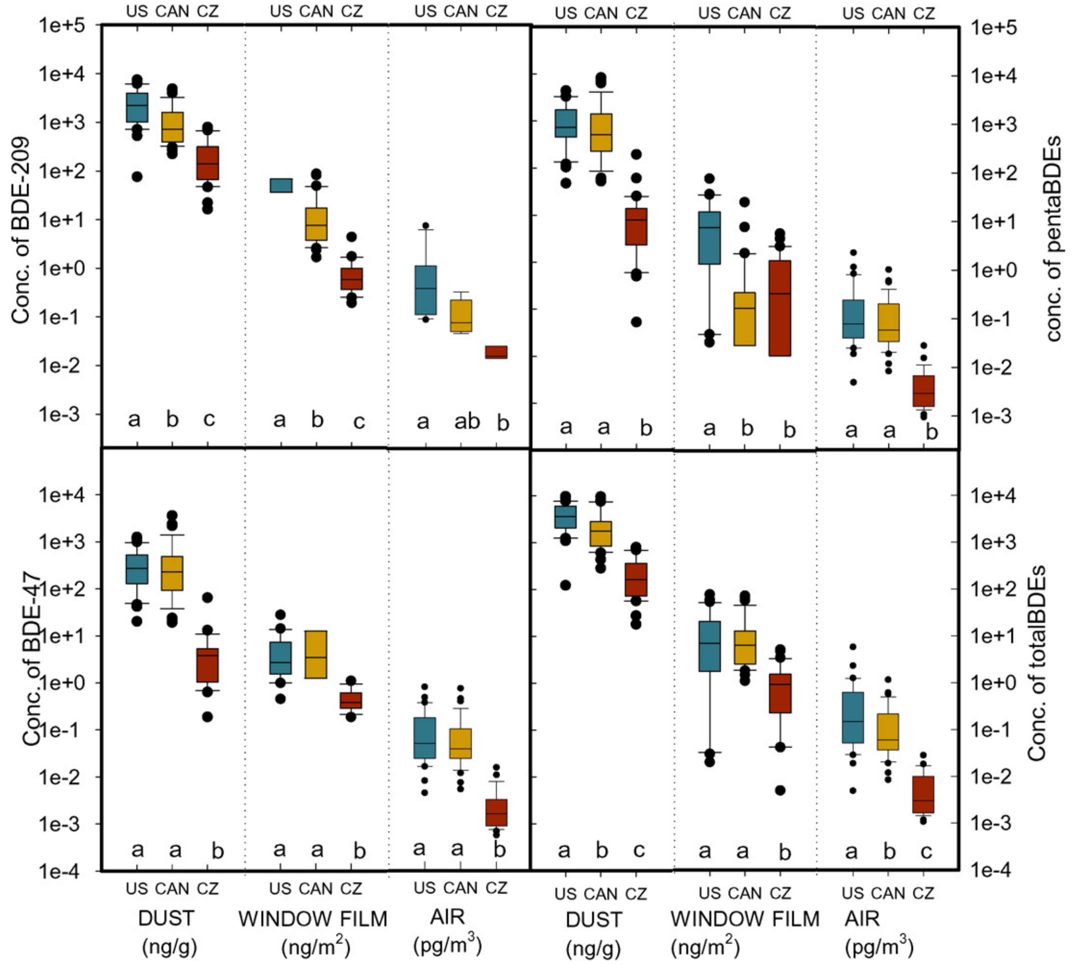

Fig. 1. Boxplots of concentrations for selected PBDEs congeners. The plots share the same y-axis concentration scales although the units are different for each matrix: dust is ng/g, window film is ng/m^2, and air is pg/m^3. Within each matrix, boxes that share the same letter are not statistically significantly different at a 5% level in an ANOVA analysis using the Tukey's test.

Penta-BDE commercial mixture (Stapleton et al., 2008). BEHTBP is also sold and used separately as a flame retardant in the commercial mixture DP-45. Recently, Abbasi et al. (2016) reported finding both these compounds in electronic equipment such as personal computers and televisions. Among the alternative flame retardants measured in this study, EHTBB and BEHTBP were the major non-PBDE components in dust and air but not in window film. Concentrations of EHTBB + BEHTBP were generally higher in the U.S. and Canada than in the Czech Republic, an observation which reflects the higher usage of FRs in North America than in Europe (see Fig. 2).

The EHTBB fraction was calculated from the concentrations of EHTBB and BEHTBP as follows: f_{EHTBB} = EHTBB / (EHTBB + BEHTBP), and these values are plotted in Fig. 2, rightmost boxes. The median ranges were 0.15–0.77 in window film, 0.14–0.71 in dust, and 0.63–0.83 in air.

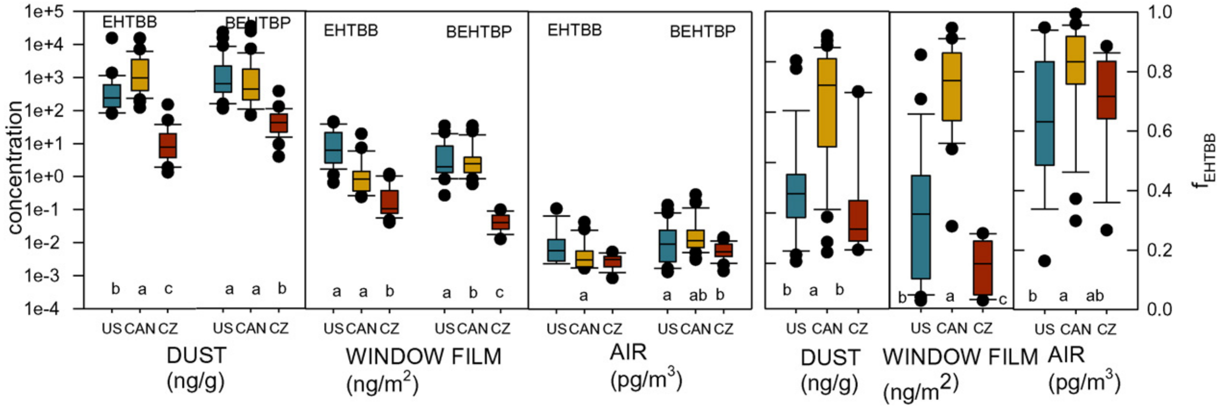

Fig. 2. Boxplots of concentrations for EHTBB, BEHTBP and the f_{EHTBB} ratio. The leftmost three plots share the same left y-axis concentration scale, and the rightmost three plots share the same right y-axis scale. The units are different for each matrix: dust is ng/g, window film is ng/m^2, and air is pg/m^3. Within each matrix, boxes that share the same letter are not statistically significantly different at a 5% level in an ANOVA analysis using the Tukey's test.

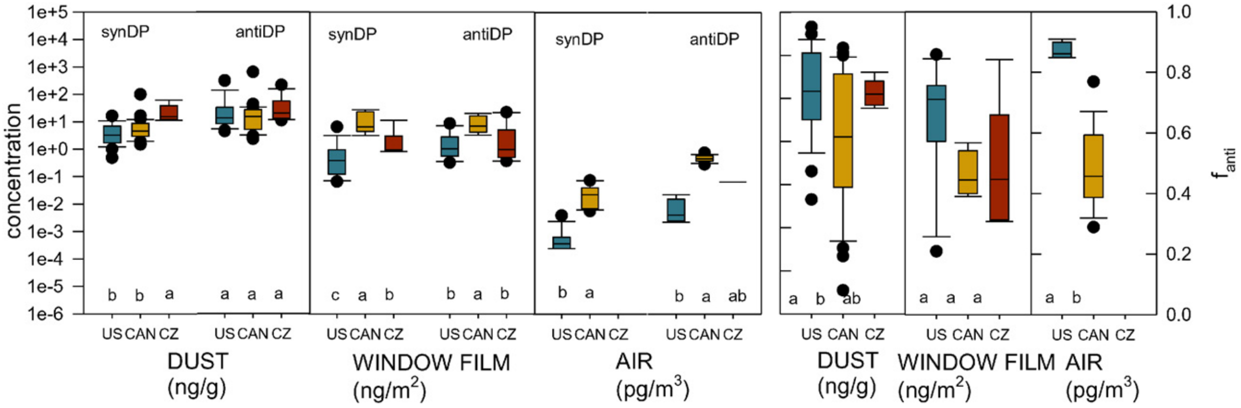

Fig. 3. Boxplots of concentrations for *syn*-DP, *anti*-DP and the f_{anti} ratio. The leftmost three plots share the same left y-axis concentration scale, and the rightmost three plots share the same right y-axis scale. The units are different for each matrix: dust is ng/g, window film is ng/m^2, and air is pg/m^3. Within each matrix, boxes that share the same letter are not statistically significantly different at a 5% level in an ANOVA analysis using the Tukey's test.

High ratios indicate the predominance of EHTBB and low ratios the predominance of BEHTBP. Low ratios were also recently observed in indoor samples from Norway (Cequier et al., 2014). In this study, the f_{EHTBB} ratio was generally highest in samples from Canada and lowest in those from the Czech Republic. Previous studies have reported values of the f_{EHTBB} ratio of 0.69 in outdoor air in Toronto (Shoeib et al., 2014), 0.70 in outdoor air in Chicago (Peverly et al., 2015), 0.26–0.54 in outdoor air from the Great Lakes area (Ma et al., 2012), and 0.59–0.81 in indoor dust (Stapleton et al., 2008). Cao et al. (2014) reported variable f_{EHTBB} ratios with different types of dust. In commercial mixtures, the f_{EHTBB} ratio was 0.77 ± 0.03 in Firemaster 550 (Ma et al., 2012) and 0.70 in Firemaster BZ-54.

The f_{EHTBB} ratios measured in this study for air are similar to those reported in previous studies, but the ratios for window wipes and dust are lower.

A higher f_{EHTBB} ratio is expected in air than in window films and dust because of the different volatilities of these two compounds: EHTBB is expected to have a higher vapor pressure than BEHTBP; hence, EHTBB is likely to be relatively more concentrated in air than BEHTBP. Another explanation for the lower ratios observed for window films and dust could be more photodegradation of EHTBB relative to BEHTBP (Davis and Stapleton, 2009).

The differences in the f_{EHTBB} ratio among countries (high in Canada and low in the Czech Republic) can be explained by looking at sources: lower f_{EHTBB} ratios could be due to BEHTBP-specific sources. BEHTBP was used as flame retardant in neoprene and polyvinylchloride (Andersson et al., 2006), and in the commercial mixture DP-45, BEHTBP was the sole component. However, the concentrations of EHTBB and BEHTBP were significantly correlated (Pearson correlation coefficient = 0.939, $p < 0.0001$; see Fig. S4), suggesting that they have a similar source. Previous studies have shown that the concentrations of EHTBB and BEHTBP were significantly correlated in dust from Belgium and from UK homes (Ali et al., 2011a), but other studies found no such correlations (Ali et al., 2011b; Shoeib et al., 2012). More data are needed to reconcile the seemingly contradictory information provided by the lower than expected ratios and yet significant correlation.

3.3. Dechlorane Plus

Dechlorane Plus (DP) is a chlorinated flame retardant used mainly in electronics and electrical cable coatings as an alternative to Deca-BDE (Dodson et al., 2012). Boxplots of the concentrations of DP in all the samples are shown in Fig. 3. DP was abundant in both the U.S. and Canadian air samples, with median total DP concentrations of 0.37 and 38 pg/m^3, respectively. DP was detected in only one Czech air sample. In this case, only the *anti* isomer was detected at a relatively high concentration of 65 pg/m^3. Interestingly, in the U.S. and Canadian samples, the *syn* isomer was detected more often than the *anti* isomer but at lower concentrations. In the window wipe samples, the highest levels of DP were found in Canada, with a median level of 8.4 ng/m^2. However, the Canadian samples also had the lowest DP detection frequency at only 24%. In U.S. and Czech window wipes samples, DP was found in 57 and 33% of samples, respectively, at similar levels (medians: 1.5 and 1.4 ng/m^2, respectively). DP was detected in most of the U.S. and Canadian dust samples, but only in half of the Czech samples. Nevertheless, levels in dust between countries are statistically indistinguishable ($p > 0.05$), with medians of 18, 22, and 20 ng/g in the U.S., Canadian, and Czech samples, respectively.

There are only a few studies that have measured DP levels in indoor samples and none that measured it in windows wipes (see Table S3). Cequier et al. (2014) reported DP levels of <10 pg/m^3 in indoor air in Norway using active samplers, levels which are similar to the results reported here. Concentrations of DP in dust varied largely based on where they were collected. Median DP values in the U.S. dust were 4.5–10 ng/g (Dodson et al., 2012; Johnson et al., 2013). In China, median DP levels were 4.0–65 ng/g in rural and urban areas, but they reached values as high as 3500 ng/g in the vicinity of an e-waste recycling facility (Chen et al., 2014; Li et al., 2015; Zheng et al., 2010, 2015). Newton et al. (2015) measured DP concentrations of <10 ng/g in dust samples in Sweden. In Canada, Abbasi et al. (2016) reported DP concentrations of up to 283 ng/g (with a geomean value of 8.6 ng/g), values which are comparable to those reported by Shoeib et al. (2012). In general, our levels for dust are similar to those previously reported for the U.S. and for rural and urban China, but our levels are much lower than those around the Chinese e-waste recycling facility.

The f_{anti} ratios (the ratio of the *anti*-DP concentration to the total DP concentration) in these samples are shown in Fig. 3, rightmost boxes. For reference, $f_{\text{anti}} = 0.75$ for the U.S. commercial product and 0.60 for the Chinese product (Sverko et al., 2011). The medians for the f_{anti} ratios in dust samples were 0.82 in the U.S. samples and 0.71 in the Canadian samples, values that were significantly different from one another ($p < 0.05$). The medians for the f_{anti} ratios in window wipes samples ranged between 0.45 for the Canadian and Czech Republic samples to 0.71 for the U.S. samples, but they were statistically indistinguishable ($p > 0.05$). The median f_{anti} ratios in the air samples were significantly higher in the U.S. samples than in the Canadian samples (0.86 vs. 0.46; $p < 0.001$). The values of the f_{anti} ratio in these samples suggest that the source of DP in the U.S. and Canada are products containing DP manufactured in North America rather than imported from China. Cequier et al. (2014) reported f_{anti} in the range 0.65–0.75. Newton et al. (2015) reported f_{anti} as low as 0.20 in a dust sample collected from a Stockholm home in an area around electrical cables. Abbasi et al. (2016) found f_{anti} of 0.69 in dust samples from 35 Toronto homes. Cao et al. (2014) reported f_{anti} of 0.8 in indoor and outdoor dust samples in Beijing, China. Several f_{anti} values lower than in the commercial mixtures were also reported in environmental compartments by Sverko et al. (i.e. outdoor air), who suggested preferential degradation of the *anti* isomer (Sverko et al., 2011).

3.4. Bromobenzenes

This group includes hexabromobenzene (HBB), pentabromoethylbenzene (PBEB), pentabromobenzene (PBBz) and tetrabromo-*p*-xylene (pTBX). Information about current and past uses of bromobenzenes is scarce, making it difficult to investigate their sources and fates. HBB is used in paper, wood, textiles, electronics, and plastic goods (Covaci et al., 2011). PBEB is an additive flame retardant used in thermoset polyester resins for applications such as circuit boards, textiles, adhesives, and wire and cable coatings (Covaci et al., 2011). Abbasi et al. (2016) reported PBBz and PBEB in about 30 and <10% of surface wipes taken from electronic products sampled in 35 and 10 Toronto homes and offices, respectively. Interestingly, they found PBBz in the casings of four cathode ray tube (CRTs) televisions that were >10 years old. In our study, the median sum of bromobenzene concentrations ranged between 7.6 pg/m^3 for Czech and 15.7 pg/m^3 in Canadian air samples, between 0.032 ng/m^2 for Czech and 11.5 ng/m^2 for Canadian window film samples, and between 0.70 ng/g for Czech and 15.7 ng/g for U.S. in dust (see Tables 1–3 for details). In general, the concentrations of total bromobenzenes were significantly lower in the Czech samples compared to U.S. and Canadian samples ($p < 0.001$). In air, bromobenzene concentrations in samples from Canada were significantly higher than those from the U.S. and from the Czech Republic ($p < 0.001$).

Among the bromobenzenes, PBEB was not detected in the Czech dust samples or in the window wipes from Canada or the Czech Republic. In air, HBB and PBEB concentrations in the U.S. and Canadian samples were similar to each other, and they were significantly higher than those in the Czech Republic ($p < 0.05$). Both the levels and the spatial trends for bromobenzenes confirm their relatively low current or past usage in the flame retardant market compared to PBDEs and other alternative compounds such as EHTBB or BEHTBP, as well as their relatively higher usage in North America compared to Europe.

3.5. Decabromodiphenylethane (DBDPE)

DBDPE was introduced in the market as a replacement for BDE-209 in the early 1990s, and it was first identified in the environment in 2003 (Ricklund et al., 2008). Environmental data suggest that BDE-209 and DBDPE have been used independently and that the substitution of DBDPE for BDE-209 has not progressed very far, especially in the U.S. -which could be due to the lack of regulations (Ricklund et al., 2008). Based on DBDPE's low vapor pressure (Dungey and Akintoye, 2007), we expected to find DBDPE mostly in the dust and window wipes samples. Rather, DBDPE was detected in 85% of the Canadian air samples, in 2 of the U.S. samples, and in none of the Czech samples (see Tables 1-3). The median level for the Canadian air samples was significantly lower than for the U.S. samples (9.2 pg/m^3 vs. 42 pg/m^3). In the dust and window film samples, the concentration of DBDPE followed the spatial trend: US > CAN > CZ ($p < 0.01$). Median concentrations in dust ranged from 148 ng/g in the U.S. to 4.73 ng/g in the Czech Republic samples and in window film from 5.40 ng/m^2 in the U.S. to 0.068 ng/m^2 in the Czech Republic. The relatively high variability in the measurement of this compound might be contributing to some of these geographic patterns (Melymuk et al., 2015).

Concentrations of DBDPE in dust in this study are consistent with previous studies that reported median concentrations of 51–138 ng/g in the U.S. (Dodson et al., 2012; Stapleton et al., 2008), a mean and geometric mean of 221 and 0.5 (range: <detection-5500) ng/g in Canada (Abbasi et al., 2016), and 10–200 ng/g in Europe (Ali et al., 2011a; Newton et al., 2015) (see Table S3 for a more extensive list of literature values). DBDPE literature data in air are scarce: Cequier et al. (2014) reported a median of 8.30 pg/m^3 in classrooms in Norway, and Newton et al. (2015) reported a median of <90 pg/m^3 in indoor air in Sweden. Results from this study are comparable. No literature data exist to date for DBDPE in window wipes.

Fig. S5 shows a strong positive relationship between the concentrations of DBDPE and BDE-209, which is the opposite of what would be expected if DBDPE was entering the flame retardant market primarily as a BDE-209 replacement. These data confirm that both these compounds have been used simultaneously both in North America and in Europe.

3.6. 1.2-Bis(2,4,6-tribromophenoxy)ethane (BTBPE)

BTBPE is an additive flame retardant marketed as FF-680 by the Great Lakes Chemical Corporation, now part of Chemtura (Covaci et al., 2011). BTBPE has been used since the 1980s as a replacement for the Octa-BDE commercial mixture.

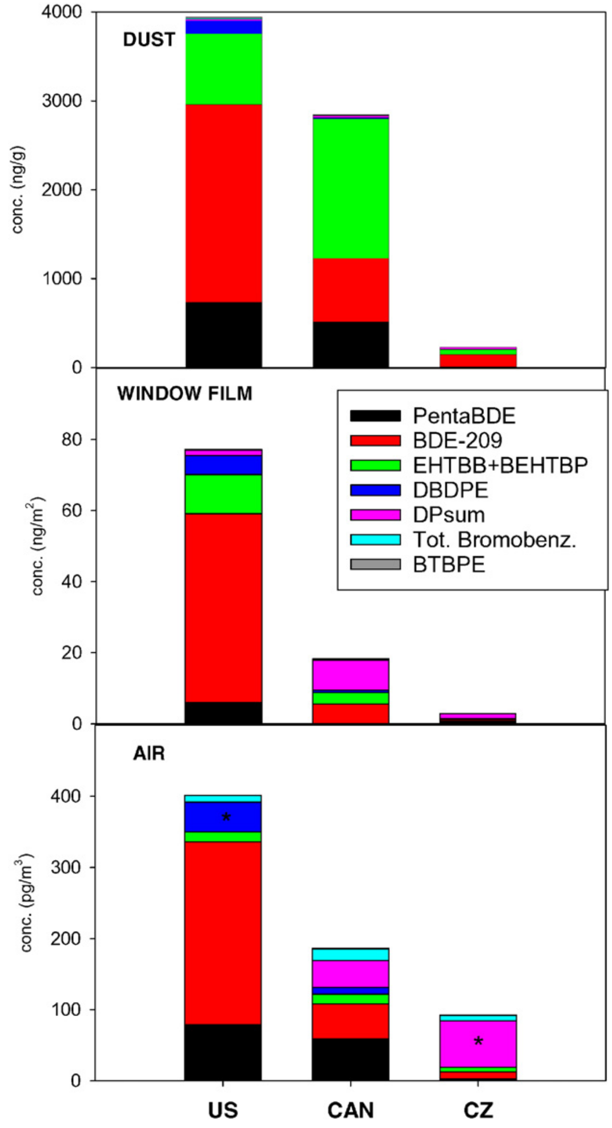

Fig. 4. Bar plot showing the median concentration of the main flame retardants targeted in this study. The values used for this plot are those reported in Tables 1-3. Asterisks indicate compounds detected with low detection frequency in the samples.

BTBPE was detected with high frequency in all three matrices in all three countries (with the exception of air in Canada and Czech Republic), suggesting that this compound is ubiquitous in indoor environments. BTBPE levels were higher in the U.S. and Canadian samples than in the Czech samples for both dust and window film ($p < 0.01$). In air, the highest levels were observed in the Canadian samples and the lowest in the Czech Republic samples ($p < 0.01$), albeit with low detection frequencies in both cases. Levels measured in this study are similar to those reported in previous studies for dust (see Table S3). No comparable data for air or window wipes currently exist. These results indicate that BTBPE was more heavily used in North America than in Europe. Concentrations of BTBPE were on the same order as those of BDE-183, for which BTBPE is a replacement.

3.7. Summary of all flame retardants

An overview of the most abundant FRs in the three phases and three countries is shown in Fig. 4, which depicts both geographical trends and compound patterns. When considering all FRs together, the median concentrations in the U.S. are higher than those in Canada and in the Czech Republic for all three phases. As mentioned above, this geographic trend reflects market and usage patterns. Historically, the U.S. has been one of the major producers and consumers of flame retardants (Abbasi et al., 2015). In Europe, in general, the United Kingdom was the largest market for FRs, explaining the relatively low levels observed for the Czech samples in this study (Abbasi et al., 2015).

Among individual compounds, Penta-BDE, BDE 209, EHTBB and BEHTBP generally showed the highest median concentrations (see Fig. 4). It is not surprising that Penta-BDE and BDE 209 still play a major role in the FR arena: Abbasi et al. (2015) estimated that total consumption of Penta-BDEs and BDE 209 from 1970 to 2020 was about 46,000 and about 380,000 t, respectively. No specific market and usage data are available for EHTBB and BEHTBP; however, because they are the major replacement for Penta-BDE, a large production volume is likely. Other compounds were measured in these samples at relatively high median concentrations (e.g. DBDPE and DP), but it should be noted that their detection frequencies were low in some phases; thus, the plot in Fig. 4 for these compounds should be interpreted with caution.

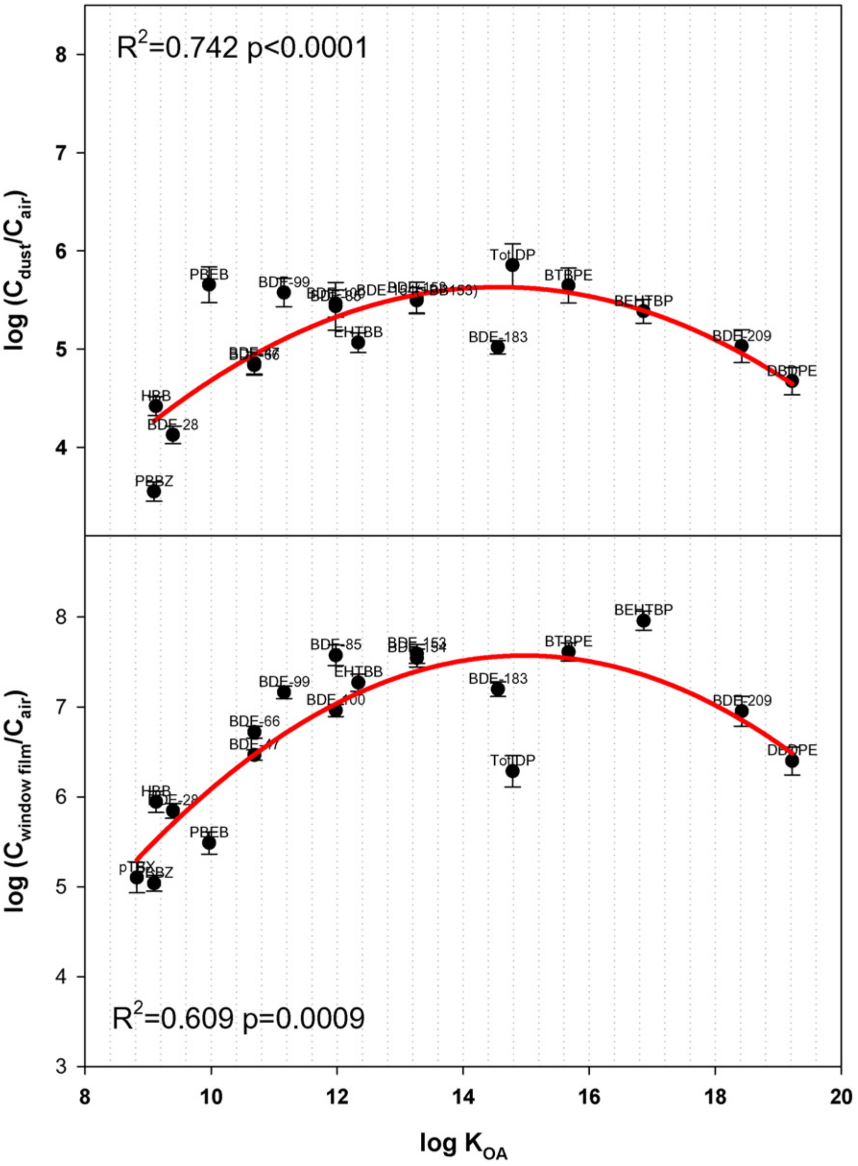

Fig. 5. Dependence of the dust/gas phase partition coefficients and of the window wipes/gas phase partition coefficients on the octanol/air partition coefficients (log K_{OA}). In both graphs, each circle represents the average, and the bars the standard error.

3.8. Within house distribution

In each country, samples from both the main bedroom and the living room area were collected in a subset of houses ($n = 10$ in each country). A side-by-side bar plot showing median concentrations in the bedrooms and living rooms for the three phases for PBDEs and for alternative flame retardants is showed in Fig. S6. While the bar plots suggest that bedrooms had higher levels of flame retardants than living rooms (especially for dust), a paired *t*-test for each matrix individually showed that the levels of FRs in bedrooms and living rooms were not significantly different ($p > 0.05$). Muenhor and Harrad (2012) and Harrad et al. (2009) reported some room-to-room differences, albeit not statistically significant, and they suggested that the dust concentrations were related to the number of specific items (such as electronic products) in each room. Coakley et al. (2013) reported an association between living room floor dust and mattress dust for BDE-183, BDE-206 and BDE-209. Ali et al. (2012) found significant differences between floor and mattress dust for BEHTBP and DBDPE but not for EHTBB or BTBPE. To our knowledge, no other study did a systematic room to room comparison on a relatively large number of samples ($n = 30$).

3.9. Correlation between air, dust, and window film concentrations

Spearman rank analysis gave good correlations between flame retardant concentrations in air, dust, and window films, especially for PBDEs (see SI Tables S4-S6). The correlation between concentrations in air and window film was the least powerful, showing significant relationships only for PBDEs, including BDE-209. The other FR concentrations were either not correlated, or their correlation was only marginally significant ($0.05 < p < 0.09$). The correlation between dust and air and dust and window film concentrations was generally stronger and encompassed a larger range of compounds. In fact, only four compounds were not significantly correlated (pTBX, PBEB, PBBz, and DPs), probably due to their relatively low detection frequencies (and hence low number of pairs for the correlation analysis). It is not surprising to find significant correlations between gas phase and window film concentrations because the thin layer of film allows for rapid equilibration between these two media (Csiszar et al., 2012). The correlation between air and dust concentrations was generally significant for lower brominated BDEs and low molecular weight compounds.

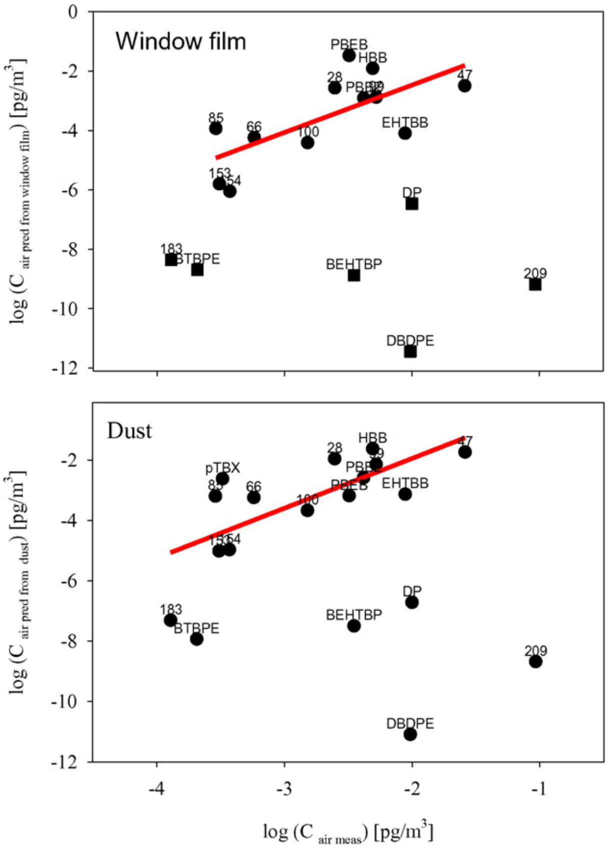

Fig. 6. Dependence of the median measured air concentrations versus the predicted concentrations using window wipes (upper) and dust (lower). Each dot represents the logarithm of the median concentration. The equation for the linear regressions are $y = 0.73 + 1.60 \times (r^2 = 0.48, p = 0.0121)$ for window film and $y = 0.094 + 1.13 \times (r^2 = 0.44, p = 0.0131)$ for dust.

Under equilibrium conditions, the partitioning between dust and air and between window film and air is expected to be proportional to log K_{OA}, which has been used to explain the sorption of chemicals to organic matter in dust (Weschler and Nazaroff, 2010). Fig. 5 shows the dependence on log K_{OA} of the partition coefficients between dust and air concentrations (top panel) and of the partition coefficients between window wipe and air concentrations (bottom panel); see Table S7 for log K_{OA} values used. Fig. 5 shows that the partitioning between dust/air and window film/air increases with the log K_{OA} for compounds with log K_{OA} up to 14, where it reaches a maximum, and it then decreases for compounds with log K_{OA} higher than 14. Zhang et al. (2011) reported a similar relationship for PCBs and PBDEs; however, their "break point" was log K_{OA} of 11. The decreasing trend of the partition coefficients with log K_{OA} for compounds with log $K_{OA} > 14$ can be due to several factors: (a) failure to reach equilibrium partitioning between air, dust, and window film; (b) overestimation of the gas-phase concentrations due to trapping of particles in the passive samplers; or (c) higher molecular weight compounds in dust as a result of physical abrasion of materials containing these chemicals, making them unavailable to exchange with gas-phase air (Zhang et al., 2011). As pointed out by Zhang et al. (2011), the latter explanation seems less likely because one would expect the ratios to be higher than the log K_{OA} rather than lower, as we have observed.

We hypothesize that in these samples, the equilibrium between air and dust and window film and dust was not reached at least for some compounds with log K_{OA}. To test this hypothesis, we used the Weschler and Nazaroff partitioning model (Weschler and Nazaroff, 2010). We first estimated air concentrations from dust and from window films, and then we compared these estimates with the measured ones. For this exercise, data from the three countries were aggregated to achieve higher statistical power. Gas-phase concentrations were calculated from dust using the following relationship:

$$C_g = \frac{X_d \rho_d}{K_{OA} f_{OM}} \tag{1}$$

where X_d is the measured dust concentration in ng/g, ρ_d is the density of dust (2.0×10^6 g/m^3 as per Weschler and Nazaroff, 2010), and f_{OM} is the fraction of organic matter in the dust [0.3, as suggested by Bennett and Furtaw (2004)]. Gas-phase air concentrations were calculated from window film levels with the following relationship:

$$C_g = \frac{S_f}{K_{OA} f_{OM} t} \tag{2}$$

where S_f is the measured window wipe concentrations (in ng/m^2), and t is the thickness of the window film (5.0×10^{-8} m) (see the Supporting Information for more details on the parameters used in calculations). Fig. 6 shows the relationship between the median measured gas-phase concentrations (pg/m^3) and the median gas-phase concentrations (pg/m^3) predicted from window film and from dust. For window film, the predicted values for compounds with log $K_{OA} < 14$ correlated with those measured ($r^2 = 0.48$) with a slope close to unity (1.60 ± 0.52) and an intercept of 0.73. For dust, the predicted values for compounds with log $K_{OA} < 14$ correlated with those measured ($r^2 = 0.44$)

with a slope close to unity (1.13 ± 0.38) and an intercept of 0.094. Both slopes were not significantly different from unity ($p > 0.05$), and the intercepts indicate that, on average, the predicted mass fractions in window films and dust were 8.3 and 1.3 times larger ($10^{0.73} = 5.4$ and $10^{0.094} = 1.2$) than the measured ones. The relatively large discrepancy between predicted and measured air concentrations from window films could be related to assuming too large a value for the fraction of organic matter (f_{OM}) or for the thickness of film layer (t). Both for dust and window films, the measured and predicted values for compounds with log $K_{OA} > 14$ did not correlate, supporting the hypothesis of Weschler and Nazaroff (2010) and Zhang et al. (2011), who suggested that the mass fraction in settled dust and window films may not have sufficient time to equilibrate with the gas phase.

On the basis of these findings, we conclude that compounds with log $K_{OA} < 14$ have reached equilibrium and they can be measured in any of the three matrices. Conversely, compounds with log $K_{OA} > 14$ have not reached equilibrium and hence they ought to be measured in each matrix separately because their values can't be predicted from one matrix alone. Sampling window films captures both gas- and particle-phase compounds, but lower concentrations lead to greater uncertainty. In these cases, window films do not provide a comprehensive picture of compound concentrations in either phase, making this a complementary technique to air and dust. In addition, all media are relevant to estimating exposure, either through inhalation (air and fine dust), ingestion (larger dust particles), and through hand contact followed by hand-to-mouth behavior (dust and window films).

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.envint.2016.04.029.

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References

Abbasi, G., Buser, A.M., Soehl, A., Murray, M.W., Diamond, M.L., 2015. Stocks and flows of PBDEs in products from use to waste in the U.S. And Canada from 1970 to 2020. Environ. Sci. Technol. 49, 1521–1528.

Abbasi, G., Saini, A., Goosey, E., Diamond, M.L., 2016. Product screening for sources of halogenated flame retardants in Canadian house and office dust. Sci. Total Environ. 545–546, 299–307.

Abdallah, M.A.-E., Harrad, S., Covaci, A., 2008. Hexabromocyclododecanes and tetrabromobisphenol-a in indoor air and dust in Birmingham, Uk: implications for human exposure. Environ. Sci. Technol. 42, 6855–6861.

Ali, N., Harrad, S., Goosey, E., Neels, H., Covaci, A., 2011a. "Novel" brominated flame retardants in Belgian and Uk indoor dust: implications for human exposure. Chemosphere 83, 1360–1365.

Ali, N., Harrad, S., Muenhor, D., Neels, H., Covaci, A., 2011b. Analytical characteristics and determination of major novel brominated flame retardants (Nbfrs) in indoor dust. Anal. Bioanal. Chem. 400, 3073–3083.

Ali, N., Dirtu, A.C., Eede, N.V.d., Goosey, E., Harrad, S., Neels, H., 't Mannetje, A., Coakley, J., Douwes, J., Covaci, A., 2012. Occurrence of alternative flame retardants in indoor dust from New Zealand: indoor sources and human exposure assessment. Chemosphere 88, 1276–1282.

Andersson, P.L., Öberg, K., Örn, U., 2006. Chemical characterization of brominated flame retardants and identification of structurally representative compounds. Environ. Toxicol. Chem. 25, 1275–1282.

Bennett, D.H., Furtaw, E.J., 2004. Fugacity-based indoor residential pesticide fate model. Environ. Sci. Technol. 38, 2142–2152.

Bennett, D.H., Moran, R.E., Wu, X., Tulve, N.S., Clifton, M.S., Colón, M., Weathers, W., Sjödin, A., Jones, R., Hertz-Picciotto, I., 2015. Polybrominated diphenyl ether (PBDE) concentrations and resulting exposure in homes in California: relationships among passive air, surface wipe and dust concentrations, and temporal variability. Indoor Air 25, 220–229.

Björklund, J.A., Sellström, U., de Wit, C.A., Aune, M., Lignell, S., Darnerud, P.O., 2012. Comparisons of polybrominated diphenyl ether and hexabromocyclododecane concentrations in dust collected with two sampling methods and matched breast milk samples. Indoor Air 22, 279–288.

Bohlin, P., Audy, O., Skrdlikova, L., Kukucka, P., Vojta, S., Pribylova, P., Prokes, R., Cupr, P., Klanova, J., 2014a. Evaluation and guidelines for using polyurethane foam (PUF) passive air samplers in double-dome chambers to assess semi-volatile organic compounds (Svocs) in non-industrial indoor environments. Environ. Sci.: Processes Impacts 16, 2617–2626.

Bohlin, P., Audy, O., Škrdlı́ková, L., Kukučka, P., Přibylová, P., Prokeš, R., Vojta, Š., Klánová, J., 2014b. Outdoor passive air monitoring of semi volatile organic compounds (Svocs): a critical evalutation of performance and limitations of polyurethane foam (PUF) disks. Environ. Sci.: Processes Impacts 16, 433–444.

Butt, C.M., Diamond, M.L., Truong, J., Ikonomou, M.G., ter Schure, A.F.H., 2004. Spatial distribution of polybrominated diphenyl ethers in southern Ontario as measured in indoor and outdoor window organic films. Environ. Sci. Technol. 38, 724–731.

Cao, Z., Xu, F., Covaci, A., Wu, M., Wang, H., Yu, G., Wang, B., Deng, S., Huang, J., Wang, X., 2014. Distribution patterns of brominated, chlorinated, and phosphorus flame retardants with particle size in indoor and outdoor dust and implications for human exposure. Environ. Sci. Technol. 48, 8839–8846.

Cequier, E., Ionas, A.C., Covaci, A., Marcé, R.M., Becher, G., Thomsen, C., 2014. Occurrence of a broad range of legacy and emerging flame retardants in indoor environments in Norway. Environ. Sci. Technol. 48, 6827–6835.

Cetin, B., Odabasi, M., 2011. Polybrominated diphenyl ethers (Pbdes) in indoor and outdoor window organic films in Izmir, Turkey. J. Hazard. Mater. 185, 784–791.

Chen, S.-J., Ding, N., Zhu, Z.-C., Tian, M., Luo, X.-J., Mai, B.-X., 2014. Sources of halogenated brominated retardants in house dust in an industrial city in southern China and associated human exposure. Environ. Res. 135, 190–195.

Coakley, J.D., Harrad, S.J., Goosey, E., Ali, N., Dirtu, A.-C., Van den Eede, N., Covaci, A., Douwes, J., Mannetje, A.t., 2013. Concentrations of polybrominated diphenyl ethers in matched samples of indoor dust and breast milk in New Zealand. Environ. Int. 59, 255–261.

Covaci, A., Harrad, S., Abdallah, M.A.E., Ali, N., Law, R.J., Herzke, D., de Wit, C.A., 2011. Novel brominated flame retardants: a review of their analysis, environmental fate and behaviour. Environ. Int. 37, 532–556.

Crimmins, B.S., Pagano, J.J., Xia, X., Hopke, P.K., Milligan, M.S., Holsen, T.M., 2012. Polybrominated diphenyl ethers (PBDEs): turning the corner in Great Lakes Trout 1980–2009. Environ. Sci. Technol. 46, 9890–9897.

Csiszar, S.A., Diamond, M.L., Thibodeaux, L.J., 2012. Modeling urban films using a dynamic multimedia fugacity model. Chemosphere 87, 1024–1031.

Davis, E.F., Stapleton, H.M., 2009. Photodegradation pathways of nonabrominated diphenyl ethers, 2-ethylhexyltetrabromobenzoate and di(2-ethylhexyl)tetrabromophthalate: identifying potential markers of photodegradation. Environ. Sci. Technol. 43, 5739–5746.

de Wit, C.A., Björklund, J.A., Thuresson, K., 2012. Tri-decabrominated diphenyl ethers and hexabromocyclododecane in indoor air and dust from Stockholm microenvironments 2: indoor sources and human exposure. Environ. Int. 39, 141–147.

Diamond, M.L., Gingrich, S.E., Fertuck, K., McCarry, B.E., Stern, G.A., Billeck, B., Grift, B., Brooker, D., Yager, T.D., 2000. Evidence for organic film on an impervious urban surface: characterization and potential teratogenic effects. Environ. Sci. Technol. 34, 2900–2908.

Dodson, R.E., Perovich, L.J., Covaci, A., Van den Eede, N., Ionas, A.C., Dirtu, A.C., Brody, J.G., Rudel, R.A., 2012. After the PBDE phase-out: a broad suite of flame retardants in repeat house dust samples from California. Environ. Sci. Technol. 46, 13056–13066.

Dodson, R.E., Camann, D.E., Morello-Frosch, R., Brody, J.G., Rudel, R.A., 2015. Semivolatile organic compounds in homes: strategies for efficient and systematic exposure measurement based on empirical and theoretical factors. Environ. Sci. Technol. 49, 113–122.

Dungey, S.M., Akintoye, L., 2007. Environmental risk evaluation report: 1,1′-(ethane-1,2-diyl)*bis*[pentabromobenzene]. Environment Agency for England and Wales.

Earnshaw, M.R., Jones, K.C., Sweetman, A.J., 2015. A First European Scale Multimedia Fate Modelling of Bde-209 from 1970 to 2020. Environ. Int. 74, 71–81.

Episuite. Us Epa, 2012. [Estimation Programs Interface Suite™ for Microsoft® Windows, V 4.11 or Insert Version Used]. United States Environmental Protection Agency, Washington, Dc, USA.

Gingrich, S.E., Diamond, M.L., Stern, G.A., McCarry, B.E., 2001. Atmospherically derived organic surface films along an urban-rural gradient. Environ. Sci. Technol. 35, 4031–4037.

Harner, T., Su, K., Genualdi, S., Karpowicz, J., Ahrens, L., Mihele, C., Schuster, J., Charland, J.-P., Narayan, J., 2013. Calibration and application of PUF disk passive air samplers for tracking polycyclic aromatic compounds (Pacs). Atmos. Environ. 75, 123–128.

Harrad, S., Abdallah, M.A.-E., 2008a. Calibration of two passive air sampler configurations for monitoring concentrations of hexabromocyclododecanes in indoor air. J. Environ. Monit. 10, 527–531.

Harrad, S., Ibarra, C., Diamond, M., Melymuk, L., Robson, M., Douwes, J., Roosens, L., Dirtu, A.C., Covaci, A., 2008b. Polybrominated diphenyl ethers in domestic indoor dust from Canada, New Zealand, United Kingdom and United States. Environ. Int. 34, 232–238.

Harrad, S., Abdallah, M.A.-E., Covaci, A., 2009. Causes of variability in concentrations and diastereomer patterns of hexabromocyclododecanes in indoor dust. Environ. Int. 35, 573–579.

Johnson, P.I., Stapleton, H.M., Mukherjee, B., Hauser, R., Meeker, J.D., 2013. Associations between brominated flame retardants in house dust and hormone levels in men. Sci. Total Environ. 445–446, 177–184.

Li, W.-L., Qi, H., Ma, W.-L., Liu, L.-Y., Zhang, Z., Zhu, N.-Z., Mohammed, M.O.A., Li, Y.-F., 2015. Occurrence, behavior and human health risk assessment of dechlorane plus and related compounds in indoor dust of China. Chemosphere 134, 166–171.

Lioy, P.J., Freeman, N.C.G., Millette, J.R., 2002. Dust: a metric for use in residential and building exposure assessment and source characterization. Environ. Health Perspect. 110, 969–983.

Lohmann, R., Klanova, J., Kukucka, P., Yonis, S., Bollinger, K., 2013. Concentration, fluxes, and residence time of PBDEs across the tropical Atlantic Ocean. Environ. Sci. Technol. 47, 13967–13975.

Ma, Y., Venier, M., Hites, R.A., 2012. 2-Ethylhexyl tetrabromobenzoate and *bis*(2-ethylhexyl) tetrabromophthalate flame retardants in the Great Lakes atmosphere. Environ. Sci. Technol. 46, 204–208.

Ma, Y., Salamova, A., Venier, M., Hites, R.A., 2013. Has the phase-out of PBDEs affected their atmospheric levels? Trends of PBDEs and their replacements in the Great Lakes atmosphere. Environ. Sci. Technol. 47, 11457–11464.

Melymuk, L., Goosey, E., Riddell, N., Diamond, M., 2015. Interlaboratory study of novel halogenated flame retardants: interflab. Anal. Bioanal. Chem. 407, 6759–6769.

Muenhor, D., Harrad, S., 2012. Within-room and within-building temporal and spatial variations in concentrations of polybrominated diphenyl ethers (PBDEs) in indoor dust. Environ. Int. 47, 23–27.

Newton, S., Sellström, U., de Wit, C.A., 2015. Emerging flame retardants, PBDEs, and Hbcdds in indoor and outdoor media in Stockholm, Sweden. Environ. Sci. Technol. 49, 2912–2920.

Peverly, A.A., Ma, Y., Venier, M., Rodenburg, Z., Spak, S.N., Hornbuckle, K.C., Hites, R.A., 2015. Variations of flame retardant, polycyclic aromatic hydrocarbon, and pesticide concentrations in Chicago's atmosphere measured using passive sampling. Environ. Sci. Technol. 49, 5371–5379.

Ricklund, N., Kierkegaard, A., McLachlan, M.S., 2008. An international survey of decabromodiphenyl ethane (Debdethane) and decabromodiphenyl ether (Decabde) in sewage sludge samples. Chemosphere 73, 1799–1804.

Saini, A., Okeme, J.O., Goosey, E., Diamond, M.L., 2015. Calibration of two passive air samplers for monitoring phthalates and brominated flame-retardants in indoor air. Chemosphere 137, 166–173.

Shoeib, M., Harner, T., 2002. Characterization and comparison of three passive air samplers for persistent organic pollutants. Environ. Sci. Technol. 36, 4142–4151.

Shoeib, M., Harner, T., Webster, G.M., Sverko, E., Cheng, Y., 2012. Legacy and current-use flame retardants in house dust from Vancouver, Canada. Environ. Pollut. 169, 175–182.

Shoeib, M., Ahrens, L., Jantunen, L., Harner, T., 2014. Concentrations in air of organobromine, organochlorine and organophosphatee flame retardants in Toronto, Canada. Atmos. Environ. 99, 140–147.

Stapleton, H.M., Allen, J.G., Kelly, S.M., Konstantinov, A., Klosterhaus, S., Watkins, D., McClean, M.D., Webster, T.F., 2008. Alternate and new brominated flame retardants detected in U.S. house dust. Environ. Sci. Technol. 42, 6910–6916.

Stapleton, H.M., Klosterhaus, S., Keller, A., Ferguson, P.L., van Bergen, S., Cooper, E., Webster, T.F., Blum, A., 2011. Identification of flame retardants in polyurethane foam collected from baby products. Environ. Sci. Technol. 45, 5323–5331.

Sverko, E., Tomy, G.T., Reiner, E.J., Li, Y.-F., McCarry, B.E., Arnot, J.A., Law, R.J., Hites, R.A., 2011. Dechlorane plus and related compounds in the environment: a review. Environ. Sci. Technol. 45, 5088–5098.

Weschler, C.J., Nazaroff, W.W., 2010. Svoc partitioning between the gas phase and settled dust indoors. Atmos. Environ. 44, 3609–3620.

Wilford, B.H., Harner, T., Zhu, J., Shoeib, M., Jones, K.C., 2004. Passive sampling survey of polybrominated diphenyl ether flame retardants in indoor and outdoor air in Ottawa, Canada: implications for sources and exposure. Environ. Sci. Technol. 38, 5312–5318.

Zhang, X., Diamond, M.L., Robson, M., Harrad, S., 2011. Sources, emissions, and fate of polybrominated diphenyl ethers and polychlorinated biphenyls indoors in Toronto, Canada. Environ. Sci. Technol. 45, 3268–3274.

Zheng, J., Wang, J., Luo, X.-J., Tian, M., He, L.-Y., Yuan, J.-G., Mai, B.-X., Yang, Z.-Y., 2010. Dechlorane plus in human hair from an E-waste recycling area in South China: comparison with dust. Environ. Sci. Technol. 44, 9298–9303.

Zheng, X., Xu, F., Chen, K., Zeng, Y., Luo, X., Chen, S., Mai, B., Covaci, A., 2015. Flame retardants and organochlorines in indoor dust from several E-waste recycling sites in South China: composition variations and implications for human exposure. Environ. Int. 78, 1–7.

APPENDIX 4

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Letter

Hands as Agents of Chemical Transport in the Indoor Environment

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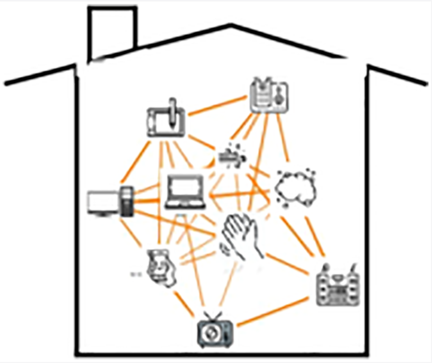

ABSTRACT: Indoor environments are important sources of exposure to chemicals intentionally added to consumer products, building materials, etc. Previous work has shown that semivolatile organic compounds (SVOCs) migrate from product/material sources to partition to indoor surfaces, including skin and hands, and that SVOCs on hands reasonably indicate nondietary exposure to indoor SVOCs. We hypothesize that the hands of indoor occupants, which contact numerous products and surfaces, transport SVOCs in the indoor environment to an extent comparable to that of fugacity-driven and advective transport. This process of “hand-based” chemical transport is analogous to that of fomite transmission of pathogens. We explore this hypothesis using a data set of halogenated flame retardants, organophosphate esters, and phthalate esters in indoor air, floor dust, and wipes of hands and surfaces of electronic devices of 51 participants. Cluster analysis shows the similarity of the SVOC profiles on all participants’ hands relative to those of all device surfaces, demonstrating the ubiquity of these SVOCs. Network analysis consistently shows the centrality of hands, followed by air, dust, and laptops, indicating that hands are most correlated with all sample types. The significance of this hypothesis lies in the ability of hands to rapidly transfer SVOCs among surfaces indoors, with implications for exposure.

■ INTRODUCTION

It is painfully clear that people are potent agents of environmental change at planetary to local scales.[1,2] People often retreat to the indoor environment for comfort, respite, and safety. In developed countries, people typically spend >90% of their time indoors and nearly 70% in indoor residential environments.[3,4] By virtue of the amount of time we spend indoors, these environments affect our health and well-being, both positively and negatively.[4−7]

Modern indoor environments contain a wide variety of chemical contaminants in air and dust that provide pathways for exposure to occupants. Although only a relatively small fraction of the complex chemical mixture that occurs indoors has been identified,[8,9] some chemicals, such as lead, formaldehyde, and some phthalate esters, have been associated with adverse health effects.[10−12]

Among the chemicals released indoors, semivolatile organic compounds (SVOCs) undergo partitioning and transport processes that result in their distribution among indoor media such as air, dust, thin organic films that accumulate on surfaces, skin, clothing and other textiles, and polyurethane foam in upholstered furniture.[13−15] The indoor environment, especially as energy-saving measures tighten building envelopes, presents fewer opportunities for chemical loss than outdoor environments,[14,16] leading to increased concentrations per unit emission relative to outdoors and relatively high chemical persistence.[17−19]

In addition to indoor products and materials, people influence indoor chemical and particle concentrations by virtue of their presence and activities.[20−22] For example, the exhaled breath of indoor occupants increases levels of carbon dioxide.[23] Adults and particularly young children resuspend dust when walking and crawling, respectively, which increases particle concentrations in air and hence inhalation exposure to particles in dust and particle-phase chemicals.[24] One more example here is that each person is surrounded by her or his personal cloud of increased particle concentrations that increase as a function of activity.[25−27]

Hands can be particularly effective in indoor transport. Hands play an essential role in transferring the personal microbiome to indoor surfaces.[28] Hands are very efficient in transporting infectious disease agents, notably the norovirus and rhinovirus, to create fomites (surfaces or objects contaminated with infectious agents) such as doorknobs, handrails, keyboards, and telephones.[29−32] Transmission of infectious agents among fomites by hands through an indoor environment can occur rapidly.[33,34]

The surfaces of hands are known to reflect our chemical environment and to be an indicator of internal body burdens of a wide range of chemicals. Bouslimani et al.[35] identified a plethora of chemicals on hands, including pesticides and

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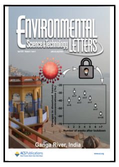

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insecticides, plasticizers, chemicals from a person's diet, and pharmaceuticals. Significant correlations have been found between several flame retardant and phthalate levels on hand wipes of adults and children and their internal body levels.[36−40] Hands are presumed to gather these chemicals by direct contact with products containing these SVOCs, contact with contaminated dust, and by air−skin partitioning.[41−43]

While hands clearly transport some infectious disease agents and a person's microbiome community, hands have not been seen as agents of transport of indoor contaminants such as SVOCs. Rather, SVOC transport is typically considered to occur by means of diffusion according to fugacity (or chemical activity) gradients between media, particle transport processes (i.e., deposition and resuspension), air advection from ventilation, heating and cooling systems, and cleaning processes that remove chemicals associated with dust and surfaces.[18,44,45]

We hypothesize that humans are agents of SVOC transport indoors and that hands play a key role in that transport process. We explored this hypothesis using a data set of SVOCs in residential air and dust and surface wipes of electronic devices and hands of 51 participants obtained through the OEHS-Homes Study (Ontario Environmental Health-Homes Study). The following SVOCs were measured: six phthalate esters,[13] 22 halogenated flame retardants (HFRs), including 14 congeners of polybrominated diphenyl ethers (PBDEs),[46] and 23 organophosphate esters (OPEs).[39] This paper is the fifth in a series published from the OEHS-Homes Study; all data along with detailed explorations of associations and explanations have been previously published as noted above.

MATERIALS AND METHODS

Chemical Concentration Data. Full details of the study design, sampling, and chemical analysis used in this analysis have been provided by Okeme and Yang et al.,[47] Yang and Jílková et al.,[13,39] and Yang et al.[46] These publications also include detailed information about household characteristics such as the air exchange rate, house volume, activities, and information provided in questionnaires by participants where such factors were significantly related to chemical concentrations and profiles. Briefly, 51 women were recruited from the Toronto and Ottawa areas in Canada, aged 18−44 and of mid-socio-economic status. Ethics approvals were obtained from the Research Ethics Boards of University of Toronto and Health Canada. The following samples for chemical analysis were taken from the bedroom (n = 51) and a subset of participants' most used room (n = 26) from February to July 2015: (1) vacuumed floor dust (n = 77), (2) air using polyurethane passive foam samplers (PUF) deployed for 3 weeks (n = 77), (3) surface wipes from hand-held (cell or mobile phones, home phones, tablets, and laptops) and non-hand-held [desktop computers, televisions (TVs), and stereos] electronic devices (n = 199), and (4) wipes of each participant's hand palms and backs (right and left were composited) taken twice, 3 weeks apart (n = 204 total). Note that we wiped the surface of the hand-held devices that came into contact with each participant's hands, which could have included cases and not the device itself. Field blanks were taken for each sample type.

The reported analytes are listed in Table S1. Analytical methods, including QA/QC procedures (e.g., analysis of SRM 2585, methods for blank and recovery correction, and deployment of duplicate PUF samplers), have been explained elsewhere.[13,39,46,47] Briefly, after extraction, all samples were analyzed by gas chromatography with a mass selective detector (Agilent 6890 GC-5975 MS). Data were blank corrected according to the criteria of Saini et al.[48] Surrogate standard recoveries were between 74% and 131%; no recovery correction was applied.

Statistical Analysis. The data set analyzed consisted of concentrations of six HFRs, 11 OPEs, and six phthalates with >40% detection frequency in dust, air, and surface wipes from hands and electronic devices. In cases in which more than one sample was collected from a home (e.g., hand wipes taken 3 weeks apart and air and dust measurements from the bedroom and the most used room), an arithmetic mean was calculated to minimize overrepresentation of any sample type in the analysis. Values of half of the method detection limit (MDL) were imputed for values < MDL. The final data set included concentrations in air (n = 51), dust (n = 51), hands (n = 51), and surface wipes from hand-held [cell phones (n = 31), home phones (n = 10), tablets (n = 27), and laptops (n = 32)] and non-hand-held [desktop computers (n = 13), TVs (n = 68), and stereos (n = 7)] electronic devices.

Cluster Analysis. We conducted cluster analysis to assess the similarities between wipe samples (hands and electronic device surfaces). Data are listed in Table S2. We normalized the wipe data to percentage contributions to reduce between-sample variations. Data were log-transformed, and the log concentration of each chemical was centered by its mean and scaled by its standard deviation to improve the distribution of the data and to minimize heteroscedasticity. We performed hierarchical clustering in MetaboAnalyst version 3.0 using Ward's method for Euclidean distances between pairs of samples. Samples were not reorganized so as to maintain natural contrast among sample types.[49]

Network Analyses. We conducted network analyses to assess relationships between sample types. We used the non-normalized final data set [air, dust, and wipes of hands and electronic device surfaces (Table S3)]. First, we calculated Spearman's correlation coefficients (ρ) between pairs of sample types, i.e., samples from the same home. Next, we used the data set to create igraph objects of arithmetic-weighted mean ρ for statistically significant correlations. We used a weighted ρ of >0.4 to generate undirected networks using the igraph package in RStudio version 3.6.1. The edges (interconnections) of the networks are proportional to the weighted ρ values, and node sizes (sample types) are proportional to the number of edges. We used the PageRank function (global, unsupervised ranking algorithm[50]) in Rstudio to calculate the centrality or importance of each node. The total PageRank scores from all nodes of a network amount to 1. Several networks were built: all chemicals grouped according to compound class [HFRs, OPEs, and phthalates (Table S4)], grouped according to log K_{OA} values of <11 or ≥11 (Table S5), and all sample types except for hands.

RESULTS AND DISCUSSION

Summary of Previous Results. Dust and surface wipes of most hand-held and non-hand-held electronic devices had high detection frequencies of most SVOCs analyzed (Figure S1).[13,39,46] This finding was contrary to our expectation that surface wipes of electronic devices would reveal one or very few dominant plasticizer(s) and/or flame retardant(s) that had been intentionally added to that product. The exception was

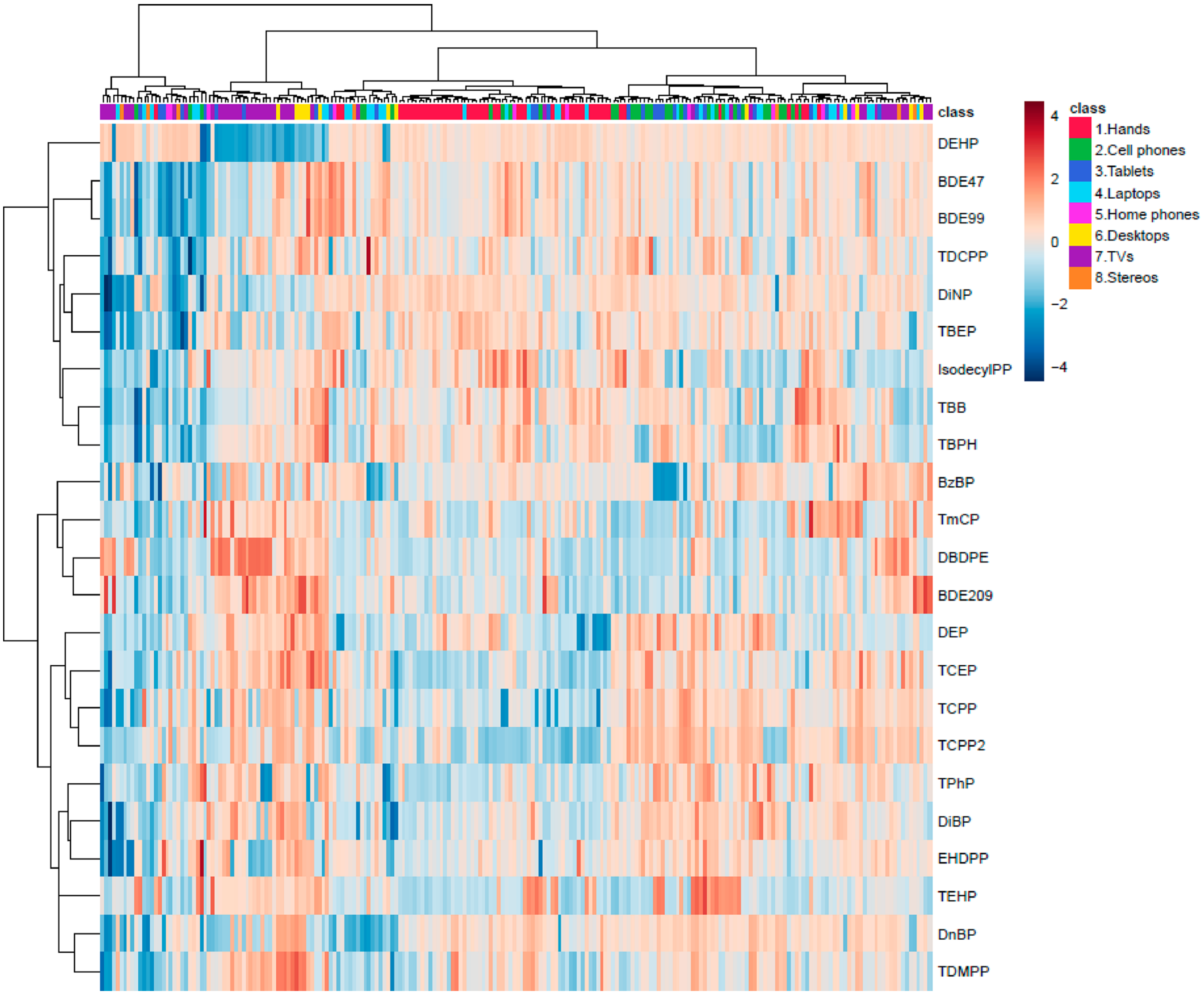
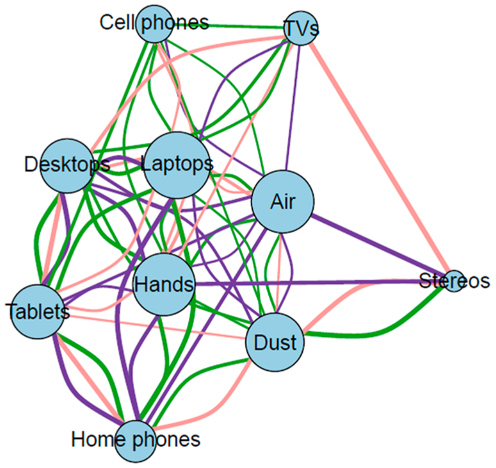


Figure 1. Hierarchical cluster analysis based on Euclidean distance for all surfaces sampled, namely, hands and hand-held and non-hand-held electronic devices (top or *x*-axis clustering and color bar). Samples were not reorganized so as to maintain natural contrast among sample types; 25% features were ranked by *t* test to show optimal contrasting patterns as described by Xia and Wishart.[49] The heat map shows clustering of the relative concentrations of chemicals (*y*-axis) in the different samples (*x*-axis). The red-to-blue color scale represents the relative concentrations of the individual phthalates, HFRs, OPEs, and phthalates as shown by the scale.

the surface wipes of TVs that had increased concentrations of DBDPE and BDE-209 that, we believe, were intentionally added at high concentrations to those casings.[46] We also found significantly higher concentrations of plasticizers and flame retardants in hand-held than non-hand-held devices, with the exceptions of DBDPE and BDE-209. We postulated that the discovery of multiple plasticizers and flame retardants in the surface wipes of all electronic devices was due to within-home partitioning of these SVOCs emitted from primary sources (e.g., TV casings, wiring, and upholstered furniture) to indoor surfaces that included other electronic devices. In addition, we postulated that the significantly higher concentrations of SVOCs (except DBDPE and BDE-209) in wipes of hand-held than non-hand-held devices were due to hands transferring these chemicals from sources to hand-held devices that are infrequently washed relative to hands. This hypothesis was supported by significant correlations between chemical profiles on hands and hand-held devices.[13,39,46] We also found that chemical profiles from wipes of hand backs and palms were significantly correlated, although concentrations were higher on palms and concentrations were consistent in samples taken 3 weeks apart.[13,39,46]

Similarity of Chemical Profiles. Although we found significant correlations between paired chemical profiles on hands and hand-held devices, the cluster analysis showed that all hand profiles clustered together, along with some hand-held devices (e.g., cell phones and tablets) (Figure 1 and Figure S1). As expected, TVs mostly clustered together because of their unusual flame retardant profiles.[46] Chemical profiles from device surfaces from individual homes did not cluster together, which was expected on the basis of within-home partitioning.[13,14,51]

The similarity of chemical profiles on hand wipes of all 51 unrelated women is consistent with the findings of Lax et al.,[28] who reported greater similarity of microbial communities on the hand samples of 18 participants versus greater differentiation of microbial communities on each participant's home surfaces (e.g., doorknobs and kitchen counters) and other area



Sample Type	Centrality (PageRank)
Hands	0.116
Air	0.113
Laptops	0.108
Dust	0.103
Tablets	0.099
Desktop computers	0.097
Televisions (TVs)	0.095
Stereos	0.095
Home phones	0.090
Cell (mobile) phones	0.084

Figure 2. Network analysis based on significant Spearman ρ correlation coefficients (>0.4) between sample types. Edges (lines connecting nodes) are green for halogenated flame retardants, pink for organophosphate esters, and purple for phthalates. The table shows the degree of centrality, expressed as PageRank, between nodes in the network analysis, arranged from most to least central.

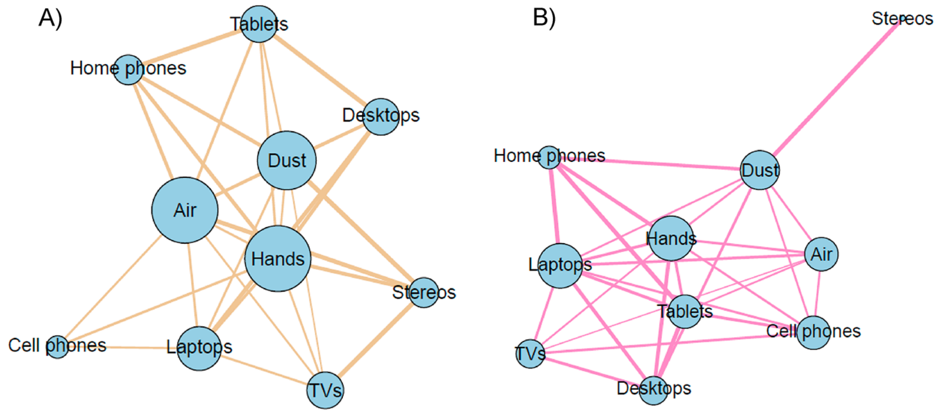


Figure 3. Network analysis based on significant Spearman ρ correlation coefficients (>0.4) between sample types for (A) more volatile compounds with log K_{OA} values of <11 and (B) less volatile compounds with K_{OA} values of ≥11. See Table S6 for PageRank values of each node.

samples such as each participant's nose and feet. They concluded that microbial populations on hands reflect the numerous inputs obtained from touching a variety of surfaces, with disruption due to washing. We suggest that the same explanation pertains here: chemical profiles reflect numerous inputs from home and other environments as hands touch numerous surfaces, sorb chemicals from air, and gather dust, with disruption from washing.

Networks of Chemical Profiles. Next, we examined the relationships among chemical profiles in all media (air, dust, and hand and device wipes) using network analysis. Here, the purpose was to understand the correlations among all sample types when seen as a connected system. We note that these networks express the central tendency of the data set, within which we expect variability due to differences in households such as differences in individual activity patterns, numbers of residents, the presence of pets, etc.

SVOC indoor distributions typically consider air, dust, and emission sources, with transport by diffusion according to fugacity gradients between air and condensed phases, and particle or dust transport; hands are not typically considered relevant vectors for transport of indoor contaminants.[14,52,53] Figure S2 shows the network of all flame retardants and plasticizers (with edges shown distinctly for HFRs, OPEs, and phthalates) based on this paradigm, excluding the hands as a possible agent of chemical transport. Air, followed by dust, was most central or important in the network as indicated by PageRank values of 0.128 and 0.127, respectively, conforming to this conventional understanding.

Figure 2 illustrates the network for the same set of chemicals with the addition of hands as a node. Hands were most central with the highest PageRank score of 0.116: hands had the greatest number and strength of significant correlations with other nodes (i.e., correlations of chemical profiles). The centrality of hands is consistent with chemical partitioning and/or chemical transfer between hands and all indoor media. Air was the next most central node (PageRank score of 0.113), and dust was fourth (PageRank score of 0.103), which conforms to the conventional understanding of the distribution of SVOCs discussed above.

The greater centrality of hand-held laptops and tablets than non-hand-held desktops, TVs, and stereos is consistent with the transfer of SVOCs from hands to hand-held devices that, we have hypothesized, augments chemical abundance attributable to only partitioning from air and accumulation from dust.[13,40] The low centrality of home phones followed by cell phones in the networks with and without hands was not expected and is not consistent with the previous explanation, especially given the frequent handling of cell phones. One explanation for the low centrality of cell phones is that their chemical profiles reflect the numerous surfaces with which they come into contact (e.g., purses and pockets) and upon which they are placed, including different environments outside the home. In addition, some people wash the exterior of their cell phones (we did not collect this information).

Finally, we examined networks produced when the flame retardants and plasticizers were separated into more volatile (with $\log K_{OA} < 11$) and less volatile ($\log K_{OA} \geq 11$) chemicals (Figure 3 and Table S6). We chose the physicochemical property of vapor pressure, expressed as K_{OA}, because of its importance in governing indoor chemical partitioning.[13,14,45] Again, hands were most central among more volatile chemicals, followed by air and dust (PageRank values of 0.142, 0.136, and 0.119, respectively). The centrality of air was expected on the basis of air-condensed phase partitioning. In the network of less volatile chemicals, hands were the second-most central, followed by dust (PageRank values of 0.131 and 0.123, respectively). The centrality of dust was expected as less volatile chemicals partition to this condensed phase that is distributed throughout the indoor environment. Laptops had the greatest centrality (PageRank value of 0.140), which could be due to their frequent touching as well as presenting a condensed phase for partitioning. Air had limited centrality (PageRank value of 0.083), as expected given the tendency of these lower-volatility chemicals to partition to condensed phases. Differences in volatility, in part, explain the difference in networks according to chemical class; i.e., the phthalates considered here as a group are more volatile than HFRs and OPEs (Figure S2 and Table S7).

Implications. Overall, hands were either most (all compound classes and chemicals with $\log K_{OA}$ values of <11) or second most (chemicals with $\log K_{OA}$ values of ≥11) central in the networks constructed, illustrating the high degree of correlation between hands and other media in the same residence, even those not frequently touched (e.g., tops of stereos and TVs). The centrality of hands is consistent with but not a test of the hypothesis that hands transport chemicals, where hands can gather chemicals from contacting multiple surfaces directly, transport adhered dust, and transport chemicals that have partitioned from air. The process of hand transport was posited to explain the higher concentrations of numerous flame retardants and plasticizers on hand-held than non-hand-held devices.[13,40,47] The similarity of flame retardant and plasticizer profiles on the hands of 51 unrelated women, as shown in the cluster analysis, is also consistent with hands coming into contact with multiple surfaces both inside and outside the home. Tangentially, these results demonstrate the ubiquity of these chemicals in modern environments, regardless of the participants' home contents, activities, or other personal characteristics.

If hands are agents of chemical transport, as hypothesized here, then we expect that this mechanism is faster than diffusive or particle-based transport mechanisms. For example, hands can rapidly transport chemicals within a home and between environments such as offices, schools, entertainment venues, etc. This hypothesis of rapid hand-based transport within and between locations is in line with the current understanding of community spread of fomites and the microbiome.[29−35]

Finally, these results have implications for human exposure. Numerous studies have shown that chemical profiles on hands are proportional to internal exposure.[37,39,54] This study advances our understanding of the ubiquity of these large-production volume chemicals with which hands come into contact. We suggest that hands likely perpetuate exposure by transporting chemicals to multiple surfaces that, unlike hands, are less frequently washed and thus can "re-contaminate" hands, in the same way that microbial and viral transfer occurs. Our results provide yet another good reason to wash your hands frequently!

■ ASSOCIATED CONTENT

Supporting Information

The Supporting Information is available free of charge at https://pubs.acs.org/doi/10.1021/acs.estlett.0c01006.

> Full names and CAS registry numbers of target compounds, data sets used to generate networks, hierarchical cluster and heat map of chemical profiles and concentrations in samples, network diagram excluding the hands node, and concentrations of compounds in samples (PDF)

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Notes

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■ REFERENCES

(1) Steffen, W.; Crutzen, P. J.; McNeill, J. R. The Anthropocene: Are Humans Now Overwhelming the Great Forces of Nature. *Ambio* **2007**, *36*, 614−621.

(2) Steffen, W.; Rockström, J.; Richardson, K.; Lenton, T. M.; Folke, C.; Liverman, D.; Summerhayes, C. P.; Barnosky, A. D.; Cornell, S. E.; Crucifix, M.; Donges, J. F.; Fetzer, I.; Lade, S. J.; Scheffer, M.; Winkelmann, R.; Schellnhuber, H. J. Trajectories of the Earth System in the Anthropocene. *Proc. Natl. Acad. Sci. U. S. A.* **2018**, *115*, 8252−8259.

(3) Matz, C.; Stieb, D.; Davis, K.; Egyed, M.; Rose, A.; Chou, B.; Brion, O. Effects of Age, Season, Gender and Urban-Rural Status on Time-Activity: Canadian Human Activity Pattern Survey 2 (CHAPS 2). *Int. J. Environ. Res. Public Health* **2014**, *11*, 2108−2124.

(4) Allen, J. G.; MacNaughton, P.; Satish, U.; Santanam, S.; Vallarino, J.; Spengler, J. D. Associations of Cognitive Function Scores with Carbon Dioxide, Ventilation, and Volatile Organic Compound Exposures in Office Workers: A Controlled Exposure Study of Green and Conventional Office Environments. *Environ. Health Perspect.* **2016**, *124*, 805−812.

(5) Schmeer, K. K.; Yoon, A. J. Home Sweet Home? Home Physical Environment and Inflammation in Children. *Soc. Sci. Res.* **2016**, *60*, 236−248.

(6) Cedeño-Laurent, J. G.; Williams, A.; MacNaughton, P.; Cao, X.; Eitland, E.; Spengler, J.; Allen, J. G. Building Evidence for Health: Green Buildings, Current Science, and Future Challenges. *Annu. Rev. Public Health* **2018**, *39*, 291−308.

(7) Logue, J. M.; Price, P. N.; Sherman, M. H.; Singer, B. C. A Method to Estimate the Chronic Health Impact of Air Pollutants in U.S. Residences. *Environ. Health Perspect.* **2012**, *120*, 216−222.

(8) Peng, H.; Saunders, D. M. V.; Jones, P. D.; Giesy, J. P. Mutagenic Azo Dyes, Rather than Flame Retardants, Are the Predominant Brominated Compounds in House Dust Response. *Environ. Sci. Technol.* **2017**, *51*, 3591−3592.

(9) Rostkowski, P.; Haglund, P.; Aalizadeh, R.; Alygizakis, N.; Thomaidis, N.; Beltran Arandes, J.; Nizzetto, P. B.; Booij, P.; Budzinski, H.; Brunswick, P.; Covaci, A.; Gallampois, C.; Grosse, S.; Hindle, R.; Ipolyi, I.; Jobst, K.; Kaserzon, S. L.; Leonards, P.; Lestremau, F.; Letzel, T.; Magner, J.; Matsukami, H.; Moschet, C.; Oswald, P.; Plassmann, M.; Slobodnik, J.; Yang, C. The Strength in Numbers: Comprehensive Characterization of House Dust Using Complementary Mass Spectrometric Techniques. *Anal. Bioanal. Chem.* **2019**, *411*, 1957−1977.

(10) Logue, J. M.; Mckone, T. E.; Sherman, M. H.; Singer, B. C. Hazard Assessment of Chemical Air Contaminants Measured in Residences. *Indoor Air* **2011**, *21*, 92−109.

(11) Dixon, S. L.; Gaitens, J. M.; Jacobs, D. E.; Strauss, W.; Nagaraja, J.; Pivetz, T.; Wilson, J. W.; Ashley, P. J. Exposure of U.S. Children to Residential Dust Lead, 1999−2004: II. The Contribution of Lead-Contaminated Dust to Children's Blood Lead Levels. *Environ. Health Perspect.* **2009**, *117*, 468−474.

(12) Radke, E. G.; Braun, J. M.; Meeker, J. D.; Cooper, G. S. Phthalate Exposure and Male Reproductive Outcomes: A Systematic Review of the Human Epidemiological Evidence. *Environ. Int.* **2018**, *121*, 764−793.

(13) Yang, C.; Harris, S. A.; Jantunen, L. M.; Kvasnicka, J.; Nguyen, L. V.; Diamond, M. L. Phthalates: Relationships between Air, Dust, Electronic Devices, and Hands with Implications for Exposure. *Environ. Sci. Technol.* **2020**, *54*, 8186−8197.

(14) Weschler, C. J.; Nazaroff, W. W. Semivolatile Organic Compounds in Indoor Environments. *Atmos. Environ.* **2008**, *42*, 9018−9040.

(15) Salthammer, T.; Zhang, Y.; Mo, J.; Koch, H. M.; Weschler, C. J. Assessing Human Exposure to Organic Pollutants in the Indoor Environment. *Angew. Chem., Int. Ed.* **2018**, *57*, 12228−12263.

(16) Zhang, X.; Diamond, M. L.; Robson, M.; Harrad, S. J. Sources, Emissions, and Fate of Polybrominated Diphenyl Ethers and Polychlorinated Biphenyls Indoors in Toronto, Canada. *Environ. Sci. Technol.* **2011**, *45*, 3268−3274.

(17) Rudel, R. A.; Perovich, L. J. Endocrine Disrupting Chemicals in Indoor and Outdoor Air. *Atmos. Environ.* **2009**, *43*, 170−181.

(18) Shin, H.-M.; McKone, T. E.; Tulve, N. S.; Clifton, M. S.; Bennett, D. H. Indoor Residence Times of Semivolatile Organic Compounds: Model Estimation and Field Evaluation. *Environ. Sci. Technol.* **2013**, *47*, 859−867.

(19) Audy, O.; Melymuk, L.; Venier, M.; Vojta, Š.; Bečanová, J.; Romanak, K.; Vykoukalová, M.; Prokeš, R.; Kukucka, P.; Diamond, M. L.; Klánová, J. PCBs and Organochlorine Pesticides in Indoor Environments - A Comparison of Indoor Contamination in Canada and Czech Republic. *Chemosphere* **2018**, *206*, 622−631.

(20) Weschler, C. J. Roles of the Human Occupant in Indoor Chemistry. *Indoor Air* **2016**, *26*, 6−24.

(21) Lunderberg, D. M.; Kristensen, K.; Liu, Y.; Misztal, P. K.; Tian, Y.; Arata, C.; Wernis, R.; Kreisberg, N.; Nazaroff, W. W.; Goldstein, A. H. Characterizing Airborne Phthalate Concentrations and Dynamics in a Normally Occupied Residence. *Environ. Sci. Technol.* **2019**, *53*, 7337−7346.

(22) Kvasnicka, J.; Cohen Hubal, E.; Ladan, J.; Zhang, X.; Diamond, M. L. Transient Multimedia Model for Investigating the Influence of Indoor Human Activities on Exposure to SVOCs. *Environ. Sci. Technol.* **2020**, *54*, 10772−10782.

(23) Stönner, C.; Edtbauer, A.; Williams, J. Real-World Volatile Organic Compound Emission Rates from Seated Adults and Children for Use in Indoor Air Studies. *Indoor Air* **2018**, *28*, 164−172.

(24) Wu, T.; Täubel, M.; Holopainen, R.; Viitanen, A.-K.; Vainiotalo, S.; Tuomi, T.; Keskinen, J.; Hyvärinen, A.; Hämeri, K.; Saari, S. E.; Boor, B. E. Infant and Adult Inhalation Exposure to Resuspended Biological Particulate Matter. *Environ. Sci. Technol.* **2018**, *52*, 237−247.

(25) Allen, J. G.; McClean, M. D.; Stapleton, H. M.; Nelson, J. W.; Webster, T. F. Personal Exposure to Polybrominated Diphenyl Ethers (PBDEs) in Residential Indoor Air. *Environ. Sci. Technol.* **2007**, *41*, 4574−4579.

(26) Rodes, C. E.; Kamens, R. M.; Wiener, R. W. The Significance and Characteristics of the Personal Activity Cloud on Exposure Assessment Measurements for Indoor Contaminants. *Indoor Air* **1991**, *1*, 123−145.

(27) You, R.; Cui, W.; Chen, C.; Zhao, B. Measuring the Short-Term Emission Rates of Particles in the "Personal Cloud" with Different Clothes and Activity Intensities in a Sealed Chamber. *Aerosol Air Qual. Res.* **2013**, *13*, 911−921.

(28) Lax, S.; Smith, D. P.; Hampton-Marcell, J.; Owens, S. M.; Handley, K. M.; Scott, N. M.; Gibbons, S. M.; Larsen, P.; Shogan, B. D.; Weiss, S.; Metcalf, J. L.; Ursell, L. K.; Vazquez-Baeza, Y.; Van Treuren, W.; Hasan, N. A.; Gibson, M. K.; Colwell, R.; Dantas, G.; Knight, R.; Gilbert, J. A. Longitudinal Analysis of Microbial Interaction between Humans and the Indoor Environment. *Science* **2014**, *345*, 1048−1052.

(29) Bloomfield, S. F.; Aiello, A. E.; Cookson, B.; O'Boyle, C.; Larson, E. L. The Effectiveness of Hand Hygiene Procedures in Reducing the Risks of Infections in Home and Community Settings Including Handwashing and Alcohol-Based Hand Sanitizers. *Am. J. Infect. Control* **2007**, *35*, S27−S64.

(30) Boone, S. A.; Gerba, C. P. Significance of Fomites in the Spread of Respiratory and Enteric Viral Disease. *Appl. Environ. Microbiol.* **2007**, *73*, 1687−1696.

(31) Zhang, N.; Miao, R.; Huang, H.; Chan, E. Y. Y. Contact Infection of Infectious Disease Onboard a Cruise Ship. *Sci. Rep.* **2016**, *6*, 38790.

(32) Kraay, A. N. M.; Hayashi, M. A. L.; Hernandez-Ceron, N.; Spicknall, I. H.; Eisenberg, M. C.; Meza, R.; Eisenberg, J. N. S. Fomite-Mediated Transmission as a Sufficient Pathway: A Comparative Analysis across Three Viral Pathogens 11 Medical and Health Sciences 1117 Public Health and Health Services. *BMC Infect. Dis.* **2018**, *18*, 540.

(33) Zhang, N.; Li, Y.; Huang, H. Surface Touch and Its Network Growth in a Graduate Student Office. *Indoor Air* **2018**, *28*, 963−972.

https://doi.org/10.1021/acs.estlett.0c01006
Environ. Sci. Technol. Lett. 2021, 8, 326−332

(34) Lei, H.; Li, Y.; Xiao, S.; Yang, X.; Lin, C.; Norris, S. L.; Wei, D.; Hu, Z.; Ji, S. Logistic Growth of a Surface Contamination Network and Its Role in Disease Spread. *Sci. Rep.* **2017**, *7*, 14826.

(35) Bouslimani, A.; Melnik, A. V.; Xu, Z.; Amir, A.; da Silva, R. R.; Wang, M.; Bandeira, N.; Alexandrov, T.; Knight, R.; Dorrestein, P. C. Lifestyle Chemistries from Phones for Individual Profiling. *Proc. Natl. Acad. Sci. U. S. A.* **2016**, *113*, E7645−E7654.

(36) Stapleton, H. M.; Kelly, S. M.; Allen, J. G.; Mcclean, M. D.; Webster, T. F. Measurement of Polybrominated Diphenyl Ethers on Hand Wipes: Estimating Exposure from Hand-to-Mouth Contact. *Environ. Sci. Technol.* **2008**, *42*, 3329−3334.

(37) Watkins, D. J.; McClean, M. D.; Fraser, A. J.; Weinberg, J.; Stapleton, H. M.; Sjödin, A.; Webster, T. F. Exposure to PBDEs in the Office Environment: Evaluating the Relationships Between Dust, Handwipes, and Serum. *Environ. Health Perspect.* **2011**, *119*, 1247−1252.

(38) Hoffman, K.; Garantziotis, S.; Birnbaum, L. S.; Stapleton, H. M. Monitoring Indoor Exposure to Organophosphate Flame Retardants: Hand Wipes and House Dust. *Environ. Health Perspect.* **2015**, *123*, 160−165.

(39) Yang, C.; Harris, S. A.; Jantunen, L. M.; Siddique, S.; Kubwabo, C.; Tsirlin, D.; Latifovic, L.; Fraser, B.; St-Jean, M.; De La Campa, R.; You, H.; Kulka, R.; Diamond, M. L. Are Cell Phones an Indicator of Personal Exposure to Organophosphate Flame Retardants and Plasticizers? *Environ. Int.* **2019**, *122*, 104−116.

(40) Gong, M.; Weschler, C. J.; Liu, L.; Shen, H.; Huang, L.; Sundell, J.; Zhang, Y. Phthalate Metabolites in Urine Samples from Beijing Children and Correlations with Phthalate Levels in Their Handwipes. *Indoor Air* **2015**, *25*, 572−581.

(41) Liu, X.; Cao, Z.; Yu, G.; Wu, M.; Li, X.; Zhang, Y.; Wang, B.; Huang, J. Estimation of Exposure to Organic Flame Retardants via Hand Wipe, Surface Wipe, and Dust: Comparability of Different Assessment Strategies. *Environ. Sci. Technol.* **2018**, *52*, 9946−9953.

(42) Wu, C.-C.; Bao, L.-J.; Tao, S.; Zeng, E. Y. Dermal Uptake from Airborne Organics as an Important Route of Human Exposure to E-Waste Combustion Fumes. *Environ. Sci. Technol.* **2016**, *50*, 6599−6605.

(43) Shi, W.; Guo, J.; Zhou, Y.; Deng, D.; Han, Z.; Zhang, X.; Yu, H.; Giesy, J. P. Phthalate Esters on Hands of Office Workers: Estimating the Influence of Touching Surfaces. *Environ. Sci. Technol. Lett.* **2017**, *4*, 1−5.

(44) Bennett, D. H.; Furtaw, E. J. Fugacity-Based Indoor Residential Pesticide Fate Model. *Environ. Sci. Technol.* **2004**, *38*, 2142−2152.

(45) Zhang, X.; Diamond, M. L.; Ibarra, C.; Harrad, S. J. Multimedia Modeling of Polybrominated Diphenyl Ether Emissions and Fate Indoors. *Environ. Sci. Technol.* **2009**, *43*, 2845−2850.

(46) Yang, C.; Rozárka Jílková, S.; Melymuk, L.; Harris, S. A.; Jantunen, L. M.; Pertili, J.; Winn, L.; Diamond, M. L. Are We Exposed to Halogenated Flame Retardants from Both Primary and Secondary Sources? *Environ. Sci. Technol. Lett.* **2020**, *7*, 585−593.

(47) Okeme, J. O.; Yang, C.; Abdollahi, A.; Dhal, S.; Harris, S. A.; Jantunen, L. M.; Tsirlin, D.; Diamond, M. L. Passive Air Sampling of Flame Retardants and Plasticizers in Canadian Homes Using PDMS, XAD-Coated PDMS and PUF Samplers. *Environ. Pollut.* **2018**, *239*, 109−117.

(48) Saini, A.; Okeme, J. O.; Goosey, E. R.; Diamond, M. L. Calibration of Two Passive Air Samplers for Monitoring Phthalates and Brominated Flame-Retardants in Indoor Air. *Chemosphere* **2015**, *137*, 166−173.

(49) Xia, J.; Wishart, D. S. Using MetaboAnalyst 3.0 for Comprehensive Metabolomics Data Analysis. *Curr. Protoc. Bioinf.* **2016**, *55*, 14.10.1−14.10.91.

(50) Sun, Y.; Han, J. Ranking Methods for Networks. In *Encyclopedia of Social Network Analysis and Mining*; Alhajj, R., Rokne, J., Eds.; Springer: New York, 2014; pp 1488−1497.

(51) Xu, Y.; Cohen Hubal, E. A.; Clausen, P. A.; Little, J. C. Predicting Residential Exposure to Phthalate Plasticizer Emitted from Vinyl Flooring: A Mechanistic Analysis. *Environ. Sci. Technol.* **2009**, *43*, 2374−2380.

(52) Rauert, C. B.; Harrad, S. J. Mass Transfer of PBDEs from Plastic TV Casing to Indoor Dust via Three Migration Pathways - A Test Chamber Investigation. *Sci. Total Environ.* **2015**, *536*, 568−574.

(53) Eichler, C. M. A.; Hubal, E. A. C.; Xu, Y.; Cao, J.; Bi, C.; Weschler, C. J.; Salthammer, T.; Morrison, G. C.; Koivisto, A. J.; Zhang, Y.; Mandin, C.; Wei, W.; Blondeau, P.; Poppendieck, D.; Liu, X.; Delmaar, C. J. E.; Fantke, P.; Jolliet, O.; Shin, H.-M.; Diamond, M. L.; Shiraiwa, M.; Zuend, A.; Hopke, P. K.; von Goetz, N.; Kulmala, M.; Little, J. C. Assessing Human Exposure to Chemicals in Materials, Products and Articles: A Modular Mechanistic Framework. *Environ. Sci. Technol.* **2021**, *55*, 25−43.

(54) Hammel, S. C.; Hoffman, K.; Phillips, A. L.; Levasseur, J. L.; Lorenzo, A. M.; Webster, T. F.; Stapleton, H. M. Comparing the Use of Silicone Wristbands, Hand Wipes, And Dust to Evaluate Children's Exposure to Flame Retardants and Plasticizers. *Environ. Sci. Technol.* **2020**, *54*, 4484−4494.

APPENDIX 5

Stuchlík Fišerová, Petra, Lisa Melymuk, Klára Komprdová, Elena Domínguez-Romero, Martin Scheringer, Jiří Kohoutek, Petra Přibylová, Lenka Andrýsková, Pavel Piler, Holger M. Koch, Martin Zvonař, Marta Esteban-López, Argelia Castaño, Jana Klánová. 2022. "Personal Care Product Use and Lifestyle Affect Phthalate and DINCH Metabolite Levels in Teenagers and Young Adults." *Environmental Research* 213: 113675. https://doi.org/10.1016/j.envres.2022.113675

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Personal care product use and lifestyle affect phthalate and DINCH metabolite levels in teenagers and young adults

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ABSTRACT

Humans are widely exposed to phthalates and their novel substitutes, and considering the negative health effects associated with some phthalates, it is crucial to understand population levels and exposure determinants. This study is focused on 300 urine samples from teenagers (aged 12–17) and 300 from young adults (aged 18–37) living in Czechia collected in 2019 and 2020 to assess 17 plasticizer metabolites as biomarkers of exposure. We identified widespread phthalate exposure in the study population. The diethyl phthalate metabolite monoethyl phthalate (MEP) and three di (2-ethylhexyl) phthalate metabolites were detected in the urine of >99% of study participants. The highest median concentrations were found for metabolites of low-molecular-weight (LMW) phthalates: mono-*n*-butyl phthalate (MnBP), monoisobutyl phthalate (MiBP) and MEP (60.7; 52.6 and 17.6 μg/L in young adults). 1,2-cyclohexanedicarboxylic acid diisononyl ester (DINCH) metabolites were present in 68.2% of the samples with a median of 1.24 μg/L for both cohorts. Concentrations of MnBP and MiBP were similar to other European populations, but 5–6 times higher than in populations in North America. We also observed large variability in phthalate exposures within the study population, with 2–3 orders of magnitude differences in urinary metabolites between high and low exposed individuals. The concentrations varied with season, gender, age, and lifestyle factors. A relationship was found between high levels of MEP and high overall use of personal care products (PCPs). Cluster analysis suggested that phthalate exposures depend on season and multiple lifestyle factors, like time spent indoors and use of PCPs, which combine to lead to the observed widespread presence of phthalate metabolites in both study populations. Participants who spent more time indoors, particularly noticeably during colder months, had higher levels of high-molecular weight phthalate metabolites, whereas participants with higher PCP use, particularly women, tended to have higher concentration of LMW phthalate metabolites.

Abbreviations: 5cx-MEPP, mono(2-ethyl-5-carboxy-pentyl) phthalate; 5OH-MEHP, mono(2-ethyl-5-hydroxy-hexyl) phthalate; 5oxo-MEHP, mono(2-ethyl-5-oxo-hexyl) phthalate; BzBP, benzyl butyl phthalate; BMI, body mass index; CELSPAC, Central European Longitudinal Studies of Parents and Children; cx-MINCH, cyclohexane-1,2-dicarboxylic acid-mono(carboxy-isooctyl) ester; cx-MiNP, 7-carboxy-(monomethyl-heptyl) phthalate; DEHP, di(2-ethylhexyl) phthalate; DEP, diethyl phthalate; DCHP, dicyclohexyl phthalate; DiBP, diisobutyl phthalate; DiDP, diisodecyl phthalate; DINCH, 1,2-cyclohexanedicarboxylic acid diisononyl ester; DiNP, diisononyl phthalate; DMP, Dimethyl phthalate; DnBP, di-*n*-butyl phthalate; DnOP, di-*n*-octyl phthalate; DPHP, bis(2-propylheptyl) phthalate; ELSPAC, European Longitudinal Study of Pregnancy and Childhood; EU, European Union; HBM, human biomonitoring; HMW, high-molecular weight; IQ, intelligence quotient; LMW, low-molecular weight; MBzP, monobenzyl phthalate; MEHP, mono(2-ethylhexyl) phthalate; MEP, monoethyl phthalate; MCHP, monocyclohexyl phthalate; MiBP, monoisobutyl phthalate; MiNP, monoisononyl phthalate; MMP, monomethyl phthalate; MnBP, mono-*n*-butyl phthalate; MnOP, mono-*n*-octyl phthalate; OH-MiDP, mono-hydroxy-isodecyl phthalate; OH-MINCH, cyclohexane-1,2-dicarboxylic acid-mono(hydroxyl-isononyl) ester; OH-MiNP, 7-hydroxy-(monomethyl-octyl) phthalate; oxo-MiDP, mono-oxo-isodecyl phthalate; oxo-MiNP, 7-oxo-(monomethyl-octyl) phthalate; PCPs, personal care products; PVC, polyvinylchloride.

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1. Introduction

Phthalate esters, commonly referred to as phthalates, are heavily used as plasticizers and solvents because of their durability and stability. Phthalates are high production volume chemicals (Heudorf et al., 2007) with worldwide production of around 5.5 million tonnes per year (Fréry et al., 2020). Phthalates with low molecular weight (LMW) and shorter chains (e.g. DMP, DEP, DBP and DiBP) are often used as solvents in personal care products (PCPs), cosmetics, and pharmaceuticals, and as plasticizers in non-polyvinyl chloride (PVC) products, including textiles, paints, adhesives, and food packaging. Phthalates with high molecular weight (HMW) and longer chains (e.g. DEHP, DiNP and DPHP) are mainly used as plasticizers in PVC products, including in medical devices and children's toys (Fréry et al., 2020; Koch and Angerer, 2012; Silano et al., 2019). Phthalates are not covalently bonded to plastic, therefore they can be emitted from consumer products and become widely distributed in humans and the environment (Husøy et al., 2019).

In humans, several phthalates are considered endocrine disruptors (WHO, 2013) associated with potential negative health effects. The effects on developmental and reproductive disorders, including neurobehavioral disorders and low IQ, respiratory problems or asthma and other allergic disorders have been reviewed elsewhere (Hlisníková et al., 2020; Katsikantami et al., 2016; Wang et al., 2019). Due to these adverse health effects, some phthalates have been legislatively banned in the EU (EC, 2009; REACH, 2006) and alternative non-aromatic, non-ortho-phthalate plasticizers are seeing increasing use (Lemke et al., 2021; Frederiksen et al., 2020; Lessmann et al., 2019). One of these plasticizers is DINCH (1,2-cyclohexanedicarboxylic acid diisononyl ester), which is used as a substitute for HMW phthalates, mainly DEHP and DiNP, particularly in sensitive products such as toys, food contact material and medical devices (Koch et al., 2013; Correia-Sá et al., 2017; Kasper-Sonnenberg et al., 2019). DINCH production is high (>10,000 t/year production and import to EU, 300,000 t/y global production), and DINCH is among the alternative plasticizers seeing substantial increases in use and exposure (Bui et al., 2016; Kasper-Sonnenberg et al., 2019). DINCH intake thresholds and hazard-based limit values suggest a lower hazard than for phthalates, however, there is particular concern about DINCH exposure in children, given its use as a replacement for HMW phthalates in toys, and evidence of exposure close to the tolerable daily intake (Bui et al., 2016).

Inhalation and dermal uptake is a relevant uptake route for the more volatile LMW phthalates, like DMP, DEP and DBP (Giovanoulis et al., 2018; Wormuth et al., 2006; Janjua et al., 2008; Lorber et al., 2017). Oral routes play the most important roles in total exposure to the HMW phthalates like DEHP and DiNP (Koch et al., 2013; Martínez et al., 2018; Correia-Sá et al., 2018). After exposure, phthalates in humans are rapidly metabolized by phase I and phase II reactions and excreted in both conjugated and free forms via urine (Silva et al., 2003), and partially also in feces (Domínguez-Romero and Scheringer, 2019). Thus, human biomonitoring typically focuses on phthalate metabolites in urine (Koch and Calafat, 2009; Calafat et al., 2013). More polar and LMW phthalates are hydrolysed to monoester forms and are eliminated mostly in free form and glucuronidated conjugates. In contrast, longer chain HMW phthalates (and DINCH) are further metabolized to secondary, alkyl chain oxidised metabolites (Fréry et al., 2020; Wittassek et al., 2011; Schütze et al., 2017).

Biomonitoring suggests that nearly all the European population is exposed to phthalates (Den Hond et al., 2015; Koppen et al., 2019). A recent study on phthalate metabolites in 140 Norwegian adults found 10 metabolites in 100% of the samples and all remaining phthalate and DINCH metabolites in 88–97% of the samples (Husøy et al., 2019). In a Slovenian study of 387 people, detection rates were 97–100% for all phthalate metabolites (Runkel et al., 2020). Several recent studies also point out the rather rapid changes in exposures to phthalates and their substitutes due to market changes and regulatory measures (Frederiksen et al., 2020; Apel et al., 2020; Lemke et al., 2021; Gyllenhammar et al., 2017). Human biomonitoring (HBM) is a useful and necessary tool to assess total exposure to phthalates in a timely and rapid manner, regardless of the route of exposure. In combination with other accompanying data (e.g. questionnaire data) HBM provides valuable information to help us better understand important exposure sources, as well as the health effects related to chemical exposure. HBM4EU is a joint EU project co-funded under Horizon 2020 to coordinate and advance human biomonitoring, involving 30 countries, the European Environment Agency and the European Commission. HBM under HBM4EU provides better evidence and understanding of the exposure of European citizens to chemicals, and the possible impact of chemical exposure on human health. In this study, we focus on plasticizer metabolites in urine samples from teenagers and young adults from the Czech Republic, collected as a part of the HBM4EU project.

2. Materials and methods

2.1. Sample information

Three hundred urine samples were obtained from 12 to 17-year-old teenagers residing in the South Moravian region of Czechia, identified as the "CELSPAC: Teenagers cohort" (TAC) (Table 1). The CELSPAC: Teenagers study was approved by the Research Ethics Committee of Masaryk University, Czech Republic (Ref. No: EKV-2019-046, dated May 27, 2019). Most urine was collected in 2019, between October and December; one urine sample was collected in January 2020. An additional 300 urine samples were obtained from participants from 18 to 37 years of age, also from South Moravia, Czechia, identified as the "CELSPAC: Young Adults cohort" (YAC) (Table 1). CELSPAC: Young Adults represents a follow-up study of longitudinal ELSPAC study in the Brno region of the Czech Republic (Piler et al., 2017) and follows ELSPAC children born in 1991 and 1992, their siblings, and spouses (the study is ongoing). The CELSPAC: Young Adults study was approved by the ELSPAC Ethics Committee (Ref. No: ELSPAC/EK/2/2019, dated March 13, 2019). YAC urine samples were collected in 2019, between March and December. All urine samples from the TAC and YAC cohorts were spot samples.

All study participants completed a questionnaire at the time of urine sample collection gathering information including age, gender, number of siblings, education and household income, time spent indoors and various parameters of the home environment, dietary information, smoking (active and passive), PCP usage and chronic illnesses. Details about the cohorts (age distribution, education and income, lifestyle variables) are given in the Supplementary Material, Figures S1-S5, Table S1.

Urine was analysed for creatinine concentration as well as for specific gravity (SG) and a wide set of chemical biomarkers, including phthalates and DINCH (presented here), as well as bisphenols, hydroxylated polycyclic aromatic hydrocarbons and metals (in preparation).

2.2. Chemicals and reagents

High purity (>97%) standards of 15 phthalate metabolites (MEP, MiBP, MBzP, MnBP, MCHP, MEHP (AccuStandard, USA), 5OH-MEHP, 5oxo-MEHP, 5cx-MEPP, (Toronto Research Chemicals, Canada), MnOP, OH-MiNP, cx-MiNP, oxo-MiNP, OH-MiDP and oxo-MiDP (Cambridge Isotope Laboratories, USA)), along with a creatinine standard solution, were purchased. OH-MINCH, cx-MINCH, OH-MINCH-d_4 and cx-MINCH-d_2 standards were provided by the Institute for Prevention and Occupational Medicine of the German Social Accident Insurance - Institute of the Ruhr-University Bochum (IPA, Germany). High purity (>98%) standards of 15 isotopically-labelled phthalate metabolites (MEP-$^{13}C_4$, MiBP-$^{13}C_4$, MBzP-$^{13}C_4$, MnBP–$^{13}C_4$, MCHP-$^{13}C_4$, 5OH-MEHP-$^{13}C_4$, 5oxo-MEHP-$^{13}C_4$, 5cx-MEPP-$^{13}C_2$, OH-MiNP-$^{13}C_2$, cx-MiNP-$^{13}C_4$, oxo-MiNP-$^{13}C_2$, OH-MiDP-$^{13}C_2$, oxo-MiDP-$^{13}C_2$, (Cambridge Isotope Laboratories, USA), MEHP-d_4 and MnOP-d_4 (Chiron, Norway)) were purchased.

The following chemicals were used: a solution of β-glucuronidase (*E. coli*, Type IX-A 1–5MU/g), acetic acid, ammonium fluoride, ammonium acetate and Milli-Q water (Sigma-Aldrich, USA). The following solvents were purchased: acetonitrile and methanol for LC-MS (Biosolve Chimie, France), dimethylsulfoxide (DMSO, Sigma-Aldrich, USA).

2.3. Sample preparation procedure

Frozen aliquots were allowed to reach laboratory temperature, homogenized with a Wizard Advanced IR vortex mixer (Velp Scientifica, USA) and 500 μL of each sample were pipetted into a 96-well sample collection plate with 2 mL square wells (Waters, Prague). Then, 500 μL of a mixture containing β-glucuronidase solution (1000 units/mL) and isotopically-labelled standards (c = 10 ng/mL) in 0.1 M acetate buffer were added to the samples. The 96-well plate was covered with foil and samples were incubated using a sample concentrator (Miu Lab, China) for 120 min at 55 °C. After incubation, 50 μL of 1% acetic acid was added. Samples were then precleaned using 96-well plate SPE (Oasis HLB; 3 mL, 60 mg; Waters, Ireland), which was previously conditioned with 1 mL of methanol and activated with 0.1% acetic acid. Samples were passed through the SPE plate, then the wells were washed with 0.1% acetic acid and dried using a Laboport vacuum pump (KNF, Germany) for at least 30 s. Finally, the samples were eluted with 1.5 mL of acetone and collected into a new 96-well plate containing 10 μL of DMSO. After the clean-up step, samples were concentrated to 10 μL. Finally, 500 μL of 50% methanol were added, samples were covered with a foil, homogenized and kept in the fridge (4 °C) until analysis.

2.4. Liquid chromatography-mass spectrometry (LC-MS) conditions

The analytical method for phthalate metabolite determination was developed on the Agilent 1290 Infinity II HPLC system (Agilent Technologies, Germany) for separation. An Acquity UPLC BEH C18 (100 × 2.1 mm, 1.7 μm, Waters, Czechia) was used as an analytical column. The mobile phases were A: 0.1 mM ammonium fluoride in Milli-Q water and B: 0.1 mM ammonium fluoride in methanol. The gradient elution was set up as follows: mobile phase A (10%) for 7 min, followed by an immediate increase to 50% for the rest of the analysis. The injection volume was 5 μL and the flow rate was set to 250 μL/min.

Agilent QQQ 6495 (Agilent Technologies, Germany) was used for analyte detection, with an electrospray ionisation (ESI) source, in negative mode. Optimal parameters for phthalate metabolites are shown in Table S2. The isotopic dilution method was used for data quantification. A corresponding isotopically-labelled standard was used for each compound as the internal standard. MassHunter software was used to process the data from instrumental analysis.

2.5. Specific gravity

For the SG measurement, a “Pocket” Urine Specific Gravity Refractometer PAL-10 S (Atago, Japan) was used. First, 300 μl of Milli-Q water was pipetted to the refractometer to reach a value of 1.000. Then, the refractometer was wiped with a paper towel. The urine sample was vortexed and 300 μL were pipetted onto the refractometer. When the temperature stabilised, the value was deducted from the refractometer. The procedure was repeated for all the samples.

2.6. Quality assurance/quality control (QA/QC)

The methods used were developed and validated under the HBM4EU framework through the successful completion of four rounds of proficiency testing (Esteban López et al., 2021). We report 10 phthalate and 2 DINCH biomarkers that are quality assured under HBM4EU (both external and internal quality assurance) and 4 biomarkers under additional internal quality assurance.

External quality assurance: For external quality assurance we participated in all four rounds of the HBM4EU Proficiency testing (Esteban López et al., 2021) and received certificates for the following biomarkers: MEP, MBzP, MiBP, MnBP, MEHP, 5OH-MEHP, 5oxo-MEHP, 5cx-MEPP, OH-MiNP, cx-MiNP (Elbers and Mol, 2019a), OH-MINCH and cx-MINCH (Elbers and Mol, 2019b).

Internal quality assurance: Procedural blanks were prepared with Milli-Q water, approximately one blank for every ten urine samples. The samples were blank subtracted based on the average mass of metabolite found in the blanks (Table S3). LODs were defined as three times the standard deviation of the blank samples (Table S3). A calibration curve was prepared in the range of 0.01–100 ng/mL, containing 10 concentration points and showing linearity. QC samples were analysed approximately every ten samples. The average recoveries of QC samples ranged from 83 to 110% (except for MnBP with 124.1%) with RSD ranging from 5.1 to 16.5 (except for MEP and MnBP with 38.1 and 40.5% respectively).

2.7. Data adjustment

SG adjustment was applied to urine samples, since creatinine correction has a higher dependence on age (Carrieri et al., 2000), time of urine sampling, sex, activity, diet (Miller et al., 2004), muscularity, BMI, disease status (Wang et al., 2015) and seasonal variations (Pearson et al., 2009). Moreover, degradation studies suggest that after only 10 days, approximately 20% of creatinine is degraded via freezing (storage at −20 °C), while SG is not (Schneider et al., 2002). Considering all this, SG correction was deemed to be the more reliable correction for the TAC and YAC samples.

2.8. Data analysis

For the correlation analyses, only values > LOD were used. For the rest of the statistical analyses, values below LOD were substituted with LOD/$\sqrt{2}$. Nonparametric Spearman’s correlation was used to determine the relationship among phthalate metabolites. To investigate seasonal variations, YAC samples were divided into two groups according to the sampling season. The cold season covered seven months from October to April, and the warm season covered five months from May to September. Basic statistical characteristics (detection frequency, median, minimal

Table 1
Profile of studied cohorts. YAC represents Young Adult cohort, TAC represents Teenage cohort, N_{mis} represents number of samples where we lack corresponding data from questionnaires.

		Young Adults (YAC) ($n = 300$)				Teenagers (TAC) ($n = 300$)		
		Median	10th90th	Minmax	N_{mis}	Median	10th90th	Minmax
Age (years)		27	26–28	18–37		13	12–15	12–17
Weight (kg)		72.6	56.5–92.5	47.3–117.3	2	Not collected		
Height (cm)		174.1	163.2–188.3	155.1–202.3	2	Not collected		
BMI (kg m^{-2})		23.3	19.9–28.7	17.3–39.2	2	Not collected		
Gender	female	155; 52%				124; 42%		
	male	145; 48%				176; 58%		

and maximal value with 5th and 95th percentiles) are shown in Table S4 for YAC and in Table S5 for TAC. Table S6 refers to number of samples and medians for categories of the exposure determinants used for non-parametric testing.

Linear regression was used to explore the relationship between phthalate metabolites and multiple exposure factors (gender, age, season of sample collection, time indoors, and sum of all PCP products used). Only metabolites with more than 75% of samples above the LOD for each cohort were used for regression analysis (MEP, MiBP, MBzP, MnBP, 5OH-MEHP, 5cx-MEPP, OH-MiNP). The regression analysis used gender and season as categorical predictors; age, time spent indoors, and sum of PCPs were used as continuous predictors. Use of individual PCPs could not be incorporated as predictors in the regression models, due to the small sample sizes in some groups stratified by e.g., cohort/age, gender and season. Metabolite concentrations were log-transformed, and outliers were excluded prior to analysis. The normal distribution of the residuals was checked by using histograms and the Kolmogorov-Smirnov test. Regressions for each exposure factor and metabolite were adjusted for other factors to explain the main effects of each factor. Another regression model for each metabolite was computed with all factors and their interactions to determine the maximum explained variability (coefficient of determination, R^2) in phthalate concentrations by the given factors. Factor interactions were added to the model because we assume that one factor can influence the effect of another factor (e.g. indoors and season) on the dependent variable (phthalate metabolite concentrations).

To supplement the key factors identified by the regression analysis, individual exposure factors were also examined, both individually, and stratified by age, season, and gender. The non-parametric Mann-Whitney *U* test was used to test for differences in concentrations of phthalate metabolites with questionnaire parameters with two categories such as gender, season (cold vs. warm), smoking (YES/NO), redecoration and renovation made at home in the last two years (YES/NO), drinking beverages from plastic bottles (YES/NO) and for the use of some PCPs (Table S7). Tests were also performed for each combination of factors.

Due to inhomogeneity of the variances and non-equal sample sizes in some categories of PCPs, nonparametric Kruskal-Wallis ANOVA was used to test for differences in concentrations of phthalate metabolites between multiple categories of frequencies of PCP use. To remove the influence of other factors, participants were first stratified by age (TAC vs YAC), gender, and season before testing of PCPs. For some frequencies of PCP use stratified by gender, age and season, the number of samples for testing was insufficient, due to the nature of the PCP use (e.g. eye make-up or nail polish is used mainly by women). The PCP use categories “never” and “sometimes” for shampoo and deodorant were combined because very few participants did not use these products at all. Tests were performed when the number of samples in each category was greater than 20 (Table S7) and only for metabolites with more than 75% samples above the LOD for each cohort (MEP, MiBP, MBzP, MnBP, 5OH-MEHP, 5cx-MEPP, OH-MiNP).

Finally, two cluster analyses were completed as an explanatory technique for cumulative phthalate exposure and cumulative PCP use:

i) First, Ward’s hierarchical clustering method with the squared Euclidean distance was used as an explanatory technique to create clusters of participants with similar phthalate exposure. Metabolite concentrations were log-transformed and standardized (*z*-score) prior to cluster analysis. The cluster analysis provides insight into the total cumulative exposure of all metabolites. Specifically, participants with similar concentrations of phthalate metabolites (e.g. participants with high/low levels of all metabolites) were clustered. Then it was determined how the exposure factors (age, gender, season, time spent indoors, PCPs) were distributed among these clusters. Differences in cumulative concentrations of LMW, HMW phthalate metabolites and DINCH between clusters were tested for by the Kruskal-Wallis ANOVA.

ii) In the second case, cluster analysis was used to create clusters of participants with similar use of personal care products. Frequencies of use of individual PCPs were expressed as number of days per week and clustered using Ward’s hierarchical clustering method with the squared Euclidean distance. In contrast to the Kruskal-Wallis ANOVA, which tests differences for each PCP separately, the cluster analysis shows the common use of all PCPs and complements the results from the regression analysis where individual PCPs and their interactions could not be included. This analysis was conducted for the YAC and TAC cohorts separately. The distribution of gender and season was determined for each cluster and differences in the concentrations of each metabolite in the clusters were tested using Kruskal-Wallis ANOVA.

3. Results and discussion

Twelve of the 17 targeted phthalate metabolites were broadly detected in TAC and YAC. The most frequently detected compounds were MEP, 5oxo-MEHP and 5cx-MEPP, detected in >99% of the YAC samples and in all TAC samples, and OH-MINP and 5OH-MEHP were often detected (>98%) in both cohorts. The detection frequencies, medians, geometric means, and ranges are shown in Table 2. Other statistical characteristics (minimum, maximum, 5th and 95th percentiles) for both cohorts are shown in Tables S4 and S5.

In the YAC, the highest median concentration was found for MnBP, followed by MiBP and MEP (60.6; 52.6 and 17.6 μg/L, respectively). All DEHP metabolites, OH-MiNP, oxo-MiNP and OH-MINCH had medians between 1.34 and 7.92 μg/L, while the other metabolites were detected with median concentrations under 1.01 μg/L (Table 2). In the TAC, the highest median concentrations were also found for MnBP, followed by MEP and MiBP (45.5; 38.1 and 28.8 μg/L, respectively). MBzP, all DEHP metabolites, the secondary DiNP metabolites and OH-MINCH were detected with the medians in the range of 1.71–13.9 μg/L. Despite the use of DINCH as a replacement for DEHP in sensitive applications (Bui et al., 2016), the DINCH metabolites are detected at lower levels and with lower detection frequency than DEHP metabolites. The other metabolites were detected in median concentrations below 1 μg/L (Table 2). MEP, MiBP, MnBP, MBzP, 5OH-MEHP, 5cx-MEPP, OH-MiNP, cx-MiNP, OH-MiDP and OH-MINCH (marked with [a] in Table 2) were selected as priority metabolites for further statistical analysis based on their appropriateness as biomarkers of the parent compounds and detection frequency. Correlations between individual phthalate metabolites were calculated to examine associations between specific phthalate metabolites and common sources. The strongest correlations were found between metabolites of the same parent compound (e.g., DEHP metabolites 5OH-MEHP, 5oxo-MEHP and 5cx-MEPP; and DINCH metabolites OH-MINCH, cx-MINCH) (Figure S6).

3.1. Determinants of phthalate exposure

The regression analysis on all data (YAC + TAC, $N = 600$) for MiBP, MEP, MBzP, MnBP, 5cx-MEPP, 5OH-MEHP, OH-MINP show the contributions of both individual factors and their interactions to metabolite concentration (Table S8a,b).

Regression analysis highlighted the differences between exposures to low and high molecular weight phthalates. Gender, sampling season, and age were significant factors for HMW phthalate metabolites 5cx-MEPP, 5OH-MEHP, OH-MINP, with explained variability around 10%. The only significant interaction for HMW metabolites were gender and age for 5cx-MEPP, time indoors, and age for 5OH-MEHP and time indoors and season for OH-MINP, but these interactions did not significantly increase R^2 (Table S8). For the LMW phthalate metabolites MiBP, MEP, MBzP, the main significant factors were similar – gender and age – with an explained variability between 10% and 16%, but the significant interaction of age and gender with total PCP use increased the explained variability of MiBP and MEP to 18% and 25%, respectively. This

Table 2
LODs, detection frequencies and median concentrations (μg/L, SG adjusted) with range (min-max) and geometric means of phthalate metabolites for both cohorts.

			YAC			TAC		
Parent phthalate	Phthalate metabolite	LOD (μg/L)	Detection frequency (%)	Median (minmax)	Geo. Mean	Detection frequency (%)	Median (minmax)	Geo. Mean
DEP	MEP [a]	0.20	99.0	17.5 (0.02–860)	22.0	100	38.1 (3.04–704)	42.2
DiBP/DnBP	MiBP [a]	0.07	99.6	52.6 (0.07–446)	64.0	97.3	28.8 (0.03–486)	26.1
BzBP	MBzP [a]	0.10	75.6	0.69 (0.10–25.3)	0.74	94.6	1.74 (0.07–112)	1.85
DnBP/BzBP	MnBP [a]	0.18	96.3	60.6 (0.18–664)	60.5	91.0	45.5 (0.08–303)	28.8
DCHP	MCHP	0.20	0	0.20 (0.20–0.20)	0.18	0.3	0.14 (0.08–1.07)	0.15
DEHP	MEHP	0.18	93.6	2.29 (0.18–40.1)	2.57	93.6	2.14 (0.11–63.6)	2.09
DEHP	5OH-MEHP [a]	0.07	99.6	7.85 (0.07–150)	9.30	99.3	13.9 (0.03–679)	14.2
DEHP	5oxo-MEHP	0.07	99.3	3.65 (0.07–113)	4.43	100	5.56 (1.33–74.5)	6.00
DEHP	5cx-MEPP [a]	0.07	99.6	5.04 (0.07–108)	6.02	100	7.52 (1.74–128)	8.32
DnOP	MnOP	0.20	0.3	0.20 (0.20–2.95)	0.18	3.3	0.14 (0.08–6.63)	0.16
DiNP	OH-MiNP [a]	0.30	98.3	7.92 (0.30–1649)	11.1	98.3	11.4 (0.16–169)	12.1
DiNP	cx-MiNP [a]	0.20	49.6	0.20 (0.20–105)	0.48	100	5.34 (0.96–503)	6.19
DiNP	oxo-MiNP	0.20	92.0	1.99 (0.20–111)	2.59	97.3	2.75 (0.16–228)	3.01
DiDP	OH-MiDP [a]	0.20	31.6	0.20 (0.20–102)	0.37	52.3	0.55 (0.09–13.4)	0.48
DiDP	oxo-MiDP	0.10	41.6	0.10 (0.10–60.4)	0.26	55.0	0.39 (0.05–14.2)	0.30
DINCH	OH-MINCH [a]	0.30	61.3	1.34 (0.30–952)	1.67	75.0	1.71 (0.15–494)	1.76
DINCH	cx-MINCH	0.25	39.6	0.25 (0.25–277)	0.74	51.0	0.76 (0.10–204)	0.80

[a] Indicates a metabolite was selected as priority biomarker for further analysis.

suggests the greater importance of PCPs use for LMW phthalate exposure (Table S8). We found no relationship between the factors and MnBP concentration. However, an important outcome of this is the limited explanatory power of these selected factors for phthalate exposure. Thus, the majority of the variance in phthalate metabolite levels remains unexplained by these factors.

We evaluated the impact of key determinants in greater detail: age, season, gender, time indoors and PCP use on phthalate metabolite levels to identify key determinants of phthalate exposure in the teenager and young adult cohorts.

3.1.1. Age

A limited relationship between age as a continuous variable and HMW metabolites was observed (Table S8), but when age was treated as a categorical variable (YAC vs. TAC) a few clear differences were apparent (Fig. 1, Table S9). Notably, MiBP was significantly higher in YAC ($p < 0.05$), with median concentrations two times higher than in TAC. MnBP was also detected in higher concentrations ($p < 0.1$) in YAC (60.65 μg/L compared to 45.47 μg/L in TAC). On the other hand, statistically significant differences were found for MEP, MBzP and 5OH-MEHP with median concentrations approximately two times higher in TAC; and the rest of the secondary DEHP metabolites were also detected in higher concentrations in TAC (Table 2), but these differences were not statistically significant. These differences may be due to lifestyle differences between teenagers and young adults, and are discussed further below.

3.1.2. Season

Analysis of the season of sample collection was stratified by cohort and gender, and only performed on the YAC, as all TAC samples were collected in the cold season. The results of the non-parametric tests showed that in YAC, statistically significant differences were found for 5OH-MEHP and OH-MiNP with 20% and 100% higher concentrations, respectively, in urine samples collected in the warm season ($p < 0.05$), and concentrations of cx-MiNP were also 50% higher ($p < 0.1$) (Tables S6 and S9). This may indicate a link between levels of phthalates in the ambient environment and human exposures (Pilka et al., 2015). Phthalates are found in outdoor air in higher concentrations in warmer months (Pilka et al., 2015; Puklová et al., 2019), attributed to increases in both primary and secondary emissions at higher temperatures

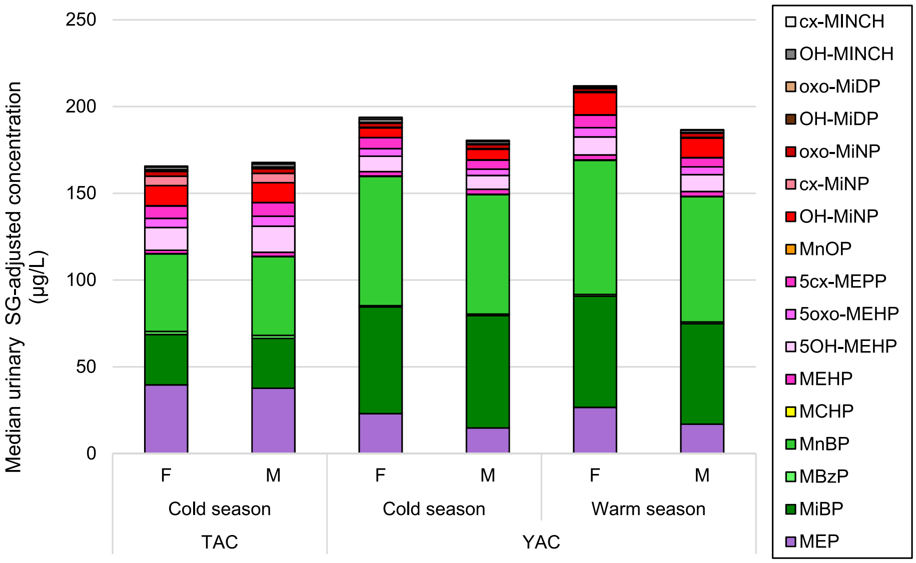

Fig. 1. Median concentration of phthalate metabolites in both cohorts. M represents males, F represents females.

(Vasiljevic et al., 2021). Higher emissions, and resulting higher environmental concentrations suggest the potential for greater human exposure in warm seasons, reflected in urinary metabolites (Puklová et al., 2019).

3.1.3. Gender

The results of the non-parametric tests show statistically significant gender differences in three phthalate metabolites (MEP, 5oxo-MEHP and 5cx-MEPP; Figure S7) in YAC, and no gender differences for any metabolite in TAC (Table S9). There are greater gender differences in lifestyles in the YAC cohort compared to the TAC: teenagers typically spend weekdays in the same location (school) regardless of gender, while there is greater differentiation in adult occupations by gender, and PCP use in teenagers is similar across males and females, but differs significantly by gender for young adults (Figures S4 and S5).

In YAC, MEP had significantly higher concentrations in females, and this difference was more significant in the cold season ($p < 0.05$ in cold season, $p < 0.1$ in whole year) (Table S9). Two DEHP metabolites (5oxo-MEHP and 5cx-MEPP) also showed significantly higher concentrations in females (Table S9). Gender-dependent differences in oxidative DEHP metabolism have been identified (Tait et al., 2020), which could lead to differences between male and female concentrations of urinary metabolites (Koch et al., 2017). Higher MEP levels in women have been noted in other studies (Runkel et al., 2020; Wormuth et al., 2006) attributed to gender differences in PCP use. This suggests an association between higher MEP levels and frequency and amount of PCP use, which the gender differences are indirectly indicating.

3.2. Lifestyle determinants of phthalates exposure

The seasonal and gender-related differences, while in some part attributable to differences in environmental levels and physiological factors, are largely due to the lifestyle differences that manifest across seasons and between different populations. The key lifestyle aspects which are hypothesized to be related to phthalate exposure are time spent indoors, indoor activities such as cleaning, use of personal care products containing phthalates, and diet. We examined the individual influence of individual exposure determinants, excluding diet, as well as possible combined effects. Diet was excluded because we had limited information on dietary patterns that would be directly relevant to phthalate exposure. We only examined a link between phthalate metabolite levels and frequency of drinking from plastic bottles, and for this variable no relationships were found.

3.2.1. Exposures via the indoor environment

When comparing urinary metabolite concentrations with self-reported time spent indoors, we did not find a statistically significant relationship between the time spent indoors and individual phthalate metabolite for either YAC or TAC. Higher MEP, MBzP, 5oxo-MEHP, 5OH-MEHP, 5cx-MEPP, OH-MiNP and cx-MiNP concentrations were observed in TAC samples, indicating higher exposures to DEP, DEHP and DiNP. This may be related to longer times spent indoors by teenagers (with a median of 14 h, compared to 12 h for YAC, Figure S3). Moreover, this is similar to the effect of inclusion of time spent indoors in the regression model, where interaction of time spent indoors with age for 5OH-MEHP and with season for OH-MINP was statistically significant but did not lead to a significant increase in explained variability (Table S8). Urinary levels of MBzP and MEP have been positively correlated with indoor levels of BzBP and DEP in several previous studies, suggesting exposure via indoor air and dust as important routes of exposure to these phthalates (Adibi et al., 2008; Bekö et al., 2013; Langer et al., 2014); however, we note that these studies did not identify indoor sources as important for HMW phthalates.

No statistically significant differences were found between any metabolites and renovations/redecorations done in the past 2 years.

3.2.2. Exposures via PCP use

Participants reported frequency of PCP use per week in the four weeks prior to sample collection. The specific PCPs identified in questionnaires were shampoo, scented products (deodorants and perfumes), lotions, nail polish, make-up/foundation creams, lip balms and eye make-up. We tested differences in phthalate metabolite concentrations for different frequencies separately stratified by age (YAC vs TAC), gender and season to remove the influence of these factors. In TAC, we found statistically significantly higher levels of MEP for both genders in participants reporting using deodorant compared with those “never” or “once a week“ using them (Table S7), as well as an increasing trend with increasing frequency of use (Fig. 2), which agrees with reports of high levels of DEP in perfumes (Guo and Kannan, 2013) and a previous study comparing urinary metabolites with PCP use (Nassan et al., 2017). In the YAC cohort, no differences in MEP concentrations were found when scented products were used because there is likely overlap with the use of other PCPs (see below).

Associations with nail polish use were observed in YAC in both the stratified and overall analyses. Women reporting use of nail polish had statistically significant higher levels of 5cx-MEPP and OH-MINP in the cold season than those reporting no use (Table S7), and when including all YAC (male and female), participants reporting “never” using nail polish had lower levels of 5cx-MEPP compared with those using nail polish four to six times per week (Figure S8). However, due to the low number of participants who used nail polish at a high frequency (7 times per week, $n = 3$), the significance of the highest use category could not be statistically tested. Nail polish use presents a documented pathway for dermal uptake of chemicals, as noted by Mendelsohn et al. (2016), and while exposure to DEHP is not typically associated with PCP use, some nail polishes have been also identified to contain DEHP (Guo and Kannan, 2013; Young et al., 2018). Moreover, nail polish is a long-term PCP exposure route, as nail polishes typically stay on the nails for days to weeks, meaning that the spot samples collected for this study may better capture this exposure source, in contrast to rinse-off PCP products. In the TAC cohort, no difference was observed in females who reported use vs. no use of nail polish (Table S7), likely because almost all participants used nail polish with a very low frequency (Figure S5A).

While the analysis comparing the use of individual products was able to identify some relationships (notably with deodorant and nail polish use), many of the product categories represent relatively small potential exposures (e.g., lip balm, eye shadow), which independently may not lead to substantial increases in phthalate exposures. Moreover, we do not know which brands of products were used by study participants, and there may be high variability in the phthalate content of products on the market, e.g., only some nail polishes contain phthalates (Guo and Kannan, 2013; Young et al., 2018). However, we hypothesized that by clustering the study participants according to patterns of PCP use, we could discern the influence of overall PCP use on human exposure to

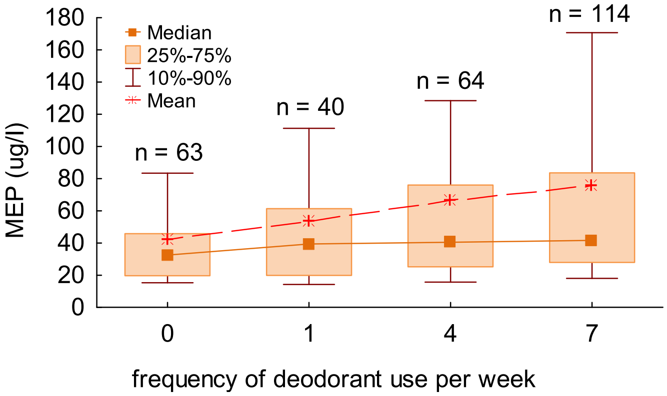

Fig. 2. Difference in MEP concentration for different frequency (times per week) of deodorant use in the TAC cohort (teenagers).

phthalates. Regression analysis shows that added interaction with the total PCP use increased the explained variability of MEP concentrations by 9%. In the case of TAC, the clusters do not provide any additional insight into PCP use patterns, mainly due to the similarity of behaviour among teenagers and low use of make-up products. However, for YAC, clear patterns of PCP use and associated exposures are visible (Figure S9). YAC data were separated into seven clusters according to PCP use, with cluster A associated with the least overall PCP use, and cluster D-G associated with frequent PCP use across most categories. Clusters A, B, and C were dominated by use of only shampoo and scented products, and included mainly male subjects, while clusters D-G included mainly female subjects and had notably higher uses of additional PCPs like lotion and decorative cosmetic products. This pattern of higher overall use of PCPs, mainly decorative cosmetic products, is connected with higher urinary concentrations of MEP (Figure S9) and presents a possible explanation for the higher levels of MEP noted in females (Table S7). However, there is no statistically significant difference in MEP concentrations between clusters, mainly due to the high variance in concentrations.

3.2.3. *Patterns in overall phthalate exposure*

In view of the complex factors leading to phthalate exposures, we combined all phthalate metabolite data (YAC + TAC, $N = 600$) to evaluate the overall pattern of phthalate metabolites and exposure determinants. We hypothesize that the cumulative effect of different lifestyles and behaviours drives phthalate exposures, rather than individual products or activities.

Cluster analysis was selected as an additional multivariate technique to provide insight into patterns of exposure and exposure determinants. Cluster analysis was used to group participants who had similar metabolite concentrations from the lowest (A) to the highest (H) according to the median metabolite concentrations in the cluster (Fig. 3). These clusters had significant differences in metabolites associated with LMW phthalates, HMW phthalates and DINCH, with Cluster H notably higher in HMW phthalate and DINCH metabolites, and Cluster G having substantially higher levels of LMW phthalate metabolites (Figure S10; Table S10). The lifestyle variables associated with each cluster were then summarized. However, it is important to note that it is not possible to strictly divide phthalate metabolites according to their source, in this case, either from PCPs or from building materials/indoor products. We note that the cluster with the highest phthalate metabolite levels (Cluster H in Fig. 3) contains mainly cold season samples from men with an average age of 19 years with high PCP use, especially of products like shampoo and lotion which have larger application quantities, and among the highest time spent indoors. Additionally, this is the only cluster where a higher concentration of the alternative plasticizer metabolite was observed: Cluster H has a much higher contribution of OH-MINCH to the total plasticizer metabolites than any other cluster. Use of DINCH as a DEHP and DINP replacement is primarily in sensitive applications such as food contact materials, medical devices and toys (Bui et al., 2016). The reasons for the predominance of OH-MINCH in Cluster H are not fully clear; it may be due to behaviours associated with increased time indoors, as well as food consumption habits which were not well captured by the determinants of exposures available.

In contrast, a low exposure cluster (Cluster C) has low use of the PCPs with large application quantities, and the lowest time spent indoors. Similarly, clusters C and E, where participants declare lower time spent indoors, have smaller contributions of HMW phthalate metabolites. Additionally, clusters A and B have low levels and were largely all collected in cold seasons, which corresponds with the identification of low environmental levels and resulting low exposures during colder months. In some cases, the relation between the metabolite levels and

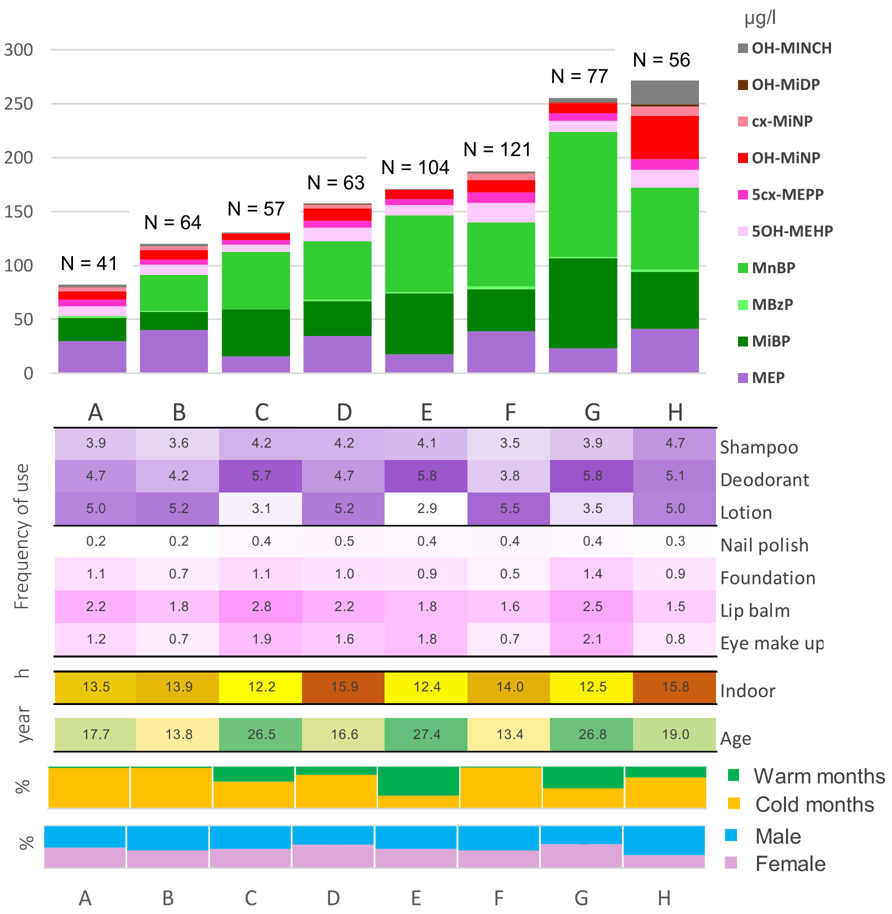

Fig. 3. Cluster analysis of metabolite concentrations in relation to lifestyle factors relating to phthalate exposure. For the metabolites, median concentrations in µg/L are given for each cluster. LMW metabolites which are often connected with PCP use are shown in tones of purple and green, HMW metabolites which are often considered “indoor exposure” biomarkers from consumer products and building materials are shown in tones of red and pink. Alternative plasticizers metabolites are shown in tones of grey. For lifestyle factors, the values indicate average PCP weekly use frequency (two-level scale – more frequently used in tones of purple, less frequently used in tones of pink), average hours per day spent indoors, and average age. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

the lifestyle factors is unclear, e.g., phthalate metabolites are high in cluster G, but that cluster has relatively low median use of PCPs, and low time spent indoors.

The results of our cluster analysis support our hypothesis that phthalate metabolite concentrations relate to a combination of separate lifestyle factors affecting phthalate exposure, rather than one dominant exposure source. Some part of this disconnect may be because we are omitting a major phthalate exposure pathway: dietary ingestion. We tested whether there was a relationship between phthalate metabolites and consumption of drinks in plastic bottles, and found no relationship, but beyond this, no factors related to dietary exposure were tested. However, dietary ingestion is known to be important for many phthalates, particularly the HMW phthalates (Giovanoulis et al., 2018; Wormuth et al., 2006). Nevertheless, cluster analysis presents a useful technique to profile the complexity of exposure factors and interrelations between variables such as age, gender, and season with PCP use and behaviours.

3.3. Comparison with other studies

We compared the median urinary concentrations of phthalates in teenagers and young adults in our study with values reported in the literature for these age groups. Specifically, median phthalate metabolite concentrations from our study were compared with data from Germany (GerES V study, 2015–2017) (Schwedler et al., 2020a, 2020b), Canada (Canadian Health Measures Survey, cycle 5, 2016–2017) (Health Canada, 2019) and USA (NHANES, 2015–2016) (CDC, 2019).

Marked differences between countries were found for MnBP and MiBP concentrations, which were around 5–6 times higher in young adults in our study than the values reported in adults from Canada and USA. In teenagers, median MnBP and MiBP concentrations were also generally higher in our study than the values reported in Canada, USA, and Germany (Fig. 4, Table S11). However, data from other studies from Czechia (Puklová et al., 2019) or other European countries (Slovakia (Pilka et al., 2015) and Denmark (Søeborg et al., 2012)) show median MnBP and MiBP concentrations more similar to those from Czechia than to those from North America (Frederiksen et al., 2010). OH-MiNP concentrations in young adults and teenagers in our study were around 15 times higher than the values reported for the same age groups in Canada, with smaller differences (2-fold) in OH-MiNP levels between Czech and German teenagers. Conversely, the median MBzP concentration in Czech young adults was around 5–6 times lower than values reported for this metabolite in Canada and the USA. DINCH metabolites are reported in all regions at similar levels.

With the general similarities in development, climate and lifestyles between Europe and North America, we attribute the differences in specific metabolites to differences in food composition; in Europe, the major sources of DEHP, DBP, DiBP and BzBP in adults is food or diet in general (Wormuth et al., 2006). Moreover, differences in PCP use, chemical legislation and building practices between North America and the EU, as well as within the EU, can also lead to differences in phthalates exposure of the general population between countries (Runkel et al., 2020). Some smaller contribution may come from differences in the identified cohort populations, e.g., teenagers are defined as 11–17 years old in Czechia, 14–17 years old in Germany, and 12–19 years old in Canada and USA.

3.4. Study limitations

Samples for both cohorts were taken in different seasons. In some cases, there are indications that self-reported data may not be correct, e.g., implausible time spent indoors. A few participants declared an average time spent indoors less than 8 h (sleep included), which seems, particularly in the case of teenagers in cold/winter months, not probable. Therefore, these values were excluded from the statistical analysis. Moreover, questionnaires asked about products used in the previous four weeks, which may not correspond directly to metabolites detected in urine samples. Diet was excluded because we had limited information on dietary patterns that would be directly relevant to phthalate exposure. Another limitation is that urine samples were spot samples collected at different times of day, with insufficient information about the timing of sample collection to allow further grouping. Values of SG were measured after one freeze-thaw cycle; the ideal measurement would be on fresh urine samples (Pearson et al., 2009). Additionally, we cannot distinguish the origin of some metabolites, particularly if the parent phthalate is DiDP or DPHP (Gries et al., 2012; Koch et al., 2017).

4. Conclusions

We identified widespread exposure to phthalate esters in the Czech population based on the quantification of phthalate metabolites in spot urine samples from teenagers and young adults. While some relationships were identified between individual exposure determinants (use of certain PCPs, season, gender) and specific phthalate metabolites, in general, the individual factors did not show a strong relationship with phthalate metabolites. We hypothesize that phthalate exposures relate

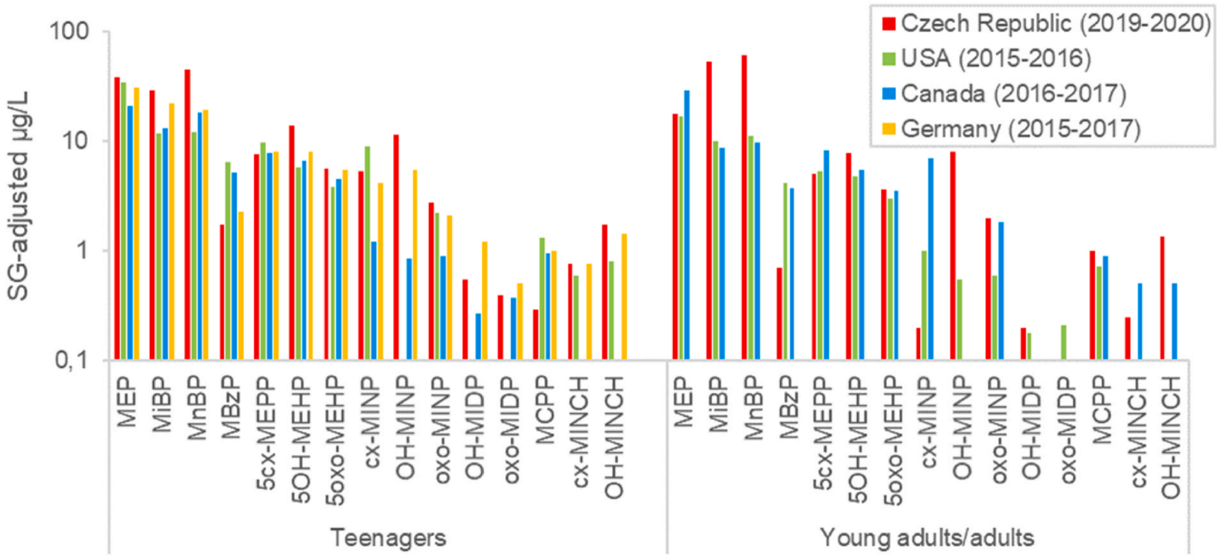

Fig. 4. Comparison of urinary concentrations of phthalate metabolites in teenagers and young adults from Czechia, Germany, Canada and USA, logarithmic scale.

to a combination of separate lifestyle factors, rather than one dominant exposure source. We tested this through cluster analyses, identifying the profiles of study participants with higher levels of phthalate metabolites. Participants who spent more time indoors, particularly noticeable during colder months, had higher levels of HMW phthalate metabolites, whereas participants with higher PCP use, particularly women, tended to have higher concentrations of LMW phthalate metabolites. We conclude that while phthalate exposure is ubiquitous, it is also very variable, according to age, gender, time spent indoors, food, and lifestyle in general.

Credit author statement

Petra Stuchlík Fišerová; – Methodology, Formal analysis, Investigation, Data curation, Writing – original draft, Writing – review & editing; **Lisa Melymuk** – Investigation, Writing – review & editing, Supervision; **Klára Komprdová**; – Data curation, Visualization; Writing – review & editing; **Elena Domínguez-Romero** – Investigation, Writing – review & editing; **Martin Scheringer** – Writing – review & editing; **Jiří Kohoutek** – Methodology, Formal analysis; **Petra Přibylová**; – Resources, Supervision; **Lenka Andrýsková**; – Resources, Supervision; **Pavel Piler** – Resources, Data curation; **Holger M. Koch** – Validation, Writing – review & editing; **Martin Zvonař**; – Supervision, Funding acquisition, Resources; **Marta Esteban-López** – Supervision, Validation, Writing – review & editing; **Argelia Castaño** – Supervision, Validation, Writing – review & editing; **Jana Klánová**; – Resources, Funding acquisition, Supervision, Writing – review & editing.

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Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envres.2022.113675.

References

Adibi, J.J., Whyatt, R.M., Williams, P.L., Calafat, A.M., Camann, D., Herrick, R., Nelson, H., Bhat, H.K., Perera, F.P., Silva, M.J., Hauser, R., 2008. Characterization of phthalate exposure among pregnant women assessed by repeat air and urine samples. Environ. Health Perspect. 116, 467–473. https://doi.org/10.1289/ehp.10749.

Apel, P., Kortenkamp, A., Koch, H.M., Vogel, N., Rüther, M., Kasper-Sonnenberg, M., Conrad, A., Brüning, T., Kolossa-Gehring, M., 2020. Time course of phthalate cumulative risks to male developmental health over a 27-year period: biomonitoring samples of the German Environmental Specimen Bank. Environ. Int. 137, 105467 https://doi.org/10.1016/j.envint.2020.105467.

Bekö, G., Weschler, C.J., Langer, S., Callesen, M., Toftum, J., Clausen, G., 2013. Children's phthalate intakes and resultant cumulative exposures estimated from urine compared with estimates from dust ingestion, inhalation and dermal absorption in their homes and daycare centers. PLoS One 8. https://doi.org/10.1371/journal.pone.0062442.

Bui, T.T., Giovanoulis, G., Cousins, A.P., Magnér, J., Cousins, I.T., de Wit, C.A., 2016. Human exposure, hazard and risk of alternative plasticizers to phthalate esters. Sci. Total Environ. 541, 451–467. https://doi.org/10.1016/j.scitotenv.2015.09.036.

Calafat, A.M., Koch, H.M., Swan, S.H., Hauser, R., Goldman, L.R., Lanphear, B.P., Longnecker, M.P., Rudel, R.A., Teitelbaum, S.L., Whyatt, R.M., Wolff, M.S., 2013. Misuse of blood serum to assess exposure to bisphenol A and phthalates. Breast Cancer Res. 15, 403. https://doi.org/10.1186/bcr3494.

Carrieri, M., Trevisan, A., Bartolucci, G.B., 2000. Adjustment to concentration-dilution of spot urine samples: correlation between specific gravity and creatinine. Int. Arch. Occup. Environ. Health 74, 63–67. https://doi.org/10.1007/s004200000190.

CDC, 2019. Fourth national report on human exposure to environmental chemicals. Updated Tables, january 2019, volume one. Fourth Natl. Rep. Hum. Expo. to Environ. Chem. 1–529.

Correia-Sá, L., Kasper-Sonnenberg, M., Pälmke, C., Schütze, A., Norberto, S., Calhau, C., Domingues, V.F., Koch, H.M., 2018. Obesity or diet? Levels and determinants of phthalate body burden - a case study on Portuguese children. Int. J. Hyg Environ. Health 221, 519–530. https://doi.org/10.1016/j.ijheh.2018.02.001.

Correia-Sá, L., Schütze, A., Norberto, S., Calhau, C., Domingues, V.F., Koch, H.M., 2017. Exposure of Portuguese children to the novel non-phthalate plasticizer di-(iso-nonyl)-cyclohexane-1,2-dicarboxylate (DINCH). Environ. Int. 102, 79–86. https://doi.org/10.1016/j.envint.2017.02.001.

Den Hond, E., Govarts, E., Willems, H., Smolders, R., Casteleyn, L., Kolossa-Gehring, M., Schwedler, G., Seiwert, M., Fiddicke, U., Castaño, A., Esteban, M., Angerer, J., Koch, H.M., Schindler, B.K., Sepai, O., Exley, K., Bloemen, L., Horvat, M., Knudsen, L.E., Joas, A., Joas, R., Biot, P., Aerts, D., Koppen, G., Katsonouri, A., Hadjipanayis, A., Krskova, A., Maly, M., Mørck, T.A., Rudnai, P., Kozepesy, S., Mulcahy, M., Mannion, R., Gutleb, A.C., Fischer, M.E., Ligocka, D., Jakubowski, M., Reis, M.F., Namorado, S., Gurzau, A.E., Lupsa, I.-R., Halzlova, K., Jajcaj, M., Mazej, D., Tratnik, J.S., López, A., Lopez, E., Berglund, M., Larsson, K., Lehmann, A., Crettaz, P., Schoeters, G., 2015. First steps toward harmonized human biomonitoring in Europe: demonstration project to perform human biomonitoring on a European scale. Environ. Health Perspect. 123, 255–263. https://doi.org/10.1289/ehp.1408616.

Domínguez-Romero, E., Scheringer, M., 2019. A review of phthalate pharmacokinetics in human and rat: what factors drive phthalate distribution and partitioning? Drug Metab. Rev. 51, 314–329. https://doi.org/10.1080/03602532.2019.1620762.

Elbers, I., Mol, H., 2019a. ICI/EQUAS REPORT; Phthalates/round_02 (2018)- Phthalate Biomarkers in Urine.

Elbers, I., Mol, H., 2019b. ICI/EQUAS REPORT; DINCH/round_02 (2018)- DINCH Biomarkers in Urine.

Esteban López, M., Göen, T., Mol, H., Nübler, S., Haji-Abbas-Zarrabi, K., Koch, H.M., Kasper-Sonnenberg, M., Dvorakova, D., Hajslova, J., Antignac, J.-P.P., Vaccher, V., Elbers, I., Thomsen, C., Vorkamp, K., Pedraza – Díaz, S., Kolossa-Gehring, M., Castaño, A., 2021. The European human biomonitoring platform - design and implementation of a laboratory quality assurance/quality control (QA/QC) programme for selected priority chemicals. Int. J. Hyg Environ. Health 234, 113740. https://doi.org/10.1016/j.ijheh.2021.113740.

Frederiksen, H., Jorgensen, N., Andersson, A.-M., 2010. Correlations between phthalate metabolites in urine, serum, and seminal plasma from young Danish men determined by Isotope dilution liquid chromatography tandem mass spectrometry. J. Anal. Toxicol. 34, 400–410. https://doi.org/10.1093/jat/34.7.400.

Frederiksen, H., Nielsen, O., Koch, H.M., Skakkebaek, N.E., Juul, A., Jørgensen, N., Andersson, A.-M., 2020. Changes in urinary excretion of phthalates, phthalate substitutes, bisphenols and other polychlorinated and phenolic substances in young Danish men; 2009–2017. Int. J. Hyg Environ. Health 223, 93–105. https://doi.org/10.1016/j.ijheh.2019.10.002.

Fréry, N., Santonen, T., Porras, S.P., Fucic, A., Leso, V., Bousoumah, R., Duca, R.C., El Yamani, M., Kolossa-Gehring, M., Ndaw, S., Viegas, S., Iavicoli, I., 2020. Biomonitoring of occupational exposure to phthalates: a systematic review. Int. J. Hyg Environ. Health 229, 113548. https://doi.org/10.1016/j.ijheh.2020.113548.

Giovanoulis, G., Bui, T., Xu, F., Papadopoulou, E., Padilla-Sánchez, J.A., Covaci, A., Haug, L.S., Cousins, A.P., Magnér, J., Cousins, I.T., de Wit, C.A., 2018. Multi-pathway human exposure assessment of phthalate esters and DINCH. Environ. Int. 112, 115–126. https://doi.org/10.1016/j.envint.2017.12.016.

Gries, W., Ellrich, D., Küpper, K., Ladermann, B., Leng, G., 2012. Analytical method for the sensitive determination of major di-(2-propylheptyl)-phthalate metabolites in human urine. J. Chromatogr. B 908, 128–136. https://doi.org/10.1016/j.jchromb.2012.09.019.

Guo, Y., Kannan, K., 2013. A Survey of phthalates and parabens in personal care products from the United States and its implications for human exposure. Environ. Sci. Technol. Just Accep.

Gyllenhammar, I., Glynn, A., Jönsson, B.A.G., Lindh, C.H., Darnerud, P.O., Svensson, K., Lignell, S., 2017. Diverging temporal trends of human exposure to bisphenols and plastizisers, such as phthalates, caused by substitution of legacy EDCs? Environ. Res. 153, 48–54. https://doi.org/10.1016/j.envres.2016.11.012.

Health Canada, 2019. Fifth Report on Human Biomonitoring of Environmentla Chemicals in Canada.

Heudorf, U., Mersch-Sundermann, V., Angerer, J., 2007. Phthalates: toxicology and exposure. Int. J. Hyg Environ. Health 210, 623–634. https://doi.org/10.1016/j.ijheh.2007.07.011.

Hlisníková, H., Petrovičová, I., Kolena, B., Šidlovská, M., Sirotkin, A., 2020. Effects and mechanisms of phthalates' action on reproductive processes and reproductive health: a literature review. Int. J. Environ. Res. Publ. Health 17, 6811. https://doi.org/10.3390/ijerph17186811.

Husøy, T., Andreassen, M., Hjertholm, H., Carlsen, M.H., Norberg, N., Sprong, C., Papadopoulou, E., Sakhi, A.K., Sabaredzovic, A., Dirven, H.A.A.M., 2019. The Norwegian biomonitoring study from the EU project EuroMix: levels of phenols and phthalates in 24-hour urine samples and exposure sources from food and personal care products. Environ. Int. 132, 105103 https://doi.org/10.1016/j.envint.2019.105103.

Janjua, N.R., Frederiksen, H., Skakkebæk, N.E., Wulf, H.C., Andersson, A.-M., 2008. Urinary excretion of phthalates and paraben after repeated whole-body topical application in humans. Int. J. Androl. 118–129. https://doi.org/10.1111/j.1365-2605.2007.00841.x.

Kasper-Sonnenberg, M., Koch, H.M., Apel, P., Rüther, M., Pälmke, C., Brüning, T., Kolossa-Gehring, M., 2019. Time trend of exposure to the phthalate plasticizer substitute DINCH in Germany from 1999 to 2017: biomonitoring data on young adults from the Environmental Specimen Bank (ESB). Int. J. Hyg Environ. Health 222, 1084–1092. https://doi.org/10.1016/j.ijheh.2019.07.011.

Katsikantami, I., Sifakis, S., Tzatzarakis, M.N., Vakonaki, E., Kalantzi, O.-I., Tsatsakis, A. M., Rizos, A.K., 2016. A global assessment of phthalates burden and related links to health effects. Environ. Int. 97, 212–236. https://doi.org/10.1016/j.envint.2016.09.013.

Koch, H.M., Angerer, J., 2012. Chapter 3A phthalates: biomarkers and human biomonitoring. In: Biomarkers and Human Biomonitoring, vol. 1. The Royal Society of Chemistry, pp. 179–233. https://doi.org/10.1039/9781849733373-00179.

Koch, H.M., Calafat, A.M., 2009. Human body burdens of chemicals used in plastic manufacture. Philos. Trans. R. Soc. B Biol. Sci. 364, 2063–2078. https://doi.org/10.1098/rstb.2008.0208.

Koch, H.M., Rüther, M., Schütze, A., Conrad, A., Pälmke, C., Apel, P., Brüning, T., Kolossa-Gehring, M., 2017. Phthalate metabolites in 24-h urine samples of the German Environmental Specimen Bank (ESB) from 1988 to 2015 and a comparison with US NHANES data from 1999 to 2012. Int. J. Hyg Environ. Health 220, 130–141. https://doi.org/10.1016/j.ijheh.2016.11.003.

Koch, H.M., Schütze, A., Pälmke, C., Angerer, J., Brüning, T., 2013. Metabolism of the plasticizer and phthalate substitute diisononyl-cyclohexane-1,2-dicarboxylate (DINCH®) in humans after single oral doses. Arch. Toxicol. 87, 799–806. https://doi.org/10.1007/s00204-012-0990-4.

Koppen, G., Govarts, E., Vanermen, G., Voorspoels, S., Govindan, M., Dewolf, M.-C., Den Hond, E., Biot, P., Casteleyn, L., Kolossa-Gehring, M., Schwedler, G., Angerer, J., Koch, H.M., Schindler, B.K., Castaño, A., López, M.E., Sepai, O., Exley, K., Bloemen, L., Knudsen, L.E., Joas, R., Joas, A., Schoeters, G., Covaci, A., 2019. Mothers and children are related, even in exposure to chemicals present in common consumer products. Environ. Res. 175, 297–307. https://doi.org/10.1016/j.envres.2019.05.023.

Langer, S., Bekö, G., Weschler, C.J., Brive, L.M., Toftum, J., Callesen, M., Clausen, G., 2014. Phthalate metabolites in urine samples from Danish children and correlations with phthalates in dust samples from their homes and daycare centers. Int. J. Hyg Environ. Health 217, 78–87. https://doi.org/10.1016/j.ijheh.2013.03.014.

Lemke, N., Murawski, A., Lange, R., Weber, T., Apel, P., Dębiak, M., Koch, H.M., Kolossa-Gehring, M., 2021. Substitutes mimic the exposure behaviour of REACH regulated phthalates – a review of the German HBM system on the example of plasticizers. Int. J. Hyg Environ. Health 236, 113780. https://doi.org/10.1016/j.ijheh.2021.113780.

Lessmann, F., Kolossa-Gehring, M., Apel, P., Rüther, M., Pälmke, C., Harth, V., Brüning, T., Koch, H.M., 2019. German Environmental Specimen Bank: 24-hour urine samples from 1999 to 2017 reveal rapid increase in exposure to the para-phthalate plasticizer di(2-ethylhexyl) terephthalate (DEHTP). Environ. Int. 132, 105102 https://doi.org/10.1016/j.envint.2019.105102.

Lorber, M., Weschler, C.J., Morrison, G., Bekö, G., Gong, M., Koch, H.M., Salthammer, T., Schripp, T., Toftum, J., Clausen, G., 2017. Linking a dermal permeation and an inhalation model to a simple pharmacokinetic model to study airborne exposure to di(n-butyl) phthalate. J. Expo. Sci. Environ. Epidemiol. 27, 601–609. https://doi.org/10.1038/jes.2016.48.

Martínez, M.A., Rovira, J., Prasad Sharma, R., Nadal, M., Schuhmacher, M., Kumar, V., 2018. Comparing dietary and non-dietary source contribution of BPA and DEHP to prenatal exposure: a Catalonia (Spain) case study. Environ. Res. 166, 25–34. https://doi.org/10.1016/j.envres.2018.05.008.

Mendelsohn, E., Hagopian, A., Hoffman, K., Butt, C.M., Lorenzo, A., Congleton, J., Webster, T.F., Stapleton, H.M., 2016. Nail polish as a source of exposure to triphenyl phosphate. Environ. Int. 86, 45–51. https://doi.org/10.1016/j.envint.2015.10.005.

Miller, R.C., Brindle, E., Holman, D.J., Shofer, J., Klein, N.A., Soules, M.R., O'Connor, K. A., 2004. Comparison of specific gravity and creatinine for normalizing urinary reproductive hormone concentrations. Clin. Chem. 50, 924–932. https://doi.org/10.1373/clinchem.2004.032292.

Nassan, F.L., Coull, B.A., Gaskins, A.J., Williams, M.A., Skakkebaek, N.E., Ford, J.B., Ye, X., Calafat, A.M., Braun, J.M., Hauser, R., 2017. Personal care product use in men and urinary concentrations of select phthalate metabolites and parabens: results from the environment and reproductive health (EARTH) study. Environ. Health Perspect. 125, 1–10. https://doi.org/10.1289/EHP1374.

Pearson, M.A., Lu, C., Schmotzer, B.J., Waller, L.A., Riederer, A.M., 2009. Evaluation of physiological measures for correcting variation in urinary output: implications for assessing environmental chemical exposure in children. J. Expo. Sci. Environ. Epidemiol. 19, 336–342. https://doi.org/10.1038/jes.2008.48.

Piler, P., Kandrnal, V., Kukla, L., Andrýsková, L., Švancara, J., Jarkovský, J., Dušek, L., Pikhart, H., Bobák, M., Klánová, J., 2017. Cohort profile: the European longitudinal study of pregnancy and childhood (ELSPAC) in the Czech republic. Int. J. Epidemiol. 46 https://doi.org/10.1093/ije/dyw091, 1379-1379F.

Pilka, T., Petrovicova, I., Kolena, B., Zatko, T., Trnovec, T., 2015. Relationship between variation of seasonal temperature and extent of occupational exposure to phthalates. Environ. Sci. Pollut. Res. Int. 22, 434–440. https://doi.org/10.1007/s11356-014-3385-7.

Puklová, V., Janoš, T., Sochorová, L., Vavrouš, A., Vrbík, K., Fialová, A., Hanzlíková, L., Černá, M., 2019. Exposure to mixed phthalates in Czech preschool and school children. Arch. Environ. Contam. Toxicol. 77, 471–479. https://doi.org/10.1007/s00244-019-00645-6.

REACH, 2006. REGULATION (EC) No 1907/2006 OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 18 December 2006 [WWW Document].

Runkel, A.A., Snoj-Tratnik, J., Mazej, D., Horvat, M., 2020. Urinary phthalate concentrations in the slovenian population: an attempt to exposure assessment of family units. Environ. Res. 186, 109548 https://doi.org/10.1016/j.envres.2020.109548.

Schneider, U., Schober, E.A., Streich, N.A., Breusch, S.J., 2002. Urinary creatinine instability falsely increases the deoxypyridinoline/creatinine quotient. Clin. Chim. Acta 324, 81–88. https://doi.org/10.1016/S0009-8981(02)00209-7.

Schütze, A., Otter, R., Modick, H., Langsch, A., Brüning, T., Koch, H.M., 2017. Additional oxidized and alkyl chain breakdown metabolites of the plasticizer DINCH in urine after oral dosage to human volunteers. Arch. Toxicol. 91, 179–188. https://doi.org/10.1007/s00204-016-1688-9.

Schwedler, G., Conrad, A., Rucic, E., Koch, H.M., Leng, G., Schulz, C., Schmied-Tobies, M.I.H., Kolossa-Gehring, M., 2020a. Hexamoll® DINCH and DPHP metabolites in urine of children and adolescents in Germany. Human biomonitoring results of the German Environmental Survey GerES V, 2014–2017. Int. J. Hyg Environ. Health 229, 113397. https://doi.org/10.1016/j.ijheh.2019.09.004.

Schwedler, G., Rucic, E., Lange, R., Conrad, A., Koch, H.M., Pälmke, C., Brüning, T., Schulz, C., Schmied-Tobies, M.I.H., Daniels, A., Kolossa-Gehring, M., 2020b. Phthalate metabolites in urine of children and adolescents in Germany. Human biomonitoring results of the German Environmental Survey GerES V, 2014–2017. Int. J. Hyg Environ. Health 225, 113444. https://doi.org/10.1016/j.ijheh.2019.113444.

Silano, V., Barat Baviera, J.M., Bolognesi, C., Chesson, A., Cocconcelli, P.S., Crebelli, R., Gott, D.M., Grob, K., Lampi, E., Mortensen, A., Rivière, G., Steffensen, I.L., Tlustos, C., Van Loveren, H., Vernis, L., Zorn, H., Cravedi, J.P., Fortes, C., Tavares Poças, M. de F., Waalkens-Berendsen, I., Wölfle, D., Arcella, D., Cascio, C., Castoldi, A.F., Volk, K., Castle, L., 2019. Update of the risk assessment of di-butylphthalate (DBP), butyl-benzyl-phthalate (BBP), bis(2-ethylhexyl)phthalate (DEHP), di-isononylphthalate (DINP) and di-isodecylphthalate (DIDP) for use in food contact materials. EFSA J. 17 https://doi.org/10.2903/j.efsa.2019.5838.

Silva, M.J., Barr, D.B., Reidy, J.A., Kato, K., Malek, N.A., Hodge, C.C., Hurtz, D., Calafat, A.M., Needham, L.L., Brock, J.W., 2003. Glucuronidation patterns of common urinary and serum monoester phthalate metabolites. Arch. Toxicol. 77, 561–567. https://doi.org/10.1007/s00204-003-0486-3.

Søeborg, T., Frederiksen, H., Andersson, A.-M., 2012. Cumulative risk assessment of phthalate exposure of Danish children and adolescents using the hazard index approach. Int. J. Androl. 35, 245–252. https://doi.org/10.1111/j.1365-2605.2011.01240.x.

Tait, S., Carli, F., Busani, L., Buzzigoli, E., Della Latta, V., Deodati, A., Fabbrizi, E., Gaggini, M., Maranghi, F., Tassinari, R., Toffol, G., Cianfarani, S., Gastaldelli, A., La Rocca, C., 2020. Biomonitoring of Bis(2-ethylhexyl)phthalate (DEHP) in Italian children and adolescents: data from LIFE PERSUADED project. Environ. Res. 185, 109428 https://doi.org/10.1016/j.envres.2020.109428.

The European parliament, The Council of the European Union, 2009. REGULATION (EC) No 1223/2009 of the EUROPEAN PARLIAMENT and of the COUNCIL of 30 November 2009 on Cosmetic Products.

Vasiljevic, T., Su, K., Harner, T., 2021. A first look at atmospheric concentrations and temporal trends of phthalates in distinct urban sectors of the Greater Toronto Area. Atmos. Pollut. Res. 12, 173–182. https://doi.org/10.1016/j.apr.2020.10.019.

Wang, B., Tang, C., Wang, H., Zhou, W., Chen, Y., Zhou, Y., Jiang, Q., 2015. Influence of body mass index status on urinary creatinine and specific gravity for epidemiological study of children. Eur. J. Pediatr. 174, 1481–1489. https://doi.org/10.1007/s00431-015-2558-9.

Wang, Y., Zhu, H., Kannan, K., 2019. A review of biomonitoring of phthalate exposures. Toxics 7, 1–28. https://doi.org/10.3390/TOXICS7020021.

Who, 2013. Endocrine Disrupting Chemicals 2012. Summary for Decision-Makers. https://doi.org/10.1002/9781118346747.ch1.

Wittassek, M., Koch, H.M., Angerer, J., Brüning, T., 2011. Assessing exposure to phthalates - the human biomonitoring approach. Mol. Nutr. Food Res. 55, 7–31. https://doi.org/10.1002/mnfr.201000121.

Wormuth, M., Scheringer, M., Vollenweider, M., Hungerbühler, K., 2006. What are the sources of exposure to eight frequently used phthalic acid esters in Europeans? Risk Anal. 26, 803–824. https://doi.org/10.1111/j.1539-6924.2006.00770.x.

Young, A.S., Allen, J.G., Kim, U.J., Seller, S., Webster, T.F., Kannan, K., Ceballos, D.M., 2018. Phthalate and organophosphate plasticizers in nail polish: evaluation of labels and ingredients. Environ. Sci. Technol. 52, 12841–12850. https://doi.org/10.1021/acs.est.8b04495.

APPENDIX 6

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Parabens and antimicrobial compounds in conventional and “green” personal care products

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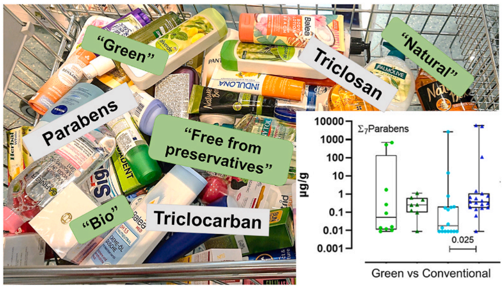
HIGHLIGHTS

- Parabens and antimicrobials were detected in 83% of personal care products.
- No difference in paraben levels between green and conventional products.
- Methylparaben was typically present at the highest concentrations.
- Estimated exclusive use of green cosmetics lowered cumulative paraben exposure.

GRAPHICAL ABSTRACT



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ABSTRACT

The personal care product (PCP) industry is a worldwide multi-billion-dollar industry. Several synthetic compounds like parabens and antimicrobial agents triclosan (TCS) and triclocarban (TCC) are ingredients in many PCPs. Due to growing public awareness of potential risks associated with parabens and other synthetic compounds, more PCPs are being marketed as “green,” “alternative,” or “natural.” We analyzed 19 green and 34 conventional PCP products obtained from a European store for seven parabens, TCC, and TCS. We found no statistically significant difference in the concentrations between green and conventional products. Only four products mentioned parabens in the list of ingredients; however, parabens were detected in 43 products, and at µg/g levels in seven PCPs. Methylparaben was typically present at the highest concentration, and one mascara exceeded the European legal concentration limit of methylparaben. Low concentrations of isopropyl-, isobutyl-, and benzylparabens, which are banned in the EU, were detected in 70% of PCPs. The cumulative estimated daily intake of parabens is an order of magnitude higher for people using only conventional products than those using green products exclusively. We propose that legislation be developed with more explicit rules on when a product can be advertised as “green” to aid consumers’ choices.

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1. Introduction

Personal care products (PCPs) have been used since ancient times for cosmetic and hygienic purposes. The global value of PCPs is estimated to reach $716.6 billion by 2025 (Grand View Research, 2018). Consumer awareness of potentially hazardous compounds in PCPs has increased (Zollo et al., 2021). As a result, global market demand for products that promote wellness, healthy aging, and environmental safety has also increased (Ghazali et al., 2017). Certain PCPs are advertised as "green" products, supposedly containing fewer and less harmful synthetic compounds than conventional products. By using these products, consumers are allegedly exposed to fewer harmful compounds (Harley et al., 2016). The most effective way for manufacturers to communicate the green status to potential consumers is by indicating it on the label. Words like "natural," "organic," or "without artificial preservatives" are used on product labels to persuade buyers to purchase the product. However, these terms are largely unregulated, with few parameters and no standardization (Dodson et al., 2012; Rubin and Brod, 2019). While some producers are truthful, others have employed green labels as a marketing strategy without significantly reducing the concentrations of harmful compounds in their products (Jog and Singhal, 2019). This strategy is known as "greenwashing" (Urbański & ul Haque, 2020). One group of compounds that are often prioritized for elimination or reduction in green products are parabens.

Parabens, esters of *p*-hydroxybenzoic acid, are low-cost, broad-spectrum antimicrobial, and antiseptic preservatives (Guo and Kannan, 2013; Li et al., 2021). They have been used widely in PCPs since the 1920s (Ana and Paula, 2016). Although parabens typically have short half-lives and rapid excretion rates in mammalian systems (Janjua et al., 2007), they have been quantified in human bodily fluids such urine, breast milk, and plasma in ng/ml concentration levels (Honda et al., 2018; Dualde et al., 2020; Sosvorova et al., 2017). The continuous presence of parabens in various bodily fluids suggests that these compounds are pseudo-persistent in the human body. Studies have found that the most prevalent route of human exposure to parabens is dermal exposure (Guo and Kannan, 2013; Liao et al., 2013). Parabens are known endocrine disrupting compounds with androgenic and estrogenic effects on humans (Darbre and Harvey, 2008). Long-chained parabens, such as butylparaben (BuP) and propylparaben (PrP), have higher endocrine disruptive potency than short-chained compounds such as methylparaben (MeP) and ethylparaben (EtP) (Oishi, 2001). Parabens with a branched structure, such as isobutyl- and isopropylparabens (iso-BuP and iso-PrP), are the most estrogenic paraben congeners (Darbre et al., 2002, 2003). Studies have linked topical application of PCPs containing parabens with the onset of breast cancer, obesity, gestational diabetes mellitus, and reduced sperm quality (Anderson, 1995; Darbre et al., 2004; Kolatorova et al., 2018; Li et al., 2019). Therefore, the European Union (EU) limited short-chained parabens to 0.4% w/w of a total product and long-chained parabens to 0.19% w/w in PCPs (Commission regulation (EU) no. 1004/2014). Isopropyl-, isobutyl-, phenyl-, benzyl-, and pentylparabens have been banned from use in PCPs in Europe since 2014 (Commission regulation (EU) no. 1004/2014)).

Triclosan (TCS) and triclocarban (TCC) are also popular additives to PCP formulations, particularly to products that hold antimicrobial and antibacterial properties (Nowak et al., 2021). TCC and TCS are known to suppress thyroid hormone production in rats and are bioaccumulative and persistent in the environment (Halden et al., 2017). More than 200 scientists and medical experts have signed the Florence Statement on Triclosan and Triclocarban in 2016, stating that potential health and environmental hazards caused by these compounds outweigh their benefits thereof (Halden et al., 2017).

The endocrine disrupting properties of PCP compounds such as parabens and TCS have been receiving increased attention. They are among the compounds included in the target list of several projects of the Horizon 2020 European Cluster to Improve Identification of Endocrine Disruptors (EURION cluster: https://eurion-cluster.eu/). Their possible associations with metabolic diseases (Kolatorova et al., 2018; Han et al., 2021; Reimann et al., 2021), for instance, have received attention in the OBERON project; a collaborative inter-European research project examining the exposures and effects of endocrine disrupting chemicals associated with metabolic disorders (https://oberon-4eu.com/). To support the understanding of adverse health effects of these compounds, current knowledge on human exposures is needed, incorporating current levels of chemical use in products as well as the consumer trends driving exposures.

The definition of a "cosmetic product" is made on a case-by-case assessment, but all products used in this study are accepted as cosmetics by Regulation (EC) no. 1223/2009 of the European Parliament and of the Council (Regulation (EC) no. 1223/2009). However, following the accepted scientific terminology, we will refer to the products as "personal care products" throughout the manuscript.

This study compares the concentrations of parabens and antimicrobial agents in PCP products marketed as a) green products and b) conventional PCPs purchased from a popular European retail store. We hypothesize that green products contain lower concentrations of parabens and other antimicrobial compounds than conventional products. We also present a determination of cumulative estimated daily intake (EDI) of parabens that consumers may be exposed to by using multiple green and conventional products.

2. Materials and methods

2.1. Sample collection

Fifty-three individual PCPs were purchased from a popular drug store (part of a multinational chain) in Brno, Czech Republic, in August 2019 (Table 1). All items are commonly used types of PCPs according to product use surveys (Wu et al., 2010). For the sake of anonymity, the store and products are not mentioned by name or brand. Green-labeled brands were identified by one of the specific marketing terms (Table 2) printed on the product packaging. After purchasing, all products were stored at room temperature until sample preparation. Samples were opened directly before sample processing.

No permits or ethical requirements were necessary to collect or analyze the samples.

2.2. Extraction and cleanup

Depending on the sample matrix, two different extraction methods were used due to the inherent variability in the composition of different PCPs. Liquid and semi-liquid products that were expected to contain low-lipid content, namely nail polish, liquid deodorant, micellar water,

Table 1
Classification and number of PCPs used for analyses.

Product	Type	Green products	Conventional products
Shampoo	Rinse-off	3	3
Conditioner	Rinse-off	2	2
Sunscreen	Leave-on	1	3
Toothpaste	Rinse-off	1	3
Shower gel	Rinse-off	3	3
Hand soap	Rinse-off	0	2
Hand cream	Leave-on	1	2
Face cream	Leave-on	0	2
Body lotion	Leave-on	2	1
Lipstick	Leave-on	1	3
Deodorant	Leave-on	2	2
Face wash	Rinse-off	1	2
Micellar water	Rinse-off	0	1
Nail polish	Leave-on	0	4
Mascara	Leave-on	1	2
	TOTAL	18	35

Table 2

Marketing terms on PCP labels that qualified the product as “green” for selection purposes.

Inclusion criteria	
Paraben free/free from parabens	Natural cosmetic
Silicone free/without silicone	Without artificial preservatives
Without artificial colorants	Without artificial scents
Bio	Certified natural cosmetics
100/90% natural origin	Without phthalates

and mascara, were analyzed using an adaptation of a sample preparation method described by Young et al. (2018). The original extraction method used by Young et al. (2018) consisted of liquid-liquid extraction of phthalates and organophosphates with methanol and acetone:ethyl acetate (1:1 v/v); we optimized it for our purposes to liquid-liquid extraction only with methanol. Samples that were highly viscous, solid, or semi-solid matrix and a medium to high-lipid content were prepared for analyses using a modified quick, easy, cheap, effective, rugged, and safe (QuEChERS) method. All samples and blanks were spiked with 100 μl of internal standard (IS; Supplementary Table S1) mix containing ^{13}C12- mass-labeled parabens (1 μg/ml) and with 100 μl of mass-labeled triclosan-^{13}C12 (1 μg/ml in isooctane) prior to extraction. A complete summary of both extraction methods is presented in the Supplementary Material.

Before instrumental analysis, samples were diluted with methanol to ensure the levels were within the calibration curve range. Low-lipid samples were diluted at 1:2; medium-lipid and turbid samples were diluted at 1:10, and high-lipid samples were diluted at 1:20. Although nail polish and mascara were considered low-lipid products, some samples exhibited turbidity until the end of the extraction and cleanup phase and were also diluted at 1:20.

2.3. Instrumental analyses

Samples were quantified for seven parabens (MeP, EtP, PrP, iso-PrP, BuP, iso-BuP, and BenzylP) and two antimicrobials (TCS and TCC). Analysis was conducted on a liquid chromatograph (Agilent 1290 Infinity II; Santa Clara, CA, USA) equipped with a vacuum degasser, binary pump, autosampler and column thermostat coupled to a mass spectrometer (ESI/QqQ Triple Quad 610; Santa Clara, CA, USA), and using Mass Hunter software. A Phenomenex Synergi Fusion C-18 end-capped (3 μm) 100 × 2.1 mm i.d. column was equipped with a Phenomenex SecureGuard C18 guard column (Phenomenex, Torrance, CA, USA). The mobile phase consisted of 5 mM ammonium acetate in water (A) and 5 mM ammonium acetate in methanol (B). The binary pump gradient increased from 30% to 40% B at 3 min, 50% B at 10 min, and finally to 100% B at 13 min, holding to 35 min, with 1 min column equilibration at initial conditions (30% B) with a flow rate of 0.12 mL/min 10 μL of the individual sample was injected for the analyses. Compounds were ionized with electrospray ionization. Ions were detected in the negative mode, and the ionization parameters were as follows: gas temperature 280 °C, gas flow 10 l/min, nebulizer 45 psi, capillary voltage 3.5 kV. Masses were quantified according to the relative response of corresponding internal standards.

2.4. Quality assurance and quality control

The analytical method was evaluated using replicates of six samples: a methanol solvent as a non-matrix blank sample, and matrix-spiked shampoo and face cream to evaluate the QuEChERS method, and a methanol solvent, and matrix-spiked mascara and face cream to evaluate the Young et al. (2018) extraction method. The spike-recovery samples were fortified at two levels (spike low and spike high) with native standards (Table S2) and analyzed as per the PCP samples. The relative recoveries for the native standards in the non-matrix spiked blanks were average 100% (range 92–115%) for parabens and 89% (64–112%) for TCS and TCC (Supplemental Figure S1). The low concentration matrix spiked sample recoveries were average 140% (range 83–406%) for parabens, and 126% (90–225%) for TCS and TCC, with the high recoveries from the spiked mascara sample (Figure S2). The high concentration matrix spiked sample recoveries were average 109% (range 95–138%) for parabens and 98% (73–139%) for TCS and TCC (Supplemental Figure S2).

Instrument calibration curves ranged from 0.1 ng/ml – 250 ng/ml with $R^2 > 0.99$ for all compounds. The researchers conducting laboratory analyses did not use any PCPs on the days that analyses or extractions were performed to limit external contamination and procedural blanks, consisting of methanol solvents spiked with IS were used to track contamination throughout the laboratory analyses. Two blanks were prepared for each extraction method and analyzed in parallel with each batch of samples. Ten blanks were processed with the QuEChERs method and two blanks with the Young et al. (2018) method. The concentrations of target analytes in blanks are presented in Table S3.

Procedural blanks were used to determine the method limit of detection (LOD). The LOD was set to equal the average of the blanks +3 times the standard deviation of the blanks. If a compound was not detected in any procedural blank, the LOD was set to equal the instrumental detection limit (Table 3). Sample values above LOD were adjusted to account for blank contamination by subtracting the average of the procedural blanks.

2.5. Statistical analyses and grouping

Graphpad Prism 8.0.2 (www.graphpad.com) was used for summary statistics. The products were grouped according to “green” or “conventional” classification, based on the marketing on the label. According to their mode of application, these groups were subdivided into “leave-on” and “rinse-off” products. We used Mann-Whitney U-tests to compare differences between the sample groups. Significance was set as $p < 0.05$. For the statistical analysis, values < LOD were substituted with LOD/2.

2.6. Estimated daily intake and hazard quotient

To contextualize the concentrations of parabens that humans are theoretically exposed to on a daily basis, the following formula of estimated daily intake (EDI) as used by Nakata et al. (2015) was adapted to determine estimated paraben exposure from PCPs in a European country.

$$\Sigma_i \frac{C_i \cdot E_i \cdot N_i \cdot A \cdot F_i}{BW}$$

C_i Concentration in product (μg/g);
E_i application quantities (g/time);
N_i application frequency (time per day applied);
A absorption factor through the skin;
F_i retention factor;
BW bodyweight.

Li et al. (2021) determined that the dermal absorption factor of parabens is 0.4. The retention factor for leave-on products is 1, and for rinse-off products, it is 0.01 (McGinty et al., 2011). The average adult body mass in the Czech Republic is 91.9 kg for men and 74.2 kg for women (WorldData, 2021). The application quantities of the selected PCPs are taken from Bremmer et al. (2006), and the application frequency of several PCPs is taken from questionnaire data from 300 Czech adults as a part of the CELSPAC: Young Adults study (Supplementary Figure S3; Fišerová et al. submitted). CELSPAC: Young Adults is an on-going follow-up study of the longitudinal ELSPAC study in the Brno region of the Czech Republic (Piler et al., 2017) and follows ELSPAC children born in 1991 and 1992, their siblings, and spouses. The CELSPAC: Young Adults study was approved by the ELSPAC Ethics Committee (Ref. No: ELSPAC/EK/2/2019, dated 13.03.2019). This cohort's

Table 3

Concentrations (µg/g) of parabens, triclosan, and triclocarban in green and conventional personal care products from a European store. LOD values are blank-based values, with the exception of BuP and iso-BuP, which are instrument detection limits. Where the quantified concentrations were lower than LOD values, the cell is marked as "<LOD", and the LOD/2 value used to calculate Σparabens is given in the bottom row.

	Product	**MeP**	**EtP**	**PrP**	**iso-PrP**	**BuP**	**iso-BuP**	**BenzylP**	**Σparabens**	**TCC**	**TCS**
Green rinse-off	Shampoo	<LOD	<LOD	0.0023	0.0031	<LOD	<LOD	<LOD	**0.012**	0.0029	<LOD
	Shampoo	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	0.007	**0.016**	<LOD	<LOD
	Shampoo	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	**0.0089**	<LOD	<LOD
	Conditioner	526	0.077	0.023	0.018	0.07	0.071	0.032	**526**	0.055	0.17
	Conditioner	0.042	<LOD	<LOD	<LOD	<LOD	<LOD	0.22	**0.26**	<LOD	<LOD
	Toothpaste	505	0.20	92	82	0.021	0.037	0.025	**679**	0.036	0.059
	Shower gel	1.1	<LOD	0.30	0.3	<LOD	<LOD	<LOD	**1.7**	0.0054	<LOD
	Shower gel	<LOD	<LOD	0.0023	<LOD	<LOD	<LOD	<LOD	**0.01**	<LOD	<LOD
	Shower gel	<LOD	0.0065	<LOD	<LOD	<LOD	<LOD	<LOD	**0.014**	<LOD	<LOD
	Face wash gel	<LOD	0.022	0.014	0.0098	0.012	0.016	0.01	**0.088**	0.024	0.09
Green leave-on	Sunscreen	0.44	0.073	0.32	0.22	0.024	0.0084	0.028	**1.1**	0.065	0.23
	Body lotion	0.13	0.031	0.01	0.014	0.023	0.018	0.02	**0.25**	0.056	0.5
	Body lotion	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	**0.0089**	<LOD	0.022
	Hand cream	0.431	0.067	0.037	0.021	0.02	0.033	0.0072	**0.62**	0.033	0.083
	Lipstick	0.25	0.094	0.02	0.032	0.034	0.038	0.016	**0.48**	0.028	0.15
	Deodorant	0.088	0.0047	<LOD	<LOD	<LOD	<LOD	<LOD	**0.095**	<LOD	<LOD
	Deodorant	0.11	0.047	0.023	0.0089	0.021	0.025	0.016	**0.25**	0.086	0.06
	Mascara	0.028	0.043	0.0086	0.0044	0.0071	0.02	0.0093	**0.12**	0.019	0.15
Conventional rinseoff	Shampoo	6.1	8.5	<LOD	<LOD	<LOD	<LOD	<LOD	**14.6**	<LOD	<LOD
	Shampoo	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	**0.0089**	<LOD	<LOD
	Shampoo	0.041	<LOD	<LOD	<LOD	<LOD	<LOD	0.14	**0.19**	<LOD	<LOD
	Conditioner	2670	0.12	0.034	0.025	0.019	0.013	<LOD	**2670**	0.019	0.6
	Conditioner	0.3	0.24	0.043	0.072	0.045	0.062	0.023	**0.78**	0.093	0.16
	Toothpaste	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	**0.0089**	<LOD	<LOD
	Toothpaste	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	**0.0089**	<LOD	<LOD
	Toothpaste	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	**0.0089**	<LOD	<LOD
	Shower gel	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	**0.0089**	<LOD	<LOD
	Shower gel	0.031	0.035	0.012	0.0098	0.027	0.019	0.0055	**0.14**	0.028	0.24
	Shower gel	0.0047	0.015	<LOD	<LOD	<LOD	<LOD	<LOD	**0.022**	<LOD	<LOD
	Face wash	0.067	0.032	0.019	0.038	0.024	0.01	0.011	**0.2**	0.057	0.1
	Face wash	0.0095	<LOD	<LOD	<LOD	<LOD	<LOD	0.00063	**0.014**	<LOD	0.05
	Micellar water	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	**0.0089**	<LOD	<LOD
	Hand soap	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	**0.0089**	<LOD	<LOD
	Hand soap	0.028	0.016	0.031	0.023	0.029	0.029	0.017	**0.17**	0.029	0.11
Conventional leave-on	Sunscreen	108	0.01	0.017	0.015	0.012	0.012	0.04	**108**	0.051	0.24
	Sunscreen	0.075	0.038	0.017	0.012	0.013	0.017	0.0086	**0.18**	0.019	0.13
	Sunscreen	0.30	0.19	0.016	0.021	0.047	0.054	0.029	**0.66**	0.065	0.26
	Body lotion	0.12	0.039	0.024	0.016	0.028	0.014	0.025	**0.27**	0.041	0.44
	Hand cream	0.067	0.07	0.018	0.027	0.048	0.055	0.021	**0.31**	0.05	0.25
	Hand cream	0.031	0.058	0.013	0.019	0.012	0.027	0.006	**0.17**	0.038	0.12
	Lipstick	0.023	0.038	<LOD	<LOD	<LOD	<LOD	<LOD	**0.064**	<LOD	0.28
	Lipstick	0.14	0.31	0.04	0.021	0.024	0.024	0.0081	**0.57**	0.022	0.33
	Lipstick	0.24	0.15	0.45	0.5	0.046	0.032	0.024	**1.4**	0.044	0.11
	Deodorant	0.15	0.077	0.05	0.0069	0.067	0.057	0.013	**0.42**	0.016	0.26
	Deodorant	0.076	0.046	0.012	0.0099	0.013	0.021	0.009	**0.19**	0.05	0.089
	Mascara	4522	0.15	464	393	223	386	0.026	**5987**	0.038	<LOD
	Mascara	0.16	0.076	0.038	0.022	0.019	0.011	0.0084	**0.33**	0.014	0.12
	Nail Polish	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	<LOD	**0.0089**	<LOD	<LOD
	Nail Polish	0.22	0.021	0.05	0.023	0.016	0.032	0.0038	**0.37**	0.018	0.42
	Nail Polish	0.63	0.063	0.068	0.075	0.093	0.099	0.0055	**1.04**	0.011	0.22
	Nail Polish	0.44	0.045	0.089	0.03	0.054	0.09	0.023	**0.77**	0.0091	0.049
	Face cream	0.064	0.012	0.017	0.012	0.019	0.014	0.0073	**0.14**	0.037	0.19
	Face cream	3615	1810	0.04	0.042	0.031	0.029	0.014	**5425**	0.037	0.15
	LOD/2	**0.0047**	**0.0016**	**0.0011**	**0.0013**	**0.000008**	**0.000003**	**0.00024**		**0.00065**	**0.00045**

🟥 >10000 µg/g 🟧 100-1000 µg/g 🟨 10-100 µg/g 🟩 1-10 µg/g 🟩 0.1-1 µg/g 🟦 0.01-0.1 µg/g 🟦 <0.001 µg/g minimum and maximum application frequency were used to estimate two paraben exposure scenarios for both green and conventional products.

3. Results and discussion

3.1. Difference between green and conventional PCPs and product labels

We tested 35 conventional and 18 green products for nine compounds of interest. The concentrations of the target compounds in the selected PCPs ranged several orders of magnitude — from <0.0001 μg/g to >1000 μg/g (Table 3). Most of the quantifiable concentrations of parabens were in the 0.01–0.1 μg/g range. TCS was present at five times higher concentrations than TCC.

Parabens were detected in 89% of green products and 77% of conventional products, while antimicrobial compounds had less frequent detection (66% of green products and 69% of conventional products). One green and eight conventional products did not contain parabens or antimicrobial compounds above the LOD. Methylparaben was found at the highest concentrations in PCPs (Table 3), with a maximum measured concentration of 4522 μg/g MeP in mascara. Similar to what was found in other studies (Dodson et al., 2012; Matwiejczuk et al., 2020), MeP had the highest detection frequency (38 of 53 products; 72%), followed by EtP (36 of 53 products; 67%). Detectable concentrations of EtP were found in a green shower gel that was labeled as "free from parabens".

It was unexpected that conventional rinse-off products had the lowest detection frequency of parabens (Table 4); however this is primarily because no parabens were present above detectable concentrations in any of the four conventional toothpaste brands (Table 3).

In all product categories, for both parabens and TCS, concentrations varied several orders of magnitude, even within the same specific group of products, e.g., conditioners (Table 3). The measured concentrations did not support our hypothesis that green products should have lower levels of parabens, TCS and TCC; we found no significant difference between the concentrations in green PCPs and the corresponding conventional PCPs for any of the target compounds. The clearer differences were not between green/conventional products but rather between the leave-on and rinse-off products across green and conventional product categories. The conventional leave-on products had significantly higher median concentrations of $\Sigma_7$parabens than conventional rinse-off products (Mann-Whitney *U* test, $p = 0.025$, Fig. 1). Similarly, for TCS, the conventional leave-on products had statistically significantly higher TCS concentrations than the conventional rinse-off products ($p = 0.0072$, Fig. 1). However, due to the small sample size employed by this study, the results should be interpreted with caution.

3.2. Legislation regarding product composition and labelling

Current legislation states that the maximum concentration of parabens in PCP products is to be 0.8% of the ready for use preparation (Commission regulation (EU) no. 1004/2014). A distinction is made between short- and long-chained parabens because of the higher endocrine disruptive potential of long-chained parabens (Oishi, 2001). One conventional mascara sample exceeded these guidelines, where 0.452% (4521 μg/g) of the ready for use preparation consisted of MeP. However, waterproof mascara is a "borderline" or "extreme cosmetic" product. Borderline products are regulated on a case-by-case basis according to the *Manual on the Scope of Application of the Cosmetic Regulation*, which can suggest whether or not a product falls within the scope of the Regulation (EC) no. 1223/2009 (Lores et al., 2018). 80% of the extreme cosmetic products analyzed by Lores et al. (2018) exceeded concentrations of various compounds as prescribed by the Regulation (EC) no. 1223/2009.

Table 4
The detection frequencies (df) of $\Sigma_7$parabens and TCS in different groups of PCPs. Different product groups are green rinse-off (GRO), green leave-on (GLO), conventional rinse-off (CRO), and conventional leave-on (CLO).

	GRO (n = 10)	GLO (n = 8)	CRO (n = 16)	CLO (n = 19)
Σ_7Parabens df	90%	88%	56%	95%
TCS df	30%	88%	38%	89%

Although the EU bans isopropyl-, isobutyl-, and benzylparabens (Commission regulation (EU) no. 1004/2014), quantifiable concentrations of these compounds were found in several products, including green and conventional mascaras, and conventional face wash gel. The highest concentration of a banned paraben was 81 μg/g isopropylparaben in the green toothpaste sample, labeled as "herbal gel for gums." The maximum concentration of TCS in a PCP is 0.3% of the total ready for use preparation and 1.5% for TCC (Regulation (EC) no. 1223/2009). No products had concentrations that were close to 0.3% or 1.5% of the product composition for either compound.

Even though parabens were quantified in 81% of the PCP samples in this study, only four products included parabens as an ingredient on the label: three conventional products (conditioner, face cream, and mascara) and one green toothpaste. Most of the products that included parabens in the ingredient list contained more than 400 μg/g of a compound. MeP was included as an ingredient in all four products, PrP in two, and EtP and BuP in one product each. One green conditioner contained 525 μg/g MeP without including MeP in the ingredient list. Neither TCC nor TCS was ever included in the ingredient lists, even in antimicrobial soap. According to the European Union cosmetics regulation (no. 1223/2009), compounds need not to be included in the ingredient list if they are present as impurities in raw materials or a subsidiary of technical materials used in the mixture but are not present in the final product (Regulation (EC) no. 1223/2009). Because most of the concentrations of parabens and other antimicrobial compounds were below 0.1 μg/g, they might have been considered impurities by the production companies or incidental ingredients from the processing and packaging procedure. Cosmetic packaging is not as strictly regulated as food packaging, which increases the chances of accidental inclusion of harmful substances from packaging materials in personal care products (Regulation (EC) no. 1272/2008).

3.3. Estimated daily intake and health risk of parabens through green and conventional PCPs

The daily use of PCPs is not restricted to a single product. Multiple PCPs are used daily to serve different cosmetic and hygienic purposes (Fisher et al., 2017). The application frequency of PCPs also differs between different groups of people. According to Czech questionnaire data (see section 2.6; Figure S3), the response group with the highest application frequency of PCPs consisted exclusively of female respondents. The group with the lowest frequency was exclusively male; as a result, we separate our exposure assessments by gender (Figure S3). EDIs were calculated for male and female exposure to parabens using only the green and conventional products sampled during this study. The cumulative EDI was calculated to determine the cumulative effect of using multiple products (see section 2.6; Table 5). No information on toothpaste, conditioner, and shower gel was included in the questionnaire. These products (marked with an asterisk in Table 5) were included in the cumulative EDI calculation under the assumption that toothpaste and shower gel is used once a day (Bremmer et al., 2006) and that conditioner is used at the same application frequency as shampoo.

Even though only a snapshot of products were included in the study, a clear pattern could be seen with exposures differing by gender and selection of green vs. conventional products due to women's higher use of PCPs (Figure S3 and Table 5). The cumulative EDI of green products was an order of magnitude lower than conventional products for both men and women (Table 4). We also note that EDI is substantially higher for women than men due to their higher use of PCPs and lower body weight.

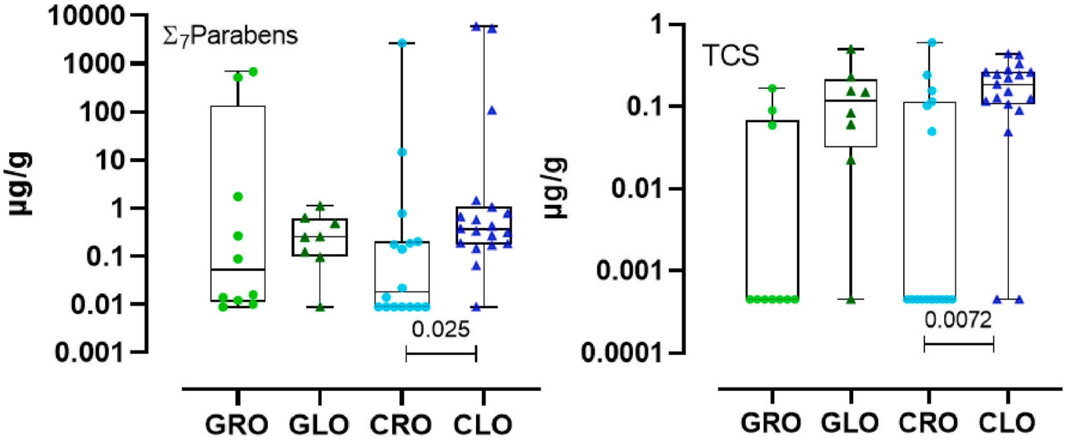

Fig. 1. Box-and-whisker plot (horizontal lines are medians, 95% confidence intervals, minima, and maxima) of the sum of parabens detected in all product groups. Different product groups are green rinse-off (GRO), green leave-on (GLO), conventional rinse-off (CRO), and conventional leave-on (CLO). The *p*-values of significant differences between product groups are indicated.

Table 5
Estimated daily intake (EDI) of parabens through conventional personal care products for men and women in the Czech Republic. An asterisk (*) indicates products that are included in the EDI calculation, but not included in the questionnaire of Figure S3.

	Product	Σparaben median	Application quantity[a]	Application frequency[b]	EDI
		(µg/g)	(g/application)	(application/day)	(µg/kg-bw/day)
Green products Female use	Shampoo	0.012	12	0.7	0.000005
	Conditioner*	263	14	0.7	0.14
	Lotion	0.25	10.5	0.9	0.013
	Deodorant	0.17	0.5	1	0.00046
	Mascara	0.12	0.025	0.8	0.00001
	Toothpaste*	679	0.08	1	0.03
	Shower gel*	0.014	8.7	1	0.00007
			Cumulative EDI		**0.18**
Conventional products Female use	Shampoo	0.19	12	0.7	0.00009
	Conditioner*	1335	14	0.7	0.71
	Lotion	0.27	10.5	0.9	0.0014
	Deodorant	0.3	0.5	1	0.0008
	Mascara	2994	0.025	0.8	0.32
	Toothpaste*	0.0089	0.08	1	0.00000004
	Shower gel*	0.022	8.7	1	0.00001
			Cumulative EDI		**1.03**
Green products Male use	Shampoo	0.012	12	0.3	0.000002
	Conditioner*	263	14	0.3	0.048
	Lotion	0.25	10.5	0.1	0.0011
	Deodorant	0.17	0.5	0.5	0.00018
	Mascara	0.12	0.025	0	0
	Toothpaste*	679	0.08	1	0.0024
	Shower gel*	0.014	8.7	1	0.000005
			Cumulative EDI		**0.052**
Conventional products Male use	Shampoo	0.19	12	0.3	0.00003
	Conditioner*	1335	14	0.3	0.24
	Lotion	0.27	10.5	0.1	0.00013
	Deodorant	0.3	0.5	0.5	0.00033
	Mascara	2994	0.025	0	0
	Toothpaste*	0.0089	0.08	1	0.00000003
	Shower gel*	0.022	8.7	1	0.000008
			Cumulative EDI		**0.24**

The cumulative EDI values were determined by summing the individual EDIs from each category. The EDIs of parabens from the individual PCPs are very low (<1 µg/kg-bw/day). The highest EDI was 0.71 µg/kg-bw/day for conventional conditioners used by women.

According to the theoretical calculation, a reduction of 82% in paraben EDI can be expected if only green products are used *in lieu* of conventional products. This contrasts with the statistical comparisons between green and conventional products (Section 3.1), where there were no statistically significant differences in paraben concentrations between the product groups. It suggests that even if the concentrations of parabens in some individual green products do not significantly differ from conventional products, the cumulative effect of reduced paraben exposure may be significant if a consumer commits to a lifestyle of using green products exclusively.

Even though the cumulative EDIs for this study were found to be relatively low and no health risks were expected, it should be noted that the cumulative EDI only accounted for seven products. Typically, people use more PCPs on a regular basis, and will have other exposure sources such as diet and pharmaceuticals (Błędzka et al., 2014; Liao et al., 2013).

3.4. Consumer attitude to green products

Due to increased public awareness of certain chemicals compounds' potential health and environmental effects, many consumers gravitate towards alternative or green products (Dodson et al., 2020). In 2019, the sale of natural cosmetics brands accounted for $ 1.6 billion in sales in the United States (NPD Group, 2018). Hence, many companies produce and promote green cosmetic products to retain a competitive advantage in the market (Luo et al., 2020).

The practice of paraben avoidance is proven to be implementable and effective. A study by Dodson et al. (2020) revealed that cosmetic users are willing to avoid potentially hazardous compounds if they have prior knowledge. The participants that actively avoided products with harmful compounds by studying product ingredient lists had lower urinary concentrations of parabens than people who did not practice avoidance (Dodson et al., 2020). Similarly, a study by Harley et al. (2016) revealed that MeP and PrP in the urine of adolescent girls decreased by 43.9% after using products labeled as "paraben-free" for three days (researchers screened these products beforehand to confirm that they were paraben-free).

However, since most consumers are not familiar with technical terms or chemical nomenclature, ingredient lists are not ideal communicators, even if the producers are transparent (Marć and Martyn, 2019). Many consumers rely on social media to choose PCPs, leading to skewed perspectives (Luo et al., 2020). Consumers also fear "greenwashed" products, where producers intentionally label a product as green, but no effort has been put into making the product green (Urbański and ul Haque, 2020). The greatest danger of green products is the fact that the term "green," "bio-," "natural," and other similar labels are haphazardly communicated and currently unregulated, with no standards in place to determine ingredient content or concentrations of chemical compounds (Dodson et al., 2012; Rubin and Brod, 2019). The label "paraben-free" is not officially registered in the EU, and producers are under no legal obligation to remove all parabens from the products (Nowak et al., 2021).

4. Conclusion

The PCP industry is a big international business. Many synthetic compounds are included in the production of PCPs, including potential endocrine disrupting compounds such as parabens, TCS, and TCC. Due to growing public awareness of potential health risks of parabens and other synthetic compounds, more PCPs are being produced and marketed as "green," "alternative," or "natural." However, due to vague regulations regarding green products, it is difficult to determine whether products marketed as green contain lower concentrations of harmful chemical. After analyzing various green and conventional PCP products from a European store for seven parabens, TCC, and TCS, we found that there was no significant difference in the paraben concentrations between the product types, thus rejecting our initial proposed hypothesis. Methylparaben was the compound present at the highest concentration and detection frequency among all target analytes. Only four products included parabens in the list of ingredients. Except for one mascara, which can be considered a borderline or extreme cosmetic product, the concentrations of all compounds were below the maximum concentrations set by the EU Regulations. It was concerning that four banned paraben compounds were quantifiable in the majority of the PCP samples. The cumulative EDI for parabens was an order of magnitude higher for conventional products than for green PCPs. Even though the EDIs for individual products were very low, the cumulative effect indicates that consumers could be exposed to fewer parabens using predominantly green PCPs in their daily routine. The potential health risks of cumulated PCP use warrant further investigation. The lack of set parameters on product labels limits the ability of consumers to make informed decisions on product purchases and may pose a risk for the consumer".

Credit author statement

Veronica van der Schyff: drafting the manuscript, data interpretation; **Lenka Suchánková:** Acquisition of data, data interpretation, manuscript revision; **Katerina Kademoglou:** Acquisition of data, data interpretation, manuscript revision, study design; **Lisa Melymuk:** Study design, data interpretation, manuscript revision, Supervision; **Jana Klánová:** Supervision, Funding acquisition, manuscript revision.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.chemosphere.2022.134019.

References

Ana, F., Paula, F.A., 2016. Parabens paradoxes in cosmetic formulations: a review. Int. J. Med. Res. Pharmaceut. Sci. 8, 1–11.

Anderson, F.A., 1995. Final report on the safety assessment of isobutylparaben and isopropylparaben. J. Am. Coll. Toxicol. 14, 364–372.

Błędzka, D., Gromadzińska, J., Wąsowicz, W., 2014. Parabens. From environmental studies to human health. Environ. Int. 67, 27–42.

Bremmer, H.J., Prud'Homme de Lodder, L.C.H., Van Engelen, J.G.M., 2006. Cosmetics Fact Sheet. To Assess the Risks for the Consumer. Updated version for ConsExpo 4. *RIVM rapport 320104001*. https://www.rivm.nl/bibliotheek/rapporten/320104001.pdf. (Accessed 16 April 2021). Date of access.

Commission Regulation (EU) no. 1004/2014 of 18 September 2014 amending Annex V to Regulation (EC) No 1223/2009 of the European Parliament and of the Council on cosmetic products. Off. J. Eur. Union 282/5. https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32014R1004&from=EN. Date accessed: 28 April 2021.

Darbre, P.D., Harvey, P.W., 2008. Paraben esters: review of recent studies of endocrine toxicity, absorption, esterase and human exposure, and discussion of potential human health risks. J. Appl. Toxicol. 28, 561–578.

Darbre, P.D., Aljarrah, A., Miller, W.R., Coldham, N.G., Sauer, M.J., Pope, G.S., 2004. Concentrations of parabens in human breast tumours. J. Appl. Toxicol.: Int. J. 24, 5–13.

Darbre, P.D., Byford, J.R., Shaw, L.E., Hall, S., Coldham, N.G., Pope, G.S., Sauer, M.J., 2003. Oestrogenic activity of benzylparaben. J. Appl. Toxicol.: Int. J. 23, 43–51.

Darbre, P.D., Byford, J.R., Shaw, L.E., Horton, R.A., Pope, G.S., Sauer, M.J., 2002. Oestrogenic activity of isobutylparaben in vitro and in vivo. J. Appl. Toxicol.: Int. J. 22, 219–226.

Dodson, R.E., Boronow, K.E., Susmann, H., Udesky, J.O., Rodgers, K.M., Weller, D., Woudneh, M., Brody, J.G., Rudel, R.A., 2020. Consumer behavior and exposure to parabens, bisphenols, triclosan, dichlorophenols, and benzophenone-3: results from a crowdsourced biomonitoring study. Int. J. Hyg Environ. Health 230, 113624.

Dodson, R.E., Nishioka, M., Standley, L.J., Perovich, L.J., Brody, J.G., Rudel, R.A., 2012. Endocrine disruptors and asthma-associated chemicals in consumer products. Environ. Health Perspect. 120, 935–943.

Dualde, P., Pardo, O., Corpas-Burgos, F., Kuligowski, J., Gormaz, M., Vento, M., Pastor, A., Yusà, V., 2020. Biomonitoring of parabens in human milk and estimated daily intake for breastfed infants. Chemosphere 240, 124829.

Fišerová, P.S., Melymuk, L., Komprdová, K., Domínguez-Romero, E., Scheringer, M., Kohoutek, J., Přibylová, P., Andrýsková, L., Piler P. & Klánová, J. PCP Usage and Lifestyle Affects Phthalate and DINCH Metabolites Levels in Teenagers and Young Adults" Submitted to *Environmental Research.*

Fisher, M., MacPherson, S., Braun, J.M., Hauser, R., Walker, M., Feeley, M., Mallick, R., Bérubé, R., Arbuckle, T.E., 2017. Paraben concentrations in maternal urine and breast milk and its association with personal care product use. Environ. Sci. Technol. 51, 4009–4017.

Ghazali, E., Soon, P.C., Mutum, D.S., Nguyen, B., 2017. Health and cosmetics: investigating consumers' values for buying organic personal care products. J. Retailing Consum. Serv. 39, 154–163.

Grand View Research, 2018. Beauty & Personal Care Products Market Worth $716.6 Billion by 2025. Date of access. https://www.grandviewresearch.com/press-release/global-beauty-personal-care-products-market. (Accessed 15 March 2021).

Guo, Y., Kannan, K., 2013. A survey of phthalates and parabens in personal care products from the United States and its implication for human exposure. Environ. Sci. Technol. 47, 14442–14449.

Halden, R.U., Lindeman, A.E., Aiello, A.E., Andrews, D., Arnold, W.A., Fair, P., Fuoco, R. E., Geer, L.A., Johnson, P.I., Lohmann, R., McNeill, K., 2017. The Florence statement on triclosan and triclocarban. Environ. Health Perspect. 125, 064501.

Han, M., Wang, Y., Tang, C., Fang, H., Yang, D., Wu, J., Wang, H., Chen, Y., Jiang, Q., 2021. Association of triclosan and triclocarban in urine with obesity risk in Chinese school children. Environ. Int. 157, 106846.

Harley, K.G., Kogut, K., Madrigal, D.S., Cardenas, M., Vera, I.A., Meza-Alfaro, G., She, J., Gavin, Q., Zahedi, R., Bradman, A., Eskenazi, B., 2016. Reducing phthalate, paraben, and phenol exposure from personal care products in adolescent girls: findings from the HERMOSA intervention study. Environ. Health Perspect. 124, 1600–1607.

Honda, M., Robinson, M., Kannan, K., 2018. Parabens in human urine from several Asian countries, Greece, and the United States. Chemosphere 201, 13–19.

Janjua, N.R., Mortensen, G.K., Andersson, A.M., Kongshoj, B., Skakkebaek, N.E., Wulf, H. C., 2007. Systemic uptake of diethyl phthalate, dibutyl phthalate, and butyl paraben following whole-body topical application and reproductive and thyroid hormone levels in humans. Environ. Sci. Technol. 41, 5564–5570.

Jog, D., Singhal, D., 2019. Pseudo green players and their greenwashing practices: a differentiating strategy for real green firms of personal care category. Strat. Dir. 35, 4–7.

Kolatorova, L., Sramkova, M., Vitku, J., Vcelak, J., Lischkova, O., Starka, L., Duskova, M., 2018. Parabens and their relation to obesity. Physiol. Res. 67, S465–S472.

Li, C., Zhao, Y., Liu, S., Yang, D., Ma, H., Zhu, Z., Kang, L., Lu, S., 2021. Exposure of Chinese adult females to parabens from personal care products: estimation of intake via dermal contact and health risks. Environ. Pollut. 272, 116043.

Li, Y., Xu, S., Li, Y., Zhang, B., Huo, W., Zhu, Y., Wan, Y., Zheng, T., Zhou, A., Chen, Z., Huang, Z., 2019. Association between urinary parabens and gestational diabetes mellitus across prepregnancy body mass index categories. Environ. Res. 170, 151–159.

Liao, C., Liu, F., Kannan, K., 2013. Occurrence of and dietary exposure to parabens in foodstuffs from the United States. Environ. Sci. Technol. 47, 3918–3925.

Lores, M., Celeiro, M., Rubio, L., Llompart, M., Garcia-Jares, C., 2018. Extreme cosmetics and borderline products: an analytical-based survey of European regulation compliance. Anal. Bioanal. Chem. 410, 7085–7102.

Luo, B., Sun, Y., Shen, J., Xia, L., 2020. How does green advertising skepticism on social media affect consumer intention to purchase green products? J. Consum. Behav. 19, 371–381.

Marć, M.M., Martyn, J., 2019. Factors determining the level of knowledge about parabens in cosmetics. Eur. J. Clin. Exper. Med. 17, 236–241.

Matwiejczuk, N., Galicka, A., Brzóska, M.M., 2020. Review of the safety of application of cosmetic products containing parabens. J. Appl. Toxicol. 40, 176–210.

McGinty, D., Letizia, C.S., Api, A.M., 2011. Fragrance material review on ethylene brassylate. Food Chem. Toxicol. 49, S174–S182.

Nakata, H., Hinosaka, M., Yanagimoto, H., 2015. Macrocyclic-, polycyclic-, and nitro musks in cosmetics, household commodities and indoor dusts collected from Japan: implications for their human exposure. Ecotoxicol. Environ. Saf. 111, 248–255.

National Purchase Diary Panel (NPD Group), 2018. U.S. Prestige Beauty Industry Sales Grow 6 Percent in 2018, Reports the NPD Group. Date of access. https://www.npd.com/wps/portal/npd/us/news/press-releases/2019/u-s–prestige-beauty-industry-sales-grow-6-percent-in-2018–reports-the-npd-group/. (Accessed 15 April 2021).

Nowak, K., Jabłońska, E., Ratajczak-Wrona, W., 2021. Controversy around parabens: alternative strategies for preservative use in cosmetics and personal care products. Environ. Res. 198, 110488.

Oishi, S., 2001. Effects of butylparaben on the male reproductive system in rats. Toxicol. Ind. Health 17, 31–39.

Piler, P., Kandrnal, V., Kukla, L., Andrýsková, L., Švancara, J., Jarkovský, J., Dušek, L., Pikhart, H., Bobák, M., Klánová, J., 2017. Cohort profile: the european longitudinal study of pregnancy and childhood (ELSPAC) in the Czech Republic. Int. J. Epidemiol. 46, 1379, 1379f.

Regulation (EC) no. 1223/2009 of the European parliament and of the council of 30 November 2009 on cosmetic products. https://eur-lex.europa.eu/legal-content/EN/TXT/HTML/?uri=CELEX:32009R1223&from=EN. Date of access: 1 July 2021.

Regulation (EC) no. 1272/2008 — classification, labelling and packaging of substances and mixtures (CLP). https://osha.europa.eu/en/legislation/directives/regulation-ec-no-1272-2008-classification-labelling-and-packaging-of-substances-and-mixtures. Date of access: 22 July 2021.

Reimann, B., Vrijens, K., Roels, H.A., Wang, C., Cosemans, C., Van Overmeire, I., Nawrot, T.S., Plusquin, M., 2021. In utero exposure to parabens and early childhood BMI z-scores – associations between placental ethyl paraben, longitudinal BMI trajectories and cord blood metabolic biomarkers. Environ. Int. 157, 106845.

Rubin, C.B., Brod, B., 2019. Natural does not mean safe—the dirt on clean beauty products. JAMA Dermatol. 155, 1344–1345.

Sosvorova, L.K., Chlupacova, T., Vitku, J., Vlk, M., Heracek, J., Starka, L., Saman, D., Simkova, M., Hampl, R., 2017. Determination of selected bisphenols, parabens and estrogens in human plasma using LC-MS/MS. Talanta 174, 21–28.

Urbański, M., ul Haque, A., 2020. Are you environmentally conscious enough to differentiate between greenwashed and sustainable items? A global consumers perspective. Sustainability 12, 1786.

WorldData, 2021. Average Sizes of Men and Women. Date of access. https://www.worlddata.info/average-bodyheight.php#by-area. (Accessed 10 May 2021).

Wu, X., Bennett, D.H., Ritz, B., Cassady, D.L., Lee, K., Hertz-Picciotto, I., 2010. Usage pattern of personal care products in California households. Food Chem. Toxicol. 48, 3109–3119.

Young, A.S., Allen, J.G., Kim, U.J., Seller, S., Webster, T.F., Kannan, K., Ceballos, D.M., 2018. Phthalate and organophosphate plasticizers in nail polish: evaluation of labels and ingredients. Environ. Sci. Technol. 52, 12841–12850.

Zollo, L., Carranza, R., Faraoni, M., Díaz, E., Martín-Consuegra, D., 2021. What influences consumers' intention to purchase organic personal care products? The role of social reassurance. J. Retailing Consum. Serv. 60, 102432.

APPENDIX 7

Demirtepe, Hale, Lisa Melymuk, Miriam L. Diamond, Lola Bajard, Šimon Vojta, Roman Prokeš, Ondřej Sáňka, Jana Klánová, Ľubica Palkovičová Murínová, Denisa Richterová, Vladimíra Rašplová, Tomáš Trnovec. 2019. "Linking Past Uses of Legacy SVOCs with Today's Indoor Levels and Human Exposure." *Environment International* 127: 653-63. https://doi.org/10.1016/j.envint.2019.04.001

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Linking past uses of legacy SVOCs with today's indoor levels and human exposure

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ABSTRACT

Semivolatile organic compounds (SVOCs) emitted from consumer products, building materials, and indoor and outdoor activities can be highly persistent in indoor environments. Human exposure to and environmental contamination with polychlorinated biphenyls (PCBs) was previously reported in a region near a former PCB production facility in Slovakia. However, we found that the indoor residential PCB levels did not correlate with the distance from the facility. Rather, indoor levels in this region and those reported in the literature were related to the historic PCB use on a national scale and the inferred presence of primary sources of PCBs in the homes. Other SVOCs had levels linked with either the activities in the home, e.g., polycyclic aromatic hydrocarbons (PAHs) with wood heating; or outdoor activities, e.g., organochlorine pesticides (OCPs) with agricultural land use and building age. We propose a classification framework to prioritize SVOCs for monitoring in indoor environments and to evaluate risks from indoor SVOC exposures. Application of this framework to 88 measured SVOCs identified several PCB congeners (CB-11, -28, -52), hexachlorobenzene (HCB), benzo(a)pyrene, and γ-HCH as priority compounds based on high exposure and toxicity assessed by means of toxicity reference values (TRVs). Application of the framework to many emerging compounds such as novel flame retardants was not possible because of either no or outdated TRVs. Concurrent identification of seven SVOC groups in indoor environments provided information on their comparative levels and distributions, their sources, and informed our assessment of associated risks.

1. Introduction

Semivolatile organic compounds (SVOCs) found in indoor environments include flame retardants (FRs), plasticizers, pesticides, combustion by-products, those added to personal care products, and their degradation products (Lucattini et al., 2018). Concentrations in indoor dust can vary over orders of magnitude from pg/g to μg/g levels (Venier et al., 2016; Vykoukalová et al., 2017; Weschler and Nazaroff, 2008). Indoor SVOC levels are affected by emission rates from consumer products, building materials and indoor activities, rates of SVOC transport from outdoor to indoor, removal rates by ventilation and cleaning, and rates of depletion/degradation indoors (Weschler, 2009; Weschler and Nazaroff, 2008; Zhang et al., 2009). Since loss rates relative to outdoor environments are low, SVOCs persist indoors more than outdoors (Shin et al., 2013; Weschler, 2009). Combined with > 90% of time spent in indoor environments, the indoor environment becomes an important source of human exposure to SVOCs.

In the environmental literature, SVOCs first gained notoriety with compounds now identified as legacy persistent organic pollutants (POPs). Specifically polychlorinated biphenyls (PCBs), dichlorodiphenyltrichloroethane (DDT), and hexachlorobenzene (HCB) were among the original "Dirty Dozen" chemicals listed on the Stockholm Convention on POPs, which entered into force in 2004. Pentachlorobenzene (PeCB) and hexachlorocyclohexane isomers (HCHs) were added in 2009, followed by commercial mixtures of polybrominated diphenyl ethers (PBDEs) in 2009 and 2017, and hexabromocyclododecane (HBCDD) in 2013 (Stockholm Convention, 2018). Intense efforts to enact controls of Stockholm POPs followed

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clear evidence of their ubiquity in the environment and biota, including humans. Many studies have documented their outdoor concentrations, in part to gauge the success of controls on new uses of legacy POPs implemented through the Stockholm Convention (Gioia et al., 2006; Harner et al., 2004; Hung et al., 2016; Wang et al., 2007). Fewer studies have documented indoor contaminant levels, and generally they are limited to PBDEs, HBCDD and PCBs (Ali et al., 2012; Harrad et al., 2009b; Marek et al., 2017; Melymuk et al., 2016b; Tao et al., 2016).

Indoor levels of legacy POPs are controlled by primary indoor sources and continuing emissions (Audy et al., 2018; Frederiksen et al., 2012; Zhang et al., 2011). SVOCs such as PCBs and FRs were used in floor coverings, electrical equipment, upholstered furnishing, building materials and carpets, while organochloride pesticides (OCPs) were used for pest control indoors and outdoors (Lucattini et al., 2018). Polycyclic aromatic hydrocarbons (PAHs) are produced unintentionally during indoor combustion processes (e.g., cooking, heating, smoking) (Maertens et al., 2008). The presence of these compounds in the indoor environment results in increased human exposure. For example, inhalation of PCBs from indoor environments was found to be a significant exposure route in school children (Ampleman et al., 2015; Marek et al., 2017). Additionally, organophosphate FRs in house dust were shown to be significantly correlated with altered hormone levels and decreased semen quality in men (Meeker and Stapleton, 2010). To reduce exposures, we need to better understand indoor sources of SVOCs.

This study was motivated by high levels of PCBs observed in breast milk (Drobná et al., 2011) and serum samples (Hovander et al., 2006; Jursa et al., 2006; Petrik et al., 2006; Strémy et al., 2019; Wimmerová et al., 2015) from people living in and around Michalovce, Slovakia. The observed serum levels were two to three times higher than that of the general population of Slovakia (Hovander et al., 2006; Jursa et al., 2006; Petrik et al., 2006). These exposures have been attributed to a nearby PCB production facility (Strémy et al., 2019). The Chemko Strážske factory, which operated between 1959 and 1984, produced 21,000 t of PCBs – 1.6% of the total global PCB volume (Breivik et al., 2007). Production resulted in the contamination of surrounding soils, sediments, ambient air and surface waters (Hiller et al., 2011; Kocan et al., 2001). For example, 53,000 ppm PCB was observed in a soil sample at asphalt-gravel mix plant in the factory complex and ambient air levels around the factory were 10 to 20 times higher than control areas (Kocan et al., 2001).

Despite high PCB levels outdoors, indoor contamination in the region had not yet been investigated. We hypothesized that levels of PCBs in homes in the region would be similarly elevated due to contamination from the factory and outdoor-to-indoor transfer of pollution. Here we report PCB levels in air and dust in 60 homes in the region, and compare them with other legacy POPs such as OCPs and PBDEs, as well as other SVOCs such as current use FRs and PAHs. We also present a framework that enables evaluation of the potential risk posed by indoor levels of SVOCs in terms of both compound toxicity and estimated human exposure.

2. Materials and methods

2.1. Sampling

Samples were collected in 60 homes in Eastern Slovakia from March–April 2015 (Fig. S1). Homes were a subset of those participating in the PCB cohort (Hertz-Picciotto et al., 2003) established in the region in 2003. Homes were distributed evenly between urban and rural areas within 40 km of the Chemko Strážske factory, covering a 750 km^2 region. The construction years of the homes ranged from 1930 to 2000, with the mean and median of 1971. One air sample and one floor dust sample was collected per home in the child's bedroom.

Single-bowl polyurethane foam passive air samplers (PUF-PAS) were deployed for 32–35 days to measure indoor ambient air concentrations. PUFs were pre-cleaned for 8 h in acetone and 8 h in dichloromethane (DCM). Indoor PUF-PAS were previously calibrated for this set of compounds (Audy et al., 2018; Venier et al., 2016; Vykoukalová et al., 2017). A generic sampling rate of 1.6 m^3/day was applied, leading to each sampler capturing approximately 50 m^3 of air during the deployment period. After deployment, PUFs were removed from the PAS housing, packed in two layers of aluminum foil and stored in a portable freezer at −18 °C for transport to the laboratory.

Dust samples were collected at the end of the PUF-PAS deployment in the same room using a household vacuum cleaner with polyester sock inserts. Socks were precleaned via Soxhlet extraction in DCM for 8 h, and before sampling and between samples the vacuum nozzle and tube were cleaned with propan-2-ol. To collect each sample, a polyester sock was inserted into the front of the vacuum tube and held in place by the vacuum nozzle, and the accessible floor surface was vacuumed (typically between 1 and 3 m^2). The sock was removed from the vacuum cleaner, packed in two layers of aluminum foil, labeled and put into a zip-lock bag, stored in a portable freezer at −18 °C for transport to the laboratory.

2.2. Analysis

The target analytes were 10 PBDE congeners (28, 47, 66, 85, 99, 100, 153, 154, 183 and 209), 3 HBCDD isomers, 22 “novel” halogenated flame retardants (NFRs) (TBP-AE, α-DBE-DBCH, β-DBE-DBCH, TBX, TBP-BAE, α-TBCO, β-TBCO, PBBZ, TBCT, DDC-CO-MA, PBT, PBEB, TBP-DBPE, HBB, PBBA, DBHCTD, EH-TBB, BTBPE, TDBP-TAZTO, s-DDC-CO, a-DDC-CO, BEH-TEBP), 18 organophosphate esters (OPEs) (TPrP, TIBP, TNBP, TCEP, TCIPP, DBPP, TPeP, BDPP, TDCIPP, TBOEP, TPHP, EHDPP, TEHP, TOTP, TMTP, TPTP, TIPPP, TDMPP), 9 PCB congeners (9, 11, 28, 52, 101, 118, 153, 138, 180), 12 OCPs (PeCB, HCB, α-HCH, β-HCH, γ-HCH, δ-HCH, o,p'-DDE, p,p'-DDE, o,p'-DDD, p,p'-DDD, o,p'-DDT, p,p'-DDT) and 27 PAHs (ACY, ACE, FLU, PHE, ANT, FLA, PYR, BAA, CHR, BBF, BKF, BAP, IcdP, DBA, BGP, RET, benzo[*b*]fluorene, benzo-naptho-thiophene, benzo[*ghi*]fluoranthene, cyclopenta[*cd*]pyrene, triphenylene, benzo[*j*]fluoranthene, benzo[*e*]pyrene, PERY, dibenzo[*ac*]anthracene, antanthrene, coronene). Full compound names, CAS numbers, and further details are given in Table S1; abbreviations of FRs are adopted from Bergman et al. (2012).

All the analytical procedures used here were published previously (Jílková et al., 2018; Venier et al., 2016; Vojta et al., 2017; Vykoukalová et al., 2017) and are described in detail in the SI. PUFs were extracted with DCM using a Büchi B-811 automated warm Soxhlet extraction system. Before extraction, dusts were sieved with a 500 μm sieve to remove coarse particles (e.g., hair, large fibres). Approximately 100 mg of the sieved dust was used for extraction. Dust samples were extracted via three times repeated sonication in 1:1 *v*/v hexane:acetone. ^{13}C-labeled or deuterated internal standards (for PBDEs, HBCDD, PAHs, OPEs) and non-environmental PCB congeners (for PCBs and OCPs) (Wellington Laboratories Inc., Canada) were added before extraction.

Dust and PUF extracts were split 70:30. The 70% aliquot was purified with a H2SO$_4$-modified silica gel column and analyzed for PBDEs, HBCDD, PCBs and OCPs. The 30% aliquot was cleaned and fractionated using a 5 g activated silica column eluted with DCM, followed by 7:3 v/v acetone:DCM. The first fraction was used for analysis of PAHs and NFRs, and the second fraction for OPEs.

PBDEs and NFRs were analyzed using gas chromatography coupled to high resolution mass spectrometry (HRGC-MS), while PCBs, OCPs and OPEs were analyzed by gas chromatography-tandem mass spectrometry. PAHs were analyzed using gas chromatography–mass spectrometry. HBCDDs were analyzed after exchanging solvent to acetonitrile by liquid chromatography electrospray ionization mass spectrometry. See SI for further details on instrumental analysis.

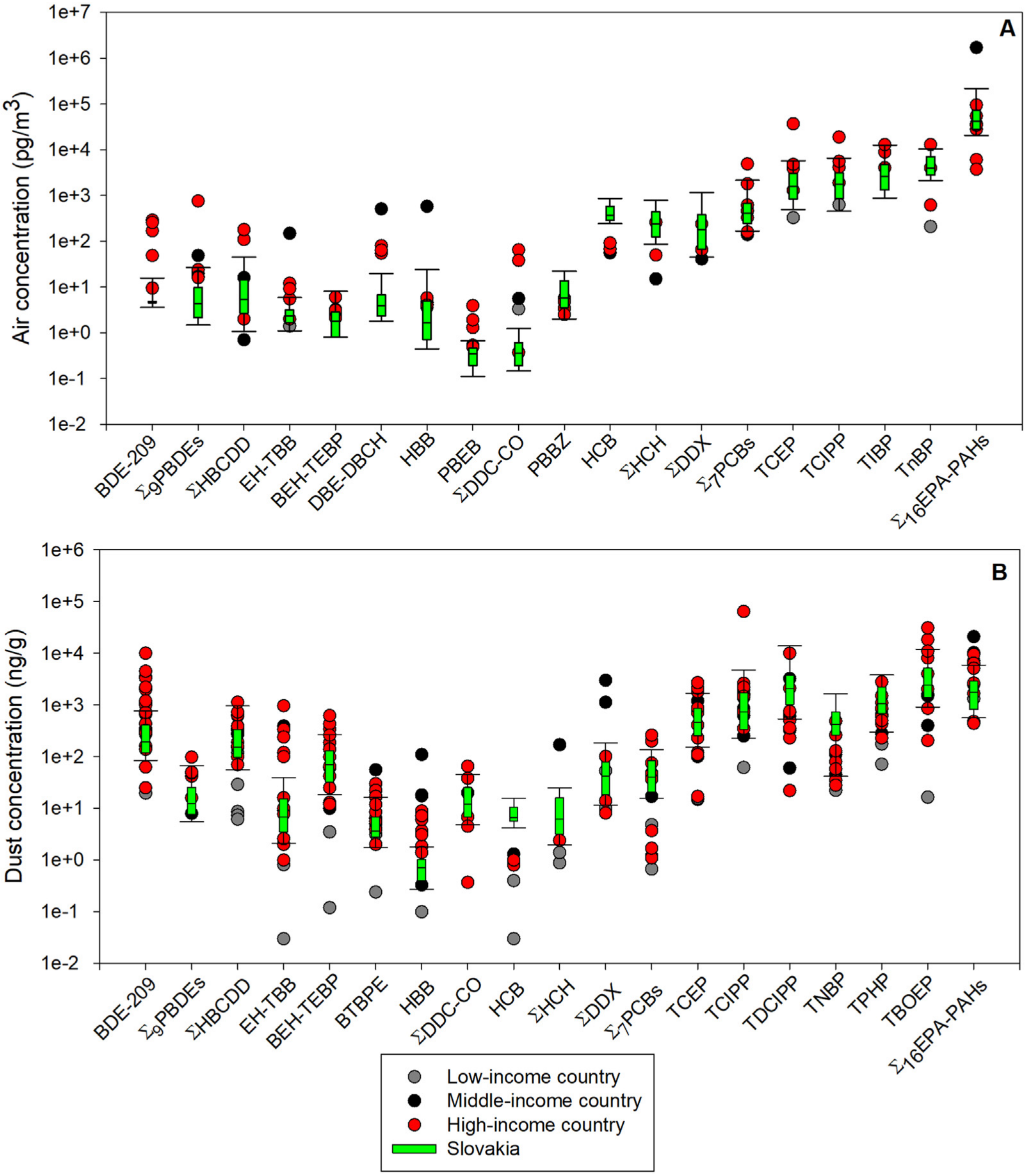

Fig. 1. Comparison of indoor concentrations for (A) air and (B) dust from Slovakia (green boxplots), and literature values from low-, middle-, and high-income countries classified based on criteria from the World Bank (2018a). Each dot represents a central tendency value (median or mean) for one country (see Tables S10–S25). For the Slovakia data, the boxes represent interquartile ranges, error bars below and above the box represent the 10th and 90th percentiles, respectively. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

2.3. QA/QC

The recoveries of individual compounds were determined using a set of pre-cleaned PUFs ($n = 8$) spiked with the native analytes prior to extraction. The recoveries were 70–120%. Sample masses were adjusted for recoveries based on the internal standards (for PCBs, OCPs, PAHs, and OPEs) or through the isotopic dilution instrumental method applied (PBDEs, NFRs, HBCDDs).

Field blanks (5 PUFs and 4 vacuum bags) were transported to the sampling site, manipulated as per the samples, and then treated as per the samples for the analytical process. Method detection limits (MDLs) were calculated as the average of the field blank plus three times the standard deviation of the blanks. The instrument detection limit (IDL) was used as the MDL for compounds that were not detected in the blanks (Tables S2-S7). The average of the blank of the corresponding matrix was subtracted from sample values that were > MDL, and values < MDL were recorded as such.

2.4. Data analysis

Statistical analyses were conducted using IBM SPSS 25. For statistical analysis, values below detection were substituted by $\sqrt{2}/2 \cdot$MDL. Non-parametric tests were applied for correlations (Spearman correlation, r_s) and comparison of differences between compounds (Mann Whitney U test) since the Shapiro-Wilk normality test showed non-normal distributions ($p < 0.001$) for dust and air data. Correlations and differences were considered significant if $p < 0.05$. Principal component analysis (PCA) and analysis of variance (ANOVA) were conducted after log-transforming the data.

3. Results and discussion

3.1. Compound concentrations and geographical comparison

The concentrations of all compounds and compound groups found in indoor dust and air from Slovakia are provided in Table S8. Median concentrations in dust ranged from 0.13 ng/g for CB-9 to 2450 ng/g for TBOEP. Generally, OPEs had the highest concentrations in dust, followed by PAHs. Median air concentrations ranged from 0.03 pg/m^3 for TBCT to 16,790 pg/m^3 for phenanthrene. PAHs, followed by OPEs, had the highest air concentrations.

We examined whether common profiles or sources for compound groups could be identified using PCA. PCA was able to explain 61.7% of the variability within the dust concentration dataset with three components. Component 1 included Σ_7PCBs, ΣHCH, ΣDDX and Σ_{27}PAH, component 2 involved Σ_9PBDEs, Σ_{14}NFRs, Σ_{14}OPEs and BDE-209, while CB-11, PeCB and HCB formed component 3. Hence, we can infer that for dust concentrations, PCBs and pesticides have similar profiles, i.e. many houses have high OCPs and high PCBs, while all flame retardants and plasticizers have similar profiles, suggesting similar sources. This clustering is consistent with differences in building age and equipment, with homes from before the restriction of PCBs and OCPs having higher levels of these compounds (Audy et al., 2018), further supported in our data analysis below (Section 3.3). Correlations among the compound groups in component 1 were also significant and strong (Table S9). Significant but relatively weaker correlations were found in component 2 between BDE-209 and Σ_9PBDEs ($r_s = 0.48$), Σ_{14}NFRs and Σ_9PBDEs ($r_s = 0.43$) and BDE-209 ($r_s = 0.27$), Σ_{14}OPEs and Σ_9PBDEs ($r_s = 0.47$) and BDE-209 ($r_s = 0.28$). These compounds all have similar uses as flame retardants. Correlations between compounds in component 3 were also significant: PeCB and HCB ($r_s = 0.73$), CB-11 and PeCB ($r_s = 0.58$), CB-11 and HCB ($r_s = 0.60$). For indoor air concentrations, PCA was able to explain 68.7% of the variability with five components, among which only component 1 constituted a meaningful cluster, containing ΣHCH, ΣDDX, HCB and Σ_7PCBs. Additionally, air concentrations of compound groups did not have significant and strong correlations with each other, except for a few pesticides, such as ΣHCH and ΣDDX ($r_s = 0.71$), HCB and ΣDDX ($r_s = 0.57$). Again, these groupings are consistent with the explanation related to building age.

We observed differences in indoor air and dust concentrations when qualitatively comparing our data with previously published indoor levels from other geographic regions (Fig. 1), supporting the findings of others (Dirtu et al., 2012; Harrad et al., 2009b; Venier et al., 2016; Vykoukalová et al., 2017). To understand the reasons for such differences, we selected the studies reporting residential indoor SVOC levels after the year 2000 regardless of the room type. For PCBs, we used the sum of 7 PCB indicator congeners (CB-28, -52, -101, -118, -153, -138, -180) and for PAHs the sum of 16 EPA-PAHs. Data for this comprehensive comparison is listed in Tables S10-S25. Slovakian levels fell within the range of those previously measured in other countries for most compounds. However, the Slovakia samples had notably higher levels of HCB in dust and air, and lower levels of PBDEs, DBE-DBCH and HBB in air compared to median levels from other countries (Fig. 1).

Next, we tested the hypothesis that country-level indoor concentrations of these legacy and current use SVOCs were related that country's economic status (categorized according to World Bank, 2018a). Typically, low-income countries (LIC) had lower concentrations of SVOCs than middle and high income countries (MIC and HIC) (Fig. 1). For example, the highest concentrations of FRs in dust were reported in HICs. In contrast, the higher levels of OCPs were reported in dust of MICs, especially when compared to HICs for which agriculture value had the lowest contribution to the gross domestic product (GDP) (World Bank, 2018b). The highest PAH levels in indoor dust and air were reported in China, attributed to residential coal combustion for cooking and heating (Lv et al., 2009; Wang et al., 2013).

For PCBs, we further hypothesized that indoor air and dust concentrations were related to historical PCB use per capita in a specific country. We explored this hypothesis by calculating PCB use per capita per country using the consumption estimates of Breivik et al. (2002). Seven PCB congeners were summed for the period of 1930 to 1984 for each country and divided by its population in 1970, reflecting the time of peak PCB use (Table S26; for Slovakia and Czech Republic, total use for Czechoslovakia and total population of two countries were used). When more than one median concentration for dust and air was reported for a country, we included all values as separate data points. We found a significant correlation between PCB use per capita and median PCB dust and air concentrations for 10 countries ($r_s = 0.59$ and 0.91, for dust and air, respectively, $p < 0.05$). We concluded that broad, national-level differences in current indoor residential PCB concentrations are related to the past national use of PCBs. It should be noted that the data from these studies are based on samples of convenience that were often of limited sample size, i.e. the number of homes varying between five and 60, which might limit the representativeness of the data for an entire country. Another limitation to this conclusion was that indoor air PCB concentrations were reported for only five countries of which most were HICs.

3.2. Polychlorinated biphenyls

The median concentration of Σ_9PCBs in dust from Slovakian homes was 58 ng/g (17.4–465 ng/g), and dust was dominated by CB-11 (30 %), -153 (20%) and -180 (13%). The median concentration of Σ_9PCBs in air was 1.09 ng/m^3 (0.28–6.92 ng/m^3), CB-11, -28 and -52 dominated indoor air. While CB-28, -52, -101, -118, -153, -138, -180 were the dominant legacy PCB congeners that were present in technical mixtures (e.g., Aroclor mixtures manufactured in USA and Delor mixtures manufactured at Chemko Strážske and used in the former Czechoslovakia), CB-11 was not present in any Delor mixtures (Taniyasu et al., 2003). Rather, CB-11 is produced unintentionally during the manufacturing of pigments, used in paints, paper products and textiles (Hu and Hornbuckle, 2010; Rodenburg et al., 2010; Vorkamp, 2016), and can be emitted from floor and cabinet surfaces (Herkert et al., 2018). Hence, the predominance of this non-legacy PCB congener in Slovakian indoor dust and air suggests the importance of modern indoor sources of PCBs, and suggests that attention should be given to a broader set of PCB congeners beyond the standard seven indicator PCBs representing only legacy PCB uses.

Correlations between PCB congeners in dust revealed that CB-9 and CB-11 correlated strongly only with CB-28, CB-52 and with each other ($r_s > 0.60$, $p < 0.001$), while the seven indicator PCBs correlated significantly with each other ($p < 0.05$), possibly related to their vapor pressure. On the other hand, indoor air concentrations of CB-9 and seven indicator PCBs correlated significantly ($p < 0.01$), whereas CB-11 did not correlate with any other PCB congeners, supporting the hypothesis of unique modern sources of CB-11 in indoor air (Hu and Hornbuckle, 2010; Rodenburg et al., 2010).

Next we tested the hypothesis that the main source of legacy PCB levels (Σ_7PCBs) to Slovakian homes was the nearby Chemko Strážske PCB production facility, where production ceased in 1984. We compared indoor levels with the distance of homes (between 1.7 and 41 km) from the Chemko Strážske PCB production facility. However, we found no significant correlation between legacy PCB levels in indoor dust or air and distance from the facility, suggesting that Chemko Strážske is not the major influence on current indoor PCB contamination in the homes. A previous study in the region, conducted 15 years after the cessation of PCB production, found that ambient outdoor air concentrations in residential areas < 5 km from the factory were up to 1700 pg/m^3 and levels decreased to regional background within 15 km (Kocan et al., 2001). Additionally, a recent study found that the elevated levels of PCBs in human serum samples from the region were associated with the factory, i.e. higher levels were detected in people living downwind from the factory when compared to other regions (Strémy et al., 2019). In our study, the home showing highest legacy PCB concentration was 6 km from the factory. However, we only sampled seven homes within 5 km from the factory, which may have limited our ability to see an effect from proximity to the factory. Rather, we found that indoor levels in this region were comparable to indoor levels in similar locations regardless of distance from the Chemko

Strážske plant, e.g., within the range of indoor levels measured in other HICs (Fig. 1).

In addition to the "macro-level" factor of country-wide PCB use, indoor legacy PCB levels can be influenced by building age and materials used (Hazrati and Harrad, 2006; Marek et al., 2017; Melymuk et al., 2016a). The homes sampled in this study were mainly made of brick ($n = 26$) and concrete ($n = 19$). The legacy PCB concentrations in indoor dust were significantly higher in brick than concrete homes (one-way ANOVA, $F = 5.67$, $df = 3$, $p < 0.05$, Fig. S2). This was contrary to our expectations given the association of PCBs with building sealants used in concrete buildings (Kohler et al., 2005; Liu et al., 2016). Other primary indoor PCB sources include paints, adhesives, wire and cable insulation manufactured before 1984, i.e. sources that cannot be easily removed and may be still present in homes. Thus, higher PCB concentrations in brick homes could be attributed to these PCB-containing materials and subsequent absorption into bricks (U.S. EPA, 2012), and the presence of materials serving as secondary sources of PCBs, such as window frames and wallpaper (Weis et al., 2003).

We also compared indoor PCB levels in homes built before and after the PCB ban in 1984 in Czechoslovakia (Holoubek et al., 2006). No significant difference was found in legacy PCB concentrations before and after the ban, however, there were only four homes built after 1984, limiting the power of this analysis.

3.3. Organochlorine pesticides

Eleven of 12 OCPs had > 50% detection frequency either in dust or in air. The median concentration of Σ_{11}OCPs in dust was 59.4 ng/g (12.2–634 ng/g) and dusts were dominated by p,p'-DDE (34.2%), p,p'-DDT (20.8%), HCB (15.1%) and γ-HCH (10%). Indoor air concentrations of Σ_{11}OCPs ranged from 336 to 7110 pg/m^3, with a median of 945 pg/m^3. The dominant compounds in air were HCB, γ-HCH, and p,p'-DDE. Dust concentrations of all eleven OCPs correlated significantly with each other ($p < 0.05$), while air concentrations of all but PeCB correlated strongly with each other ($r_s > 0.50$, $p < 0.01$), suggesting either similar sources of OCPs or from homes of similar age.

We hypothesized that the major sources of OCPs in the homes were either outdoor secondary sources from nearby agricultural lands with past use of OCPs, and/or past indoor uses for indoor insect control (Audy et al., 2018; Booij et al., 2016; Holt et al., 2017). We tested this hypothesis by considering: (1) percentage of agricultural land in the vicinity of each home, and (2) building age (as a proxy for past indoor chemical use). The percentage of agricultural land within a 1 km radius of each home was determined using a geographic information system (GIS) and the Corine Land Cover database (European Environment Agency, 2012).

ΣHCH and ΣDDX concentrations in both dust and air were significantly correlated with the % agricultural land within a 1 km radius of the homes ($r_s > 0.31$, $p < 0.01$). Except for α-HCH in air, individual isomers of HCH and DDX also correlated significantly with % agricultural land ($r_s > 0.29$, $p < 0.03$). Additionally, homes having ≥65% agricultural land within 1 km radius had significantly higher ΣHCH and ΣDDX concentrations than homes with lower % agricultural land (one-way ANOVA, $p < 0.05$, Fig. S3). This relationship suggests that past OCP use in agricultural lands in the vicinity of homes significantly contributes to indoor OCP levels.

If past direct use of OCPs indoors is a significant source, we hypothesized that higher OCP levels would be expected in older homes (Audy et al., 2018). ΣDDX concentrations in dust and air were significantly higher in homes built before than after the 1973 ban (Mann-Whitney test $U = 61.0$, $z = -2.47$, $p < 0.05$ and $U = 47.0$, $z = -3.00$ $p < 0.05$; Fig. 2a and b). Concentrations of individual isomers and ΣHCH were also significantly higher in homes built before the ban on HCH in 1977 ($U = 31.0$, $z = -3.21$, $p < 0.05$ for dust and $U = 35.0$, $z = -3.05$, $p < 0.05$ for air; Fig. 2c and d). This suggests that use of DDX and HCH in homes > 40 years ago remains a significant contributor to present-day indoor levels. Indoor use of OCPs can result in extremely high levels indoors at the time of use, and elevated levels many years after use (Booij et al., 2016; Holt et al., 2017). Moreover, SVOCs associated with indoor dust do not readily degrade due to limited microbial and abiotic degradation processes (Hwang et al., 2008). Hence, OCPs used in the indoor environment may persist. No significant difference was found in HCB concentrations according to building age. Furthermore, we tested the relation between the two variables affecting OCP concentrations, which were % agricultural land and building age. We found no significant correlations between these two variables ($p > 0.05$), which suggested that they independently affect indoor OCP levels.

3.4. Flame retardants

The median Σ_9PBDEs concentration was 12.3 ng/g (3.2–197 ng/g) in indoor dust. The median concentration of BDE-209 was 193 ng/g (24.8 to 2965 ng/g). BDE-209 was the predominant congener, accounting for 38–99.6% of Σ_{10}PBDEs. Other dominant congeners in dust were BDE-99 and -47. The median concentration of Σ_9PBDEs in air was 4.3 pg/m^3 (0.70–113 pg/m^3). As expected, BDE-209 had a low detection frequency in air (45%) but concentrations ranged from < MDL–420 pg/m^3. The dominant congeners in air were BDE-47, -28 and -99.

The median concentration of ΣHBCDD in dust was 153 ng/g (20.2–1900 ng/g) with the greatest contribution from γ-HBCDD in 47 homes, whereas α-HBCDD dominated in 13 homes. The γ-isomer constitutes > 70% of the technical HBCDD mixture, hence its dominance in dust samples was expected. However, many studies reported higher indoor levels of α-isomer, possibly due to photolytic isomerization in dust (Harrad et al., 2009a) and/or interconversion between isomers during manufacturing of flame-retarded products (Abdallah et al., 2016; Dirtu et al., 2012). Median ΣHBCDD in air was 5.4 pg/m^3 (0.46 to 239 pg/m^3). Similar to the dust concentrations, air concentrations were dominated by γ-HBCDD in all but seven homes that were dominated by the α-isomer, three of which were the same as homes where α-isomer dominated in dust.

Fourteen out of 22 NFRs had > 50% detection frequency either in dust or in air. The median dust concentration of Σ_{14}NFRs was 122 ng/g (29.6–798 ng/g), dominated by BEH-TEBP (61%) and α- and β-DDC-CO (15%). The median concentration of Σ_{14}NFRs in air was 40.8 pg/m^3 (11.2–407 pg/m^3). PBT, PBBZ, HBB and α- and β-DBE-DBCH made up 71% of the total NFR air levels.

Among the 18 OPEs analyzed, 14 OPEs had detection frequencies > 50% either in dust or air. The concentrations of median Σ_{14}OPEs in dust was 12,400 ng/g (3080–312,000 ng/g). These levels were two orders of magnitude higher than other flame retardants (i.e., BDE-209, HBCDD and NFRs), which is consistent with other reports (Ali et al., 2013; Dirtu et al., 2012; Okeme et al., 2018). This greater abundance of OPEs levels has been ascribed to the use of some compounds as plasticizers (e.g., non-chlorinated OPEs) (Marklund et al., 2003), the need to use higher concentrations than brominated flame retardants to achieve the same flame retardancy, and the higher vapor pressures of some of the OPEs relative to brominated compounds (Sühring et al., 2016; Zhang et al., 2016). The abundant OPE species in dust in the Slovakian homes were TBOEP, TDCIPP and TPHP, constituting nearly 70% of all OPEs. OPE distributions in Slovakia were similar to those observed in other European countries (Dirtu et al., 2012; García et al., 2007; Van Den Eede et al., 2012), and Canada (Yang et al., 2019) in that TBOEP was the greatest contributor in dust samples. This profile differs from typical profiles in Asia where TDCIPP or TCIPP were the highest contributors (Tan et al., 2017; Zheng et al., 2017).

Indoor air concentrations of Σ_{14}OPEs were a median of 13.7 ng/m^3 (5.25–74.7 ng/m^3). These levels were three orders of magnitude higher than NFRs. The main contributors of OPEs in indoor air were TNBP, TIBP, TCEP and TCIPP.

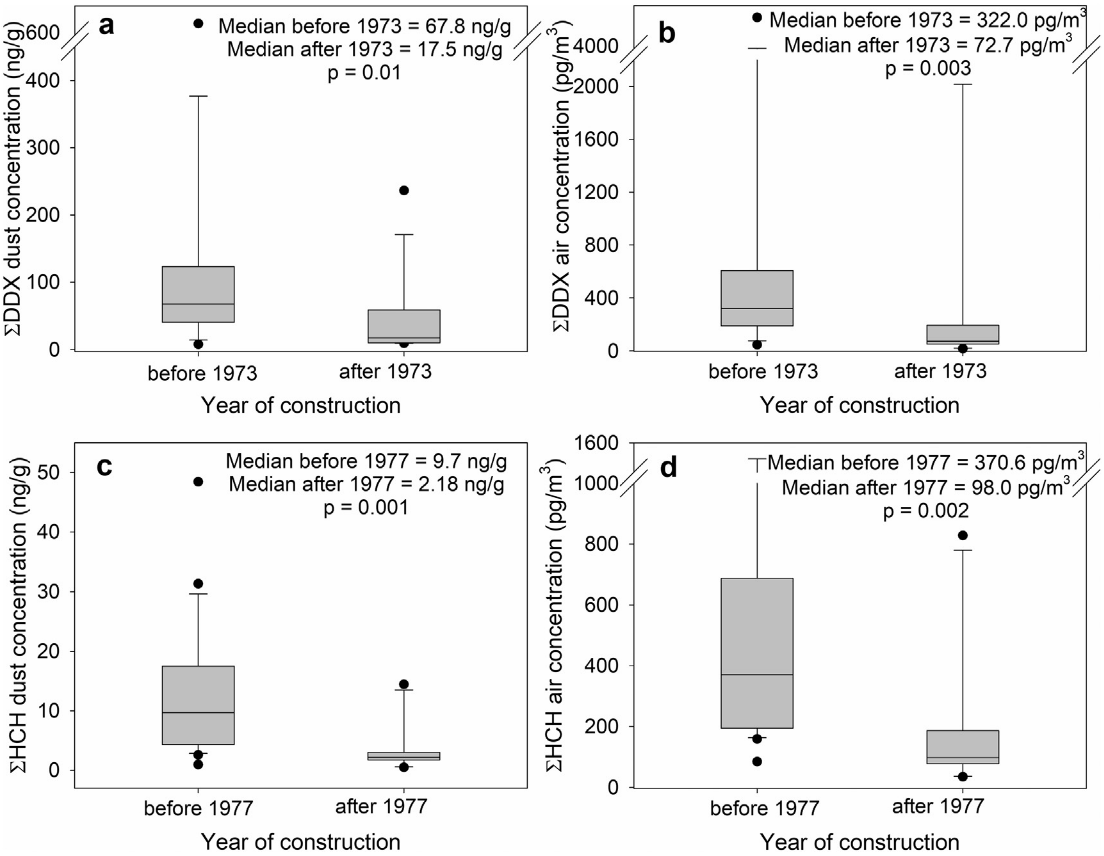

Fig. 2. Differences in indoor dust and air concentrations of ΣDDX (a and b) and ΣHCH (c and d) measured in homes built before (n = 18 for ΣDDX, n = 22 for ΣHCH) and after (n = 14 for ΣDDX, n = 10 for ΣHCH) OCPs controls were implemented in Czechoslovakia.

All PBDE congeners in dust correlated well with each other, indicating common emission sources, while air concentrations showed correlations among lower brominated and among higher brominated congeners separately, due to the different volatility of congeners. HBCDD isomers in dust correlated with each other but not with other FRs in dust, while air concentrations correlated with lower brominated PBDE congeners in air. Strong correlations were found between BDE-183 and BTBPE in dust and air (dust $r_s = 0.71$, air $r_s = 0.61$, $p < 0.001$); and most of the other individual FRs correlated with each other, but these were weak correlations.

Additionally, correlations between FRs showed an unexpected difference between urban and rural homes. Σ_9PBDEs in dust correlated significantly with Σ_{14}NFRs ($r_s = 0.61$, $p < 0.05$) and Σ_{14}OPEs ($r_s = 0.48$, $p < 0.05$) for urban homes, while no correlations were found between these groups for rural homes. The reason for this difference between urban and rural homes is unclear, but may suggest differences in the timing of product replacement and new product purchase between urban and rural homes.

As mentioned above, indoor FR levels are expected to be influenced by indoor sources and building characteristics. We found no significant correlations between FRs in air or dust and the prevalence of PVC flooring, wall-to-wall carpeting or new furniture in the home. However, we were able to identify regional differences in replacement FRs used by comparing the Slovak indoor compound distributions with those from North American indoor environments. FireMaster-550 (FM-550) was a common substitute for penta-BDE mixture after its restriction in North America (Stapleton et al., 2008). The ratio of the brominated compounds in FM-550, i.e. f_{EHTBB} = EH-TBB/EH-TBB + BEH-TEBP, can indicate whether these compounds are emitted from a material containing FM-550 (Venier et al., 2016). Before calculating the ratio, air concentrations were corrected for the volatilities of compounds by using PUF-air partition coefficient, $K_{\mathrm{PUF\text{-}air}}$, according to Francisco et al. (2017) and Okeme et al. (2017). The median of f_{EHTBB} was 0.09 (0.02–0.54) and 0.001 (0.00–0.02) for dust and air concentrations, respectively. These ratios were both much lower than that of FM-550, which is 0.77 (Ma et al., 2012). Lower f_{EHTBB} ratios have been typically observed in European indoor samples (Cequier et al., 2014; Venier et al., 2016) than in North American samples (Dodson et al., 2012; Venier et al., 2016). One explanation is the use of BEH-TEBP individually as a flame retardant and plasticizer in Europe, compared with greater use of FM-550 in North America (Venier et al., 2016). For example, another commercial mixture, DP-45 which is 100% BEH-TEBP, is listed as a flame retardant and plasticizer for PVC applications, e.g. wire and cable insulation, coating fabrics, and sheeting and flexible polyurethane (Lanxess, 2016). Both FM-550 and DP-45 are high production volume chemicals in USA, whereas in Europe FM-550 is not registered but DP-45 is manufactured and/or imported at volumes similar to USA (ECHA, 2018). The wide application of BEH-TEBP in consumer products in Europe likely explains its abundance relative to EH-TBB (medians = 69.5 and 6.8 ng/g in dust, respectively). In fact, EH-TBB does not have an EC number and it has no REACH record. Hence, direct use of EH-TBB in commercial products in Europe still remains unidentified. Although f_{EHTBB} suggested that EH-TBB and BEH-TEBP had different sources, the two compounds were significantly correlated ($r_s = 0.46$, $p < 0.001$). Therefore, we speculate that the low amount of EH-TBB detected in homes likely originates from the limited use of FM-550 in products in Europe, e.g., imported as finished products.

3.5. Polycyclic aromatic hydrocarbons

Σ_{27}PAHs had a median dust concentration of 2008 ng/g (505 to

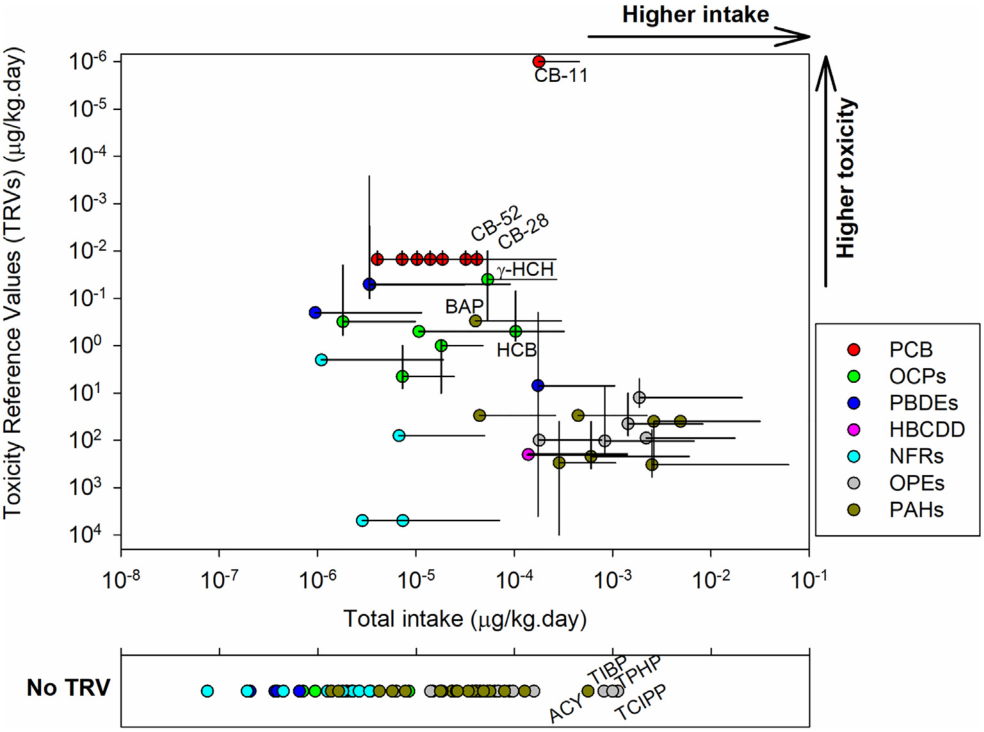

Fig. 3. Total human intake for child age 6–11 vs toxicity reference values (TRVs) of indoor SVOCs. Vertical lines represent the range of toxicity values from literature and regulatory agencies, horizontal lines represent the high intake scenario, and points represent median exposure scenario and median values of TRVs. Compounds having both high intake and high toxicity (i.e. highest priority) are labeled. The box below the graph shows compounds with no available TRVs.

37,018 ng/g). The dust concentrations were dominated by phenanthrene, fluoranthene and pyrene. The median of Σ_{27}PAHs in indoor air was 43.1 ng/m^3 (17.2 to 1923 ng/m^3), dominated by acenaphthene, fluorene and phenanthrene. All PAH compounds in dust correlated well with each other, indicating similar sources for all, while air concentrations were not correlated for some PAHs (notably between acenaphthene and PAHs with > 4 rings), possibly due to differences in volatility.

Indoor PAH contamination is typically attributed to outdoor sources such as vehicle or industrial emissions, or indoor sources such as smoking or domestic biomass burning (Fromme et al., 2004; Whitehead et al., 2013; Yadav et al., 2018). The nearest industrial PAH emission source to the Slovak homes was a steel mill also located at the Chemko Strážske site. Nevertheless, no meaningful correlation was found between indoor PAH levels and distance to the steel mill. We also tested the influence of vehicle emissions using the total length of roads within a 1 km radius of each home derived using GIS. However, road length was also not useful for explaining indoor PAH levels.

Rather, we identified domestic heating as the likely major indoor source of PAH. Σ_{27}PAHs in dust, but not air, was significantly higher in homes heated with wood (median = 3100 ng/g) than those using central or gas heating (median = 1679 ng/g) (Mann-Whitney test $U = 170$, $z = -2.69$, $p < 0.05$, Fig. S4). Additionally, levels in wood-burning homes were significantly higher for 18 individual PAHs in dust, seven of which were identified as having indoor emissions due to wood burning (Gustafson et al., 2008). We also considered whether cigarette smoking ($n = 27$) affected indoor PAH levels, however, no correlation between PAH concentrations and indoor smoking was found. Hence, indoor wood burning was the dominant factor controlling indoor PAH concentrations.

3.6. Human exposure to SVOCs and associated risk

A challenge in evaluating implications of compounds detected in indoor environments and prioritizing monitoring is the wide range of concentrations in different indoor matrices, which are associated with different human exposure pathways, in addition to the wide range of toxicities of the various compounds. Compounds are often highlighted due to elevated levels in indoor environments (e.g., OPEs); however, this type of prioritization does not address the risk, i.e., exposure combined with toxicity. Several attempts have been made to address the risk associated with residential indoor exposure (Bonvallot et al., 2010; Mitro et al., 2016; Pelletier et al., 2018; Shin et al., 2012; Smith et al., 2016). We propose a classification framework to merge information on human exposure and toxicity to evaluate risk for indoor-relevant compounds. Further, we have applied it to prioritize risk from indoor exposures from the levels reported in Slovakian homes.

To address the combined influence of indoor air and dust on human exposure, human intake of SVOCs was calculated as the sum of exposure doses via air inhalation, dust ingestion, and dermal contact with dust. The daily exposure dose formulas and assumptions are provided in Table S27, and Fig. S5 summarizes the % distribution of exposure routes for individual compounds. Intakes were calculated for median and high (95th percentile) scenarios for a child between ages six and eleven, as the samples were taken in children's bedrooms. We also calculated intake for an adult female for comparison. Total intakes of SVOCs for both scenarios are given in Tables S28 and S29.

The total intake for individual compounds ranged between $7.50 * 10^{-5}$ ng/kg/d and $2.13 * 10^{-5}$ ng/kg/d for TBX to 4.9 ng/kg/d and 1.7 ng/kg/d for phenanthrene according to the median intake scenario, for a child and an adult, respectively. The highest exposures were observed for PAHs > OPEs > PCBs. Estimated exposure for the high intake scenario for higher chlorinated PCB congeners, DDTs, BDE-99, PBT, HBB, acenaphthene and cyclopenta(cd)pyrene were more than one order of magnitude higher than their median intake scenario rates. Lower chlorinated PCBs, OCPs except DDT, PBBZ, PBT, TIBP, TNBP and lower molecular weight PAHs had higher exposure doses via inhalation than dust ingestion and dermal dust contact due to their higher concentration in indoor air (Figs. S5 and S6). As expected, the child had higher estimated intakes of SVOCs than the adult female.

To evaluate the relative toxicities of the target compounds, we compiled non-carcinogenic oral chronic (or sub-chronic if the chronic value was not available) toxicity reference values (TRVs) from the literature and regulatory sources. TRVs are defined as a maximum dose of a compound that can be ingested daily without risk for people's health. We selected those for which the details on the calculation method were accessible for further assessment (see supplementary information for methods and Table S29).

Finally, intake estimates and TRVs were visualized on a scatterplot to allow prioritization of compounds for monitoring and evaluating the risk in the indoor environment for a child (Fig. 3). When several selected TRVs were available for one compound, the median and the full range of values were shown. The high intake scenario is reflected as half error bars. For comparison, intake scenarios for an adult female are presented in Fig. S7. Priority compounds are those in the upper right corner that have the highest intake and highest toxicity (lowest TRVs). The highest priority compound in our study was CB-11. It does not have an individual TRV; however, it was assigned the tolerable daily intake value for dioxins and planar PCBs (Baars et al., 2001) since it is a planar PCB congener. It has not been frequently monitored in indoor environments since CB-11 is a non-legacy PCB congener. However, its emission indoors from paints and indoor materials has been reported (see Section 3.2). This linking of intake and TRVs emphasizes that CB-11 should be prioritized for indoor monitoring and evaluation of associated risk.

Other high priority SVOCs were HCB, γ-HCH, CB-28 and-52, and benzo[*a*]pyrene. It is important to note that new uses of HCB, γ-HCH and PCBs were restricted decades ago, yet they still exhibit high human intake due to their past usage. Individual PCB congeners have no specific TRV. As such, we assigned the values reported for non-planar PCBs and/or for Aroclors to individual PCB congeners except for CB-11. Benzo[*a*]pyrene is one of the many compounds emitted from indoor wood burning (Gustafson et al., 2008) and has a higher TRV than other PAHs which had higher intake. The risk from PAHs is probably even higher, as we are considering only non-carcinogenic TRVs, but note that benzo[a]pyrene is classified as carcinogenic to humans (Group 1) according to the International Agency for Research on Cancer (IARC) (IARC, 2010). Bonvallot et al. (2010) ranked compounds present in indoor dust depending on the selected TRVs and concentrations in dust and similarly highlighted PCBs and γ-HCH (Bonvallot et al., 2010).

Some SVOCs did not appear to present a high risk because they had low human intake in this study. However, some of these chemicals had low TRVs, e.g., BDE-99, -47, and -153, PCBs, β-HCH and DDT, suggesting that these compounds require attention in indoor environments with higher levels.

Prioritization was not possible for 60% of the 88 compounds evaluated, and these are shown in Fig. 3 according to only their estimated intake. TCIPP, TPHP, TIBP, acenaphthylene, EHDPP, retene, TPrP and chrysene had high intake but no reliable TRVs. More effort should be given to toxicological studies of these compounds, especially those for which the current literature suggests high toxicity. For example, Behl et al. (2016) found that TPHP was the second most active reproductive toxicant for *C. elegans* after penta-BDE commercial mixture DE-71. Additionally, four OPEs had the highest intake but current TRVs suggested low risk. However, the TRVs of these compounds are highly uncertain because of insufficient and/or outdated testing. For example, a cursory view of TDCIPP and TCEP might suggest minimal risk, however their TRVs were derived from studies conducted almost 30 years ago (ATSDR, 2012), while much of the research on their toxicity was done in the last 10 years but TRVs have not yet been determined. We also noted that different calculation methods of TRVs could produce values differing by up to several orders of magnitude, as seen for BDE-99 and -209 (ATSDR, 2018; National Research Council, 2000; USEPA, 2018). For BDE-99, RIVM (Baars et al., 2001) provides a maximal allowed intake that is 4 orders of magnitude lower than the reference dose from the US EPA (IRIS), based on a different experimental value and a different calculation method that takes into account known bioaccumulation properties of the chemical in human as compared to rodents. It is also surprising that TRVs for OPEs, such as TDCIPP, are several orders of magnitude higher than for PBDEs such as BDE-47 or -99, although they were shown to have comparable levels of toxicity when tested in parallel in experimental systems (Behl et al., 2015; Jarema et al., 2015; Noyes et al., 2015). Finally, epidemiology studies indicate that several OPEs may represent a risk for people's health. For example, two epidemiological studies reported a statistical association between levels of TNBP (one of the most abundant compounds in air and dust) and the risk of asthma, mucosal symptoms of SHS (sick house syndrome) or allergic rhinitis (Araki et al., 2014; Kanazawa et al., 2010). Other epidemiological studies indicate associations between TCEP or TPHP exposure with neurobehavioral defects in children and thyroid cancer, and between TDCIPP or TPHP exposure and decrease in male fertility (Carignan et al., 2018; Castorina et al., 2017; Hoffman et al., 2017; Hutter et al., 2013; Meeker et al., 2013; Meeker and Stapleton, 2010). However, these outcomes are not captured in the available TRVs.

The use of non-carcinogenic oral TRVs was the most appropriate to estimate the toxicity for most of the compounds analyzed, but, in some cases, it could underestimate the risk, for example, for chemicals that are classified as carcinogenic to humans (Group 1) according to IARC, such as benzo[*a*]pyrene or PCBs (IARC, 2010, 2016). Also, the compounds having higher exposure via inhalation than ingestion (see Fig. S5) should be investigated further in terms of their reference dose for inhalation (if available) to have a better understanding of their toxicity via this exposure route. Among the compounds with higher inhalation intake, only γ-HCH has an inhalation reference dose (0.1 μg/kg/d) (US Environmental Protection Agency, 2018). This value is within the range of TRVs used in our study for γ-HCH, and hence does not change the conclusion on the risk associated with this compound.

In our framework to prioritize and evaluate the risk of individual compounds, we did not distinguish the main health effects associated with each chemical, as done by others (Mitro et al., 2016). Future iterations of this framework could be adapted to consider this aspect, which could be especially useful for identifying potential mixture effects.

4. Conclusions and implications

Indoor air and dust levels of 88 SVOCs from seven compound classes in Slovakia appeared to be related to a country's income level (e.g., PCBs), outdoor uses (OCPs), and/or indoor activities (e.g., PAHs). In particular, a comparison by country with published indoor levels of PCBs and FRs showed higher levels in high-income countries. Indoor PCBs concentrations in Slovakian homes nearby a former PCB production facility fell within the range of previously reported values from other countries, and were not uniquely elevated due to proximity to the PCB factory. Indoor OCP levels were correlated with the percentage of agricultural lands in the vicinity, and with the age of the sampled homes, both taken as surrogates for past outdoor and indoor OCP use, respectively. FR concentrations and distributions were consistent with the findings of previous European studies. Indoor PAH levels were associated with wood burning for home heating and not smoking.

The highest indoor exposures for a child between ages 6–11 via air inhalation, dust ingestion and dermal contact were for PAHs > OPEs > PCBs. We presented a classification framework to prioritize the risk of these compounds by considering human intake and toxicity where toxicity was expressed as the highest recommended intake per day (TRV). According to this classification, CB-11, HCB, γ-HCH, CB-28, CB-52 and benzo[*a*]pyrene should be prioritized for indoor monitoring and evaluation of the associated risks. However, 53 compounds (60% of the total number of compounds considered) could not be evaluated due to a lack of reliable toxicological information.

Identification of a wide range of compounds in the indoor environment enabled comparing not only the levels of various compound groups, but also the associated risks. Overall, we concluded that exposure via indoor routes can continue long after home construction and the enactment of restrictions on *new* uses of the compounds, e.g., exposure to legacy PCBs, OCPs. We also concluded that other current indoor sources could be targeted for reduction, e.g., PAH from home heating by wood burning. Further, industrial manufacturing processes leading to inadvertent production of PCB, especially CB-11, should be adapted and improved to reduce PCB emissions from new commercial products. Finally, attention should be given to SVOCs that had high human intake but had no reliable TRVs, e.g., TCIPP, TPHP, TIBP, acenaphthylene, EHDPP, retene, TPrP, and chrysene, or had TRVs that may underestimate their toxicity, e.g. TDCIPP and TCEP. A "next step" would be to consider the mixture effects of the wide range of SVOCs with significant human exposure in residential indoor environments.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envint.2019.04.001.

References

Abdallah, M.A.E., Bressi, M., Oluseyi, T., Harrad, S., 2016. Hexabromocyclododecane and tetrabromobisphenol-A in indoor dust from France, Kazakhstan and Nigeria: implications for human exposure. Emerg. Contam. 2, 73–79. https://doi.org/10.1016/j.emcon.2016.03.006.

Ali, N., Van Den Eede, N., Dirtu, A.C., Neels, H., Covaci, A., 2012. Assessment of human exposure to indoor organic contaminants via dust ingestion in Pakistan. Indoor Air 22, 200–211. https://doi.org/10.1111/j.1600-0668.2011.00757.x.

Ali, N., Ali, L., Mehdi, T., Dirtu, A.C., Al-Shammari, F., Neels, H., Covaci, A., 2013. Levels and profiles of organochlorines and flame retardants in car and house dust from Kuwait and Pakistan: implication for human exposure via dust ingestion. Environ. Int. 55, 62–70. https://doi.org/10.1016/j.envint.2013.02.001.

Ampleman, M.D., Martinez, A., DeWall, J., Rawn, D.F.K., Hornbuckle, K.C., Thorne, P.S., 2015. Inhalation and dietary exposure to PCBs in urban and rural cohorts via congener-specific measurements. Environ. Sci. Technol. 49, 1156–1164. https://doi.org/10.1021/es5048039.

Araki, A., Saito, I., Kanazawa, A., Morimoto, K., Nakayama, K., Shibata, E., Tanaka, M., Takigawa, T., Yoshimura, T., Chikara, H., Saijo, Y., Kishi, R., 2014. Phosphorus flame retardants in indoor dust and their relation to asthma and allergies of inhabitants. Indoor Air 24, 3–15. https://doi.org/10.1111/ina.12054.

ATSDR, 2012. Toxicological Profile for Phosphate Ester Flame Retardants. https://doi.org/10.1007/s00244-006-0094-3.

ATSDR, 2018. Agency for Toxic Substances & Disease Registry Minimal Risk Levels (MRLs) for Hazardous Substances [WWW Document]. URL. https://www.atsdr.cdc.gov/mrls/mrllist.asp, Accessed date: 27 June 2018.

Audy, O., Melymuk, L., Venier, M., Vojta, S., Becanova, J., Romanak, K., Vykoukalova, M., Prokes, R., Kukucka, P., Diamond, M.L., Klanova, J., 2018. PCBs and organochlorine pesticides in indoor environments - a comparison of indoor contamination in Canada and Czech Republic. Chemosphere 206, 622–631. https://doi.org/10.1016/j.chemosphere.2018.05.016.

Baars, A.J., Theelen, R.M.C., Janssen, P.J.C.M., Hesse, J.M., van Apeldoorn, M.E., Meijerink, M.C.M., Verdam, L., Zeilmaker, M.J., 2001. Re-evaluation of human-toxicological maximum permissible risk levels, RIVM report 711701 025. doi:https://doi.org/10.1051/vetres:2008046.

Behl, M., Hsieh, J.H., Shafer, T.J., Mundy, W.R., Rice, J.R., Boyd, W.A., Freedman, J.H., Hunter, E.S., Jarema, K.A., Padilla, S., Tice, R.R., 2015. Use of alternative assays to identify and prioritize organophosphorus flame retardants for potential developmental and neurotoxicity. Neurotoxicol. Teratol. 52, 181–193. https://doi.org/10.1016/j.ntt.2015.09.003.

Behl, M., Rice, J.R., Smith, M.V., Co, C.A., Bridge, M.F., Hsieh, J.-H., Freedman, J.H., Boyd, W.A., 2016. Comparative toxicity of organophosphate flame retardants and polybrominated diphenyl ethers to Caenorhabditis elegans. Toxicol. Sci. 154 (2), 241–252.

Bergman, A., Rydén, A., Law, R.J., de Boer, J., Covaci, A., Alaee, M., Birnbaum, L.S., Petreas, M.X., Rose, M., Sakai, S., Van den Eede, N., van der Veen, I., 2012. A novel abbreviation standard for organobromine, organochlorine and organophosphorus flame retardants and some characteristics of the chemicals. Environ. Int. 49, 57–82.

Bonvallot, N., Mandin, C., Mercier, F., Le Bot, B., Glorennec, P., 2010. Health ranking of ingested semi-volatile organic compounds in house dust: an application to France. Indoor Air 20, 458–472. https://doi.org/10.1111/j.1600-0668.2010.00667.x.

Booij, P., Holoubek, I., Klánová, J., Kohoutek, J., Dvorská, A., Magulová, K., Al-Zadjali, S., Čupr, P., 2016. Current implications of past DDT indoor spraying in Oman. Sci. Total Environ. 550, 231–240. https://doi.org/10.1016/j.scitotenv.2015.12.044.

Breivik, K., Sweetman, A., Pacyna, J.M., Jones, K.C., 2002. Towards a global historical emission inventory for selected PCB congeners - a mass balance approach: 1. Global production and consumption. Sci. Total Environ. 290, 181–198. https://doi.org/10.1016/S0048-9697(01)01075-0.

Breivik, K., Sweetman, A., Pacyna, J.M., Jones, K.C., 2007. Towards a global historical emission inventory for selected PCB congeners–a mass balance approach 3. An update. Sci. Total Environ. 377, 296–307. https://doi.org/10.1016/j.scitotenv.2007.02.026.

Carignan, C.C., Mínguez-Alarcón, L., Williams, P.L., Meeker, J.D., Stapleton, H.M., Butt, C.M., Toth, T.L., Ford, J.B., Hauser, R., 2018. Paternal urinary concentrations of organophosphate flame retardant metabolites, fertility measures, and pregnancy outcomes among couples undergoing in vitro fertilization. Environ. Int. 111, 232–238. https://doi.org/10.1016/j.envint.2017.12.005.

Castorina, R., Bradman, A., Stapleton, H.M., Butt, C.M., Avery, D., Harley, K.G., Gunier, R.B., Holland, N., Eskenazi, B., 2017. Current-use flame retardants: maternal exposure and neurodevelopment in children of the CHAMACOS cohort. Chemosphere 189, 574–580. https://doi.org/10.1016/j.chemosphere.2017.09.037.

Cequier, E., Ionas, A.C., Covaci, A., Marce, R.M., Becher, G., Thomsen, C., 2014. Occurrence of a broad range of legacy and emerging flame retardants in indoor environments in Norway. Environ. Sci. Technol. 48, 6827–6835. https://doi.org/10.1021/es500516u.

Dirtu, A.C., Ali, N., Van den Eede, N., Neels, H., Covaci, A., 2012. Country specific comparison for profile of chlorinated, brominated and phosphate organic contaminants in indoor dust. Case study for Eastern Romania, 2010. Environ. Int. 49, 1–8. https://doi.org/10.1016/j.envint.2012.08.002.

Dodson, R.E., Perovich, L.J., Covaci, A., Van Den Eede, N., Ionas, A.C., Dirtu, A.C., Brody, J.G., Rudel, R.A., 2012. After the PBDE phase-out: a broad suite of flame retardants in repeat house dust samples from California. Environ. Sci. Technol. 46, 13056–13066. https://doi.org/10.1021/es303879n.

Drobná, B., Fabišiková, A., Čonka, K., Chovancová, J., Dömötörová, M., Wimmerová, S., Šovčíková, E., Kočan, A., 2011. Differences between dioxin-like PCB, non-dioxin-like PCB, polychlorinated dibenzo-p-dioxin and dibenzofuran intake from human milk and infant milk formula by infants in the Michalovce district (Slovakia). J. Food Nutr. Res. 50, 106–117.

ECHA, 2018. European Chemicals Agency Information on Chemicals. [WWW Document]. URL. https://echa.europa.eu/information-on-chemicals, Accessed date: 29 June 2018.

European Environment Agency, 2012. Corine Land Cover 2012 [WWW Document].

Francisco, A.P., Harner, T., Eng, A., 2017. Measurement of polyurethane foam – air partition coefficients for semivolatile organic compounds as a function of temperature: application to passive air sampler monitoring. Chemosphere 174, 638–642. https://doi.org/10.1016/j.chemosphere.2017.01.135.

Frederiksen, M., Meyer, H.W., Ebbehøj, N.E., Gunnarsen, L., 2012. Polychlorinated biphenyls (PCBs) in indoor air originating from sealants in contaminated and uncontaminated apartments within the same housing estate. Chemosphere 89, 473–479.

Fromme, H., Lahrz, T., Piloty, M., Gebhardt, H., Oddoy, A., Rüden, H., 2004. Polycyclic aromatic hydrocarbons inside and outside of apartments in an urban area. Sci. Total Environ. 326, 143–149. https://doi.org/10.1016/j.scitotenv.2004.02.002.

García, M., Rodríguez, I., Cela, R., 2007. Microwave-assisted extraction of organophosphate flame retardants and plasticizers from indoor dust samples. J. Chromatogr. A 1152, 280–286. https://doi.org/10.1016/j.chroma.2006.11.046.

Gioia, R., Steinnes, E., Thomas, G.O., Mejier, S.N., Jones, K.C., 2006. Persistent organic pollutants in European background air: derivation of temporal and latitudinal trends. J. Environ. Monit. 8, 700–710. https://doi.org/10.1039/b604821h.

Gustafson, P., Östman, C., Sällsten, G., 2008. Indoor levels of polycyclic aromatic hydrocarbons in homes with or without wood burning for heating. Environ. Sci. Technol. 42, 5074–5080. https://doi.org/10.1021/es800304y.

Harner, T., Shoeib, M., Diamond, M., Stern, G., Rosenberg, B., 2004. Using passive air samplers to assess urban-rural trends for persistent organic pollutants. 1. Polychlorinated biphenyls and organochlorine pesticides. Environ. Sci. Technol. 38, 4474–4483. https://doi.org/10.1021/es040302r.

Harrad, S., Abdallah, M.A.E., Covaci, A., 2009a. Causes of variability in concentrations and diastereomer patterns of hexabromocyclododecanes in indoor dust. Environ. Int. 35, 573–579. https://doi.org/10.1016/j.envint.2008.10.005.

Harrad, S., Ibarra, C., Robson, M., Melymuk, L., Zhang, X., Diamond, M., Douwes, J., 2009b. Polychlorinated biphenyls in domestic dust from Canada, New Zealand, United Kingdom and United States: implications for human exposure. Chemosphere 76, 232–238. https://doi.org/10.1016/j.chemosphere.2009.03.020.

Hazrati, S., Harrad, S.J., 2006. Causes of variability in concentrations of polychlorinated biphenyls and polybrominated diphenyl ethers in indoor air. Environ. Sci. Technol. 40, 7584–7589.

Herkert, N.J., Jahnke, J.C., Hornbuckle, K.C., 2018. Emissions of Tetrachlorobiphenyls (PCBs 47, 51, and 68) from polymer resin on kitchen cabinets as a non-Aroclor source to residential air. Environ. Sci. Technol. 52, 5154–5160. https://doi.org/10.1021/acs.est.8b00966.

Hertz-Picciotto, I., Trnovec, T., Kočan, A., Charles, M.J., Čižnar, P., Langer, P., Sovčikova, E., James, R., 2003. PCBs and early childhood development in Slovakia: study design and background. Fresenius Environ. Bull. 12, 208–214.

Hiller, E., Zemanová, L., Sirotiak, M., Jurkovič, L., 2011. Concentrations, distributions, and sources of polychlorinated biphenyls and polycyclic aromatic hydrocarbons in bed sediments of the water reservoirs in Slovakia. Environ. Monit. Assess. 173, 883–897. https://doi.org/10.1007/s10661-010-1431-6.

Hoffman, K., Lorenzo, A., Butt, C.M., Hammel, S.C., Henderson, B.B., Roman, S.A., Scheri, R.P., Stapleton, H.M., Sosa, J.A., 2017. Exposure to flame retardant chemicals and occurrence and severity of papillary thyroid cancer: a case-control study. Environ. Int. 107, 235–242. https://doi.org/10.1016/j.envint.2017.06.021.

Holoubek, I., Adamec, V., Bartoš, M., Bláha, K., Bláha, L., Budňáková, M., Černa, M., Čupr, P., Demnerová, K., Drápal, J., Hajšlová, J., Hanzálková, M., Holoubková, I., Hrabětová, S., Jech, L., Klánová, J., Kohoutek, J., Kužílek, V., Machálek, P., Matějů, V., Matoušek, J., Matoušek, M., Mestřík, V., Novák, J., Ocelka, T., Pekárek, V., Petrlík, J., Petira, O., Provazník, K., Punčochář, M., Rieder, M., Ruprich, J., Sáňka, M., Tomaniová, M., Vácha, R., Volka, K., Zbíral, J., 2006. The National Implementation Plan for Implementation of the Stockholm Convention in the Czech Republic [WWW Document].

Holt, E., Audy, O., Booij, P., Melymuk, L., Prokes, R., Klánová, J., 2017. Organochlorine pesticides in the indoor air of a theatre and museum in the Czech Republic: inhalation exposure and cancer risk. Sci. Total Environ. 609, 598–606. https://doi.org/10.1016/j.scitotenv.2017.07.203.

Hovander, L., Linderholm, L., Athanasiadou, M., Athanassiadis, I., Bignert, A., Fängström, B., Kocan, A., Petrik, J., Trnovec, T., Bergman, A., 2006. Levels of PCBs and their metabolites in the serum of residents of a highly contaminated area in Eastern Slovakia. Environ. Sci. Technol. 40, 3696–3703. https://doi.org/10.1021/es0525657.

Hu, D., Hornbuckle, K.C., 2010. Inadvertent Polychlorinated Biphenyls in Commercial Paint Pigments 44. pp. 2822–2827. https://doi.org/10.1021/es902413k.

Hung, H., Katsoyiannis, A.A., Brorström-Lundén, E., Olafsdottir, K., Aas, W., Breivik, K., Bohlin-Nizzetto, P., Sigurdsson, A., Hakola, H., Bossi, R., Skov, H., Sverko, E., Barresi, E., Fellin, P., Wilson, S., 2016. Temporal trends of persistent organic pollutants (POPs) in Arctic air: 20 years of monitoring under the Arctic monitoring and assessment Programme (AMAP). Environ. Pollut. 217, 52–61. https://doi.org/10.1016/j.envpol.2016.01.079.

Hutter, H.-P., Haluza, D., Piegler, K., Hohenblum, P., Fröhlich, M., Scharf, S., Uhl, M., Damberger, B., Tappler, P., Kundi, M., Wallner, P., Moshammer, H., 2013. Semivolatile compounds in schools and their influence on cognitive performance of children. Int. J. Occup. Med. Environ. Health 26, 628–635. https://doi.org/10.2478/s13382-013-0125-z.

Hwang, H.M., Park, E.K., Young, T.M., Hammock, B.D., 2008. Occurrence of endocrine-disrupting chemicals in indoor dust. Sci. Total Environ. 404, 26–35. https://doi.org/10.1016/j.scitotenv.2008.05.031.

IARC, 2010. IARC Monographs on the Evaluation of Carcinogenic Risks to Humans: Some Non-heterocyclic Polycyclic Aromatic Hydrocarbons and some Related Exposures, IARC Monographs on the Evaluation of Carcinogenic Risks to Humans. (Lyon, France).

IARC, 2016. IARC Monographs on the Evaluation of Carcinogenic Risks to Humans: Polychlorinated Biphenyls and Polybrominated Biphenyls. Lyon, France.

Jarema, K.A., Hunter, D.L., Shaffer, R.M., Behl, M., Padilla, S., 2015. Acute and developmental behavioral effects of flame retardants and related chemicals in zebrafish. Neurotoxicol. Teratol. 52, 194–209. https://doi.org/10.1016/j.ntt.2015.08.010.

Jílková, S., Melymuk, L., Vojta, Š., Vykoukalová, M., Bohlin-Nizzetto, P., Klánová, J., 2018. Small scale spatial variability of flame retardants in indoor dust and implications for dust sampling. Chemosph 206, 132–141. https://doi.org/10.1016/j.chemosphere.2018.04.146.

Jursa, S., Chovancová, J., Petrík, J., Lokša, J., 2006. Dioxin-like and non-dioxin-like PCBs in human serum of Slovak population. Chemosphere 64, 686–691. https://doi.org/10.1016/j.chemosphere.2005.10.048.

Kanazawa, A., Saito, I., Araki, A., Takeda, M., Ma, M., Saijo, Y., Kishi, R., 2010. Association between indoor exposure to semi-volatile organic compounds and building-related symptoms among the occupants of residential dwellings. Indoor Air 20, 72–84. https://doi.org/10.1111/j.1600-0668.2009.00629.x.

Kocan, A., Petrik, J., Jursa, S., Chovancova, J., Drobna, B., 2001. Environmental contamination with polychlorinated biphenyls in the area of their former manufacture in Slovakia. Chemosphere 43, 595–600. https://doi.org/10.1016/S0045-6535(00)00411-2.

Kohler, M., Tremp, J., Zennegg, M., Seiler, C., Minder-Kohler, S., Beck, M., Lienemann, P., Wegmann, L., Schmid, P., 2005. Joint sealants: an overlooked diffuse source of polychlorinated biphenyls in buildings. Environ. Sci. Technol. 39, 1967–1973. https://doi.org/10.1021/es048632z.

Lanxess, 2016. DP-45 - Halogenated Flame Retardant [WWW Document].

Liu, X., Guo, Z., Krebs, K.A., Greenwell, D.J., Roache, N.F., Stinson, R.A., Nardin, J.A., Pope, R.H., 2016. Laboratory study of PCB transport from primary sources to building materials. Indoor Built Environ 25, 635–650. https://doi.org/10.1177/1420326X15623355.

Lucattini, L., Poma, G., Covaci, A., de Boer, J., Lamoree, M.H., Leonards, P.E.G., 2018. A review of semi-volatile organic compounds (SVOCs) in the indoor environment: occurrence in consumer products, indoor air and dust. Chemosphere 201, 466–482. https://doi.org/10.1016/j.chemosphere.2018.02.161.

Lv, J., Xu, R., Wu, G., Zhang, Q., Li, Y., Wang, P., Liao, C., Liu, J., Jiang, G., Wei, F., 2009. Indoor and outdoor air pollution of polycyclic aromatic hydrocarbons (PAHs) in Xuanwei and Fuyuan, China. J. Environ. Monit. 11, 1368. https://doi.org/10.1039/b900382g.

Ma, Y., Venier, M., Hites, R.A., 2012. 2-Ethylhexyl tetrabromobenzoate and Bis(2-ethylhexyl) tetrabromophthalate flame retardants in the great lakes atmosphere. Environ. Sci. Technol. 46, 204–208. https://doi.org/10.1021/es203251f.

Maertens, R.M., Yang, X., Zhu, J., Gagne, R.W., Douglas, G.R., White, P.A., 2008. Mutagenic and carcinogenic hazards of settled house dust I: polycyclic aromatic hydrocarbon content and excess lifetime cancer risk from preschool exposure. Environ. Sci. Technol. 42, 1747–1753. https://doi.org/10.1021/es702449c.

Marek, R.F., Thorne, P.S., Herkert, N.J., Awad, A.M., Hornbuckle, K.C., 2017. Airborne PCBs and OH-PCBs inside and outside urban and rural U.S. schools. Environ. Sci. Technol. 51, 7853–7860. https://doi.org/10.1021/acs.est.7b01910.

Marklund, A., Andersson, B., Haglund, P., 2003. Screening of organophosphorus compounds and their distribution in various indoor environments. Chemosphere 53, 1137–1146. https://doi.org/10.1016/S0045-6535(03)00666-0.

Meeker, J.D., Stapleton, H.M., 2010. House dust concentrations of organophosphate flame retardants in relation to hormone levels and semen quality parameters. Environ. Health Perspect. 118, 318–323. https://doi.org/10.1289/ehp.0901332.

Meeker, J.D., Cooper, E.M., Stapleton, H.M., Hauser, R., 2013. Exploratory analysis of urinary metabolites of phosphorus-containing flame retardants in relation to markers of male reproductive health. Endocr. Disruptors 1, e26306. https://doi.org/10.4161/endo.26306.

Melymuk, L., Bohlin-Nizzetto, P., Kukučka, P., Vojta, Š., Kalina, J., Čupr, P., Klánová, J., 2016a. Seasonality and indoor/outdoor relationships of flame retardants and PCBs in residential air. Environ. Pollut. 218, 392–401. https://doi.org/10.1016/J.ENVPOL.2016.07.018.

Melymuk, L., Bohlin-Nizzetto, P., Vojta, Š., Krátká, M., Kukučka, P., Audy, O., Přibylová, P., Klánová, J., 2016b. Distribution of legacy and emerging semivolatile organic compounds in five indoor matrices in a residential environment. Chemosphere 153, 179–186. https://doi.org/10.1016/j.chemosphere.2016.03.012.

Mitro, S.D., Dodson, R.E., Singla, V., Adamkiewicz, G., Elmi, A.F., Tilly, M.K., Zota, A.R., 2016. Consumer product chemicals in indoor dust: a quantitative meta-analysis of U.S. studies. Environ. Sci. Technol. 50, 10661–10672. https://doi.org/10.1021/acs.est.6b02023.

National Research Council, 2000. Toxicological Risks of Selected Flame-Retardant Chemicals. National Academy Press, Washington, DC.

Noyes, P.D., Haggard, D.E., Gonnerman, G.D., Tanguay, R.L., 2015. Advanced morphological - behavioral test platform reveals neurodevelopmental defects in embryonic zebrafish exposed to comprehensive suite of halogenated and organophosphate flame retardants. Toxicol. Sci. 145, 177–195. https://doi.org/10.1093/toxsci/kfv044.

Okeme, J.O., Webster, E.M., Parnis, J.M., Diamond, M.L., 2017. Approaches for estimating PUF-air partitions coefficient for semi-volatile organic compounds: a critical comparison. Chemosphere 168, 199–204. https://doi.org/10.1016/j.chemosphere.2016.10.001.

Okeme, J.O., Yang, C., Abdollahi, A., Dhal, S., Harris, S.A., Jantunen, L.M., Tsirlin, D., Diamond, M.L., 2018. Passive air sampling of flame retardants and plasticizers in Canadian homes using PDMS, XAD-coated PDMS and PUF samplers. Environ. Pollut. 239, 109–117. https://doi.org/10.1016/j.envpol.2018.03.103.

Pelletier, M., Glorennec, P., Mandin, C., Le Bot, B., Ramalho, O., Mercier, F., Bonvallot, N., 2018. Chemical-by-chemical and cumulative risk assessment of residential indoor exposure to semivolatile organic compounds in France. Environ. Int. 117, 22–32. https://doi.org/10.1016/j.envint.2018.04.024.

Petrik, J., Drobna, B., Pavuk, M., Jursa, S., Wimmerova, S., Chovancova, J., 2006. Serum PCBs and organochlorine pesticides in Slovakia: age, gender, and residence as determinants of organochlorine concentrations. Chemosphere 65, 410–418. https://doi.org/10.1016/j.chemosphere.2006.02.002.

Rodenburg, L.A., Guo, J., Du, S., Cavallo, G.J., 2010. Evidence for unique and ubiquitous environmental sources of 3,3′-dichlorobiphenyl (PCB 11). Environ. Sci. Technol. 44, 2816–2821. https://doi.org/10.1021/es901155h.

Shin, H.M., McKone, T.E., Bennett, D.H., 2012. Intake fraction for the indoor environment: a tool for prioritizing indoor chemical sources. Environ. Sci. Technol. 46, 10063–10072. https://doi.org/10.1021/es3018286.

Shin, H.-M., McKone, T.E., Tulve, N.S., Clifton, M.S., Bennett, D.H., 2013. Indoor residence times of semivolatile organic compounds: model estimation and field evaluation. Environ. Sci. Technol. 47, 859–867.

Smith, M.N., Grice, J., Cullen, A., Faustman, E.M., 2016. A toxicological framework for the prioritization of children's safe product act data. Int. J. Environ. Res. Public Health 13. https://doi.org/10.3390/ijerph13040431.

Stapleton, H.M., Allen, J.A., Kelly, S.M., Konstantinov, A., Klosterhaus, S., Watkins, D., Mcclean, M.D., Webster, T.F., 2008. Alternate and new brominated flame retardants detected in U. S. House dust. Environ. Sci. Technol. 42, 6910–6916. https://doi.org/10.1021/es801070p.

Stockholm Convention, 2018. Stockholm Convention on Persistent Organic Pollutants [WWW Document]. URL. http://chm.pops.int/, Accessed date: 10 January 2018.

Strémy, M., Šutová, Z., Murínová, Ľ.P., Richterová, D., Wimmerová, S., Čonka, K., Drobná, B., Fábelová, L., Jurečková, D., Jusko, T.A., Tihányi, J., Trnovec, T., 2019. The spatial distribution of congener-specific human PCB concentrations in a PCB-polluted region. Sci. Total Environ. 651, 2292–2303. https://doi.org/10.1016/j.scitotenv.2018.10.123.

Sühring, R., Diamond, M.L., Scheringer, M., Wong, F., Pućko, M., Stern, G., Burt, A., Hung, H., Fellin, P., Li, H., Jantunen, L.M., 2016. Organophosphate esters in Canadian Arctic air: occurrence, levels and trends. Environ. Sci. Technol. 50, 7409–7415. https://doi.org/10.1021/acs.est.6b00365.

Tan, H., Peng, C., Guo, Y., Wang, X., Wu, Y., Chen, D., 2017. Organophosphate flame retardants in house dust from South China and related human exposure risks. Bull. Environ. Contam. Toxicol. 99, 344–349. https://doi.org/10.1007/s00128-017-2120-8.

Taniyasu, S., Kannan, K., Holoubek, I., Ansorgova, A., Horii, Y., Hanari, N., Yamashita, N., Aldous, K.M., 2003. Isomer-specific analysis of chlorinated biphenyls, naphthalenes and dibenzofurans in Delor: polychlorinated biphenyl preparations from the former

Czechoslovakia. Environ. Pollut. 126, 169–178. https://doi.org/10.1016/S0269-7491(03)00207-0.

Tao, F., Abdallah, M.A.-E., Harrad, S., 2016. Emerging and legacy flame retardants in UK indoor air and dust: evidence for replacement of PBDEs by emerging flame retardants? Environ. Sci. Technol. 50, 13052–13061. https://doi.org/10.1021/acs.est.6b02816.

U.S. EPA, 2012. Literature Review of Remediation Methods for PCBs in Buildings.

US Environmental Protection Agency, 2018. US Environmental Protection Agency Chemistry Dashboard. (WWW Document).

U.S. EPA, 2018. US Environmental Protection Agency Chemistry Dashboard, [WWW Document]. [WWW Document]. https://comptox.epa.gov/dashboard, Accessed date: 6 January 2018.

Van Den Eede, N., Dirtu, A.C., Ali, N., Neels, H., Covaci, A., 2012. Multi-residue method for the determination of brominated and organophosphate flame retardants in indoor dust. Talanta 89, 292–300. https://doi.org/10.1016/j.talanta.2011.12.031.

Venier, M., Audy, O., Vojta, Š., Bečanová, J., Romanak, K., Melymuk, L., Krátká, M., Kukučka, P., Okeme, J., Saini, A., Diamond, M.L., Klánová, J., 2016. Brominated flame retardants in the indoor environment - comparative study of indoor contamination from three countries. Environ. Int. 94, 150–160. https://doi.org/10.1016/j.envint.2016.04.029.

Vojta, Š., Melymuk, L., Klánová, J., 2017. Changes in flame retardant and legacy contaminant concentrations in indoor air during building construction, furnishing, and use. Environ. Sci. Technol. 51, 11891–11899. https://doi.org/10.1021/acs.est.7b03245.

Vorkamp, K., 2016. An overlooked environmental issue? A review of the inadvertent formation of PCB-11 and other PCB congeners and their occurrence in consumer products and in the environment. Sci. Total Environ. 541, 1463–1476. https://doi.org/10.1016/j.scitotenv.2015.10.019.

Vykoukalová, M., Venier, M., Vojta, Š., Melymuk, L., Bečanová, J., Romanak, K., Prokeš, R., Okeme, J.O., Saini, A., Diamond, M.L., Klánová, J., 2017. Organophosphate esters flame retardants in the indoor environment. Environ. Int. 106, 97–104. https://doi.org/10.1016/j.envint.2017.05.020.

Wang, J., Guo, L., Li, J., Zhang, G., Lee, C.S.L., Li, X., Jones, K.C., Xiang, Y., Zhong, L., 2007. Passive air sampling of DDT, chlordane and HCB in the Pearl River Delta, South China: implications to regional sources. J. Environ. Monit. 9, 582–588. https://doi.org/10.1039/b700798a.

Wang, W., Wu, F., Zheng, J., Wong, M.H., 2013. Risk assessments of PAHs and Hg exposure via settled house dust and street dust, linking with their correlations in human hair. J. Hazard. Mater. 263, 627–637. https://doi.org/10.1016/j.jhazmat.2013.10.023.

Weis, N., Kohler, M., Zorn, C., 2003. Highly PCB-contaminated schools due to PCB-containing roughcast. In: Health Buildings 2003, Proceedings of the 7th International Conference on Healthy Buildings, International Society of Indoor Air Quality and Climate (ISIAQ). Singapore, December 7–11 2003, pp. 283–288.

Weschler, C.J., 2009. Changes in indoor pollutants since the 1950s. Atmos. Environ. 43, 153–169. https://doi.org/10.1016/j.atmosenv.2008.09.044.

Weschler, C.J., Nazaroff, W.W., 2008. Semivolatile organic compounds in indoor environments. Atmos. Environ. 42, 9018–9040. https://doi.org/10.1016/j.atmosenv.2008.09.052.

Whitehead, T.P., Metayer, C., Petreas, M., Does, M., Buffler, P.A., Rappaport, S.M., 2013. Polycyclic aromatic hydrocarbons in residential dust: sources of variability. Environ. Health Perspect. 121, 543–550. https://doi.org/10.1289/ehp.1205821.

Wimmerová, S., Watson, A., Drobná, B., Šovčíková, E., Weber, R., Lancz, K., Patayová, H., Richterová, D., Koštiaková, V., Jurečková, D., Závacký, P., Strémy, M., Jusko, T.A., Palkovičová Murínová, Ľ., Hertz-Picciotto, I., Trnovec, T., 2015. The spatial distribution of human exposure to PCBs around a former production site in Slovakia. Environ. Sci. Pollut. Res. 22, 14405–14415. https://doi.org/10.1007/s11356-015-5047-9.

World Bank, 2018a. World Bank Country and Lending Groups [WWW Document]. URL. https://datahelpdesk.worldbank.org/knowledgebase/articles/906519-world-bank-country-and-lending-groups, Accessed date: 1 May 2018.

World Bank, 2018b. Agricultural Value Added (% of GDP) [WWW Document]. URL. https://data.worldbank.org/indicator/NV.AGR.TOTL.ZS, Accessed date: 1 May 2018.

Yadav, I.C., Devi, N.L., Li, J., Zhang, G., 2018. Polycyclic aromatic hydrocarbons in house dust and surface soil in major urban regions of Nepal: implication on source apportionment and toxicological effect. Sci. Total Environ. 616–617, 223–235. https://doi.org/10.1016/j.scitotenv.2017.10.313.

Yang, C., Harris, S.A., Jantunen, L.M., Siddique, S., Kubwabo, C., Tsirlin, D., Latifovic, L., Fraser, B., St-Jean, M., De La Campa, R., You, H., Kulka, R., Diamond, M.L., 2019. Are cell phones an indicator of personal exposure to organophosphate flame retardants and plasticizers? Environ. Int. 122, 104–116. https://doi.org/10.1016/j.envint.2018.10.021.

Zhang, X., Diamond, M.L., Ibarra, C., Harrad, S., 2009. Multimedia modeling of polybrominated diphenyl ether emissions and fate indoors. Environ. Sci. Technol. 43, 2845–2850. https://doi.org/10.1021/es802172a.

Zhang, X., Diamond, M.L., Robson, M., Harrad, S., 2011. Sources, emissions, and fate of polybrominated diphenyl ethers and polychlorinated biphenyls indoors in Toronto, Canada. Environ. Sci. Technol. 45, 3268–3274. https://doi.org/10.1021/es102767g.

Zhang, X., Sühring, R., Serodio, D., Bonnell, M., Sundin, N., Diamond, M.L., 2016. Novel flame retardants: estimating the physical-chemical properties and environmental fate of 94 halogenated and organophosphate PBDE replacements. Chemosphere 144, 2401–2407. https://doi.org/10.1016/j.chemosphere.2015.11.017.

Zheng, X., Qiao, L., Covaci, A., Sun, R., Guo, H., Zheng, J., Luo, X., Xie, Q., Mai, B., 2017. Brominated and phosphate flame retardants (FRs) in indoor dust from different microenvironments: implications for human exposure via dust ingestion and dermal contact. Chemosphere 184, 185–191. https://doi.org/10.1016/j.chemosphere.2017.05.167.

APPENDIX 8

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Organochlorine pesticides in the indoor air of a theatre and museum in the Czech Republic: Inhalation exposure and cancer risk

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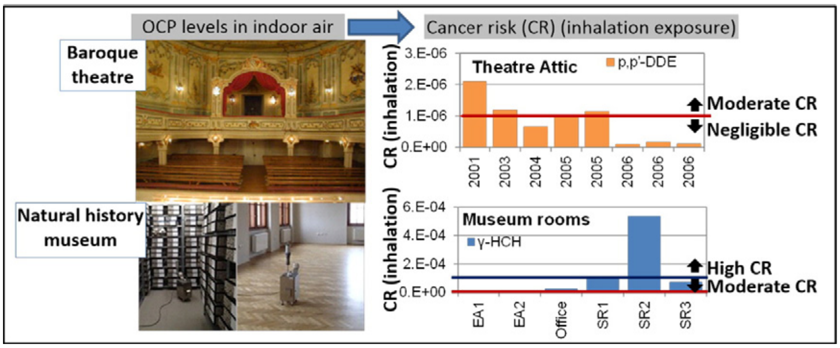
HIGHLIGHTS

- Indoor air in buildings of cultural and historical importance had high OCP levels
- Cancer risk (CR) from inhalation exposure to OCPs determined for workers
- CR from DDE exposure reduced by orders of magnitude after theatre restoration
- CR for museum workers from γ-HCH exposure were significant as to require mitigation
- Museum worker CR was reduced, but still moderate where γ-HCH granules were removed

GRAPHICAL ABSTRACT



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ABSTRACT

Organochlorine pesticides (OCPs) have been used to preserve the integrity of historical buildings or to protect collections of artefacts at potentially large volumes and often without detailed application records. Previous research has focused on the efficiency of remediation at contaminated sites (where identified), as well as improvement of preservation techniques and workplace health and safety. Few studies have assessed the human health risks from occupational exposure to OCPs in buildings of cultural and historical importance. Thus, potential risks may remain unidentified. In the present study, OCPs in indoor air were measured in a baroque theatre and a natural history museum in the Czech Republic, both of which had suspected past indoor application. In the theatre attic *p,p′*-dichlorodiphenyldichloroethylene (*p,p′*-DDE) levels in air were up to 190 ng m^{-3}, confirming past indoor use of *p,p′*-dichlorodiphenyltrichloroethane (*p,p′*-DDT). There was also evidence of γ-hexachlorocyclohexane (γ-HCH) use in the theatre (max γ-HCH in air of 56 ng m^{-3}). Yet, the cancer risk (CR) from occupational exposure via inhalation (Exp_i) to OCPs in the theatre was low ($CR < 4.0 \times 10^{-6}$). γ-HCH was found at elevated levels in air of the museum (max γ-HCH in air of 15,000 ng m^{-3}). CR from Exp_i in the museum was moderate to high ($>1 \times 10^{-4}$). Our results show the CR through Exp_i to OCPs in buildings, such as museums can still be significant enough to warrant mitigation measures, e.g., remediation.

1. Introduction

Inorganic and organic pesticides are regularly used as a protectant for wood used in building materials, e.g., impregnated into wood or used as a coating for wooden building materials (Unger et al., 2001). In the past, these wood protectant pesticides included organochlorine pesticides (OCPs) (e.g., γ-hexachlorocyclohexane (γ-HCH)), pentachlorophenol, dichlorodiphenyltrichloroethane (DDT) (Unger et al., 2001), most of which have since been replaced by other pesticides (e.g., synthetic pyrethroids) (Unger et al., 2001). The use of pesticides was also common practice amongst collectors of artefacts, biological specimens,

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etc. to prevent damage from pests, such as insects or fungi (Palmer et al., 2003; Unger et al., 2001). A range of pesticides has been detected in artefacts or significant artistic, cultural, historical, or scientific objects, as well as in the buildings (e.g., museums, herbariums) that store such items (Fellowes et al., 2011; Goldberg, 1996; Marcotte et al., 2014; Musshoff et al., 2010; Palmer et al., 2003; Schieweck et al., 2007). However, records of past pesticide application to protect collections from insects and other pests are often limited (Goldberg, 1996; Schieweck et al., 2007). For example, in Germany use of organic and inorganic pesticides in museum collections was largely undocumented or poorly documented prior to the 1970s (Schieweck et al., 2007). Some of the knowledge of the pesticides used is incidental, obtained through word-of-mouth; and the type of pesticides used varied according to the type of collection, efficacy of the pesticide, as well as human health and safety (Goldberg, 1996; Schieweck et al., 2007). Determination of pesticide use in artefacts or for the treatment of wooden building materials is therefore difficult.

Levels of pesticides in museum objects can be high due to repeated application (Wörle et al., 2012). In one case, objects shown in a museum were so heavily treated with DDT they emitted a strong odour, and decontamination methods for the artefacts was sought so they were safer to display (Wörle et al., 2012). Yet, evaluation of health outcomes from exposure to pesticides through indoor air in buildings housing important artefacts is rare, despite the potential for serious health effects. In the past, pesticides were applied by museum staff and hence exposure to the pesticides may have occurred during application, leading to acute effects such as nausea (Musshoff et al., 2010). Museum staff may also be exposed to the pesticides during their everyday work, which may include handling or cleaning contaminated objects (Glastrup, 2000; Marcotte et al., 2014; Musshoff et al., 2010; Schieweck et al., 2007).

Dermal uptake and ingestion through handling contaminated objects or contact with contaminated dust, as well as inhalation of particle-associated or volatile compounds are expected exposure routes in indoor environments such as museums (Covaci et al., 2006; Glastrup, 2000; Purewal, 2000; Schieweck et al., 2007). Exposure via dermal uptake or inhalation of particles or gas-phase compounds may also occur from museum artefacts used for ceremonial purposes (Palmer et al., 2003). Marcotte et al. (2014) assessed the daily inhalation and ingestion exposure in The Natural History Museum of Rouen, and neither exposure pathway was found to be significant in the short-term (i.e., inhalation and ingestion exposures were orders of magnitude less than occupational guidelines and permissible uptake limits, respectively). However, to date no other studies have assessed multiple exposure pathways (dermal uptake, inhalation and ingestion) to OCPs for workers or visitors in museums, or in historical buildings where people may work or visit, such as theatres or castles. In addition, long-term exposure in these indoor environments has been neglected. Therefore, the relative importance of these different exposure pathways in such environments is unknown.

The main objective of the present study was to assess potential risks related to pesticide exposure via inhalation exposure in a largely unexplored occupational setting in the Czech Republic – buildings of cultural and historical importance, specifically a theatre and a museum. In the Czech Republic, OCPs were used extensively in agriculture in the past and their volatilization from secondary sources such as contaminated soils into the ambient air has been well documented (Beránek and Havel, 2006; Beránek and Petrlík, 2005; Dvorska et al., 2008; Holoubek et al., 2007). Technical HCH (mixture of HCH isomers) and γ-HCH (also known as lindane) were also used for wood treatment (Beránek and Havel, 2006), which may be a source of this compound in indoor air (Kohoutek et al., 2005). Formulations such as Pentalidol (containing 2% DDT but also 5% pentachlorophenol (PCP) and 0.1% γ-HCH) were produced and used in the former Czechoslovakia for the treatment of different kinds of wood, construction material, furniture, floors and roofs against insects, fungi and mold in the late 1970s and early 1980s (Beránek and Havel, 2006; Kohoutek et al., 2005). Yet, there is no information available in the published literature on the use of OCPs in historical buildings of cultural significance or in museum collections in the Czech Republic. This is despite evidence of OCP use and residues in museum air and artefacts in other regions (Fellowes et al., 2011; Goldberg, 1996; Marcotte et al., 2014; Musshoff et al., 2010; Palmer et al., 2003; Schieweck et al., 2007).

We have investigated the levels of selected organochlorine pesticides (*p,p′*-DDT and its metabolites *p,p′*-dichlorodiphenyldichloroethylene (*p,p′*-DDE) and *p,p′*-dichlorodiphenyltrichloroethane (*p,p′*-DDD) and HCH isomers γ-, α- β- and δ-HCH), as well as industrial chemicals (hexachlorobenzene (HCB) and pentachlorobenzene (PeCB)) in a baroque theatre and a natural history museum. In the theatre, OCPs were used for preservation of the wooden structure of the building itself, and in the museum they were applied to protect historical collections. Air quality data were used to estimate occupational inhalation exposure and risk to the health (i.e., cancer risk (CR)) of workers at the baroque theatre and at the museum.

2. Methods

2.1. Sampling sites

2.1.1. Baroque theatre, Český Krumlov

Air sampling sites were located in the stage area and attic of the theatre (see Fig. S1). The site history is described in detail in Kohoutek et al. (2006, 2005). In brief, in the 1950s, wooden materials in the theatre were affected by dry rot and between 1966 and 1997, the theatre was closed to public due to restorations. During this time various fungicides, insecticides and conservation treatments were applied, but the type of pesticide used, the application volume, frequency, and technique was not well documented. Pentalidol (2% DDT; 5% PCP and 0.1% γ-HCH) use was documented since 1980, but this pesticide was most likely used earlier at the site; and Pentalidol use was restricted in 1983. It is suspected that pesticide application occurred on dusty surfaces in the theatre thus creating a large volume of heavily polluted mobile particles, which resulted in contamination of the theatre with a complex mixture of chemicals. In the early 1990s, there were reports of eye, membrane and skin irritation from people who had spent time in the theatre. Consequently, ~5 t of contaminated material, including contaminated dust, was removed from the theatre but acute health issues were still being reported when visitors returned. A sampling campaign occurred in summer 2001 to isolate contamination sources. Consequently, wooden floors, tarred cardboard and insulation were removed from the attic between 2002 and 2003, and the attic floor was covered with plastic to prevent particles from penetrating into the auditorium of the theatre. In summer 2003, wooden beams in the attic were tested to determine whether the white surface layers required removal. Trials carried out in spring 2006 showed the most appropriate procedure to remove the whitish surface layer of the timber, and reduce contamination, was dry planing and subsequent sanding of corners and other hard-to-access places. Planing and sanding was coupled with a high quality exhaust system to capture and collect dust. Dry planing allowed for the preservation of the authentic shape of the original material. However, this remediation was never completed. Results of building material and air sampling discussed below span from 2001, when acute health issues from past pesticide use were apparent, to 2006, a few years after site restoration and remediation trials. No further analysis of air or building materials occurred in the theatre since 2006. Given the persistent nature of the OCPs analyzed in the present study, as well as the fact that no further building works or site remediation have occurred in the theatre, it is expected that levels of OCPs are still high in indoor air.

2.1.2. Museum of South Bohemia, České Budějovice

The Museum of South Bohemia, located in České Budějovice, is an important historical building, part of which had been subject to interior renovations at the time of sampling. As with the theatre, records of past pesticide use in the museum are poor. Rooms in two different buildings (A and B), were sampled using the methods described below in Section 2.2.2. In the botanical and zoological sciences storage areas (storage rooms 1, 2 and 3) of building A some plant and animals specimens were treated with γ-HCH (Prokeš et al., 2015). Granulated γ-HCH was applied in drawers of specimens up to 40 years ago and visible signs of this application remained up to and including at the time of sampling in at least one of the storage rooms (storage room 2) (Prokeš et al., 2015). γ-HCH applied in storage rooms 1 and 3, building A had been removed prior to sampling, but residues of past application of this pesticide were expected to remain (Prokeš et al., 2015). γ-HCH was also applied to the floor of an office in the same building (A) in approximately 2011 or 2012, although there are no specific details on the application process and rate (Prokeš et al., 2015). There may also have been some past application of γ-HCH in exhibition room 1 of building B but there are no details or of when this occurred or the application rate and process. There was no history of pesticide use in exhibition room 2 of building B. For further details of the site rooms refer to Table S1 and Fig. S2 in the Supplementary data (SD).

2.2. Sampling

2.2.1. Baroque theatre, Český Krumlov

The original aim of indoor air monitoring in this theatre was to investigate the source of OCPs (2001 sampling). In 2002 to 2003, air and building material were monitored to inform restoration works and to deal with the acute health issues of theatre visitors. Air monitoring in 2004 to 2006 was to assess how OCP levels changed after restoration and determine whether remediation was required for OCP embedded in building materials (trial remediation occurred in spring of 2005). Indoor temperature and humidity were recorded alongside collection of air samples. Samples of building materials were also collected to determine the potential source of pesticides in indoor air.

Air sampling was carried out based on internationally recognized methods (Kohoutek et al., 2005). In brief, air was sampled using the US EPA test method TO-13 (US EPA, 1988) and a high volume active air sampler (HV AAS (Fig. S2, SD); PS-1 (Thermo, USA)) (Kohoutek et al., 2005). The particle phase was captured on quartz fiber filters (QFF) (Whatman, USA), and the gas phase was collected on polyurethane foam (PUF) plugs. Sampling was carried out in parallel in the stage, backstage and auditorium area, as well as in the attic in July 2001, August 2003 and 2004, September 2005 and August 2006. Each AAS sampling campaign consisted of one, two or three consecutive periods of approximately 12 h, each time collecting volumes of approximately 200 m^3. Details of air samples (room and year) are included in Table 2. In 2001, there were two consecutive samples taken in the stage area, and one air sample each was taken in the auditorium, upper ropes and attic areas of the theatre. One air sample was taken per year in the stage and attic areas of the theatre in 2003 and 2004. In 2005, two consecutive air samples were taken in the stage area and attic, and in 2006, three consecutive air samples were taken in the stage area and attic of the theatre. The PUF plugs and quartz filters used in air sampling were collected, wrapped in two layers of aluminum foil, labeled, sealed in polythene bags and transported to the laboratory at a temperature of approximately 5 °C. In the laboratory, the samples were stored at −18 °C before processing.

Temperature and humidity are key factors likely to influence the intensity of secondary emissions of target compounds from any potentially contaminated building materials in the baroque theatre and their redistribution through air between rooms in gaseous form. Therefore, all sampling was carried out in the same season, summer, when there were days with longer periods of sunshine and higher temperatures of the air inside and outside the theatre than at other times of the year.

Twenty-six grab samples of building materials such as wood (shaved or scraped), plaster and insulation were taken from the attic, stage area and auditorium of the theatre from 2001 to 2006. Samples were taken with the assistance of theatre staff to help determine the source of compounds identified in indoor air. Solid building material samples were taken from the surface layer (scraped off to a depth of 1 mm). All samples were wrapped in foil, placed in Ziploc bags and transported to the laboratory at a temperature of approximately 5 °C. In the laboratory, the samples were stored at −18 °C before processing.

2.2.2. Museum of South Bohemia, České Budějovice

The main aim of the sampling campaign was to measure the levels of γ-HCH and a number of other OCPs in indoor air. At the museum, air sampling was carried out using a medium volume (MV) AAS (Leckel MVS6, Sven Leckel, Germany), over a two-day period at six sites within the museum offices, collection storage area and public access areas/exhibition areas. Sampling times for AAS were based on the volumes of individual rooms, i.e., 40 h for a smaller room and 50 h for a larger room. Sampling was carried out in September 2014. No additional air sampling has been carried out in the museum since 2014. Yet, given the persistent nature of the OCPs sampled, and continued presence of the lindane granules, it is not expected that levels of these compounds in the museum air have changed significantly in the past three years.

In total there were six samples taken (one per room) in the museum. Active sampling was carried out according to the method outlined in Czech technical standards 16000-01, 16000-12 and 16000-32. The sampler collected particles <10 μm (PM10). The AAS sampler was fitted with a QFF (Whatman, QMA, United Kingdom) for collecting the particle-bound compounds and two PUF plugs for sampling gas-phase compounds (Molitan a.s., Czech Republic). The flow rate of the sampler was 2.3 $m^3\ h^{-1}$. PUF plugs were pre-cleaned by hot Soxhlet extraction for 8 h in acetone followed by 8 h in dichloromethane.

Samples were collected, transported to the laboratory, and stored prior to analysis in the same manner as those from the baroque theatre.

2.3. Sample processing, clean-up and analysis

2.3.1. Baroque theatre, Český Krumlov and Museum of South Bohemia, České Budějovice

Analyses of target OCPs ((a) α-, β-, γ-, δ- and ε-HCH, (b) *o,p′*-DDT and *p,p′*-DDT, *o,p′*-DDE, *p,p′*-DDE, *o,p′*-DDD and *p,p′*-DDD and (c) HCB and PeCB) was carried out in a similar manner to analysis described in Lohmann et al. (2012). In brief, all sample media (QFFs, PUF plugs, and 5 g of solid matrices) were extracted with 120 mL of dichloromethane in a Büchi System B-811 automatic extractor (Büchi Labortechnik AG, Switzerland) for 40 min followed by a 20-minute thimble wash and condensation. AAS (QFF and PUF plug) were spiked with recovery standards, PCB 30 and 185 (Wellington Laboratories Inc., Canada) prior to extraction. Sample extract volumes were reduced under a gentle nitrogen stream at ambient temperature. The extracts were subject to clean-up on a H_2SO_4 modified silica gel column (44% w/w) (eluted with 30 mL of 1:1 hexane/dichloromethane). Analytes were eluted from the column with a mixture of *n*-hexane and dichloromethane (1:1). Extracts were concentrated under a gentle stream of nitrogen, transferred to nonane and an internal standard of PCB 121 (Wellington Laboratories Inc., Canada) was added.

Baroque theatre solid and air samples were analyzed using a gas chromatograph (HP 6890) coupled with a mass spectrometer (HP 5975) in selected ion monitoring (SIM) mode. The separation took place on a J&W Scientific fused silica column (DB-5MS; 60 m × 0.25 mm × 0.25 μm) with helium carrier gas at a flow of 1.5 mL min^{-1}. Sample volumes of 1 μL were injected in splitless mode at 280 °C^{-1}. The GC temperature program was as follows: 80 °C (hold for 1 min), 15 °C min^{-1} to 180 °C, 5 °C min^{-1} to 310 °C (10 min). Analysis for the Museum of South Bohemia air samples was carried out using gas chromatography with tandem mass spectroscopy (GC–MS/MS; Agilent GC 7890/MS-MS Triple Quadrupole 7000B) in multiple reaction monitoring (MRM) mode. The separation took place on a SGE Analytical

Table 1
Input values for the inhalation exposure calculation for workers in the theatre and museum.

Parameter	Value	Description
CA	See Table 2 and Table 4	Lower bound values (i.e., 0) where air concentrations were <LOQ
ET	10 h day^{-1}	It was estimated people spend 8 to 10 h a day at work and the more conservative value was used for the calculation. Percentage of the day worked is 42%.
EF	241 days $year^{-1}$	It was assumed 4 weeks holiday were taken by workers and work was undertaken from Monday to Friday (i.e., 5 days a week).
ED	30 years	The exposure duration was defined as 30 years, which is the reasonable maximum exposure for an adult resident and is based on an established hazardous waste facility with a useful lifetime of 30 years. 30 years was assumed to represent a reasonable maximum period of employment and time spent by a worker in the theatre or museum (US EPA, 2005a).
AT	613,200 h	The averaging time (AT) is the period over which exposure is averaged. AT was estimated to be lifetime in years × 365 days/year × 24 h/day, where lifetime is the average lifetime for chemicals of potential concern for carcinogenicity of 70 years (US EPA, 2009).

Science HT-8 column (60 m × 0.25 mm internal diameter × 0.25 μm; 8% phenyl polycarbonate-siloxane) with helium as a carrier gas at a flow rate of 1.5 mL min^{-1}. The GC temperature program was as follows: 80 °C (hold for 1.5 min), 40 °C min^{-1} to 200 °C (hold for 18 min), 5 °C min^{-1} to 305 °C. Splitless injection method was executed at 280 °C.

2.4. Quality assurance/quality control (QA/QC)

2.4.1. Baroque theatre, Český Krumlov and Museum of South Bohemia, České Budějovice

Compounds were identified and quantified with an eight point calibration of all target compounds (i.e., at concentrations of 1, 4, 10, 40, 100, 400, 1000 and 4000 ng mL^{-1}). Linearity was achieved for the entire concentration range. The OCP analytical method for AAS has been previously evaluated using a certified reference material (ASLAB soil standard, Czech Republic) (Lohmann et al., 2012). Recovery of all native OCPs from the reference material ranged from 75 to 98% at the time of sample analysis for the theatre and the museum.

Recoveries for the baroque theatre air and solid samples, based on PCB 30 and 185 added prior to extraction, were >71%. For the Museum of South Bohemia the recoveries ranged from 47 to 84% for PCB 30 and 76 to 104% for PCB 185. Recovery factors were not applied to any of the data from either the theatre or the museum, i.e., no correction was made to the results for recovery.

One trip blank and one solvent blank were analyzed with each batch of air samples (2 to 6 samples per batch) from the theatre and the museum; and one solvent blank was analyzed for each batch of solid samples (2 to 6 samples per batch) from the theatre. Field blanks were extracted and analyzed in the same way as the samples, and the levels of OCPs in field blanks never exceeded 1% of the quantities detected in samples from either the theatre or museum for any OCPs, indicating minimal contamination during the transport, storage, and analysis. As a result, no blank corrections were performed on the data. Limits of quantification (LOQ) were determined using the lowest external calibration point. The LOQ was defined as the level at which signal-to-noise (S/N) ratio was 10:1 (Peak-to-Peak).

2.5. Human health risk assessment

2.5.1. Cancer risk assessment

Cancer risk (CR) was calculated for workers potentially exposed to α-HCH, γ-HCH, DDT, DDD, DDE[1] and HCB in the museum and theatre through inhalation. Occupational exposure concentration was calculated using the following equation (US EPA, 2009),

$$EC = (CA \times ET \times EF \times ED)/AT \qquad (1)$$

Where, EC is the exposure concentration (μg m^{-3}), CA is contaminant concentration in air (μg m^{-3}), ET is exposure time (hours day^{-1}), EF is exposure frequency (days $year^{-1}$), ED is exposure duration (year), and AT is averaging time (hours). The values for each of these parameters are included in Table 1.

The estimated occupational exposure via inhalation was multiplied by the appropriate toxicity factor - Inhalation Unit Risk Factor (URFi) or Inhalation Unit Risk (IUR; $(\mu g\ m^{-3})^{-1}$) in order to derive an estimate of the potential CR associated with exposure,

$$CR = EC \times URFi \text{ or } IUR \qquad (2)$$

IUR for α-HCH and DDT were obtained from the Integrated Risk Information System (IRIS) database (US EPA, 2003) (in Table S2, SD). The IUR for γ-HCH was derived by the California (2005). The URFi values for DDD, DDE and HCB originated from the ACCESS database supplied by the US EPA as part of the Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities (US EPA, 2005b). The procedures used to calculate the URFi or IUR values are described in California (2005); US EPA (2003) and US EPA (2005b).

According to the United States Environmental Protection Agency (US EPA) a CR of $<1 \times 10^{-6}$ (or 1 in one million) is considered to be very low or negligible (US EPA, 1991). A CR of $>1 \times 10^{-4}$ (or 1 in ten thousand) is considered large enough to warrant risk management actions such as mitigation or reduction measures (e.g., remediation) (US EPA, 1991). CR within the range of 1×10^{-4} and 1×10^{-6}, where risk is low to moderate, may not be considered large enough to warrant risk management actions but should be evaluated on a case-by-case basis (US EPA, 1991).

3. Results and discussion

3.1. Levels and patterns of pesticides in air and building materials

3.1.1. Baroque theatre, Český Krumlov

Levels of *p,p′*-DDE and *p,p′*-DDT in air were higher in the attic than in the theatre (the stage area and auditorium) (Table 2). For example, *p,p′*-DDE ranged from 45 ng m^{-3} to 190 ng m^{-3} in the attic and 5.5 ng m^{-3} to 26 ng m^{-3} in the stage and auditorium areas, respectively. The source of the *p,p′*-DDE and *p,p′*-DDT in air of the attic was suspected to be the contaminated materials (e.g., insulation) in the attic (Table 3). Indoor air levels of *p,p′*-DDE, *p,p′*-DDD and *p,p′*-DDT decreased in the attic once insulation, etc. was removed from the attic between 2002 and 2003 (Table 2). There was also a significant negative linear relationship between the air concentrations of these pollutants and year sampled ($p < 0.05$) (Fig. S3). However, the wooden beams coated in white powder and shown to contain high levels of *p,p′*-DDE (untreated beam = 1600 to 27,000 ng g^{-1}; Table 3) and *p,p′*-DDT (untreated beam = 260 to 8600 ng g^{-1}; Table 3) remained in the attic. As a result, relatively high levels of *p,p′*-DDE in particular were still detected in attic air from 2004 to 2006 (DDE: 8.0 to 100 ng g^{-1}; Table 2), particularly in 2005. The use of plastic sheeting in 2003 to prevent particle-associated DDT and DDE entering the theatre does not appear to have reduced

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[1] The *p,p′*-DDT, *p,p′*-DDE and *p,p′*-DDD isomers were measured in air at the baroque theatre and *o,p′*- and *p,p′*-DDT, DDE and DDD isomers were measured in air at the museum.

Table 2
Levels (ng m^{-3}) of selected OCPs in bulk indoor air in the baroque theatre.

Year	2001[a]					2003		2004		2005				2006					
Sample location	Stage 1	Stage 2	Auditorium	Upper ropes	Attic	Stage	Attic	Stage	Attic	Stage 1	Stage 2	Attic 1	Attic 2	Stage 1	Stage 2	Stage 3	Attic 1	Attic 2	Attic 3
α-HCH	NA	NA	NA	NA	NA	0.41	0.16	0.55	0.27	1.1	1.1	1.5	1.1	0.26	0.21	0.32	0.10	0.18	0.13
β-HCH	NA	NA	NA	NA	NA	0.047	0.016	0.032	0.067	0.29	0.23	0.56	0.36	0.036	0.026	0.035	0.012	0.015	0.012
γ-HCH	5.2	25	14	28	17	24	11	12	4.5	37	36	5.2	4.4	6.2	5.5	8.0	2.6	5.2	3.2
δ-HCH	NA	NA	NA	NA	NA	0.038	0.047	<LOQ	<LOQ	0.077	0.065	0.14	0.097	<LOQ	<LOQ	<LOQ	<LOQ	<LOQ	<LOQ
p,p′-DDD	NA	NA	NA	NA	NA	3.1	14	0.55	0.44	2.6	2.8	5.0	4.6	0.31	0.34	0.40	0.22	0.60	0.26
p,p′-DDE	5.5	22	8.5	18	190	20	110	8.9	58	26	12	92	100	6.8	4.7	9.3	8.0	14	9.4
p,p′-DDT	1.1	3.4	1.9	3.1	21	4.6	15	1.2	1.8	2.3	2.6	2.9	2.7	1.6	1.5	2.3	1.2	3.1	1.5
HCB	NA	NA	NA	NA	NA	1.1	0.59	1.8	0.72	1.5	1.8	0.88	0.96	1.2	0.94	1.6	0.68	1.2	0.67
PeCB	NA	NA	NA	NA	NA	0.094	0.022	0.11	0.031	0.47	0.72	0.27	0.28	0.39	0.39	0.43	0.31	0.48	0.20

NA = not available. LOQ = 0.00050 ng m^{-3}.
[a] All values as reported in Kohoutek et al. (2006).

p,p′-DDE and *p,p′*-DDT levels in theatre air and the relationship between air concentrations of these compounds and year sampled was not significant ($p > 0.05$) (Fig. S3).

HCH concentrations in air were generally higher in the stage and auditorium area (5.2 to 37 ng m^{-3}) than the attic (2.6 to 17 ng m^{-3}) (Table 2). The source of the γ-HCH in the indoor air around the stage area was most likely previous indoor application of the pesticide. This was supported by analysis of wood scrapings from the backstage area around the lower machinery and stage areas of the theatre, which had much higher γ-HCH levels than in other materials tested in the building (Table 3). The linear relationship between the air concentrations and time was negative and significant ($p < 0.05$) suggesting that concentrations of γ-HCH in the attic decreased after the removal of building materials (between 2002 and 2003) and laying of plastic sheeting on the attic floor in 2003 (Fig. S3).

PeCB and HCB concentrations in air were similar in the attic, stage and auditorium areas of the theatre. The most likely source of the chlorobenzenes in the indoor air is outdoor air. Most of the HCB present in outdoor air of the Czech Republic is thought to have re-volatilized from past soil contamination (Dvorska et al., 2008). Other potential sources of HCB in the environment of the Czech Republic are the manufacture of chlorinated solvents and pesticides, as well as the application of HCB-contaminated pesticides and inadequate incineration of chlorine-containing wastes (Dvorska et al., 2008).

As expected from past application of DDT, the *p,p′*-DDT metabolites (*p,p′*-DDE and *p,p′*-DDD) dominated relative to *p,p′*-DDT in air and building materials. The average *p,p′*-DDT metabolite contribution to $\sum$ *p,p′*-DDT, *p,p′*-DDE and *p,p′*-DDD in air was 0.84 and was largely unchanged in air over the sampling period from 2001 to 2006 (ranging from 0.68 to 0.98). The HCH fingerprint (contribution of each isomer to the $\sum_4$HCH, i.e., α-, β-, γ- and δ-HCH) was dominated by γ-HCH in the air and the solid building materials in both the stage area and in the theatre attic, with only a few exception where the α- or β-HCH isomers dominated. This suggests that the source of the HCH in the theatre is a γ-HCH formulation rather than technical HCH.

Annual fluctuations in the temperature and humidity of the theatre were expected to affect the air concentrations of OCPs, but this was not evaluated because all sampling occurred at approximately the same time of the year, i.e., summer. At the time of sampling, temperature and humidity in the theatre ranged from 17 °C to 18 °C and 63% to 67%, respectively. Throughout the year in the theatre the temperature and humidity ranged from −0.50 °C to 21 °C and 49% to 82%,

Table 3
Levels (ng g^{-1}) of selected OCPs in solid materials from the baroque theatre[a].

Year	Material	Location	α HCH	β-HCH	γ HCH	δ-HCH	p,p′-DE	p,p′-DDD	p,p′-DDT	HCB	PeCB
2001	Cardboard insulation	A	NA	NA	NA	NA	250	NA	2900	NA	NA
	Fiberglass insulation		NA	NA	NA	NA	100	NA	41	NA	NA
	Wood		NA	NA	NA	NA	580	NA	2700	NA	NA
	Crushed stone		NA	NA	NA	NA	86	NA	380	NA	NA
2003	Non-treated beam, scraping	A	96	6.4	170	3.1	21,000	1100	8600	7.1	NQ
			11	NQ	38	1.5	1600	51	260	1	0.14
	Floor, wood shavings		0.38	NQ	5.0	0.18	290	12	66	0.26	0.13
2004	Lower machinery, scraping	LM	8.4	2	400	NQ	4300	300	1100	11	6.8
	Stage, scraping	S	13	7.9	450	NQ	2900	190	590	14	40
	Upper rope, scraping	UR	16	NQ	NQ	NQ	3600	510	1900	29	65
	Non-treated beam, scraping	A	NQ	NQ	54	NQ	8700	280	930	NQ	NQ
2005	Lower machinery, scraping	LM	32	23	490	NQ	3800	690	6400	1.6	NQ
	Stage, scraping	S	27	34	320	NQ	2300	310	1800	0.59	NQ
	Upper rope, scraping	UR	26	94	260	NQ	4100	350	2300	1.1	NQ
	Upper rope, plaster	UR	2.7	18	12	NQ	0.63	34	0.44	NQ	NQ
	Non-treated beam, scraping	A	360	14	150	NQ	18,000	920	1700	13	NQ
		A	590	42	240	NQ	27,000	1200	2400	21	23
	Ceiling boards, scraping	A	0.49	NQ	41	NQ	3200	46	70	2.8	NQ
		A	170	120	130	NQ	5600	100	190	40	20
		A	NQ	NQ	140	NQ	6000	90	170	33	17
		A	12	14	28	NQ	4200	84	590	0.20	NQ
	Plaster	A	4.1	30	11	NQ	610	59	260	0.10	NQ
	Clean beam, scaping	A	NQ	NQ	29	NQ	730	70	160	NQ	NQ
		A	NQ	NQ	23	NQ	590	100	330	NQ	NQ
2006	Clean beam	A	NQ	0.11	4.1	NQ	130	20	81	NQ	0.31
		A	NQ	NQ	11	NQ	180	35	140	NQ	0.15

NA = not available. A = attic, S = stage, LM = lower machinery and UR = upper rope. NQ = not quantified due to interference in analysis.
[a] All values as reported in Kohoutek et al. (2006).

respectively. As the target OCPs are fairly volatile, the levels of p,p′-DDE, p,p′-DDT and γ-HCH measured in the indoor air of the theatre are therefore likely to represent the highest annual levels, i.e., a worst case scenario for the year. Rapid heating during the daytime and high temperatures throughout the day would have allowed volatilization of p,p′-DDT from treated attic beams thus increasing concentrations in air.

Table 4
Levels of selected OCPs (ng m^{-3}) in bulk indoor air in the Museum of South Bohemia.

Compound	Exhibition area 1	Exhibition area 2	Office	Storage room 1	Storage room 2	Storage room 3
α-HCH	1.5	0.078	2.6	7.4	23	18
β-HCH	<0.0013	<0.0013	<0.0016	<0.0013	<0.0016	<0.0013
γ-HCH	200	7.9	630	2800	15,000	1900
δ-HCH	0.018	0.0051	0.033	0.11	0.65	0.23
ε-HCH	<0.0010	<0.0010	0.022	0.058	0.17	0.10
o,p′-DDE	0.0094	0.010	0.011	0.0062	0.0072	0.013
p,p′-DDE	0.041	0.056	0.049	0.026	0.026	0.11
o,p′-DDD	<0.00068	0.0016	<0.00085	0.0020	0.0014	0.0081
p,p′-DDD	<0.00059	<0.00060	<0.00075	<0.00059	<0.00074	<0.00059
o,p′-DDT	0.013	0.016	0.033	0.032	0.033	0.15
p,p′-DDT	0.0031	0.0058	0.017	0.0085	0.0064	0.068
HCB	0.31	0.28	0.29	0.49	0.48	0.32

3.1.2. Museum of South Bohemia, České Budějovice

Organochlorine pesticides were detected in air samples in all rooms of the Museum of South Bohemia (Table 4). Levels of $\sum_5$HCH (α-, β-, γ-, δ- and ε-HCH) in air varied markedly between rooms (average of 3400 ng m^{-3} and range of 8.0 to 15,000 ng m^{-3}) and most of the $\sum_5$HCH was γ-HCH. Levels were highest in the storage rooms where γ-HCH was applied in the past as a granulated formulation for the preservation of the museums botanical and zoological collections, and a significant residue of the applied formulation remains to the present day. γ-HCH levels were also at least 3 times higher in the air in the office compared to the exhibition areas, most likely due to the application of a γ-HCH formulation in the office a few years prior to the air sampling. Average and range of $\sum_6$DDT (o,p′- and p,p′-DDT, DDE and DDD) levels in air (0.13 ng m^{-3} and 0.066–0.35 ng m^{-3}) were orders of magnitude lower than $\sum_5$HCH. There is no record of DDT use in the museum. DDT (sum of o,p′- and p,p′-DDT isomers) was >3 times higher in one of the rooms used to house the museum's collection of zoological specimens (storage room 3) than other rooms of the museum, suggesting that this pesticide may have been used here in the past. However, in general, the source of DDT and its metabolites in the indoor air of the museum is likely to be outdoor air, considering that the levels in most of the rooms are similar. $\sum_6$DDT was dominated by p,p′-DDE in the indoor air of the museum because DDT has not been used for several decades in the Czech Republic. HCB levels were similar in all rooms, indicating that outdoor air was also the source of the indoor contamination of this compound.

3.1.3. Comparison to OCPs in indoor air of other museums and historical buildings

There are limited studies on OCP levels in indoor air of museums (air, dust) and artefacts (Table 5). To the authors' knowledge, there is no information on the levels of OCPs in air of historical buildings such as the baroque theatre (e.g., castles, churches, etc.). In general, levels of DDT and γ-HCH were higher in artefacts or museum air in other studies than in the present study. p,p′-DDT in all materials from the theatre included in the present study were in the range of 41 to 29,000 ng g^{-1} with a median of 590 ng g^{-1}. These p,p′-DDT levels are much lower than DDT levels in fur- and skin-based artefacts from the National Museum of Denmark, where the median levels were 870,000 ng g^{-1} and 370,000 ng g^{-1} before and after cleaning, respectively (Glastrup, 2000) or in Native American artefacts (DDT levels detected up to

Table 5
Levels of OCPs in museum air and dust from museums, and artefacts.

Sample description		Results	Reference
Air	Air measured with AAS in invertebrate, bird and mammal galleries, 2 stockrooms and a laboratory in Natural History Museum, Rouen.	DDT + DDD: <0.5. 0.9, 9.0, <0.5, <0.5 and <0.5 μg m^{-3} γ-HCH: 0.5, 4.8, 32, 1.8, <0.3 and <0.3 μg m^{-3}	(Marcotte et al., 2014)[b]
	Air measured in 2-year period on persons in preservation (PS), maintenance (MS) and inventory (IS) services at the Natural History Museum of Rouen.	DDT + DDD: 546 (PS 1st year), <0.5 (PS 2nd year), 46 (MS 1st year), <0.5 (MS 2nd year) and 10 (IS) μg m^{-3} γ-HCH: 2.8 (PS 1st year), <0.3 (PS 2nd year), 2.1 (MS 1st year), <0.3 (MS 2nd year) and <0.3 (IS) μg m^{-3}	
Dust	Dust samples from the invertebrate, bird and mammal galleries, as well as a stockroom in the Natural History Museum of Rouen.	DDT + DDD: 968, 403, <0.5 and 159 μg g^{-1} γ-HCH: 1.5, 0.7, <0.3 and 0.4 μg g^{-1}	(Marcotte et al., 2014)
	Settled dust of German museum rooms, where wooden sculptures are displayed. Collected by manual wiping.	DDT: <1–100 μg g^{-1} γ-HCH: 5–50 μg g^{-1}	(Schieweck et al., 2007)[a]
	Dust, dirt off exhibits and storage areas were collected from several locations in the Reiss-Engelhorn-Museen, Mannheim in 2006.	DDT, DDE and DDD detected. Levels not reported	(Musshoff et al., 2010)[b]
Artefact/object	Fur/skin scraped or cut off 16 artefacts (1983) kept at the National Museum of Denmark. Artefacts cleaned with compressed air.	DDT range and median: <39–5100 and 870 μg g^{-1} (before clean). <39–7100 and 370 μg g^{-1} (after clean)	(Glastrup, 2000)[b]
	Biocide concentrations in the diorama (soil) of the Lower Saxony State Museum, Hanover.	DDT: <1–10 μg g^{-1} γ-HCH: <1–4 μg g^{-1}	(Schieweck et al., 2007)[b]
	Artefacts from (a) Lower Saxony State Museum, Hanover, (b) an unknown museum and (c) a museum in southwest Germany	DDT: not detected – 21 μg g^{-1} γ-HCH: <1–38 μg g^{-1}	
	Museum objects repatriated to the Hupa Tribe of California	DDT: not detected – 130 μg g^{-1} γ-HCH: not detected – 30 μg g^{-1}	(Palmer et al., 2003)[b]
	Extract of unknown museum object.	DDT detected. Not quantified	(Palmer, 2000)[b]
	Stores and collection dust and object samples in 1987.	DDT detected in all samples. Levels not reported	(Schmidt, 2000)[b]
	Artefact - bear paw pouch	DDT detected. Levels not reported	(Sirois, 2000)[b]
	Artefact - fly catcher	γ-HCH detected. Levels not reported	
	Wooden printing blocks from the Plantin-Moretus Museum, Antwerp. Objects were treated circa 1950.	Detected in all 10 samples. Levels not reported Not detected in all 10 samples	(Covaci et al., 2006)[b]
	Wooden sculptures and masks from the Ethnographic Museum, Antwerp. Objects were treated circa 1950.	DDT not detected in all 7 samples γ-HCH not detected in all 7 samples	
	Walnut veneered spruce cabinet from Swiss National Museum.	DDT crystals observed. DDT levels not quantified.	(Wörle et al., 2012)
	Washing/abrasion from 2 artefacts from the Reiss-Engelhorn-Museen, Mannheim in 2006.	DDT, DDE and DDD detected. Levels not reported	(Musshoff et al., 2010)[b]

[a] The DDT values represent the sum of isomers and derivatives (o,p′-DDD, p,p′-DDD, o,p′-DDE, p,p′-DDE, o,p′-DDT, and p,p′-DDT).

[b] Note that the isomers of DDT, DDE or DDD analyzed were not identified for these studies.

13,000,000 ng g^{-1}) (Palmer et al., 2003). Yet, *p,p′*-DDT in theatre attic beam scrapings (this study) are similar to soil from inside a diorama and artefacts from German museums, where levels of DDT ranged from <1000 to 38,000 ng g^{-1} (Schieweck et al., 2007). Levels of γ-HCH, *p,p′*-DDT and *p,p′*-DDD in the theatre air measured in the present study ranged from 3.0 to 56 ng m^{-3}, 1.0 to 25 ng m^{-3} and 3.0 to 14 ng m^{-3}, respectively. Therefore, the OCPs in the baroque theatre were well below those measured in air of the Natural History Museum of Rouen, where levels of γ-HCH were up to 32,000 ng m^{-3} and levels of DDT + DDD were up to 9000 ng m^{-3} (Marcotte et al., 2014). Levels of γ-HCH in the storage rooms of the Museum of South Bohemia (median = 1300 ng m^{-3}; this study) and the galleries of the Natural History Museum of Rouen (median = 3300 ng m^{-3} (Marcotte et al., 2014)) were similar.

3.1.4. Comparison to OCPs in non-impacted outdoor air in the Czech Republic

Indoor air from the present study was compared with OCPs measured in outdoor air at the Czech European Monitoring and Evaluation Program (EMEP) site, the Košetice Observatory, situated in a largely rural area in central Czech Republic (Table 6) (Dvorská et al., 2009; Dvorska et al., 2008; Holoubek et al., 2007; Holt et al., 2017). Levels of some OCPs in the indoor air of the theatre and museum were orders of magnitude higher than the outdoor air at the background site in the Czech Republic. Average *p,p′*-DDT, *p,p′*-DDE, *p,p′*-DDD in the air of the stage area and attic were at least 3 orders of magnitude higher than the average background outdoor air levels in the Czech Republic, even after building restoration and dust removal to reduce indoor air contamination. γ-HCH was also significantly higher (2 orders of magnitude) in the air of the theatre than in outdoor air. γ-HCH in the indoor air of the museum was between 100 and 1 × 10^5 fold the outdoor air levels.

There were, however, some similarities between indoor air levels of OCPs in the theatre and museum, and background outdoor air in the Czech Republic, supporting the hypothesis of outdoor air as a potential source of some OCPs in indoor air. PeCB levels in the museum air and HCB levels in both the theatre and museum air were generally within the same range or slightly above the levels in background outdoor air. β-HCH in the theatre and museum also likely originated from outdoor air. In contrast with the theatre, levels of *p,p′*-DDT and its metabolites in the museum air were also in the range of background outdoor air. This suggests that for the OCPs not used in indoor preservation, outdoor air is the primary source of indoor air contamination.

Table 6
Levels of OCPs in air (ng m^{-3}; measured using AAS), Košetice observatory from 2003 to 2012 (Data provided by the RECETOX laboratory upon request. Košetice air monitoring data have previously been published in Dvorská et al. (2009); Dvorska et al. (2008); Holoubek et al. (2007); Holt et al. (2017)).

Compound	Min	Max	Mean	Median
α-HCH	<LOQ[a]	0.12	0.015	0.011
β-HCH	<LOQ[a]	0.096	0.0086	0.0058
γ-HCH	<LOQ[a]	0.13	0.022	0.017
p,p′-DDE	0.0014	0.064	0.018	0.016
p,p′-DDD	<LOQ[a]	0.026	0.0023	0.0012
p,p′-DDT	<LOQ[a]	0.045	0.0050	0.0033
PeCB	<LOQ[a]	0.078	0.011	0.0086
HCB	<LOQ[a]	0.50	0.093	0.067

[a] From 2003 to 2010 average LOQ for all compounds (0.00050 ng m^{-3}) was reported by the RECETOX laboratory. From 2011 laboratory reporting changed to a range of LOQ values for each compound: α-HCH (0.000030–0.0010 ng m^{-3}), β-HCH (0.000030–0.0017 ng m^{-3}), γ-HCH (0.000030–0.0022 ng m^{-3}), *p,p′*-DDD (0.000030–0.0015 ng m^{-3}), *p,p′*-DDE (0.000030–0.00067 ng m^{-3}), *p,p′*-DDT (0.000030–0.0028 ng m^{-3}), PeCB (0.000030–0.00050 ng m^{-3}) and HCB (0.000030–0.00062 ng m^{-3}).

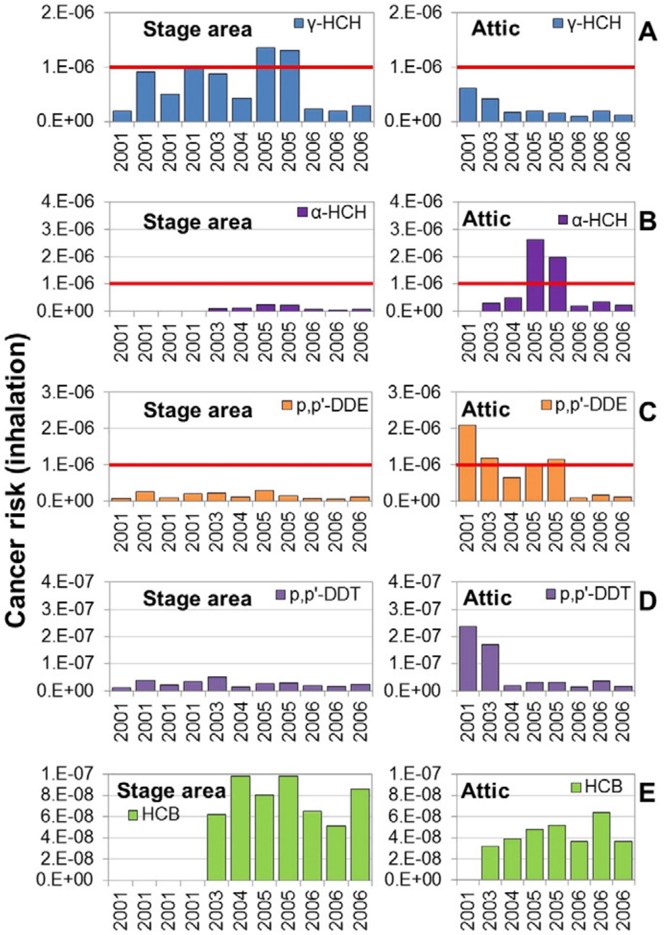

Fig. 1. Estimated CR for employees from exposure to (a) γ-HCH, (b) α-HCH, (c) *p,p′*-DDE, (d) *p,p′*-DDT and (e) HCB compounds in indoor air of the baroque theatre, Český Krumlov. *y*-axes are on a logarithmic scale. Below the red line (1 × 10^{-6}) risk is considered very low or negligible and above the red line the risk is considered moderate and should be subject to case specific evaluation (US EPA, 1991). Missing values from 2001 are where air concentrations were not available.

3.2. Inhalation cancer risk

3.2.1. Baroque theatre, Český Krumlov

The CR from exposure through inhalation to OCPs, particularly *p,p′*-DDE from past treatment of wood in the theatre, was estimated (Fig. 1). The CR for employees was negligible (<1 in 1000,000) to low (between 1 in 100,000 and 1 in 1000,000) for *p,p′*-DDE prior to theatre restoration (2002 to 2003), with CR depending on the location of the building, i.e., CR was higher in the attic. CR decreased to negligible in the attic after site restoration, which involved the removal of contaminated building materials including insulation. γ-HCH and HCB were also estimated to pose a low risk to theatre workers. γ-HCH was also lower after theatre restoration despite the fact that higher levels were found in wood from the stage area rather than materials that were eventually removed from the attic.

3.2.2. Museum of South Bohemia, České Budějovice

The CR from occupational inhalation exposure to γ- and α-HCH, *p,p′*-DDE and HCB (Fig. 2) was estimated using air monitoring data from the Museum of South Bohemia. CR from γ-HCH in the storage rooms (CR > 1 × 10^{-4}) is substantial enough to warrant some form of action to reduce or eliminate risks (e.g., remediation), according to the US EPA classification for contaminated sites (US EPA, 1991). There also appears to be higher CR from HCB (Fig. 2E), which is from outdoor rather than indoor

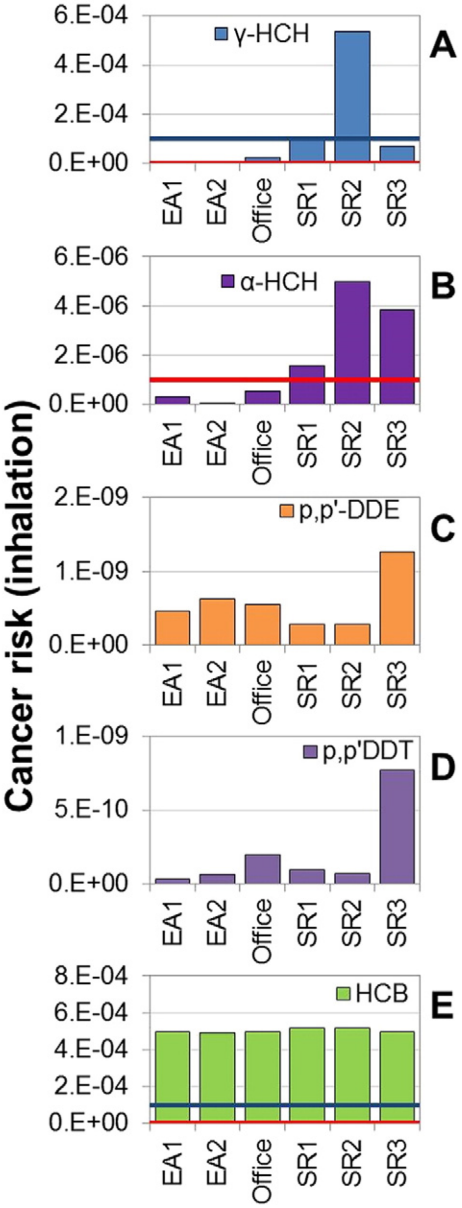

Fig. 2. Estimated CR through inhalation exposure to (a) γ-HCH, (b) α-HCH, (c) *p,p′*-DDE and (d) HCB in indoor air of the Museum of South Bohemia. *y*-axis is on a logarithmic scale. Below the red line (1×10^{-6}) risk is considered very low or negligible and above blue line (1×10^{-4}) risk may require mitigation measures to reduce risk; between red and blue lines the risk is considered moderate and should be subject to case-specific evaluation (US EPA, 1991).

sources. The CR from HCB is ~5×10^{-4} in all rooms where air sampling occurred, and is high enough to prompt actions to reduce or eliminate potential risks. CR for α-HCH ranges between 1×10^{-4} and 1×10^{-6} (moderate risk) in the storage rooms. Therefore CR for α-HCH may not be large enough to warrant risk management actions but should be evaluated on a case-by-case basis (US EPA, 1991). CR for *p,p′*-DDE and *p,p′*-DDT was negligible ($CR << 1 \times 10^{-6}$).

3.2.3. Human health risk (cancer risk) assessment results in context

Few other studies have investigated exposure to OCPs in similar indoor environments. Daily (short-term) inhalation exposure to γ-HCH, DDT and DDD was determined for Natural History Museum of Rouen staff and was orders of magnitude lower than occupational exposure guidelines (Marcotte et al., 2014). Yet, no studies have investigated potential long-term exposure and quantitatively estimated long-term risk in museums or historical buildings such as the baroque theatre. Therefore, direct comparisons of long-term health outcomes were not possible.

However, while short-term exposure to high levels of OCPs may be insignificant or low risk for human health, prolonged exposure to OCPs, even at lower levels can be a significant risk for human health. The exposure for staff at the Natural History Museum of Rouen was predicted to be insignificant despite the fact that levels of γ-HCH and DDT and DDD were high in some rooms (up to 32,000 ng m^{-3} and 9000 ng m^{-3}, respectively) (Marcotte et al., 2014). In contrast, at the Museum of South Bohemia (present study) long-term health risks from inhalation exposure to γ-HCH (15,000 ng m^{-3}) was high.

While the present study quantified levels of OCPs in indoor air, and thus focused on inhalation exposure, occupational exposure via ingestion of dust particles and dermal contact with contaminated surfaces could also be significant in both the theatre and museum, and would have to be separately estimated to determine overall occupational exposure.

4. Conclusions

While OCPs such as DDT, γ-HCH or technical HCH are no longer applied in many countries, apart from specifically controlled uses, the legacy of previous application of these compounds is that their release in indoor environments can continue due to the large volumes used in the past and their persistence in the environment. Levels of OCPs in air, dust, and building materials, as well as on or embedded in artefacts are poorly characterized for settings such as buildings of cultural and historical significance. Such buildings can to be hotspots for OCP contamination but the extent of the problem is largely unknown. To date there is very little information on human occupational exposure and the potential human health risk from past use of OCPs in museum collections or in historical buildings. The present study provides evidence of high levels of OCPs in indoor air of important historical and cultural buildings in the Czech Republic, such as the baroque theatre in Český Krumlov and the Museum of South Bohemia, from pesticide use that occurred decades ago. OCP contamination in indoor air may lead to an increased CR to exposed workers to the point where remediation may be required, as is the case with the Museum of South Bohemia.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.scitotenv.2017.07.203.

References

Beránek, M., Havel, M., 2006. Lindane in the Czech Republic. International Persistent Organic Pollutants Elimination Network (IPEN), pp. 1–31.
Beránek, M., Petrlík, J., 2005. POPs Pesticides in the Czech Republic. International Persistent Organic Pollutant Elimination Network (IPEN), pp. 1–21.
California, E.P.A., 2005. Air Toxics Hot Spots Program Risk Assessment Guidelines: part II Technical Support Document for Describing Available Cancer Potency Factors. California Environmental Protection Agency.
Covaci, A., Kawaki, P., Indekeu, C., Schepens, P., Neels, H., 2006. Highly chlorinated toxic contaminants in pesticide-treated wooden art objects. Arch. Environ. Occup. Health 61, 25–248.
Dvorska, A., Lammel, G., Klanova, J., Holoubek, I., 2008. Kosetice, Czech Republic–ten years of air pollution monitoring and four years of evaluating the origin of persistent organic pollutants. Environ. Pollut. 156, 403–408.

Dvorská, A., Lammel, G., Holoubek, I., 2009. Recent trends of persistent organic pollutants in air in central Europe - air monitoring in combination with air mass trajectory statistics as a tool to study the effectivity of regional chemical policy. Atmos. Environ. 43, 1280–1287.

Fellowes, J.W., Pattrick, R.A.D., Green, D.I., Dent, A., Lloyd, J.R., Pearce, C.I., 2011. Use of biogenic and abiotic elemental selenium nanospheres to sequester elemental mercury released from mercury contaminated museum specimens. J. Hazard. Mater. 189, 660–669.

Glastrup, J., 2000. The effectiveness of compressed air in removal of pesticides from ethnographic objects. In: Caldararo, N., Davis, L., Palmer, P., Waddington, J. (Eds.), The Contamination of Museum Materials and the Repatriation Process for Native California: Proceedings of a Working Conference at the San Francisco State University. vol. 16. Collection Fourm, San Francisco State University, pp. 19–22.

Goldberg, L., 1996. A history of pest control measures in the anthropology collections, national museum of natural history, Smithsonian Institution. J. Am. Inst. Conserv. 35, 23–43.

Holoubek, I., Klanova, J., Jarkovsky, J., Kohoutek, J., 2007. Trends in background levels of persistent organic pollutants at Kosetice observatory, Czech Republic. Part I. Ambient air and wet deposition 1996–2005. J. Environ. Monit. 9, 557–563.

Holt, E., Bohlin Nizzetto, P., Borůvková, J., Harner, T., Kalina, J., Melymuk, L., et al., 2017. Using long-term air monitoring of semi-volatile organic compounds to evaluate the uncertainty in polyurethane-disk passive sampler-derived air concentrations. Environ. Pollut. 220 (Part B), 1100–1111.

Kohoutek, J., Holoubek, I., Klánová, J., Zachař, P., Tříska, J., 2005. Vývoj Znečištění Vnitřních Prostor Barokního Divadla V Českém Krumlově Chemickými Látkami Ochrana Ovzduší (in Czech). 5 pp. 1–7.

Kohoutek, J., Holoubek, I., Klánová, J., Zachař, P., 2006. Tocoen Report No. 312: Vývoj znečitění vnitřních prostor barokního divadla v Českém Krumlově v letech 2001–2006 (in Czech). TOCOEN s.r.o., Brno, Czech Republic.

Lohmann, R., Klanova, J., Kukucka, P., Yonis, S., Bollinger, K., 2012. PCBs and OCPs on a east-to-west transect: the importance of major currents and net volatilization for PCBs in the Atlantic Ocean. Environ. Sci. Technol. 46, 10471–10479.

Marcotte, S., Estel, L., Leboucher, S., Minchin, S., 2014. Occurrence of organic biocides in the air and dust at the Natural History Museum of Rouen, France. J. Cult. Herit. 15, 68–72.

Musshoff, F., Gottsmann, S., Mitschke, S., Rosendahl, W., Madea, B., 2010. Potential occupational exposures in the Reiss-Engelhorn-Museen Mannheim/Germany. Bull. Environ. Contam. Toxicol. 85, 638–641.

Palmer, P., 2000. A review of analytical methods for the determination of mercury, arsenic, and pesticide residues on museum objects. In: Caldararo, N., Davis, L., Palmer, P., Waddington, J. (Eds.), The Contamination of Museum Materials and the Repatriation Process for Native California: Proceedings of a Working Conference. vol. 16. Collection Fourm, San Francisco State University, pp. 25–39.

Palmer, P.T., Martin, M., Wentworth, G., Caldararo, N., Davis, L., Kane, S., et al., 2003. Analysis of pesticide residues on museum objects repatriated to the Hupa Tribe of California. Environ. Sci. Technol. 37, 1083–1088.

Prokeš, R., Čupr, P., Vaňková, L., Přibylová, P., Audy, O., Stanovení, Klánová J., 2015. vybraných organických polutantů ve vnitřním ovzduší Jihočeského muzea v Českých Budějovicích, 2014 (in Czech). Centre for Toxic Compounds in the Environment (RECETOX), Brno, Czech Republic, pp. 1–33.

Purewal, V., 2000. The identification of four persistent and hazardous residues present on historic plant collections housed within the National Museum and Galleries of Wales. In: Caldararo, N., Davis, L., Palmer, P., Waddington, J. (Eds.), The Contamination of Museum Materials and the Repatriation Process for Native California: Proceedings of a Working Conference. vol. 16. Collection Fourm, San Francisco State University, pp. 77–86.

Schieweck, A., Delius, W., Siwinski, N., Vogtenrath, W., Genning, C., Salthammer, T., 2007. Occurrence of organic and inorganic biocides in the museum environment. Atmos. Environ. 41, 3266–3275.

Schmidt, O., 2000. Insecticide contamination at the National Museum of Denmark: a case study. In: Caldararo, N., Davis, L., Palmer, P., Waddington, J. (Eds.), The Contamination of Museum Materials and the Repatriation Process for Native California: Proceedings of a Working Conference. vol. 16. Collection Fourm, San Francisco State University, pp. 92–95.

Sirois, P., 2000. The analysis of museum objects for the presence of arsenic and mercury: non-destructive analysis and sample analysis. In: Caldararo, N., Davis, L., Palmer, P., Waddington, J. (Eds.), The Contamination of Museum Materials and the Repatriation Process for Native California: Proceedings of a Working Conference. vol. 16. Collection Fourm, San Francisco State University, pp. 65–75.

Unger, A., Schniewind, A., Unger, W., 2001. Conservation of Wood Artifacts. Sringer-Verlag, Berlin.

US EPA, 1988. TO-13: Polycyclic Aromatic Hydrocarbons (PAH).

US EPA, 1991. OSWER Directive 9355.0-30: Role of the Baseline Risk Assessment in Superfund Remedy Selection Decisions. United States Environmental Protection Agency.

US EPA, 2003. Integrated risk information system. http://www.epa.gov/iris.

US EPA, 2005a. Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities (Final). pp. 1–810.

US EPA, 2005b. Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities (Final) - The Hazardous Waste Companion Database (ACCESS). 2016. United States Environmental Protection Agency.

US EPA, 2009. Risk Assessment Guidance for Superfund Volume I: Human Health Evaluation Manual (Part F, Supplemental Guidance for Inhalation Risk Assessment). United States Environmental Protection Agency, Washington, D.C., pp. 1–68.

Wörle, M., Hubert, Vera, Hildbrand, E., Hunger, K., Lehmann, E., Mayer, I., et al., 2012. Evaluation of decontamination methods of pesticide contaminated wooden objects in museum collections: efficiency of the treatments and influence on the wooden structure. J. Cult. Herit. 135, S209–S215.

APPENDIX 9

Bečanová, Jitka, Lisa Melymuk, Šimon Vojta, Klára Komprdová, and Jana Klánová. 2016. "Screening for Perfluoroalkyl Acids in Consumer Products, Building Materials and Wastes." *Chemosphere* 164: 322-29. https://doi.org/10.1016/j.Chemosphere.2016.08.112

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Screening for perfluoroalkyl acids in consumer products, building materials and wastes

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HIGHLIGHTS

- 15 target PFAAs were analyzed in 126 samples of consumer products and building materials.
- In 88% of samples at least one PFAA was detected; dominated by PFOS.
- The highest Σ_{15}PFAA concentration (77.61 μg kg^{-1}) was found in a textile.
- Several materials contained high levels of unregulated short-chain PFAAs.
- C5-C8-chain PFCAs in wood-based building materials were identified for the first time.

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ABSTRACT

Perfluoroalkyl and polyfluoroalkyl substances (PFASs) are a large group of important chemical compounds with unique and useful physico-chemical properties, widely produced and used in many applications. However, due to the toxicity, bioaccumulation and long-range transport potential of certain PFASs, they are of significant concern to scientists and policy makers. To assess human exposure to PFASs, it is necessary to understand the concentrations of these emerging contaminants in our environment, and particularly environments where urban population spend most of their time, i.e. buildings and vehicles.

A total of 126 samples of building materials, consumer products, car interior materials and wastes were therefore analyzed for their content of key PFASs - 15 perfluoroalkyl acids (PFAAs). At least one of the target PFAAs was detected in 88% of all samples. The highest concentration of Σ_{15}PFAAs was found in textile materials (77.61 μg kg^{-1}), as expected, since specific PFAAs are known to be used for textile treatment during processing. Surprisingly, PFAAs were also detected in all analyzed composite wood building materials, which were dominated by perfluoroalkyl carboxylic acids with 5–8 carbons in the chain (Σ_4PFCAs up to 32.9 μg kg^{-1}). These materials are currently widely used for building refurbishment, and this is the first study to find evidence of the presence of specific PFASs in composite wood materials. Thus, in addition to consumer products treated with PFASs, materials used in the construction of houses, schools and office buildings may also play an important role in human exposure to PFASs.

1. Introduction

Perfluoroalkyl and polyfluoroalkyl substances (PFASs) are a diverse and large group of chemicals which consists of various polymers and nonpolymer substances (Buck et al., 2011). PFASs are applied to a wide range of commercial and industrial materials in order to change their physico-chemical properties (Banks, 1994; Kissa, 2001) resulting in lowered interfacial or/and surface tension and enhanced resistance to water, stains, oil, and fire. Major producers of PFASs are 3M, DuPont, Clariant, Daikin and Asahi Glass and applied in large volumes on a global scale: e.g. the total production of perfluorooctane sulfonyl fluoride from 1970 to 2002 was estimated to be about 100 000 metric tons (Paul et al., 2009). PFASs are specifically used in applications such as surface treatment of textiles (e.g. carpets, furniture materials and clothing), leather products, paper and packaging, coating additives, cleaning agents and firefighting foams (Paul et al., 2009; Prevedouros et al., 2006; Wang et al., 2014). In addition, specific PFASs are also used in the photographic industry, photolithography and manufacturing of

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semiconductors (OECD, 2002; EFSA, 2008). Since PFASs are typically used to treat surface layers they can be directly released from final products into the environment during their lifecycle. Wang et al. (2014) estimated global emissions of perfluoroalkyl carboxylic acids (PFCAs) (C_4 to C_{14}) from direct and indirect sources between 1951 and 2015 of up to 21 400 metric tons.

The fate of PFASs in the environment as well as their biological activity are determined by their physico-chemical properties, which vary greatly with molecular structure (e.g. functional group and carbon chain length) (Kissa, 2001). In general, neutral PFASs (e.g. fluorotelomer alcohols, perfluorinated sulfonamides and their derivatives) may undergo long range atmospheric transport (Lai et al., 2016) and may be partially degraded into perfluoroalkyl acids (PFAAs) in the presence of OH radicals (Ellis et al., 2004). These PFAAs, such as PFCAs and perfluoroalkane sulfonates (PFSAs) have been shown to persist in the environment (Scheringer et al., 2014) and have no known significant natural degradation mechanism (Sulbaek Andersen et al., 2003.). Typically they can undergo long-range transport and therefore they are widespread in a range of environmental compartments around the globe (Butt et al., 2007; Giesy and Kannan, 2001). Humans are exposed to PFASs through various pathways (e.g. dust ingestion, drinking water, diet and inhalation) (Enault et al., 2015; Haug et al., 2011; Miralles-Marco and Harrad, 2015). Similarly to the legacy persistent organic pollutants (POPs, e.g. polychlorinated biphenyls, organochlorine pesticides), the neutral PFASs accumulate in lipid-rich tissues (Chu et al., 2016). In contrast, the ionic PFASs, particularly the longer chain PFAS, bind to blood proteins and accumulate in the liver, kidney and bile secretions (Ng and Hungerbühler, 2014). Almost all human blood samples collected around the world were found to contain ionic PFASs at levels of ng mL^{-1} (Gebbink et al., 2015; Miralles-Marco and Harrad, 2015; Wang et al., 2015b). Information about the effects on humans are primarily available for PFOA and PFOS and are related to early menopause, changes in a sperm quality and increased risk of cancer (Bonefeld-Jorgensen et al., 2011; Joensen et al., 2009).

In 2009, PFOS and related compounds were listed under Annex B of the Stockholm Convention on Persistent Organic Pollutants, which restricts manufacturing and use to a few specific applications (Stockholm Convention, 2009). Within the European Union, the use of PFOS is regulated by European Commission Regulation No. 122/2006 (EU, 2006a), 552/2009 (EU, 2009) and 757/2010 (EU, 2010) Intermediate products or articles with concentrations of PFOS greater than 0.1% by weight are banned since 2007. For textiles or other coated materials, the amount of PFOS must be lower than 1 $\mu g\ m^{-2}$. Moreover, PFOA, other C9-C14 PFCAs and their salts and precursors have been identified by the European Chemicals Agency as a Substance of Very High Concern due to their irreversible effects on the environment and human health (ECHA, 2015) and presence of PFOA in articles on the European market at levels higher than 0.1% must be reported (EU Regulation No. 1907/2006 (EU, 2006b)). The group of regulated PFAAs includes also several other identified precursors and intermediates (OECD, 2007), which are currently being replaced by the manufacturers with alternative PFASs; e.g. PFAAs with shorter (C4 to C6) fluorinated chains or PFASs with hetero atoms (N, S, O) in their carbon chains (EPA, 2012; Wang et al., 2013). Use volumes and the environmental and human impact of these alternative PFASs are currently under scientific focus (Blum et al., 2015).

Both regulated and non-regulated PFASs are frequently found to be present in consumer products used on a daily basis in homes, schools and workplaces. While intensive research has focused on food packaging materials (Begley et al., 2005; Gallart-Ayala et al., 2013; Herzke et al., 2012; Martanez-Moral and Tena, 2012; Poothong et al., 2012; Sinclair et al., 2007; Trier et al., 2011a, 2011b; Vestergren et al., 2008; Zafeiraki et al., 2014), information regarding levels found in other daily-use consumer products is limited (Herzke et al., 2012; Liu et al., 2014; Trier et al., 2011a; Vestergren et al., 2015; Ye et al., 2015) and no information regarding levels in building materials is currently available. However, the indoor environment is a potential source of human PFAS exposure, especially in view of the fact that people in urban areas spend more than 20 h per day in indoor spaces (Klepeis et al., 2001).

Quantifying PFAS exposure is a complex and challenging process, not least due to a lack of knowledge concerning the exact PFAS composition in materials. Thus, a necessary component of quantifying PFAS exposure is obtaining information on their concentrations in broad range of materials types, since these materials release PFASs and contribute to their elevated levels in indoor air and dust. In this study we analyzed 126 individual samples of building materials, consumer products, car interior materials and wastes in order to provide insight into the distribution and amounts of PFASs added to indoor materials and to identify the potential sources of specific PFASs to the indoor environment.

2. Experimental

2.1. Sample collection

The aim of this study was to obtain PFAA levels in a representative selection of different types of materials. Both new and used materials were included to cover the widest range of materials used in the construction of buildings, household equipment, interior of cars and wastes. These included mainly construction materials used in the past three decades, new and used electrical devices, flooring, fabric, upholstery and other daily-use materials. Samples of recycled materials were also included, as their contamination from primary materials remains unclear. New materials were purchased while older and used materials were collected from various sources. These materials were also analyzed for flame retardants, presented elsewhere (Vojta et al., in prep).

A total of 126 samples were split into four categories according to use and composition. The first category included household equipment; i.e. (1A) textiles, (1B) floor coverings, (1C) electrical & electronic equipment (EEE), and (1D) plastics. The second category included building materials; i.e. (2A) oriented strand board (OSB), other composite wood and wood, (2B) insulation materials, (2C) mounting and sealant foam, (2D) facade materials, (2E) polystyrene and (2F) air conditioner components. Some building materials were supplied by a company dealing with the construction of low-energy houses. In the third category were car interior materials. The final group consisted of wastes of electrical & electronic equipment (WEEE) collected at a sorting plant. See Supplementary Materials Table SI.2 for a detailed list of sample categorization.

2.2. Sample preparation and extraction

Samples of solid materials were crushed, chopped or ground while samples of foam and fabrics were cut into small pieces. After grinding or cutting, 5 g of each sample was extracted with methanol with the addition of ammonium acetate (final concentration 5 mM) using warm Soxhlet extraction (60 min warm Soxhlet followed by 30 min of solvent rinsing) in a B-811 extraction unit (Büchi, Switzerland). Following extraction, samples were cleaned up according to the procedures for PFAAs analysis described in detail elsewhere (Karásková et al., 2016). Briefly, concentrated extracts were filtered using a syringe filter (nylon membrane, 13 mm diameter and 0.45 µm pore size). The filtrate was concentrated using a stream of nitrogen in a TurboVap II (Caliper LifeSciences, USA) concentrator unit to 500 µL and transferred to a minivial.

Finally, samples were diluted using a solution of ammonium acetate in water (concentration 5 mM) up to a final volume (50/50, ammonium acetate in water/ammonium acetate in methanol, v/v). Prior to final analysis, syringe standards ($^{13}C_4$ PFOA, $^{13}C_4$ PFOS) were added to all samples.

2.3. Quality assurance and quality control

The extraction efficiency of individual PFAAs from analyzed materials was tested by repeated warm Soxhlet extraction cycles. Since no PFAAs were detected in repeat extracts, we assumed that the sample preparation method described above was sufficient to remove all extractable amounts of PFAAs from materials and therefore to obtain realistic levels of PFAAs present in samples.

The accuracy of method was evaluated using a set of spiked solid blank materials (polyurethane foam ($n = 6$) and sand matrix blank ($n = 10$)). Obtained recoveries varied between 70% and 120% for individual compounds with relative standard deviation up to 21% for PFCAs and 8.3% for PFSAs (extraction method details are provided in Supplementary Materials and results for individual compounds are in Table SI.1b). Laboratory procedural blanks were also analyzed ($n = 10$) using empty extraction cartridges and no laboratory contamination was detected for any of the analyzed compounds. For quantification of target compounds, set of calibration standards solution (CS) were analyzed and the instrumental limit of quantification (ILQ) was calculated from the signal to noise ratio ($S/N = 9$) for the lower calibration point (Table SI.1a). Method quantitation limits (MQL) were calculated from spiked solid blank materials as a lowest quantifiable concentration and were recalculated to the sample weights. Precision of measurement was determined by repetitive injection of QA/QC standard solution as described in Bečanová et al. (2016).

2.4. Instrumental analysis

The separation and detection of 15 PFAAs: short-chain (C5-C7) PFCAs, long-chain (C8-C14) PFCAs and C4, C6, C7, C8, C10 PFSAs (listed in full in Table SI.1a) were performed by liquid chromatography using an Agilent 1100 (Agilent Technologies, Palo, Alto, CA) equipped with a SYNERGI 4μ Fusion RP 80Ä 50 mm × 2 mm column (Phenomenex, USA) coupled to a mass spectrometer QTRAP 5500 (ABSciex, CA, USA) interfaced with an electrospray ionization source (HPLC-ESI-MS/MS). The mass spectrometer was operated in negative ion mode (EI-) using two MRM transitions for each compound. For detailed mass spectrometer conditions see Supplementary Materials Tables SI.3 and SI.4. Aliquots of 10 μL were injected on the column and eluted using a gradient of 200 μL min^{-1} methanol and water (with 5 mM ammonium acetate). The initial gradient was set at 45/55 methanol/water, with methanol content increased to 80% over two minutes (then held for 10 min) and further increased to 100% methanol (with 5 mM ammonium acetate) and held for 3 min. The column was equilibrated using the initial content of the mobile phase for 5 min between injections.

2.5. Statistical analysis

Prior to data treatment, an anomalous sample of a 7-year-old polyurethane foam with 2 times higher concentrations of target chemicals compared to the second highest concentrated one was excluded from the dataset due to detectable levels of all of the target compounds at unusually high levels (see Table SI.2).

The Kruskal-Wallis test and multiple comparisons of mean ranks were used for statistical differences of sum PFAAs among material categories according to use and composition. To identify PFAA concentration patterns in different types of materials, cluster analysis was performed. Only PFAAs with more than 10% of values above detection limits were used for this analysis. The concentrations were standardized to three levels. Values below or close to limits of quantification were assigned zeros, values with normal distribution range were marked as 1 and outlier and extreme values were marked as 2. This type of transformation allowed direct comparison of different levels of PFAAs in materials. The samples without any detected concentrations (with all zeros, in total 33) were excluded from clustering. The Ward's method with Euclidean distance was used as the clustering method. All statistical analyses were performed in STATISTICA (version 12, StatSoft, Inc.).

3. Results and discussion

3.1. PFAA detection frequency and general composition profiles

Two classes of PFAAs (PFSAs and PFCAs) were quantified in 126 samples. PFAAs were found in all but eleven samples (88%). As expected, the most frequently detected compound was PFOS, with a detection frequency of 64% (Figure SI.1). The detection frequency of the remaining PFSAs varied between 27 and 45%. The detection frequency of the short-chain PFCAs (C5–C7) and PFOA ranged from 21 to 44%; by contrast, the detection frequency of PFCAs with 9–14 carbons in their chain was only 2–5%. Although the low frequency of PFCAs with carbon chain ≥10 was predictable due to their recent limited production and usage in consumer products (Wang et al., 2014), their high bioaccumulation potential (Martin et al., 2004) warrants their continued monitoring in consumer products.

As a result of the relation between their median concentration and detection frequency, PFAAs were split into two major groups (Fig. 1).

The first group (I.) comprises PFAAs found at low concentrations with a low detection frequency, and contains long-chain PFCAs with 11–14 carbons in their chain, as expected considering there is no reported intensive use of long-chain PFCAs in consumer products or building materials. The second group (II.) contains compounds with detection frequencies between 25 and 45% and median concentrations of up to 0.6 μg kg^{-1} of material. This group includes both PFCAs and PFSAs with short carbon chains. Some of these compounds are used as alternatives to banned C8 compounds (EPA, 2012); moreover, they were identified as impurities occurring during C8 compound production (Buck et al., 2011). The remaining ungrouped substances are PFOS, PFOA and PFHpA. PFOS (median concentration 0.12 μg kg^{-1} and a detection frequency of 64%) suggests broad usage in materials, as previously reported (Herzke et al., 2012; Washburn et al., 2005). The C7 and C8 PFCAs have a similar detection frequency (20 and 14%, respectively) but different median concentrations (0.88 and 0.37 μg kg^{-1}). Similar results for short-chain PFCAs have been reported in various applications, i.e. firefighting foams (Herzke et al., 2012), textiles and food packaging materials (Begley et al., 2005; Vestergren et al., 2015; Ye et al., 2015).

3.2. Levels of PFAA in consumer product groups

Total PFAA concentrations in consumer products with detectable levels ranged from 0.01 to 77.6 μg kg^{-1}. All materials were split into product use groups (as described in Section 2.1, and Table SI.1) to generalize the measured concentrations (Table 1).

In the first category (household equipment) there were no statistically significant differences between individual groups (1A, 1B, 1C, and 1D). The textile group (1A) showed high variability due to the relatively high number of extremes and outliers, and contained the highest concentration of $\sum_{15}$PFAAs (77.6 μg kg^{-1}). Floor coverings (1B) also covered large ranges, with $\sum_{15}$PFAAs from 0.3 to

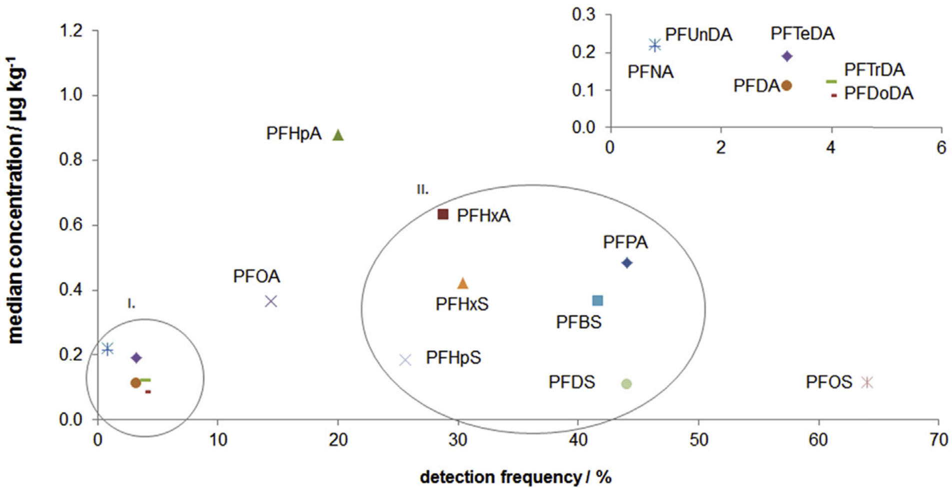

Fig. 1. Median concentration vs. detection frequency of individual PFAAs in 125 samples.

Table 1
Comparison of Σ_{15}PFAA concentrations within four categories of materials.

Category	Group		n_{total} [a] ($n_{positive}$)	min - max (median)[b] µg kg^{-1}	Intra-category Kruskal-Wallis test [c]
Household equipment	Textile	1A	23 (19)	<MQL[d] - 77.6 (1.15)	n.s.
	Floor covering	1B	9 (9)	0.310–38.4 (2.09)	n.s.
	EEE	1C	18 (16)	<MQL - 11.7 (0.384)	n.s.
	Plastics	1D	4 (3)	<MQL - 0.384 (0.029)	n.s.
Building materials	OSB and wood	2A	14 (**14**)	1.39–18.3 (**4.87**)	* (2C, 2E, 2F)
	Insulation materials	2B	16 (16)	**0.068**–**34.3** (3.55)	*(2E)
	Mounting and sealing foam	2C	6 (4)	<MQL - 1.28 (0.230)	*(2A)
	Facade materials	2D	8 (6)	0–24.5 (0.618)	**n.s.**
	Polystyrene	2E	5 (1)	<MQL - 0.18 (0.00)	*(2A, 2B)
	Air conditioning	2F	5 (5)	0.057–0.295 (0.135)	*(2A)
Car interior materials	–	3	10 (10)	**0.033**–**35.5** (1.34)	n.a.
WEEE	–	4	7 (7)	0.046–2.20 (1.42)	n.a.

[a] n_{total} - total number of samples, $n_{positive}$ - number of positive detected samples within each group.
[b] of Σ_{15}PFAAs.
[c] *$p < 0.05$; n.s. - not statistically significant; n.a. - not applicable; (p values in Table SI.5).
[d] MQL – *method quantification limit (specified in* Table SI.1*).*

38.4 µg kg^{-1}. Within the second category (building materials), statistically significant differences were found between the OSB and wood group (2A) and the other three (2C, 2E, 2F) groups. Group 2A exhibited the highest median Σ_{15}PFAA concentration (4.87 µg kg^{-1}) with a narrow distribution and 100% detection frequency. This uniformity could be caused by similarities in the production for all materials within this group. In the third category (car interior materials) although all samples were from car interiors, the materials themselves were very diverse (e.g. plastics, foams and textiles), and this is reflected in the wide range (from 0.03 to 35.5 µg kg^{-1}) of Σ_{15}PFAA concentrations. Similar ranges were also found for groups 1B and 2B, which we hypothesize, is due to differences in materials type, ages and production technologies within these groups as well. In contrast, the last category (WEEE) had a smaller range of Σ_{15}PFAA concentrations, attributed to similar type of material (mainly plastics) within this group and the homogenization that occurs as products enter the waste sorting stream.

3.3. *PFAA composition of specific samples in individual consumer products categories*

As discussed further in section 3.4, for most materials it is not possible to identify whether the PFAAs originate from production of the materials or from the product lifetime. However, some materials exhibited either considerable concentrations of one specific PFAA or a compound-specific pattern was observed for a particular group of materials. In the following section the materials with patterns indicating the intentional addition of PFAAs or their precursors during production are investigated in detail.

3.3.1. *Household equipment*

Of the new textile materials, the sample with the highest concentration of PFAAs was stain resistant upholstery material produced in 2010, with a predominant contribution of PFHpS (73.8 µg kg^{-1}) at levels 23 times higher than PFOS (3.2 µg kg^{-1}). No toxicological characteristics, data on associated human risk or regulated levels of PFHpS are currently available. Additionally two carpet samples produced in 2006 and listed as used in Table SI.2 contained high concentrations of $\sum_{15}$PFAA (8.54 and 38.4 µg kg^{-1}) with a majority of PFOS (6.08 and 19.6 µg kg^{-1}). Following a recalculation based on the typical face weight of carpets used in Europe (500–2000 g m^{-2}; Fung, 2002), the surface concentrations in the carpets were determined to be 16–44 µg m^{-2} and 4.8–13.2 µg m^{-2}, respectively. These levels exceeded the PFOS limit (1 µg per m^2 of coated fabrics) stipulated by the EU directive by more than 5 times (EU, 2006a). However, since both carpets were used prior to the analysis, measured concentrations may originate from both treatment of carpeting with PFAA-containing stain-proofing sprays/washes during carpet use or intentional addition during carpet manufacturing. Presence and levels of PFOS and its alternatives in various textile materials (especially in floor covering and upholstery materials) are of interest, since direct contact with these materials is an important exposure pathway, particularly for small children.

The last group of household equipment (1D - plastics) included plastics made from recycled materials which might be potential sources of indoor contamination. However, there is a lack of data regarding the potential content of PFAAs in these recycled materials. In our study, no PFAAs were detected in recycled plastics. Thus, while recycled plastics may be potential sources of compounds such as brominated flame retardants to the indoor environment (Hirai and Sakai, 2007), our analysis suggests that they may not be a significant source of PFAAs.

3.3.2. Building materials

All 14 samples in group 2A (OSB and wood) contained primarily short-chain PFCAs and PFOA at concentrations ranging from 1.38 to 13.9 μg kg^{-1} for Σ_4PFCAs. The pattern and composition of those PFCAs were similar in all samples within this group. Although wood-based material technology has undergone significant development in many countries over the past 50 years (Buehlmann et al., 2000) and composition of additives used in composite wood manufacturing depends on the producer, the most commonly used resins are based on urea- or phenol-formaldehyde due to their good performance and low cost (Tang et al., 2009). However, no information about PFAS concentrations in these resins is available from their producers. The only information about the usage of PFOS-related substances in analogous application is their addition in sealants and adhesive products (POPRC, 2011, 2012). Therefore, we considered the addition of these adhesives or other additives to wooden strips or plates during the production of OSB and wood-based materials to be a possible source of PFASs. A detailed investigation of these composite wood materials may be useful in the future, especially when worldwide production has been growing steadily since 2008 (FAO, 2009) (74 million m^3 per year in 2012 in Europe and over 117 million m^3 in China (FAO, 2013)), as well as due to the recycling potential of these materials.

In group 2B, two samples of wood fibre insulation (produced in 2010) contained high amounts of PFHpA (20.6 and 28.4 μg kg^{-1}) and other 5- to 8-carbon chain PFCAs (12.3 and 5.8 μg kg^{-1}). Both insulations were produced by a company manufacturing environmentally-friendly materials and are regularly used for the insulation of floors and walls in modern buildings and especially in low-energy houses. Following a recalculation using insulation area and material density, the PFHpA concentration was found to be 61.5 μg m^{-2} (wall insulation) and 181.8 μg m^{-2} (floor insulation). Although these short–chain PFOS alternatives are currently manufactured by major producers, there are no current regulations on short-chain PFCAs in indoor materials in the European Union. However, at the same time there is scientific pressure on manufacturers to inform consumers about the usage of these alternatives (Cousins et al., 2015).

Materials with high concentrations of PFOS were also found in group 2B. A sample of phenolic insulation foam produced in 2010, used as an insulating layer on interior walls, contained 22.8 μg kg^{-1} of PFOS, which corresponds to a concentration of 54.7 μg m^{-2} after recalculation for area. There is no limit for PFOS in building materials, but the EU legislative limit for carpets (1 μg m^{-2}) as a floor covering was thus exceeded 50 times in this case. While insulation is not a strictly equivalent surface layer to carpet, the comparison with the EU limits for PFOS for carpets as a surface covering suggests that the levels of PFOS in insulation may be a problematic contamination source in the indoor environment, especially if used in open applications.

3.3.3. Car interior materials

We investigated ten samples of plastics, textile and upholstery materials used for the interiors of two cars: Škoda and Hyundai, each from ~1995 (Table SI.1). We found PFAAs in all samples in the same range as in facade and insulation materials (Table 1). One of the foams used as upholstery material contained 35.5 μg kg^{-1} of PFOS. As no other PFAAs were present in this sample, we suspect that PFOS was added during the production of this foam.

3.3.4. Wastes of electrical & electronic equipment (WEEE)

Within this group all materials contained PFOS in the range of 0.07–0.43 μg kg^{-1} and also other PFASs and PFCAs at detectable levels. Since the materials are homogenized in the waste sorting systems and therefore the origin and age of individual materials is not known, the origin of PFAAs in WEEE cannot be identified.

3.4. Possible sources of PFAAs in consumer products

Generally, in a majority of materials (Table SI.2), the origin of detected PFAAs is not clear and they may originate either from production of selected materials or from transport and usage. We can further divide this into four types of PFAA sources: (1) intentional addition during production to achieve properties associated with the use of PFAAs; (2) contamination with by-products/

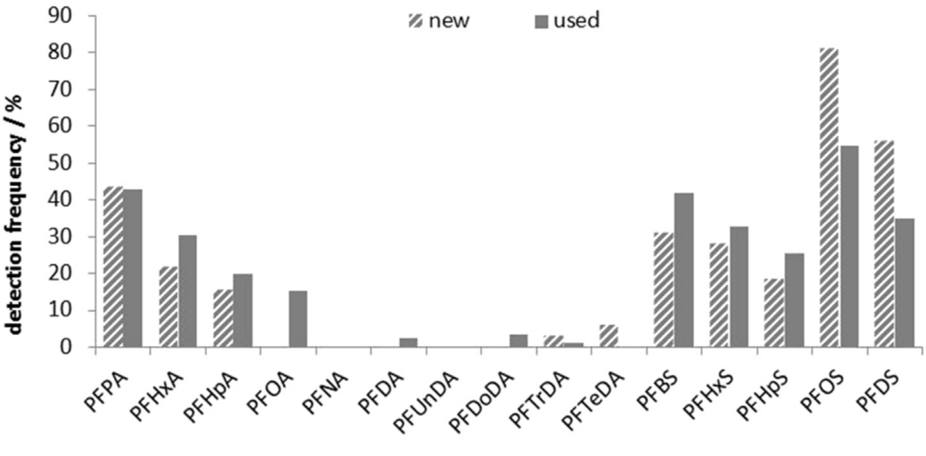

Fig. 2. Detection frequency of PFAAs in new vs. previously used materials.

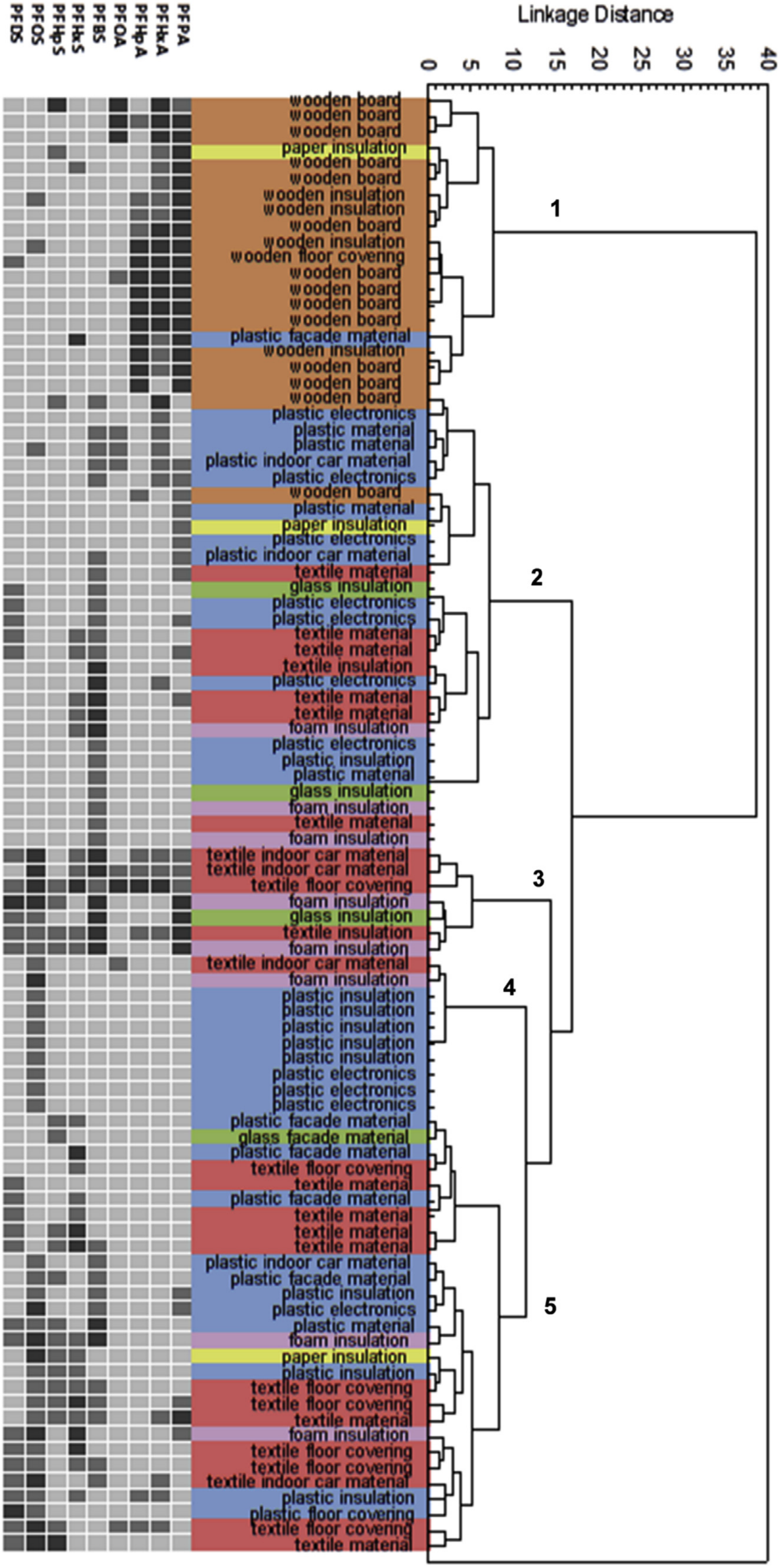

Fig. 3. Dendrogram of material samples (according to raw material and final product application) based on selected PFAA concentrations. The gray scale on the left reflects three concentration levels (darker grey indicates higher concentrations). Color coding indicates similar materials, and numbers 1 to 5 denote clusters of different composition described further in the text.

impurities during production or product lifetime; (3) intentional addition during product use, such as application of stain-proofing sprays to carpets, and (4) contamination from the surrounding environment during the product lifetime. In a few cases, products can be easily linked with one of the PFAA source types, for example, the car interior foam mentioned above, containing only PFOS, suggests intentional addition during manufacturing; while the presence of PFHpA, which is predominantly a breakdown product of coatings on carpets, textiles and food packaging (Wang et al., 2015a) suggests breakdown of coatings, potentially beginning rapidly after product manufacture.

However, in most cases it is very challenging to identify the exact source of the detected PFAAs. A comparison of new and used materials offers some insight (Fig. 2). The higher detection frequency of short-chain PFSAs in older materials suggests possible uptake of PFSAs during the product lifetime. However, further interpretation of the differences between new and previously used products is challenging as the composition of the new and used product groups was not identical so these differences may also be influenced by the presence of different product types in new vs. used groupings (Figure SI.2).

3.5. *PFAA patterns in material based on composition*

Due to the uncertainties in the origin of PFAAs reported in many materials (see above), statistical analyses of materials based on composition of materials was utilized. The goal of this approach was to link the pattern of PFAAs with the type of manufacturing of the materials. Firstly, PFAA concentrations were grouped according to three concentration levels, and secondly a clustering technique (Ward's method with Euclidean distance) was utilized. Samples were hierarchically clustered depending on their level of PFAA concentrations; the resulting dendrogram thus reflects patterns of materials with similar PFAA content (Fig. 3).

In the dendrogram, the top cluster (marked as number 1) contains mainly composite wood materials previously grouped in different categories, i.e. primarily OSB but also wooden floor covering and wood-based insulation. These samples exhibited the highest concentrations of short-chain PFCAs, whose likely sources were discussed in section 3.2. The lower cluster of the dendrogram contains two main subgroups. The first subgroup (cluster 2) includes primarily plastics and textiles, all of which contained mainly C4 and C6 PFASs, i.e. replacements for PFOS. This contrasts with cluster 4, which contains mainly plastics produced before the 2009 PFOS ban and which thus have a high PFOS content. Cluster 3 contains textile floor coverings (from both cars and buildings) and foam insulation. The presence of a broad range of PFAS in this group suggests that these compounds were intentionally added in order to change textile and foam surface properties. Although the sorption of PFASs during use may contribute to their concentration, the presence of both new and used materials in this group thus suggests intentional addition during manufacturing. The last group (cluster 5) consists predominantly of plastics and textiles with C4 to C9 PFSAs. Despite the fact that some of these materials were produced after 2009, samples in this group also contained PFOS. However, the levels of short-chain PFSAs were found to be similar or higher than levels of PFOS, which further supports above-discussed changes in PFAS production. All above-mentioned findings indicate that the PFAA content is associated with material to a greater extent than with usage.

4. Conclusions

We analyzed perfluoroalkyl substances in 126 individual samples of building materials, consumer products, car interior materials and wastes which potentially affect indoor environments where people spend most of their time. A WEEE category was also included in the study to facilitate the monitoring of PFAA concentrations throughout the lifespan of consumer products. The concentrations of PFAAs in the majority of studied materials suggested that the presence of these compounds was not caused by the intentional addition of PFAAs or their precursors to the materials during the manufacturing, as the levels were typically low and product groups contained multiple assorted and unrelated PFAAs. However, regardless of the origin of PFAAs in specific consumer products (e.g. impurities or degradation of precursors) knowledge about their levels is important when assessing human exposure.

However in a few particular materials high concentrations of banned PFOS and unregulated short-chain PFCAs and PFSAs were found. The concentrations of some individual samples exceeded the EU limit for PFOS content in consumer products. The highest concentrations of PFASs were found in construction and textile materials. As construction materials are normally installed during building construction or renovation, they may not be under the control of the occupant. Moreover, some materials, like wood-based building materials, are not typically considered as potential sources of PFAAs. Therefore, better consumer awareness of the use of perfluorinated additives in these materials is desirable. Better information and awareness is also useful for household equipment, where the choice and use depends on a resident and potential unknown indoor exposure may thus be controlled to a greater extent. Detailed monitoring of indoor environments along with research regarding new materials and potential testing of PFOS alternatives is needed. Increased attention should be paid especially to new polymers and nonpolymer substances introduced by key PFAS producers, e.g. Novec™, Scotchguard™ and FORAFAC™, which are currently unregulated.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.chemosphere.2016.08.112.

References

Banks, R.E., 1994. Organofluorine Chemistry: Principles and Commercial Applications. Plenum Press.

Bečanová, J., Komprdová, K., Vrana, B., Klánová, J., 2016. Annual dynamics of perfluorinated compounds in sediment: a case study in the Morava River in Zlín district, Czech Republic. Chemosphere 151, 225–233.

Begley, T.H., White, K., Honigfort, P., Twaroski, M.L., Neches, R., Walker, R.A., 2005. Perfluorochemicals: potential sources of and migration from food packaging. Food Addit. Contam. 22, 1023–1031.

Blum, A., Balan, S.A., Scheringer, M., Trier, X., Goldenman, G., Cousins, I.T., Diamond, M., Fletcher, T., Higgins, C., Lindeman, A.E., Peaslee, G., de Voogt, P., Wang, Z.Y., Weber, R., 2015. The Madrid statement on poly- and perfluoroalkyl substances (PFASs). Environ. Health Perspect. 123, A107–A111.

Bonefeld-Jorgensen, E.C., Long, M., Bossi, R., Ayotte, P., Asmund, G., Kruger, T., Ghisari, M., Mulvad, G., Kern, P., Nzulumiki, P., Dewailly, E., 2011. Perfluorinated compounds are related to breast cancer risk in Greenlandic Inuit: a case control study. Environ. Health 10.

Buck, R.C., Franklin, J., Berger, U., Conder, J.M., Cousins, I.T., de Voogt, P., Jensen, A.A., Kannan, K., Mabury, S.A., van Leeuwen, S.P.J., 2011. Perfluoroalkyl and polyfluoroalkyl substances in the environment: terminology, classification, and origins. Integr. Environ. Assess. Manag. 7, 513–541.

Buehlmann, U., Ragsdale, C.T., Gfeller, B., 2000. A spreadsheet-based decision support system for wood panel manufacturing. Decis. Support Syst. 29, 207–227.

Butt, C.M., Mabury, S.A., Muir, D.C.G., Braune, B.M., 2007. Prevalence of long-chained perfluorinated carboxylates in seabirds from the Canadian Arctic between 1975 and 2004. Environ. Sci. Technol. 41, 3521–3528.

Chu, S., Letcher, R.J., McGoldrick, D.J., Backus, S.M., 2016. A new fluorinated surfactant contaminant in biota: perfluorobutane sulfonamide in several fish species. Environ. Sci. Technol. 50, 669–675.

Cousins, I.T., Balan, S.A., Scheringer, M., Weber, R., Wang, Z.Y., Blum, A., Diamond, M., Fletcher, T., Goldenman, G., Higgins, C., Lindeman, A.E., Peaslee, G., Trier, X., De Voogt, P., 2015. Comment on “Fluorotechnology is critical to modern life: the fluoro council counterpoint to the Madrid statement”. Environ. Health Perspect. 123, A170.

ECHA, 2015. ANNEX XV Proposal for a Restriction – Perfluorooctanoic Acid (PFOA), PFOA Salts and PFOA-related Substances Committee for Risk Assessment and Committee for Socio-economic Analysis.

EFSA, 2008. Perfluorooctane sulfonate (PFOS), perfluorooctanoic acid (PFOA) and their salts. EFSA J. 6, 1–131.

Ellis, D.A., Martin, J.W., De Silva, A.O., Mabury, S.A., Hurley, M.D., Andersen, M.P.S., Wallington, T.J., 2004. Degradation of fluorotelomer alcohols: a likely atmospheric source of perfluorinated carboxylic acids. Environ. Sci. Technol. 38, 3316–3321.

Enault, J., Robert, S., Schlosser, O., de Thé, C., Loret, J.-F., 2015. Drinking water, diet, indoor air: comparison of the contribution to environmental micropollutants exposure. Int. J. Hyg. Environ. Health 218, 723–730.

EPA, 2012. Emerging Contaminants – Perfluorooctane Sulfonate (PFOS) and Perfluorooctanoic Acid (PFOA). United States Environmental Protection Agency.

EU, 2006a. European Union Directive 2006/122/EC of the European Parliament and of the Council.

EU, 2006b. Regulation (EC) No 1907/2006 of the European Parliament and of the Council.

EU, 2009. Commission Regulation (EC) No 552/2009.

EU, 2010. Commission Regulation (EU) No. 757/2010.

FAO, 2009. Global Demand for Wood Products. State World's Forestry. 2009, p. 144.

FAO, 2013. Global Forest Products Facts and Figures. State of the World’s Forests 2009. FAO Forestry Department, p. 144.

Fung, W., 2002. Coated and Laminated Textiles, first ed. Woodhead Publishing.

Gallart-Ayala, H., Nunez, O., Lucci, P., 2013. Recent advances in LC-MS analysis of food-packaging contaminants. TrAC Trends Anal. Chem. 42, 99–124.

Gebbink, W.A., Glynn, A., Berger, U., 2015. Temporal changes (1997-2012) of perfluoroalkyl acids and selected precursors (including isomers) in Swedish human serum. Environ. Pollut. 199, 166–173.

Giesy, J.P., Kannan, K., 2001. Global distribution of perfluorooctane sulfonate in wildlife. Environ. Sci. Technol. 35, 1339–1342.

Haug, L.S., Huber, S., Becher, G., Thomsen, C., 2011. Characterisation of human exposure pathways to perfluorinated compounds - comparing exposure estimates with biomarkers of exposure. Environ. Int. 37, 687–693.

Herzke, D., Olsson, E., Posner, S., 2012. Perfluoroalkyl and polyfluoroalkyl substances (PFASs) in consumer products in Norway - a pilot study. Chemosphere 88, 980–987.

Hirai, Y., Sakai, S.-i, 2007. Brominated Flame Retardants in Recycled Plastic Products. BFR, Amsterdam, The Netherlands.

Joensen, U.N., Bossi, R., Leffers, H., Jensen, A.A., Skakkebæk, N.E., Jørgensen, N., 2009. Do Perfluoroalkyl compounds impair human semen quality? Environ. Health Perspect. 117, 923–927.

Karásková, P., Venier, M., Melymuk, L., Bečanová, J., Vojta, Š., Prokeš, R., Diamond, M.L., Klánová, J., 2016. Perfluorinated alkyl substances (PFASs) in household dust in Central Europe and North America. Environ. Int. 94, 315–324.

Kissa, E., 2001. Fluorinated Surfactants and Repellents. Marcel Dekker Inc., New York.

Klepeis, N.E., Nelson, W.C., Ott, W.R., Robinson, J.P., Tsang, A.M., Switzer, P., Behar, J.V., Hern, S.C., Engelmann, W.H., 2001. The National Human Activity Pattern Survey (NHAPS): a resource for assessing exposure to environmental pollutants. J. Expo. Anal. Environ. Epidemiol. 11, 231–252.

Lai, S., Song, J., Song, T., Huang, Z., Zhang, Y., Zhao, Y., Liu, G., Zheng, J., Mi, W., Tang, J., Zou, S., Ebinghaus, R., Xie, Z., 2016. Neutral polyfluoroalkyl substances in the atmosphere over the northern South China Sea. Environ. Pollut. 214, 449–455.

Liu, X.Y., Guo, Z.S., Krebs, K.A., Pope, R.H., Roache, N.F., 2014. Concentrations and trends of perfluorinated chemicals in potential indoor sources from 2007 through 2011 in the US. Chemosphere 98, 51–57.

Martanez-Moral, M.P., Tena, M.T., 2012. Determination of perfluorocompounds in popcorn packaging by pressurised liquid extraction and ultra-performance liquid chromatography-tandem mass spectrometry. Talanta 101, 104–109.

Martin, J.W., Smithwick, M.M., Braune, B.M., Hoekstra, P.F., Muir, D.C.G., Mabury, S.A., 2004. Identification of long-chain perfluorinated acids in biota from the Canadian Arctic. Environ. Sci. Technol. 38, 373–380.

Miralles-Marco, A., Harrad, S., 2015. Perfluorooctane sulfonate: a review of human exposure, biomonitoring and the environmental forensics utility of its chirality and isomer distribution. Environ. Int. 77, 148–159.

Ng, C.A., Hungerbühler, K., 2014. Bioaccumulation of perfluorinated alkyl acids: observations and models. Environ. Sci. Technol. 48, 4637–4648.

OECD, 2002. Hazard assessment of perfluorooctane sulfonate (PFOS) and its salts. Co-operation on existing chemicals. Organ. Econ. Co-oper. Dev. 362.

OECD, 2007. Report of an OECD Workshop on Perfluorocarboxylic Acids (PFCAs) and Precursors. Organisation for Economic Co-operation and Development, p. 82.

Paul, A.G., Jones, K.C., Sweetman, A.J., 2009. A first global production, emission, and environmental inventory for perfluorooctane sulfonate. Environ. Sci. Technol. 43, 386–392.

Poothong, S., Boontanon, S.K., Boontanon, N., 2012. Determination of perfluorooctane sulfonate and perfluorooctanoic acid in food packaging using liquid chromatography coupled with tandem mass spectrometry. J. Hazard. Mater. 205–206, 139–143.

POPRC, 2011. Guidance on Alternatives to Perfluorooctane Sulfonic Acid and its Derivatives. Report of the Persistent Organic Pollutants Review Committee on the Work of its Sixth Meeting. Stockholm Convention on Persistent Organic Pollutants, Geneva.

POPRC, 2012. Technical Paper on the Identification and Assessment of Alternatives to the Use of Perfluorooctane Sulfonic Acid, its Salts, Perfluorooctane Sulfonyl Fluoride and Their Related Chemicals in Open Applications. Stockholm Convention, Geneva.

Prevedouros, K., Cousins, I.T., Buck, R.C., Korzeniowski, S.H., 2006. Sources, fate and transport of perfluorocarboxylates. Environ. Sci. Technol. 40, 32–44.

Scheringer, M., Trier, X., Cousins, I.T., de Voogt, P., Fletcher, T., Wang, Z., Webster, T.F., 2014. Helsingør Statement on poly- and perfluorinated alkyl substances (PFASs). Chemosphere 114, 337–339.

Sinclair, E., Kim, S.K., Akinleye, H.B., Kannan, K., 2007. Quantitation of gas-phase perfluoroalkyl surfactants and fluorotelomer alcohols released from nonstick cookware and microwave popcorn bags. Environ. Sci. Technol. 41, 1180–1185.

Stockholm Convention. In: Parties, C.o.t. (Ed.), 2009. Stockholm convention on persistent organic pollutants (POPs), p. 64.

Sulbaek Andersen, M.P., Hurley, M.D., Wallington, T.J., Ball, J.C., Martin, J.W., Ellis, D.A., Mabury, S.A., 2003. Atmospheric chemistry of C2F5CHO: mechanism of the C2F5C(O)O2 + HO2 reaction. Chem. Phys. Lett. 381, 14–21.

Tang, X.J., Bai, Y., Duong, A., Smith, M.T., Li, L.Y., Zhang, L.P., 2009. Formaldehyde in China: production, consumption, exposure levels, and health effects. Environ. Int. 35, 1210–1224.

Trier, X., Granby, K., Christensen, J.H., 2011a. Polyfluorinated surfactants (PFS) in paper and board coatings for food packaging. Environ. Sci. Pollut. Res. 18, 1108–1120.

Trier, X., Nielsen, N.J., Christensen, J.H., 2011b. Structural isomers of polyfluorinated di- and tri-alkylated phosphate ester surfactants present in industrial blends and in microwave popcorn bags. Environ. Sci. Pollut. Res. 18, 1422–1432.

Vestergren, R., Cousins, I.T., Trudel, D., Wormuth, M., Scheringer, M., 2008. Estimating the contribution of precursor compounds in consumer exposure to PFOS and PFOA. Chemosphere 73, 1617–1624.

Vestergren, R., Herzke, D., Wang, T., Cousins, I.T., 2015. Are imported consumer products an important diffuse source of PFASs to the Norwegian environment? Environ. Pollut. 198, 223–230.

Vojta, Š., Melymuk, L., Bečanová, J., Kukučka, P., Kohoutek, J., Klánová, J., 2016. Screening for halogenated flame retardants in european consumer products. Build. Mater. Wastes (in. prep).

Wang, Z., Cousins, I.T., Scheringer, M., Hungerbühler, K., 2013. Fluorinated alternatives to long-chain perfluoroalkyl carboxylic acids (PFCAs), perfluoroalkane sulfonic acids (PFSAs) and their potential precursors. Environ. Int. 60, 242–248.

Wang, Z.Y., Cousins, I.T., Scheringer, M., Buck, R.C., Hungerbuhler, K., 2014. Global emission inventories for C-4-C-14 perfluoroalkyl carboxylic acid (PFCA) homologues from 1951 to 2030, Part I: production and emissions from quantifiable sources. Environ. Int. 70, 62–75.

Wang, F., Shih, K., Ma, R., Li, X.-y, 2015a. Influence of cations on the partition behavior of perfluoroheptanoate (PFHpA) and perfluorohexanesulfonate (PFHxS) on wastewater sludge. Chemosphere 131, 178–183.

Wang, T.Y., Wang, P., Meng, J., Liu, S.J., Lu, Y.L., Khim, J.S., Giesy, J.P., 2015b. A review of sources, multimedia distribution and health risks of perfluoroalkyl acids (PFAAs) in China. Chemosphere 129, 87–99.

Washburn, S.T., Bingman, T.S., Braithwaite, S.K., Buck, R.C., Buxton, L.W., Clewell, H.J., Haroun, L.A., Kester, J.E., Rickard, R.W., Shipp, A.M., 2005. Exposure assessment and risk characterization for perfluorooctanoate in selected consumer articles. Environ. Sci. Technol. 39, 3904–3910.

Ye, F., Zushi, Y., Masunaga, S., 2015. Survey of perfluoroalkyl acids (PFAAs) and their precursors present in Japanese consumer products. Chemosphere 127, 262–268.

Zafeiraki, E., Costopoulou, D., Vassiliadou, I., Bakeas, E., Leondiadis, L., 2014. Determination of perfluorinated compounds (PFCs) in various foodstuff packaging materials used in the Greek market. Chemosphere 94, 169–176.

APPENDIX 10

Vojta, Šimon, Jitka Bečanová, Lisa Melymuk, Klára Komprdová, Jiří Kohoutek, Petr Kukučka, and Jana Klánová. 2017. "Screening for Halogenated Flame Retardants in European Consumer Products, Building Materials and Wastes." *Chemosphere* 168: 457-66. https://doi.org/10.1016/j.Chemosphere.2016.11.032

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Screening for halogenated flame retardants in European consumer products, building materials and wastes

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HIGHLIGHTS

- PBDE, HBCDD and NFR concentrations were determined in 137 consumer products.
- Differences in HFR content between product groups, esp. textiles, plastic and EEE.
- Differences in HFRs between recycled and virgin plastics.
- Recycled plastics contained low levels of a wide range of HFRs.

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ABSTRACT

To fulfill national and international fire safety standards, flame retardants (FRs) are being added to a wide range of consumer products and building materials consisting of flammable materials like plastic, wood and textiles. While the FR composition of some products and materials has been identified in recent years, the limited global coverage of the data and the large diversity in consumer products necessitates more information for an overall picture of the FR composition in common products/materials.

To address this issue, 137 individual samples of various consumer products, building materials and wastes were collected. To identify and characterize potential sources of FRs in indoor environment, all samples were analyzed for content of polybrominated diphenyl ethers (PBDEs), hexabromocyclododecanes (HBCDDs) and novel flame retardants (NFRs).

The most frequently detected were HBCDDs (85%), with the highest median concentration of Σ_4HBCDDs of 300 mg kg^{-1} in polystyrenes. The highest median concentration of Σ_{10}PBDEs was found in recycled plastic materials, reaching 4 mg kg^{-1}. The lowest concentrations were observed for NFRs, where the median of Σ_{12}NFRs reached 0.4 mg kg^{-1} in the group of electrical & electronic equipment wastes. This suggests that for consumer products and building materials that are currently in-use, legacy compounds still contribute to the overall burden of FRs. Additionally, contrasting patterns of FR composition in recycled and virgin plastics, revealed using principle component analysis (PCA), suggest that legacy flame retardants are reentering the market through recycled products, perpetuating the potential for emissions to indoor environments and thus for human exposure.

1. Introduction

Halogenated flame retardants (HFRs) are chlorine or bromine based compounds added to a broad range of commercial products to increase their fire resistance. They are used to meet fire safety regulations in flammable petroleum-based materials such as polymers – which have increased in production in the past several decades and which surround us in the form of clothing, furniture, electronics, cars and computers as well as in the form of combustible materials such as wood, paper and textiles.

Brominated compounds are frequently used organohalogenated flame retardants due to their higher trapping efficiency for free radicals and lower decomposition temperature in comparison with chlorinated compounds. Brominated flame retardants (BFRs) make up 22% of the global FR market (Posner et al., 2011), and despite recent shifts towards non-halogenated flame retardants such as

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organophosphate esters (OPEs), BFRs still dominate (>40 000 metric tons/year consumed globally in 2008, compared to ~ 20 000 tons/year for OPEs; (Posner et al., 2011)). Throughout the 1990s and early 2000s, the BFR market was dominated by polybrominated diphenyl ethers (PBDEs), hexabromocyclododecane (HBCDD) and tetrabromobisphenol A (TBBPA). However, due to environmental and human health concerns, two commercial mixtures of PBDEs were added to the annexes of Stockholm Convention on Persistent Organic Pollutants in 2009 (UNEP, 2009), while HBCDD was added in 2013 (UNEP, 2013). National bans on PBDEs (EU, 2004, 2010; Wager et al., 2012), Stockholm Convention restrictions and voluntary manufacturer phase-outs have led to a shift in current BFR usage and the consequent introduction of alternative flame retardants (FRs) to the market. The usage of HBCDD is projected to continue to 2020 (ECHA, 2014) while TBBPA and structurally-related compounds remain high-use BFRs. However, a diverse set of alternative HFRs, often structurally similar to the restricted/legacy compounds, is currently seeing wider use. These "novel" flame retardants (NFRs) include a wide variety of halogenated formulas that are new to the market or newly observed in the environment and often serve as replacements for banned PBDE formulations. Currently, there is a large number of different aromatic and cycloaliphatic HFRs with varying physico-chemical properties designed to achieve compatibility with polymers and stability during product lifetimes. Some of these substances (like TBBPA) can be covalently bound to materials during the production process, but most legacy FRs and NFRs are additive compounds, i.e. they are not bound to the material and thus have a greater potential for release from final products during their lifecycle. Many of these compounds are persistent, lipophilic and have been shown to accumulate in the environment (Besis and Samara, 2012; Covaci et al., 2009; de Wit et al., 2010; Law et al., 2003; Papachlimitzou et al., 2012; Venier et al., 2010).

HFRs are used in materials typically found in indoor environments (homes, vehicles and workplaces) where we typically spend over 20 h per day (Schweizer et al., 2007). The presence of indoor sources leads to elevated indoor concentrations of FRs, exacerbated by lower removal and degradation rates in the indoor environments due to low advective air movement, reduced photolysis and biodegradation and controlled climate. In addition to acting as a source of such compounds to outdoor environment (Bjorklund et al., 2012; Melymuk et al., 2016), the indoor environments also represent a significant potential exposure route for humans. In view of the toxicity of HFRs (Bruchajzer et al., 2011; Darnerud, 2003, 2008; Lyche et al., 2015; van der Ven et al., 2009), understanding indoor exposure is a key research concern. Indoor air and dust are frequently analyzed (Blanchard et al., 2014; Brommer et al., 2012; Brown et al., 2014; Cequier et al., 2014; Dodson et al., 2012; Fromme et al., 2014; Harrad et al., 2010; Shoeib et al., 2012; Sjödin et al., 2008; Takeuchi et al., 2014; Thuresson et al., 2012; Venier et al., 2016; Whitehead et al., 2015) to cover the main exposure routes: air inhalation and dust ingestion. Nevertheless, while the link between consumer product sources and measured indoor concentrations has been extensively investigated to define the mechanisms of this transfer (Rauert et al., 2015, 2014a, 2014b; Stubbings and Harrad, 2014), a comprehensive characterization of the levels in these sources is a missing piece. Although some work focusing on selected specific consumer products (Ionas et al., 2015; Kajiwara and Takigami, 2013; Peaslee et al., 2014; Rani et al., 2014; Stapleton et al., 2011, 2012) has been carried out, only a limited number of studies deal with a full range of products/materials commonly present in the indoor environment (Abbasi et al., 2016; Kumari et al., 2014). Knowledge of the concentrations of these chemicals in their presumed sources such as building materials and consumer products allows us to evaluate their contribution to concentrations of HFRs in indoor air and dust, and is thus a necessary component of human exposure assessment.

To determine the typical composition and levels of HFRs in indoor materials in the Czech Republic, 137 individual samples of construction materials, electrical and electronic devices, flooring, fabric, upholstery and other daily use materials including wastes and recycled products were investigated in order to identify and characterize potential sources of HFRs in indoor environments. All materials were analyzed for their content of PBDEs, HBCDDs and NFRs to provide a broad profile of the composition of HFRs in materials with different intended applications.

2. Materials and methods

2.1. Chemicals

The target analytes included 10 PBDEs (congeners 28, 47, 66, 85, 99, 100, 153, 154, 183 and 209), 4 HBCDD isomers and 12 NFRs (TBX, TBP-BAE, PBT, PBEB, TBP-DBPE, HBB, BTBPE, s-DDC-CO, a-DDC-CO, DBE-DBCH, TBCO, DBHCTD). See Table S 1 for further details. ^{13}C labeled internal standards of PBDEs and HBCDDs were added before extraction. All the analytical standards (native and ^{13}C labeled standards) were purchased from Wellington Laboratories Inc. (Canada). Pesticide residue grade solvents were obtained from Lab-scan (Poland) and Silica Gel 60 (70–230 mesh) from Merck (Germany).

2.2. Sample collection

Both new and used materials were included to cover the widest range of materials used in building construction, household equipment and cars interiors. These included mainly construction materials from the past three decades (including some supplied from a company specializing in "green" low-energy buildings), new and used electrical and electronic devices, flooring, fabric, upholstery and other daily-use materials. Samples of recycled materials were also included, as their contamination from primary materials remains unclear. New materials were purchased while older and used materials were collected from various sources. A subset of 126 of these materials was also analyzed for perfluoroalkyl substances (PFASs), presented elsewhere (Bečanová et al., 2016).

A total of 137 samples were split into four categories according to use and composition. See Supplementary Material and Table S 2 for categorization and sample details.

2.3. Sample preparation, extraction and instrumental analysis

Samples of solid materials were crushed, chopped or ground while samples of foam and fabrics were cut into small pieces about 1–5 mm in diameter. After grinding or cutting, 5 g of each sample was extracted with dichloromethane (DCM) using warm Soxhlet extraction (60 min warm Soxhlet followed by 30 min of solvent rinsing) in a B-811 extraction unit (Büchi, Switzerland). Concentrated extracts were cleaned up using H_2SO_4 modified silica column chromatography (elution with 40 mL DCM:n-hexane mixture 1:1). The eluate was concentrated using a stream of nitrogen in a TurboVap II (Caliper LifeSciences, USA) concentrator unit and transferred to a GC vial insert. Solvent was exchanged to nonane for gas chromatography/mass spectrometry (GC/MS) analysis and consequently to acetonitrile for liquid chromatography/mass spectrometry (LC/MS) analysis. Prior to instrumental analysis, injection standards (^{13}C BDE-77, 138) were added to all samples.

PBDEs and NFRs were analyzed by GC-MS/MS using a 7890A GC (Agilent, USA) equipped with a 15 m × 0.25 mm x 0.10 μm DB5 column (Agilent J&W, USA) coupled to a Quattro Micro MS (Waters, Micromass, UK). Separation and detection of HBCDDs was performed by HPLC-ESI-MS/MS using an Agilent 1100 (Agilent, USA) equipped with a LUNA C-18 endcapped (3 μm) 100 mm × 2 mm column (Phenomenex, Torrance, USA) coupled to a tandem mass spectrometer QTRAP 5500 (ABSciex, USA). See SM for more details.

2.4. *Quality assurance and quality control*

The efficiency of the extraction method was tested by repeated warm Soxhlet extraction cycles of selected materials. No target analytes were detected in subsequent extracts and thus we assumed that all extractable compounds were extracted in the first cycle.

The recoveries of individual compounds were determined using a set of pre-cleaned (8 h in acetone and 8 h in dichloromethane) polyurethane foam samples ($n = 8$) spiked with the native analytes prior to extraction. The recoveries of the 26 target compounds varied between 70% and 120% (Table S 2).

To calculate limits of detection (LODs) of individual compounds, eight laboratory procedural blanks were processed using empty extraction cartridges. An average value plus three times the standard deviation of the blanks was used as LOD, and the average of the procedural blanks was subtracted from the samples above LOD. When a compound was not detected in the blanks, instrumental LODs were used.

2.5. *Statistical analysis*

For statistical analysis, relevant compounds were summed according to their usage and properties, resulting in three groups, i.e. Σ_{10}PBDEs, Σ_4HBCDDs and Σ_{12}NFRs. Additionally, Σ_{10}PBDEs was subdivided either into the three PBDE commercial mixtures or considered as BDE-209 vs. Σ_9PBDEs. The Kruskal-Wallis test and multiple comparisons of mean ranks were used to identify statistical differences in BDE-209, Σ_9PBDEs, Σ_4HBCDDs and Σ_{12}NFRs among material categories according to use and composition. Principle component analysis (PCA) based on a correlation matrix was performed for visualization of relevant sums of FRs in relation to material type and usage. The sums were logarithmically transformed before analysis due to their log-normal distribution. As a first step, the PCA with all variables was computed and the most important compounds or their sums (estimates by eigenvalues) were selected. The final PCA was performed on selected sums of FRs (penta-BDE, octa-BDE, deca-BDE, Σ_4HBCDDs and Σ_{12}NFRs) and the others (Σ_9PBDEs, Σ_{10}PBDEs) were visualized in the same ordination space for interpretation purposes. Additionally, histograms created along the most important axes were used for description of variability and levels of FRs in individual consumer products. For PCA, all the non-detects were substituted by LOD/2. All statistical analyses were performed with STATISTICA (version 12, StatSoft, Inc.).

3. Results and discussion

3.1. *Overall distribution of FRs*

To address the overall compound distribution among all material samples, median concentrations and detection frequency were compared (Fig. 1). Six main clusters were identified based on the combination of concentration, detection frequency and compound structure.

The most frequently detected compounds (~80% of samples) and with considerably higher concentrations were the three main diastereoisomers of HBCDD (Formation I, Fig. 1). The fourth HBCDD diastereoisomer (δ-HBCDD) was detected only in 6.5% of samples, mostly alongside exceptionally high concentrations of other diastereoisomers, corresponding with its presence in the commercial HBCDD mixture only as a trace impurity (Arsenault et al., 2007). Generally, HBCDDs had the highest maximum concentrations of all the compounds, reaching 4.4 g kg^{-1} of α–HBCDD in a sample of sealant foam. Isomer profiles of HBCDDs in these consumer products are discussed in greater detail in Okonski et al. (Submitted). The two isomers of Dechlorane Plus were detected in ~30% of samples at concentrations up to 40 mg kg^{-1} (Formation II). The PBDE congeners formed two formations. Formation III is formed by the main components of the penta-BDE commercial mixture (BDEs 47, 99 and 100), with a detection frequency of about 20–30%, other penta-BDE congeners (BDEs 28, 66 and 85) and BDE-153 and 154, which are present in both penta and octa-BDE commercial mixtures. Together with BDE-183, BDE-153 and 154 formed a sub-set of octa-BDE components (Formation IV) with a detection frequency around 10%. The last two formations belong to the NFRs. Bromobenzenes (Formation V, Fig. 1), including HBB, PBT, PBEB and TBX, had low concentrations and low detection frequencies. Formation VI contained the tribromophenoxy and related compounds: TBP-DBPE, TBP-BAE and BTBPE. TBP-DBPE is a flame retardant produced in Germany in 1980s and TBP-BAE is a transformation product of TBP-DBPE (Ma et al., 2012). TBP-DBPE and TBP-BAE were infrequently detected (<5%), while considerable amounts of BTBPE were found in electronics-related categories.

The remaining three structurally exceptional NFRs were found only in samples containing recycled plastic materials or mixed waste of small electronic devices, which means materials containing broad spectra of compounds. Although DBHCTD exhibits relatively high median concentrations, it was detected in only four samples. Considering the lack of information about this compound containing both bromine as well as chlorine atoms, its rare occurrence was expected. A similar scenario applies for DBE-DBCH and TBCO. Finally, BDE-209, the main component of the deca-BDE commercial mixture, was found in a relatively small number of samples, in contrast to its presumed use in broad range of polymers (Alaee et al., 2003; Stubbings and Harrad, 2014). However, the maximum amounts were the second highest (after HBCDDs) of all the compounds, reaching 0.6 g kg^{-1} in a sample of pipe insulation.

3.2. *FR concentrations in individual product categories*

All samples were categorized according to their usage or composition (see Table S 1), as described in section 2.2. Of all analyzed materials, only two (linoleum floor covering and a polystyrene disposable food box) did not contain any of the target chemicals. An additional ten samples had only one detectable FR. On the other hand, four samples contained more than 18 out of 26 (70%) target compounds. These materials included recycled plastics and heavily used polyurethane foam (further discussed in section 3.4).

An overview of the basic descriptive statistics for individual product categories is displayed in Table 1. Generally, among all categories, Σ_{10}PBDEs ranged from 0.03 μg kg^{-1} to 600 000 μg kg^{-1}, while Σ_4HBCDDs ranged from 0.7 μg kg^{-1} to 5 800 000 μg kg^{-1} and Σ_{12}NFRs reached up to 65 000 μg kg^{-1}.

3.3. *Household equipment*

In textiles (Group 1A, containing 24 samples), the only notable FRs were penta-BDE congeners and HBCDDs, both with detection frequencies over 80%. Interestingly, penta-BDEs were dominated by BDE-28, found in 88% of the samples, while two main components of the commercial mixture (BDE-47, 99) were detected only in 29 and 13%, respectively. This may indicate sorption of more volatile congeners to fabrics during their use period, as suggested by Saini

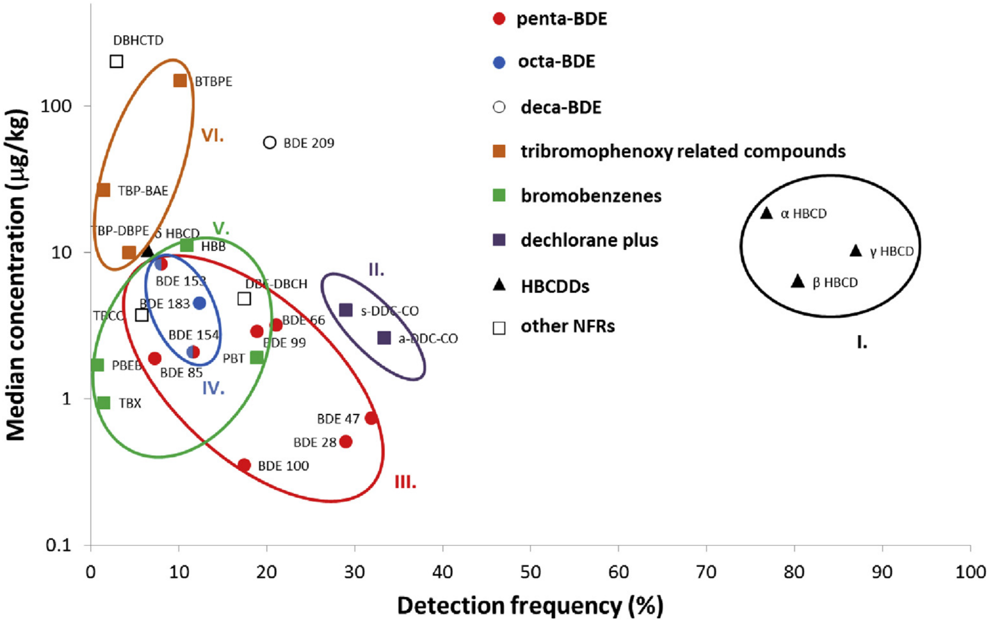

Fig. 1. Median concentration vs. detection frequency of individual HFRs in all analyzed samples.

et al. (2016). Although frequently detected, the concentrations of PBDEs were low, with a median of Σ_9PBDEs of 0.616 µg kg^{-1} only, which contrasts with findings in North America (Stapleton et al., 2012), potentially reflecting market differences. Surprisingly, we found very low amounts of BDE-209 and HBCDDs, major HFRs used in textiles in USA and Japan (Shaw et al., 2014). The highest concentration of Σ_4HBCDDs (544 µg kg^{-1}) was in a curtain sample, while BDE-209 was only detected in one textile sample at 1.66 µg kg^{-1}.

Although the concentrations were very low, a more consistent pattern of penta-BDE was observed in 13 floor covering samples (group 1B), dominated by BDE-47, 99 and 100, while no octa- and deca-BDEs were detected. Significantly higher ($p < 0.05$) were the amounts of Σ_4HBCDDs, especially in three samples of carpets, reaching a maximum of 1140 µg kg^{-1}. NFR concentrations were negligible; only DDC-CO was detected.

In addition to penta-BDEs, similar concentrations of octa-BDEs and a few detections of BDE-209 were found in electric and electronic equipment (EEE, Group 1C, 19 samples), reflecting the higher portion of plastic materials in this group, e.g., samples of keyboards, screens, TV sets and printed circuit boards. The presence of octa-BDE in EEE reflects its typical use for flame retarding of thermoplastics like acrylonitrile butadiene styrene (ABS) resins, widely used in such applications (Stubbings and Harrad, 2014). Additionally, BTBPE was also detected in EEE, reaching a maximum of 317 µg kg^{-1} which corresponds with the use of BTBPE as a replacement for octa-BDE in ABS, high impact polystyrene (HIPS) and other thermoplastics (Covaci et al., 2011). Lower, but still notable, were the amounts of DDC-COs, while concentrations and detection frequency of HBCDDs were less than in floor coverings or textiles.

Finally, in the plastics group (1D), two out of the six samples had substantial concentrations of a wide variety of target chemicals (18 and 22 out of 26 analytes), typically at higher levels (medians of hundreds to thousands µg.kg^{-1}) than in the other groups of household equipment.

3.4. *Building materials*

Building materials were dominated by HBCDDs and BDE-209; no other flame retardants had any significant levels. The concentrations of HBCDDs far exceeded the levels in household plastics (Groups 1C and 1D) and the only other non-recycled materials with comparable levels of HBCDDs were four carpet samples from the floor covering group (1B), which could also be considered to overlap with the building material category.

Only four out of 14 samples of oriented strand board (OSB) and wood (Group 2A) exhibited elevated concentrations of Σ_4HBCDDs, with a maximum of 483 µg kg^{-1}. Such a low abundance of FRs in OSB was quite unexpected, considering their use as building materials and the content of additive adhesives and resins.

Insulation materials (Group 2B, 18 samples) were dominated by BDE-209, exceeding 1000 µg kg^{-1} in three samples of glass fiber, foam glass and phenolic foam insulation and 500 mg kg^{-1} in a sample of tube insulation foam, discussed in section 3.4. The only other notable sample from this group was stone-fiber insulation containing 8470 µg kg^{-1} of Σ_4HBCDDs.

The highest levels of HBCDDs were found in mounting and sealing foams (Group 2C, 8 samples). With one exception, discussed in section 3.4., maxima of hundreds of mg kg^{-1} for Σ_4HBCDDs were common in this group, as well as in facade materials (2D), polystyrene (2E) and heating, ventilation and air conditioning (HVAC) components (2F), These four groups had the highest levels of HBCDDs of all the products. The highest median of 299 mg kg^{-1} was found in polystyrene (group 2E, 5 samples) which agrees with typical use of HBCDDs (Alaee et al., 2003; Marvin et al., 2011; Stubbings and Harrad, 2014) as well as with previously published data (Rani et al., 2014). Additionally, HVAC components (Group 2F, 5 samples) contained considerable amounts (350–5890 µg kg^{-1}) of BDE-209.

3.5. *Car interior materials and WEEE*

These two heterogeneous groups had no BDE-209 and low concentrations of HBCDDs, contrasting with frequent detection and relatively higher levels of NFRs and other PBDEs. This pattern was

Table 1
Summary of results for four FR groups in individual product categories.

Category	Group		BDE-209				Σ_9PBDEs				Σ_4HBCDDs				Σ_{12}NFRs			
			n_{total} (n_{detect})	Min - max (median) µg kg^{-1}	Median µg kg^{-1}	Detection frequency %	n_{total} (n_{detect})	Min - max (median) µg kg^{-1}	Median µg kg^{-1}	Detection frequency %	n_{total} (n_{detect})	Min - max (median) µg kg^{-1}	Median µg kg^{-1}	Detection frequency %	n_{total} (n_{detect})	Min - max (median) µg kg^{-1}	Median µg kg^{-1}	Detection frequency %
Household equipment	Textiles	1A	24 (1)	1.66–1.66 (1.66)	10.5	14.5	24 (21)	0.119–155 (0.616)	0.720	71.0	24 (20)	4.90–544 (21.2)	19.1	85.5	24 (8)	0.588–3910 (3.53)	4.49	50.0
	Floor coverings	1B	13 (2)	3.77–10.5 (7.12)			13 (6)	0.377–6.73 (1.60)			13 (12)	2.85–1140 (42.5)			13 (6)	0.541–23.7 (1.43)		
	EEE	1C	19 (3)	2.20–37.8 (2.47)			19 (15)	0.069–178 (0.721)			19 (17)	0.654–288 (4.65)			19 (13)	0.115–359 (40.7)		
	Plastics	1D	6 (3)	17.6–17 500 (3360)			6 (2)	708-3910 (2310)			6 (4)	12-1160 (367)			6 (4)	0.138–18 600 (497)		
Building materials	OSB and wood	2A	14 (0)	ND (N/A)	80.0	32.8	14 (2)	0.044–0.132 (0.088)	0.324	19.0	14 (14)	9.18–483 (34.0)	108	94.8	14 (8)	0.307–36.1 (5.58)	4.05	48.3
	Insulation materials	2B	18 (13)	1.77 –626 000 (73.4)			18 (5)	0.324–20.8 (1.02)			18 (16)	2.47–8470 (63.2)			18 (7)	1.28–22.0 (2.10)		
	Mounting and sealing foam	2C	8 (0)	ND (N/A)			8 (0)	ND (N/A)			8 (8)	31.4–5 810 000 (1340)			8 (6)	0.526–64.8 (5.71)		
	Facade materials	2D	8 (3)	2.42–79.9 (70.2)			8 (1)	0.184–0.184 (0.184)			8 (8)	6.47 –233 000 (179)			8 (2)	3.96–5.38 (4.67)		
	Polystyrene	2E	5 (0)	ND (N/A)			5 (1)	0.085–0.085 (0.085)			5 (4)	33.4 –468 000 (299 000)			5 (2)	0.769–45.5 (23.1)		
	HVAC components	2F	5 (3)	350-5890 (394)			5 (2)	0.217–18.9 (9.58)			5 (5)	21.3 –531 000 (16,300)			5 (3)	0.845–1090 (9.46)		
Car interior materials	–	3	10 (0)	ND (N/A)	(N/A)	0.00	10 (8)	0.034–54.0 (1.22)	1.22	80.0	10 (10)	0.709–134 (21.7)	21.7	100	10 (9)	2.13–121 (11.3)	11.3	90.0
WEEE	–	4	8 (0)	ND (N/A)	(N/A)	0.00	8 (8)	0.068–32 700 (144)	144	100	8 (8)	1.49–485 (12.9)	12.9	100	8 (7)	5.74–64 600 (426)	426	87.5

especially clear in waste electrical and electronic equipment (WEEE), consisting of homogenized mixed electronic waste samples. Both penta and octa-BDE congeners were frequently detected, with a median concentration of Σ_9PBDEs of 1.22 µg kg^{-1} for car interior materials and 144 µg kg^{-1} for WEEE. Beside the main use of penta-BDE formulations in polyurethane foams, textile and automotive applications (European Chemicals Bureau, 2000), considerable amounts detected in electronic waste also suggest their use in plastics. As car interior materials were considered something of a crossover category as it was composed of ten samples of plastics, textile and upholstery materials from two car interiors, relatively low amounts of target flame retardants, comparable to household equipment category (Table 1) were expected. The lack of detection of BDE-209 in contrast with frequent detection of penta- and octa-BDE congeners may indicate differences in the timeline of deca-BDE usage, suggesting that the main stocks of deca-BDE did not reach the waste disposal/recycling point of the product lifecycle at the time of sampling.

Together with plastics (1D) and EEE (1C), the highest concentrations of NFRs (Table 1) were observed in the WEEE category. NFRs were dominated by DDC-COs and BTBPE, which may reflect wide use of DDC-CO as a flame retardant in electrical hard plastic connectors in televisions and computer monitors, wire coatings, and furniture (Betts, 2006; Hoh et al., 2006; Sverko et al., 2011) and use of BTBPE as a replacement for octa-BDE in ABS (Hoh et al., 2005; WHO, 1997). Low levels of bromobenzenes were also detected in plastics, EEE and WEEE, corresponding with the fact that these compounds were produced in relatively low amounts and used locally in specific applications. HBB was used mainly in Japan as an additive flame retardant to paper, wood, textiles, electronic and plastic goods (Covaci et al., 2011). In our samples, HBB was found mainly in WEEE, one sample of air conditioning heat exchanger and the abovementioned heavily used polyurethane foam. PBT, produced in Israel, USA and China and used in polyethylene, polypropylene, polystyrene, latex, textile, rubber and ABS (Covaci et al., 2011) was found in very low amounts across all categories, but mainly in plastics, polystyrene, car interior materials and WEEE. Only one sample of WEEE contained PBEB, which agrees with the fact that PBEB was produced mainly in 1970s and 1980s in Israel and USA and used in circuit boards, textiles, adhesives, wire and cable coatings and polyurethane foam (Covaci et al., 2011; Hoh et al., 2005; WHO, 1997).

3.6. *Identification of FR patterns by PCA*

When PCA was used to investigate the relationship and overall distribution of sums of FRs among all product samples, two main gradients distributed along two PCA axes explaining a high amount of variability (70%) were distinguished and identified with corresponding product classification. The visualization of the distribution of products between these axes according to the categorization is displayed in Fig. 2a. The directions of compound sums are plotted together with ellipses indicating 95% of the samples associated with the corresponding category. The gradient distributed along the first axis (Factor I) reflects increasing concentrations of sums of penta- and octa-BDEs, NFRs and Σ_9PBDEs. Increasing concentration gradients of these compounds were found in WEEE and also in household equipment, with both having similar patterns of relatively low concentrations of a wide variety of these chemicals. Although less distinct due to even lower concentrations, a similar pattern was observed for car interior materials. The second axis (Factor II) represents the correlation with the gradient of the Σ_4HBCDDs and BDE-209. A contrasting pattern of high concentrations of only these two compounds was found in building materials. Moreover, while an even stronger relationship was identified between the subgroup of insulation (2B) and BDE-209, samples of household equipment were also distributed along the gradient of Σ_{10}PBDEs, reflecting the subgroup of plastics (1D) where recycled materials containing both lower and fully brominated PBDEs were present.

With respect to the individual samples, the PCA revealed two distinct patterns. We hypothesize that this corresponds to two types of plastics: recycled and virgin (non-recycled) plastics. When applying these findings to the same PCA ordination space grouped by material composition (textile, wood, plastics, paper) rather than product use, these two different plastics can be clearly identified (Fig. 2b).

Use of penta- and octa-BDEs declined since the early 2000s (Stapleton et al., 2011) and thus their presence in recycled materials more likely comes from original materials or usage period than from intended use in the sampled consumer product/material. However, recently restricted BDE-209 and HBCDDs (Stubbings and Harrad, 2014) are more likely to be found in primary use. The low levels of NFRs suggest that no consumer product with direct application of NFRs was included in this study; they follow the pattern of penta- and octa-BDEs reflecting their low production volumes and rather local or limited use in specific applications. Generally the lower concentrations of all compounds occurred in non-plastic materials like textile, paper and wood (Fig. 2b).

Overall, the PCA suggests three main material types according to flame retardant content and composition (Fig. 2b). Plastic materials can be divided into (1) virgin plastics with relatively higher amounts of one or a few particular FRs, added intentionally to provide a flame retarding effect and (2) recycled plastics with relatively lower amounts of a wide spectra of FRs not added intentionally, but rather coming from the original source materials. The last type (3) contains all remaining non-plastic materials with negligible target FRs.

The distribution of FRs according to product age was also considered; the highest concentrations in insulation (2B) and household equipment were found mainly in new samples (produced after 2010), however middle and low concentrations were independent of product age.

For a more comprehensive description of FR content, the histograms of individual groups were visualized for PCA axes (Fig. 3) and supplemented by a Kruskal-Wallis test for identification of intra-category statistical significance (Table S 3). In accordance with concentration gradients displayed in Fig. 2, the individual factors in Fig. 3 show the frequency of specific concentrations of the samples from each particular category for the compound gradient correlating with the corresponding factor. Thus, the broad histograms along the Factor I axis indicate large concentration ranges in products with higher sums of penta and octa-BDE, NFRs and Σ_9PBDEs, especially in WEEE (4), plastics (1D) and the fraction of EEE (1C) made of recycled plastics (extracted as a subgroup from EEE and identified as EEE (1C)R in Fig. 3). Similarly, the contribution of recycled plastics to the product groups with higher concentrations on the Factor I axis is clearly revealed in combination with Fig. 2b. The remaining product groups, especially from the category of building materials (2) have rather lower concentrations of these compounds.

Histograms along the second axis, representing the distribution of concentrations of BDE-209 and HBCDDs, indicate non-recycled plastics (Fig. 3) and clearly reveal of the dominance of building materials. Five main groups of HVAC system components (2F), insulation materials (2B), mounting and sealant foams (2C), facade materials (2D) and polystyrene (2F) exhibited a typical non-recycled plastics pattern.

Due to the high variability of results, no significant intra-category differences between individual material groups were

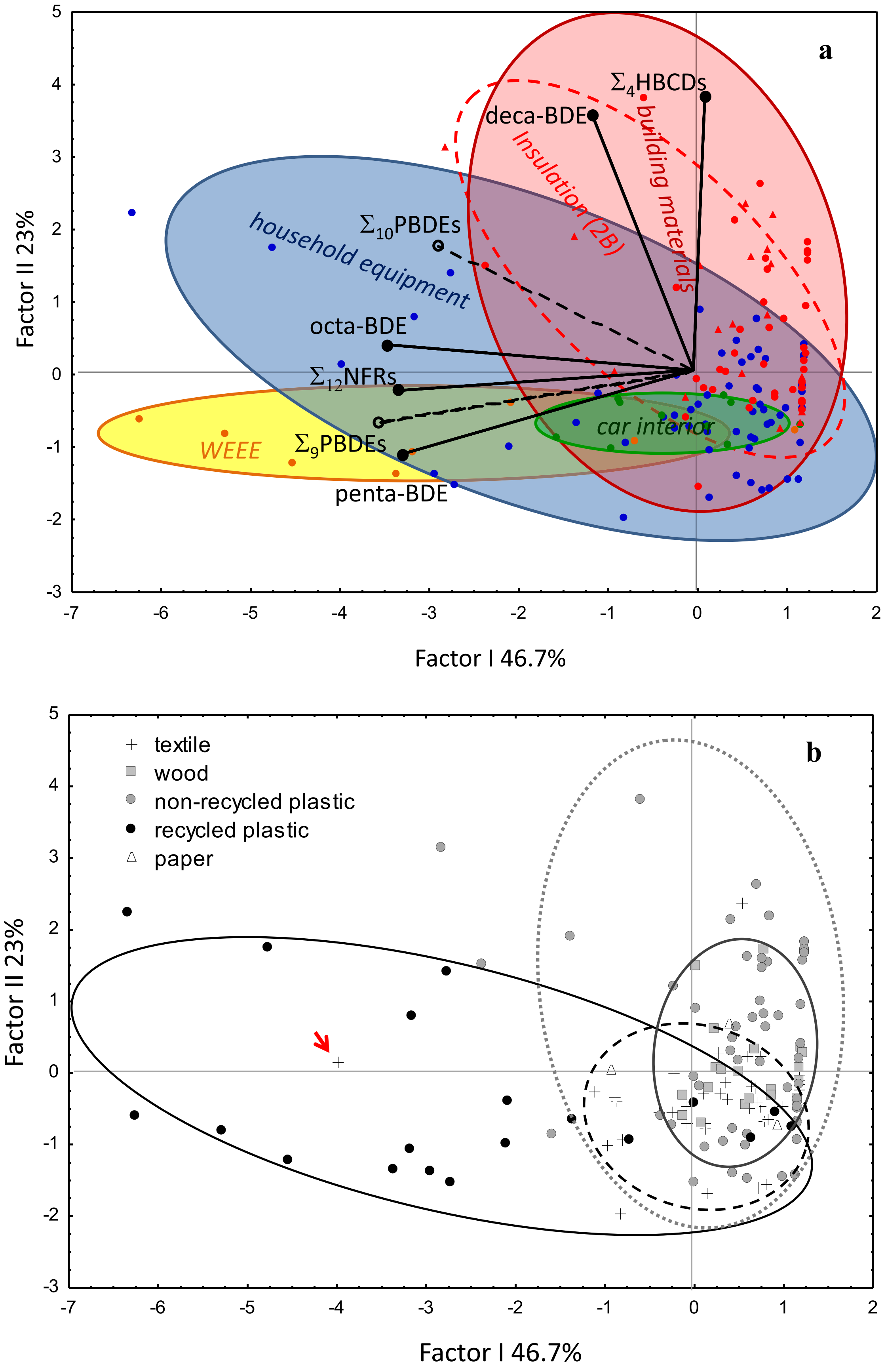

Fig. 2. Results of PCA showing (a) the relationships and main directions of FR sums and distribution variability among main categories of usage and (b) distribution variability among different material types in the same ordination space. Ellipses indicate 95% of samples from different categories. Solid lines indicate the direction of increasing compound concentrations. Dashed lines show supplementary variables (only for visualization, not PCA calculations). The red arrow highlights the sample of highly used polyurethane foam, discussed in section 3.4. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

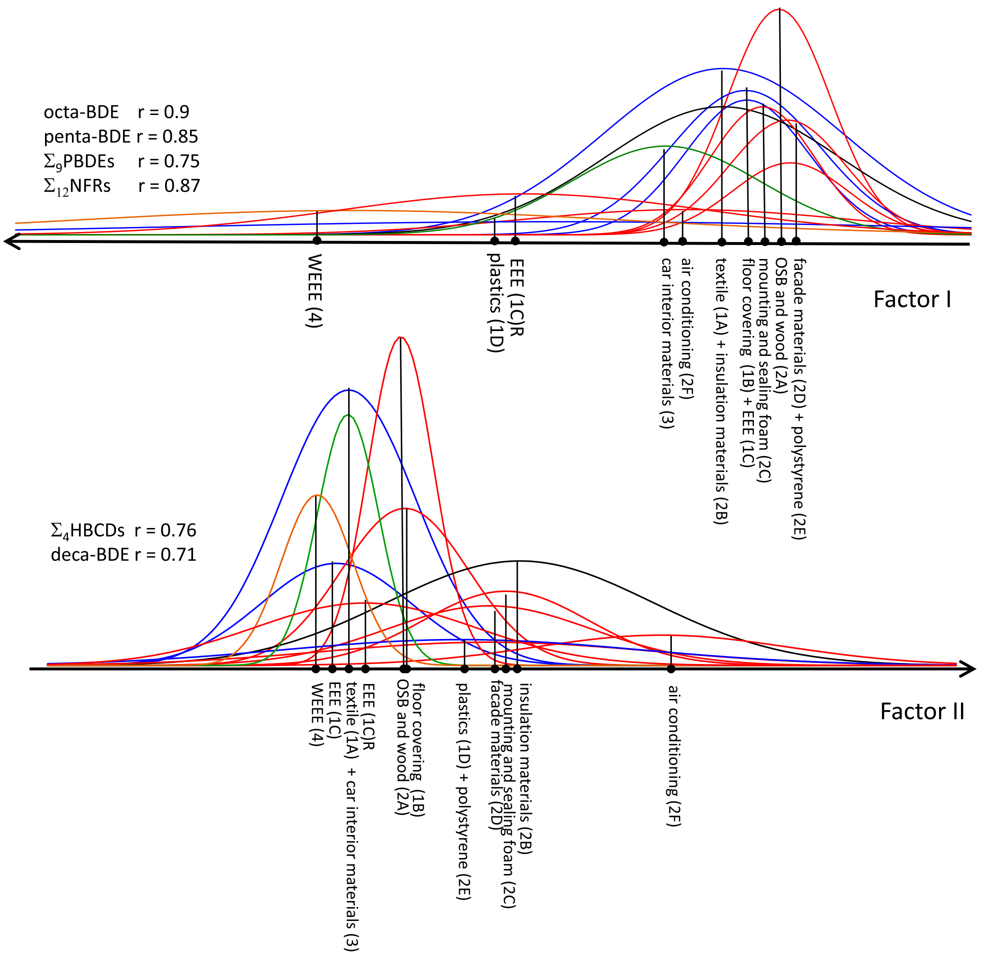

Fig. 3. Histograms of samples of consumer products along first and second PCA axes (same as Fig. 2). Arrows indicate the direction of increasing compound concentrations (right to left for Factor I and left to right for Factor II) and r signifies correlation of compound with axis.

found (Kruskal-Wallis, $p < 0.05$, Table S 3), except for the case of BDE-209, where the exceptionally high concentrations in insulation materials (2B) significantly differed from OSB and wood (2A), mounting and sealant foam (2C) and polystyrene (2E), which had no positive detections. Similarly, Σ_9PBDEs was below detection in mounting and sealant foam (2C), significantly differing from other groups.

3.7. *FRs concentrations in individual samples*

As suggested by the PCA, samples of household equipment, car interior materials and WEEE frequently consisted of recycled plastic materials containing a wide variety of FRs. The two samples most representative of this pattern were plastics which contained 70 and 85% of target analytes. The first sample was a DVD case and the second one was a poorly engineered recycled plastic part with visible chunks of original materials. Although the concentrations of PBDEs in these samples were the highest of all household equipment, even the highest level of 21 400 µg kg^{-1} of Σ_{10}PBDEs in the recycled plastic part represents only 0.002% of the material weight, which is much lower than amounts typically applied (3–18% for penta-BDE and 10–18% for octa-BDE) to the materials when PBDEs are intentionally used as FRs (Alaee et al., 2003; Gallen et al., 2014; Stapleton et al., 2011; Stubbings and Harrad, 2014). Similar amounts of PBDEs were previously reported in electronics components and other plastics (Chen et al., 2010; Kajiwara et al., 2011; Kumari et al., 2014). On the other hand, consistently higher amounts were found in Australian consumer products (Gallen et al., 2014), North American and Chinese baby products (Chen et al., 2009; Stapleton et al., 2011) and Japanese electronics (Kajiwara et al., 2011).

Another sample with many FRs was a heavily used seven-year-old polyurethane foam, containing 21 out of the 26 (81%) target chemicals. Although this resembles a typical pattern of recycled plastics (Fig. 2b), we assume that this contamination is more likely originating from the usage period of the material rather than from its source materials, and demonstrates the typical passive sampling behavior of foam, with a higher contribution of lower molecular weight compounds: the sum of penta-BDE congeners was 129 µg kg^{-1}, the sum of octa-BDEs was 25.3 µg kg^{-1} and BDE-209 was only 1.66 µg kg^{-1}. Since these levels are far below penta-BDE amounts typically used in polyurethane foam (Stubbings and Harrad, 2014), we assume this sample was not originally penta-BDE treated. Furthermore, there is the possibility that PUF and similar materials could change from acting as a chemical sink to a secondary source when equilibrium is affected by changes of surrounding conditions, thus emitting legacy FRs to the indoor environment at a later stage (Zhang et al., 2009).

As described previously, building materials were dominated by

HBCDDs and BDE-209, typically present in non-recycled plastic materials. Of all the FRs in this study, the highest amount was found for Σ_4HBCDDs, which reached 5.81 g kg^{-1} (0.6% w/w) in a sample of sealing foam. This is close to the minimum concentration added to expanded or extruded polystyrene (0.8–4.0%; (Alaee et al., 2003; Marvin et al., 2011; Stubbings and Harrad, 2014). The highest concentration of BDE-209 from all samples (626 mg kg^{-1}), found in tube insulation foam, represents 0.06% of the material mass, which is much less than the typically added levels of deca-BDE of 10–25% (Weil and Levchik, 2008). However, similar amounts have been reported in samples of textiles (Ionas et al., 2015), where the flame retarding function was obviously also neglected.

4. Conclusions

Low levels of FRs were found in most consumer products and building materials, usually at levels of units of μg kg^{-1}, which are far below levels which would ensure a flame retarding function. This finding is consistent with previous studies (Abbasi et al., 2016; Ionas et al., 2015). The highest mass fraction was found for HBCDDs (0.6% w/w) in a sealing foam sample, and the highest level of BDE-209 (0.06% w/w) was found in a tube insulation sample. A maximum of 0.003% w/w of both penta and octa-BDEs was found in a sample of mixed electronic waste from a waste sorting system. Similarly, the NFR with the highest observed concentrations, Dechlorane Plus, typically added in amounts of 10–35%, was found at only 0.006% of material mass in a sample also originating from a waste sorting system.

Based on revealed FR composition patterns, two main types of polymeric materials were identified. The first was virgin plastic containing elevated levels of either one or a few FRs, typically BDE-209 and HBCDDs, which we hypothesize had been added intentionally to provide a flame retarding effect. This pattern was predominantly observed in insulation, mounting and facade materials, polystyrene and air conditioning system components. The second type was recycled plastic containing lower amounts of a wide variety of FRs. This pattern was most pronounced in the case of EEE, WEEE and car interior materials. Since FR concentrations in these types of products were far below levels capable of providing any flame retarding effect, this suggests unintended accumulation of FRs due to recycling of originally flame retarded materials or uptake from the environment during product use. Although the overall concentrations of FRs in consumer products and materials was lower than expected, this suggests both that recycled and virgin plastics may both act as sources to indoor environments, possibly releasing a wide suite of FRs, including those that are not in current use. Thus, recycled plastics may be inadvertently and unintentionally contaminating the indoor environment with unwanted and in some cases even currently banned synthetic chemicals with no desired function. This presents a challenge, as recycling of plastics is generally a desirable activity, but re-contamination of indoor environments with currently banned FRs may be a negative side-effect, and this requires further attention. Moreover, recycled plastics can contribute a mixture of FRs (and potentially other plastic additives) to indoor environments, potentially necessitating consideration of mixture effects rather than only individual compounds. Due to their reintroduction from existing stocks (Abbasi et al., 2014) throughout the recycling phase of the original materials, these chemicals may be observed not only long after legislative restrictions have been imposed, but also in places where they were never employed for any primary purpose. Gearhart and Miller (2016) indicated that the quality of the recycling process might also influence the amount and availability of FRs in final recycled product, suggesting bigger concerns in case of cheaper plastics. Overall, there is a lack of knowledge regarding the recycling process, the extent of this issue and potential for indoor contamination.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.chemosphere.2016.11.032.

References

Abbasi, G., Buser, A.M., Soehl, A., Murray, M.W., Diamond, M.L., 2014. Stocks and flows of PBDEs in products from use to waste in the U.S. And Canada from 1970 to 2020. Environ. Sci. Technol. 49, 1521–1528.

Abbasi, G., Saini, A., Goosey, E., Diamond, M.L., 2016. Product screening for sources of halogenated flame retardants in Canadian house and office dust. Sci. Total Environ. 545, 299–307.

Alaee, M., Arias, P., Sjodin, A., Bergman, A., 2003. An overview of commercially used brominated flame retardants, their applications, their use patterns in different countries/regions and possible modes of release. Environ. Int. 29, 683–689.

Arsenault, G., Konstantinov, A., Marvin, C.H., MacInnis, G., McAlees, A., McCrindle, R., Riddell, N., Tomy, G.T., Yeo, B., 2007. Synthesis of the two minor isomers, delta- and epsilon-1,2,5,6,9,10-hexabromocyclododecane, present in commercial hexabromocyclododecane. Chemosphere 68, 887–892.

Bečanová, J., Melymuk, L., Vojta, Š., Komprdová, K., Klánová, J., 2016. Screening for perfluoroalkyl acids in consumer products, building materials and wastes. Chemosphere 164, 322–329.

Besis, A., Samara, C., 2012. Polybrominated diphenyl ethers (PBDEs) in the indoor and outdoor environments-A review on occurrence and human exposure. Environ. Pollut. 169, 217–229.

Betts, K.S., 2006. A new flame retardant in the air. Environ. Sci. Technol. 40, 1090–1091.

Bjorklund, J.A., Thuresson, K., Cousins, A.P., Sellstrom, U., Emenius, G., de Wit, C.A., 2012. Indoor air is a significant source of tri-decabrominated diphenyl ethers to outdoor air via ventilation systems. Environ. Sci. Technol. 46, 5876–5884.

Blanchard, O., Glorennec, P., Mercier, F., Bonvallot, N., Chevrier, C., Ramalho, O., Mandin, C., Le Bot, B., 2014. Semivolatile organic compounds in indoor air and settled dust in 30 french dwellings. Environ. Sci. Technol. 48, 3959–3969.

Brommer, S., Harrad, S., Van den Eede, N., Covaci, A., 2012. Concentrations of organophosphate esters and brominated flame retardants in German indoor dust samples. J. Environ. Monit. 14, 2482–2487.

Brown, F.R., Whitehead, T.P., Park, J.-S., Metayer, C., Petreas, M.X., 2014. Levels of non-polybrominated diphenyl ether brominated flame retardants in residential house dust samples and fire station dust samples in California. Environ. Res. 135, 9–14.

Bruchajzer, E., Frydrych, B., Sporny, S., Szymanska, J.A., 2011. The effect of short-term intoxication of rats with pentabromodiphenyl ether (in mixture mimic commercial products). Hum. Exp. Toxicol. 30, 363–378.

Cequier, E., Ionas, A.C., Covaci, A., Maria Marce, R., Becher, G., Thomsen, C., 2014. Occurrence of a broad range of legacy and emerging flame retardants in indoor environments in Norway. Environ. Sci. Technol. 48, 6827–6835.

Chen, S.-J., Ma, Y.-J., Wang, J., Tian, M., Luo, X.-J., Chen, D., Mai, B.-X., 2010. Measurement and human exposure assessment of brominated flame retardants in household products from South China. J. Hazard. Mater. 176, 979–984.

Chen, S.J., Ma, Y.J., Wang, J., Chen, D., Luo, X.J., Mai, B.X., 2009. Brominated flame retardants in Children's toys: concentration, composition, and Children's exposure and risk assessment. Environ. Sci. Technol. 43, 4200–4206.

Covaci, A., Harrad, S., Abdallah, M.A.E., Ali, N., Law, R.J., Herzke, D., de Wit, C.A., 2011. Novel brominated flame retardants: a review of their analysis, environmental fate and behaviour. Environ. Int. 37, 532–556.

Covaci, A., Voorspoels, S., Abdallah, M.A.E., Geens, T., Harrad, S., Law, R.J., 2009. Analytical and environmental aspects of the flame retardant tetrabromobisphenol-A and its derivatives. J. Chromatogr. A 1216, 346–363.

Darnerud, P.O., 2003. Toxic effects of brominated flame retardants in man and in wildlife. Environ. Int. (The State-of-Science and Trends of BFRs in the Environment) 29, 841–853.

Darnerud, P.O., 2008. Brominated flame retardants as possible endocrine disrupters. Int. J. Androl 31, 152–160.

de Wit, C.A., Herzke, D., Vorkamp, K., 2010. Brominated flame retardants in the Arctic environment - trends and new candidates. Science of the Total

Environment 408, 2885–2918.

Dodson, R.E., Perovich, L.J., Covaci, A., Van den Eede, N., Ionas, A.C., Dirtu, A.C., Brody, J.G., Rudel, R.A., 2012. After the PBDE phase-out: a broad suite of flame retardants in repeat house dust samples from California. Environ. Sci. Technol 46, 13056–13066.

ECHA, 2014. Template for Third Party Submission of Information on Alternatives for Applications for Authorisation HBCD Use in EPS for Building Applications. ICL-IP Europe BV.

EU, 2004. Directive 850/2004 of the European Parliament and of the Council.

EU, 2010. Directive 757/2010 of the European Parliament and of the Council.

European Chemicals Bureau, 2000. European Union Risk Assessment Report, Diphenyl Ether, Pentabromo Derivative (Pentabromodiphenylether).

Fromme, H., Hilger, B., Kopp, E., Miserok, M., Volkel, W., 2014. Polybrominated diphenyl ethers (PBDEs), hexabromocyclododecane (HBCD) and “novel” brominated flame retardants in house dust in Germany. Environ. Int 64, 61–68.

Gallen, C., Banks, A., Brandsma, S., Baduel, C., Thai, P., Eaglesham, G., Heffernan, A., Leonards, P., Bainton, P., Mueller, J.F., 2014. Towards development of a rapid and effective non-destructive testing strategy to identify brominated flame retardants in the plastics of consumer products. Science of the Total Environment 491, 255–265.

Gearhart, J., Miller, G., 2016. Evidence of Electronic Waste, Including Flame Retardants Chemicals, Recycled into New Consumer Products. BFR2016.

Harrad, S., de Wit, C.A., Abdallah, M.A.E., Bergh, C., Bjorklund, J.A., Covaci, A., Darnerud, P.O., de Boer, J., Diamond, M., Huber, S., Leonards, P., Mandalakis, M., Oestman, C., Haug, L.S., Thomsen, C., Webster, T.F., 2010. Indoor contamination with hexabromocyclododecanes, polybrominated diphenyl ethers, and perfluoroalkyl compounds: an important exposure pathway for people? Environ. Sci. Technol 44, 3221–3231.

Hoh, E., Zhu, L.Y., Hites, R.A., 2005. Novel flame retardants, 1,2-bis(2,4,6-tribromophenoxy)ethane and 2,3,4,5,6-pentabromoethylbenzene, in United States' environmental samples. Environ. Sci. Technol 39, 2472–2477.

Hoh, E., Zhu, L.Y., Hites, R.A., 2006. Dechlorane plus, a chlorinated flame retardant, in the Great Lakes. Environ. Sci. Technol 40, 1184–1189.

Ionas, A.C., Gomez, A.B., Uchida, N., Suzuki, G., Kajiwara, N., Takata, K., Takigami, H., Leonards, P.E.G., Covaci, A., 2015. Comprehensive characterisation of flame retardants in textile furnishings by ambient high resolution mass spectrometry, gas chromatography-mass spectrometry and environmental forensic microscopy. Environmental Research 142, 712–719.

Kajiwara, N., Noma, Y., Takigami, H., 2011. Brominated and organophosphate flame retardants in selected consumer products on the Japanese market in 2008. Journal of Hazardous Materials 192, 1250–1259.

Kajiwara, N., Takigami, H., 2013. Emission behavior of hexabromocyclododecanes and polybrominated diphenyl ethers from flame-retardant-treated textiles. Environ. Sci.-Process Impacts 15, 1957–1963.

Kumari, K., Sharma, J.K., Kanade, G.S., Kashyap, S.M., Juwarkar, A.A., Wate, S.R., 2014. Investigation of polybrominated diphenyl ethers in old consumer products in India. Environ. Monit. Assess 186, 3001–3009.

Law, R.J., Alaee, M., Allchin, C.R., Boon, J.P., Lebeuf, M., Lepom, P., Stern, G.A., 2003. Levels and trends of polybrominated diphenylethers and other brominated flame retardants in wildlife. Environment International (The State-of-Science and Trends of BFRs in the Environment) 29, 757–770.

Lyche, J.L., Rosseland, C., Berge, G., Polder, A., 2015. Human health risk associated with brominated flame-retardants (BFRs). Environ. Int 74, 170–180.

Ma, Y.N., Venier, M., Hites, R.A., 2012. Tribromophenoxy flame retardants in the great lakes atmosphere. Environ. Sci. Technol 46, 13112–13117.

Marvin, C.H., Tomy, G.T., Armitage, J.M., Arnot, J.A., McCarty, L., Covaci, A., Palace, V., 2011. Hexabromocyclododecane: current understanding of chemistry, environmental fate and toxicology and implications for global management. Environ. Sci. Technol 45, 8613–8623.

Melymuk, L., Bohlin-Nizzetto, P., Kukučka, P., Vojta, Š., Kalina, J., Čupr, P., Klánová, J., 2016. Seasonality and indoor/outdoor relationships of flame retardants and PCBs in residential air. Environmental Pollution 218, 392–401.

Okonski K., Melymuk L., Kohoutek J. and Klánová J., Hexabromocyclododecane: concentrations and isomer profiles from sources to environmental sinks, Submitted

Papachlimitzou, A., Barber, J.L., Losada, S., Bersuder, P., Law, R.J., 2012. A review of the analysis of novel brominated flame retardants. Journal of Chromatography A 1219, 15–28.

Peaslee, G., Savagian, L., Maley, A., Greve, E., O'Connell, H., Gearhart, J., Pena, K., Roberts, B., 2014. A Study of Halogenated Flame Retardants in Automotive Seats. BFR2014.

Posner, S., Roos, S., Olsson, E., 2011. Exploration of Management Options for HBCD. Swerea IVF Project Report 10/11.

Rani, M., Shim, W.J., Han, G.M., Jang, M., Song, Y.K., Hong, S.H., 2014. Hexabromocyclododecane in polystyrene based consumer products: an evidence of unregulated use. Chemosphere 110, 111–119.

Rauert, C., Harrad, S., Stranger, M., Lazarov, B., 2015. Test chamber investigation of the volatilization from source materials of brominated flame retardants and their subsequent deposition to indoor dust. Indoor Air 25, 393–404.

Rauert, C., Harrad, S., Suzuki, G., Takigami, H., Uchida, N., Takata, K., 2014a. Test chamber and forensic microscopy investigation of the transfer of brominated flame retardants into indoor dust via abrasion of source materials. Science of the Total Environment 493, 639–648.

Rauert, C., Lazarov, B., Harrad, S., Covaci, A., Stranger, M., 2014b. A review of chamber experiments for determining specific emission rates and investigating migration pathways of flame retardants. Atmospheric Environment 82, 44–55.

Saini, A., Rauert, C., Simpson, M.J., Harrad, S., Diamond, M.L., 2016. Characterizing the sorption of polybrominated diphenyl ethers (PBDEs) to cotton and polyester fabrics under controlled conditions. Science of the Total Environment 563, 99–107.

Shaw, S.D.H., Jennifer, H., Berger, Michelle L., Subedi, B., Kannan, K., 2014. Brominated flame retardants and their replacements in food packaging and household products: uses, human exposure, and health effects. In: Snedeker, S.M. (Ed.), Toxicants in Food Packaging and Household Plastics. Springer-Verlag, London.

Shoeib, M., Harner, T., Webster, G.M., Sverko, E., Cheng, Y., 2012. Legacy and current-use flame retardants in house dust from Vancouver, Canada. Environmental Pollution 169, 175–182.

Schweizer, C., Edwards, R.D., Bayer-Oglesby, L., Gauderman, W.J., Ilacqua, V., Jantunen, M.J., Lai, H.K., Nieuwenhuijsen, M., Kunzli, N., 2007. Indoor time-microenvironment-activity patterns in seven regions of Europe. Journal of Exposure Science and Environmental Epidemiology 17, 170–181.

Sjödin, A., Päpke, O., McGahee, E., Focant, J.-F., Jones, R.S., Pless-Mulloli, T., Toms, L.-M.L., Herrmann, T., Müller, J., Needham, L.L., Patterson Jr., D.G., 2008. Concentration of polybrominated diphenyl ethers (PBDEs) in household dust from various countries. Chemosphere – halogenated persistent organic pollutants dioxin 2005-Selected papers from the 25th international symposium on halogenated environmental organic pollutants and POPs held in toronto. Canada 73, S131–S136. August 2005.

Stapleton, H.M., Klosterhaus, S., Keller, A., Ferguson, P.L., van Bergen, S., Cooper, E., Webster, T.F., Blum, A., 2011. Identification of flame retardants in polyurethane foam collected from baby products. Environ. Sci. Technol 45, 5323–5331.

Stapleton, H.M., Sharma, S., Getzinger, G., Ferguson, P.L., Gabriel, M., Webster, T.F., Blum, A., 2012. Novel and high volume use flame retardants in US couches reflective of the 2005 PentaBDE phase out. Environ. Sci. Technol 46, 13432–13439.

Stubbings, W.A., Harrad, S., 2014. Extent and mechanisms of brominated flame retardant emissions from waste soft furnishings and fabrics: a critical review. Environ. Int 71, 164–175.

Sverko, E., Tomy, G.T., Reiner, E.J., Li, Y.F., McCarry, B.E., Arnot, J.A., Law, R.J., Hites, R.A., 2011. Dechlorane plus and related compounds in the environment: a review. Environ. Sci. Technol 45, 5088–5098.

Takeuchi, S., Kojima, H., Saito, I., Jin, K., Kobayashi, S., Tanaka-Kagawa, T., Jinno, H., 2014. Detection of 34 plasticizers and 25 flame retardants in indoor air from houses in Sapporo, Japan. Science of the Total Environment 491, 28–33.

Thuresson, K., Bjorklund, J.A., de Wit, C.A., 2012. Tri-decabrominated diphenyl ethers and hexabromocyclododecane in indoor air and dust from Stockholm microenvironments 1: levels and profiles. Science of the Total Environment 414, 713–721.

UNEP, 2009. Stockholm Convention on Persistent Organic Pollutants (POPs). Amendments to Annexes A, B and C. Secretariat of the Stockholm Convention.

UNEP, 2013. Stockholm Convention on Persistent Organic Pollutants (POPs). Amendments to Annex A. Secretariat of the Stockholm Convention.

van der Ven, L.T.M., van de Kuil, T., Leonards, P.E.G., Slob, W., Lilienthal, H., Litens, S., Herlin, M., Hakansson, H., Canton, R.F., van den Berg, M., Visser, T.J., van Loveren, H., Vos, J.G., Piersma, A.H., 2009. Endocrine effects of hexabromocyclododecane (HBCD) in a one-generation reproduction study in Wistar rats. Toxicology Letters 185, 51–62.

Venier, M., Audy, O., Vojta, Š., Bečanová, J., Romanak, K., Melymuk, L., Krátká, M., Kukučka, P., Okeme, J., Saini, A., Diamond, M.L., Klánová, J., 2016. Brominated flame retardants in the indoor environment — comparative study of indoor contamination from three countries. Environ. Int 94, 150–160.

Venier, M., Wierda, M., Bowerman, W.W., Hites, R.A., 2010. Flame retardants and organochlorine pollutants in bald eagle plasma from the Great Lakes region. Chemosphere 80, 1234–1240.

Wager, P.A., Schluep, M., Muller, E., Gloor, R., 2012. RoHS regulated substances in mixed plastics from waste electrical and electronic equipment. Environ. Sci. Technol 46, 628–635.

Weil, E.A., Levchik, S.V., 2008. Flame retardants in commercial use or development for polyolefins. Journal of Fire Sciences 26, 5–43.

Whitehead, T.P., Smith, S.C., Park, J.-S., Petreas, M.X., Rappaport, S.M., Metayer, C., 2015. Concentrations of persistent organic pollutants in California women's serum and residential dust. Environmental Research 136, 57–66.

WHO, 1997. Flame Retardants: a General Introduction. 192. Environmental Health Criteria.

Zhang, X.M., Diamond, M.L., Ibarra, C., Harrad, S., 2009. Multimedia modeling of polybrominated diphenyl ether emissions and fate indoors. Environ. Sci. Technol 43, 2845–2850.

APPENDIX 11

Vojta, Šimon, Lisa Melymuk, and Jana Klánová. 2017. "Changes in Flame Retardant and Legacy Contaminant Concentrations in Indoor Air during Building Construction, Furnishing, and Use." *Environmental Science & Technology* 51 (20): 11891-99. https://doi.org/10.1021/acs.est.7b03245

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Changes in Flame Retardant and Legacy Contaminant Concentrations in Indoor Air during Building Construction, Furnishing, and Use

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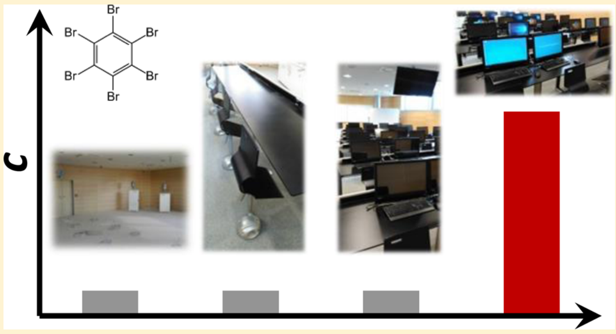
Supporting Information

ABSTRACT: A newly constructed university building was selected for targeted assessment of changes in the levels of flame retardants and legacy contaminants during the installation of building equipment, furniture, electronics, and first year of building use. Indoor air samples were collected during several periods of intensive equipment installation to determine a relationship between newly introduced equipment and changes in the concentrations and profiles of contaminants in indoor air. Samples were analyzed for polybrominated diphenyl ethers (PBDEs), hexabromocyclododecanes (HBCDDs), and new types of flame retardants: brominated (BFRs) and organophosphate esters (OPEs). Additionally, typical outdoor contaminants such as polychlorinated biphenyls (PCBs) and organochlorine pesticides (OCPs) were also analyzed for comparison. From the set of 90 compounds analyzed here, hexabromobenzene (HBB) and tris(2-chloroisopropyl)phosphate (TCIPP) showed a significant concentration increase in indoor air concentrations during computer installation and operation, suggesting emission by operating computers, while an order of magnitude concentration increase in tris(1,3-dichloro-2-propyl)phosphate (TDCIPP) and tri-m-cresyl phosphate (TMTP) was observed after the furniture and carpet was introduced to the computer room, suggesting furniture or carpet as a source. However, the majority of compounds had no systematic change in concentrations during equipment installation, indicating that no sources of target compounds were introduced or, that source introduction was not reflected in indoor air concentrations. Generally, low levels of legacy flame retardants compared to their novel alternatives were observed.



INTRODUCTION

Flame retardants (FRs) are synthetic compounds added to a wide variety of building materials and consumer products[1] in order to reduce flammability, prevent combustion, and delay the spread of fire after ignition. They are also frequently required by many national and international fire safety codes, standards and regulations. Most FRs are typically added to a material after polymer manufacture, which considerably lowers their affinity to the final product and allows them to be released during the entire lifecycle of the product. Therefore, they have become widespread,[2−13] especially in indoor microenvironments such as homes, offices, cars, or schools.[14]

Brominated flame retardants (BFRs) are historically the most widely used group of organic FRs due to their high efficiency and low cost. Although there are concerns about the persistence, bioaccumulation and potential health effects of many BFRs,[15−17] only polybrominated diphenyl ethers (PBDEs) and hexabromocyclododecane (HBCDD) are currently legislatively restricted. Penta and octaBDE technical mixtures and HBCDD are currently banned in North America and the European Union, and they have been listed by the Stockholm Convention on Persistent Organic Pollutants.[18,19] DecaBDE, which consists mainly of decabromodiphenyl ether (BDE-209), was phased out of use in the EU in 2008[20] and North America in 2013,[21] and it was added to the Stockholm Convention in 2017.[22]

To replace restricted FRs while continuing to fulfill existing international fire safety standards, worldwide production has turned to alternative flame retarding compounds, namely novel (alternative) halogenated flame retardants (NFRs)[15,23] and chemicals based on organophosphate esters (OPEs).[24] For example, the pentaBDE formulation has been replaced by Firemaster commercial mixtures, containing 2-ethylhexyl-2,3,4,5-tetrabromobenzoate (EH-TBB), bis(2-ethylhexyl)-tetrabromophthalate (BEH-TEBP) and triphenyl phosphate (TPHP), while octaBDE has been replaced by, for example, 1,2-bis(2,4,6-tribromophenoxy)ethane (BTBPE), commercially known as FF680, and decaBDE has been replaced by decabromodiphenyl ethane (DBDPE).[10,11,15,25] Halogenated OPEs are usually used as flame retardants, whereas the nonhalogenated OPEs are primarily used as plasticizers. In

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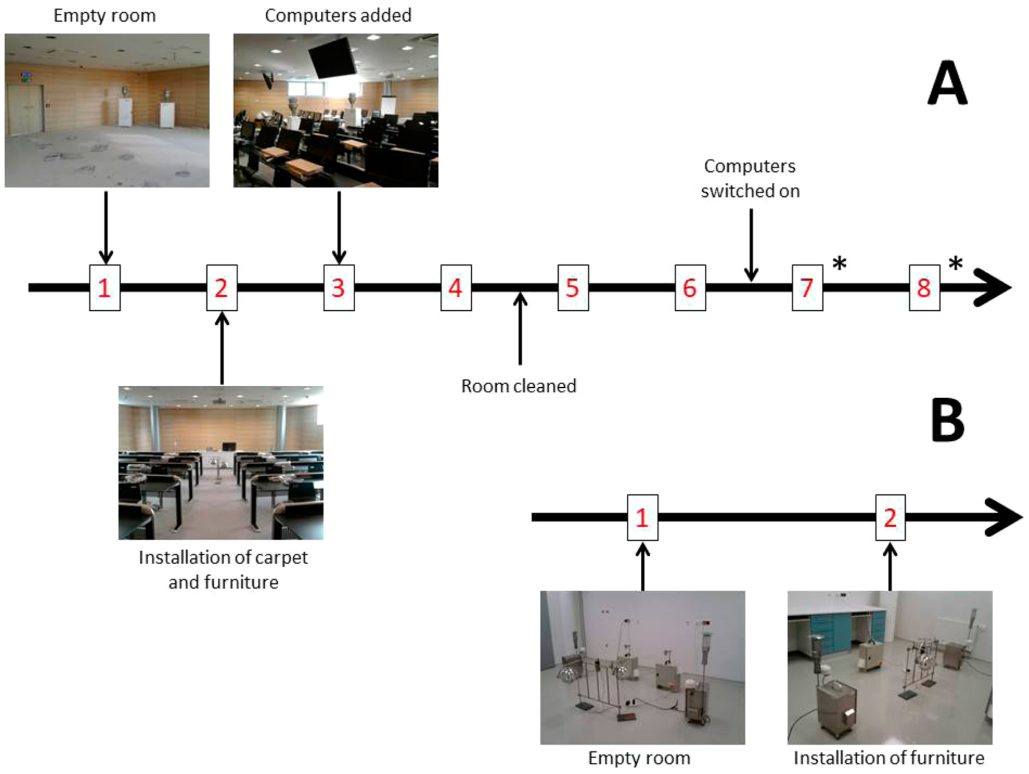

Figure 1. Sampling scheme for computer room (A) and laboratory (B). The asterisk indicates the events when the lecture room was also sampled. The sample timing for the laboratory also applies for the office.

both cases, OPEs are being added to a wide range of various materials resulting in elevated concentrations detected in the environment.[24,26] Although they are considered as less persistent than above-mentioned halogenated FRs, they can still undergo long-range atmospheric transport.[27,28] Moreover, some OPEs were identified as carcinogenic or neurotoxic[29] and due to their higher hydrophilicity, they are reported to be more accessible via inhalation exposure.[30] Although these alternatives have been chosen as substitutes for banned chemicals, many have very similar structures and physicochemical properties and raise similar environmental and health concerns.[15,28,29,31−33]

Since FRs are found in many consumer products, their indoor concentrations are often higher than outdoors,[3,34] leading to indoor exposure to FRs via inhalation of indoor air and ingestion of indoor dust;[35,36] furthermore, the indoor environment can act as a source of FRs to the outdoor environment.[37]

Indoor air levels of FRs and other semivolatile organic compounds (SVOCs) are typically measured in in-use buildings.[14,38,39] In these studies, indoor levels are often attributed to building materials or indoor equipment, however without measurements of emissions from the materials themselves, these links between single time-point measurements and sources are often circumstantial. Although elevated concentrations of FRs are generally considered to be connected to the presence of electronics or furniture,[35,40−42] correlations between indoor levels and potential sources are difficult to identify[35,43] and links have only been made in a few studies with repeated indoor measurements during changes in room equipment or furnishings.[44,45] Stronger links have been made between FR sources and indoor concentrations, for example, presence of a television and dust concentrations of HBCDD[46] and presence of a computer and levels of PBDEs in air,[47,48] however these are limited to legacy FRs. Recently, Dodson et al.[49] identified differences in FR profiles in response to different flammability standards, suggesting both regulations and equipment affect the profile of FRs indoors.

Useful information concerning source identification can be obtained from chamber experiments, which represent a powerful tool for determining source emission characteristics, but they are difficult to apply to real indoor conditions.[35] Evaluating levels of FRs and other semivolatile organic compounds (SVOCs) during building construction and furnishing provides a unique opportunity to identify compound sources as either building materials, furnishings, consumer products, or some other aspect of room use. In this study, indoor air samples were collected during several periods of extensive equipment installation in a newly constructed empty building in order to identify potential sources of FRs and other SVOCs.

We hypothesized that initial concentrations of FRs in the building reflect FRs associated with the building materials themselves, whereas any increases in FRs during installation of building equipment and furnishing indicate emissions from the newly added products. In contrast, SVOCs with no expected indoor sources, that is, polychlorinated biphenyls (PCBs), organochlorine pesticides (OCPs), and polycyclic aromatic hydrocarbons (PAHs) should show no changes in indoor air concentrations in response to equipping of the building. The strategy of sampling during the stages of construction has previously been applied to measurements of volatile organic compounds,[50−52] but not yet to SVOCs.

■ MATERIALS AND METHODS

Sampling Design. Indoor air was measured in four rooms of a newly constructed university building (in Brno, Czech

Republic) during equipment and furniture installation. The rooms were chosen according to their properties and functions to represent various indoor environments and identify potential differences in air composition: a computer room (3rd floor, 400 m^3), lecture room (2nd floor, 400 m^3), office (4th floor, 62 m^3), and laboratory (1st basement, 65 m^3). Indoor air in the two larger rooms was collected using high-volume active air samplers (HVAAS) while the smaller office and laboratory were sampled using low-volume air samplers (LVAAS). The HVAAS was a Digitel DH-77 sampler (Digitel Elektronik, Austria) with a flow rate of 30 m^3/h equipped with one glass fiber filter (47 mm GFF, Whatman) and two polyurethane foam (PUF) plugs (110 mm diameter, 50 mm length, 0.030 g cm^{-3} density, T-3037 Molitan a.s., Czech Republic) to collect both particulate matter and the gas phase. The LVAAS was a LVS3 (Sven Leckel Ingenieurburo GmbH, Germany) with a flow rate of 2.3 m^3/h, equipped with 1 GFF and 2 PUFs (55 mm diameter, 50 mm length).

In the computer room, sampling was repeated eight times (Figure 1). The first sampling was carried out in the empty room (event one), followed by subsequent sample collection as carpet and furniture were added (event two), after computers were installed (events three to six including room cleaning episode at event five) and after computers were switched on (events seven and eight). The room was equipped with 56 all-in-one computers, 56 wood composite chairs, and 12 long desks. The lecture room was considered as a control without electronic devices, as it is an identical room to the computer room. Sample collection occurred after the room was equipped with carpet and furniture (80 chairs, 60 desks). In the office and laboratory, air samples were collected twice: before and after furniture installation. The indoor temperature over the sampling period was stable, varying by less than ±2 °C. See Supporting Information (SI) Table S1 for further sampling details.

All sampling media were precleaned by Soxhlet extraction in acetone and dichloromethane, 8 h each. After sampling, exposed media were wrapped in aluminum foil and sealed in plastic bags for transport to the laboratory, where they were stored at −18 °C until analysis. Detailed methods for PUF/GFF preparation and deployment have been described by Bohlin et al.[53,54]

Analysis. The target analytes were 10 PBDEs (congeners 28, 47, 66, 85, 99, 100, 153, 154, 183, and 209), 3 HBCDD isomers, 23 NFRs (TBP-AE, TBX, TBP-BAE, PBT, PBBZ, PBEB, TBCT, DDC−CO-MA, PBB-Acr, TBP-DBPE, HBB, BTBPE, EH-TBB, TDBP-TAZTO, s-DDC−CO, a-DDC−CO, *α*-DBE-DBCH, *β*-DBE-DBCH, *α*-TBCO, *β*-TBCO, BEH-TEBP, DBHCTD, DBDPE), 15 non-chlorinated OPEs (non-Cl-OPEs: TPrP, TiBP, TnBP, DBPP, TPeP, BDPP, TBOEP, TPHP, EHDPP, TEHP, TOTP, TMTP, TPTP, TIPPP, TDMPP), 3 chlorinated OPEs (Cl-OPEs: TCEP, TCIPP, TDCIPP), 9 PCBs (congeners 9, 11, 28, 52, 101, 118, 138, 153, 180), 12 OCPs (PeCB, HCB, *α*-HCH, *β*-HCH, *γ*-HCH, *δ*-HCH, *o,p*′-DDE, *p,p*′-DDE, *o,p*′-DDD, *p,p*′-DDD, *o,p*′-DDT, *p,p*′-DDT), and 15 PAHs (ACY, ACE, FLU, PHE, ANT, FLA, PYR, BAA, CHR, BBF, BKF, BAP, IDP, DBA, BGP). See SI Table S2 for full compound names and further details; abbreviations of FRs are adopted from Bergman et al.[55] ^{13}C-labeled or deuterated internal standards were added before extraction. All the analytical standards were purchased from Wellington Laboratories Inc. (Canada). Pesticide residue grade solvents were purchased from Lab-scan (Poland) and Silica Gel 60 (70−230 mesh) from Merck (Germany).

The exposed PUFs and GFFs were extracted with dichloromethane using an automated warm Soxhlet extraction system and the extracts were purified using column chromatography.

PBDEs and NFRs were analyzed using gas chromatography coupled to high resolution mass spectrometry, whereas PCBs, OCPs, and OPEs were analyzed by gas chromatography-tandem mass spectrometry systems. HBCDDs were analyzed after exchanging solvent to acetonitrile by liquid chromatography electrospray ionization mass spectrometry. All the analytical procedures used here were published previously[39,56−59] and are described in detail in the SI.

QA/QC. The recoveries of individual compounds were determined using a set of pre-cleaned (8 h in acetone and 8 h in dichloromethane) PUFs ($n = 8$) spiked with the native analytes prior to extraction. The recoveries of individual compounds are given in SI Table S2.

To calculate method detection limits (MDLs) of individual compounds for each matrix, eight laboratory procedural blanks were processed using precleaned sampling media. An average value plus three times the standard deviation of the blanks was used as MDL, and the average of the procedural blanks was subtracted from the samples above MDL. When a compound was not detected in the blanks, instrumental detection limits were used (SI Table S2).

For statistical data analysis, values below detection were substituted by $\sqrt{2}/2{\cdot}\mathrm{MDL}$.[60] The Kruskal−Wallis test and multiple comparisons of mean ranks were used to investigate statistical differences. All statistical analyses were performed with STATISTICA (version 13, StatSoft, Inc.) or Microsoft Excel.

RESULTS AND DISCUSSION

Overview of Indoor Air Concentrations. Order of magnitude differences were observed between the different compound groups, reflecting differences in use and sources (Figure 2; summary statistics for individual compounds are given in SI Tables S4−S7). The lowest concentrations were found for PBDEs; in all four rooms, the $\sum_{10}$PBDEs in bulk (gas +particle phase) air ranged from 0.803 pg m^{-3} to 9.83 pg m^{-3} with a median of 2.28 pg m^{-3}, and the dominant congeners were BDE 47 and 99. These indoor air levels were very low compared to North America ($\sum$PBDEs = 1260 pg m^{-3} in a U.S. institutional building,[61] 148 pg m^{-3} in U.S. apartments, 60 pg m^{-3} in Canadian apartments[39]) and other European homes[62] ($\sum$PBDEs = 223 pg m^{-3}), which was expected considering that the building was newly constructed and equipped after the restrictions on PBDEs. On the other hand, similar or even lower amounts of PBDEs have been reported in family houses and apartments in France[63] and Czech Republic.[39]

A similar result was observed for HBCDDs, where consistently low concentrations were found. The $\sum_4$HBCDDs was dominated by *α*-HBCDD and bulk air concentrations ranged from 0.0988 pg m^{-3} to 13.6 pg m^{-3} with a median of 5.63 pg m^{-3}. These relatively low levels are in agreement with the building documentation, specifying that no polystyrene was used for building insulation.

Twelve of the 23 target NFRs were consistently detected in indoor matrices: DBE-DBCH, TBP-BAE, PBBZ, PBT, PBEB, TBP-DBPE, HBB, PBB-Acr, EH-TBB, BEH-TEBP, BTBPE, and DDC−CO. In contrast to the restricted PBDEs and

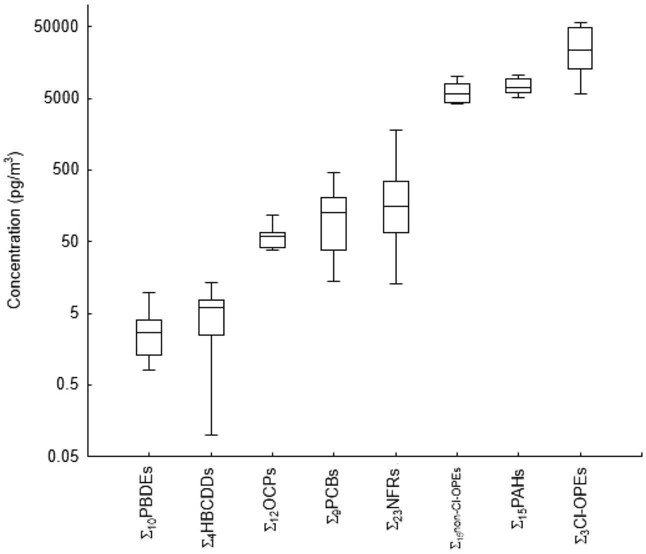

Figure 2. Indoor air concentrations (gas + particle phase) of PBDEs, HBCDDs, OCPs, PCBs, NFRs, OPEs, and PAHs. The central lines indicate the median, the boxes show the interquartile range (25th/75th percentiles), and the whiskers indicate the maximum and minimum concentrations.

HBCDD, elevated levels of many replacement NFRs were observed, in particular for five compounds: HBB, PBBZ, PBT, TBP-DBPE, and DBE-DBCH. The concentrations of $\sum_{23}$NFRs ranged from 12.3 pg m^{-3} to 1820 pg m^{-3}, with a median of 158 pg m^{-3}. The highest median concentration of an individual NFR was for HBB, at 97.0 pg m^{-3} (range 1.29 to 1400 pg m^{-3}). Concentrations of NFRs in this study are in general agreement with those published previously,[25,39,62,64] with the exception of two compounds. The range of concentrations and median of HBB is larger than those reported by Cequier et al.[62] (median: 4.11 pg m^{-3} in Norway), Newton et al.[25] (median: 3.6 pg m^{-3} in Sweden), Tao et al.[64] (median 53 pg m^{-3} in UK) and Venier et al.[39] (medians of 4.0, 5.8, and 4.6 pg m^{-3} in the U.S., Canada, and Czech Republic, respectively). The median concentration of DBE-DBCH of 9.28 pg m^{-3} from our study is lower than that found by Cequier et al.[62] (median: 77.9 pg m^{-3}), Newton et al.[25] (median: 55 pg m^{-3}) and Tao et al.[64] (median 280 pg m^{-3}).

The indoor concentrations of banned PCBs and OCPs were of a similar order of magnitude to the in-use NFRs. The median concentration of $\sum_9$PCBs was 147 pg m^{-3} (range 14.1 pg m^{-3} to 452 pg m^{-3}), dominated by congeners 11 and 28; while the median concentration of $\sum_{12}$OCPs was 59.8 pg m^{-3} (range 38.4 pg m^{-3} to 118 pg m^{-3}), and this was dominated by HCB, α-HCH, γ-HCH, and p,p'-DDE.

Two compound classes were orders of magnitude higher: the OPEs and PAHs (Figure 2). Twelve out of the 18 OPEs were consistently detected: TIBP, TNBP, TCEP, TCIPP, DBPP, TDCIPP, TBOEP, TPHP, EHDPP, TEHP, TMTP, and TIPPP. The $\sum_{18}$OPEs ranged from 12 900 pg m^{-3} to 63 300 pg m^{-3} with a median of 30 700 pg m^{-3}, with the major contribution from TCIPP (75% on average). Similar levels of TCIPP, the highest OPE contributor in our study (median 22 500 pg m^{-3}) were reported in Cequier et al.[62] (median 42 300 pg m^{-3}), but Cequier et al. observed 20 times higher concentrations of TBOEP in school classrooms (median 12 900 pg m^{-3}) than in residential living rooms (median 598 pg m^{-3}), attributed to floor polish. Our findings, with a median of 1230 pg m^{-3} for TBOEP, suggest no floor polish during the sampling period, as expected considering that the rooms are carpeted. The amounts of TCEP found in our study were 1−2 orders of magnitude lower compared to available literature data,[62,65,66] corresponding with its gradual substitution with TCIPP prior to the building construction.[24,27,30]

The median of $\sum_{15}$PAHs was 7200 pg m^{-3} (range 5200 pg m^{-3} to 10 420 pg m^{-3}) with the highest contribution from phenanthrene. These order of magnitude higher concentrations for OPEs and PAHs are typical for these compound classes,[62,63,67] and emphasize that the strongest control on overall indoor levels is the magnitude of use and/or emission. Further, a comparison with outdoor levels from a concurrent study at the same location[54] provides an additional indication of compound sources. Concentrations of PBDEs, OCPs, and PCBs were similar in indoor and outdoor air, excepting for PCBs 28 and 52, which were 10× higher indoors, whereas concentrations of NFRs were consistently higher indoors, and concentrations of PAHs were 10× higher outdoors. This suggests the outdoor air as a source to indoors for PAHs, while indoor environments are a source for in-use NFRs, in keeping with the hypothesis of the study.

The majority of the target compounds were found predominantly (>75%) in the gaseous phase (i.e., sorbed to the PUF in the active air samplers). Only HBCDDs and all OPEs had significant particle phase contributions (see SI

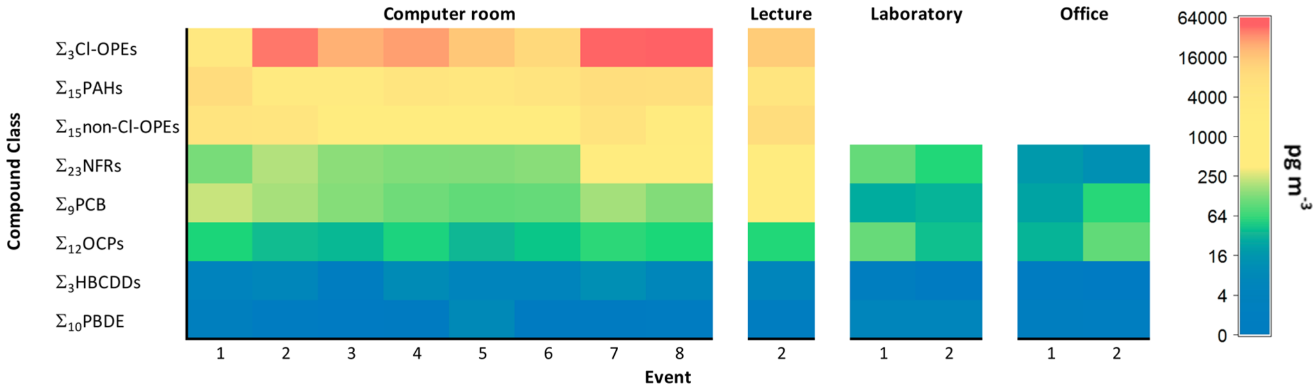

Figure 3. Concentration changes of sums of target compounds during the equipment installation. The event numbers correspond to the sampling scheme given in Figure 1. PAH and OPE data for laboratory and office are not available.

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Tables S4–S7). There were no significant spatial or temporal differences in the gas-particle partitioning.

Temporal Changes in Concentrations. In keeping with the hypothesis of outdoor sources of the legacy compounds and PAHs, and indoor sources of the emerging FRs, we expected temporal changes over the course of equipment installation for NFRs and OPEs, and no such changes for PCBs, OCPs, PAHs, and PBDEs. The measurements in the computer room at different time-points support this hypothesis. No significant change in concentrations of PBDEs was observed during equipment installation except for event five in the computer room where increased activity in the room (due to room cleaning) is reflected in slightly higher concentrations. We assume that no direct sources of PBDEs to the air were present in building materials or introduced by the furniture or equipment. Similarly, no temporal trend was observed for HBCDDs, PAHs, PCBs, or OCPs (Figure 3).

Six NFRs had notable temporal trends, and these are shown in more detail in Figure 4. An order of magnitude increase in the concentration of HBB in bulk air was observed at events seven and eight, that is, after computers were switched on. The same pattern was observed also for PBBZ and TBP-DBPE, although at lower concentrations. A similar pattern was also identified for TBP-DBPE-related tribromophenoxy compounds: TBP-AE and TBP-BAE, and for PBB-Acr, another bromobenzene compound, at much lower concentrations (median about 0.1 pg m^{-3}). Concurrently, another concentration increase was observed between event one and two for PBT and DBE-DBCH (Figure 4). No significant temporal changes were observed for PBEB, EH-TBB, BEH-TEBP, BTBPE, and DDC–CO. This suggests that three bromobenzene compounds, dominated by HBB, and three tribromophenoxy compounds, dominated by TBP-DBPE, were emitted by operating computers. This is supported by concentrations from the laboratory and office, where no electronics were present, and only background concentrations of these compounds were found (Figure 3). Moreover, intermediate concentrations (median of 338 pg m^{-3} for HBB) were found in the lecture room, which only contained one computer and overhead projector (Figure 3).

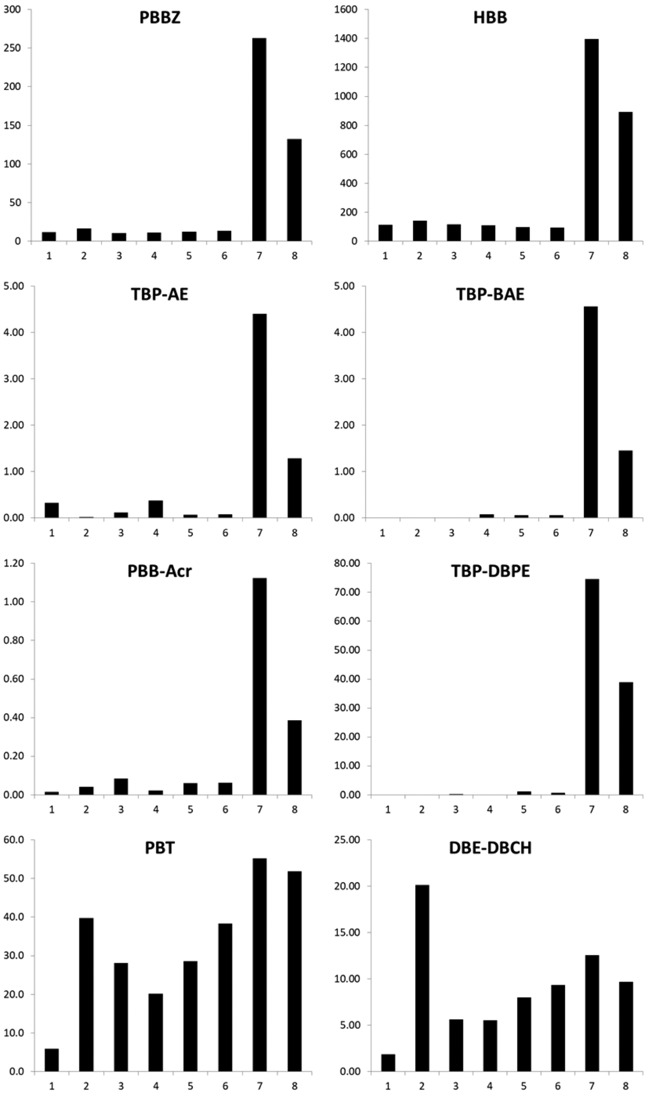

Figure 4. Changes in air concentrations (pg m^{-3}) of NFRs over time in computer room. Sample numbers (x-axis) correspond to the sampling events identified in Figure 1.

Different OPEs had different temporal trends. Similarly to PBT and DBE-DBCH, an order of magnitude concentration increase in four OPEs (TDCIPP, TCIPP, TEHP, and TMTP) was observed after the furniture and carpet was introduced to the computer room (Figure 5), which directly suggests furniture, carpet (or associated products used in carpet installation) as a source of these OPEs and NFRs. TIPPP and DBPP exhibited high concentrations in the initial sample, followed by a gradual decrease over time until Event 7 (computers switched on), where another substantial increase occurred (Figure 5). TCIPP also had a distinct increase in concentrations at sample 7. Thus, a computer-related source is suggested for TIPPP, DBPP, and TCIPP. TCEP and EHDPP also had concentration peaks at Event 7. TBOEP decreased gradually over time. No temporal trends were observed for TiBP, TnBP and TPHP.

Spatial Differences between Rooms. As mentioned above for HBB, the comparison of rooms within the same building should further allow us to distinguish between sources due to building materials, which are common to all rooms, vs sources from room equipment and furnishings, which are unique to individual rooms. No striking difference between the room types was observed in the case of PBDEs (medians of $\sum_{10}$PBDEs from 1.41 pg m^{-3} to 6.61 pg m^{-3}), HBCDDs (medians of $\sum_{3}$HBCDDs from 0.7841 pg m^{-3} to 6.54 pg m^{-3}), OCPs (medians of $\sum_{12}$OCPs from 55.6 pg m^{-3} to 82.6 pg m^{-3}), or PAHs (medians of $\sum_{15}$PAHs from 6931 pg m^{-3} to 7270 pg m^{-3}) (Figure 3). A slightly higher concentration of $\sum_{9}$PCBs was noted in the lecture room, due to high concentrations of PCB 28 and 52, as mentioned above. The median of $\sum_{9}$PCBs in the lecture room was 420 pg m^{-3} compared to 155 pg m^{-3} in the computer room and 35.4 pg m^{-3} the office and laboratory. The higher concentrations of lighter PCBs in the lecture room may be due to emission from nearby excavation and construction, but no source can be concretely identified.

Concentrations of NFRs in the office and laboratory were similar to the concentrations in the computer room before the addition of the computers (Figure 3), supporting the result of computer operation as a source of HBB, TBP-AE, TBP-BAE, PBB-Acr, PBBZ, and TBP-DBPE, rather than any source from building materials or flooring. For example, when electronics were switched on in the computer room in our study,

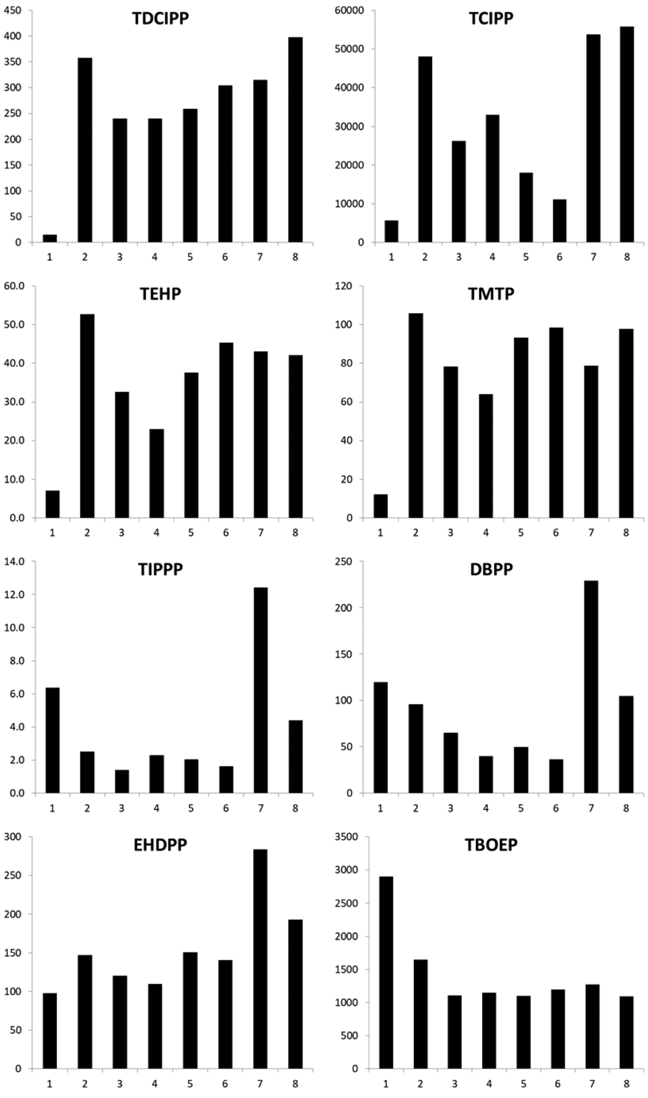

Figure 5. Changes in air concentrations (pg m^{-3}) of OPEs over time in computer room. Sample numbers (x-axis) correspond to the sampling events identified in Figure 1.

concentrations of HBB jumped from hundreds to thousands of pg m^{-3}, while the median concentration of HBB in non-electronics equipped rooms was only 6.71 pg m^{-3}. Similarly, a maximum concentration of 297 pg m^{-3} and median of 4.11 pg m^{-3} from 37 dwellings in Cequier et al.[62] suggest varying numbers of electronics in the study environments. The median concentration of $\sum_{18}$OPEs from the lecture room (26 300 pg m^{-3}) agrees with the median (26 900 pg m^{-3}) of events 1 to 6 (before the computers were turned on) from the computer room, likely reflecting the same technology used in both rooms.

Identification of Potential FRs Sources. Although the levels of flame retardants in indoor environments are often considered to be connected to presence of electronics or upholstered furniture in a particular microenvironment, evidence is usually indirect.[33,35] The identification of potential sources of contamination is often hindered by factors including sampling techniques,[3] source misclassification,[35] or experimental conditions,[43] such as temperature, a key parameter for the emission of FRs from operating electronics.[68] Moreover, the life-cycle stage of the product plays an important role in the emission rates of the compounds, especially in case of more volatile flame retardants.[35,45,68,69] This also applies for another important group of consumer products often considered a source of FRs, that is, building materials.[1,70,71] Materials initially have a higher emission for volatilization emissions of SVOCs,[16,44] and thus sampling in a recently constructed building with the presence of new materials provides a stronger opportunity to identify emissions from building materials and furnishings.

Operating computers were identified as the most notable source of FRs to the indoor air, primarily for HBB and TCIPP. HBB emissions from computers agree with its suggested use in electronic and plastic goods,[15] but possible emission of TCIPP from computers is quite surprising, considering that TCIPP is mainly used as FR in insulation and foams.[24,71] Additionally, temporal trends suggest computers as a source of TBP-DBPE and related tribromophenoxy compounds (TBP-AE and TBP-BAE), PBB-Acr and PBBZ. TBP-DBPE is used as an FR in polypropylene[72] and as an additive FR to acrylonitrile-butadiene-styrene (ABS) plastics.[73] Although TBP-AE is used in expanded polystyrene, considering the low concentration levels detected, together with TBP-BAE, is a possible transformation product or impurity[10,74] of TBP-DBPE.[10,74] PBB-Acr is a reactive FR which little is known about,[75] whereas PBBZ may be a reactive debromination product of HBB,[75] and has been identified in 30% of electronic product wipes in a study in Canada.[76] Thus, it could be that elevated operating temperatures within computers could be resulting in the formation of some brominated products from the additive FRs, and the concurrent volatilization losses of both the parent FRs and products. However, more specific studies (e.g., emission chambers) are necessary to confirm this hypothesis. Finally, elevated concentrations of DBPP, TIPPP, TCEP, and EHDPP may also (in part) be connected to emissions from operation of computers. These compounds are used as FR in PVC, polyester resins and coatings (for TCEP), PVC and thermoplastics (for TIPPP) or as plasticizers (EHDPP).

Moreover, we emphasize that the increase in emissions comes from operation of the computers, not simple presence of the computers; no significant increase in concentrations was observed in the room with the introduction of 56 brand new idle computers (Event 6); rather, the jump in air concentrations occurred when computers were switched on (Event 7). We also note that compound physicochemical properties are expected to have an influence on the patterns and time trends of emissions. The two main chemicals detected in the indoor air, HBB and TCIPP might be emitted from the source primarily in the initiation phase due to their high volatility and other less volatile chemicals might be released as time goes on, gradually increasing their total mass emitted.

Two NFRs (DBE-DBCH and PBT) and four OPEs (TDCIPP, TCIPP, TEHP, and TMTP) are suggested as having sources in either furniture or carpet. The room did not contain any significant foam-containing furniture, which contrasts with typically reported uses of TCIPP in insulation and foam. TDCIPP, TEHP, and TMTP are used in plastic, textile and PUF (TDCIPP), PVC, cellulose, paints and coatings, rubber, and PUF (TEHP), and hydraulic fluids, cellulose, coatings, polystyrene, plastics, and thermoplastics (TMTP);[24] thus, use in carpet textile, carpet backing or furniture coatings may account for their increase in emissions in conjunction with installation of furniture and carpet. Sources of DBE-DBCH and PBT are less clear. DBE-DBCH is an additive FR in polystyrene and polyurethane, electronic cable coating and construction materials, whereas PBT is used in various polymers, latex, textiles, rubber, and ABS.[75]

Overall, operation of computers increased total air concentrations of NFRs by 886%, from 167 pg m^{-3} (average before) to 1480 pg m^{-3} (average after), and installation of flooring and furniture increased total air concentrations of OPEs by 310%, from 12 900 pg m^{-3} (before) to 40 000 pg m^{-3} (after). While the elevated concentrations may not persist in the long term, this suggests that the indoor burden of FRs is strongly linked to the products and materials used indoors, and consequently, FR burdens could be reduced with appropriate selection of products that use methods other than synthetic organic flame retardants to meet flammability standards.

However, even in this targeted study with samples in conjunction with changes in room equipment, definitive identification of sources is difficult, emphasizing the general challenges in correlating between putative sources and FRs concentrations in air.[35,43] Besides the main issue with source classification, the main condition to achieve a good correlation is that the presence of the FR must be clearly reflected in the air concentration. Better correlations with potential sources are frequently reported for indoor dust,[35] as elevated concentrations and higher detection frequencies are usually obtained due to the presence of the entire material particles introduced by the abrasion migration pathway.[77] Analysis of such source material particles provides a good link with a potential source, but also has the disadvantage of spatial variability driven by the distance from the source.[3,67,78,79] Thus, regardless of the correlation with the sources, indoor air and dust sampling provides complementary information to characterize the indoor environment, covering various real-world factors (sinks, mass dynamics), which are difficult to simulate in chamber emission experiments. While chamber experiments are a useful tool for characterizing sources by determining specific emission rates, environmental information is more relevant for human exposure. Moreover, material samples are usually investigated by chamber emission experiments, rather than complex dynamic systems such as operating computers.

A source introduction causing even a marginal increase in air concentrations of chemicals is still relevant from the viewpoint of long-term human exposure. For example, the scenario demonstrated in this study typically applies to open-concept office environments and other computer rooms, where people often spend substantial time. For example, the daily inhalation exposure to HBB for a 70 kg adult, spending 8 h per day in the computer room from our study will increase from 205 pg kg^{-1} day^{-1} to 2094 pg kg^{-1} day^{-1} (see SI for the details) when the computers are switched on.

Moreover, while both brominated and organophosphate FRs were identified as having in-room sources, there are large concentration differences between the two compound groups; the median concentration of TCIPP found in this study was more than 200 times higher than that of HBB, emphasizing the importance of OPEs as a potential concern in indoor environments, as well as the necessity of indoor air monitoring, since inhalation is considered as an important exposure route for chlorinated OPEs.[30] On the other hand, 20 and 50 times lower concentrations of banned HBCDDs and PBDEs, respectively, compared to NFRs, show the decline of legacy FRs in the indoor environments.

■ ASSOCIATED CONTENT

Ⓢ Supporting Information

The Supporting Information is available free of charge on the ACS Publications website at DOI: 10.1021/acs.est.7b03245.

Analytical method details, QA/QC details, summary statistics, exposure assessment calculations (PDF)

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Notes

The authors declare no competing financial interest.

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■ REFERENCES

(1) Vojta, Š.; Bečanová, J.; Melymuk, L.; Komprdová, K.; Kohoutek, J.; Kukučka, P.; Klánová, J. Screening for halogenated flame retardants in European consumer products, building materials and wastes. *Chemosphere* **2017**, *168*, 457−466.

(2) Harrad, S.; Ibarra, C.; Diamond, M.; Melymuk, L.; Robson, M.; Douwes, J.; Roosens, L.; Dirtu, A. C.; Covaci, A. Polybrominated diphenyl ethers in domestic indoor dust from Canada, New Zealand, United Kingdom and United States. *Environ. Int.* **2008**, *34* (2), 232−238.

(3) Harrad, S.; de Wit, C. A.; Abdallah, M. A. E.; Bergh, C.; Bjorklund, J. A.; Covaci, A.; Darnerud, P. O.; de Boer, J.; Diamond, M.; Huber, S.; Leonards, P.; Mandalakis, M.; Oestman, C.; Haug, L. S.; Thomsen, C.; Webster, T. F. Indoor Contamination with Hexabromocyclododecanes, Polybrominated Diphenyl Ethers, and Perfluoroalkyl Compounds: An Important Exposure Pathway for People? *Environ. Sci. Technol.* **2010**, *44* (9), 3221−3231.

(4) Law, R. J.; Barry, J.; Bersuder, P.; Barber, J. L.; Deaville, R.; Reid, R. J.; Jepson, P. D. Levels and Trends of Brominated Diphenyl Ethers in Blubber of Harbor Porpoises (Phocoena phocoena) Stranded or Bycaught in the UK: An Update to 2006. *Environ. Sci. Technol.* **2010**, *44* (12), 4447−4451.

(5) Law, R. J.; Allchin, C. R.; de Boer, J.; Covaci, A.; Herzke, D.; Lepom, P.; Morris, S.; Tronczynski, J.; de Wit, C. A. Levels and trends of brominated flame retardants in the European environment. *Chemosphere* **2006**, *64* (2), 187−208.

(6) Law, R. J.; Covaci, A.; Harrad, S.; Herzke, D.; Abdallah, M. A. E.; Femie, K.; Toms, L.-M. L.; Takigami, H. Levels and trends of PBDEs and HBCDs in the global environment: Status at the end of 2012. *Environ. Int.* **2014**, *65*, 147−158.

(7) Muir, D. C. G.; de Wit, C. A. Trends of legacy and new persistent organic pollutants in the circumpolar arctic: Overview, conclusions, and recommendations. *Sci. Total Environ.* **2010**, *408* (15), 3044−3051.

(8) Salamova, A.; Hermanson, M. H.; Hites, R. A. Organophosphate and Halogenated Flame Retardants in Atmospheric Particles from a European Arctic Site. *Environ. Sci. Technol.* **2014**, *48* (11), 6133−6140.

(9) Venier, M.; Ma, Y. N.; Hites, R. A. Bromobenzene Flame Retardants in the Great Lakes Atmosphere. *Environ. Sci. Technol.* **2012**, *46* (16), 8653−8660.

(10) Ma, Y. N.; Venier, M.; Hites, R. A. Tribromophenoxy Flame Retardants in the Great Lakes Atmosphere. *Environ. Sci. Technol.* **2012**, *46* (24), 13112−13117.

(11) Ma, Y. N.; Venier, M.; Hites, R. A. 2-Ethylhexyl Tetrabromobenzoate and Bis(2-ethylhexyl) Tetrabromophthalate Flame Retardants in the Great Lakes Atmosphere. *Environ. Sci. Technol.* **2012**, *46* (1), 204−208.

(12) Shoeib, M.; Ahrens, L.; Jantunen, L.; Harner, T. Concentrations in air of organobromine, organochlorine and organophosphate flame retardants in Toronto, Canada. *Atmos. Environ.* **2014**, *99*, 140−147.

(13) Möller, A.; Xie, Z. Y.; Caba, A.; Sturm, R.; Ebinghaus, R. Occurrence and air-seawater exchange of brominated flame retardants and Dechlorane Plus in the North Sea. *Atmos. Environ.* **2012**, *46*, 346−353.

(14) Harrad, S.; Goosey, E.; Desborough, J.; Abdallah, M. A. E.; Roosens, L.; Covaci, A. Dust from UK Primary School Classrooms and Daycare Centers: The Significance of Dust As a Pathway of Exposure of Young UK Children to Brominated Flame Retardants and Polychlorinated Biphenyls. *Environ. Sci. Technol.* **2010**, *44* (11), 4198−4202.

(15) Covaci, A.; Harrad, S.; Abdallah, M. A. E.; Ali, N.; Law, R. J.; Herzke, D.; de Wit, C. A. Novel brominated flame retardants: A review of their analysis, environmental fate and behaviour. *Environ. Int.* **2011**, *37* (2), 532−556.

(16) Stubbings, W. A.; Harrad, S. Extent and mechanisms of brominated flame retardant emissions from waste soft furnishings and fabrics: A critical review. *Environ. Int.* **2014**, *71*, 164−175.

(17) van der Ven, L. T. M.; van de Kuil, T.; Leonards, P. E. G.; Slob, W.; Lilienthal, H.; Litens, S.; Herlin, M.; Hakansson, H.; Canton, R. F.; van den Berg, M.; Visser, T. J.; van Loveren, H.; Vos, J. G.; Piersma, A. H. Endocrine effects of hexabromocyclododecane (HBCD) in a one-generation reproduction study in Wistar rats. *Toxicol. Lett.* **2009**, *185* (1), 51−62.

(18) UNEP. *Stockholm Convention on Persistent Organic Pollutants (POPs). Amendments to Annexes A, B and C*, 2009.

(19) UNEP. *Stockholm Convention on Persistent Organic Pollutants (POPs). Amendments to Annex A. Secretariat of the Stockholm Convention*, 2013.

(20) EU. Directive 2002/95/EC and Commission Decision 2005/717/EC; 2008. In European Court of Justice: Cases C-14/06 and C-295/06, Judgement of the Court, 2008.

(21) USEPA. *DecaBDE phase-out initiative*, 2010.

(22) UNEP. *Draft Decision SC-8/[ ]: Decabromodiphenyl ether (commercial mixture, c-decaBDE)*, 2017.

(23) Sverko, E.; Tomy, G. T.; Reiner, E. J.; Li, Y. F.; McCarry, B. E.; Arnot, J. A.; Law, R. J.; Hites, R. A. Dechlorane Plus and Related Compounds in the Environment: A Review. *Environ. Sci. Technol.* **2011**, *45* (12), 5088−5098.

(24) van der Veen, I.; de Boer, J. Phosphorus flame retardants: Properties, production, environmental occurrence, toxicity and analysis. *Chemosphere* **2012**, *88* (10), 1119−1153.

(25) Newton, S.; Sellström, U.; de Wit, C. A. Emerging Flame Retardants, PBDEs, and HBCDDs in Indoor and Outdoor Media in Stockholm, Sweden. *Environ. Sci. Technol.* **2015**, *49* (5), 2912−2920.

(26) Schreder, E. D.; La Guardia, M. J. Flame Retardant Transfers from U.S. Households (Dust and Laundry Wastewater) to the Aquatic Environment. *Environ. Sci. Technol.* **2014**, *48* (19), 11575−11583.

(27) Sühring, R.; Diamond, M. L.; Scheringer, M.; Wong, F.; Pućko, M.; Stern, G.; Burt, A.; Hung, H.; Fellin, P.; Li, H.; Jantunen, L. M. Organophosphate Esters in Canadian Arctic Air: Occurrence, Levels and Trends. *Environ. Sci. Technol.* **2016**, *50* (14), 7409−7415.

(28) Zhang, X. M.; Suhring, R.; Serodio, D.; Bonnell, M.; Sundin, N.; Diamond, M. L. Novel flame retardants: Estimating the physical-chemical properties and environmental fate of 94 halogenated and organophosphate PBDE replacements. *Chemosphere* **2016**, *144*, 2401−2407.

(29) Wei, G. L.; Li, D. Q.; Zhuo, M. N.; Liao, Y. S.; Xie, Z. Y.; Guo, T. L.; Li, J. J.; Zhang, S. Y.; Liang, Z. Q. Organophosphorus flame retardants and plasticizers: Sources, occurrence, toxicity and human exposure. *Environ. Pollut.* **2015**, *196*, 29−46.

(30) Schreder, E. D.; Uding, N.; La Guardia, M. J. Inhalation a significant exposure route for chlorinated organophosphate flame retardants. *Chemosphere* **2016**, *150*, 499−504.

(31) Darnerud, P. O. Brominated flame retardants as possible endocrine disrupters. *Int. J. Androl.* **2008**, *31* (2), 152−160.

(32) Kuramochi, H.; Takigami, H.; Scheringer, M.; Sakai, S. I. Estimation of physicochemical properties of 52 non-PBDE brominated flame retardants and evaluation of their overall persistence and long-range transport potential. *Sci. Total Environ.* **2014**, *491*, 108−117.

(33) Liagkouridis, I.; Cousins, A. P.; Cousins, I. T. Physical-chemical properties and evaluative fate modelling of 'emerging' and 'novel' brominated and organophosphorus flame retardants in the indoor and outdoor environment. *Sci. Total Environ.* **2015**, *524*, 416−426.

(34) Rudel, R. A.; Dodson, R. E.; Perovich, L. J.; Morello-Frosch, R.; Camann, D. E.; Zuniga, M. M.; Yau, A. Y.; Just, A. C.; Brody, J. G. Semivolatile Endocrine-Disrupting Compounds in Paired Indoor and Outdoor Air in Two Northern California Communities. *Environ. Sci. Technol.* **2010**, *44* (17), 6583−6590.

(35) Rauert, C.; Lazarov, B.; Harrad, S.; Covaci, A.; Stranger, M. A review of chamber experiments for determining specific emission rates and investigating migration pathways of flame retardants. *Atmos. Environ.* **2014**, *82*, 44−55.

(36) Webster, T. F.; Stapleton, H. M.; McClean, M. D. Exposure to Polybrominated Diphenyl Ethers in the Indoor Environment. *Fire Technol.* **2015**, *51* (1), 85−95.

(37) Bjorklund, J. A.; Thuresson, K.; Cousins, A. P.; Sellstrom, U.; Emenius, G.; de Wit, C. A. Indoor Air Is a Significant Source of Tri-decabrominated Diphenyl Ethers to Outdoor Air via Ventilation Systems. *Environ. Sci. Technol.* **2012**, *46* (11), 5876−5884.

(38) Melymuk, L.; Bohlin-Nizzetto, P.; Kukučka, P.; Vojta, Š.; Kalina, J.; Čupr, P.; Klánová, J. Seasonality and indoor/outdoor relationships of flame retardants and PCBs in residential air. *Environ. Pollut.* **2016**, *218*, 392−401.

(39) Venier, M.; Audy, O.; Vojta, Š.; Bečanová, J.; Romanak, K.; Melymuk, L.; Krátká, M.; Kukučka, P.; Okeme, J.; Saini, A.; Diamond, M. L.; Klánová, J. Brominated flame retardants in the indoor environment — Comparative study of indoor contamination from three countries. *Environ. Int.* **2016**, *94*, 150−160.

(40) D'Hollander, W.; Roosens, L.; Covaci, A.; Cornelis, C.; Reynders, H.; Campenhout, K. V.; Voogt, P. d.; Bervoets, L. Brominated flame retardants and perfluorinated compounds in indoor dust from homes and offices in Flanders, Belgium. *Chemosphere* **2010**, *81* (4), 478−487.

(41) Mandalakis, M.; Atsarou, V.; Stephanou, E. G. Airborne PBDEs in specialized occupational settings, houses and outdoor urban areas in Greece. *Environ. Pollut.* **2008**, *155* (2), 375−382.

(42) Watkins, D. J.; McClean, M. D.; Fraser, A. J.; Weinberg, J.; Stapleton, H. M.; Sjodin, A.; Webster, T. F. Exposure to PBDEs in the Office Environment: Evaluating the Relationships Between Dust, Handwipes, and Serum. *Environ. Health Perspect.* **2011**, *119* (9), 1247−1252.

(43) Liagkouridis, I.; Cousins, I. T.; Cousins, A. P. Emissions and fate of brominated flame retardants in the indoor environment: A critical review of modelling approaches. *Sci. Total Environ.* **2014**, *491*, 87−99.

(44) Kemmlein, S.; Hahn, O.; Jann, O. Emissions of organophosphate and brominated flame retardants from selected consumer products and building materials. *Atmos. Environ.* **2003**, *37* (39−40), 5485−5493.

(45) Bako-Biro, Z.; Wargocki, P.; Weschler, C. J.; Fanger, P. O. Effects of pollution from personal computers on perceived air quality, SBS symptoms and productivity in offices. *Indoor Air* **2004**, *14* (3), 178−187.

(46) Harrad, S.; Abdallah, M. A. E.; Covaci, A. Causes of variability in concentrations and diastereomer patterns of hexabromocyclododecanes in indoor dust. *Environ. Int.* **2009**, *35* (3), 573−579.

(47) de Wit, C. A.; Bjorklund, J. A.; Thuresson, K. Tri-decabrominated diphenyl ethers and hexabromocyclododecane in indoor air and dust from Stockholm microenvironments 2: Indoor sources and human exposure. *Environ. Int.* **2012**, *39* (1), 141−147.

(48) Zhang, X. M.; Diamond, M. L.; Ibarra, C.; Harrad, S. Multimedia Modeling of Polybrominated Diphenyl Ether Emissions and Fate Indoors. *Environ. Sci. Technol.* **2009**, *43* (8), 2845−2850.

(49) Dodson, R. E.; Rodgers, K. M.; Carey, G.; Cedeno Laurent, J. G.; Covaci, A.; Poma, G.; Malarvannan, G.; Spengler, J. D.; Rudel, R. A.; Allen, J. G. Flame Retardant Chemicals in College Dormitories: Flammability Standards Influence Dust Concentrations. *Environ. Sci. Technol.* **2017**, *51* (9), 4860−4869.

(50) Liang, W.; Wang, C.; Yang, C.; Yang, X. Volatile organic compounds in different interior construction stages of an apartment. *Building and Environment* **2014**, *81*, 380−387.

(51) Park, J. C.; Kwon, Y. C.; Do Jun, H. A Study on the Improvement of Indoor Air Quality of Newly-Built Apartment Houses Using Low Emission Building Materials. *J. Asian Archit. Build. Eng.* **2011**, *10* (1), 235−240.

(52) Pang, S. K.; Cho, H.; Sohn, J. Y.; Song, K. D. Assessment of the emission characteristics of VOCs from interior furniture materials during the construction process. *Indoor Built Environ.* **2007**, *16* (5), 444−455.

(53) Bohlin, P.; Audy, O.; Skrdlikova, L.; Kukucka, P.; Vojta, S.; Pribylova, P.; Prokes, R.; Cupr, P.; Klanova, J. Evaluation and guidelines for using polyurethane foam (PUF) passive air samplers in double-dome chambers to assess semi-volatile organic compounds (SVOCs) in non-industrial indoor environments. *Environ. Sci.-Process Impacts* **2014**, *16* (11), 2617−2626.

(54) Bohlin, P.; Audy, O.; Skrdlikova, L.; Kukucka, P.; Pribylova, P.; Prokes, R.; Vojta, S.; Klanova, J. Outdoor passive air monitoring of semi volatile organic compounds (SVOCs): a critical evaluation of performance and limitations of polyurethane foam (PUF) disks. *Environ. Sci.-Process Impacts* **2014**, *16* (3), 433−444.

(55) Bergman, A.; Ryden, A.; Law, R. J.; de Boer, J.; Covaci, A.; Alaee, M.; Birnbaum, L.; Petreas, M.; Rose, M.; Sakai, S.; Van den Eede, N.; van der Veen, I. A novel abbreviation standard for organobromine, organochlorine and organophosphorus flame retardants and some characteristics of the chemicals. *Environ. Int.* **2012**, *49*, 57−82.

(56) Lohmann, R.; Klanova, J.; Pribylova, P.; Liskova, H.; Yonis, S.; Bollinger, K. PAHs on a West-to-East Transect Across the Tropical Atlantic Ocean. *Environ. Sci. Technol.* **2013**, *47* (6), 2570−2578.

(57) Lohmann, R.; Klanova, J.; Kukucka, P.; Yonis, S.; Bollinger, K. Concentrations, Fluxes, and Residence Time of PBDEs Across the Tropical Atlantic Ocean. *Environ. Sci. Technol.* **2013**, *47* (24), 13967−13975.

(58) Lohmann, R.; Klanova, J.; Kukucka, P.; Yonis, S.; Bollinger, K. PCBs and OCPs on a East-to-West Transect: The Importance of Major Currents and Net Volatilization for PCBs in the Atlantic Ocean. *Environ. Sci. Technol.* **2012**, *46* (19), 10471−10479.

(59) Vykoukalova, M.; Venier, M.; Vojta, S.; Melymuk, L.; Becanova, J.; Romanak, K.; Prokes, R.; Okeme, J. O.; Saini, A.; Diamond, M. L.; Klanova, J. Organophosphate esters flame retardants in the indoor environment. *Environ. Int.* **2017**, *106*, 97−104.

(60) Antweiler, R. C. Evaluation of Statistical Treatments of Left-Censored Environmental Data Using Coincident Uncensored Data Sets. II. Group Comparisons. *Environ. Sci. Technol.* **2015**, *49* (22), 13439−13446.

(61) Batterman, S.; Godwin, C.; Chernyak, S.; Jia, C.; Charles, S. Brominated flame retardants in offices in Michigan, U.S.A. *Environ. Int.* **2010**, *36* (6), 548−556.

(62) Cequier, E.; Ionas, A. C.; Covaci, A.; Maria Marce, R.; Becher, G.; Thomsen, C. Occurrence of a Broad Range of Legacy and Emerging Flame Retardants in Indoor Environments in Norway. *Environ. Sci. Technol.* **2014**, *48* (12), 6827−6835.

(63) Blanchard, O.; Glorennec, P.; Mercier, F.; Bonvallot, N.; Chevrier, C.; Ramalho, O.; Mandin, C.; Le Bot, B. Semivolatile Organic Compounds in Indoor Air and Settled Dust in 30 French Dwellings. *Environ. Sci. Technol.* **2014**, *48* (7), 3959−3969.

(64) Tao, F.; Abdallah, M. A. E.; Harrad, S. Emerging and Legacy Flame Retardants in UK Indoor Air and Dust: Evidence for Replacement of PBDEs by Emerging Flame Retardants? *Environ. Sci. Technol.* **2016**, *50* (23), 13052−13061.

(65) Bergh, C.; Torgrip, R.; Emenius, G.; Ostman, C. Organophosphate and phthalate esters in air and settled dust - a multi-location indoor study. *Indoor Air* **2011**, *21* (1), 67−76.

(66) Hartmann, P. C.; Burgi, D.; Giger, W. Organophosphate flame retardants and plasticizers in indoor air. *Chemosphere* **2004**, *57* (8), 781−787.

(67) Melymuk, L.; Bohlin-Nizzetto, P.; Vojta, S.; Kratka, M.; Kukucka, P.; Audy, O.; Pribylova, P.; Klanova, J. Distribution of legacy and emerging semivolatile organic compounds in five indoor matrices in a residential environment. *Chemosphere* **2016**, *153*, 179−186.

(68) Carlsson, H.; Nilsson, U.; Ostman, C. Video display units: An emission source of the contact allergenic flame retardant triphenyl phosphate in the indoor environment. *Environ. Sci. Technol.* **2000**, *34* (18), 3885−3889.

(69) Ni, Y.; Kumagai, K.; Yanagisawa, Y. Measuring emissions of organophosphate flame retardants using a passive flux sampler. *Atmos. Environ.* **2007**, *41* (15), 3235−3240.

(70) Duan, H.; Yu, D. F.; Zuo, J.; Yang, B.; Zhang, Y. K.; Niu, Y. N. Characterization of brominated flame retardants in construction and demolition waste components: HBCD and PBDEs. *Sci. Total Environ.* **2016**, *572*, 77−85.

(71) Babrauskas, V.; Lucas, D.; Eisenberg, D.; Singla, V.; Dedeo, M.; Blum, A. Flame retardants in building insulation: a case for re-evaluating building codes. *Build. Res. Informat.* **2012**, *40* (6), 738−755.

(72) CECBP *Brominated and Chlorinated Organic Chemical Compounds Used as Flame Retardants*; Meeting of the California Environmental Contaminant Biomonitoring Program (CECBP) Scientific Guidance Panel (SGP): 2009.

(73) von der Recke, R.; Vetter, W. Synthesis and characterization of 2,3-dibromopropyl-2,4,6-tribromophenyl ether (DPTE) and structurally related compounds evidenced in seal blubber and brain. *Environ. Sci. Technol.* **2007**, *41* (5), 1590−1595.

(74) Harju, M. *Emerging "New" Brominated Flame Retardants in Flame Retarded Products and the Environment*; The Norwegian Pollution Control Authority: Oslo, 2009.

(75) EFSA. EFSA Panel on Contaminants in the Food Chain (CONTAM). Scientific Opinion on Emerging and Novel Brominated Flame Retardants (BFRs) in Food. *EFSA Journal* 2012, *10* (10), 2908, 133 pp.

(76) Abbasi, G.; Saini, A.; Goosey, E.; Diamond, M. L. Product screening for sources of halogenated flame retardants in Canadian house and office dust. *Sci. Total Environ.* **2016**, *545*, 299−307.

(77) Rauert, C.; Harrad, S.; Suzuki, G.; Takigami, H.; Uchida, N.; Takata, K. Test chamber and forensic microscopy investigation of the transfer of brominated flame retardants into indoor dust via abrasion of source materials. *Sci. Total Environ.* **2014**, *493*, 639−648.

(78) Muenhor, D.; Harrad, S. Within-room and within-building temporal and spatial variations in concentrations of polybrominated diphenyl ethers (PBDEs) in indoor dust. *Environ. Int.* **2012**, *47*, 23−27.

(79) Allen, J. G.; McClean, M. D.; Stapleton, H. M.; Nelson, J. W.; Webster, T. F. Personal exposure to Polybrominated Diphenyl Ethers (PBDEs) in residential indoor air. *Environ. Sci. Technol.* **2007**, *41* (13), 4574−4579.

APPENDIX 12

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Organophosphate esters flame retardants in the indoor environment

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ABSTRACT

Concentrations of 13 organophosphate ester flame retardants (OPEs) were measured in air, dust and window wipes from 63 homes in Canada, the Czech Republic and the United States in the spring and summer of 2013 to look for abundances, differences among regions, and partitioning behavior. In general, we observed the highest concentrations for halogenated OPEs, particularly TCEP, TCIPP and TDCIPP, and also non-halogenated TPHP. Differences between regions strongly depended on the matrix. The concentrations of OPEs in dust were significantly higher in the US than in Canada (CAN) and Czech Republic (CZ). CZ had the highest concentrations in window film and CAN in air. ΣOPE concentrations were 2–3 and 1–2 orders of magnitude greater than ΣBFRs in air, and dust and window films, respectively. We found a significant relationship between the concentrations in dust and air, and between the concentrations in window film and air for OPEs with log K_{OA} values < 12, suggesting that equilibrium was reached for these compounds but not for those with log $K_{OA} > 12$. This hypothesis was confirmed by a large discrepancy between values predicted using a partitioning model and the measured values for OPEs with log K_{OA} values > 12.

1. Introduction

Polybrominated diphenyl ethers (PBDEs) are the most thoroughly studied class of flame retardants (FRs). They were widely used in numerous household products until the early 2000s when two of the commercial mixtures, Penta and OctaBDE, were withdrawn from the market in the US due to mounting evidence of adverse health effects and widespread environmental presence. This resulted in an increased use of alternative compounds, mostly either brominated compounds (e.g. 2-Ethylhexyl-2,3,4,5-tetrabromobenzoate (EHTBB), Bis(2-ethyl-1-hexyl) tetrabromophthalate (BEHTBP) or organophosphate esters (OPEs).

OPEs are phosphoric acid esters used as FRs, plasticizers and antifoaming agents (Rauert and Harrad, 2015). Their application in products ranges from textiles, polyurethane foam (PUF) upholstered furniture, and electronics to construction materials (e.g., building insulation) and vehicles (Marklund et al., 2003; van der Veen and de Boer, 2012; Wei et al., 2015). While chlorinated and brominated OPEs see widespread use mostly as FRs, the non-halogenated OPEs are used also as plasticizers, lubricants and pore size regulators (Andresen et al., 2004). Since OPEs are typically used as additive chemicals and are therefore not covalently bound to polymeric materials, they can easily migrate from products into the environment by means of volatilization, leaching and abrasion, and direct transfer to dust (Marklund et al., 2003; van der Veen and de Boer, 2012; Wei et al., 2015).

Due to their physical and chemical characteristics, OPEs are ubiquitous in various environmental compartments worldwide and have been detected in abiotic matrices such as sediment (Cao et al., 2012), surface and groundwater water (Regnery et al., 2010; Regnery et al., 2011; Venier et al., 2014), outdoor air including remote locations (Salamova et al., 2016; Sühring et al., 2016a), indoor air (Marklund et al., 2005; van der Veen and de Boer, 2012), and house dust (Dodson et al., 2012; Stapleton et al., 2009). They are also found in biota (van der Veen and de Boer, 2012) and human breast milk (Sundkvist et al., 2010), indicating that these compounds are bioavailable and might bioaccumulate (Greaves et al., 2016). Studies on the toxicity of OPEs are still limited but some OPEs have been reported to be mutagenic, carcinogenic, and neurotoxic, as well as potential developmental and

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reproductive toxins (Behl et al., 2015; Hendriks and Westerink, 2015; Schweizer et al., 2007; van der Veen and de Boer, 2012).

Due to its persistence, bioaccumulative potential and toxicity, Tris (2-chloroethyl) phosphate (TCEP) has been restricted from use in concentrations greater than 5 mg/kg in toys for children up to 3 years of age and for any toys intended for mouthing and has been designated as a substance of very high concern in the European Union (Toy Safety Directive, 2009/48/EC, European Chemicals Agency, 2015). As such, it is no longer produced in Europe (Green et al., 2008; Schreder et al., 2016; Sühring et al., 2016a). Some reports suggest that it has been replaced by other FRs, primarily Tris(1-chloro-2-propyl) phosphate (TCIPP). In 2001, this structurally similar and relatively cheap replacement for TCEP, represented approximately 80% of the chlorinated OPEs used in Europe (Leisewitz et al., 2001). In 2014 Canada prohibited new use of TCEP in products containing PUF intended for children under 3 years of age (Canada Gazette, 2014). TCEP was restricted for new uses in several US states starting in 2011 (Safer States) and is listed as a carcinogen under California's Proposition 65. TCIPP has been recommended for designation as “toxic” under the Canadian Environmental Protection Act (Environment and Climate Change Canada, 2016). The proposed risk management measures are limiting its use to < 0.1% in mattresses and upholstered furniture.

Tris(1,3-dichloro-2-propyl) phosphate (TDCIPP) appears to be used as a substitute for brominated FRs. TDCIPP is primarily used in foams in the automotive industry and to some extent also in upholstered furniture (Cooper et al., 2016). Tris(2,3-dibromopropyl) phosphate (TDBPP), a brominated analog for TDCIPP, was banned in children's sleepwear in the US in 1977 due to carcinogenicity concerns after mutagenic metabolites were detected in children's urine (Gold et al., 1978) and it is listed as a carcinogen in California's Proposition 65. TDCIPP is also listed as a carcinogen in California's Proposition 65 but has not been recommended for designation as “toxic” under the Canadian Environmental Protection Act (Environment and Climate Change Canada, 2016).

Triphenyl phosphate (TPHP), one of the most effective FRs used in polymers (van der Veen and de Boer, 2012), is used also in hydraulic fluids (Andresen et al., 2004). TPHP is a component in the Firemaster (FM) 550 mixture, introduced in 2004 as a replacement for the PentaBDE commercial mixture used in upholstered furniture. It is also used as a plasticizer in, for example, nail polish (Mendelsohn et al., 2016) and electronic components such as televisions and monitor screens (Kajiwara et al., 2011). EHDPP is mainly used as a flame retardant/plasticizer in flexible PVC (e.g., wire and cable insulators, connectors) and in certain food packaging in the US (Brooke et al., 2009).

In this study, we measured the concentrations of 13 OPEs - four halogenated (TCEP, TCIPP, TDCIPP and TDBPP) and nine non-halogenated (TPHP, EHDPP, TEHP, TNBP, o-TMPP, p-TMPP, TIPPP, TDMPP and TBPP) in air, dust and window film from homes in three different countries (Czech Republic, Canada and USA) during the spring and summer of 2013. These three matrices were chosen as the most relevant for human exposure, as well as being novel and convenient. The goals of this project were to evaluate the partitioning behavior of OPEs in the indoor environment, to compare within-house differences, and to examine regional differences between Central Europe and North America. Two companion papers reported on levels of brominated flame retardants (Venier et al., 2016) and perfluorinated alkyl substances (Karásková et al., 2016) in the same homes.

2. Experimental section

2.1. Sample collection

Air, dust and window film samples were collected from a total of 63 houses and apartments (20 homes each in Brno, Czech Republic and Bloomington, IN, US and 23 in Toronto, ON, Canada) during a sampling period of 28 days in May–August 2013. Samples from one room, usually the bedroom, were collected in each home while a second room, usually the living room, was sampled in at least nine randomly chosen homes in each country. Participants were recruited as a “sample of convenience” among colleagues, friends, relatives and acquaintances.

Sampling involved deploying polyurethane foam (PUF) passive air samplers to estimate air concentrations, collecting settled floor dust into nylon socks by vacuuming, and collecting interior window films as representative of surface films on interior surfaces using Kimwipes. Details of sample collection can be found elsewhere (Venier et al., 2016) and only a brief description is provided here. Before sampling, all matrices (PUF disks, Kimwipes and nylon vacuum socks) were precleaned in a Soxhlet extractor (8 h in acetone, then 8 h in toluene), dried and packed in aluminum foil and transported to the sites. Field blanks were collected by exposing pre-cleaned matrices during sample retrieval.

On day 1, floors were vacuumed and windows were cleaned with Kimwipes moistened with 2-propanol until no dirt was visible (Kimwipes were not saved at this time), PUF passive air samplers were deployed, and participants were asked not to vacuum rooms or wash the windows where the samplers were located until the end of the campaign. PUF disks were exposed to indoor air using a single-bowl in US and Canada or double-bowl in the Czech Republic (see Fig. S1) passive sampler housing for 28 days. Sampling rates for each passive air sampler configuration were calculated in a separate experiment by simultaneously deploying single- and double-bowl samplers along with active air samplers (Venier et al., 2016). More details on the calibration of the passive air samplers are reported in the Supporting Information. For this study, we used a sampling rate of 1.6 m^3/day for the single-bowl sampler and 0.82 m^3/day for the double-bowl sampler. Given the difficulty of calculating accurate and meaningful sampling rates, this approach seems reasonable.

On day 28, PUF discs were retrieved and window wipe samples were collected using a pre-cleaned Kimwipe moistened with 2-propanol (Kimwipes were saved at this time). Windows were wiped until no dirt was visible on the Kimwipe. The sampled area averaged at 0.32 m^2 for Canada, 0.93 m^2 for US, and 1.79 m^2 for the Czech Republic. Floor dust samples were collected using a pre-cleaned nylon sampling sock inserted into the tube of a conventional household vacuum cleaner, vacuuming the largest possible floor area and recording the area. After collection, all samples were wrapped in aluminum foil and sealed in plastic bags for transport to the laboratory. Samples were stored at − 18 °C until extraction and analysis.

2.2. Target compounds and chemicals

The list of target compounds is reported in Table 1. The following OPEs standards were purchased from Wellington Laboratories (Guelph, ON, Canada): Tri-*n*-butylphosphate (TnBP), Tris(2-chloroethyl) phosphate (TCEP), Tris(1-chloro-2-propyl) phosphate (TCIPP), Tris(1,3-dichloro-2-propyl) phosphate (TDCIPP), Triphenyl phosphate (TPHP), 2-Ethylhexyl-diphenyl phosphate (EHDPP), Tris(2-ethylhexyl) phosphate (TEHP), Tri-*o*-tolyl phosphate (o-TMPP), Tri-*p*-tolyl phosphate (p-TMPP), Tris(2-isopropylphenyl) phosphate (TIPPP), and Tris(3,5-dimethylphenyl) phosphate (TDMPP). Tris(2,3-dibromopropyl) phosphate (TDBPP) was purchased from AccuStandard (New Haven, CT). Tris(4-*tert*-butylphenyl) phosphate (TBPP) was purchased from Sigma-Aldrich (St. Louis, MO). d_{12}-Tris(2-chloroethyl) phosphate (d_{12}-TCEP), $^{13}C_{18}$-triphenyl phosphate (MTPP) and $^{13}C_{12}$-BDE-77 were purchased from Wellington. The internal quantitation standards, d_{10}-anthracene, d_{12}-benz[*a*]anthracene, and d_{12}-perylene, were obtained from Chem Service (West Chester, PA). All solvents were HPLC or Optima grade. Silica gel (100–200 mesh, 75–150 μm, Grade 644) and granular anhydrous sodium sulfate (Na_2SO_4) were purchased from Fisher Scientific (Pittsburgh, PA).

Table 1
OPEs names, abbreviations (alternate name between parentheses), CAS numbers and log K_{OA} values (EPI Suite log K_{OA} values were used in this paper; the others are provided for reference only).

Name	Abbreviations	CAS number	log K_{OA} (Sühring et al., 2016b)		
			EPI Suite	SPARC	Absolv
Tris(2-chloroethyl) phosphate	TCEP	115–96-8	7.6	7.0	8.9
Tris(1-chloro-2-propyl) phosphate	TCIPP (TCPP)	13674–84-5	8.5	7.6	10.0
Tris(1,3-dichloro-2-propyl) phosphate	TDCIPP (TDCPP)	13674–87-8	10.6	10.3	12.0
Tris(2,3-dibromopropyl) phosphate	TDBPP	126–72-7	14.1	12.9	16.3
Tri-phenylphosphate	TPHP (TPP)	115–86-6	10.5	10.3	14.0
2-ethylhexyldiphenylphosphate	EHDPP	1241–94-7	11.3	10.6	14.1
Tris(2-ethylhexyl) phosphate	TEHP	78–42-2	11.9	12.0	14.5
Tri n-butyl phosphate	TNBP (TBP)	126–73-8	7.7	7.0	9.5
Tri-o-tolylphosphate	o-TMPP (TOTP)	78–30-8	12.0	11.6	15.1
Tri-p-tolylphosphate	p-TMPP (TPTP)	78–32-0	12.0	11.8	15.1
Tris(2-isopropylphenyl) phosphate	TIPPP (T2IPPP)	64532–95-2	14.0	13.5	17.5
Tris(3,5-dimethylphenyl) phosphate	TDMPP (T35DMPP)	25653–16-1	13.5	12.9	16.2
Tris(4-*tert*-butylphenyl) phosphate	TBPP	78–33-1	15.0	14.7	18.1

2.3. Extraction

The US samples were analyzed in Bloomington, IN at Indiana University (IU) and the Canadian and Czech samples were analyzed in Brno, Czech Republic at RECETOX. Details of the analytical protocols are given in the SI and are summarized here.

Before extraction, all samples were spiked with known amounts of recovery standards [$^{13}C_{18}$-triphenyl phosphate (MTPP) at RECETOX and $^{13}C_{18}$-triphenyl phosphate (MTPP) and d_{12}-Tris(2-chloroethyl) phosphate (d_{12}-TCEP) at IU]. PUF and Kimwipes were extracted in 250 mL dichloromethane (DCM) using automated warm Soxhlet extraction (three cycles: 40 min warm Soxhlet followed by 20 min of solvent rinsing and concentration of the extract) at RECETOX and with 400 mL of acetone: *n*-hexane (1:1, v/v) using Soxhlet extraction for 24 h at IU. In both labs, socks with dust were weighed, the dust was sieved to < 500 μm, approximately 100 mg were weighed and the remaining dust was wrapped in foil for future use. The sock rinsate (30 mL hexane:acetone, 1:1) and the sieved fraction of dust were sonicated with 30 mL acetone: n-hexane (1:1, *v*/v). The dust was allowed to settle, the liquid extract was removed, and this procedure was repeated twice.

2.4. Sample fractionation and clean-up

At RECETOX, 30% of the extract for each sample (measured by weighing; the rest of the sample was used for the analysis of other compounds) underwent non-destructive clean-up on a 5 g non-modified activated silica column topped with 1 cm of Na_2SO_4 using 20 mL DCM for the first fraction and 20 mL acetone:DCM (7:3, v/v) for the second fraction. OPEs eluted in the second fraction. The OPE fraction was concentrated under N_2 and transferred into a 1 mL vial, solvent exchanged to nonane, further concentrated to approximately 0.5 mL and spiked with the injection standard $^{13}C_{12}$-BDE-77. At IU, extracts were reduced in volume to about 2 mL by rotary evaporation. The solvent was exchanged to hexane and the extracts were fractionated on a column containing 3.5% water deactivated silica gel. The column was eluted with 25 mL of 7:3 acetone in DCM. After N_2 blow down to about 1 mL, the samples were spiked with the quantitation internal standards (d_{10}-anthracene, d_{12}-benz[*a*]anthracene and d_{12}-perylene).

2.5. Instrumental analysis

At RECETOX, OPEs were analyzed using an Agilent 7890A gas chromatograph (GC) equipped with a 15 m Restek RTX 1614 column (with 60 cm retention gap) coupled to a Waters APGC XEVO TQ-S tandem mass spectrometer (MS/MS). The mass spectrometer was operated in positive atmospheric pressure ionization mode (APGC +) using multiple reaction monitoring mode (MRM). At least two transitions were recorded for each compound. The following transitions were used for quantification (confirmation): 267 > 99 (267 > 211) for TNBP, 287 > 99 (249 > 99), for TCEP, 327 > 99 (329 > 99) for TCIPP, 431 > 99 (431 > 209) for TDCIPP, 327 > 215 (327 > 152) for TPHP, 362 > 251 (362 > 94) for EHDPP, 435 > 99 (435 > 323) for TEHP, 368 > 277 (369 > 278) for o-TMPP, 368 > 261 (369 > 165) for m-TMPP and p-TMPP, 453 > 118 (453 > 251) for TIPPP, 411 > 194 (411 > 179) for TDMPP, 618 > 137 (698 > 99) for TDBPP, 495 > 327 (494 > 211) for TBPP, 345 > 277 (345 > 164) for $^{13}C_{12}$-TPHP and 498 > 338 (496 > 336) for $^{13}C_{12}$-BDE-77. Additional parameters were set as follows: source temperature 150 °C, cone gas flow 160 L/h and auxillary gas flow 220 L/Hr. Injection was splitless 1 μL at 250 °C, with He as the carrier gas at 1.2 mL min^{-1}. The GC temperature program was 80 °C with a 1 min hold, then 30 °C min^{-1} to 140 °C, followed by 4 °C min^{-1} to 175 °C, then 8 °C min^{-1} to 270 °C and 15 °C min^{-1} to 325 °C with a 7 min hold. Makeup gas pressure was 60 psi and transfer line temperature was set to 300 °C.

At IU, an Agilent 6890 series GC coupled to an Agilent 5973 MS was used to quantify OPEs. The MS was operated in the electron impact mode. The GC resolution was achieved with a 30 m × 250 μm i.d. × 0.25 μm film thickness DB-5MS Ultra Inert capillary column (Agilent Technologies, Santa Clara, CA). One μL of the sample was injected in the pulsed splitless mode at 280 °C. The GC–MS interface was kept at 300 °C. Temperatures of the ion source and quadrupole were set at 230 °C and 150 °C, respectively. High purity helium (99.999%; Liquid Carbonic, Chicago) was used as the carrier gas. The GC oven temperature was held at 90 °C for 1 min, increased to 170 °C at 10 °C/min, held for 3 min, then increased to 230 °C at 10 °C/min, held for 4 min, then increased to 260 °C at 5 °C/min, finally increased to 300 °C at 10 °C/min, and held for 4 min. TCIPP and TEHP were monitored using m/z 99 and 125. Monitoring ions for the other OPEs were: m/z 155 and 99 for TNBP, m/z 249 and 63 for TCEP, m/z 75 and 191 for TDCIPP, m/a 326 and 325 for TPHP, m/z 251 and 250 for EHDPP, m/z 410 and 411 for TDMPP, m/z 368 and 367 for o-TMPP and p-TMPP, m/z 368 and 165 for m-TMPP, m/z 201 and 119 for TDBPP, m/z 494 and 479 for TBPP. For the surrogate and internal standards, the following ions were monitored: MTPP, m/z 343; d_{12}-TCEP, m/z 261; d_{10}-anthracene, m/z 188; d_{12}-benz[a]anthracene, m/z 240, and d_{12}-perylene, m/z 264. OPE data were quantitated using the internal standard method at both labs.

2.6. Quality control and quality assurance

Several measures were taken to ensure data comparability between the different laboratories and the accuracy and reliability of the measurements. For dust samples, spiked ($n = 3$) and non-spiked ($n = 3$) house dust SRM 2585 were analyzed and compared with literature data (Table SI2). The results were also used to ensure the data comparability between the laboratories.

The same blank treatment procedure and method limit of detection (LOD) calculation was used as described elsewhere (Karásková et al., 2016; Venier et al., 2016). Solvent blanks were used to evaluate contamination from the laboratory procedures. Solvent blank levels were low and no correction was necessary. Three field blanks per matrix per country were collected and analyzed. The mass of the target compounds in each sample were then compared to the average mass in the field blanks (on a country and matrix specific basis – see Table S1 in the Supporting Information for more details) and treated as follows: If the blank level was < 10% of the measured level, there was no correction. If the blank level was 10–35% of the measured level, the blank level was subtracted from the measured level. If the blank level was > 35% of the measured level, the value was reported as "non-detect."

For air and window film samples, the recoveries of individual compounds were determined using a set of pre-cleaned (8 h in acetone and 8 h in DCM) sampling media ($n = 8$) spiked with the native analytes prior to extraction. The recoveries of target compounds in matrix spike experiments varied between 70% and 120%. Average surrogate recoveries were mostly within the 50–150% range, which is considered acceptable. Final values were not recovery corrected.

For statistical data analysis, values below detection were replaced by $\sqrt{2}/2$*LOD (Antweiler, 2015). Compounds with low detection (generally < 60% detection frequency in a given matrix/room) were not used for statistical analysis.

2.7. *Data analysis*

Basic calculations, descriptive statistics and graphs were made using Microsoft Excel software and Statistica 12. Central trends were described with median concentrations because the data were not normally distributed. Differences between sample populations were determined using a nonparametric Kruskal-Wallis, Mann-Whitney U and Wilcoxon tests. Differences were considered significant at $p < 0.05$.

3. Results and discussion

A summary of the results for dust, window film and air from all three countries is reported in Table 2. An overview of the concentrations of the main targeted OPEs is shown in Fig. S2, which depicts both regional trends and compound patterns. Box plots for selected compounds are shown in Fig. S3. In general, we observed the highest concentrations for TCEP, TCIPP, TDCIPP and TPHP. Interestingly, all three matrices in all the three countries were dominated by halogenated OPEs.

Regional patterns of OPE concentrations followed those seen for PBDEs and novel BFRs (Harrad et al., 2008; Venier et al., 2016). For OPEs, we note that the comparison of differences country-to-country must be interpreted with caution due to the challenges associated with OPE measurement. We found that the regional trends of OPEs strongly depended on the matrix. The concentrations of OPEs in dust were significantly higher in the US than in Canada (CAN) and Czech Republic (CZ) (see Table 2). CZ had the highest concentrations in window film and CAN in air.

In particular, dust samples were dominated by TDBPP in the US (median 4530 ng/g, range ND - 11,000 ng/g), TPHP in CAN (median 2350 ng/g, range 377–31,900 ng/g) and TCIPP in CZ (median 1860 ng/g, range 163–26,700 ng/g). Although TCIPP was also prevalent in CZ window film samples (median 566 ng/m^2, range 15.0–6670 ng/m^2), US and CAN window film samples primarily had TDCIPP (medians 83.2 ng/m^2 and 82.3 ng/m^2, ranges ND – 542 ng/m^2 and 20.8–3590 ng/m^2, respectively). In all three countries, air concentrations were dominated by TCIPP, with the highest levels found in CAN (median 73.6 ng/m^3, range 7.68–4190 ng/m^3), followed by the US (median 26.3 ng/m^3, range 0.280–226 ng/m^3) and the CZ (median 16.4 ng/m^3, range 3.62–139 ng/m^3). See Fig. S2 and Table 2 for details. The abundance of TCIPP in air is consistent with its common use and relatively high vapor pressure (log Pl − 2.1 EPI Suite V4.1; − 1.3 Pa, ABSOLV; Zhang et al., 2016).

3.1. *Dust*

The highest concentrations of halogenated OPEs were found in US dust, with medians ranging from 1440 ng/g (TCEP) to 4530 ng/g (TDBPP), followed by CAN and CZ. With the exception of TCIPP, the differences between US and the other two countries were significant ($p < 0.05$). TDCIPP was significantly lower in CZ than in CAN and US while TDBPP was not detected in CZ at all.

Among non-halogenated OPEs, TPHP was generally most abundant. TPHP was significantly lower in CZ (median 811 ng/g) than in the US (median 3040 ng/g) and in CAN (median 2350 ng/g). In contrast, EHDPP did not follow this trend, with similar concentrations in all countries. For TNBP, TBPP and TIPPP, levels were significantly higher in US than in the other two countries. Both isomers of TMPP were significantly higher in CZ than in CAN. In the US, p-TMPP showed similar concentrations to CZ, while o-TMPP was not detected in the US. TDMPP was only detected in a single US dust sample.

Concentrations of OPEs in dust samples measured here were comparable to those reported in the literature, with a few exceptions (see Table S3 for a comprehensive list of literature data). Remarkably higher levels of TCEP and TDCIPP were found in Sweden by Bergh et al. (2011). TCEP concentrations in Spain (García et al., 2007) were higher than those in CZ homes reported here but sampling occurred at a much earlier year, before any restrictions were put in place. Dust measured in California by Dodson et al. (2012) was higher than our data from Indiana, which could reflect the legacy of more stringent flammability standards in California. TDBPP in dust was only reported by Dodson et al. (2012) for California and were lower than those measured here. Somewhat higher TNBP concentrations than here were reported by several European studies (Bergh et al., 2011; García et al., 2007; Marklund et al., 2003; Van den Eede et al., 2011) and higher TPHP values were found in the US by Stapleton et al. (2009) compared with our results. Conversely, lower TPHP values were found in Germany by Brommer et al. (2012) and Sweden (Marklund et al., 2003) relative to results from all our locations. While TEHP values measured in dust were higher in our study compared to the literature, o-TMPP were lower than those reported by Bergh et al. (2011) for Sweden.

3.2. *Window film*

The highest median concentration (566 ng/m^2, TCIPP) among halogenated OPEs was found in CZ window film samples. TDBPP was only detected in a single CAN sample. While TDCIPP values were found to be significantly higher in the US and CAN than in CZ, a reverse trend was apparent in the case of TCIPP. Finally, TCEP was found to be significantly lower in CAN than in the US and CZ.

For non-halogenated OPEs, the highest median concentrations in window film samples were measured for TNBP in CAN and CZ (29.6 and 72.6 ng/m^2, respectively) and EHDPP in the US (48.1 ng/m^2). The remaining non-halogenated OPEs (o-TMPP, p-TMPP, TIPPP, TDMPP and TBPP) were measured at low concentration levels (range: 0.134 to 70.5 ng/m^2) and with lower detection frequencies. Most of the non-halogenated OPEs in window film were significantly higher in North America than in CZ, with the exception of TNBP, for which concentrations were significantly higher in CAN and CZ. To our knowledge, no literature data exist for OPEs in window film.

Table 2

Summary results for dust (ng/g), window film (ng/m^2) and air (ng/m^3) samples including median, minimum, maximum, percentage of samples above the detection limit (%), and Kruskal-Wallis test results. The Kruskal-Wallis results are to be read across countries for each compound; compound concentrations from countries that have different letter are significantly different from one another ($p < 0.05$). ND indicates that a compound was not detected in the samples.

		US						CAN						CZ					
		Median	Range			%		Median	Range			%		Median	Range			%	
Dust [ng/g]	TCEP	1440	123	–	9080	100	a	181	73.7	–	6750	100	b	155	41.8	–	1680	100	b
	TCIPP	2790	177	–	26,500	100	a	1470	270	–	39,300	100	a	1860	163	–	26,700	100	a
	TDCIPP	3680	ND	–	8940	97	a	917	206	–	9530	100	b	183	56	–	1220	100	c
	TDBPP	4530	ND	–	11,000	60	a	0.193	ND	–	41.4	26	b	ND					
	TPHP	3040	22.5	–	24,800	100	a	2350	377	–	31,900	100	a	811	72.8	–	11,000	100	b
	EHDPP	889	ND	–	4760	97	a	754	150	–	8400	100	a	836	144	–	8230	100	a
	TEHP	1360	ND	–	18,600	97	a	101	18.6	–	611	100	b	153	37.2	–	10,300	100	b
	TNBP	114	81.6	–	4270	100	a	63	25.5	–	288	100	b	51.6	6.81	–	160	100	b
	o-TMPP	ND						0.712	ND	–	7.02	71	a	3.34	ND	–	249		b
	p-TMPP	82.5	ND	–	453	47	a	6.18	0.871	–	116	100	b	201	14	–	2500	100	a
	TIPPP	52.4	ND	–	317	77	a	20.4	4.26	–	74.3	100	b	11.9	1.64	–	508	100	b
	TDMPP	35.2	ND	–	146	3	a	ND						ND					
	TBPP	41.8	ND	–	500	73	a	4.85	0.959	–	57.7	100	b	6.49	0.254	–	39.6	100	b
Window film [ng/m^2]	TCEP	56.5	8.17	–	267	100	a	12.5	5.09	–	258	97	b	46.1	2.35	–	1630	97	a
	TCIPP	41.7	4.42	–	959	100	b	57.9	24.6	–	3620	97	b	566	15	–	6670	97	a
	TDCIPP	83.2	ND	–	542	90	a	82.3	20.8	–	3590	97	a	13.6	1.01	–	181	97	b
	TDBPP	ND						0.060*	ND	–	1.31	3	a	ND				0	
	TPHP	39.6	4.2	–	1050	100	a	20.2	8.82	–	426	97	a	4.56	0.961	–	25.3	97	b
	EHDPP	48.1	3.66	–	656	100	a	27.1	9.68	–	491	97	a	9.88	ND	–	103	93	b
	TEHP	23.2	ND	–	283	63	a	3.72	0.978	–	49.6	97	b	6.85	0.749	–	91.9	97	b
	TNBP	8.65	3.14	–	123	100	b	29.8	12.1	–	1530	97	a	72.6	2.93	–	538	97	a
	o-TMPP	7.45	ND	–	51.2	7	a	0.354	ND	–	7.76	51	b	0.006	ND	–	1.17		c
	p-TMPP	12.7	ND	–	22.1	7	a	0.18	ND	–	13	43	b	0.02	ND	–	1.57	47	b
	TIPPP	6.36	ND	–	70.5	7	a	0.34	0.134	–	29.2	97	b	0.084	ND	–	0.223	90	c
	TDMPP	ND						ND						0.401	ND	–	0.814	3	a
	TBPP	6.33	ND	–	39.8	7	a	0.089	ND	–	1.89	63	b	0.009	ND	–	39.8	10	c
Air [ng/m^3]	TCEP	6.81	ND	–	28.9	97	a	6.35	1.34	–	145	97	a	2.96	0.781	–	36.4	97	b
	TCIPP	26.3	0.28	–	226	100	b	73.6	7.68	–	4190	97	a	16.4	3.62	–	139	97	b
	TDCIPP	0.372	ND	–	6.46	80	a	0.525	0.045	–	5.5	97	a	0.311	0.228	–	0.572	97	a
	TDBPP	23.6	ND	–	45		a							ND				0	
	TPHP	0.799	ND	–	3.31	97	a	0.723	0.21	–	2.99	97	a	0.592	0.304	–	0.91	97	a
	EHDPP	0.739	ND	–	9.58	97	b	1.71	0.369	–	66.5	97	a	0.375	0.193	–	1.09	97	c
	TEHP	0.376	ND	–	24.7	53	a	0.042	0.014	–	2.38	97	b	0.037	0.02	–	0.163	97	b
	TNBP	6.07	ND	–	57	97	ab	6.2	2.61	–	461	97	a	2.34	0.745	–	33.1	97	b
	o-TMPP	0.142	ND	–	0.142	3	a	0.005	ND	–	0.05	77	b	0.001	ND	–	0.025		c
	p-TMPP	0.256	ND	–	0.256	7	a	0.002	ND	–	0.023	71	b	0.002	ND	–	0.016	63	b
	TIPPP	0.121	ND	–	0.121	7	a	0.007	0.002	–	0.034	97	b	0.005	0.003	–	0.023	97	c
	TDMPP	0.121	ND	–	0.121		a	ND						ND					
	TBPP	0.439	ND	–	1.1	97	a	ND						0.001	ND	–	0.001	3	b

3.3. *Air*

As mentioned above, the highest median concentrations among halogenated OPEs in air were measured for TCIPP, with CAN values (73.6 ng/m^3) significantly higher than those of the US (26.3 ng/m^3) and CZ (16.4 ng/m^3). TCEP concentrations in CAN and the US were significantly higher than in CZ which is consistent with regulatory actions taken against TCEP in Europe and Canada. No significant differences between countries were found for TDCIPP. While TDBPP was not detected in CAN or CZ, the US detection frequency was up to 90% and the median concentration was 23.6 ng/m^3.

We observed higher concentrations in the US and CAN compared to the CZ for TNBP (6.20 ng/m^3 in CAN), and EHDPP (1.71 ng/m^3in CAN), while TEHP was found to be significantly higher in US only (0.376 ng/m^3). No significant country trend was observed for TPHP with median concentrations ranging from 0.592 ng/m^3 in CZ to 0.799 ng/m^3 in the US. Similarly to window film samples, the remaining non-halogenated OPEs were detected rarely or at concentrations just above the limit of quantification. The only exception was TBPP, which was frequently detected in US, reaching a median concentration of 0.439 ng/m^3.

Similarly to previous studies, TCEP air concentrations were substantially lower than TCIPP concentrations (see Table SI3), which may be explained by the recent restrictions on the uses of TCEP and the associated rising usage of TCIPP as discussed above (Leisewitz et al., 2001). Lower TCIPP than TCEP concentrations were reported for dust only in Sweden (Bergh et al., 2011) and in the US (Dodson et al., 2012).

TCIPP was the most frequently measured halogenated OPE in our study. This reflects its use as a major FR globally. To the authors' knowledge, no data exist for TDBPP in indoor home air. TDCIPP concentrations below the detection limit were reported in previous studies (Bergh et al., 2011; Bergh et al., 2010; Marklund et al., 2005), which is consistent with the very low concentrations found in our study and its low vapor pressure (log Pl of − 4.40, EPI Suite V4.1; − 4.44 ABSOLV; Zhang et al., 2016). US, CAN and CZ air concentrations of TCEP and TCIPP were similar or slightly higher than those reported by previous studies from both continents (Bergh et al., 2011; Bergh et al., 2010; Marklund et al., 2005; Staaf and Ostman, 2005). No comparable data for all measured non-halogenated OPEs in indoor home air currently exist, with the exception of TNBP. Concentrations of TNBP in the literature are comparable to those reported here (see Table S3).

3.4. *Correlations among concentrations*

The results of the Spearman correlation analysis between the log transformed concentrations of all the OPEs from this study and of all the brominated flame retardants (BFRs) reported by Venier et al. (2016) are included in the Supporting Information (see Table S4). Correlations

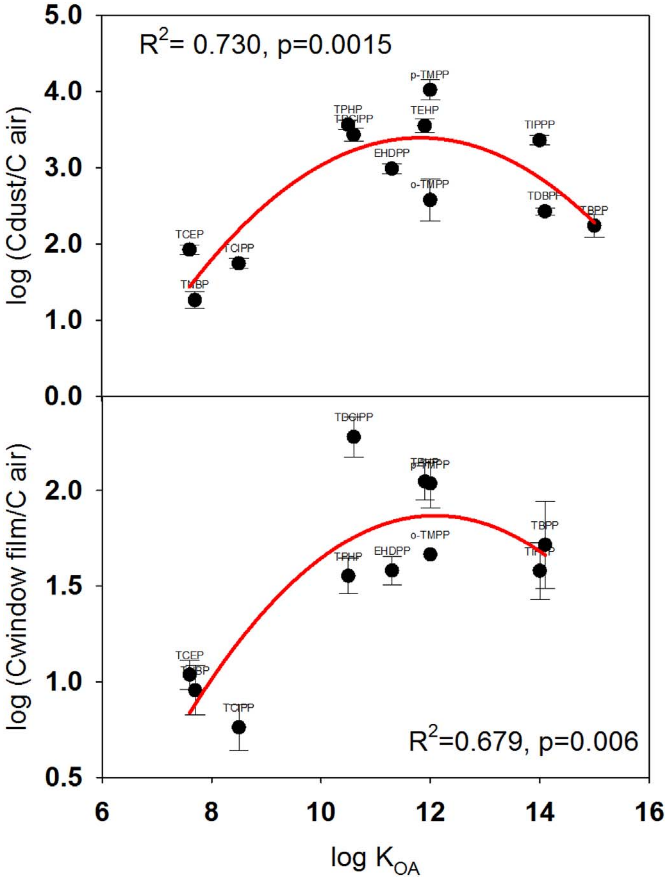

Fig. 1. Dependence of the dust/air partition coefficients and of window film/air partition coefficients on the octanol/air partition coefficient (log K_{OA}). In both graphs each circle represents the average and the bars the standard error. The equations of the regression curves were: $Y = -11.59 + 2.59\ X - 0.1099 \times {}^{2}$ (top) and $Y = -5.60 + 1.24\ X - 0.0511 \times {}^{2}$ (bottom).

were done for each matrix separately and only one room per house was included (i.e. the living room, since not all the houses had samples from the bedroom).

The strongest correlations between OPE concentrations and between OPE and BFR concentrations were measured in dust, which was probably due to higher concentrations and higher detection frequency in dust than in air or window film. Concentrations of most of the OPEs were correlated with one another, with a few exceptions. For example, EHDPP in dust correlated only with the other non-haolgenated OPEs TPHP, TIPPP and TBPP. EHDPP was not correlated with any of the PBDE congeners and with very few of the alternative FRs (namely Pentabromoetheyl benzene (PBEB) and Dechlorane Plus). TCIPP in dust and window film were not correlated with any of the PBDEs, except for BDE209 and a few non PBDEs FRs. The opposite trend was evident in air concentrations. Conversely, TDCIPP was significantly correlated with PBDEs in dust and window film but not in air.

TPHP was significantly correlated with both EHTBB and BEHTBP in dust and window film ($r = 0.32$, $p = 0.02$ for EHTBB in dust; $r = 0.28$, $p = 0.006$ for BEHTBP in dust; $r = 0.70$ $p < 0.0001$ for EHTBB in window film; $r = 0.58$, $p < 0.0001$ for BEHTBP in window film), suggesting that in these samples these three chemicals could share the same source. Differing from our study, Brandsma et al. (2014) did not find a relationship between TPHP and the two brominated components of FM 550 in house and car dust in Europe. They speculated that TPHP had sources other than FM 550 (e.g., electronics). This result seems to suggest that FM 550 was also used in Europe, since the correlation was significant also for CZ samples alone (although only for window film samples). TPHP in dust and window film was also significantly correlated with BDE 209 and DBDPE. TPHP is a by-product in the technical mixture of resorcinol bis and bisphenol A bis, both alternatives to BDE 209, which was also used in conjunction with DBDPE (Ballesteros-Gómez et al., 2016; Brandsma et al., 2014).

A comparison of OPE concentrations with BFR concentrations, using data reported by Venier et al. (2016), revealed that OPEs were generally higher than BFRs in these houses. In particular, the sum of all OPEs was 1 to 2 orders of magnitude higher than the sum of all BFRs for all countries for window film and dust, and 2 to 3 orders of magnitude higher than the sum of all BFRs for all countries for air.

3.5. *Comparison of compound distributions between rooms*

In order to assess variability of FR concentrations between rooms within one home, results for sample pairs collected from both the main bedroom and the living room area were compared for at least nine homes in each country. Differences between rooms were tested for each matrix and country separately using the Mann-Whitney *U* test (for pooled samples) and the Wilcoxon matched pairs test (for room pairs).

The Mann-Whitney *U* test results showed no significant differences ($p > 0.05$) between individual OPEs with the exception of TDCIPP ($p = 0.014$) in CZ dust samples (see Table S4), where concentrations were systematically higher in bedrooms. The Wilcoxon matched pairs test, which is more powerful when evaluating differences within a single home, showed an additional six significant cases: TCIPP, TDBPP and TBPP in US dust samples, TDCIPP and TIPPP in CAN air samples, and TCIPP in CZ air samples ($p < 0.05$, see Table SI). While the concentrations of TCIPP, TDBPP and TBPP in US dust samples and TDCIPP in CAN air samples were higher in bedrooms, concentrations of TIPPP in CAN air samples and TCIPP in CZ air samples were higher in living rooms.

Overall, these results suggest that differences between individual rooms are negligible, similarly to what was previously reported by Venier et al. (2016) for BFR in these samples. The only other study on this topic (Muenhor and Harrad, 2012), reported some room-to-room differences, albeit not statistically significant. They suggested that the dust concentrations were related to the number of specific items (such as electronic products) in each room. To our knowledge, no other study has done a systematic room-to-room comparison on a relatively large number of samples ($n = 30$).

3.6. *Correlations among air, dust and window film*

Next, we considered the relationship among chemical concentrations in air, dust and window film for two reasons. First, this comparison tested the hypothesis for OPEs that under equilibrium conditions, the partitioning between dust and air and between window film and air is expected to be proportional to a chemical's K_{OA}. This relationship has been used for characterizing the sorption of chemicals to organic matter in dust and window films (Butt et al., 2004; Weschler and Nazaroff, 2010). Second, if equilibrium has been achieved, then the concentrations of a chemical can be assessed from measurements in one rather than more media, as a way to reduce sampling effort.

Fig. 1 shows the dependence on log K_{OA} of the partition coefficients between dust and air (top) and between window film and air (bottom). The log K_{OA} values were those from the US EPA's EPI Suite (see Table 1). Fig. 1 shows that the partition coefficients between dust and air, and window film and air increased for compounds with log K_{OA} between 7 and 12, beyond which the coefficients decreased. Similar partitioning behavior was observed for PBDEs and BFRs with a “break point” of $K_{OA} > 14$ (Venier et al., 2016) and for PBDEs and PCBs with a “break point” of $K_{OA} > 11$ (Zhang et al., 2011). Surprisingly, the quadratic curve used to describe this partitioning behavior for OPEs was quite different from that of BFRs. For example, for OPEs, the maximum value of the log dust/air partition coefficient was 3.4 at log K_{OA} 12 whereas the value for BFRs from Venier et al. (2016) was 7.6 at

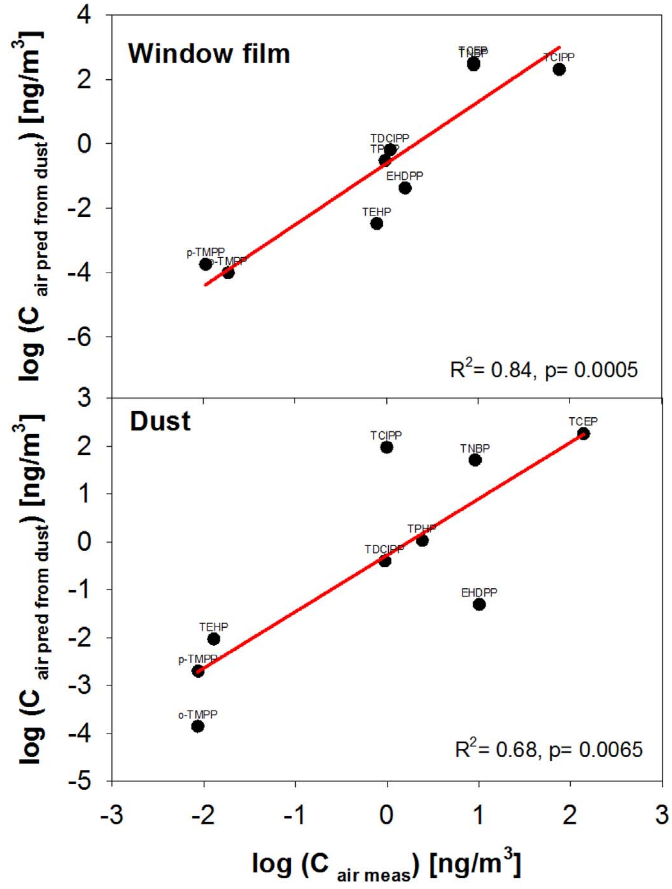

Fig. 2. Dependence of the average measured air concentrations versus the concentrations predicted using window wipes (top) and dust (bottom). Each dot represents the logarithm of the mean concentration. Only compounds with log $K_{OA} > 12$ are included. The equations for the linear regressions are $Y = -0.60 + 1.91 X$ ($r^2 = 0.84$, $p = 0.0005$) for window film and $Y = -0.28 + 1.18 X$ ($r^2 = 0.68$, $p = 0.0065$) for dust.

log K_{OA} 14 (see Fig. 5 in Venier et al., 2016). Taking into account the logarithmic scale, these differences translate to about 4 orders of magnitude difference in the value of the partition coefficient at a 2 order of magnitude offset in log K_{OA}. This large difference in partition coefficients suggests that OPEs have a significantly stronger tendency than BFRs to partition into the air than in dust, assuming that values of K_{OA} are accurate and that OPEs follow expected gas-particle partitioning behavior, as reflected in PUF passive sampler collection. A similar difference was observed also for window film/air, with the logarithms of the partition coefficient reaching a maximum of 1.9 for OPEs at log K_{OA} 12 compared with 5.64 for BFRs at log K_{OA} 14. Again, this result is surprising since one would expect similar partitioning behavior of OPEs and BFRs at the same values of K_{OA}. The difference seen in partitioning behavior is consistent with the lower sampling rates for OPEs than for BFRs for the PUF passive air samplers used (0.82 m^3/day for double bowl and 1.6 m^3/day for single bowl for OPEs versus 1.6 m^3/day and 2.9 m^3/day, respectively, for BFRs).

Based on the analysis described above, we hypothesize that equilibrium between air and dust and between window film and dust was not reached for compounds with log $K_{OA} > 12$. Alternatively, since the total concentration on PUF was used in the calculations, air concentrations could have over-estimated the gas phase by capturing also particle-phase compounds (Abdollahi et al., 2017). OPEs have been found in high percentages in the particle phase (Sühring et al., 2016b). The data presented here do not allow for distinguishing between these two hypotheses.

Using the same approach described in our companion paper (Venier et al., 2016), we estimated air concentrations using dust and window film for compounds with log $K_{OA} < 12$ (see the Supporting Information for details on the calculations). Fig. 2 shows the relationship between the median measured air concentrations and the median gas-phase concentrations predicted from dust (top) and window film (bottom) for compounds that appeared to reach equilibrium. For both window film and dust, the predicted and measured values were correlated ($r^2 = 0.84$ and 0.68, respectively). For both the window film and dust, the slopes of the linear regression were close to unity (1.91 ± 0.31 and 1.18 ± 0.31, respectively). The negative values of the intercepts (-0.60 for window film and -0.28 for dust), albeit not statistically significant, indicated that, in general, the air concentrations predicted from either window film or dust are lower than the measured air concentrations ($10^{-0.60} = 0.25$ and $10^{-0.28} = 0.52$). This discrepancy between measured and predicted air concentration might be due to assuming too large value for the organic matter fraction in both media or for the thickness of the film layer. However, based on these results, we recommend that for OPEs with log $K_{OA} < 12$ that either air, dust or window films can be sampled, but not all are necessary.

The partitioning analysis suggests that several OPEs compounds (i.e. those with log $K_{OA} > 12$) did not reach equilibrium between air, dust and window film and, differently from BFRs, the three matrices used here are not equivalent. The partitioning behavior suggests that OPEs tend to accumulate more in air than dust or window film compared to PBDEs and BFRs.

3.7. Limitations

OPEs are challenging compounds to measure, as reflected in the large variability in the measurements of a standard reference material (see Table S2 and discussion in the Supporting Information). While different analytical methods between IU and RECETOX could have contributed to differences or lack thereof, the regional trends are in agreement with the usage history of FRs (US > CAN > CZ). If differences due to analytical methods would be at play here, they would not affect the partitioning and prediction modeling nor the correlations or comparisons with BFRs. It should be noted that, although samples in the three countries were collected in a relatively short time window (May–August), we acknowledge that seasonal variability might play a role in the differences we have observed. Hoffman et al. (2017) reported higher exposure concentrations and hence exposures for TDCIPP and TPHP in the summer than in winter.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.envint.2017.05.020.

References

Abdollahi, A., Eng, A., Jantunen, L.M., Ahrens, L., Shoeib, M., Parnis, J.M., Harner, T., 2017. Characterization of polyurethane foam (PUF) and sorbent impregnated PUF (SIP) disk passive air samplers for measuring organophosphate flame retardants.

Chemosphere 167, 212–219.

Andresen, J.A., Grundmann, A., Bester, K., 2004. Organophosphorus flame retardants and plasticisers in surface waters. Sci. Total Environ. 332, 155–166.

Antweiler, R.C., 2015. Evaluation of statistical treatments of left-censored environmental data using coincident uncensored data sets: II. Group Comparisons. Environ. Sci. Technol. 49, 13439–13446.

Ballesteros-Gómez, A., Aragón, Á., Van den Eede, N., de Boer, J., Covaci, A., 2016. Impurities of resorcinol Bis(diphenyl phosphate) in plastics and dust collected on electric/electronic material. Environ. Sci. Technol. 50, 1934–1940.

Behl, M., Hsieh, J.-H., Shafer, T.J., Mundy, W.R., Rice, J.R., Boyd, W.A., Freedman, J.H., Hunter III, E.S., Jarema, K.A., Padilla, S., Tice, R.R., 2015. Use of alternative assays to identify and prioritize organophosphorus flame retardants for potential developmental and neurotoxicity. Neurotoxicol. Teratol. 52, 181–193.

Bergh, C., Torgrip, R., Östman, C., 2010. Simultaneous selective detection of organophosphate and phthalate esters using gas chromatography with positive ion chemical ionization tandem mass spectrometry and its application to indoor air and dust. Rapid Commun. Mass Spectrom. 24, 2859–2867.

Bergh, C., Torgrip, R., Emenius, G., Östman, C., 2011. Organophosphate and phthalate esters in air and settled dust – a multi-location indoor study. Indoor Air 21, 67–76.

Brandsma, S.H., de Boer, J., van Velzen, M.J.M., Leonards, P.E.G., 2014. Organophosphorus flame retardants (PFRs) and plasticizers in house and car dust and the influence of electronic equipment. Chemosphere 116, 3–9.

Brommer, S., Harrad, S., Van den Eede, N., Covaci, A., 2012. Concentrations of organophosphate esters and brominated flame retardants in German indoor dust samples. J. Environ. Monit. 14, 2482–2487.

Brooke, D.N., Crookes, M.J., Quarterman, P., Burns, J., 2009. Environmental risk evaluation report: 2-Ethylhexyl diphenyl phosphate (CAS no. 1241-94-7) - SCHO0809BQTY-E-P. UK Environment Agency, Bristol, UK.

Butt, C.M., Diamond, M.L., Truong, J., Ikonomou, M.G., ter Schure, A.F.H., 2004. Spatial distribution of Polybrominated diphenyl ethers in southern Ontario as measured in indoor and outdoor window organic films. Environ. Sci. Technol. 38, 724–731.

Cao, S.X., Zeng, X.Y., Song, H., Li, H.R., Yu, Z.Q., Sheng, G.Y., Fu, J.M., 2012. Levels and distributions of organophosphate flame retardants and plasticizers in sediment from Taihu Lake, China. Environ. Toxicol. Chem. 31, 1478–1484.

Cooper, E.M., Kroeger, G., Davis, K., Clark, C.R., Ferguson, P.L., Stapleton, H.M., 2016. Results from screening polyurethane foam based consumer products for flame retardant chemicals: assessing impacts on the change in the furniture flammability standards. Environ. Sci. Technol. 50, 10653–10660.

Dodson, R.E., Perovich, L.J., Covaci, A., Van den Eede, N., Ionas, A.C., Dirtu, A.C., Brody, J.G., Rudel, R.A., 2012. After the PBDE phase-out: a broad suite of flame retardants in repeat house dust samples from California. Environ. Sci. Technol. 46, 13056–13066.

Environment and Climate Change Canada, 2016. List of Toxic Substances Managed Under CEPA (Schedule 1).

European Chemicals Agency, 2015. List of Substances Included in Annex XIV of REACH ("Authorisation List").

García, M., Rodríguez, I., Cela, R., 2007. Microwave-assisted extraction of organophosphate flame retardants and plasticizers from indoor dust samples. J. Chromatogr. A 1152, 280–286.

Gazette, Canada, 2014. Regulations amending schedule 2 to the Canada consumer product safety act (TCEP). http://www.gazette.gc.ca/rp-pr/p2/2014/2014-04-23/html/sor-dors79-eng.php (accessed on 5/12/2017).

Gold, M., Blum, A., Ames, B., 1978. Another flame retardant, tris-(1,3-dichloro-2-propyl)-phosphate, and its expected metabolites are mutagens. Science 200, 785–787.

Greaves, A.K., Letcher, R.J., Chen, D., McGoldrick, D.J., Gauthier, L.T., Backus, S.M., 2016. Retrospective analysis of organophosphate flame retardants in herring gull eggs and relation to the aquatic food web in the Laurentian Great Lakes of North America. Environ. Res. 150, 255–263.

Green, N.S., Bakke, M., Brevik, T., Dye, E.M., Herzke, C., Huber, D., Plosz, S., Remberger, B., Schoyen, M., Uggerud, H.T., Vogelsang, C., 2008. Screening of selected metals and new organic contaminants 2007. Norwegian Pollution Control Agency.

Harrad, S., Ibarra, C., Diamond, M., Melymuk, L., Robson, M., Douwes, J., Roosens, L., Dirtu, A.C., Covaci, A., 2008. Polybrominated diphenyl ethers in domestic indoor dust from Canada, New Zealand, United Kingdom and United States. Environ. Int. 34, 232–238.

Hendriks, H.S., Westerink, R.H.S., 2015. Neurotoxicity and risk assessment of brominated and alternative flame retardants. Neurotoxicol. Teratol. 52 (Part B), 248–269.

Hoffman, K., Butt, C.M., Webster, T.F., Preston, E.V., Hammel, S.C., Makey, C., Lorenzo, A.M., Cooper, E.M., Carignan, C., Meeker, J.D., Hauser, R., Soubry, A., Murphy, S.K., Price, T.M., Hoyo, C., Mendelsohn, E., Congleton, J., Daniels, J.L., Stapleton, H.M., 2017. Temporal trends in exposure to organophosphate flame retardants in the United States. Environ. Sci. Technol. Lett. 4 (3), 112–118.

Kajiwara, N., Noma, Y., Takigami, H., 2011. Brominated and organophosphate flame retardants in selected consumer products on the Japanese market in 2008. J. Hazard. Mater. 192, 1250–1259.

Karásková, P., Venier, M., Melymuk, L., Bečanová, J., Vojta, Š., Prokeš, R., Diamond, M.L., Klánová, J., 2016. Perfluorinated alkyl substances (PFASs) in household dust in Central Europe and North America. Environ. Int. 94, 315–324.

Leisewitz, A., Kruse, H., Schramm, E., 2001. Substituting Environmentally Relevant Flame Retardants: Assessment Fundamentals (Report No. UBA-FB 000171/1). Federal Environmental Agency (Umweltbundesamt).

Marklund, A., Andersson, B., Haglund, P., 2003. Screening of organophosphorus compounds and their distribution in various indoor environments. Chemosphere 53, 1137–1146.

Marklund, A., Andersson, B., Haglund, P., 2005. Traffic as a source of organophosphorus flame retardants and plasticizers in snow. Environ. Sci. Technol. 39, 3555–3562.

Mendelsohn, E., Hagopian, A., Hoffman, K., Butt, C.M., Lorenzo, A., Congleton, J., Webster, T.F., Stapleton, H.M., 2016. Nail polish as a source of exposure to triphenyl phosphate. Environ. Int. 86, 45–51.

Muenhor, D., Harrad, S., 2012. Within-room and within-building temporal and spatial variations in concentrations of polybrominated diphenyl ethers (PBDEs) in indoor dust. Environ. Int. 47, 23–27.

Rauert, C., Harrad, S., 2015. Mass transfer of PBDEs from plastic TV casing to indoor dust via three migration pathways — a test chamber investigation. Sci. Total Environ. 536, 568–574.

Regnery, J., Puttmann, W., Merz, C., Berthold, G., 2011. Occurrence and distribution of organophosphorus flame retardants and plasticizers in anthropogenically affected groundwater. J. Environ. Monit. 13, 347–354.

Salamova, A., Peverly, A.A., Venier, M., Hites, R.A., 2016. Spatial and temporal trends of particle phase organophosphate ester concentrations in the atmosphere of the Great Lakes. Environ. Sci. Technol.

Schreder, E.D., Uding, N., La Guardia, M.J., 2016. Inhalation a significant exposure route for chlorinated organophosphate flame retardants. Chemosphere 150, 499–504.

Schweizer, C., Edwards, R.D., Bayer-Oglesby, L., Gauderman, W.J., Ilacqua, V., Jantunen, M.J., Lai, H.K., Nieuwenhuijsen, M., Kunzli, N., 2007. Indoor time-microenvironment-activity patterns in seven regions of Europe. J. Expo. Sci. Environ. Epidemiol. 17, 170–181.

Staaf, T., Ostman, C., 2005. Indoor air sampling of organophosphate triesters using solid phase extraction (SPE) adsorbents. J. Environ. Monit. 7, 344–348.

Stapleton, H.M., Klosterhaus, S., Eagle, S., Fuh, J., Meeker, J.D., Blum, A., Webster, T.F., 2009. Detection of organophosphate flame retardants in furniture foam and US house dust. Environ. Sci. Technol. 43, 7490–7495.

Sühring, R., Diamond, M.L., Scheringer, M., Wong, F., Pućko, M., Stern, G., Burt, A., Hung, H., Fellin, P., Li, H., Jantunen, L.M., 2016a. Organophosphate esters in Canadian Arctic air: occurrence levels and trends. Environ. Sci. Technol. 50, 7409–7415.

Sühring, R., Wolschke, H., Diamond, M.L., Jantunen, L.M., Scheringer, M., 2016b. Distribution of organophosphate esters between the gas and particle phase–model predictions vs measured data. Environmental Science & Technology. 50, 6644–6651.

Sundkvist, A.M., Olofsson, U., Haglund, P., 2010. Organophosphorus flame retardants and plasticizers in marine and fresh water biota and in human milk. J. Environ. Monit. 12, 943–951.

Toy Safety Directive, 2009. Directive 2009/48/EC of the European Parliament and of the Council of 18 June 2009 on the safety of toys. /48/EC. http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2009:170:0001:0037:en:PDF (accessed on 5/12/2017).

Van den Eede, N., Dirtu, A.C., Neels, H., Covaci, A., 2011. Analytical developments and preliminary assessment of human exposure to organophosphate flame retardants from indoor dust. Environ. Int. 37, 454–461.

van der Veen, I., de Boer, J., 2012. Phosphorus flame retardants: properties, production, environmental occurrence, toxicity and analysis. Chemosphere 88, 1119–1153.

Venier, M., Dove, A., Romanak, K., Backus, S., Hites, R., 2014. Flame retardants and legacy Chemicals in Great Lakes' water. Environ. Sci. Technol. 48, 9563–9572.

Venier, M., Audy, O., Vojta, Š., Bečanová, J., Romanak, K., Melymuk, L., Krátká, M., Kukučka, P., Okeme, J., Saini, A., 2016. Brominated flame retardants in the indoor environment—comparative study of indoor contamination from three countries. Environ. Int. 94, 150–160.

Wei, G.L., Li, D.Q., Zhuo, M.N., Liao, Y.S., Xie, Z.Y., Guo, T.L., Li, J.J., Zhang, S.Y., Liang, Z.Q., 2015. Organophosphorus flame retardants and plasticizers: sources, occurrence, toxicity and human exposure. Environ. Pollut. 196, 29–46.

Weschler, C.J., Nazaroff, W.W., 2010. SVOC partitioning between the gas phase and settled dust indoors. Atmos. Environ. 44, 3609–3620.

Zhang, X., Diamond, M.L., Robson, M., Harrad, S., 2011. Sources, emissions, and fate of Polybrominated diphenyl ethers and polychlorinated biphenyls indoors in Toronto, Canada. Environ. Sci. Technol. 45, 3268–3274.

Zhang, X., Sühring, R., Serodio, D., Bonnell, M., Sundin, N., Diamond, M.L., 2016. Novel flame retardants: estimating the physical-chemical properties and environmental fate of 94 halogenated and organophosphate PBDE replacements. Chemosphere 144, 2401–2407.

APPENDIX 13

Melymuk, Lisa, Pernilla Bohlin-Nizzetto, Petr Kukučka, Šimon Vojta, Jiří Kalina, Pavel Cupr, Jana Klánová, Pavel Čupr, and Jana Klánová. 2016. "Seasonality and Indoor/Outdoor Relationships of Flame Retardants and PCBs in Residential Air." *Environmental Pollution* 218: 392-401. https://doi.org/10.1016/j.envpol.2016.07.018

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Seasonality and indoor/outdoor relationships of flame retardants and PCBs in residential air☆

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ABSTRACT

This study is a systematic assessment of different houses and apartments, their ages and renovation status, indoors and outdoors, and in summer vs. winter, with a goal of bringing some insight into the major sources of semivolatile organic compounds (SVOCs) and their variability. Indoor and outdoor air concentrations of polychlorinated biphenyls (PCBs), polybrominated diphenyl ethers (PBDEs) and novel flame retardants (NFRs) were determined at 17–20 homes in Czech Republic in winter and summer.

Indoor concentrations were consistently higher than outdoor concentrations for all compounds; indoor/outdoor ratios ranged from 2–20, with larger differences for the current use NFRs than for legacy PCBs. Seasonal trends differed according to the use status of the compounds: the PCBs had higher summer concentrations both indoors and outdoors, suggesting volatilization as a source of PCBs to air. PBDEs had no seasonal trends indoors, but higher summer concentrations outdoors. Several NFRs (TBX, PBT, PBEB) had higher indoor concentrations in winter relative to summer. The seasonal trends in the flame retardants suggest differences in air exchange rates due to lower building ventilation in winter could be driving the concentration differences.

Weak relationships were found with building age for PCBs, with higher concentrations indoors in buildings built before 1984, and with the number of electronics for PBDEs, with higher concentrations in rooms with three or more electronic items. Indoor environments are the primary contributor to human inhalation exposure to these SVOCs, due to the high percentage of time spent indoors (>90%) combined with the higher indoors levels for all the studied compounds. Exposure via the indoor environment contributed ~96% of the total chronic daily intake via inhalation in summer and ~98% in winter.

1. Introduction

The quality of air in a non-industrial indoor environment (e.g., a home, school, office, public building) is affected by a wide range of factors, including pollutants from sources inside the buildings as well as from outdoor air. Knowledge on the fate and distribution of indoor air pollutants is crucial since 80–90% of our total life time is spent in different indoor environments. Moreover, people who are most susceptible to the adverse effects of air pollutants (e.g. infants, elderly, etc.) spend even more time indoors, and thus inhalation of indoor air can significantly contribute to the total human exposure to pollutants. Until recently, the focus on indoor air quality has been on volatile organic compounds (VOCs), inorganics, radon and particulate matter, and the World Health Organization (WHO) Regional Office for Europe has published indoor air quality guidelines values for these pollutants (WHO European Centre for Environment and Health, 2010).

A more recent focus is the semivolatile organic compounds (SVOCs), which also have a plethora of indoor sources. Many products containing SVOCs are used predominantly in indoor environments and many indoor sources remain active despite bans and regulations (e.g., Stockholm Convention). Indoor environments also contain hydrophobic, sorptive materials that may act as reservoirs and thereby sources of secondary emissions, even after the removal of primary sources. This may be enhanced by increased

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persistence of these compounds indoors due to limited exposure to direct sunlight, OH radicals and bacterial activity. In addition, physical insulation and reduced ventilation in many indoor environments may also enhance indoor concentrations. SVOCs are therefore prone to accumulate and persist in indoor environments, contributing to long term exposure. Indoor concentrations of many SVOCs are several times higher than outdoor concentrations (Menichini et al., 2007; Rudel et al., 2010; Zhang et al., 2011), emphasizing the importance of the indoor environment as an exposure route for these compounds. Furthermore, as the composition of chemicals on the market changes due to regulation, e.g., a shift from polybrominated diphenyl ethers (PBDEs) to replacement “novel” halogenated flame retardants (NFRs), the chemicals to which we are exposed indoors also shifts, and thus may lead to differences between older and newer dwellings.

With this study we analyse: (i) the relationship between indoor and outdoor air concentrations, (ii) seasonal variability, and (iii) the relationship between indoor air concentrations and house characteristics (i.e. year of construction, renovation status, and type of house) and house contents (e.g., floor type, furniture, electronics). Moreover, indoor data from Czech Republic increases the knowledge on the global spatial distribution of indoor air measurements in the less frequently studied area of Central and Eastern Europe. We focus on SVOCs with links to building materials and/or consumer products that cover a range of legacy and current sources (Table 1): polychlorinated biphenyls (PCBs), PBDEs and NFRs.

2. Materials and methods

2.1. Sampling site

Air sampling was conducted in residential areas around Brno, Czech Republic (metropolitan area with a population of 600,000). The sampling sites were private dwellings located both in the urban area and nearby semi-rural areas (maximum 40 km from city centre). The residents were informed about the study and invited to participate on a voluntary basis.

2.2. Experimental design

Twenty private houses were selected to cover a range of specific characteristics: (i) house age, (ii) house status, (i.e. original construction or renovated), and (iii) type of house (i.e. apartment/flat in a multi-unit building, row house, or detached house). Polyurethane foam passive air samplers (PUF-PAS) were deployed concurrently inside and outside the homes for 28 days. The indoor samplers were deployed in the main living area of the house at ~2 m height and the outdoor samplers on the adjacent balcony or the garden of the house, with a distance of 0.5–10 m from the building wall. In three homes, samplers were deployed both in living rooms and in bedrooms to address spatial variability within the residence. Two sampling campaigns were carried out to address seasonal variability: one in summer (July–August 2010, avg. outdoor air temperature 19 °C) and one in winter (February–March 2011, avg. outdoor air temperature 0 °C).

A questionnaire, completed by the residents at the end of the sampling, provided additional information on type and age of building materials (floors, walls, windows, insulation, etc.), type and age of products in the home (furniture, carpets, electronics, heating systems, water boiler, fireplaces, etc.), ventilation systems and frequency of aeration (how often windows are kept open), and living habits (cleaning, cooking, smoking, etc.). Details of the buildings and questionnaires are presented in Supplementary Information (SI).

2.3. Passive air sampling

The PUF-PAS disks were 15 cm diameter, 1.5 cm thickness, 424 cm^2 total surface area, 0.030 g/cm^3 density (type T-3037, Molitan a.s., Czech Republic), and were deployed in protective chambers consisting of two stainless steel bowls (upper 30 cm diameter and lower 24 cm diameter). Methods for PUF-PAS preparation and deployment are described by Bohlin et al. (2014a,b). Two generic PUF-PAS sampling rates were used to convert sampled masses to air concentrations: 1.4 m^3/day for indoor samplers (Bohlin et al., 2014b) and 3.5 m^3/day for outdoor samplers (Bohlin et al., 2014a).

Field validations of PUF-PAS have shown good performance for gas-phase compounds such as PCBs both indoors and outdoors, yet they have been shown to be a reliable sampling tool for only a limited number of PBDE congeners (Bohlin et al., 2014a,b; Hazrati and Harrad, 2007). Consistent results can be obtained for BDE 28, 47, 66, 85, 99, and 100, but the heavier, largely particle-bound BDEs may not have a consistent sampling rate with the PUF-PAS,

Table 1
List of compounds.

Compound class	Compound/congener	Primary/possible uses in Czech Republic
PCBs	CB-28, 52, 101, 118, 138, 153, 180	Electrical and industrial equipment, paints, pre-1984 (Holoubek et al., 2006)
PBDEs	BDE-28, 47, 66, 85, 99, 100, 153, 154, 183, and 209	Flame retardant (FR) in polyurethane foams, electrical and electronic equipment, textiles, carpets, draperies, plastic furniture, roofing, insulation, piping/ducting/hoses, cables (European Chemicals Agency, 2014; Prevedouros et al., 2004)
NFRs	2,3-dibromopropyl 2,4,6-tribromophenyl ether (TBP-DBPE)	FR in polypropylene (CECBP, 2009), manufactured in Germany, production ceased in 1980s (Ma et al., 2012)
	2-bromoallyl 2,4,6-tribromophenyl ether (TBP-BAE)	Transformation product of TBP-DBPE (Ma et al., 2012)
	tetrabromo-*p*-xylene (TBX)	Additive FR in polymers and resins (Venier et al., 2012)
	pentabromoethylbenzene (PBEB)	FR in polyurethane foam, textiles, adhesives, wire and cable coatings, polyester resins, thermoset polyester resins (Harju et al., 2009; Salamova and Hites, 2011)
	pentabromotoluene (PBT)	FR in bulk moulded polyesters (Harju et al., 2009)
	hexabromobenzene (HBB)	FR in polymers, wood, textiles, electronics (Salamova and Hites, 2011; Schlabach et al., 2011)
	1,2-bis(2,4,6-tribromophenoxy)ethane (BTBPE)	FR in thermoplastics, replacement for octaBDE (Schlabach et al., 2011)
	Hexachlorocyclopentadienyl-dibromocyclooctane (DBHCTD)	FR in styrenic polymers (Covaci et al., 2011; Riddell et al., 2008)
	syn- and anti-Dechlorane Plus (s-DDC-CO, a-DDC-CO)	FR in electrical wires, cables and polymer products (Schlabach et al., 2011)
	1,2-dibromo-4-(1,2-dibromoethyl) cyclohexane (DBE-DBCH)	FR in polystyrene, polyethylene, building insulation (Schlabach et al., 2011)

particularly in indoor environments with limited air flow, and thus may have higher uncertainty, notably for BDE-209. Thus, when discussing seasonal and site-related trends in PBDEs, we exclude any detected levels of BDE-209, although we note that BDE-209 was detected at levels above the limit of detection (LOD) in 16% of samples (Table S3).

2.4. Analysis

Laboratory and analytical methods are given in full in the SI and have been previously published (Bohlin et al., 2014b). The target compounds are listed in Table 1. Briefly, PUF-PAS were extracted with dichloromethane (DCM) using automated warm Soxhlet extraction, cleaned-up via a H_2SO_4 modified silica column eluted with 1:1 DCM:*n*-hexane. Samples were analysed for PCBs using a gas chromatograph (GC)-mass spectrometry (MS) system (Agilent 6890N GC coupled to Micromass Quattro Micro GC tandem quadrupole MS) with a 60 m DB5-MS column. Samples were analysed for PBDEs and NFRs using a GC-high resolution MS system (Agilent 7890A GC coupled to Micromass AutoSpec Premier sector MS) with a 15 m DB5 column.

2.5. QA/QC

Two field blanks consisting of pre-cleaned PUFs were collected for each summer and winter campaign (4 total). These field blanks, along with laboratory and solvent blanks, were analysed as per the samples. The LOD was calculated as the average field blanks plus twice the standard deviation of the blanks. The samples above the LOD were blank subtracted based on the average of the field blanks. For compounds that were not detected in the blanks, instrumental LODs were used.

Further information on QA/QC is given in Table S4.

2.6. Statistical analyses

Statistical Analyses were performed in GraphPad Prism (version 5.00), Microsoft Excel (Microsoft Office 2010), R studio (version 0.99.491 based on R version 3.2.3) and Maplesoft Maple (version 16). Concentrations were not normally distributed, thus non-parametric tests were used (e.g., Mann-Whitney *U*-test) at $\alpha = 0.05$. For the statistical analyses, values < LOD were substituted by 0.5*LOD.

3. Results and discussion

3.1. PCBs

3.1.1. PCB air concentrations

PCBs were detected in all but one indoor air sample. Concentrations of Σ_7PCB (sum of PCBs 28, 52, 101, 118, 138, 153, and 180) are presented in Table 2. The variability of the air concentrations among different indoor sites was relatively small for PCBs (3–10 fold). Mean indoor air concentrations of Σ_7PCB were 89 pg/m^3 in summer (median: 82 pg/m^3) and 61 pg/m^3 in winter (median: 61 pg/m^3). These levels are 4–10 × lower than PCB air concentrations in indoor environments in Canada (Zhang et al., 2011), Denmark (Frederiksen et al., 2012), France (Alliot et al., 2014), UK, and Sweden (Bohlin et al., 2008), while comparable to indoor levels in Mexico (Bohlin et al., 2008) and offices in Czech Republic (Bohlin et al., 2014b) (Fig. 1).

Mean outdoor air concentrations of Σ_7PCB were 38 pg/m^3 in summer (median: 34 pg/m^3) and 17 pg/m^3 in winter (median: 19 pg/m^3). The winter concentrations were consistent with outdoor air concentrations from urban and rural sites in the Czech Republic (Bohlin et al., 2014a; Holoubek et al., 2007), Ontario, Canada (Melymuk et al., 2012), Sweden, UK and Mexico (Bohlin et al., 2008).

The low indoor air concentrations of PCBs in this study relative to those measured in regions with known use of PCBs in building materials (e.g., in building sealants, Frederiksen et al., 2012; Kohler et al., 2005) supports the hypothesis of limited indoor primary sources of PCBs in Czech Republic. Moreover, while evidence from Western Europe and North America suggests PCB-containing building sealants were typically used in larger concrete buildings (Klosterhaus et al., 2014; Robson et al., 2010), air concentrations here do not suggest any difference between the larger prefabricated concrete residential buildings and single family homes. However, we caution that with a small sample size, further work is necessary to confirm this result. In agreement with this, no reports exist of the use of PCBs in non-military building applications in Czech Republic. However, although the indoor air concentrations of PCBs in this study are lower than many other countries, the concentrations are higher than the indoor air concentrations of PBDEs and NFRs, which have known indoor use. Moreover, the measured outdoor air concentrations of PCBs are similar to those in urban areas in other countries. We hypothesize while there may be fewer direct sources to residential indoor environments in this region, some do exist, and there also exists a similar level of sources to outdoors compared to other places, e.g., release of PCBs from electrical and industrial equipment, of which there are documented existing stocks (Holoubek et al., 2006), and volatilization from a range of secondary sources. However, it is apparent that the indoor contamination from PCBs in Czech Republic is not fully characterized, and further work to identify possible indoor (undocumented) sources is needed, especially in non-residential buildings.

3.1.2. PCB congener profiles

In individual houses, the congener profiles did not vary significantly between summer and winter (Fig. S1), however the congener profile differed between houses, ranging from those dominated by tri-tetra PCBs (e.g., houses 11, 16 and 20) and those dominated by hepta-hexa PCBs (e.g., houses 3, 4, and 13). The congener profiles were more homogeneous outdoors (Fig. S1), dominated by PCBs 28 and 153.

This range of profiles indoors is similar to the Delor technical mixtures, the PCB technical mixtures produced in the former Czechoslovakia (Holoubek et al., 2006). Thus, linear optimization was used to estimate which Delor technical mixtures could be the primary contributors to the individual indoor air samples by predicting the composition of the indoor air samples based on the known composition of the four Delor mixtures (Taniyasu et al., 2003), adjusted for vapour pressure, and thereby identifying the fraction contribution of each Delor to the indoor air samples (details in SI).

Delors 103 and 106 best approximated the composition of congeners in air in all in differing combinations (Fig. S2). Houses 3, 4, 13, 14, and 17 had estimated contributions of >80% from Delor 106, while Houses 5–11, and 15 had contributions of 20–40% Delor 103, as well as significant contributions from Delors 105 and 106. Only two homes had estimated contributions from Delor 104 (~10%) (mainly due to relatively high level of PCB 52 relative to PCB 28). Outdoor air profiles were most similar to Delor 103 and 106 profiles (Fig. S2).

Delor 103 was used as a dielectric fluid in capacitors and as an insulator, Delor 105 in transformer fluid and Delor 106 primarily as a paint additive (Table S6). Thus, the dominance of Delors 103 and 106 suggests paints, adhesives, sealants and capacitors as the primary sources, which is plausible for residential indoor environments. Delor 104 had minimal production and use as a capacitor

Table 2
Air concentrations (pg/m^3) estimated from PUF-PAS in the studied sites ($n = 17$ in summer and $n = 20$ in winter). DBE-DBCH is the sum of the α, β, γ, and δ isomers, and DDC-CO is the sum of the syn and anti isomers.

Compound	Parameter	Indoor summer	Indoor winter	Outdoor summer	Outdoor winter
Σ_7PCBs	Mean ± SD	89 ± 33	61 ± 23	38 ± 13	17 ± 5.9
	Median	82	61	34	19
	Range	44–160	<LOD-110	26–82	7.5–29
Σ_9PBDEs	Mean ± SD	9.1 ± 10	8.5 ± 11	1.8 ± 2.2	0.49 ± 0.21
	Median	4.5	5.8	1.3	0.42
	Range	1.4–41	0.79–49	0.50–9.7	<LOD-1.1
TBX	Mean ± SD	0.076 ± 0.094	0.16 ± 0.11	0.023 ± 0.009	0.028 ± 0.013
	Median	0.046	0.10	0.023	0.029
	Range	0.020–0.41	0.051–0.36	<LOD-0.036	<LOD-0.055
PBT	Mean ± SD	2.1 ± 3.1	3.5 ± 3.1	0.21 ± 0.10	0.11 ± 0.058
	Median	1.4	3.0	0.20	0.11
	Range	0.22–14	0.32–13	0.065–0.39	0.045–0.21
PBEB	Mean ± SD	0.064 ± 0.083	0.10 ± 0.079	0.026 ± 0.070	0.013 ± 0.010
	Median	0.041	0.075	0.0082	0.0092
	Range	0.007–0.35	0.031–0.37	0.003–0.30	0.004–0.035
HBB	Mean ± SD	1.4 ± 1.2	1.7 ± 1.7	0.13 ± 0.072	0.082 ± 0.051
	Median	0.94	1.1	0.14	<LOD
	Range	0.17–4.4	0.33–5.7	0.041–0.33	<LOD-0.21
TBP-DBPE	Mean ± SD	1.5 ± 1.2	1.3 ± 1.2	0.58 ± 0.50	0.25 ± 0.12
	Median	1.4	<LOD	0.47	<LOD
	Range	<LOD-5.7	<LOD-4.8	<LOD-2.2	<LOD-0.75
BTBPE	Mean ± SD	0.29 ± 0.39	0.22 ± 0.37	0.062 ± 0.099	0.065 ± 0.13
	Median	0.23	<LOD	<LOD	<LOD
	Range	<LOD-1.6	<LOD-1.2	<LOD-0.27	<LOD-0.49
DBE-DBCH	Mean ± SD	150 ± 450	65 ± 120	96 ± 360	2.2 ± 2.6
	Median	32	20	7.8	0.85
	Range	6.5–1900	7.7–530	1.7–1500	0.85–12
ΣDDC-CO	Mean ± SD	1800 ± 6900	66 ± 110	53 ± 120	30 ± 52
	Median	54	<LOD	15	6.0
	Range	13–29,000	<LOD-440	5.1–490	6.0–170

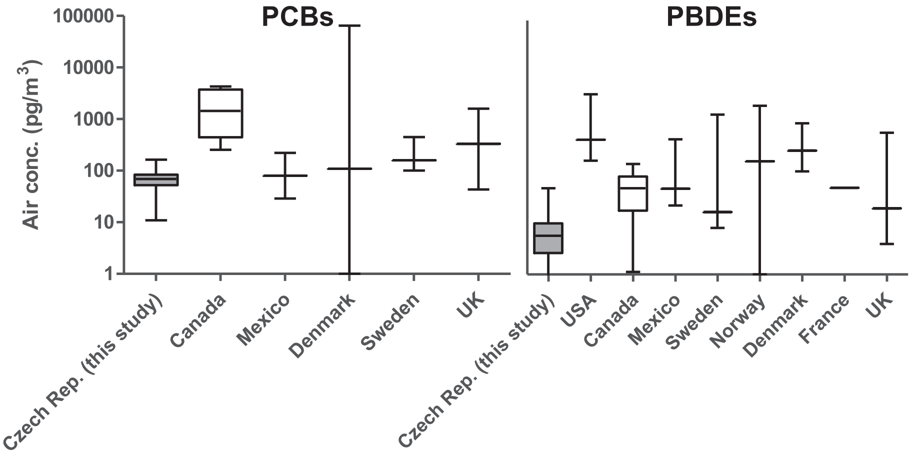

Fig. 1. Comparison between medians and range of indoor air concentrations of PCBs and PBDEs measured in this study with previously published concentrations from Canada (Zhang et al., 2011), USA (Allen et al., 2007), Mexico (Bohlin et al., 2008), Sweden (Bjorklund et al., 2012; Bohlin et al., 2008), Norway (Cequier et al., 2014), Denmark (Frederiksen et al., 2012; Vorkamp et al., 2011), France (Alliot et al., 2014) and UK (Bohlin et al., 2008; Harrad et al., 2006).

fluid (<0.2% of total PCBs produced in former Czechoslovakia, Holoubek et al., 2006), thus its limited contribution as a source is expected. No relationships could be found between the Delor compositions and building parameters such as age, dwelling type, renovation status or location. This may be due to the limitation of using only seven congeners to estimate the Delor compositions, or because the factors contributing to the differences in sources between the buildings were not captured by the building survey and questionnaire data. This emphasizes that the sources of PCBs to indoor environments in the Czech Republic are still not well understood, despite this being the highest concentration SVOC of the targeted compounds in these 20 homes, and further work is needed to determine the plausibility of open indoor sources (paints, sealants) in residential environments.

3.1.3. Indoor-outdoor PCB relationships

Indoor air concentrations of PCBs (both individual congeners and sum) were significantly higher than outdoor PCB concentrations in both seasons (Mann-Whitney *U* test, $p < 0.05$), except in one home. The higher indoor concentrations resulted in positive indoor to outdoor ratios (I/O) in both summer and winter (Table 3). Winter I/O ratios were significantly higher than summer I/O ratios (Mann-Whitney *U* test, $p < 0.0001$), reflecting temperature differences and the influence of volatilization, and also supporting the

Table 3
Indoor/outdoor (I/O) and summer/winter (S/W) ratios. Ratios were calculated from sites where the compound in question was >LOD in both of the samples in question.

Compound	Parameter	I/O in summer	I/O in winter	S/W indoors	S/W outdoors
Σ_7PCBs	Median	2.4	3.9	1.3	2.1
	Range	1.0–3.9	0.41–8.5	0.77–7.8	1.2–3.1
Σ_9PBDEs	Median	4.9	13	1.4	2.4
	Range	1.7–23	0.23–98	0.32–2.7	0.35–8.8
TBX	Median	2.9	4.6	0.58	0.7
	Range	0.55–18	1.4–43	0.19–1.3	0.23–1.6
PBT	Median	6.7	20	0.60	1.3
	Range	1.1–74	2.4–81	0.076–13	0.55–4.1
PBEB	Median	6.7	5.0	0.51	0.55
	Range	0.078–46	1.1–24	0.099–2.3	0.21–10
HBB	Median	6.7	7.3	0.71	0.91
	Range	1.8–49	3.5–9.4	0.30–10	0.65–1.2
DBE-DBCH	Median	3.1	9.5	1.4	5.2
	Range	1.1–29	2.6–46	0.31–5.8	3.4–128
ΣDDC-CO	Median	5.8	6.1	1.7	0.84
	Range	1.9–58	0.76–48	0.072–65	0.096–53

hypothesis that ventilation differences in indoor environments (e.g., lower air exchange in winter) influence SVOC concentrations indoors (MacIntosh et al., 2012; Zhang et al., 2009). These ratios fall in the lower end of the range of I/O = 2–50 previously reported for PCBs in Europe and North America (Bohlin et al., 2008; Menichini et al., 2007; Rudel and Perovich, 2009; Schulz, 2012), suggesting that although some building-related sources of PCBs likely exist, the relative contribution of these compared to the sources emitting directly to outdoor air is lower than in many other regions, as also suggested by the concentrations and congener profiles.

3.1.4. *Seasonal variability of PCB concentrations*

The indoor concentrations of Σ_7PCB and of all individual congeners were significantly higher in summer than in winter (Mann-Whitney *U* test, $p < 0.01$ for Σ_7PCB, Fig. 2a). A similar pattern was observed in outdoor air; summer outdoor concentrations were consistently higher than winter for all congeners and Σ_7PCB (Mann-Whitney *U* test, $p < 0.0001$ for Σ_7PCB, Fig. 2a). However, the summer/winter ratios (S/W) were higher and more consistent in outdoor air than indoors (Table 3), indicating stronger seasonal influences in outdoor air. Temperature-dependent outdoor seasonality is typical for PCBs (Halsall et al., 1995; Holoubek et al., 2007; Klánová et al., 2006) due to enhanced volatilization from both primary and secondary PCB sources at warmer temperatures; we assume the same influence here as the primary driver of the seasonal difference in concentrations. It is interesting that this seasonality is mimicked in indoor air, although clearly seasonal variations indoors are less marked than in outdoor air. This suggests either an influence of outdoor air on indoor environments, possibly enhanced in summer due to more ventilation, and seasonal temperature influences acting on indoor environments. The variation in temperature between summer and winter seasons is much smaller indoors (differences of 5–10 °C) than outdoors (20–40 °C), but this may contribute.

3.1.5. *Influence of house characteristics on PCB concentrations*

Despite the small range of air concentrations observed across houses, there was a difference between houses built before and after 1984, the end of PCB use in Czech Republic (Holoubek et al., 2006). Concentrations of Σ_7PCBs in indoor air in homes built before 1984 were ~35% higher than in those built after 1984, which may translate into differences in PCB exposure (Egsmose et al., 2016), however on a seasonal basis this difference was only statistically significant in summer (Fig. 3, Mann-Whitney *U* test, $p < 0.05$). This again suggests that while the buildings themselves may be a source of PCBs, perhaps the PCBs are in materials from the outside of the buildings rather than directly within rooms, and thus act as a stronger source at higher temperatures. But the consistently higher indoor concentrations also suggest that the indoor environment may concentrate/accumulate the PCBs. We note that indoor concentrations of Σ_7PCBs were consistently higher than outdoor concentrations even for new buildings which we presume to have no sources of PCBs. In fact, the I/O ratios had no relationship with building construction date. The status of the house (i.e. in original form or renovated) also did not affect PCB concentrations; the year of the original construction was the strongest influence on the air concentrations. The differences in outdoor air near pre- and post-1984 buildings were not significant compared to the seasonal differences in outdoor air (Fig. 3).

3.2. *PBDEs*

3.2.1. *PBDE air concentrations*

PBDEs were detected in all indoor samples, although the higher molecular weight PBDEs were rarely detected. Concentrations and ranges of Σ_9PBDE (sum of BDE 28, 47, 66, 85, 99, 100, 153, 154 and 183) are presented in Table 2. BDEs 28, 47, 99, and 100 were most frequently detected while BDEs 153, 154 and 183 were only detected in a limited number of samples (Table S4). Air concentrations of PBDEs varied by 20–100 × among different houses; this range was much higher than that observed for PCBs. Mean indoor air concentration of Σ_9PBDEs was 9.1 pg/m^3 in summer (median: 4.5 pg/m^3) and 8.5 pg/m^3 in winter (median: 5.8 pg/m^3). These levels are in the lower range of what has been typically reported for indoor residential environments in North America (Allen et al., 2007; Zhang et al., 2011) and some European sites (Cequier et al., 2014; Harrad et al., 2006; Vorkamp et al., 2011) but comparable to those measured elsewhere in Europe (Alliot et al., 2014; Bjorklund et al., 2012; Bohlin et al., 2008) as well as in an indoor office in Czech Republic (Bohlin et al., 2014b) (Fig. 1).

Mean outdoor air concentrations of Σ_9PBDE were 1.8 pg/m^3 in summer (median: 1.3 pg/m^3) and 0.49 pg/m^3 in winter (median 0.42 pg/m^3). The concentrations were consistent with outdoor air concentrations in Czech Republic (Bohlin et al., 2014a; Jarkovský et al., 2015), Sweden and UK (Bohlin et al., 2008), while lower than in Canada (Melymuk et al., 2012; Shoeib et al., 2014), USA (Salamova and Hites, 2011) and Mexico (Bohlin et al., 2008).

Congener profiles of PBDEs were difficult to evaluate because of the limited detection of the higher molecular weight PBDEs. However, the data shows some variability between individual homes, with higher contributions of penta-to hepta-BDEs in four homes. The average profiles were similar in summer, both indoor and outdoor, and winter indoor. Only winter outdoor air had a different congener profile, with a statistically significant shift

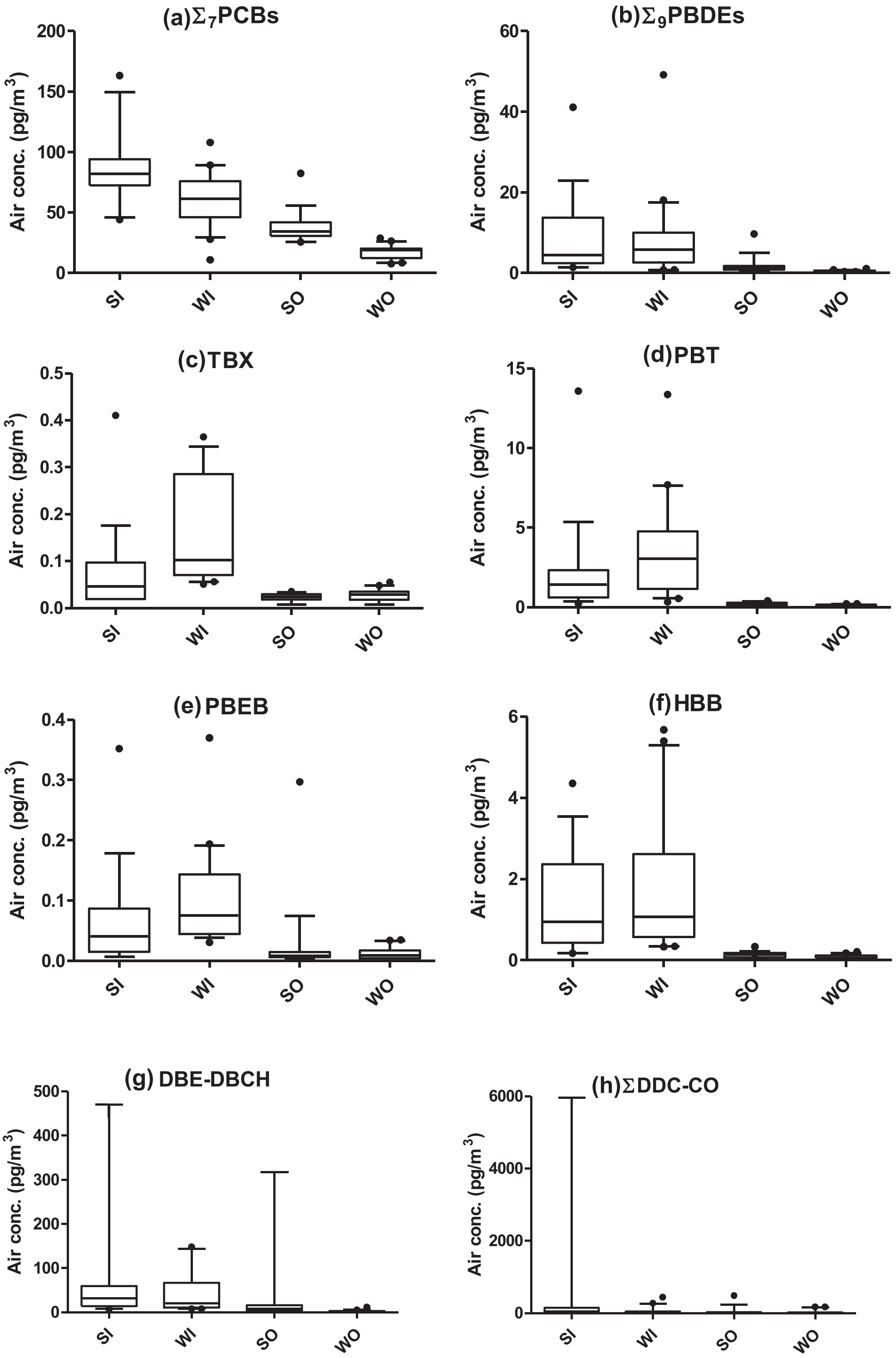

Fig. 2. Concentration ranges for (a) PCBs, (b) PBDEs, and selected NFRs: (c) TBX, (d) PBT, (e) PBEB, (f) HBB, (g) DBE-DBCH, and (h) DDC-CO. Boxes represent medians and 25th/75th percentiles, and whiskers are 10th and 90th percentiles. Outlying measurements are indicated by points. The x-axis labels indicate the season and location of samples (S = summer, W = winter, I = indoors, O = outdoors).

towards heavier congeners (BDE-99 and -100; Fig. S3), attributed to lower outdoor winter temperatures.

3.2.2. Indoor-outdoor PBDE relationships

The indoor air concentrations of Σ_9PBDEs were significantly higher than the corresponding outdoor air concentrations in both summer and winter (Mann-Whitney *U* test, $p < 0.0001$), with positive I/O ratios in both summer and winter (Table 3). As with PCBs, I/O ratios were significantly higher in winter, suggesting a stronger indoor-outdoor gradient in winter than in summer, presumably caused by a combination of increased building ventilation rates (due to open windows) and increased volatilization outdoors in summer. Indoor and outdoor concentrations at each site were significantly correlated (Spearman $r = 0.52$ in summer, and $r = 0.47$

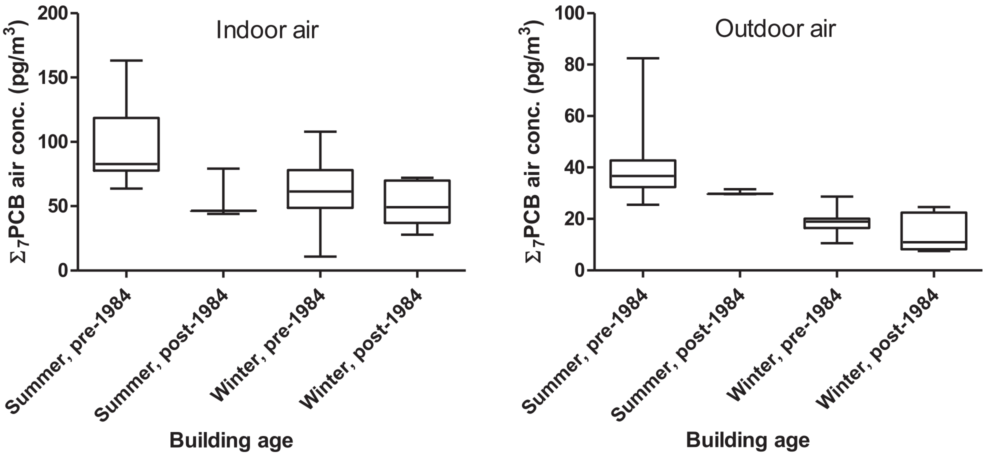

Fig. 3. Differences in PCB concentrations in indoor and outdoor air between buildings constructed before and after the ban on PCB use.

in winter, $p < 0.05$) suggesting indoor environments acting as a source to local outdoor environments, particularly in summer.

3.2.3. *Seasonal variability of PBDE concentrations*

Indoors, there was no statistically significant difference between air concentrations in winter and summer. The summer and winter indoor concentrations were also highly correlated for individual homes (Spearman $r = 0.81$, $p < 0.0001$) suggesting concentrations are more strongly linked to the specific individual environment (and presumably the indoor products and materials, discussed below), and that seasonal influences on PBDE concentrations indoors are minimal.

In contrast to the indoor concentrations, summer outdoor concentrations were 2.7 × higher than winter outdoor (Mann-Whitney *U* test, $p < 0.0001$). However, by site the outdoor summer and winter concentrations were again highly correlated (Spearman $r = 0.76$, $p < 0.0005$), suggesting that outdoor concentrations are driven by the magnitude of emissions from the local sources (e.g., higher emissions from indoor-to-outdoor transfer or from volatilization in summer), but that the distribution of sources stays the same. The importance of indoor-to-outdoor transfer is also suggested by the I/O ratios; median I/O ratios in summer (4.9) were significantly lower than in winter (13) suggesting that in summer, indoor environments are acting as a source and air exchange decreases indoor concentrations and elevates outdoor concentrations, while in winter the reduced air exchange between indoor and outdoor environments leads to a greater difference between indoor and outdoor concentrations.

3.2.4. *Influence of house characteristics on PBDE concentrations*

Although the variability of indoor PBDE concentrations was high between individual houses, there were no correlations between PBDE concentrations and building parameters (age, type, renovation status, window material, furniture, flooring, etc.), While the indoor/outdoor and summer/winter ratios clearly suggest indoor environments as a major source to outdoors, the amount of complexity in the materials and products in indoor environments makes this difficult to quantitatively relate to the presence or absence of certain products. For example, concentrations of ΣPBDEs in building insulation purchased between 2005 and 2010 had a 10,000-fold range, from below detection to >600 µg/g (Vojta et al., in prep), suggesting that simple information on the type and age of building materials or consumer products is insufficient to link to indoor concentrations. However, elevated concentrations were found in homes with higher numbers of electronics, based on a general count of the number of electronic items (e.g., TVs, computers, stereos). Median concentrations were 3.1 pg/m³, 6.5 pg/m³ and 8.8 pg/m³ for rooms with one, two, and three or more electronic items (Fig. 4), however there was only a statistically significant difference between rooms with one and three or more electronic items.

3.3. *NFRs*

3.3.1. *NFR air concentrations*

Of the 10 targeted NFRs, two (TBP-BAE and DBHCTD) were not detected in any sample. Concentrations of the detected NFRs are presented in Table 2. TBP-DBPE and BTBPE were infrequently detected so we do not discuss their trends in further detail. Syn- and anti-DDC-CO were quantified separately, but no systematic variation was observed in the isomer ratios (~37% anti) in relation to either season or location, so hereafter we discuss DDC-CO as the sum of both isomers.

As with the PBDEs, large variability existed between different houses (generally 7–150 × differences between highest and lowest concentration houses). DDC-CO had one very high outlier observation: summer indoor concentrations in one home were 500 × higher than the median indoor summer DDC-CO concentration. Indoors in winter, median concentrations of DDC-CO were lower, but concentrations in the same home remained anomalously high: 30 × higher than the winter indoor median. DBE-DBCH (sum

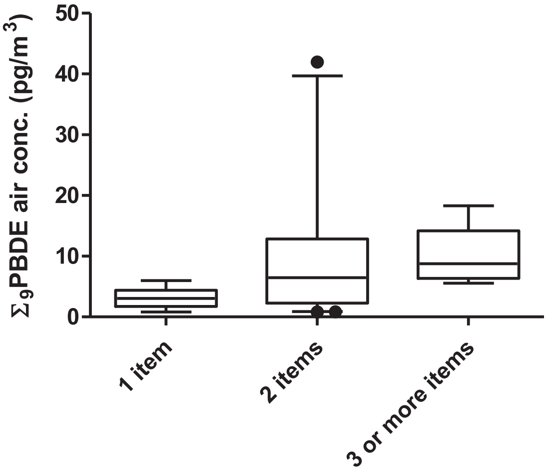

Fig. 4. Relationship between the indoor PBDE concentrations and number of electronics per room.

of α, β, γ, and δ isomers) and DDC-CO were detected at the highest levels; 10–1000 × higher than the other NFRs, and neither exhibited a significant difference in indoor concentrations between summer and winter. Mean indoor air concentrations of DDC-CO were 840 pg/m^3 (median: 26 pg/m^3) and DBE-DBCH was 100 pg/m^3 (median: 28 pg/m^3). TBX, PBEB, PBT, and HBB were at much lower levels, with concentrations of 0.1–10 pg/m^3. NFR levels are comparable to those in an office in Czech Republic (Bohlin et al., 2014b) and indoor residential levels in Norway (Cequier et al., 2014), with the exception of DDC-CO, which was much higher in the Czech indoor samples, and TBX, which was much higher in Norwegian samples (Cequier et al., 2014). The levels of individual NFRs indoors were generally much lower than the banned PCBs, suggesting indoor "persistence" of legacy SVOCs through continuous primary or secondary sources.

Large spatial variations in NFR concentrations also existed outdoors, with differences of 5–900 × between sites in summer and 4–50 × in winter. The largest spatial differences were in the compounds found at the highest concentrations: DBE-DBCH and DDC-CO. Mean outdoor air concentrations of DBE-DBCH were 96 pg/m^3 in summer (median: 7.8 pg/m^3) and 2.2 pg/m^3 in winter (median: 0.85 pg/m^3). Mean outdoor air concentrations of DDC-CO were 53 pg/m^3 in summer (median: 15 pg/m^3) and 30 pg/m^3 in winter (median: 6.0 pg/m^3). As indoors, the concentrations of TBX, PBEB, PBT, and HBB were much lower, with concentrations of 0.01–0.5 pg/m^3. The concentrations were generally consistent with other outdoor air concentrations (Bohlin et al., 2014a; Shoeib et al., 2014; Venier and Hites, 2008).

3.3.2. Indoor-outdoor NFR relationships

The indoor levels of all NFRs were significantly higher than outdoor levels in both summer and winter (Mann-Whitney *U* test, $p < 0.05$). Average I/O ratios for NFRs were 2.9–6.7 in summer and 4.6–20 in winter. As with the PBDEs, the I/O ratios suggest that outdoor concentrations strongly depend on indoor sources: I/O ratios in summer were significantly lower than in winter, suggesting indoor environments in summer are acting as a source to outdoors; in winter reduced air exchange between indoor and outdoor environments led to a greater difference between indoor and outdoor concentrations.

3.3.3. Seasonal variability of NFR concentrations

As mentioned above, DBE-DBCH and DDC-CO had similar indoor concentrations in winter and summer. HBB also had no statistically significant difference. In contrast, TBX, PBEB, and PBT had higher indoor concentrations in winter than in summer (Mann-Whitney *U* test, $p < 0.05$), with winter concentrations two to three times higher than summer.

Indoors, summer and winter concentrations were generally correlated (Spearman $r > 0.63$, $p < 0.01$ for DBE-DBCH, TBX, PBEB, and HBB) suggesting that, as with the PBDEs, concentrations are strongly linked to the specific individual environment (and presumably the indoor products and materials, discussed below). Correlations for PBT and DDC-CO were weaker and not statistically significant ($p = 0.1$) due to a higher incidence of non-detects for these compounds.

Outdoors, the seasonal trends contrasted with those indoors. Summer outdoor concentrations of DBE-DBCH, PBT and HBB were significantly higher than winter outdoor concentrations (Mann-Whitney *U* test, $p < 0.01$). For TBX, PBEB, and DDC-CO, summer outdoor concentrations were also higher than winter outdoors, but the differences were not statistically significant. By site, outdoor summer and winter concentrations were correlated for the high concentration NFRs: DDC-CO (Spearman $r = 0.51$, $p < 0.05$) and DBE-DBCH (Spearman $r = 0.80$, $p < 0.0001$). This suggests that outdoor levels were driven by the magnitude of emissions from the local sources but that the distribution of sources stayed the same, as observed for PBDEs. The lack of correlation for the lower concentration compounds may be due to the higher number of sites with levels below limits of detection, particularly outdoors in winter.

3.3.4. Influence of house characteristics on NFR concentrations

As with PBDEs, although the variability in NFR concentrations was high between houses, there were no apparent correlations between NFR levels and the house parameters (age, type, reconstruction status, window material, furniture, flooring, etc.). The most notable range was in DDC-CO and DBE-DBCH; DDC-CO concentrations in one home were ~100 × higher than in the home with the second highest levels, and DBE-DBCH concentrations in a different home were 10 × higher than the second highest levels. DDC-CO is an additive flame retardant in many polymer products, most notably in electrical wires and cables and plastic building materials, and can be 10–35% of a product composition (Schlabach et al., 2011; Xian et al., 2011). DBE-DBCH is an additive flame retardant commonly used in polystyrene building insulation at ~1% of composition (Covaci et al., 2011; Schlabach et al., 2011). However information on electrical products and building materials in the homes gave no insight into the uniquely elevated levels of these flame retardants in these two homes. This emphasizes the complexity and challenges in linking indoor concentrations of FRs and other SVOCs with specific building materials and/or consumer products. For example, a 10,000-fold range was observed in concentrations of BFRs in televisions from the same country and similar time period (Gallen et al., 2014). The wide range of flame retardants in use and the lack of distinctive patterns of use currently prevent us from expanding beyond broad generalizations in linking indoor levels with consumer product levels.

3.4. Within-house variability

In order to assess the variability within individual homes, bedrooms in three homes were sampled in both winter and summer, in addition to the living room samples that form the core of this study. In all cases the within-home variability was less than the between-home variability, e.g., the typical range of concentrations observed within a home was smaller than the typical range of concentrations observed when comparing different homes. We evaluated the following hypothesis: that variability between rooms is greater for compounds with consumer products (electronics, furniture) as sources rather than building materials or outdoor air.

While these results are based on only three homes, they suggest that for PCBs, the concentrations were relatively uniform within a home (Fig. S4), with generally <50% difference between Σ_7PCB concentrations within the same home. Conversely for PBDEs, while concentrations in two homes had <50% difference in Σ_9PBDEs between rooms, in one home concentrations were more than two times higher in the living room than in the bedroom (Fig. S4). The differences for NFRs were even more striking. While some homes had relatively uniform concentrations (e.g., HBB in House 9, Fig. S4), for other compounds/homes the air concentrations of NFRs differed more than 10-fold between rooms in the same house (e.g., PBT and HBB in House 11, Fig. S4). This indicates that compounds with outdoor or building material sources (in this case PCBs) may have more uniform concentrations within the house, while compounds with sources unique to individual rooms (flame retardants) may have more heterogeneous distributions. Moreover, this suggests greater spatial heterogeneity in compounds with significant current use (NFRs), as has also been observed within an individual room (Melymuk et al., 2016).

There was no consistent pattern related to room type or season.

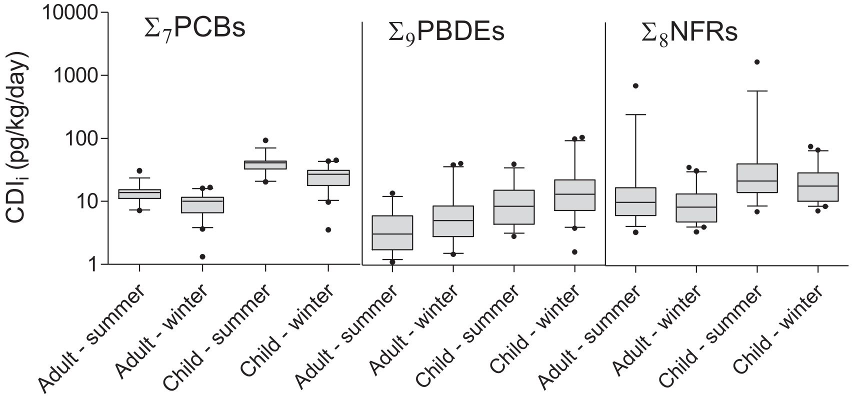

Fig. 5. Seasonal differences in estimated inhalation exposure for children and adults. Boxes show median ± standard error, whiskers show 10th and 90th percentiles, and outliers are shown by points. For summer n = 17 and for winter n = 20 (details in Table S7).

In some cases the variability in summer was greater and in other cases winter, and in some cases the living room had higher concentrations while in some cases bedrooms.

4. Implications for human exposure

The seasonal differences in the indoor environment are expected to also lead to seasonal differences in human exposure. To demonstrate these differences, we estimated seasonal chronic daily intake (CDI) via inhalation to PCBs and FRs in winter and summer. CDI_i is the time-weighted average exposure to a chemical, for inhalation intake only. We considered both the differences in concentrations between summer and winter, both indoor and outdoor, and the difference in human activity patterns between winter and summer (Leech et al., 2002; Schweizer et al., 2007; further described in SI), and average inhalation and body mass parameters for adults (31–61 year old) and children (3–6 year old) (US EPA, 2011). Calculations are described further in the SI. Exposure via the indoor environment contributed ~96% of the total CDI_i in summer and ~98% in winter (Fig. 5), emphasizing the importance of the indoor air for human exposure. This is related to high percentage of time spent indoors (>90%) combined with the higher indoors levels for all the studied compounds. Moreover, the differences between median exposure and high-level exposure (in this case 90th percentile) were much greater for the current use compounds than legacy compounds; for example, there were 100x differences in median and 90th percentile exposure for NFRs, while PCBs had only 50–60% differences between median and 90th percentile exposure.

5. Conclusions

Elevated indoor concentrations of PCBs in Czech homes suggest a diverse set of sources, including indoor and outdoor, primary and secondary. Indoor sources, while apparently minor compared to those observed in North America and Western Europe, can contribute to differences in air concentrations between older and newer homes. Both indoor and outdoor air concentrations have seasonal influences; summer concentrations are higher due to enhanced volatilization at warmer temperatures. The sources may be linked to the age of buildings, and with more sources in areas where buildings were constructed before 1984, leading to higher indoor air concentrations of PCBs. The elevated indoor air concentrations relative to outdoor may suggest some use of PCBs indoors in Czech Republic, which has previously not been identified, in addition to the presence of secondary indoor sources. This requires further investigation in both residential and non-residential buildings.

PBDE sources are primarily in the indoor environment, leading to higher indoor concentrations. Indoor concentrations are generally constant throughout the year. The distributions suggest that indoor environments can be a source to outdoors due to building ventilation; when windows are open and there is greater air exchange between indoor and outdoor environments, the indoor-outdoor concentration gradient is lower, presumably because indoor environments are releasing PBDEs to the local outdoor air, increasing outdoor PBDE air concentrations.

NFRs behave similarly to PBDEs in terms of indoor/outdoor distributions and seasonal trends, although with even more source-specific heterogeneity between homes due to their on-going use.

It is challenging to identify the specific sources of any of the compounds from this analysis, however some insight is provided. A key gap in our understanding of PCB sources relates to the relative contribution of building vs. electrical/industrial sources. While the majority of the PCB inventory in Czech Republic is in electrical/industrial sources, there remains a lack of information about any use indoors, although the elevated indoor concentrations in pre-1984 buildings suggest this may be worthy of further investigation. PBDEs and NFRs are known to be related to specific materials, but the complexity of the range of materials with FRs, and range of types and levels of FRs used in consumer products make it very challenging to identify specific products as drivers of indoor levels, or make generalizations based on building parameters. Linking indoor concentrations with FR distributions in consumer products, compound regulations and global market distribution remains a key gap in our understanding of FRs in indoor environments.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.envpol.2016.07.018.

References

Allen, J.G., McClean, M.D., Stapleton, H.M., Nelson, J.W., Webster, T.F., 2007. Personal exposure to polybrominated diphenyl ethers (PBDEs) in residential indoor air. Environ. Sci. Technol. 41, 4574–4579.

Alliot, F., Moreau-Guigon, E., Bourges, C., Desportes, A., Teil, M.-J., Blanchard, M., Chevreuil, M., 2014. A multi-residue method for characterization of endocrine disruptors in gaseous and particulate phases of ambient air. Atmos. Environ. 92, 1–8.

Bjorklund, J.A., Thuresson, K., Cousins, A.P., Sellström, U., Emenius, G., de Wit, C.A., 2012. Indoor air is a significant source of tri-decabrominated diphenyl ethers to outdoor air via ventilation systems. Environ. Sci. Technol. 46, 5876–5884.

Bohlin, P., Audy, O., Škrdlíková, L., Kukučka, P., Přibylová, P., Prokeš, R., Vojta, Š., Klánová, J., 2014a. Outdoor passive air monitoring of semi volatile organic compounds (SVOCs): a critical evaluation of performance and limitations of polyurethane foam (PUF) disks. Environ. Sci. Process Impacts 16, 433–444.

Bohlin, P., Audy, O., Škrdlíková, L., Kukučka, P., Vojta, Š., Přibylová, P., Prokeš, R., Čupr, P., Klánová, J., 2014b. Evaluation and guidelines for using polyurethane foam (PUF) passive air samplers in double-dome chambers to assess semi-volatile organic compounds (SVOCs) in non-industrial indoor environments. Environ. Sci. Process. Impacts 16, 2617–2626.

Bohlin, P., Jones, K.C., Tovalin, H., Strandberg, B., 2008. Observations on persistent organic pollutants in indoor and outdoor air using passive polyurethane foam samplers. Atmos. Environ. 42, 7234–7241.

CECBP, 2009. Brominated and Chlorinated Organic Chemical Compounds Used as Flame Retardants. Sacremento, California.

Cequier, E., Ionas, A.C., Covaci, A., Marcé, R.M., Becher, G., Thomsen, C., 2014. Occurrence of a broad range of legacy and emerging flame retardants in indoor environments in Norway. Environ. Sci. Technol. 48, 6827–6835.

Covaci, A., Harrad, S.J., Abdallah, M.A.E., Ali, N., Law, R.J., Herzke, D., de Wit, C.A., 2011. Novel brominated flame retardants: a review of their analysis, environmental fate and behaviour. Environ. Int. 37, 532–556.

Egsmose, E.L., Bräuner, E.V., Frederiksen, M., Mørck, T.A., Siersma, V.D., Hansen, P.W., Nielsen, F., Grandjean, P., Knudsen, L.E., 2016. Associations between plasma concentrations of PCB 28 and possible indoor exposure sources in Danish school children and mothers. Environ. Int. 87, 13–19. http://dx.doi.org/10.1016/j.envint.2015.11.005.

European Chemicals Agency, 2014. No Title [WWW Document]. URL http://echa.europa.eu/web/guest (accessed 02.24.15.).

Frederiksen, M., Meyer, H.W., Ebbehøj, N.E., Gunnarsen, L., 2012. Polychlorinated biphenyls (PCBs) in indoor air originating from sealants in contaminated and uncontaminated apartments within the same housing estate. Chemosphere 89, 473–479.

Gallen, C., Banks, A., Brandsma, S., Baduel, C., Thai, P., Eaglesham, G., Heffernan, A., Leonards, P., Bainton, P., Mueller, J.F., 2014. Towards development of a rapid and effective non-destructive testing strategy to identify brominated flame retardants in the plastics of consumer products. Sci. Total Environ. 491–492, 255–265.

Halsall, C.J., Lee, R.G.M., Coleman, P.J., Burnett, V., Harding-Jones, P., Jones, K.C., 1995. PCBs in U.K. urban air. Environ. Sci. Technol. 29, 2368–2376.

Harju, M.T., Heimstad, E.S., Herzke, D., Sandanger, T., Posner, S., Wania, F., 2009. Current State of Knowledge and Monitoring Requirements: Emerging “new” Brominated Flame Retardants in Flame Retarded Products and the Environment (TA-2462/2008). Norwegian Pollution Control Authority SFT.

Harrad, S.J., Hazrati, S., Ibarra, C., 2006. Concentrations of polychlorinated biphenyls in indoor air and polybrominated diphenyl ethers in indoor air and dust in Birmingham, United Kingdom: implications for human exposure. Environ. Sci. Technol. 40, 4633–4638.

Hazrati, S., Harrad, S.J., 2007. Calibration of polyurethane foam (PUF) disk passive air samplers for quantitative measurement of polychlorinated biphenyls (PCBs) and polybrominated diphenyl ethers (PBDEs): factors influencing sampling rates. Chemosphere 67, 448–455.

Holoubek, I., Adamec, V., Bartoš, M., Bláha, K., Bláha, L., Budňáková, M., Černa, M., Čupr, P., Demnerová, K., Drápal, J., Hajšlová, J., Hanzálková, M., Holoubková, I., Hrabětová, S., Jech, L., Klánová, J., Kohoutek, J., Kužílek, V., Machálek, P., Matějů, V., Matoušek, J., Matoušek, M., Mestřík, V., Novák, J., Ocelka, T., Pekárek, V., Petrlík, J., Petira, O., Provazník, K., Punčochář, M., Rieder, M., Ruprich, J., Sáňka, M., Tomaniová, M., Vácha, R., Volka, K., Zbíral, J., 2006. The National Implementation Plan for Implementation of the Stockholm Convention in the Czech Republic. Brno.

Holoubek, I., Klánová, J., Jarkovský, J., Kohoutek, J., 2007. Trends in background levels of persistent organic pollutants at Kosetice observatory, Czech Republic.1) Part I. Ambient air and wet deposition 1996-2005. J. Environ. Monit. 9, 557–563.

Jarkovský, J., Dušek, L., Klánová, J., Hůlek, R., Šebková, K., Borůvková, J., Kalina, J., Gregor, J., Bednářová, Z., Novák, R., Šalko, M., Hřebíček, J., Holoubek, I., 2015. Multi-matrix Online Data Browser for Environmental Analysis and Assessment [online] [WWW Document]. Masaryk Univ.

Klánová, J., Kohoutek, J., Hamplová, L., Urbanová, P., Holoubek, I., 2006. Passive air sampler as a tool for long-term air pollution monitoring: Part 1. Performance assessment for seasonal and spatial variations. Environ. Pollut. 144, 393–405.

Klosterhaus, S.L., McKee, L.J., Yee, D., Kass, J.M., Wong, A., 2014. Polychlorinated biphenyls in the exterior caulk of San Francisco Bay area buildings, California, USA. Environ. Int. 66, 38–43.

Kohler, M., Tremp, J., Zennegg, M., Seiler, C., Minder-Kohler, S., Beck, M., Lienemann, P., Wegmann, L., Schmid, P., 2005. Joint sealants: an overlooked diffuse source of polychlorinated biphenyls in buildings. Environ. Sci. Technol. 39, 1967–1973.

Leech, J.A., Nelson, W.C., Burnett, R.T., Aaron, S., Raizenne, M.E., 2002. It's about time: a comparison of Canadian and American time-activity patterns. J. Expo. Anal. Environ. Epidemiol. 12, 427–432.

Ma, Y., Venier, M., Hites, R.A., 2012. Tribromophenoxy flame retardants in the Great Lakes atmosphere. Environ. Sci. Technol. 46, 13112–13117.

MacIntosh, D.L., Minegishi, T., Fragala, M.A., Allen, J.G., Coghlan, K.M., Stewart, J.H., McCarthy, J.F., 2012. Mitigation of building-related polychlorinated biphenyls in indoor air of a school. Environ. Health 11, 24.

Melymuk, L., Bohlin-Nizzetto, P., Vojta, Š., Krátká, M., Kukučka, P., Audy, O., Přibylová, P., Klánová, J., 2016. Distribution of legacy and emerging semivolatile organic contaminants in five indoor matrices in a residential environment. Chemosphere 153, 179–186.

Melymuk, L., Robson, M., Helm, P.A., Diamond, M.L., 2012. PCBs, PBDEs, and PAHs in Toronto air: spatial and seasonal trends and implications for contaminant transport. Sci. Total Environ. 429, 272–280.

Menichini, E., Iacovella, N., Monfredini, F., Turrio-Baldassarri, L., 2007. Relationships between indoor and outdoor air pollution by carcinogenic PAHs and PCBs. Atmos. Environ. 41, 9518–9529.

Prevedouros, K., Jones, K.C., Sweetman, A.J., 2004. Estimation of the production, consumption, and atmospheric emissions of pentabrominated diphenyl ether in Europe between 1970 and 2000. Environ. Sci. Technol. 38, 3224–3231.

Riddell, N., Arsenault, G., Lough, A., McAlees, A., McCrindle, R., Meissner, J., Robertson, V., 2008. The three-dimensional structural characterization of hexachlorocyclopentenyl-dibromocyclooctane (HCDBCO). Chemosphere 73, 479–483.

Robson, M., Melymuk, L., Csiszar, S.A., Giang, A., Diamond, M.L., Helm, P.A., 2010. Continuing sources of PCBs: the significance of building sealants. Environ. Int. 36, 506–513.

Rudel, R.A., Dodson, R.E., Perovich, L.J., Morello-Frosch, R., Camann, D.E., Zuniga, M.M., Yau, A.Y., Just, A.C., Brody, J.G., 2010. Semivolatile endocrine-disrupting compounds in paired indoor and outdoor air in two northern California communities. Environ. Sci. Technol. 44, 6583–6590.

Rudel, R.A., Perovich, L.J., 2009. Endocrine disrupting chemicals in indoor and outdoor air. Atmos. Environ. 43, 170–181.

Salamova, A., Hites, R.A., 2011. Discontinued and alternative brominated flame retardants in the atmosphere and precipitation from the Great Lakes basin. Environ. Sci. Technol. 45, 8698–8706.

Schlabach, M., Remberger, M., Kaj, L., Herzke, D., Borgen, A., Harju, M.T., 2011. Brominated Flame Retardants (BFR) in the Nordic Environment. Nordic Council of Ministers, Copenhagen.

Schulz, T.J., 2012. Comparison of PCBs in East Chicago, Indiana and Columbus Junction, Iowa in Indoor and Outdoor Air. University of Iowa.

Schweizer, C., Edwards, R.D., Bayer-Oglesby, L., Gauderman, W.J., Ilacqua, V., Jantunen, M.J., Lai, H.K., Nieuwenhuijsen, M., Künzli, N., 2007. Indoor time-microenvironment-activity patterns in seven regions of Europe. J. Expo. Sci. Environ. Epidemiol. 17, 170–181.

Shoeib, M., Ahrens, L., Jantunen, L., Harner, T., 2014. Concentrations in air of organobromine, organochlorine and organophosphate flame retardants in Toronto, Canada. Atmos. Environ. 99, 140–147.

Taniyasu, S., Kannan, K., Holoubek, I., Ansorgova, A., Horii, Y., Hanari, N., Yamashita, N., Aldous, K.M., 2003. Isomer-specific analysis of chlorinated biphenyls, naphthalenes and dibenzofurans in Delor: polychlorinated biphenyl preparations from the former Czechoslovakia. Environ. Pollut. 126, 169–178.

US EPA, 2011. Exposure Factors Handbook: 2011 Edition. U.S. Environ. Prot. Agency EPA/600/R-, 1–1466 doi:EPA/600/R-090/052F.

Venier, M., Hites, R.A., 2008. Flame retardants in the atmosphere near the Great Lakes. Environ. Sci. Technol. 42, 4745–4751.

Venier, M., Ma, Y., Hites, R.A., 2012. Bromobenzene flame retardants in the Great Lakes atmosphere. Environ. Sci. Technol. 46, 8653–8660.

Vojta, Š., Melymuk, L., Bečanová, J., Kukučka, P., Kohoutek, J., Klánová, J., n.d. Halogenated flame retardants in building materials, consumer products and e-waste on the European market. in prep.

Vorkamp, K., Thomsen, M., Frederiksen, M., Pedersen, M., Knudsen, L.E., 2011. Polybrominated diphenyl ethers (PBDEs) in the indoor environment and associations with prenatal exposure. Environ. Int. 37, 1–10.

WHO European Centre for Environment and Health, 2010. Guidelines for Indoor Air Quality – Selected Pollutants. World Health Organization Regional Office for Europe, Copenhagen.

Xian, Q., Siddique, S., Li, T., Feng, Y.-L., Takser, L., Zhu, J., 2011. Sources and environmental behavior of dechlorane plus—a review. Environ. Int. 37, 1273–1284.

Zhang, X., Diamond, M.L., Ibarra, C., Harrad, S.J., 2009. Multimedia modeling of polybrominated diphenyl ether emissions and fate indoors. Environ. Sci. Technol. 43, 2845–2850.

Zhang, X., Diamond, M.L., Robson, M., Harrad, S.J., 2011. Sources, emissions, and fate of polybrominated diphenyl ethers and polychlorinated biphenyls indoors in Toronto, Canada. Environ. Sci. Technol. 45, 3268–3274.

APPENDIX 14

Demirtepe, Hale, Lisa Melymuk, Garry Codling, Ľubica Palkovičová Murínová, Denisa Richterová, Vladimíra Rašplová, Tomáš Trnovec, and Jana Klánová. 2021. "Targeted and Suspect Screening of Plasticizers in House Dust to Assess Cumulative Human Exposure Risk." *Science of The Total Environment* 781: 146667. https://doi.org/10.1016/j.scitotenv.2021.146667

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Targeted and suspect screening of plasticizers in house dust to assess cumulative human exposure risk

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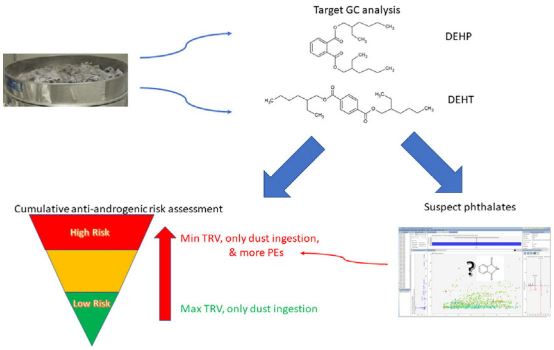
HIGHLIGHTS

- DEHP & DEHT dominated phthalate & alternative plasticizer dust levels, respectively.
- Acceptable risks estimated for anti-androgenic effects of PEs via dust ingestion.
- Uncertainty in toxicity data and other exposure routes may alter the risk.
- Non-target analysis suggests >50 PEs in dust; many are ignored in risk assessment.

GRAPHICAL ABSTRACT



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ABSTRACT

Indoor dust is an important exposure route to anthropogenic chemicals used in consumer products. Plasticizers are common product additives and can easily leach out of the product and partition to dust. Investigations of plasticizers typically focus on a subset of phthalate esters (PEs), but there are many more PEs in use, and alternative plasticizers (APs) are seeing greater use after recognition of adverse health effects of PEs. In this study we use full scan high resolution mass spectrometry for targeted and suspect screening of PEs and APs in house dust and to assess the potential risk of human exposure. House dust samples from Eastern Slovakia were investigated and concentrations of $\sum_{12}$PEs and $\sum_{5}$APs ranged 12–2765 μg/g and 45–13,260 μg/g, respectively. APs were at similar levels to PEs, indicating common usage of these compounds in products in homes.

Evaluation of individual compound toxicity combined with human intake via dust ingestion suggested PEs are of lower priority compared to semivolatile organic compounds such as polychlorinated biphenyls due to their lower toxicity. However, cumulative risk assessment (CRA) is a more appropriate evaluation of risk, considering the presences of many PEs in dust and their similar toxic mode of action. CRA based on median toxicity reference values (TRVs) suggested acceptable risks for dust ingestion, however, the wide range of literature-derived TRVs is a large uncertainty, especially for the APs. Use of newer TRVs suggest risk from dust ingestion alone, i.e. not even considering diet, inhalation, and dermal contact. Additionally, screening of full-scan instrumental spectra identified a further 40 suspect PE compounds, suggesting the CRA based on the 12 target PEs underestimates the risk.

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1. Introduction

Due to human and industrial needs, synthetic materials have become ubiquitous in our lives, and most notably plastic products. Global production of plastics reached almost 370 Mtons in 2019 with extensive usage in consumer products (Plastics Europe, 2020). Plastic materials contain a wide range of additives to impart the properties needed in the final products, and this frequently includes plasticizers, forming up to 55% by weight of the material (Narvaez Rincon and Suarez Palacios, 2015). Phthalate esters (PEs) are the most commonly used plasticizers, used to improve flexibility and durability in many consumer products (Kutz, 2017; Stanley et al., 2003). PEs are found in polyvinyl chloride (PVC) products, toys, flooring, carpets, wall coverings, food packaging, medical products, glue and paint (ATSDR, 1997, 2001, 2019; Kutz, 2017; Zhang et al., 2020), as well as cosmetics and insecticides, mainly containing low molecular weight (LMW) PEs such as diethyl phthalate (DEP) and dimethyl phthalate (DMP) (ATSDR, 1995). Many PEs are considered high production volume (HPV) compounds: yearly production and import of bis(2-ethylhexyl)phthalate (DEHP) in the European economic area (EEA) is between 10^4 and 10^5 t, while that of di-*n*-butyl phthalate (DnBP), DEP and DMP is between 10^3 and 10^4 t (ECHA, 2018) (Table S1).

Since plasticizers are applied as additives to a product, they can easily leach out of the product and be emitted to the environment where the products are used or when they are disposed as wastes, hence PEs are broadly detected in the global environment (Net et al., 2015). Due to widespread usage in indoor-related products and physicochemical properties allowing for significant partitioning to particulates, PEs can be found ubiquitously in indoor dust, which is an important human exposure route to PEs, especially to high molecular weight compounds considering their higher abundance in dust than in indoor air (Giovanoulis et al., 2018) and more specifically for infants and toddlers (Wormuth et al., 2006).

Exposure to PEs has been associated with negative effects on reproductive and developmental systems, endocrine disruption (Hauser and Calafat, 2005; Meeker et al., 2009; Swan et al., 2005), allergies and asthma in children (Ait Bamai et al., 2016; Bekö et al., 2015; Bertelsen et al., 2013), obesity and cardiometabolic risk factors in children and adolescents (Amin et al., 2018). Due to the evidence of adverse health effects, especially in children, the use of DEHP, DnBP, butylbenzyl phthalate (BBP), di-isobutyl phthalate (DiBP), di-isononyl phthalate (DINP), di-isodecyl phthalate (DIDP) and di-*n*-octyl phthalate (DnOP) has been restricted by the European Commission to not exceed 0.1% by weight of plasticized material in toys and childcare products (REACH, 2018). The same restriction was applied in the USA, but also included di-*n*-pentyl phthalate (DPP), di-*n*-hexyl phthalate (DHP) and dicyclohexyl phthalate (DCHP), but not DIDP and DnOP (CPSC, 2017).

The restrictions on the use of legacy PEs has led to the introduction of alternatives with lower toxic potential. Among these alternative plasticizers (APs), bis(2-ethylhexyl)terephthalate (DEHT) and bis(2-propylheptyl)phthalate (DPHP) have yearly production and import volume up to 10^6 t to the EEA, while acetyl tributyl citrate (ATBC), bis(2-ethylhexyl)adipate (DEHA) and 1,2-*cyclo*hexane dicarboxylic acid diisononyl ester (DINCH) are individually produced and imported in the range from 10^4 to 10^5 t (ECHA, 2018). The larger production volumes for APs compared with legacy PEs suggest similar abundance in indoor environments to the legacy PEs, especially in indoor dust due to their partitioning coefficients similar to PEs (Bui et al., 2016). Therefore, their exposure and potential toxicity should be evaluated carefully. While the APs considered for target analysis in this study have been examined for developmental and reproductive toxicity, there is limited information compared to the PEs. ECHA reports no observed adverse effect levels (NOAELs) based on developmental and reproductive toxicity for DPHP and ATBC and derived no-effect levels (DNEL) based on repeated dose toxicity for all APs, except for DINCH which is based on carcinogenicity (ECHA, 2018). However, despite the perception of lower hazard for the APs, ATBC was found to have endocrine disrupting potential and neurotoxicity (Bui et al., 2016).

Evaluation of the risk associated with indoor exposure is crucial, especially considering the range of semi-volatile organic compounds (SVOCs) typically present indoors, with wide variations in concentrations and toxicity. We have previously proposed a framework combining human intake and toxicity reference values (TRVs) to prioritize SVOCs for risk assessment (Demirtepe et al., 2019). The framework, relying on median TRVs reported in literature and by regulatory agencies, enabled a comparative evaluation of risks of individual SVOCs. However, this framework ignores the possibility of mixture effects based on chemicals having similar toxicological endpoints. PEs are known to have common developmental and reproductive health effects (Howdeshell et al., 2008), particularly anti-androgenic effects (Pelletier et al., 2018; Radke et al., 2019). Anti-androgenic effects include decreased fetal testosterone, reduced anogenital distance, reduced reproductive organ weights, retained nipples, decreased sperm production and Leydig cell adenomas (Gray et al., 2000; Kortenkamp and Faust, 2010). Multiple PEs are typically present indoors; therefore cumulative risk assessment (CRA) can more accurately estimate the human exposure risk (Kortenkamp and Faust, 2010; Pelletier et al., 2018).

This study employs ultra-high resolution mass spectrometry coupled with gas chromatography operating in full scan with the aim of (i) identifying the concentrations of target PEs and APs in Slovakian indoor dust, considered representative of European SVOC levels (Demirtepe et al., 2019), and investigating their associations with home characteristics, (ii) assessing human exposure to PEs and APs via dust ingestion, and cumulative anti-androgenic risk of PEs, and (iii) using non-target screening for the evaluation of unquantified/unknown compounds with PE structure.

2. Methods

2.1. Sample collection and preparation

Indoor dust samples from 60 homes in Eastern Slovakia were collected in March–April 2015. Details on sampling location are presented in a previous study (Demirtepe et al., 2019). The dust samples were collected using a household vacuum cleaner with polyester sock inserted in the front of the vacuum tube and vacuuming 1 to 3 m^2 floor surface. The sock was removed from vacuum cleaner, packed in aluminum foil, put into a zip-lock bag, stored in freezer at −18 °C for transport to the laboratory.

The analytical procedure used was published previously (Demirtepe et al., 2019; Jílková et al., 2018; Venier et al., 2016; Vojta et al., 2017; Vykoukalová et al., 2017), and is briefly described here. Dusts were sieved with a 500 μm sieve to remove course particles, and ~ 100 mg were taken for extraction. Dust samples were sonicated three times in 1:1 *v*/*v* hexane:acetone, and extracts were split 70:30. The 30% aliquot was cleaned and fractionated using activated silica column eluted with DCM (1st fraction), followed by 7:3 v/v acetone:DCM (2nd fraction). The fractionated extracts were exchanged to nonane and stored at −18 °C. Concentrations of polychlorinated biphenyls (PCBs), organochlorine pesticides (OCPs), polycyclic aromatic hydrocarbons (PAHs), organophosphate esters (OPEs) and halogenated flame retardants (FRs) in these dust samples have been previously reported (Demirtepe et al., 2019). In this study, 55 of the second fraction extracts were available for analysis of plasticizers, i.e. 13 PEs and five APs, listed in Table S1 with CAS numbers and acronyms.

2.2. Instrumental analysis

The extracts were fortified with benzo(*e*)pyrene-d_{12} (Wellington Laboratories Inc., Canada) and PEs and APs were analyzed with a Trace 1300 series gas chromatograph (Thermo Scientific) equipped with 30 m × 0.25 mm × 0.25 μm Rxi5-SIL-MS column coupled to a Q-Exactive GC Orbitrap (Thermo Scientific). The operation was full-scan

(70–1000 amu) in EI mode with a mass resolution of 60,000. Injection was splitless 1 μL at 280 °C. The GC temperature program was 80 °C (3 min hold), then 7 °C min^{-1} to 320 °C (23 min hold). Samples were quantified based on a 15-point internal calibration curve, with R^2 greater than 0.99 for all compounds. Mass accuracy was <2 ppm and the instrument was checked for drift in mass every 12 h during instrumental run. The calibration range covered 4 orders of magnitude in order to include the concentration range expected for some PEs.

2.3. QA/QC

These extracts were not initially extracted for PEs; therefore, the potential for elevated blank contamination was high, especially with the ubiquitous contamination in a typical laboratory. However, field blanks (4 vacuum socks) were treated as per the samples. Method detection limits (MDLs) were calculated as the average of the field blank samples plus three times the standard deviation of the blanks (ng/sample). MDLs were converted to μg/g dust by dividing by a nominal sample amount of 0.1 g. The instrument detection limits (IDLs) were derived from the lowest concentration of the analyte detected in the calibration curve. The IDL was used as the MDL for compounds that were not detected in the blanks (Table S2). For compounds having >50% detection frequency, average blank contamination corresponded to 1.2 ± 0.5% of mean concentration of each analyte. The average of the blanks was subtracted from sample values that were > MDL, and values <MDL were recorded as such. Additionally, indicative method quantification limits (MQLs) were calculated as the average of the field blank samples plus ten times the standard deviation of the blanks. Blank levels, MDLs and MQLs are reported in Table S2.

The 1st fraction extracts were also checked for PEs & APs, but the target peak areas were lower than 1% of that in 2nd fraction, hence they were not included in further analysis. DiBP contamination in the calibration curve was high and calibration for this compound cannot be accurately performed; thus, it is not discussed further in this study.

We analyzed NIST standard reference material (SRM) 2585 for PEs & APs and compared the measured concentrations with previously reported values in seven studies since SRM 2585 is not certified for plasticizers (Table S3). PEs were within the range of SRM values previously reported, although we measured consistently lower DEHP concentration than previous studies except for one (Christia et al., 2019). For APs, only two studies have reported SRM concentrations of DEHT, and one study ATBC and DEHA. Our values for DEHA and DEHT were consistent with those of Kademoglou et al. (2018) and Christia et al. (2019), but for ATBC were significantly greater than those of Christia et al. (Table S3). The SRM samples were spiked with recovery standards before extraction and the average recoveries were 95.2%, 80.2% and 81.7% ($n = 3$) for DMP-d4, DnBP-d4, and DEHP-d4, respectively.

2.4. Data analysis

Statistical analyses were conducted using IBM SPSS 25. For statistical analysis, values below detection were substituted by $\sqrt{2}/2 \cdot \text{MDL}$. Shapiro-Wilk normality test showed non-normal distribution for indoor dust data ($p < 0.001$) hence non-parametric tests, i.e. Spearman correlation (r_s) and Mann Whitney *U* test, were applied. Analysis of variance (ANOVA) was conducted after log-transforming the data. $p < 0.05$ was considered the threshold for statistical significance.

2.4.1. Cumulative risk assessment

Human intake of PEs and APs via dust ingestion was calculated using the daily exposure dose (DED) formula and exposure parameters given in the SI based on the US EPA Exposure Factors Handbook (U.S. EPA, 2011). Intakes were calculated for median and high (95th percentile) intake scenarios for a child aged between six and eleven, as dust was collected from children's bedrooms, and for an adult male. For compounds whose median is <MDL, we used substituted values as defined above for median scenario. Toxicity reference values (TRVs), defined as the maximum dose of a compound that a person can be exposed daily without a health risk, were collected as non-carcinogenic oral doses from literature and regulatory sources (Table S4). Then, for cumulative risk assessment of PEs, relative potency factors (RPFs) and reference doses (RfD) were compiled from literature for individual PEs (Table 1). Since DEHP is the common compound for studies presenting RPFs, it was selected as the index compound and median of its TRVs was used as RfD_{DEHP} in cumulative hazard quotient (HQ) calculation, given below (Pelletier et al., 2018):

$$Cumulative\ HQ = \frac{\sum_{i=1}^{n} DED_i \times RPF_i}{RfD_{DEHP}} \tag{1}$$

where *DED* and *RfD* were given in μg/kg/d, and *RPF* was unitless.

2.4.2. Screening for suspect PEs

Full scan MS spectra of indoor dust samples, standard mixtures and blanks were transferred to MS-DIAL software for deconvolution, alignment and peak identification (Tsugawa et al., 2015). The details of MS-DIAL settings and parameters are provided in the SI.

Data from MS-DIAL were processed using Python Pandas software. A total of 5412 features were found in samples, standards, and blanks, though some features are likely to be the same compound separated by retention time drift. To remove contaminant features found in the laboratory blanks, the MDL (mean plus 3 times the standard deviation) of nonane instrumental blanks ($n = 5$) was applied to all samples. Features below the MDL were excluded from further study. Each feature detected by MS-DIAL included a comprehensive list of ions; this was reduced to include just those within 10% of the most abundant ion and then the top 10 of these were retained for preliminary investigation. Where two ions were detected with less than 0.0005 amu mass difference, the most abundant was retained in any feature.

The primary structure of PEs is the 1,2-benzenedicarboxylic acid with multiple possible alkyl side chains (Fig. S1A). Therefore, the mass 149.0226 (+/− 0.0003 amu), due to protonated phthalic anhydride structure (Fig. S1B), is a characteristic feature and often the most abundant, with the exception of dimethyl phthalates and the

Table 1
Cumulative hazard quotients calculated with various RPFs. RPFs in bold indicates the reference compound used by the corresponding study.

References	RPF					ΣHQ			
						Child		Adult	
	DEHP	BBP	DEP	DnBP	DPP	median	high	median	high
Benson (2009)	**1.00**	0.21		0.64	1.26	1.1×10^{-2}	0.64×10^{-1}	1.73×10^{-3}	1.00×10^{-2}
German Federal Environment Agency (2011)	1.00	1.00	0	1.00	3.00	1.1×10^{-2}	0.70×10^{-1}	1.79×10^{-3}	1.10×10^{-2}
Hannas et al. (2011)	0.11				**1.00**	1.2×10^{-3}	0.62×10^{-2}	1.82×10^{-4}	0.97×10^{-3}
Fournier et al. (2016)[a]	**1.00**	0.088	21			1.2×10^{-2}	0.75×10^{-1}	1.81×10^{-3}	1.17×10^{-2}
Howdeshell et al. (2008)	**1.00**	0.83		0.96	2.93	1.1×10^{-2}	0.69×10^{-1}	1.78×10^{-3}	1.08×10^{-2}
Gray et al. (2000)	**1.00**	1.00	0	0.5		1.1×10^{-2}	0.64×10^{-1}	1.73×10^{-3}	1.00×10^{-2}
Varshavsky et al. (2016)	0.61	0.26	0.024	**1.00**		0.73×10^{-2}	0.47×10^{-1}	1.14×10^{-3}	0.73×10^{-2}

[a] Calculated with DEHP as reference since cypermethrin was used as reference in Fournier et al. (2016).

diphenyl-isophthalates. For dimethyl phthalates, two CH_3 groups are on the alkyl side chain, and the diphenyl *iso*phthalates have two benzene rings, giving characteristic fragments of 163.0382 and 225.0552, respectively. After filtering the ion list for these masses and eliminating the features corresponding to the targeted PEs, the remaining features were considered "suspect PEs".

3. Results and discussion

3.1. Targeted PEs and APs in indoor dust

Nine out of 12 PEs and three out of five APs had >50% detection frequency in dust samples from Slovakian homes. The median concentration of $\sum_{12}$PEs was 376 μg/g, with the greatest contribution from DEHP (80%), followed by DnBP and DnOP. $\sum_5$APs had a median concentration of 200 μg/g. DEHT was the dominant AP, with a mean contribution of 37%. The median concentrations of DEHT, ATBC and DEHA were higher than that of six frequently detected PEs, indicating abundant use of APs in homes. Descriptive statistics for PEs and APs are presented in Table 2.

When compared to the SVOCs identified in our previous study (Demirtepe et al., 2019), the median concentrations of $\sum_{12}$PEs (376 μg/g) and $\sum_5$APs (200 μg/g) were an order of magnitude greater than that of $\sum_{14}$OPEs (12.4 μg/g) and two orders of magnitude greater than that of $\sum_{27}$PAHs (2.0 μg/g). This was consistent with previous publications where PEs were identified at one to two orders of magnitude greater concentration than OPEs (Bergh et al., 2011; He et al., 2016; Luongo and Östman, 2016; Yang et al., 2020, 2019).

The median PE concentrations in indoor dust in this study were comparable to the concentrations reported in existing publications on PEs in house dust (Table S5). In most studies DEHP was the most frequently detected compound (>70% of samples) and found at the highest concentration. DEHP concentrations from literature are compared in Fig. 1 grouped by continent and in Fig. S2 by country. In Europe, the median concentration across all studies investigated here was 270 μg/g, with the lowest median observed in Belgium (62 μg/g) and the greatest in Bulgaria (1050 μg/g). The European median for DEHP is similar to the median in this study (319 μg/g), and within the range for other regions (137 μg/g for North America, $n_{NAmer} = 6$ and 435.5 μg/g for Asia, $n_{Asia} = 14$; Table S5). Thailand, with the second highest median concentration of 1739.3 μg/g (Promtes et al., 2019), has to date no DEHP restrictions (Sedtasiriphokin et al., 2017) though a new toy safety proposal on DEHP has been proposed in 2020.

Table 2
Descriptive statistics for PEs and APs for Slovakian homes (*n* = 55, *μg/g*).

	Median	Geomean	Mean	SD	Min	Max	% Frequency
Phthalate esters (PEs)							
DEEP	<MDL	0.012	0.027	0.064	<MDL	0.43	31
DEHP	319	245	470	524	<MDL	2615	98
DMEP	<MDL	0.057	0.072	0.066	<MDL	0.36	18
BBP	2.94	2.81	9.14	16.7	<MDL	87.9	93
DCHP	0.98	1.01	2.34	3.74	0.038	19.4	100
DEP	1.42	1.53	7.36	27.3	<MDL	201	80
DHP	0.19	0.23	0.85	2.20	<MDL	12.3	95
DMP	0.071	0.11	0.37	0.82	<MDL	5.44	96
DnBP	24.3	24.6	78.7	196	<MDL	1160	84
DnOP	13.2	12.6	20.2	20.3	<MDL	119	95
DNP	0.94	0.79	1.52	1.82	<MDL	8.16	96
DPP	<MDL	<MDL	0.039	0.11	<MDL	0.64	36
Total $\sum_{12}$PEs	376	309	590	655	11.8	2765	
Alternative plasticizers (APs)							
ATBC	13.4	14.1	41.5	65.8	<MDL	307	96
DEHA	7.50	6.91	21.2	48.6	<MDL	274	93
DEHT	71.3	66.0	186	629	<MDL	4713	95
DPHP	<MDL	33.2	49.5	64.8	<MDL	382	38
DINCH	<MDL	45.7	300	1770	<MDL	13,200	33
Total $\sum_5$APs	200	227	597	1870	45.4	13,260	

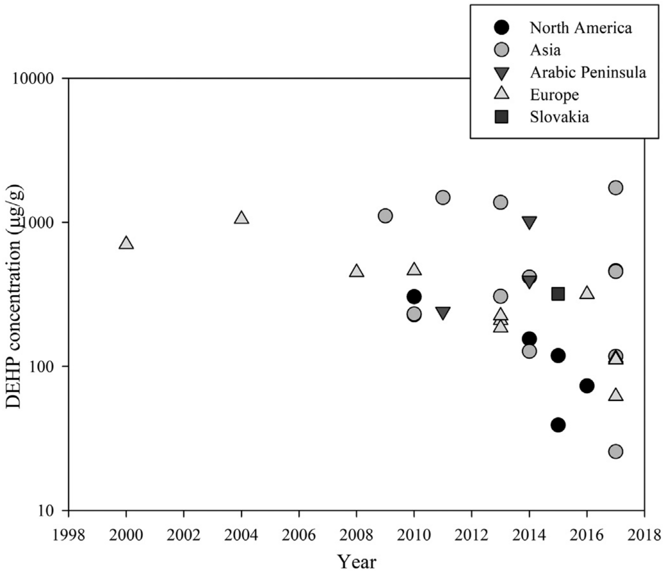

Fig. 1. Median DEHP concentrations in indoor dust samples from various continents with respect to sampling year. The studies reporting these levels are listed in Table S5.

An important observation was that DEHP concentrations reported in indoor dust show a significant decrease over time ($n = 33$, $r_s = -0.44$, $p < 0.05$) (Fig. 1). The regulatory agencies in the EU and US prohibited DEHP use in children's products in 2005 and listed it in the candidate list for substance of very high concern in 2008 (CPSC, 2017; ECHA, 2020). More recent DEHP concentrations reported in dust from Europe and North America are generally lower than those in Asia, and the decrease in reported dust concentrations of DEHP is much stronger when considering only European and North American records ($n = 17$, $r_s = -0.88, p < 0.001$; Fig. S3). This suggests that stocks and indoor uses of DEHP are being rapidly removed from indoor environments and that restrictions in use can be effective in reducing DEHP exposure. Nevertheless, a more in-depth literature survey would be required to make a definite conclusion on trends observed.

So far, few studies have identified APs in house dust, although we found AP concentrations to be comparable to PEs (Table S5). Median indoor DEHT, ATBC and DEHA concentrations from Slovakia were greater than those from Norway (Kademoglou et al., 2018), Ireland, Belgium and Netherlands, except for DEHT from Ireland (Christia et al., 2019). Median DEHT concentrations from Slovakia were also higher than that from Germany (Nagorka et al., 2011), and within the range of the studies reporting DEHT in USA (Hammel et al., 2019; Shin et al., 2020). DEHA and ATBC concentrations from the US (Subedi et al., 2017) were higher than those from Slovakia, while DEHA levels from Japan (Kishi et al., 2018) and Qatar (Nayef et al., 2019) were similar to that from Slovakia. Finally, median ATBC concentration from Slovakia was higher than that from China (Tang et al., 2020) and the US (Shin et al., 2020).

3.2. Possible sources of PEs and APs

Spearman's Rank correlations between PEs and APs suggested similar indoor sources (Table S6). For example, strong correlations were observed between DEHP, DnOP and DEHA, all of which are known to be used in wires and cables (ATSDR, 1997, 2019; Lowell Center for Sustainable Production, 2011) and DEHP and DnBP, which are commonly used in furniture (Lowell Center for Sustainable Production, 2011). We also explored the correlations between PEs, APs and OPEs measured in our previous study (Demirtepe et al., 2019). We found weak significant correlations for some compounds, such as tri-n-butyl phosphate correlated with DEP ($r_s = 0.28$, $p < 0.05$) and DMP ($r_s = 0.40$, $p < 0.01$) which have common uses as plasticizers in cellulose lacquers, plastics, and vinyl resins (Lyche, 2017; PubChem, 2020).

A questionnaire provided ancillary information on characteristics of the Slovakian homes. We hypothesized that presence of PVC flooring would result in higher PE concentrations in dust, as has been previously noted (Bornehag et al., 2005). However, no significant difference in individual PEs and total PE concentrations was found between homes with ($n = 17$) and without ($n = 33$) PVC flooring. A possible reason can be that the contribution of PVC flooring to PEs found in the dust may be limited compared to other PE sources in homes, such as furniture, carpets, wires and cables, packaging materials, etc. (ATSDR, 2019, 2001; Lowell Center for Sustainable Production, 2011). On the other hand, we found that DnBP, DnOP, DEHP, $\sum$PEs, ATBC and DEHT had significantly greater concentrations in homes with carpeting ($n = 8$) than homes without carpeting ($n = 42$) (Mann-Whitney U test, $p < 0.05$); these compounds are used in carpets (ATSDR, 2001, 1997; US EPA, 2019), which may be acting as a source to indoor dust. Lastly, we explored the correlations with building age. Only DMP ($r_s = 0.44$, $p < 0.05$) and DEP ($r_s = 0.35$, $p < 0.05$) correlated with building age, having greater concentrations in older homes. Additionally, DMP concentrations were significantly higher in homes more than 45 years old than homes less than 45 years old (one-way ANOVA, $F = 2.96$, $p < 0.05$), which we attribute to past use of DMP as a solvent in insecticides (Lyche, 2017).

3.3. Exposure and toxicity assessment of PEs and APs

PEs are known or suspected to have a range of health effects including altered immune system, developmental and reproductive effects, endocrine disruption, liver and kidney toxicity (ATSDR, 2019; Mitro et al., 2016), among which developmental and reproductive effects have been the most studied on laboratory animals and also in humans (Johnson et al., 2012; Kay et al., 2014; Wang et al., 2019). On the other hand, the toxicological information for APs is relatively scarce. Studies have examined their developmental and reproductive toxicity potentials and carcinogenicity, and APs have DNELs (ECHA, 2018). We first estimated exposure to PEs and APs and compared them with respect to their individual toxicity; we then evaluated the mixture effects of PEs via a cumulative risk assessment (CRA).

The available toxicity reference values (TRVs) for PEs and APs are given in Table S4. Comparison of median TRVs for PEs revealed the highest toxicity for DEHP, followed by DnOP, DnBP, DCHP, BBP, DEP, DMP, while APs have higher TRVs than PEs except for DEP and DMP (Table S4). DEHP has a large range of TRVs reported, which might be due to different endpoints used, e.g. ATSDR provided an MRL of 0.1 μg/kg/d for a developmental endpoint and IRIS provided RfD of 20 μg/kg/d for increase in liver weight. Additionally, structure-activity relationship (SAR) studies showed that straight-chain PEs with four to seven carbons have higher potency for developmental and reproductive toxicity, and branching of the side chain and unsaturation of the side chain increases the potency, while cyclic side chain does not (Li et al., 2019). Accordingly, DHP, DPP, DnBP should have higher toxic potency among the straight chain PEs, while DiBP and DEHP should have higher potency among branched chain PEs (Li et al., 2019). BBP, having a mixed carbon chain in the structure, is equipotent to DnBP and DEHP (Howdeshell et al., 2008). On the other hand, DnOP does not have developmental and reproductive toxicity at the highest doses tested (Li et al., 2019), although has the second lowest median TRV. Hence, usage of multiple endpoints in different toxicity assessments creates uncertainty in evaluating human risk.

In our previous study, we used TRVs derived from literature and regulatory sources to evaluate relative toxicities of SVOCs and merged this information with indoor exposure estimates for prioritization of compound risks (Demirtepe et al., 2019). Using this previously developed framework, we added PEs and APs into the evaluation from Demirtepe et al. (2019) (Fig. 2) to compare exposure and toxicity of PEs and APs to other SVOCs via dust ingestion. The estimated intakes for median and high intake scenarios and for a child and an adult male are provided in Table S7 for individual compounds. The human intake via dust ingestion for a child ranged between <0.01 ng/kg/d for DPP and 211 ng/kg/d for DEHP according to the median intake scenario. DEHP had a two order of magnitude higher intake than tris(2-butoxyethyl) phosphate, which had the highest intake via dust ingestion (1.62 ng/kg/d) among previously reported SVOCs from these same samples (which included OPEs, PCBs, PAHs, OCPs and halogenated FRs) (Demirtepe et al., 2019). PEs and APs have higher TRVs (indicating lower toxicity) than PCBs,

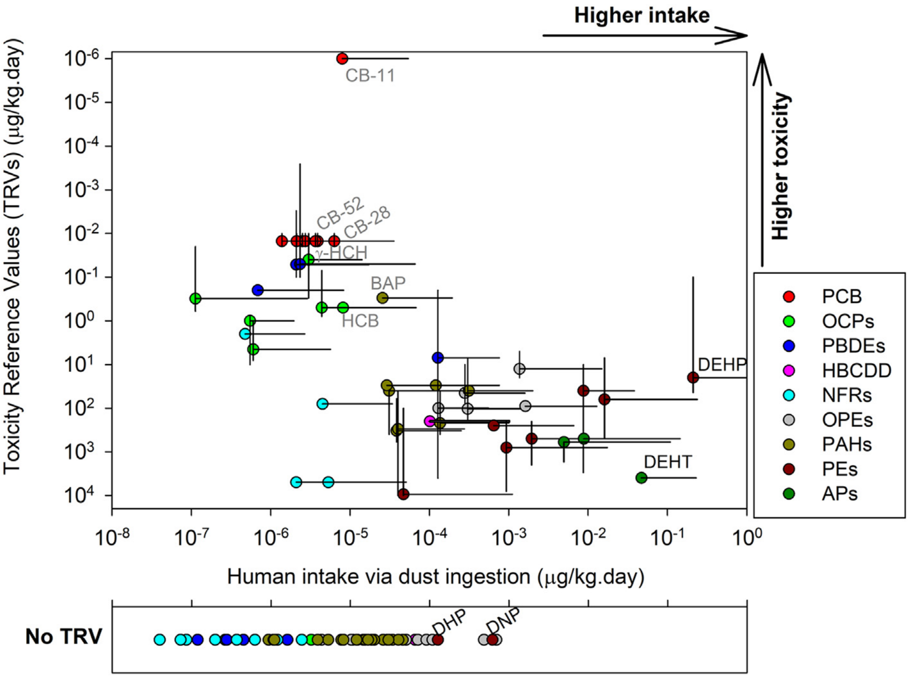

Fig. 2. Human intake via dust ingestion vs toxicity reference values of indoor SVOCs, PEs and APs. Vertical lines represent the range of TRVs reported in the literature, horizontal lines represent the high intake scenario, and points represent median exposure scenario and median TRVs. The box below the graph shows compounds with no available TRVs. Modified from Demirtepe et al. (2019).

OCPs and PBDEs, but similar TRVs to many PAHs, OPEs and NFRs (Fig. 2). Hence, although PEs and APs had the highest human intake among SVOCs, this assessment did not place them among the high priority compounds for indoor risk assessment, although, as with the OPEs, the uncertainty regarding the TRVs combined with their high exposures suggests potential for concern.

Moreover, co-occurrence of PEs in the indoor environment leads to human exposure to multiple PEs with possible common health effects. Several PEs affect the reproductive development of fetal male rat via a common mode of toxicity (Howdeshell et al., 2008), and studies on the mixture effects of PEs showed that the effect was best predicted by dose addition (Hannas et al., 2011; Howdeshell et al., 2008). Hence, CRA was employed to understand the risks associated with the total PE contamination of indoor dust. We focused on additive anti-androgenic effects, and calculated the cumulative hazard quotients (ΣHQ) given in Eq. (1) by using the Tier 2 approach presented by Pelletier et al. (2018).

ΣHQs were calculated for a child and an adult male for median and high intake scenarios using the RPFs available for five PEs (Table 1). ΣHQs were < 1 for each set of RPFs for a child and an adult male for both scenarios. ΣHQ values calculated by using RPFs from different studies given in Table 1 were on the same order of magnitude, except for Hannas et al. (2011) in which only two RPFs were available for PEs. Percent contribution of PEs to ΣHQs was highest for DEHP, i.e. more than 89% for all set of RPFs, followed by DnBP (Fig. 3). This was expected since DEHP is the dominant PE and has higher RPF than other PEs. Overall, the CRA identified no unacceptable risks for children and male adults regarding anti-androgenic effects due to indoor exposure via dust ingestion.

Some uncertainties and limitations are associated with use of the CRA. First, the CRA results represented only indoor dust ingestion exposure pathway, while dermal contact with dust and inhalation of indoor air might also contribute to total human exposure to PEs. Additionally, other SVOCs having anti-androgenic effects, such as p,p'-DDE, BDE-99 (Kortenkamp and Faust, 2010), and benzo(*a*)pyrene (Fournier et al., 2016) were identified in the same dust samples but were not included in this CRA since the first two have no RPFs available. Yet these compounds should also contribute to the ΣHQs. Another important contribution might come from DiBP, which has RPFs ranging 0.15–1.00 (Benson, 2009; German Federal Environment Agency, 2011; Hannas et al., 2011; Howdeshell et al., 2008). However, DiBP was not quantified in this study due to high background contamination.

Second, the RPFs derived from different studies may be incomparable since the experimental parameters and uncertainty factors used to derive reference and benchmark doses might be different (Benson, 2009; Fournier et al., 2016). Subsequently, the differences in specific endpoints such as the decrease in fetal testosterone vs. reduced reproductive organ weights might create uncertainty in comparability of data. However, the variation in RPFs did not lead to large variations among calculated HQs since daily exposure dose to DEHP was one to four orders of magnitude higher than other PEs, i.e. HQs were dominated by DEHP intake. It is also important to note that the median of available TRVs for DEHP was used as RfD_{DEHP} in the calculation of HQs (Eq. (1)). If we had used the minimum of reported TRVs, which is 0.1 μg/kg/d, and is also the most recent reported value (ATSDR, 2019), we would have obtained HQs > 1. Therefore, uncertainty in the TRVs is an important factor impacting HQ calculation and the CRA. Data available for many compounds were inconsistent, e.g. DEP was found 21 times more potent than DEHP according to Fournier et al. (2016) but zero potent according to most of the studies in Table 1. Positive correlations were found between DEP exposure and effects in human studies, but no associations were found in most of the animal studies, which might be explained by co-exposure to other PEs in human studies (NRC, 2008).

This highlights the importance of placing indoor exposure measurements into context by acknowledging the differing toxicities and toxic endpoints of individual compounds, at minimum through a combined assessment of exposures and TRVs (e.g., Fig. 2), but ideally incorporating the importance of mixture effects through use of HQs or a similar technique. Yet in all aspects, the biggest limitation at present remains the quality and comparability of TRVs, which are lacking for many compounds, or, when present, often have order-of-magnitude ranges, leading to similar uncertainty in the risk evaluations for which they are used.

3.4. Screening for other suspect PEs

Approximately 40 PEs are considered important for monitoring due to their use and HPV (Staples, 2003), however fewer are regularly monitored. This may be due to the limitations on standard mixtures, instrumental detection and sample contamination such as observed for DiBP in this study. All sample data in this study was collected on a GC-Orbitrap operating in full scan, allowing for screening of other PEs not included in the target analysis. MS-DIAL software was used to identify peaks, deconvolute and group the samples providing the opportunity to assess the number of peaks which are suspect PEs according to common PE feature masses. In total 5411 features were identified by MS-DIAL. After removing features reflecting background contamination, and selecting only the 10 most abundant ions, the features were filtered for target feature masses (149.0226, 163.0382 and 225.0552, +/− 0.0003 amu), which resulted in 63 features. Of the three masses, only the 149.0226 mass had any detectable peaks. Removing those features with low frequency of detection, i.e. < 10% of samples, 48 suspect compounds remained. Based on ion masses and retention times, seven of

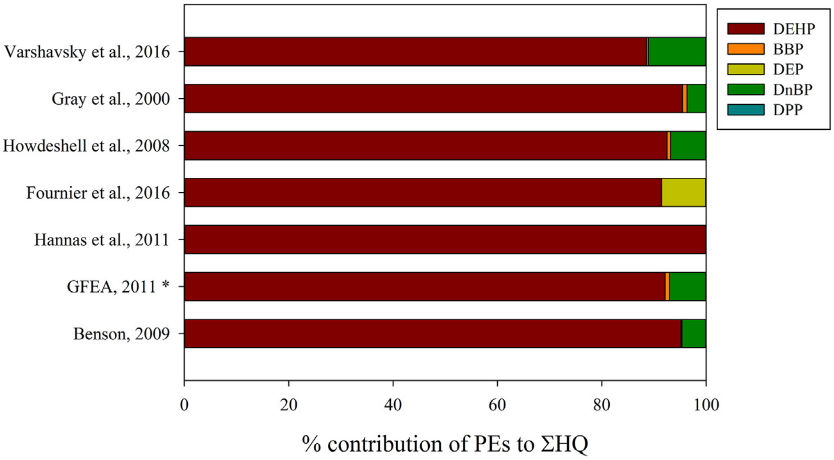

Fig. 3. Percent contribution of PEs to ΣHQs calculated by using RPFs from studies given in the y-axis. * GFEA refers to German Federal Environment Agency.

these were found to correspond to the target PEs already included in the study, leaving 41 suspect PEs with the protonated phthalic anhydride structure.

A list of 65 candidate PEs (Table S8) was generated from literature including the compounds detected by target analysis ($n = 13$ target compounds; Table S1). Many more PEs have been published but were excluded if no published retention time index (RTI) or retention time (RT) were available and if no GC spectra had been published. From the list of PEs, 54 had published n-alkane scale RTIs and for the majority also multiple published RTs were available. Additionally, the methods using a similar GC column of the current study (30 m with 5% diphenyl) were given priority. For the remaining compounds, a linear regression analysis between the RT of corresponding compounds in our target method and the method in the literature was used to estimate an RTI value and RT. For most of the remaining compounds more than two published RTs were used in estimation of RTI.

Using the estimated RT, 12 additional PEs were tentatively identified (1,2-benzenedicarboxylic acid monopentyl ester (MPeP), 2-{[(4-methylpentyl)oxy]carbonyl}benzoate, bis(2-ethylhexyl) *iso*phthalate, bis(4-methyl-2-pentyl) phthalate isomer (BMPP), butyl *cyclo*hexyl phthalate, diallyl phthalate (DAP), di-*n*-heptyl phthalate, di-isoheptyl phthalate, di-isononyl phthalate (DINP), hexyl decyl phthalate, monomethyl phthalate (MMP), octyl hydrogen phthalate (MnOP)). As many PEs have the same masses (149, 150, 105 and 104), confirmation ions may be of relatively low intensity compared to the primary ion, and many of the protonated phthalic anhydride masses detected were within 10–20s of each other, a more in-depth study would be needed to confirm identification.

Among the tentatively identified PEs, a few are particularly notable. For example, DINP is produced and imported annually between 10^5 and 10^6 t in EEA (ECHA, 2018) and is restricted for certain uses (REACH, 2018). Additionally, DINP is an endocrine disrupting chemical at high doses, although it has a lower toxic potential than others such as DPP and DEHP (Hannas et al., 2011). Di-*n*-heptyl phthalate and di-isoheptyl phthalate have lower production volumes, but a higher toxic potential than some of our target PEs. SAR studies found that di-*n*-heptyl phthalate and di-isoheptyl phthalate have higher potency for developmental toxicity and testosterone production in fetal testis (Li et al., 2019). These compounds may be of concern if they have high human intake via dust ingestion, and if they were included in overall assessments of PE mixture toxicity, the mixture toxicity would be higher. Although we have estimated a low risk based on the set of target analytes in our study, the presence of more than 40 suspect PEs suggests an underestimation of both human exposure to PEs and the associated PE mixture toxicity. This suggests for a more complete evaluation of PE exposure and toxicity, PEs with HPV and higher toxicity potential should be prioritized in lists of targeted analysis for indoor dust and other environmental matrices.

4. Conclusion

This study reports the results of targeted (12 PEs and five APs) and suspect screening of plasticizers in indoor residential dust. Dust concentrations of APs were comparable to, or in some homes even higher than, the concentrations of PEs. While we found no association of plasticizer concentrations in dust with presence of PVC flooring, homes with carpeting had higher concentrations of PEs and APs than homes without, suggesting carpeting as an important source of plasticizers in dust. We also found significant correlations between DMP and DEP and building age, which might be due to past use of these LMW PEs as insecticides in older homes.

We evaluated the risk of indoor plasticizer exposure via dust ingestion by combining toxicity and human intake estimates. DEHP was found to have the highest human intake but a lower toxicity compared to other SVOCs, e.g. PCBs and OCPs. However, a cumulative anti-androgenic risk assessment is more meaningful since mixture effects can prevail when multiple PEs with a common mode of toxicity occur. CRA estimated acceptable risks for children and male adults, however the wide range of reported TRVs is a large source of uncertainty and the biggest limitation in risk evaluation and can directly change whether the CRA crosses the hazard threshold. Furthermore, inclusion of other exposure routes, especially dietary intake, would lead to higher hazard quotients associated with PE intake. Lastly, more than 40 suspect PEs were identified through suspect screening, suggesting higher mixture toxicity if a larger set of PEs is considered. All PEs with high production volumes and high toxicity potential are recommended to be quantified in indoor studies, together with APs which have comparable indoor levels to PEs.

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CRediT authorship contribution statement

Hale Demirtepe: Investigation, Formal analysis, Writing – original draft, Writing – review & editing. **Lisa Melymuk:** Writing – review & editing, Supervision. **Garry Codling:** Methodology, Writing – review & editing. **Ľubica Palkovičová Murínová:** Supervision, Funding acquisition, Writing – review & editing. **Denisa Richterová:** Investigation, Resources, Writing – review & editing. **Vladimíra Rašplová:** Investigation, Resources, Writing – review & editing. **Tomáš Trnovec:** Conceptualization, Supervision, Funding acquisition. **Jana Klánová:** Conceptualization, Supervision, Funding acquisition, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2021.146667.

References

Ait Bamai, Y., Araki, A., Kawai, T., Tsuboi, T., Saito, I., Yoshioka, E., Cong, S., Kishi, R., 2016. Exposure to phthalates in house dust and associated allergies in children aged 6–12 years. Environ. Int. 96, 16–23. https://doi.org/10.1016/j.envint.2016.08.025.

Amin, M.M., Ebrahimpour, K., Parastar, S., Shoshtari-Yeganeh, B., Hashemi, M., Mansourian, M., Poursafa, P., Fallah, Z., Rafiei, N., Kelishadi, R., 2018. Association of urinary concentrations of phthalate metabolites with cardiometabolic risk factors and obesity in children and adolescents. Chemosphere 211, 547–556. https://doi.org/10.1016/j.chemosphere.2018.07.172.

ATSDR, 1995. Toxicological Profile for Diethyl Phthalate. Atlanta, Georgia.

ATSDR, 1997. Toxicological Profile for Di-n-Octylphthalate. Atlanta, Georgia.

ATSDR, 2001. Toxicological Profile for Di-n-Butyl Phthalate. Atlanta, Georgia.

Agency for Toxic Substances and Disease Registry (ATSDR), 2019. Toxicological profile for Di(2-ethylhexyl)phthalate (DEHP). U.S. Department of Health and Human Services, Public Health Service, Atlanta, GA.

Bekö, G., Callesen, M., Weschler, C.J., Toftum, J., Langer, S., Sigsgaard, T., Høst, A., Kold Jensen, T., Clausen, G., 2015. Phthalate exposure through different pathways and allergic sensitization in preschool children with asthma, allergic rhinoconjunctivitis and atopic dermatitis. Environ. Res. 137, 432–439. https://doi.org/10.1016/j.envres.2015.01.012.

Benson, R., 2009. Hazard to the developing male reproductive system from cumulative exposure to phthalate esters-dibutyl phthalate, diisobutyl phthalate, butylbenzyl phthalate, diethylhexyl phthalate, dipentyl phthalate, and diisononyl phthalate. Regul. Toxicol. Pharmacol. 53, 90–101. https://doi.org/10.1016/j.yrtph.2008.11.005.

Bergh, C., Torgrip, R., Emenius, G., Östman, C., 2011. Organophosphate and phthalate esters in air and settled dust - a multi-location indoor study. Indoor Air 21, 67–76. https://doi.org/10.1111/j.1600-0668.2010.00684.x.

Bertelsen, R.J., LØdrup Carlsen, K.C., Calafat, A.M., Hoppin, J.A., Håland, G., Mowinckel, P., Carlsen, K.H., LØvik, M., 2013. Urinary biomarkers for phthalates associated with asthma in Norwegian children. Environ. Health Perspect. 121, 251–256. https://doi.org/10.1289/ehp.1205256.

Bornehag, C.G., Lundgren, B., Weschler, C.J., Sigsgaard, T., Hagerhed-Engman, L., Sundell, J., 2005. Phthalates in indoor dust and their association with building characteristics. Environ. Health Perspect. 113, 1399–1404. https://doi.org/10.1289/ehp.7809.

Bui, T.T., Giovanoulis, G., Cousins, A.P., Magnér, J., Cousins, I.T., de Wit, C.A., 2016. Human exposure, hazard and risk of alternative plasticizers to phthalate esters. Sci. Total Environ. 541, 451–467. https://doi.org/10.1016/j.scitotenv.2015.09.036.

Christia, C., Poma, G., Harrad, S., de Wit, C.A., Sjostrom, Y., Leonards, P., Lamoree, M., Covaci, A., 2019. Occurrence of legacy and alternative plasticizers in indoor dust from various EU countries and implications for human exposure via dust ingestion and dermal absorption. Environ. Res. 171, 204–212. https://doi.org/10.1016/j.envres.2018.11.034.

CPSC, 2017. 16 CFR Part 1307 Prohibition of Children's Toys and Child Care Articles Containing Specified Phthalates.

Demirtepe, H., Melymuk, L., Diamond, M.L., Bajard, L., Vojta, Š., Prokeš, R., Sáňka, O., Klánová, J., Palkovičová Murínová, Ľ., Richterová, D., Rašplová, V., Trnovec, T., 2019. Linking past uses of legacy SVOCs with today's indoor levels and human exposure. Environ. Int. 127, 653–663. https://doi.org/10.1016/j.envint.2019.04.001.

ECHA, 2018. European Chemicals Agency - Information on Chemicals [WWW Document]. URL https://echa.europa.eu/brief-profile/-/briefprofile/100.043.099 (accessed 9.8.18).

ECHA, 2020. Candidate List of Substances of Very High Concern for Authorisation [WWW Document]. URL https://echa.europa.eu/candidate-list-table (accessed 6.1.20).

Fournier, K., Tebby, C., Zeman, F., Glorennec, P., Zmirou-Navier, D., Bonvallot, N., 2016. Multiple exposures to indoor contaminants: derivation of benchmark doses and relative potency factors based on male reprotoxic effects. Regul. Toxicol. Pharmacol. 74, 23–30. https://doi.org/10.1016/j.yrtph.2015.11.017.

German Federal Environment Agency, 2011. Substance Monograph: Phthalates – New and Updated Reference Values for Monoesters and Oxidised Metabolites in Urine of Adults and Children.

Giovanoulis, G., Bui, T., Xu, F., Papadopoulou, E., Padilla-Sanchez, J.A., Covaci, A., Haug, L.S., Cousins, A.P., Magnér, J., Cousins, I.T., de Wit, C.A., 2018. Multi-pathway human exposure assessment of phthalate esters and DINCH. Environ. Int. 112, 115–126. https://doi.org/10.1016/j.envint.2017.12.016.

Gray, L.E., Ostby, J., Furr, J., Price, M., Veeramachaneni, D.N.R., Parks, L., 2000. Perinatal exposure to the phthalates DEHP, BBP, and DINP, but not DEP, DMP, or DOTP, alters sexual differentiation of the male rat. Toxicol. Sci. 58, 350–365. https://doi.org/10.1093/toxsci/58.2.350.

Hammel, S.C., Levasseur, J.L., Hoffman, K., Phillips, A.L., Lorenzo, A.M., Calafat, A.M., Webster, T.F., Stapleton, H.M., 2019. Children's exposure to phthalates and non-phthalate plasticizers in the home: the TESIE study. Environ. Int. 132, 105061. https://doi.org/10.1016/j.envint.2019.105061.

Hannas, B.R., Lambright, C.S., Furr, J., Howdeshell, K.L., Wilson, V.S., Gray, L.E., 2011. Dose-response assessment of fetal testosterone production and gene expression levels in rat testes following in utero exposure to diethylhexyl phthalate, diisobutyl phthalate, diisoheptyl phthalate, and diisononyl phthalate. Toxicol. Sci. 123, 206–216. https://doi.org/10.1093/toxsci/kfr146.

Hauser, R., Calafat, A.M., 2005. Phthalates and human health. Occup. Environ. Med. 62, 806–818. https://doi.org/10.1136/oem.2004.017590.

He, R., Li, Y., Xiang, P., Li, C., Zhou, C., Zhang, S., Cui, X., Ma, L.Q., 2016. Organophosphorus flame retardants and phthalate esters in indoor dust from different microenvironments: bioaccessibility and risk assessment. Chemosphere 150, 528–535. https://doi.org/10.1016/j.chemosphere.2015.10.087.

Howdeshell, K.L., Wilson, V.S., Furr, J., Lambright, C.R., Rider, C.V., Blystone, C.R., Hotchkiss, A.K., Gray, L.E., 2008. A mixture of five phthalate esters inhibits fetal testicular testosterone production in the Sprague-Dawley rat in a cumulative, dose-additive manner. Toxicol. Sci. 105, 153–165. https://doi.org/10.1093/toxsci/kfn077.

Jílková, S., Melymuk, L., Vojta, Š., Vykoukalová, M., Bohlin-Nizzetto, P., Klánová, J., 2018. Small scale spatial variability of flame retardants in indoor dust and implications for dust sampling. Chemosph. 206, 132–141. https://doi.org/10.1016/j.chemosphere.2018.04.146.

Johnson, K.J., Heger, N.E., Boekelheide, K., 2012. Of mice and men (and rats): phthalate-induced fetal testis endocrine disruption is species-dependent. Toxicol. Sci. 129, 235–248. https://doi.org/10.1093/toxsci/kfs206.

Kademoglou, K., Giovanoulis, G., Palm-Cousins, A., Padilla-Sanchez, J.A., Magnér, J., De Wit, C.A., Collins, C.D., 2018. In vitro inhalation bioaccessibility of phthalate esters and alternative plasticizers present in indoor dust using artificial lung fluids. Environ. Sci. Technol. Lett. 5, 329–334. https://doi.org/10.1021/acs.estlett.8b00113.

Kay, V.R., Bloom, M.S., Foster, W.G., 2014. Reproductive and developmental effects of phthalate diesters in males. Crit. Rev. Toxicol. 44, 467–498. https://doi.org/10.3109/10408444.2013.875983.

Kishi, R., Ketema, R.M., Ait Bamai, Y., Araki, A., Kawai, T., Tsuboi, T., Saito, I., Yoshioka, E., Saito, T., 2018. Indoor environmental pollutants and their association with sick house syndrome among adults and children in elementary school. Build. Environ. 136, 293–301. https://doi.org/10.1016/j.buildenv.2018.03.056.

Kortenkamp, A., Faust, M., 2010. Combined exposures to anti-androgenic chemicals: steps towards cumulative risk assessment. Int. J. Androl. 33, 463–472. https://doi.org/10.1111/j.1365-2605.2009.01047.x.

Kutz, M., 2017. Applied Plastics Engineering Handbook. Elsevier, Second Edi. ed.

Li, X., Mo, J., Zhu, Q., Ni, C., Wang, Y., Li, H., Lin, Z. kun, Ge, R.S., 2019. The structure–activity relationship (SAR) for phthalate-mediated developmental and reproductive toxicity in males. Chemosphere 223, 504–513. doi:https://doi.org/10.1016/j.chemosphere.2019.02.090.

Lowell Center for Sustainable Production, 2011. Phthalates and their Alternatives: Health and Environmental Concerns (Lowell).

Luongo, G., Östman, C., 2016. Organophosphate and phthalate esters in settled dust from apartment buildings in Stockholm. Indoor Air 26, 414–425. https://doi.org/10.1111/ina.12217.

Lyche, J.L., 2017. Phthalates. In: Gupta, R. (Ed.), Reproductive and Developmental Toxicology. Academic Press.

Meeker, J.D., Calafat, A.M., Hauser, R., 2009. Urinary metabolites of di(2-ethylhexyl) phthalate are associated with decreased steroid hormone levels in adult men. J. Androl. 30, 287–297. https://doi.org/10.2164/jandrol.108.006403.

Mitro, S.D., Dodson, R.E., Singla, V., Adamkiewicz, G., Elmi, A.F., Tilly, M.K., Zota, A.R., 2016. Consumer product chemicals in indoor dust: a quantitative meta-analysis of U.S. Studies. Environ. Sci. Technol. 50, 10661–10672. https://doi.org/10.1021/acs.est.6b02023.

Nagorka, R., Conrad, A., Scheller, C., Süßenbach, B., Moriske, H.J., 2011. Diisononyl 1,2-cyclohexanedicarboxylic acid (DINCH) and Di(2-ethylhexyl) terephthalate (DEHT) in indoor dust samples: concentration and analytical problems. Int. J. Hyg. Environ. Health 214, 26–35. https://doi.org/10.1016/j.ijheh.2010.08.005.

Narvaez Rincon, P.C., Suarez Palacios, O.Y., 2015. Plasticizers. In: Palsule, S. (Ed.), Encyclopedia of Polymers and Composites. Springer, Berlin Heidelberg, pp. 1–13 https://doi.org/10.1007/978-3-642-37179-0_73-1.

Nayef, N., Qasmi, A., Al-thaiban, H., Helaleh, M.I.H., 2019. Indoor phthalates from household dust in Qatar : implications for non-dietary human exposure. Environ. Sci. Pollut. Res. 26, 421–430. https://doi.org/10.1007/s11356-018-3604-8.

Net, S., Sempéré, R., Delmont, A., Paluselli, A., Ouddane, B., 2015. Occurrence, fate, behavior and ecotoxicological state of phthalates in different environmental matrices. Environ. Sci. Technol. 49, 4019–4035. https://doi.org/10.1021/es505233b.

NRC, 2008. Phthalates and Cumulative Risk Assessment: The Tasks Ahead. National Academies Press, Washington, DC.

Pelletier, M., Glorennec, P., Mandin, C., Le Bot, B., Ramalho, O., Mercier, F., Bonvallot, N., 2018. Chemical-by-chemical and cumulative risk assessment of residential indoor exposure to semivolatile organic compounds in France. Environ. Int. 117, 22–32. https://doi.org/10.1016/j.envint.2018.04.024.

Plastics Europe, 2020. Plastics - the facts 2020.

Promtes, K., Kaewboonchoo, O., Kawai, T., Miyashita, K., Panyapinyopol, B., Kwonpongsagoon, S., Takemura, S., 2019. Human exposure to phthalates from house dust in Bangkok. Thailand. J. Environ. Sci. Heal. - Part A Toxic/Hazardous Subst. Environ. Eng. 54, 1269–1276. https://doi.org/10.1080/10934529.2019.1637207.

PubChem, 2020. U.S. National Library of Medicine National Center for Biotechnology Information [WWW Document]. URL https://pubchem.ncbi.nlm.nih.gov/ (accessed 1.31.20).

Radke, E.G., Glenn, B.S., Braun, J.M., Cooper, G.S., 2019. Phthalate exposure and female reproductive and developmental outcomes: a systematic review of the human epidemiological evidence. Environ. Int. 130, 104580. https://doi.org/10.1016/j.envint.2019.02.003.

REACH, 2018. Commission Regulation (EU) 2018/2005 of 17 December 2018 Amending Annex XVII to Regulation (EC) no 1907/2006 of the European Parliament and of the Council Concerning the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH) as Regar.

Sedtasiriphokin, N., Supornsilchai, V., Jantarat, C., Nosoongnoen, W., 2017. Phthalate exposure in Thai children and adolescents. Asian Biomed. 11, 343–352. https://doi.org/10.1515/abm-2018-0006.

Shin, H.M., Moschet, C., Young, T.M., Bennett, D.H., 2020. Measured concentrations of consumer product chemicals in California house dust: implications for sources, exposure, and toxicity potential. Indoor Air 30, 60–75. https://doi.org/10.1111/ina.12607.

Stanley, M.K., Robillard, K.A., Staples, C.A., 2003. Introduction. In: Staples, C.A. (Ed.), Phthalate Esters - Handbook of Environmental Chemistry. Springer, Berlin Heidelberg, pp. 1–7.

Staples, C.A., 2003. Phthalate esters. Handbook of Environmental Chemistry. Springer. https://doi.org/10.1007/b11460.

Subedi, B., Sullivan, K.D., Dhungana, B., 2017. Phthalate and non-phthalate plasticizers in indoor dust from childcare facilities, salons, and homes across the USA. Environ. Pollut. 230, 701–708. https://doi.org/10.1016/j.envpol.2017.07.028.

Swan, S.H., Main, K.M., Liu, F., Stewart, S.L., Kruse, R.L., Calafat, A.M., Mao, C.S., Redmon, J.B., Ternand, C.L., Sullivan, S., Teague, J.L., The Study for Future Families Research Team, 2005. Decrease in anogenital distance among male infants with prenatal phthalate exposure. Environ. Health Perspect. 113, 1056–1061. https://doi.org/10.1289/ehp.8100.

Tang, B., Christia, C., Malarvannan, G., Liu, Y.E., Luo, X.J., Covaci, A., Mai, B.X., Poma, G., 2020. Legacy and emerging organophosphorus flame retardants and plasticizers in indoor microenvironments from Guangzhou, South China. Environ. Int. 143, 105972. https://doi.org/10.1016/j.envint.2020.105972.

Tsugawa, H., Cajka, T., Kind, T., Ma, Y., Higgins, B., Ikeda, K., Kanazawa, M., VanderGheynst, J., Fiehn, O., Arita, M., 2015. MS-DIAL: data independent MS/MS deconvolution for comprehensive metabolome analysis. Nat. Methods 12, 523–526. https://doi.org/10.1038/nmeth.3393.

U.S. EPA, 2011. Exposure Factors Handbook 2011 Edition (Final Report). Washington, DC.

US EPA, 2019. Proposed Designation of Di-Ethylhexyl Phthalate (DEHP) (1,2-Benzene- Dicarboxylic Acid, 1,2-Bis (2-Ethylhexyl) Ester) (CASRN 117-81-7) as a High-Priority Substance for Risk Evaluation.

Varshavsky, J.R., Zota, A.R., Woodruff, T.J., 2016. A novel method for calculating potency-weighted cumulative phthalates exposure with implications for identifying racial/ethnic disparities among U.S. reproductive-aged women in NHANES 2001-2012. Environ. Sci. Technol. 50, 10616–10624. https://doi.org/10.1021/acs.est.6b00522.

Venier, M., Audy, O., Vojta, Š., Bečanová, J., Romanak, K., Melymuk, L., Krátká, M., Kukučka, P., Okeme, J., Saini, A., Diamond, M.L., Klánová, J., 2016. Brominated flame retardants in the indoor environment - comparative study of indoor contamination from three countries. Environ. Int. 94, 150–160. https://doi.org/10.1016/j.envint.2016.04.029.

Vojta, Š., Melymuk, L., Klánová, J., 2017. Changes in flame retardant and legacy contaminant concentrations in indoor air during building construction, furnishing, and use. Environ. Sci. Technol. 51, 11891–11899. https://doi.org/10.1021/acs.est.7b03245.

Vykoukalová, M., Venier, M., Vojta, Š., Melymuk, L., Bečanová, J., Romanak, K., Prokeš, R., Okeme, J.O., Saini, A., Diamond, M.L., Klánová, J., 2017. Organophosphate esters flame retardants in the indoor environment. Environ. Int. 106, 97–104. https://doi.org/10.1016/j.envint.2017.05.020.

Wang, Y., Ni, C., Li, X., Lin, Z., Zhu, Q., Li, L., Ge, R.S., 2019. Phthalate-induced fetal leydig cell dysfunction mediates male reproductive tract anomalies. Front. Pharmacol. 10, 1–11. https://doi.org/10.3389/fphar.2019.01309.

Wormuth, M., Scheringer, M., Vollenweider, M., Hungerbühler, K., 2006. What are the sources of exposure to eight frequently used phthalic acid esters in Europeans? Risk Anal. 26, 803–824. https://doi.org/10.1111/j.1539-6924.2006.00770.x.

Yang, C., Harris, S.A., Jantunen, L.M., Siddique, S., Kubwabo, C., Tsirlin, D., Latifovic, L., Fraser, B., St-Jean, M., De La Campa, R., You, H., Kulka, R., Diamond, M.L., 2019. Are cell phones an indicator of personal exposure to organophosphate flame retardants and plasticizers? Environ. Int. 122, 104–116. https://doi.org/10.1016/j.envint.2018.10.021.

Yang, C., Harris, S.A., Jantunen, L.M., Kvasnicka, J., Nguyen, L.V., Diamond, M.L., 2020. Phthalates: relationships between air, dust, electronic devices, and hands with implications for exposure. Environ. Sci. Technol. https://doi.org/10.1021/acs.est.0c00229.

Zhang, Qinghao, Sun, Y., Zhang, Qingnan, Hou, J., Wang, P., Kong, X., Sundell, J., 2020. Phthalate exposure in Chinese homes and its association with household consumer products. Sci. Total Environ. 719, 136965. https://doi.org/10.1016/j.scitotenv.2020.136965.

APPENDIX 15

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Application of land use regression modelling to describe atmospheric levels of semivolatile organic compounds on a national scale

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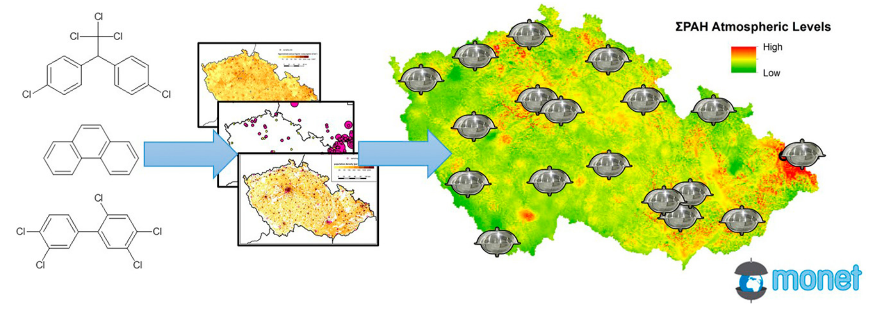
HIGHLIGHTS

- Spatial models developed from atmospheric levels and GIS data at 29 sampling sites
- PAHs related to fuel consumption, industrial sources, and topography ($R^2 = 0.68$)
- PCBs related to elevation and soil concentrations ($R^2 = 0.62$)
- National maps created from PAH and PCB models identified pollutant hotspots

GRAPHICAL ABSTRACT



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ABSTRACT

Despite the success of passive sampler-based monitoring networks in capturing global atmospheric distributions of semivolatile organic compounds (SVOCs), their limited spatial resolution remains a challenge. Adequate spatial coverage is necessary to better characterize concentration gradients, identify point sources, estimate human exposure, and evaluate the effectiveness of chemical regulations such as the Stockholm Convention on Persistent Organic Pollutants. Land use regression (LUR) modelling can be used to integrate land use characteristics and other predictor variables (industrial emissions, traffic intensity, demographics, etc.) to describe or predict the distribution of air concentrations at unmeasured locations across a region or country. While LUR models are frequently applied to data-rich conventional air pollutants such as particulate matter, ozone, and nitrogen oxides, they are rarely applied to SVOCs.

The MONET passive air sampling network (RECETOX, Masaryk University) continuously measures atmospheric SVOC levels across Czechia in monthly intervals. Using monitoring data from 29 MONET sites over a two-year period (2015–2017) and a variety of predictor variables, we developed LUR models to describe atmospheric levels and identify sources of polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs) and DDT across the country. Strong and statistically significant ($R^2 > 0.6$; $p < 0.05$) models were derived for PAH and PCB levels on a national scale. The PAH model retained three predictor variables – heating emissions represented by domestic fuel consumption, industrial PAH point sources, and the hill:valley index, a measure of site topography. The PCB model retained two predictor variables – site elevation, and secondary sources of PCBs represented by soil concentrations. These models were then applied to Czechia as a whole, highlighting the spatial variability of atmospheric SVOC levels, and providing a tool that can be used for further optimization of sampling network design, as well as evaluating potential human and environmental chemical exposures.

1. Introduction

Semivolatile organic compounds (SVOCs) are a broad class of atmospheric pollutants that include polycyclic aromatic hydrocarbons

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(PAHs), polychlorinated biphenyls (PCBs), and organochlorine pesticides (OCPs). An ongoing challenge in monitoring SVOCs in many regions is the lack of sufficient spatial coverage to characterize atmospheric concentration gradients, identify sources, evaluate human exposure, and assess the effectiveness of national and international regulations such as the Stockholm Convention on Persistent Organic Pollutants (POPs). While concentrations of conventional air quality pollutants such as nitrogen oxides (NO_X), ozone (O_3), and particulate matter (PM) can be rapidly measured in "on-line" mode resulting in continuous monitoring, SVOCs require complex equipment and daily/weekly time resolution for active air sampling. Widespread use of passive air sampling for SVOCs has helped to improve spatial coverage, providing a reliable technique to quantify SVOCs across different site types and conditions without the use of complex equipment (Bohlin-Nizzetto et al., 2020; Herkert et al., 2018; Wania and Shunthirasingham, 2020). Yet except for national air monitoring networks in Czechia (Kalina et al., 2018), Norway (Tørseth et al., 2012), Spain (Muñoz-Arnanz et al., 2016), and the United Kingdom (Graf et al., 2016), most countries in Europe have only 1 or 2 long-term SVOC monitoring sites, with monitoring of POPs being particularly scarce (Aas and Bohlin-Nizzetto, 2018; Hulek et al., 2014).

Spatial modelling can be a powerful tool to fill gaps in the geographic coverage of monitoring networks. Land use regression (LUR) is a spatial modelling technique that can be used to describe the environmental distribution of chemical concentrations at unmeasured locations across a region based on land use and related predictor variables (e.g., industrial emission records, traffic intensities, demographics). LUR methods have been used extensively in spatial studies of conventional air quality pollutants such as NO_X, O_3, and PM (Beelen et al., 2013; Eeftens et al., 2012; Gulliver et al., 2018; Hoek et al., 2008; Jedynska et al., 2017, 2014; Lu et al., 2020; Van Nunen et al., 2017), as well as some volatile organic compounds (Amini et al., 2017). While these studies typically investigate small-scale spatial variation within individual cities to evaluate human exposure, some studies have used LUR models to predict atmospheric concentrations on larger scales, such as across Europe (de Hoogh et al., 2018, 2016; Vienneau et al., 2013; Vizcaino and Lavalle, 2018) and even globally (Larkin et al., 2017). However, due to the greater complexity of measuring SVOCs and the relative scarcity of permanent monitoring sites compared to conventional air pollutants, few attempts have been made to apply LUR models to SVOCs. Some urban atmospheric LUR studies have included PAHs, but typically in the context of characterizing PM composition where the models reflect particle-phase concentrations in air (Jedynska et al., 2014; Noth et al., 2011; Polidori et al., 2010); only two LUR studies have considered gas-phase PAHs (Masri et al., 2018; Noth et al., 2016). Furthermore, to our knowledge there has only been one previous study to apply LUR methods to atmospheric PCB concentrations (Melymuk et al., 2013), and none for OCPs.

The MONET passive air sampling network (RECETOX, Masaryk University) is the largest national SVOC monitoring network in Europe, and includes 29 permanent sampling sites spread across Czechia that continuously measure atmospheric SVOC levels in 28-day intervals (Kalina et al., 2018). Additional MONET sampling extends across the rest of Europe as well as Africa (Přibylová et al., 2012; White et al., 2020). However, the relatively dense spatial coverage of the MONET network in Czechia (29 sites compared with 1–2 sites in most other European countries), provides an opportunity to apply LUR methods to SVOC monitoring data at a national scale. In this study we use spatial predictor variables extracted using geographic information systems (GIS) and two years of passive air sampling data (2015–2017) from the 29 Czech sites in the MONET network to develop LUR models for PAHs, PCBs, and the OCP dichlorodiphenyltrichloroethane (DDT). We demonstrate the strength of LUR for developing generalized national-scale models of SVOCs that can help to improve sampling network design for optimal coverage of sources, identify regional differences for human and environmental exposure assessment, and assist in evaluating the effectiveness of chemical regulations.

2. Methods

2.1. Air sampling

Continuous passive air sampling of SVOCs in Czechia is coordinated by the MONET network at RECETOX (Masaryk University, Brno) and has occurred at more than 60 sites across the country since 2006, with 29 sites operating as of 2017 (Fig. 1). These sites cover a wide range of land types and regions including remote mountainous sites, rural agricultural sites, urban city sites, and industrial sites (Table S1). Air monitoring data for all 29 sites were used for model development, covering a two-year period (July 2015 to June 2017) to reduce the potential influence of seasonality and variance in atmospheric levels within a single year, while avoiding the influence of long-term temporal trends previously identified for SVOCs at some of these sites (Kalina et al., 2018). Hourly meteorological conditions over the two-year monitoring period were assessed using the MERRA-2 model (Gelaro et al., 2017), as previously described for passive air sampling studies (Bohlin-Nizzetto et al., 2020; Herkert et al., 2018; White et al., 2020), and were relatively uniform between all 29 sites. Median site temperatures ranged from 7.0–9.4 °C and median site wind speeds ranged from 1.0–2.7 m/s. Based on continuous monitoring in 28-d passive sampling intervals, this two-year data selection resulted in 26 samples per site – except for Churáňov, Rudolice, and Rýchory with only 25, 25, and 24 samples, respectively – for a total of 750 samples. The sampling periods and deployment dates for all samples at each site included in this study are depicted in Fig. S1. Ninety percent of samples were deployed for exactly 28 d, and 99% were deployed for 26 to 30 d (Fig. S2).

MONET passive samplers consist of two stainless steel bowls containing a 15 cm × 1.5 cm polyurethane foam (PUF) disk and are typically deployed at 1–2 m above ground level. After the sampling period, PUF disks are collected and analyzed for a full suite of SVOCs at the RECETOX Trace Analytical Laboratory. Analytical methods are described in detail in the Supplemental Materials (SM Section 2 – Analytical Methods). In this study we examined seven indicator PCBs: 28, 52, 101, 118, 138, 153, and 180; fourteen PAHs: acenaphthylene (ACY), acenaphthene (ACE), fluorene (FLU), phenanthrene (PHE), anthracene (ANT), pyrene (PYR), fluoranthene (FLA), chrysene (CHR), benz[*a*]anthracene (BAA), benzo[*b*]fluoranthene (BBF), benzo[*k*]fluoranthene (BKF), benzo[*a*]pyrene (BAP), benzo[*ghi*]perylene (BGP), and indeno [1,2,3-*cd*]pyrene (IP); and DDT and its degradation products (o,p'-DDD, o,p'-DDE, o,p'-DDT, p,p'-DDD, p,p'-DDE, p,p'-DDT). Several of these compounds, particularly the high molecular weight PAHs and PCBs, were often present within the PUF disks at concentrations below their instrumental limit of quantification (LOQ, 3.3× the limit of detection; Table S2). For the purposes of data analysis and model development, these left-censored values were substituted by $(\sqrt{2}/2)$ x LOQ (Antweiler, 2015). Levels of SVOCs measured by passive sampling are reported in ng per PUF disk per sampling period. Although SVOC levels from passive sampling are often converted to concentrations (i.e., ng/m^3), this conversion may be biased by a number of sources of uncertainty (Holt et al., 2017). We chose to work with the primary data in units of ng/PUF/d as these values are still appropriate for spatial assessment and analysis of SVOCs (Kalina et al., 2018). To address atmospheric levels at a national scale, and in consideration of regulations and policies that typically address groups of chemicals of concern, we chose to develop simplified models using the sums of SVOC compound classes ($\sum_7$PCB, $\sum_{14}$PAH, and $\sum_6$DDT).

2.2. Predictor variables

Tool functions in ArcGIS Desktop software (v.10.5, Esri) were used to determine values of predictor variables for all sampling sites, either at the site itself (e.g., elevation) or in buffer zones of defined sizes (1 km and 5 km) around each site (e.g., population, traffic density). Both buffer zone sizes were examined for buffered variables, and the buffer with the

Site Code	
1	Bily Kriz
2	Blansko
3	Brno Kotlarska
4	Brno Lisen
5	Brno Turany
6	Churanov
7	Hodonin
8	Jesenik
9	Jested
10	Klet
11	Kosetice
12	Kosor
13	Kucharovice
14	Kyjov
15	Libis
16	Mokra Horakov
17	Planavy
18	Prague Libus
19	Prague Radotin
20	Prebuz
21	Primda
22	Rudolice
23	Rychory
24	Sedlec
25	Sedlonov
26	Sivice
27	Sneznik
28	Svratouch
29	Vernovice

Fig. 1. MONET passive air sampling sites in Czechia included in this study. Shading indicates topography, and cities with population > 100,000 people are identified.

strongest single variable correlation was used in model development. Atmospheric SVOC levels and most predictor variables were log-normally distributed and thus log-transformed prior to modelling, except for emission point sources and the hill-valley index (HVI). Determination of the predictor variables is summarized in Table 1 and detailed in the Supplementary Materials (SM Section 3 – Predictor Variables).

Annual consumption of several heating fuel types (propane-butane, natural gas, bituminous coal, oil, coke, lignite, and wood) was estimated for all 6354 administrative units of Czechia based on the relationship between population and fuel consumption records from a subset of 145 administrative units located in the north of the country. This region was previously used as an input for PAH air concentration modelling (Sáňka et al., 2014) and is considered representative of the country as a whole, as it covers a diversity of settlement types (large city of >100,000 people, as well as smaller villages and towns). Population (counts) and annual fuel consumption (masses) for each administrative unit were log-transformed to normalize distributions, and a linear regression was applied (Fig. S10). The linear relationships observed between the log-transformed fuel consumption and population (Table S3) were then used to extrapolate annual fuel consumption in the remaining administrative units according to settlement size (Fig. S11).

PAH and PCB emission factors for the various fuel types were estimated according to published studies (Tables S4 and S5). While a significant amount of uncertainty exists in the emission factors due to differences in heater types, efficiencies, operating procedures and fuel qualities, they convey the relative differences according to fuel efficiencies and emissions (e.g., PAH emissions from lignite burning are 200× greater than from home heating oil and 2000× greater than from natural gas). As a result of these findings, coke, lignite, wood, and bituminous coal were deemed to be the four major domestic fuel types for PAH and PCB emissions across Czechia. In addition to the consumption of these four individual fuel types being used as predictor variables in model development, the sum of all four (total fuel consumption) was also included.

2.3. Model development

Model development followed a similar approach to the standard LUR method that has previously been applied in atmospheric studies of PM, NO_X, and PAHs (Beelen et al., 2013; de Hoogh et al., 2018, 2016, 2014, 2013; Eeftens et al., 2012; Jedynska et al., 2014). Multiple linear regression models were derived for atmospheric levels of $\sum_{14}$PAH, $\sum_{7}$PCB, and $\sum_{6}$DDT via a multidirectional stepwise method based on Akike information criteria (AIC) selection using the stepAIC() function in R software (v.3.4.2). This function adds and removes predictor variables iteratively until it identifies a statistically significant final model that maximizes the explained variance (adjusted R^2) in the dependent variable (atmospheric SVOC levels). However, while the final model may be statistically significant, individual included predictor variables may not be, or may only negligibly affect the overall explained variance. In the previously mentioned LUR studies, individual site- and compound-specific models were developed to explain as much of the variance in atmospheric levels as possible. As a result, optimization of these final models was relatively lenient with the inclusion of all predictor variables with p-values ≤ 0.10 that each increased the adjusted R^2 by ≥0.01 (Beelen et al., 2013; de Hoogh et al., 2014, 2013; Eeftens et al., 2012; Jedynska et al., 2014). However, our aim was to develop simplified models for SVOC groups on a national scale to identify the most important predictor variables. As a result, we used a more stringent optimization process in which only predictor variables with p-values ≤ 0.05 that increased the adjusted R^2 by ≥0.05 were retained in the final models selected by the stepAIC() function. Validation of each optimized model was performed using a standard leave-one-out cross validation (LOOCV) method (Beelen et al., 2013; de Hoogh et al., 2014, 2013; Eeftens et al., 2012; Jedynska et al., 2014). Spatial autocorrelation of the residuals from each optimized model were subsequently assessed using the Moran's I statistic (Beelen et al., 2013; de Hoogh et al., 2013; Eeftens et al., 2012; Jedynska et al., 2014). Finally, normality of residuals for each optimized model was assessed using the Shapiro-Wilk test.

Table 1
Predictor variables and data sources.

Predictor variable	Description and data source
Fuel consumption *(ConsFuel)* PAH emissions from local heating *(HeatEmissPAH)* PCB emissions from local heating *(HeatEmissPCB)*	See Section 2.2
PAH industrial source density *(PointSourcePAH)* PM10 industrial source density *(PointSourcePM10)*	The European Environmental Agency's European Pollutant Release and Transfer Register (*E*-PRTR, http://prtr.ec.europa.eu) was used to estimate industrial air pollution. All pollution sources within Czechia and neighbouring regions of adjacent countries with either PAH, PCB, or PM10 records were extracted and imported into GIS. Only one official record of PCB sources was identified, thus PCBs were not considered further. A kernel density function was applied to PAH and PM10 sources, with the emission intensity of pollution sources selected as the population attribute. The search radius of the kernel density tool was set to 40 km. Rasters were created for both PAH and PM10 emissions, and the corresponding values were extracted for each sampling site for use as predictor variables (Fig. S3).
PCB concentrations in soil (*SoilPCB*)	Kubošová et al. (2009) used regression trees to predict concentrations of 5 PCB congeners (101, 118, 138, 153, and 180) in soil across Czechia at a 1 km spatial resolution. Predicted soil concentrations were imported to GIS and average values were extracted for each sampling site using a 5 km radius buffer.
Traffic density *(DensTraf)*	Road traffic density (vehicles/day) was taken from the Country Traffic Census performed by the Czech Roads and Motorways Directorate (http://scitani2010.rsd.cz/pages/informations/default.aspx) and spatially attributed in GIS. Exact values were available for a subset of roads, including all major roads. Roads of lower classes without exact values were assigned average values based on roads of the same class. Small streets inside cities and villages were excluded. A GIS line density tool was used to create a grid of traffic density across the country, and the sum of all traffic density grid values was calculated in a 1500 m radius buffer around each sampling site (Fig. S4).
Road density *(LengRoad)*	Road density was determined using an OpenStreetMap dataset from Geofabrik's server (http://download.geofabrik.de/). Only roads with car traffic were included in the analysis. The total length of roads within a 1 km and 5 km radius buffer of each sampling site was calculated (Fig. S5).
Building density *(AreadBuild)*	Building density was determined using a dataset from Geofabrik's server (http://download.geofabrik.de/). The total percentage of built-up area within a 1 km and 5 km radius of each sampling site was calculated. It is important to note that this dataset did not include building heights, only land coverage, which could limit its explanatory potential (Fig. S6).
Population *(Pop)*	The population count of each administrative unit in Czechia was taken from ArcData Prague (https://www.arcdata.cz/produkty/geograficka-data/arccr-500). Administrative units were reshaped in GIS by excluding water bodies and forests to achieve a more realistic population distribution. The total population was calculated within a 1 km and 5 km radius of each sampling site (Fig. S7).
Land use *(AreaCrop)* *(AreaIndus)*	The Corine Land Cover 2006 database (http://www.eea.europa.eu/data-and-maps/data/clc-2006-vector-4) was used to determine spatial coverage of classes 121 (industry) and 211, 221 and 222 (agriculture) within a 1 km and 5 km radius of each sampling site (Fig. S8).
Elevation *(Elevation)*	The elevation of each sampling site was acquired using GPS measurement during sampler deployment.
Hill-valley index (*HVI*)	To incorporate the effect of topography into the LUR without the use of more complex contaminant air transport models, a 'hill-valley index' (HVI) was developed to quantify the position of each sampling site relative to the elevation of the surrounding terrain. The HVI at each site was calculated by dividing the elevation of the site by the median elevation of a 1.5 km radius buffer around each site (acquired from Shuttle Radar Topographic Mission, http://www.diva-gis.org/gdata). Thus, HVI <1 indicates the site is located in a valley, while HVI >1 indicates the site is located on a hill (see Fig. S9).

3. Results & discussion

3.1. Atmospheric SVOC levels

Average atmospheric levels of each SVOC compound class were similar across most sites. The lowest average levels for PAHs, PCBs, and DDT were detected at sites in the remote mountainous regions around the border of the country, many of which were previously identified as being the least polluted sites in an assessment of Czech background air monitoring (Kalina et al., 2018), while the sites with the highest average levels differed by compound class (Fig. 2).

Average atmospheric $\sum_{14}$PAH levels varied over approximately one order of magnitude, ranging from 14–124 ng/PUF/d at all sites except Prague Radotin (255 ng/PUF/d) and Věřňovice (394 ng/PUF/d) (Fig. 2A). Both of these sites are heavily influenced by local sources, with Prague Radotin located at a cement factory and Věřňovice located on the Czech-Polish border in the centre of a coal mining and industrial region with significant coal use for domestic heating. Phenanthrene was the most abundant individual PAH across all 29 sites, comprising 46 ± 3% of the average $\sum_{14}$PAH levels, followed by fluoranthene, fluorene, and pyrene. Together these four PAHs alone comprised 90 ± 3% of the average $\sum_{14}$PAH levels at all sites (Table S7).

Average atmospheric $\sum_{7}$PCB levels ranged from 0.04–0.25 ng/PUF/d, with only Brno Kotlarska significantly elevated above all others (0.35 ng/PUF/d; Fig. 2B). Similarly, the relative proportions of individual PCB congeners contributing to $\sum_{7}$PCB levels were comparable among all sites, except for a significantly higher average level of PCB 28 at Brno Kotlarska (Table S8).

Average atmospheric $\sum_{6}$DDT levels ranged from 0.03–0.70 ng/PUF/d across all sites (Fig. 2C) and were dominated by p,p'-DDE (74 ± 7%). Four of the five sites with the highest DDT levels are located in the South Moravia region and surrounded by large areas of agricultural land, reflecting historical agricultural application of DDT. The other site with high levels (Libiš) is located less than 1 km away from a former pesticide manufacturing plant, suggesting environmental contamination from past industrial production. This difference in dominant sources was apparent from the differences in the composition of $\sum_{6}$DDT, with the four agricultural sites having the highest p,p'-DDE levels, while Libiš had the highest levels of the other five DDT compounds (Table S9).

3.2. Model development

Statistically significant multiple linear regression models were derived for atmospheric $\sum_{14}$PAH and $\sum_{7}$PCB levels across Czechia, while only a simple linear regression model could be derived for $\sum_{6}$DDT levels. Of all individual predictor variables, sampling site elevation was able to explain the greatest amount of variance in the atmospheric levels of all SVOCs, with a moderately strong negative correlation (adjusted R^2 ranging from 0.45–0.49), indicating higher levels at lower elevations (Table S10). Most background air sampling sites within Czechia are located in mountainous regions around the border of the country while large cities are located at lower elevations towards the centre. As a result, elevation of the sampling sites was also significantly negatively correlated with most of the other selected predictor variables: building, crop, and industrial land coverage,

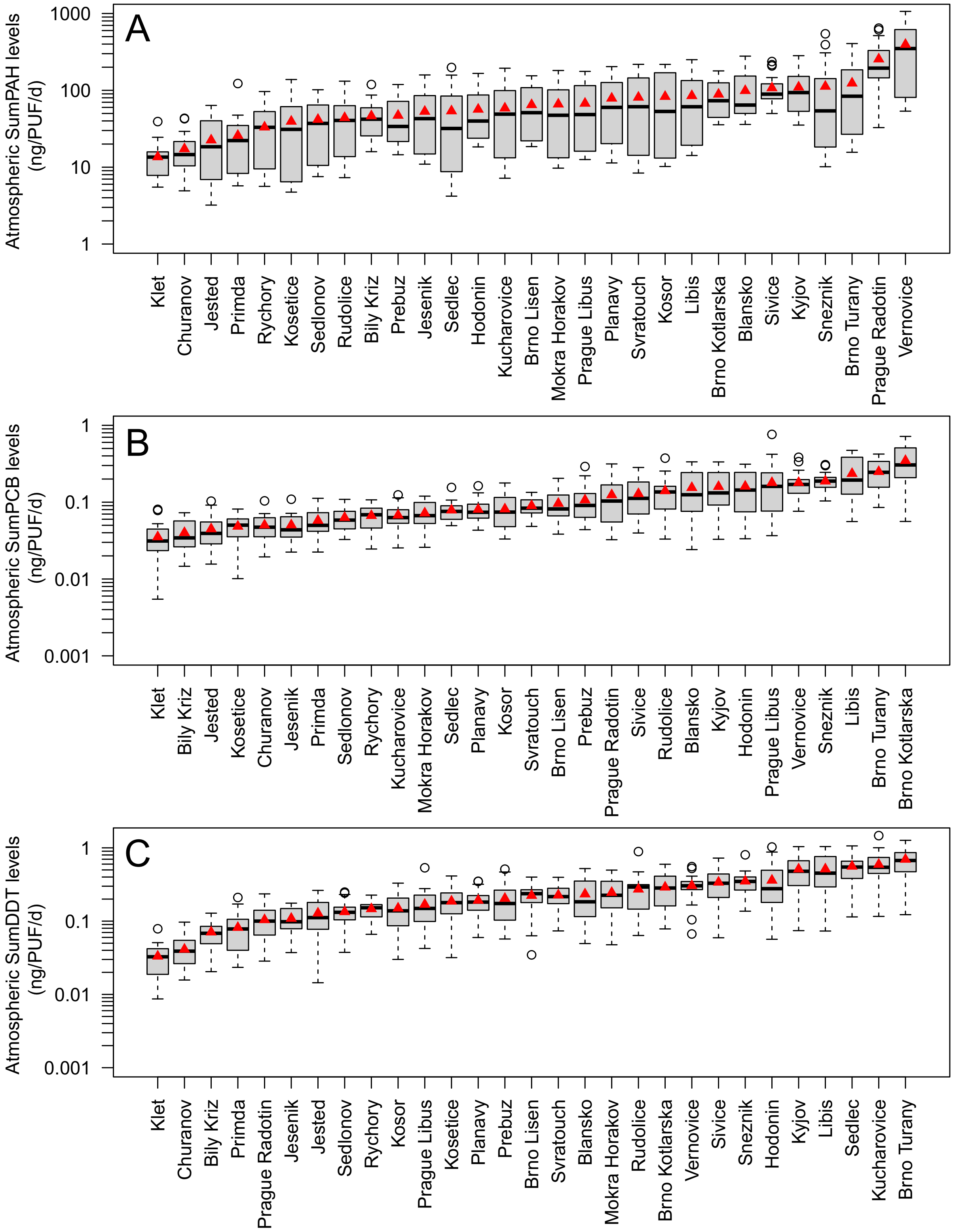

Fig. 2. Log-boxplots of atmospheric levels (ng per PUF disk per day) of A) $\sum_{14}$PAH; B) $\sum_{7}$PCB; and C) $\sum_{6}$DDT. Levels measured monthly (28-d) by passive sampling over a two-year period (July 2015–June 2017) at 29 sites across Czechia ($n = 24$–26 at each site). Boxplots depict the interquartile range (IQR): box upper limit is the 75th percentile, horizontal line is the 50th percentile (median), box lower limit is the 25th percentile; whiskers above and below depict 1.5 x IQR; data points outside this range (outliers) are depicted as hollow circles. Sites are listed in order of increasing average level (depicted by triangles).

population, total fuel consumption, road length, and traffic density (Table S11). Elevation is therefore a moderately strong predictor of anthropogenic activity within Czechia. Population within 5 km of each sampling site was also significantly correlated with atmospheric levels of all SVOCs, however the strength of the correlation was much lower than that of elevation (adjusted R^2 ranging from 0.10–0.25). Similar to elevation, population was also significantly correlated with most of the other selected predictor variables. Since the strong correlation between atmospheric SVOC levels and site elevation is a result of the specific topography and population distribution across Czechia, we anticipate that population density would likely be a more general predictor of anthropogenic activity if similar LUR methods were applied to atmospheric SVOC levels in other countries.

While $\sum_6$DDT was only significantly correlated with elevation and population, $\sum_{14}$PAH and $\sum_7$PCB were both significantly correlated with many of the other predictor variables to varying degrees. However, most were not retained in the final models selected by stepAIC, and the subsequent optimization simplified the models even further. Stepwise removal of predictor variables that did not meet selection criteria during optimization of each model is summarized in Table S12. Overall, the models for $\sum_{14}$PAH and $\sum_7$PCB were relatively strong (adjusted $R^2 > 0.60$) with no spatial autocorrelation (Moran's I of 0.003 and 0.048, respectively). Conversely, the model explained variance for $\sum_6$DDT was lower (adjusted $R^2 = 0.45$) and weakly spatially autocorrelated (Moran's I of 0.142, $p < 0.05$). Despite this, the LOOCV R^2 was <10% lower than the adjusted R^2 for all three SVOC models, indicating stable models with minimal influence of individual sites (Beelen et al., 2013; de Hoogh et al., 2014, 2013; Eeftens et al., 2012; Jedynska et al., 2014). Furthermore, all three final models passed the Shapiro-Wilk test for normality on model residuals (all >0.95) indicating appropriate log transformations.

3.2.1. PAH model

The final optimized $\sum_{14}$PAH model included three predictor variables – total domestic fuel consumption (sum of lignite, bituminous coal, coke, and wood), PAH point sources, and the HVI – and explained nearly 70% of the variance in atmospheric levels (Eq. (1)).

$$\log\left(\sum\nolimits_{14}\mathrm{PAH}\right) = 0.37\,\log(\mathit{ConsFuel}) + 0.28\,\log(\mathit{PointSourcePAH}) - 1.83(\mathit{HVI}) + 2.35 \quad \left[\text{Adjusted } R^2 = 0.68(p < 0.05);\ \text{LOOCV } R^2 = 0.62(p < 0.05)\right] \tag{1}$$

Surprisingly, neither road length nor traffic density were included in our final optimized model. Previous LUR models developed for PAHs have found traffic-related predictor variables to explain a significant portion of the variance in atmospheric levels (Noth et al., 2016), though the strengths of the correlations were still generally weaker than those for other air pollutants (Jedynska et al., 2014; Melymuk et al., 2013). Melymuk et al. (2013) attributed the weaker explanatory power of their PAH model to more spatially complex emission sources and dynamics, as well as greater uncertainty in passive sampling of heavier particle-associated compounds. Jedynska et al. (2014) attributed the poorer performance of their PAH model to a smaller influence of traffic emissions on atmospheric PAH levels relative to other less characterized sources, such as wood burning and industrial emissions, for which no data were available. For our study, enough data were available in Czechia to provide estimates for both of these primary emission sources, with total domestic fuel consumption and industrial PAH point sources both individually explaining modest amounts of the variance in atmospheric levels (30% and 21%, respectively).

Jedynska et al. (2014) also proposed the incorporation of chimney height into PAH models to account for the influence of distance and vertical transport of PAHs from point sources to sampling sites. The atmospheric lifetime of PAHs is generally much shorter than that of other SVOCs such as PCBs, due to dry deposition of particle-associated compounds as well as photochemical degradation (Lammel et al., 2009). As a result, atmospheric levels of PAHs are strongly affected by the altitude at which emissions occur (such as chimney height; Wu et al., 2006) and sharply decline with increasing altitude if sources are predominantly at ground level (Farrar et al., 2005; Tao et al., 2007). Surrounding topography can therefore significantly compound these effects, with higher levels measured in the lowest atmospheric layers in valleys compared to those at mountaintops, even when affected by the same local source (Kalberer et al., 2004). Some influence of this vertical distribution on atmospheric levels in Czechia is reflected in the $\sum_{14}$PAH model through the inclusion of the HVI predictor variable. In agreement with these previous studies, atmospheric PAH levels within Czechia were significantly negatively correlated with HVI, indicating higher levels at sampling sites located in valleys compared to those located on hills or mountaintops.

Overall, the model is slightly concentration-dependent and overestimates $\sum_{14}$PAH at the two sites with the lowest atmospheric levels (Klet' and Churáňov), although it accurately describes the atmospheric levels at the two most heavily polluted sites (Prague Radotin and Věřňovice) (Fig. S12). Overestimation of LUR models for SVOCs has previously been attributed to the possible removal of emission sources and the resulting time-dependency of emission estimates (Melymuk et al., 2013), which may explain the discrepancy at the sites with low atmospheric PAH levels. However, it is also possible that Klet' and Churáňov are more significantly affected by the negative correlation between elevation and $\sum_{14}$PAH levels than is captured by the HVI, as they are the highest altitude sites included in this study (1060 m and 1121 m, respectively). For the remaining sites, there appears to be a tendency for the model to underestimate concentrations to a greater degree than those it overestimates, suggesting additional sources of PAHs that are not reflected in the model. In addition to primary emission sources, Melymuk et al. (2013) also observed a correlation between atmospheric PAH levels and proximity to contaminated sites due to high levels of PAHs in soil which may act as a secondary source to air via volatilization. National data on PAHs in soil were not available for model development, thus the influence of soil and other secondary emission sources on atmospheric $\sum_{14}$PAH levels in Czechia may be partially responsible for the remaining variance (~30%) not explained by primary emission sources and HVI in our model.

3.2.2. PCB and DDT models

Sampling site elevation was strongly negatively correlated with atmospheric levels of PCBs and DDT in Czechia and was the only predictor variable retained in the final $\sum_6$DDT model (Eq. (2)), and one of two variables retained in the final $\sum_7$PCB model (Eq. (3)).

$$\log\left(\sum\nolimits_{6}\mathrm{DDT}\right) = -0.84\,\log(\mathit{Elevation}) + 1.53 \quad \left[\text{Adjusted } R^2 = 0.45(p < 0.05);\ \text{LOOCV } R^2 = 0.37(p < 0.05)\right] \tag{2}$$

$$\log\left(\sum\nolimits_{7}\mathrm{PCB}\right) = -0.99\,\log(\mathit{Elevation}) + 1.26\,\log(\mathit{SoilPCB}) + 0.67 \quad \left[\text{Adjusted } R^2 = 0.62(p < 0.05);\ \text{LOOCV } R^2 = 0.55(p < 0.05)\right] \tag{3}$$

As previously discussed, elevation is strongly negatively correlated with anthropogenic activity within the country, with high elevation sites generally characterized by low population density and limited influence of industry, agriculture, or traffic. Atmospheric levels of PCBs and DDT at these sites are likely determined by long-range transport and more influenced by levels in the atmospheric mixing layer of Central Europe than local sources (Lammel et al., 2009). This is apparent from the relative homogeneity in the atmospheric PCB and DDT levels at the least polluted sites despite large differences in their geographic locations, suggesting they are capturing the same regional background level. However, elevation still only explained 47% and 45% of the variability in atmospheric $\sum_7$PCB and $\sum_6$DDT levels, respectively, suggesting the influence of local emission sources as well.

Although the production and use of PCBs in Czechia has been banned since 1984, and despite national efforts to eliminate PCB-containing devices and waste, PCBs remain in use in thousands of old electrical devices and building materials, as well as in surface layers of contaminated industrial sites (Bláha et al., 2017; Holoubek et al., 2006). Releases from older products and materials remaining in use can still play a significant role in atmospheric levels; for example, Melymuk et al. (2013) found that 75% of the variability in urban/suburban atmospheric PCB levels could be explained by just the mass of PCBs in use, storage, and building sealants within a 1 km radius around each sampling site. Thus, while rural and remote Czech sites are likely more influenced by long-range transport and background levels, urban and surburban sites with higher levels such as Brno Kotlarska may still be heavily influenced by nearby primary emissions or secondary emissions from contaminated soils.

We were able to obtain $\sum_5$PCB levels in soil across the country from a previous study (Kubošová et al., 2009). When these soil concentrations were included in the LUR analysis for $\sum_7$PCB as a predictor variable, they were retained in the final $\sum_7$PCB model in addition to elevation, and resulted in a substantial increase in the model explained variance (Eq. (3)). Many SVOCs are highly persistent in soils, which can act as a significant secondary emission source to the atmosphere if conditions favour volatilization (Holoubek et al., 2009). Within Central Europe, soil generally acts as a sink for high molecular weight (highly chlorinated) PCBs, but as a source for more volatile (less chlorinated) PCBs, particularly in summer months when temperatures are higher (Růžičková et al., 2008). However, soil-air transfer is highly site-specific and depends on the physicochemical properties of both the compound and soil, as well the contaminant burden within the soil and local meteorological conditions (Růžičková et al., 2008). Based on previous studies of soil contamination across Czechia (Holoubek et al., 2009; Kubošová et al., 2009; Růžičková et al., 2008) we expect that if we had spatially distributed records of $\sum_6$DDT in soil, the inclusion of these soil concentrations into the $\sum_6$DDT model would result in a similar or greater increase in the explained variance as that observed for the $\sum_7$PCB model, due to the historical application of DDT directly to agricultural land.

3.2.3. Model limitations

There are several limitations to these models related to the LUR method itself, as well as its implementation for the compounds selected in this study. Our study was limited to 29 sampling sites, and previous studies have shown that LUR models developed for air pollutants based on a small number of sampling sites have poor predictive power when validated against external datasets, even if both the adjusted R^2 and LOOCV R^2 values are high (Basagaña et al., 2012; Wang et al., 2012), and suggest LUR models should be developed using at least 40 (Wang et al., 2012) or even 80 sites (Basagaña et al., 2012). It should be noted that these authors offered significantly more predictor variables to their models (76 and 106, respectively, vs. 12 in this study), and restricting the number of predictor variables offered to LUR models can improve the predictive power of models developed using a small number of sampling sites, though some overestimation of R^2 is still observed (Basagaña et al., 2012).

In addition to the small number of sampling sites, the predictive power of our models is likely also reduced by our decision to develop models for sum classes of SVOCs rather than individual compounds/congeners at specific sites as is common in other LUR studies. While broad LUR models may be more valuable for national policy and regulation, the influence of specific sources on individual compounds may be lost or obfuscated. For example, Melymuk et al. (2013) found that the distribution of metal refining and processing industries explained 60% of the variance in atmospheric PCB concentrations, but only for tri- and tetra-chlorinated congeners, while paint and pigment manufacturing was associated with other congeners. Similar differences can be observed in PAHs, with phenanthrene being the dominant compound in bituminous coal-, lignite-, and wood-burning emissions, while fluoranthene dominates coke-burning emissions (Table S4). Moreover, the influence of degradation in ambient air and during sampling can be important for PAHs under certain conditions (Keyte et al., 2013; Melymuk et al., 2016), however this was not accounted for in our generalized model for Σ_{14}PAHs.

Site selection is also an important consideration when applying LUR models on a large spatial scale, rather than for a single city or urban area as is more common. To optimize their predictive power, models should be developed using sites that are representative of the population to which the model will be applied (Wang et al., 2012). For our national-scale models, it was therefore important to include a broad range of site classifications including remote, rural, suburban, urban, and industrial sites. Although this was achieved to some extent, approximately two thirds of the sites used in the development of the models were background rural sites simply due to these sites being more prevalent in the MONET network. As a result, our LUR models are likely biased towards rural sites and may be less accurate for more industrial or impacted sites. This is further compounded by the lack of primary and secondary emission source data within the country, as previously discussed.

3.3. Model application

To demonstrate the ability of LUR modelling to improve understanding of atmospheric SVOC distributions and optimize design of air monitoring networks, the two relatively strong models ($\sum_{14}$PAH and $\sum_7$PCB) were applied at a national scale using ArcGIS software to estimate levels across Czechia (Fig. 3). Due to the model limitations we chose to depict atmospheric SVOC levels semi-quantitatively (high to low) to prevent misinterpretation, however, this still provides sufficient information to identify suspected emission hotspots and source regions. The higher granularity of the PAH map reflects the dependence of the model on local topography (represented by the HVI predictor variable), suggesting that atmospheric distributions of PAHs may be more heterogeneous than PCBs. This is supported by the fact that $\sum_{14}$PAH levels include several highly particle-bound PAHs that have more limited air transport and thus deposit closer to sources, while PCBs are more volatile and have greater potential for long-range transport. Furthermore, the $\sum_7$PCB model suggests that secondary emissions (e.g., soil) may be the major source of atmospheric PCB levels, while PAHs are still primarily affected by on-going primary emissions, with greater diversity of combustion sources.

In both maps there is a tendency towards higher levels in urban regions compared to rural/background regions, as frequently reported for these compounds (Cetin et al., 2017; Jamshidi et al., 2007; Melymuk et al., 2012; Motelay-Massei et al., 2005; Venier et al., 2019). In particular, the PAH map highlights elevated atmospheric levels in the Ostrava region, which has been identified as a hotspot of PAH contamination due to coal mining and heavy industry (Jiřík et al., 2016; Sram et al., 2013). Industrial emissions in this region are also linked to elevated atmospheric PCB levels, as observed in Fig. 3. Higher PAH levels are also observed in the North Bohemia basin, northwest of Prague, potentially associated with extensive coal (lignite) surface mining in this area.

In addition to the industrial emissions in the Ostrava region, Fig. 3 also identifies large areas of South Moravia (Morava River basin, southeast of Brno) and North Bohemia (Prague) as having the highest atmospheric PCB levels across Czechia. In the 1990s, this region of South Moravia had the highest PCB levels in all environmental matrices across the country due to the Colorlak paint factory; Colorlak was the primary consumer of PCBs produced in the former Czechoslovakia and resulted in PCB-containing paints being heavily used in this region for decades (Holoubek et al., 2007a). This was also the region most affected by a major flood event in 1997 that damaged industrial and agricultural chemical storage facilities and remobilized PCB burdens from sediments – the largest environmental sink for SVOCs in Czechia (Holoubek et al., 2007b) – to surface soils and then to air (Holoubek et al., 2007a). Similarly, a major flood event in 2002 occurred in central Bohemia and led to significantly increased atmospheric levels of pesticides in the region, including DDT (Holoubek et al., 2007a). The success of the LUR-based maps in identifying known/plausible concentration gradients and hotspots reflects well on their utility for sampling design and source tracking.

However, the PCB map also shows an area of high predicted air levels northeast of Prague, which corresponds to a protected sandstone landscape. The plausibility of this area having high PCB levels is not clear; air sampling has not been directly performed in the region, but the high

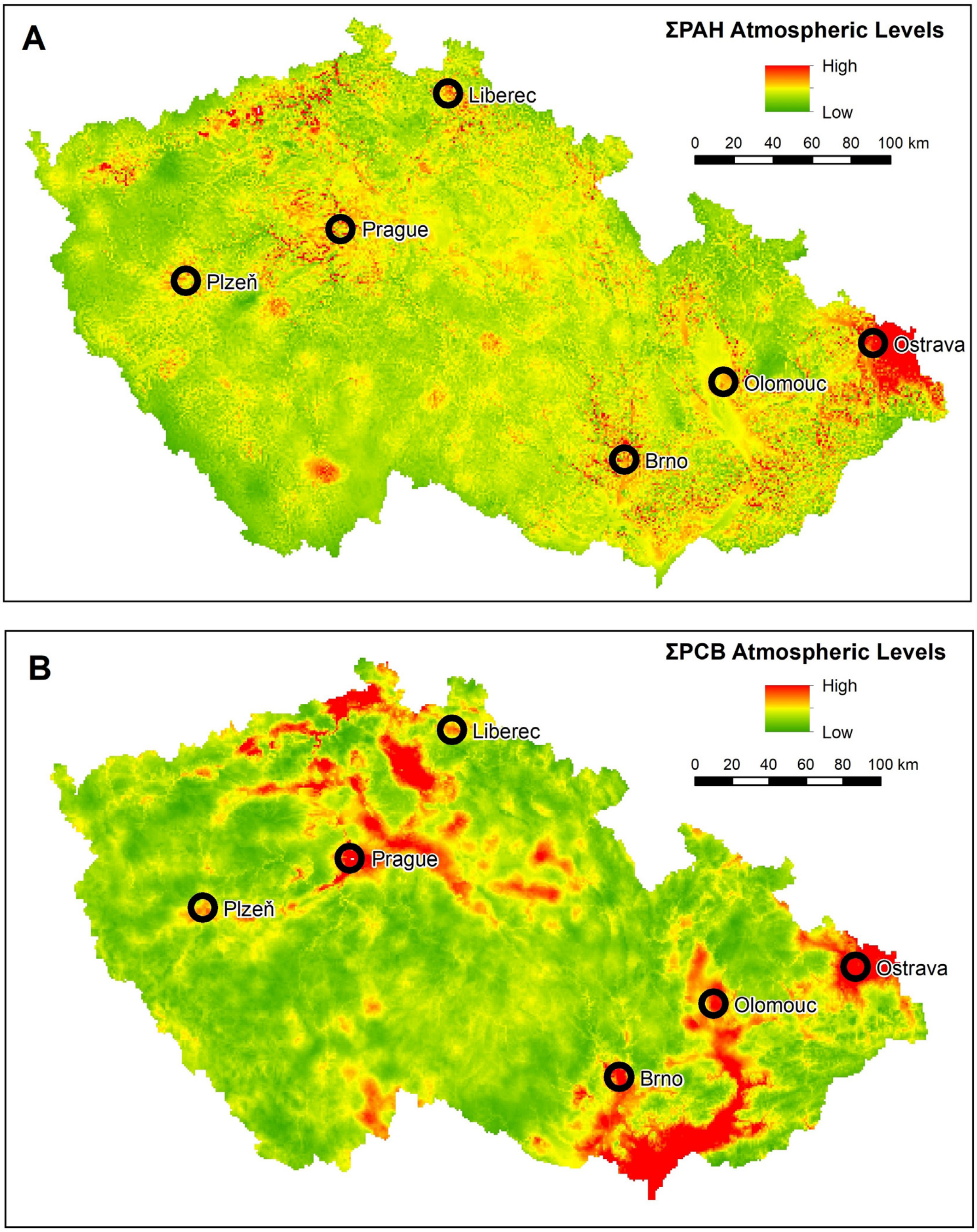

Fig. 3. Application of the LUR models across all of Czechia, indicating the estimated gradient of low to high atmospheric levels of A) Σ_{14}PAH, ($R^2 = 0.68$), and B) Σ_7PCB, ($R^2 = 0.62$). Cities with population > 100,000 are indicated on the map.

levels may be an artifact of the soil organic carbon influence on PCB soil concentrations in the Kubošová et al. (2009) soil model. The cluster analysis by Kubošová et al. did not distinguish between mountain and sandstone podzol soils, and it is unclear whether the predicted high PCB burden of the mountain podzols is also applicable to the sandstone podzols. This would require further investigation in the form of air and/or soil sampling. Thus, while there are some clear limitations to the application of LUR for predicting atmospheric levels, largely related to the assumptions of the input data, the application of the PAH and PCB models across the entire country presents strong evidence of their utility as a tool for better understanding sources and distributions of SVOCs in air.

4. Conclusions and policy implications

Broad spatial coverage in air monitoring of SVOCs remains challenging, even with the widespread use of passive air sampling. Significant spatial heterogeneity can exist in concentrations on a small scale (a few km) causing gradients in human exposure, but can also provide crucial information about point sources. LUR modelling is an ideal tool to better understand regional SVOC levels by relying on high density spatial data (population, industrial land use, topography) to highlight atmospheric gradients of SVOCs since such spatial data are more widely available than SVOC concentrations. We have demonstrated the success of this technique for describing atmospheric levels of PAHs and PCBs at a national scale for the first time. However, the regression equations have not been tested with an independent data set, so we consider these models to be primarily descriptive rather than predictive. As such, they are useful to identify specific predictor variables that correlate with atmospheric levels, and thereby suggest potential important sources of PAHs and PCBs in Czechia.

The PAH model suggests active primary sources, particularly domestic heating, are the most important driver of PAHs in Czech air; thus, reductions to emissions from domestic heating (low-emission boilers, transitioning away from wood and coal for fuel) should reduce their atmospheric burden. In contrast, the PCB model suggests secondary sources, particularly emission from soils and sediments, may be a key driver of levels in Czech air. These sources are much more challenging to actively reduce and will largely depend on environmental cycling and the relatively slow process of environmental degradation. This is consistent with the recently observed increasing environmental half-lives of PCBs and slowly declining/stabilizing atmospheric concentrations (Graf et al., 2016; Muñoz-Arnanz et al., 2016; Shunthirasingham et al., 2016), and in contrast to the previously established understanding (up to ~2010) that primary sources were the main contributor to atmospheric PCB levels (Melymuk et al., 2012; Robson and Harrad, 2004; Schuster et al., 2010). While the influence of primary vs. secondary sources likely differs by region/country, recent active reduction of PCB stocks in many countries in response to the Stockholm Convention deadline of 2025 for elimination of in-use PCBs (UNEP, 2017) may have shifted the dominant sources of PCBs to the atmosphere from primary to secondary. We also clearly show that high quality surrogate variables covering the range of possible sources (both primary and secondary) is a necessary part of successful LUR model development for SVOCs, and a lack of such data, as in the case of our DDT model, hinders the development and use of LUR as a technique.

Knowledge of atmospheric SVOC levels is necessary to assess the effectiveness of actions for reducing SVOCs in the environment, in particular the global Stockholm Convention on Persistent Organic Pollutants, and the Convention on Long-Range Transboundary Air Pollution (CLRTAP) within Europe. To these ends, it is crucial to monitor background sites for long-term trends without the influence of local point sources. At the same time, global monitoring needs to be cost-effective and sustainable for decades which requires minimizing the number of monitoring sites needed to capture both temporal trends and regional gradients in concentrations. LUR modelling can help improve selection of appropriate background sites, and identify areas influenced by local/point sources to be avoided in monitoring networks. At the national and local scales, monitoring is a crucial tool for regulators and policymakers targeting local air quality and requires a complete overview of influences (hot spots, legacy contamination, urban-rural gradients, agricultural influences, etc.). Thus, LUR modelling can also help to improve local, targeted air quality monitoring by pinpointing areas where such impacts would be expected. In both cases, incorporation of LUR enhances continuous atmospheric SVOC monitoring, which should include on-going optimization of network design to improve our understanding of sources and distributions in the longer term.

CRediT authorship contribution statement

Kevin B. White: Methodology, Validation, Investigation, Formal analysis, Writing – original draft. **Ondřej Sáňka:** Methodology, Visualization, Writing – review & editing. **Lisa Melymuk:** Conceptualization, Methodology, Supervision, Writing – review & editing. **Petra Přibylová:** Resources, Project administration. **Jana Klánová:** Funding acquisition, Writing – review & editing, Supervision.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2021.148520.

References

Aas, W., Bohlin-Nizzetto, P., 2018. EMEP/CCC-Report 3/2020-Heavy Metals and POP Measurements, 2018. Kjeller, Norway.

Amini, H., Yunesian, M., Hosseini, V., Schindler, C., Henderson, S.B., Künzli, N., 2017. A systematic review of land use regression models for volatile organic compounds. Atmos. Environ. 171, 1–16. https://doi.org/10.1016/j.atmosenv.2017.10.010.

Antweiler, R.C., 2015. Evaluation of statistical treatments of left-censored environmental data using coincident uncensored data sets. II. Group comparisons. Environ. Sci. Technol. 49, 13439–13446.

Basagaña, X., Rivera, M., Aguilera, I., Agis, D., Bouso, L., Elosua, R., Foraster, M., de Nazelle, A., Nieuwenhuijsen, M., Vila, J., Künzli, N., 2012. Effect of the number of measurement sites on land use regression models in estimating local air pollution. Atmos. Environ. 54, 634–642. https://doi.org/10.1016/j.atmosenv.2012.01.064.

Beelen, R., Hoek, G., Vienneau, D., Eeftens, M., Dimakopoulou, K., Pedeli, X., Tsai, M.Y., Künzli, N., Schikowski, T., Marcon, A., Eriksen, K.T., Raaschou-Nielsen, O., Stephanou, E., Patelarou, E., Lanki, T., Yli-Tuomi, T., Declercq, C., Falq, G., Stempfelet, M., Birk, M., Cyrys, J., von Klot, S., Nádor, G., Varró, M.J., Dedele, A., Gražulevičiene, R., Mölter, A., Lindley, S., Madsen, C., Cesaroni, G., Ranzi, A., Badaloni, C., Hoffmann, B., Nonnemacher, M., Krämer, U., Kuhlbusch, T., Cirach, M., de Nazelle, A., Nieuwenhuijsen, M., Bellander, T., Korek, M., Olsson, D., Strömgren, M., Dons, E., Jerrett, M., Fischer, P., Wang, M., Brunekreef, B., de Hoogh, K., 2013. Development of NO 2 and NO x land use regression models for estimating air pollution exposure in 36 study areas in Europe - the ESCAPE project. Atmos. Environ. 72, 10–23. https://doi.org/10.1016/j.atmosenv.2013.02.037.

Bláha, K., Borůvková, J., Budňáková, M., Čupr, P., Eyblová, D., Gadas, P., Gruntorád, J., Hlavatý, E., Holoubek, I., Hůlek, R., Jarolím, O., Klánová, J., Kolář, J., Krech, L., Krejčová, V., Prokeš, R., Přibylová, P., Ptáčková, Ž., Puklová, V., Straková, H., Šebková, K., Špringar, Z., Štěpánek, P., Udatný, M., Vávra, T., Veselý, I., Vytopilová, M., 2017. Updated National Implementation Plan for Implementation of Stockholm Convention on Persistent Organic Pollutants in the Czech Republic for the Period 2018-2023. Prague.

Bohlin-Nizzetto, P., Melymuk, L., White, K.B., Kalina, J., Madadi, V.O., Adu-Kumi, S., Prokeš, R., Přibylová, P., Klánová, J., 2020. Field- and model-based calibration of polyurethane foam passive air samplers in different climate regions highlights differences in sampler uptake performance. Atmos. Environ. 238. https://doi.org/10.1016/j.atmosenv.2020.117742.

Cetin, B., Ozturk, F., Keles, M., Yurdakul, S., 2017. PAHs and PCBs in an Eastern Mediterranean megacity, Istanbul: their spatial and temporal distributions, air-soil exchange and toxicological effects. Environ. Pollut 220, 1322–1332. https://doi.org/10.1016/J.ENVPOL.2016.11.002.

Eeftens, M., Beelen, R., de Hoogh, K., Bellander, T., Cesaroni, G., Cirach, M., Declercq, C., Dėdelė, A., Dons, E., de Nazelle, A., Dimakopoulou, K., Eriksen, K., Falq, G., Fischer, P., Galassi, C., Gražulevičienė, R., Heinrich, J., Hoffmann, B., Jerrett, M., Keidel, D., Korek, M., Lanki, T., Lindley, S., Madsen, C., Mölter, A., Nádor, G., Nieuwenhuijsen, M., Nonnemacher, M., Pedeli, X., Raaschou-Nielsen, O., Patelarou, E., Quass, U., Ranzi, A., Schindler, C., Stempfelet, M., Stephanou, E., Sugiri, D., Tsai, M.-Y., Yli-Tuomi, T., Varró, M.J., Vienneau, D., Klot, S. Von, Wolf, K., Brunekreef, B., Hoek, G., 2012. Development of land use regression models for PM 2.5 , PM 2.5 absorbance, PM 10 and PM coarse in 20 European study areas; results of the ESCAPE project. Environ. Sci. Technol. 46, 11195–11205. https://doi.org/10.1021/es301948k.

Farrar, N.J., Harner, T., Shoeib, M., Sweetman, A.J., Jones, K.C., 2005. Field deployment of thin film passive air samplers for persistent organic pollutants: a study in the urban atmospheric boundary layer. Environ. Sci. Technol. 39, 42–48.

Gelaro, R., McCarty, W., Suárez, M.J., Todling, R., Molod, A., Takacs, L., Randles, C.A., Darmenov, A., Bosilovich, M.G., Reichle, R., Wargan, K., Coy, L., Cullather, R., Draper, C., Akella, S., Buchard, V., Conaty, A., da Silva, A.M., Gu, W., Kim, G.K., Koster, R., Lucchesi, R., Merkova, D., Nielsen, J.E., Partyka, G., Pawson, S., Putman, W., Rienecker, M., Schubert, S.D., Sienkiewicz, M., Zhao, B., 2017. The modern-era retrospective analysis for research and applications, version 2 (MERRA-2). J. Clim. 30, 5419–5454. https://doi.org/10.1175/JCLI-D-16-0758.1.

Graf, C., Katsoyiannis, A., Jones, K.C., Sweetman, A.J., 2016. The TOMPs ambient air monitoring network – continuous data on UK air quality for over 20 years. Environ. Pollut. 217, 42–51. https://doi.org/10.1016/j.envpol.2016.01.033.

Gulliver, J., Morley, D., Dunster, C., McCrea, A., van Nunen, E., Tsai, M.Y., Probst-Hensch, N., Eeftens, M., Imboden, M., Ducret-Stich, R., Naccarati, A., Galassi, C., Ranzi, A., Nieuwenhuijsen, M., Curto, A., Donaire-Gonzalez, D., Cirach, M., Vermeulen, R., Vineis, P., Hoek, G., Kelly, F.J., 2018. Land use regression models for the oxidative potential of fine particles (PM2.5) in five European areas. Environ. Res. 160, 247–255. https://doi.org/10.1016/j.envres.2017.10.002.

Herkert, N.J., Spak, S.N., Smith, A., Schuster, J.K., Harner, T., Martinez, A., Hornbuckle, K.C., 2018. Calibration and evaluation of PUF-PAS sampling rates across the Global Atmospheric Passive Sampling (GAPS) network. Environ Sci Process Impacts 20, 210–219. https://doi.org/10.1039/c7em00360a.

Hoek, G., Beelen, R., de Hoogh, K., Vienneau, D., Gulliver, J., Fischer, P., Briggs, D.J., 2008. A review of land-use regression models to assess spatial variation of outdoor air pollution. Atmos. Environ. 42, 7561–7578.

Holoubek, I., Adamec, V., Bartoš, M., Bláha, K., Bláha, L., Budňáková, M., Černá, M., Čupr, P., Demnerová, K., Drápal, J., Hajšlová, J., Hanzálková, M., Holoubková, I., Hrabětová, S., Jech, L., Klánová, J., Kohoutek, J., Kužílek, V., Machálek, P., Matějů, V., Matoušek, J., Matoušek, M., Mestřík, V., Novák, J., Ocelka, T., Pekárek, V., Petrlík, J., Petira, O., Provazník, K., Punčochář, M., Rieder, M., Ruprich, J., Sáňka, M., Tomaniová, M., Vácha, R., Volka, K., Zbíral, J., 2006. The National Implementation Plan for Implementation of the Stockholm Convention in the Czech Republic. Brno.

Holoubek, I., Klánová, J., Jarkovský, J., Kohoutek, J., Jarkovský, J., Kohoutek, J., Jarkovský, J., Kohoutek, J., 2007a. Trends in background levels of persistent organic pollutants at Kosetice observatory, Czech Republic. Part I. Ambient air and wet deposition 1996-2005. J. Environ. Monit. 9, 557–563. https://doi.org/10.1039/b700750g.

Holoubek, I., Klánová, J., Jarkovský, J., Kubík, V., Helešic, J., 2007b. Trends in background levels of persistent organic pollutants at Kosetice observatory, Czech Republic. Part II. Aquatic and terrestrial environments 1996-2005. J. Environ. Monit. 9, 564–571. https://doi.org/10.1039/b701096f.

Holoubek, I., Dusek, L., Sáňka, M., Hofman, J., Cupr, P., Jarkovský, J., Zbíral, J., Klánová, J., 2009. Soil burdens of persistent organic pollutants–their levels, fate and risk. Part I. Variation of concentration ranges according to different soil uses and locations. Environ. Pollut. 157, 3207–3217. https://doi.org/10.1016/j.envpol.2009.05.031.

Holt, E., Bohlin-Nizzetto, P., Borůvková, J., Harner, T., Kalina, J., Melymuk, L., Klánová, J., 2017. Using long-term air monitoring of semi-volatile organic compounds to evaluate the uncertainty in polyurethane-disk passive sampler-derived air concentrations. Environ. Pollut. 220, 1100–1111. https://doi.org/10.1016/j.envpol.2016.11.030.

de Hoogh, K., Wang, M., Adam, M., Badaloni, C., Beelen, R., Birk, M., Cesaroni, G., Cirach, M., Declercq, C., Dėdelė, A., Dons, E., De Nazelle, A., Eeftens, M., Eriksen, K.T., Eriksson, C., Fischer, P., Gražulevičienė, R., Gryparis, A., Hoffmann, B., Jerrett, M., Katsouyanni, K., Iakovides, M., Lanki, T., Lindley, S., Madsen, C., Mölter, A., Mosler, G., Nádor, G., Nieuwenhuijsen, M., Pershagen, G., Peters, A., Phuleria, H., Probst-Hensch, N., Raaschou-Nielsen, O., Quass, U., Ranzi, A., Stephanou, E., Sugiri, D., Schwarze, P., Tsai, M.Y., Yli-Tuomi, T., Varró, M.J., Vienneau, D., Weinmayr, G., Brunekreef, B., Hoek, G., 2013. Development of land use regression models for particle composition in twenty study areas in Europe. Environ. Sci. Technol. 47, 5778–5786. https://doi.org/10.1021/es400156t.

de Hoogh, K., Korek, M., Vienneau, D., Keuken, M., Kukkonen, J., Nieuwenhuijsen, M.J., Badaloni, C., Beelen, R., Bolignano, A., Cesaroni, G., Pradas, M.C., Cyrys, J., Douros, J., Eeftens, M., Forastiere, F., Forsberg, B., Fuks, K., Gehring, U., Gryparis, A., Gulliver, J., Hansell, A.L., Hoffmann, B., Johansson, C., Jonkers, S., Kangas, L., Katsouyanni, K., Künzli, N., Lanki, T., Memmesheimer, M., Moussiopoulos, N., Modig, L., Pershagen, G., Probst-Hensch, N., Schindler, C., Schikowski, T., Sugiri, D., Teixidó, O., Tsai, M.Y., Yli-Tuomi, T., Brunekreef, B., Hoek, G., Bellander, T., 2014. Comparing land use regression and dispersion modelling to assess residential exposure to ambient air pollution for epidemiological studies. Environ. Int. 73, 382–392. https://doi.org/10.1016/j.envint.2014.08.011.

de Hoogh, K., Gulliver, J., Donkelaar, A. van, Martin, R.V., Marshall, J.D., Bechle, M.J., Cesaroni, G., Pradas, M.C., Dedele, A., Eeftens, M., Forsberg, B., Galassi, C., Heinrich, J., Hoffmann, B., Jacquemin, B., Katsouyanni, K., Korek, M., Künzli, N., Lindley, S.J., Lepeule, J., Meleux, F., de Nazelle, A., Nieuwenhuijsen, M., Nystad, W., Raaschou-Nielsen, O., Peters, A., Peuch, V.H., Rouil, L., Udvardy, O., Slama, R., Stempfelet, M., Stephanou, E.G., Tsai, M.Y., Yli-Tuomi, T., Weinmayr, G., Brunekreef, B., Vienneau, D., Hoek, G., 2016. Development of West-European PM 2.5 and NO 2 land use regression models incorporating satellite-derived and chemical transport modelling data. Environ. Res. 151, 1–10. https://doi.org/10.1016/j.envres.2016.07.005.

de Hoogh, K., Chen, J., Gulliver, J., Hoffmann, B., Hertel, O., Ketzel, M., Bauwelinck, M., van Donkelaar, A., Hvidtfeldt, U.A., Katsouyanni, K., Klompmaker, J., Martin, R.V., Samoli, E., Schwartz, P.E., Stafoggia, M., Bellander, T., Strak, M., Wolf, K., Vienneau, D., Brunekreef, B., Hoek, G., 2018. Spatial PM 2.5 , NO2 , O3 and BC models for Western Europe – evaluation of spatiotemporal stability. Environ. Int 120, 81–92. https://doi.org/10.1016/j.envint.2018.07.036.

Hulek, R., Boruvkova, J., Grehugor, J., Kalina, J., Bednářová, Z., Sebkova, K., Melkes, O., Salko, M., Novak, R., Jarkovský, J., Dusek, L., Klanova, J., 2014. Global Monitoring Plan of the Stockholm Convention on Persistent Organic Pollutants: Visualisation and on-Line Analysis of Global Levels of Chemicals in Air, Water, Breast Milk and Blood [WWW Document]. Masaryk Univ http://www.pops-gmp.org/visualization-2014 (accessed 6.17.20).

Jamshidi, A., Hunter, S., Hazrati, S., Harrad, S.J., Hazrati, S., Harrad, S.J., 2007. Concentrations and chiral signatures of polychlorinated biphenyls in outdoor and indoor air and soil in a major U.K. conurbation. Environ. Sci. Technol. 41, 2153–2158.

Jedynska, A., Hoek, G., Wang, M., Eeftens, M., Cyrys, J., Keuken, M., Ampe, C., Beelen, R., Cesaroni, G., Forastiere, F., Cirach, M., de Hoogh, K., De Nazelle, A., Nystad, W., Declercq, C., Eriksen, K.T., Dimakopoulou, K., Lanki, T., Meliefste, K., Nieuwenhuijsen, M.J., Yli-Tuomi, T., Raaschou-Nielsen, O., Brunekreef, B., Kooter, I.M., 2014. Development of land use regression models for elemental, organic carbon, PAH, and Hopanes/Steranes in 10 ESCAPE/TRANSPHORM European study areas. Environ. Sci. Technol. 48, 14435–14444. https://doi.org/10.1021/es502568z.

Jedynska, A., Hoek, G., Wang, M., Yang, A., Eeftens, M., Cyrys, J., Keuken, M., Ampe, C., Beelen, R., Cesaroni, G., Forastiere, F., Cirach, M., de Hoogh, K., De Nazelle, A., Nystad, W., Akhlaghi, H.M., Declercq, C., Stempfelet, M., Eriksen, K.T., Dimakopoulou, K., Lanki, T., Meliefste, K., Nieuwenhuijsen, M., Yli-Tuomi, T., Raaschou-Nielsen, O., Janssen, N.A.H., Brunekreef, B., Kooter, I.M., 2017. Spatial variations and development of land use regression models of oxidative potential in ten European study areas. Atmos. Environ. 150, 24–32. https://doi.org/10.1016/j.atmosenv.2016.11.029.

Jiřík, V., Machaczka, O., Miturová, H., Tomášek, I., Šlachtová, H., Janoutová, J., Velická, H., Janout, V., 2016. Air pollution and potential health risk in Ostrava region – a review. Cent. Eur. J. Public Health 24, S4–S17. https://doi.org/10.21101/cejph.a4533.

Kalberer, M., Henne, S., Prevot, A.S.H., Steinbacher, M., 2004. Vertical transport and degradation of polycyclic aromatic hydrocarbons in an Alpine Valley. Atmos. Environ. 38, 6447–6456. https://doi.org/10.1016/j.atmosenv.2004.06.021.

Kalina, J., Scheringer, M., Borůvková, J., Kukucka, P., Přibylová, P., Sáňka, O., Melymuk, L., Vana, M., Klánová, J., 2018. Characterizing spatial diversity of passive sampling sites for measuring levels and trends of semivolatile organic chemicals. Environ. Sci. Technol. 52, 10599–10608. https://doi.org/10.1021/acs.est.8b03414.

Keyte, I.J., Harrison, R.M., Lammel, G., 2013. Chemical reactivity and long-range transport potential of polycyclic aromatic hydrocarbons–a review. Chem. Soc. Rev. 42, 9333–9391.

Kubošová, K., Komprda, J., Jarkovský, J., Sáňka, M., Hájek, O., Dušek, L., Holoubek, I., Klánová, J., 2009. Spatially resolved distribution models of POP concentrations in soil: a stochastic approach using regression trees. Environ. Sci. Technol. 43, 9230–9236. https://doi.org/10.1021/es902076y.

Lammel, G., Klánová, J., Kohoutek, J., Prokeš, R., Ries, L., Stohl, A., 2009. Observation and origin of organochlorine compounds and polycyclic aromatic hydrocarbons in the free troposphere over central Europe. Environ. Pollut. 157, 3264–3271. https://doi.org/10.1016/j.envpol.2009.05.028.

Larkin, A., Geddes, J.A., Martin, R.V., Xiao, Q., Liu, Y., Marshall, J.D., Brauer, M., Hystad, P., 2017. Global land use regression model for nitrogen dioxide air pollution. Environ. Sci. Technol. 51, 6957–6964. https://doi.org/10.1021/acs.est.7b01148.

Lu, M., Soenario, I., Helbich, M., Schmitz, O., Hoek, G., van der Molen, M., Karssenberg, D., 2020. Land use regression models revealing spatiotemporal co-variation in NO2, NO, and O3 in the Netherlands. Atmos. Environ. 223, 117238. https://doi.org/10.1016/j.atmosenv.2019.117238.

Masri, S., Li, L., Dang, A., Chung, J.H., Chen, J.C., Fan, Z.H. (Tina), Wu, J., 2018. Source characterization and exposure modeling of gas-phase polycyclic aromatic hydrocarbon (PAH) concentrations in Southern California. Atmos. Environ. 177, 175–186. https://doi.org/10.1016/j.atmosenv.2018.01.014.

Melymuk, L., Robson, M., Helm, P.A., Diamond, M.L., 2012. PCBs, PBDEs, and PAHs in Toronto air: spatial and seasonal trends and implications for contaminant transport. Sci. Total Environ. 429, 272–280. https://doi.org/10.1016/j.scitotenv.2012.04.022.

Melymuk, L., Robson, M., Helm, P.A., Diamond, M.L., 2013. Application of land use regression to identify sources and assess spatial variation in urban SVOC concentrations. Environ. Sci. Technol. 47, 1887–1895. https://doi.org/10.1021/es3043609.

Melymuk, L., Bohlin-Nizzetto, P., Prokeš, R., Kukučka, P., Klánová, J., 2016. Sampling artifacts in active air sampling of semivolatile organic contaminants: comparing theoretical and measured artifacts and evaluating implications for monitoring networks. Environ. Pollut. 217, 97–106. https://doi.org/10.1016/j.envpol.2015.12.015.

Motelay-Massei, A., Harner, T., Shoeib, M., Diamond, M.L., Stern, G.A., Rosenberg, B., 2005. Using passive air samplers to assess urban-rural trends for persistent organic pollutants and polycyclic aromatic hydrocarbons. 2. Seasonal trends for PAHs, PCBs, and organochlorine pesticides. Environ. Sci. Technol. 39, 5763–5773.

Muñoz-Arnanz, J., Roscales, J.L., Ros, M., Vicente, A., Jiménez, B., 2016. Towards the implementation of the Stockholm Convention in Spain: five-year monitoring (2008–2013) of POPs in air based on passive sampling. Environ. Pollut. 217, 107–113. https://doi.org/10.1016/j.envpol.2016.01.052.

Noth, E.M., Hammond, S.K., Biging, G.S., Tager, I.B., 2011. A spatial-temporal regression model to predict daily outdoor residential PAH concentrations in an epidemiologic study in Fresno, CA. Atmos. Environ. 45, 2394–2403. https://doi.org/10.1016/j.atmosenv.2011.02.014.

Noth, E.M., Lurmann, F., Northcross, A., Perrino, C., Vaughn, D., Hammond, S.K., 2016. Spatial and temporal distribution of polycyclic aromatic hydrocarbons and elemental carbon in Bakersfield, California. Air Qual. Atmos. Health 9, 899–908. https://doi.org/10.1007/s11869-016-0399-y.

Polidori, A., Kwon, J., Turpin, B.J., Weisel, C., 2010. Source proximity and residential outdoor concentrations of PM 2.5, OC, EC, and PAHs. J. Expo. Sci. Environ. Epidemiol. 20, 457–468. https://doi.org/10.1038/jes.2009.39.

Přibylová, P., Kares, R., Borůvková, J., Cupr, P., Prokeš, R., Kohoutek, J., Holoubek, I., Klánová, J., 2012. Levels of persistent organic pollutants and polycyclic aromatic hydrocarbons in ambient air of Central and Eastern Europe. Atmos. Pollut. Res. 3, 494–505.

Robson, M., Harrad, S.J., 2004. Chiral PCB signatures in air and soil: implications for atmospheric source apportionment. Environ. Sci. Technol. 38, 1662–1666.

Růžičková, P., Klánová, J., Cupr, P., Lammel, G., Holoubek, I., 2008. An assessment of air-soil exchange of polychlorinated biphenyls and organochlorine pesticides across central and southern Europe. Environ. Sci. Technol. 42, 179–185.

Sáňka, O., Melymuk, L., Čupr, P., Dvorská, A., Klánová, J., 2014. Dispersion modeling of selected PAHs in urban air: a new approach combining dispersion model with GIS and passive air sampling. Atmos. Environ. 96, 88–95. https://doi.org/10.1016/j.atmosenv.2014.07.002.

Schuster, J.K., Gioia, R., Sweetman, A.J., Jones, K.C., 2010. Temporal trends and controlling factors for polychlorinated biphenyls in the UK atmosphere (1991-2008). Environ. Sci. Technol. 44, 8068–8074. https://doi.org/10.1021/es102134d.

Shunthirasingham, C., Gawor, A., Hung, H., Brice, K.A., Su, K., Alexandrou, N., Dryfhout-Clark, H., Backus, S., Sverko, E., Shin, C., Park, R., Noronha, R., 2016. Atmospheric concentrations and loadings of organochlorine pesticides and polychlorinated biphenyls in the Canadian Great Lakes Basin (GLB): spatial and temporal analysis (1992–2012). Environ. Pollut. 217, 124–133. https://doi.org/10.1016/j.envpol.2016.01.039.

Sram, R.J., Dostal, M., Libalova, H., Rossner, P., Rossnerova, A., Svecova, V., Topinka, J., Bartonova, A., 2013. The European hot spot of B[a]P and PM 2.5 exposure—the Ostrava Region, Czech Republic: health research results. ISRN Public Health 2013, 1–12. https://doi.org/10.1155/2013/416701.

Tao, S., Wang, Y., Wu, S., Liu, S., Dou, H., Liu, Y., Lang, C., Hu, F., Xing, B., 2007. Vertical distribution of polycyclic aromatic hydrocarbons in atmospheric boundary layer of Beijing in winter. Atmos. Environ. 41, 9594–9602. https://doi.org/10.1016/j.atmosenv.2007.08.026.

Tørseth, K., Aas, W., Breivik, K., Fjeraa, a.M., Fiebig, M., Hjellbrekke, A.-G., Lund Myhre, C., Solberg, S., Yttri, K.E., 2012. Introduction to the European Monitoring and Evaluation Programme (EMEP) and observed atmospheric composition change during 1972-2009. Atmos. Chem. Phys. 12, 5447–5481. https://doi.org/10.5194/acp-12-5447-2012.

UNEP, 2017. Stockholm Convention on Persistent Organic Pollutants (POPs).

Van Nunen, E., Vermeulen, R., Tsai, M.Y., Probst-Hensch, N., Ineichen, A., Davey, M., Imboden, M., Ducret-Stich, R., Naccarati, A., Raffaele, D., Ranzi, A., Ivaldi, C., Galassi, C., Nieuwenhuijsen, M., Curto, A., Donaire-Gonzalez, D., Cirach, M., Chatzi, L., Kampouri, M., Vlaanderen, J., Meliefste, K., Buijtenhuijs, D., Brunekreef, B., Morley, D., Vineis, P., Gulliver, J., Hoek, G., 2017. Land use regression models for ultrafine particles in six European areas. Environ. Sci. Technol. 51, 3336–3345. https://doi.org/10.1021/acs.est.6b05920.

Venier, M., Salamova, A., Hites, R.A., 2019. How to distinguish urban vs. agricultural sources of persistent organic pollutants? Curr. Opin. Environ. Sci. Heal 8, 23–28. https://doi.org/10.1016/J.COESH.2019.01.005.

Vienneau, D., De Hoogh, K., Bechle, M.J., Beelen, R., Van Donkelaar, A., Martin, R.V., Millet, D.B., Hoek, G., Marshall, J.D., 2013. Western european land use regression incorporating satellite- and ground-based measurements of NO2 and PM10. Environ. Sci. Technol. 47, 13555–13564. https://doi.org/10.1021/es403089q.

Vizcaino, P., Lavalle, C., 2018. Development of European NO2 Land Use Regression Model for present and future exposure assessment: implications for policy analysis. Environ. Pollut. 240, 140–154. https://doi.org/10.1016/j.envpol.2018.03.075.

Wang, M., Beelen, R., Eeftens, M., Meliefste, K., Hoek, G., Brunekreef, B., 2012. Systematic evaluation of land use regression models for NO2. Environ. Sci. Technol. 46, 4481–4489. https://doi.org/10.1021/es204183v.

Wania, F., Shunthirasingham, C., 2020. Passive air sampling for semi-volatile organic chemicals. Environ Sci Process Impacts 22, 1925–2002. https://doi.org/10.1039/D0EM00194E.

White, K.B., Kalina, J., Scheringer, M., Přibylová, P., Kukučka, P., Kohoutek, J., Prokeš, R., Klánová, J., 2020. Temporal trends of persistent organic pollutants across Africa after a decade of MONET passive air sampling. Environ. Sci. Technol. https://doi.org/10.1021/acs.est.0c03575.

Wu, S.P., Tao, S., Liu, W.X., 2006. Particle size distributions of polycyclic aromatic hydrocarbons in rural and urban atmosphere of Tianjin, China. Chemosphere 62, 357–367. https://doi.org/10.1016/j.chemosphere.2005.04.101.

APPENDIX 16

Melymuk, Lisa, Pernilla Bohlin-Nizzetto, Ondřej Sáňka, Karla Pozo, and Jana Klánová. 2014. "Current Challenges in Air Sampling of Semivolatile Organic Contaminants: Sampling Artifacts and Their Influence on Data Comparability." *Environmental Science & Technology* 48 (24): 14077-91. https://doi.org/10.1021/es502164r

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Current Challenges in Air Sampling of Semivolatile Organic Contaminants: Sampling Artifacts and Their Influence on Data Comparability

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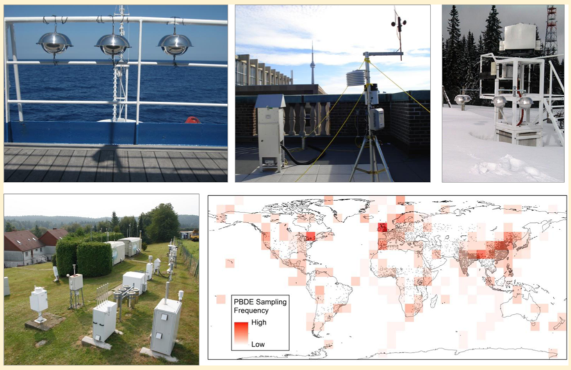
Supporting Information

ABSTRACT: With current science and policy needs, more attention is being given to expanding and improving air sampling of semivolatile organic contaminants (SVOCs). However, a wide range of techniques and configurations are currently used (active and passive samplers, different deployment times, different sorbents, etc.) and as the SVOC community looks to assess air measurements on a global scale, questions of comparability arise. We review current air sampling techniques, with a focus on sampling artifacts that can lead to uncertainties or biases in reported concentrations, in particular breakthrough, degradation, meteorological influences, and assumptions regarding passive sampling. From this assessment, we estimate the bias introduced for SVOC concentrations from all factors. Due to the effects of breakthrough, degradation, particle fractions and sampler uptake periods, some current passive and active sampler configurations may underestimate certain SVOCs by 30−95%. We then recommend future study design, appropriateness of sampler types for different study goals, and finally, how the SVOC community should move forward in both research and monitoring to best achieve comparability and consistency in air measurements.



INTRODUCTION AND BACKGROUND

Recent scientific and policy discussion has addressed air sampling of semivolatile organic compounds (SVOCs)/persistent organic pollutants (POPs), particularly regarding long-term monitoring networks and the data needs of the SVOC community.[1−3] SVOCs are organic compounds with vapor pressures typically between ∼1 and 10^{-10} Pa. Many of the SVOCs are classified as POPs, and are subject to international agreements and conventions such as the Aarhus protocol on POPs (1998) under the Convention on Long-Range Transboundary Air Pollution (LRTAP, 1979) (http://www.unece.org/env/lrtap/) and the Stockholm Convention (SC) on POPs (2001/2004) (www.pops.int). These conventions aim to eliminate or restrict the production and emission of a range of SVOCs, notably organochlorine pesticides (OCPs), polychlorinated biphenyls (PCBs), polychlorinated dibenzo-*p*-dioxins and furans (PCDD/Fs), polybrominated diphenyl ethers (PBDEs), polycyclic aromatic hydrocarbons (PAHs), per- and polyfluorinated compounds (PFCs). To provide information on long-term trends and effectiveness of regulatory actions, air monitoring strategies have been established,[1] including the European Monitoring and Evaluation Programme (EMEP) for LRTAP compounds (1999) and the Global Monitoring Plan (GMP) for SC compounds (2008).[1] Monitoring networks require appropriate sampling strategies to provide reliable and broadly comparable air data and current regional monitoring networks, with differences in sampling methodology and procedures, cannot adequately provide the data needed for global assessments, such as the GMP.[1] Furthermore, scientific concern is also growing regarding currently unregulated SVOCs such as endocrine disrupting compounds (EDCs).[4]

Beyond the policy demands of international conventions, the SVOC scientific community also requires air sampling to understand the fate and transport of SVOCs and to provide necessary information for scientific models and human exposure assessments.[2] This type of data is often produced through smaller-scale case studies or local- or regional-scale monitoring networks. While less standardized than long-term monitoring networks, some very critical knowledge of SVOCs has come from case studies, as they allow specific questions to be addressed in a focused manner. In particular, they provide a platform for testing samplers (e.g., polyurethane foam (PUF) passive air samplers,[5] XAD-passive air samplers[6]) and provide data in vulnerable areas that are not covered by long-term monitoring networks (e.g., e-waste sites in China[7] or Africa, intensive agricultural regions in India or South America[8−10]).

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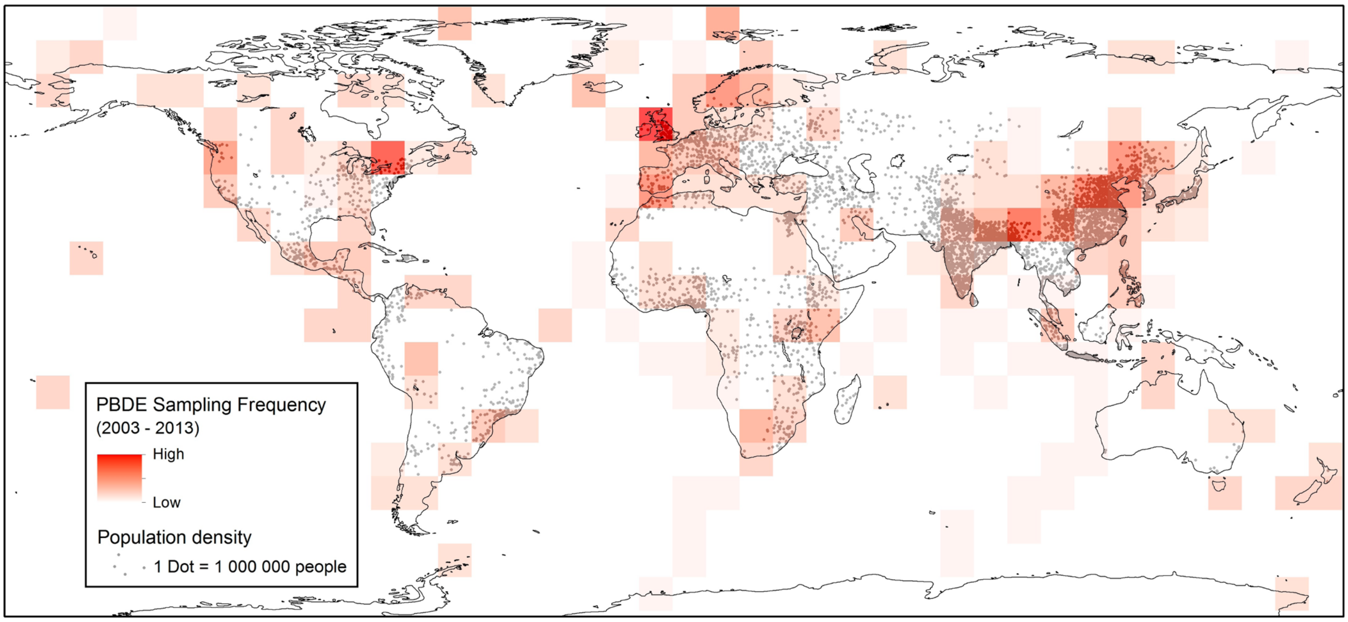

Figure 1. Global distribution of PBDE measurements in air (active and passive) from 2003 to 2013, as an example of the unbalanced spatial distribution of SVOC sampling. Shading is used to aid in visualization of aggregated sampling distribution and frequency. Darker orange indicates a high density of sampling (e.g., regular sampling over many years), while the lightest orange indicates collection of a single event sample (e.g., ship cruises). Dots reflect global population density as of 2000 (http://sedac.ciesin.columbia.edu/gpw).

Together, monitoring networks and case studies provide our information on the global distribution of SVOCs, their atmospheric transport, fate, and relationship with sources. However, there are still crucial gaps and biases in this information which may lead to uncertainties in comparability of data, namely spatial distribution and temporal coverage of measurements, and lack of sampling standardization within and between networks/case-studies. Problems related to spatial distributions of established networks are discussed by Hung et al.[2] and Klánová and Harner.[1] The uneven spatial distribution is somewhat improved when one considers the contribution of case studies (Supporting Information (SI) Figure S1b) and new/planned monitoring networks in Australia, Southeast Asia, and South America,[2] but the issue of data comparability remains. Additionally, there are discrepancies in the spatial distribution of measurements and the spatial scale of concentration variations; SVOCs vary on small scales[11−13] and current sampling distributions infrequently address this. There are also discrepancies in the frequency of sampling at each site, with some regions having repeated continuous monitoring capturing seasonal trends and short-term variability, and others with only one or two reported measurements in the past ten years (Figure 1). Although individual monitoring networks are effective at accomplishing their respective goals (e.g., providing data for modeling exercises, or broad spatial coverage) the different network structures may lead to problems with the intercomparability of scientific results. With increasing international collaboration and data sharing (e.g., the GMP) the question of inter-network comparability is brought to the forefront.[1] There are a wide range of techniques used across different networks and case studies, including active and passive air samplers, and differences in sampling volumes, sorbents, sampling length and sampling frequencies. There has never been an agreed mandate to establish standardized procedures within and between SVOC networks and there is a lack of quality standard operating procedures for SVOC sampling. While some studies have assessed comparability between samplers or sampling networks,[14−21] efforts never went beyond these to address the potential implications. Most current discussion focuses on the problems related to spatial distributions, as well as analytical quality assurance and quality control (e.g., interlaboratory or inter-network comparisons[14,16,17,22,23]), but little attention is given to sampling artifacts or spatial intercomparability. Herein, we review and discuss influences on the most frequently used SVOC air sampling methods/techniques in both established networks and case studies and analyze how they impact measurements and thus data comparability. We then make recommendations on what sampling techniques are appropriate for particular study goals, how to enhance data comparability, and what is needed to improve SVOC air measurements.

■ SAMPLING TECHNIQUES

Active Air Sampling Methods. Active air samplers (AAS) are currently perceived as the most accurate method of obtaining SVOC air concentrations as they accumulate both gas and particle phase compounds under a controlled flow. Despite identified sampling artifacts,[24] this technique is almost unchanged over the past 40 years.[25−27]

AAS are commonly classified as either high volume or low volume samplers, with the main differences being the flow rate and sample collection time. In high volume samplers, flow rates are typically 15−80 m^3/hour[16,28,29] yielding total sample volumes of >400 m^3, although there is a large variation in what is considered a high volume sample. For example, the Integrated Atmospheric Deposition Network (IADN) typically collects ~820 m^3 of air at American sites, and ~350 m^3 at Canadian sites,[16] while some Arctic sampling networks collect up to 13000 m^3.[30] Low volume air samplers typically have flow rates of <3 m^3/hour.[15,31−34] This often results in small sample volumes (e.g., <200 m^3), but low volume samplers can also be used for longer deployment times, for example, 7−14 days of

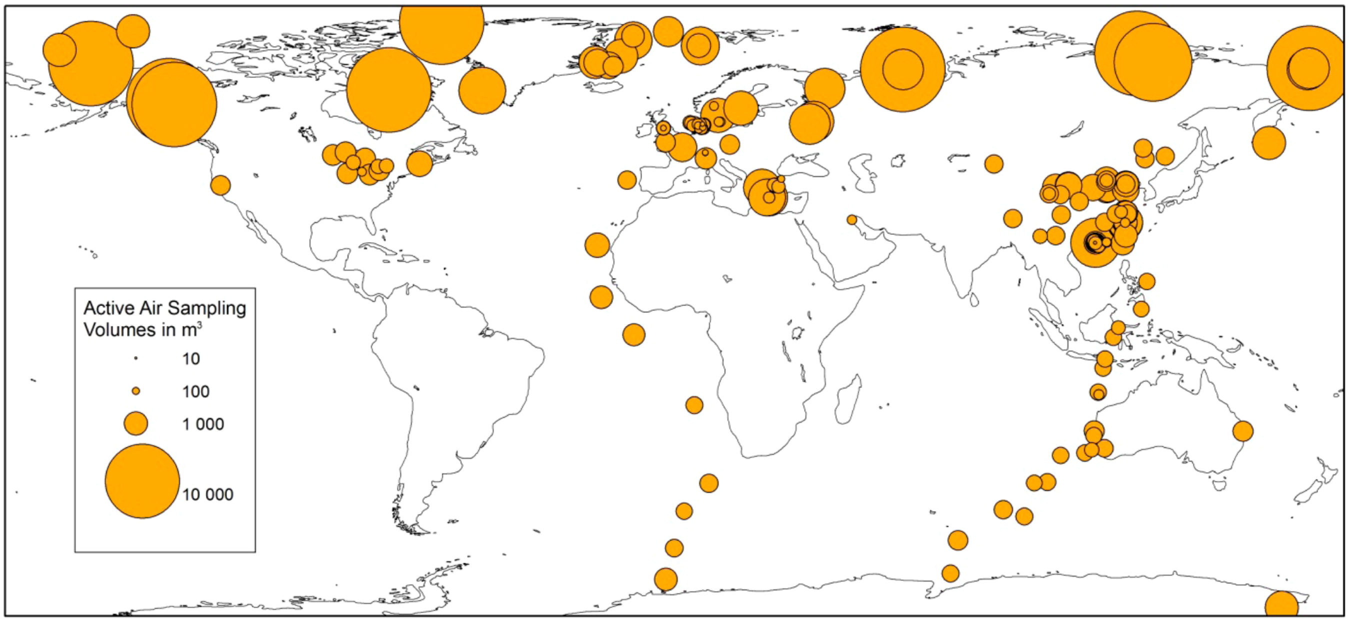

Figure 2. Distribution of active air sampling volumes used in measurements of PBDEs from 2003 to 2013. Studies that did not report a sample volume are excluded. There is a general trend of larger volumes collected in remote areas, particularly the Arctic, however there is also significant variability in air volumes that is not directly related to the expected concentrations at a given site and this variability may affect what is reported as concentrations in a given location (e.g., occurrence of either breakthrough or lack of detection).

Table 1. Common Active and Passive Sampling Techniques for SVOCs

sampler	high volume sampler	low volume sampler	double-bowl PUF PAS	XAD PAS	SIP PAS
type of sampler	active	active	passive	passive	passive
typical sampling volume (m^3)	600 m^3	100 m^3	80–400 m^3	100–200 m^3	100–500 m^3
typical sampling rate/flow rate	15–80 m^3/hour	<3 m^3/hour	0.3–20 m^3/day	0.4–0.8 m^3/day	4–12 m^3/day
sampling time	hours to month	days to weeks	weeks to months	months to year	weeks to months
gas-phase sorbent	PUF, XAD or combination of PUF, XAD	PUF, XAD or combination of PUF, XAD	PUF	XAD	XAD-impregnated PUF
particle-phase sampling	GFF or QFF	GFF or QFF	debatable. inconsistent. with lower sampling rate than gas-phase (10%).	no	debatable. inconsistent. with lower sampling rate than gas-phase (10%).
use in long-term monitoring networks?	yes (inter alia EMEP, IADN, AMAP, MONAIRNET)	rarely (selected OSPAR stations)	yes (GAPS, MONET)	yes (GAPS)	yes (GAPS)

continuous sampling resulting in air volumes of 500–1000 m^3.[31−33] Low volume sampling is rarely used in long-term sampling networks; only the Coordinated Atmospheric Monitoring Programme of OSPAR and EMEP use low volume samplers at selected stations.[31,35]

Environmental parameters and SVOC concentrations vary widely, for example, very low concentrations in remote/polar regions or high temperatures in tropical regions, and thus it can be problematic to use identical sampling configurations in all locations. The use of different sampling configurations results in different sampling artifacts. Sampling events can only be compared if a sufficient measure of accuracy has been reached. Current sampling procedures/protocols have wide variations in sampled volumes that may not always reflect the best choice for the temperature and concentrations in a region, and rather may simply rely on established sampling protocols not tailored to specific sampling conditions (Figure 2). While these choices allow individual networks to have long-term temporal data continuity, they may reduce the accuracy of results and comparability between networks or studies. A greater analysis of the influences of sample volume and sampling media is given in the following sections.

Passive Air Sampling Methods. Passive air sampling techniques for SVOCs were introduced in the end of 1990s to simplify and reduce the costs for air monitoring of SVOCs. Unlike AAS, passive air samplers (PAS) do not need electricity; instead chemicals are trapped by diffusive uptake to a sorbent material. PAS have enabled broader monitoring network distribution and, as a result, increased the spatial breadth of SVOC measurements.[1,36] Several types of PAS have been evaluated and used during the last 10–15 years, and a few of them have received greater attention and been implemented in monitoring networks (Table 1). Currently, the disk-shaped PUF–PAS,[5] and the XAD-resin based PAS[6] are most used in monitoring networks. Recently, an XAD sorbent-impregnated PUF (SIP) PAS has also been implemented in the Global Atmospheric Passive Sampling network (GAPS).[37] Other PAS samplers have been used (e.g., POGs,[38] SPMDs[39]), and new techniques are under development,[40] however these are largely used in case studies. The available PAS differ in characteristics and applicability, which affects data comparability, as described below. For an in-depth review of PAS theory and application we recommend dedicated articles on theoretical framework,[41] calibration,[42] indoor use,[43] new developments and techniques.[40,44]

SAMPLING INFLUENCES

Below, we describe and discuss some of the major sampling influences for both AAS and PAS. All air sampling techniques have biases that may affect measurements and thereby also the comparability of results between different sampling events. We consider: (i) is the sampling technique accurate—does it provide the real picture of atmospheric levels of the target contaminants, (ii) is the sampling technique robust—how sensitive is it to environmental conditions, and (iii) are data from different techniques comparable—do we get the same number when using different sampling techniques and different sampling approaches? A focus is often on analytical procedures and intercomparisons of chemical analyses;[17,23] comparisons of sampling procedures/methodologies/techniques are less common. Yet, given the range of sampling techniques, the parameters of sample collection (e.g., sorbent type, sample volumes) should be carefully considered, as they have the potential to introduce significant biases. A summary of the advantages and disadvantages of individual sampler configurations is given in SI Table S1.

Sampling Influences—Active Air Sampling. *Sampling Head.* The most commonly used sampling heads in AAS are the TSP (total suspended particulates) and PM_{10} (particles with aerodynamic diameter <10 μm). TSP theoretically includes all airborne particulate matter, although in practice, the upper particle size cutoff of a TSP sampling head is between 50 and 100 μm, dependent on flow rate and ambient conditions.[45] As a result of new legislations within the European Union (EU), recently $PM_{2.5}$ (particles <2.5 μm) has also been suggested as a sampling threshold for monitoring networks. Different sampling heads and a specific particle cut-point can be useful, as particles <10 μm are most important for atmospheric chemistry and physics[46] while particles <2.5 μm are most relevant for human health and risk assessment estimates,[47] but may result in lower data comparability. The choice of sampling head should not result in large differences in reported SVOC concentrations, as the majority of SVOCs are typically associated with the finest particles (e.g., <1.5 μm), which are sampled by the TSP, PM_{10} and $PM_{2.5}$ sampling heads.[48−52] On average $PM_{2.5}$ represents >75% of TSP by mass and most of the SVOCs (>80%) are found in this fraction.[51,52] However, there may be circumstances when the use of different sampling heads leads to very different results, such as environments with high TSP, or industrial settings where SVOCs are found on coarser particles.

Sorbent and Filter Type. A range of different sorbent types are used for sampling gas-phase SVOCs in AAS (e.g., PUF, XAD, Tenax TA, polydimethylsiloxane), but published data on SVOCs is dominated by sampling with PUF and XAD, and we focus our discussion on these. PUF is a low density foam consisting of a polymer bound by urethane links that has been used in SVOC air sampling for over 40 years.[25] XAD is a styrene-divinylbenzene copolymer more recently used in AAS, either independently or in conjunction with PUF. Two types of XAD are commonly used in SVOC air sampling:[53] (i) Amberlite XAD-2, commonly used in AAS and XAD-PAS,[6,54,55] and (ii) Amberlite XAD-4, in active denuder samplers and SIP-PAS.[56,57] The main difference between XAD and PUF is sorptive capacity; for equal amounts of sorbent material, PUF has a much lower capacity than XAD, largely due to differences in surface area. PUF has a specific surface area of 0.007−0.035 m^2/g while the surface area of XAD-2 is 300−600 m^2/g.[53] On a practical basis, this means that samplers using XAD as the gas-phase sorbent can collect larger volumes of air before breakthrough occurs (for AAS) or be deployed for longer periods of time before equilibrium is reached (for PAS). However, PUF does have the advantage of lower cost and ease-of-use compared to XAD, as manipulation of the PUF is simple and blanks are often lower.

There are also differences in the surface bonding of particular compounds to PUF and XAD.[58,59] XAD is a stronger sorbent for PFCs, current-use pesticides (CUPs), and other emerging chemicals,[59,60] yet the differences are not fully understood, and research is ongoing. Both sorbents are influenced by temperature and humidity, but the amount of variability introduced by these parameters is unclear and needs further research.[53,59] Generally, they are linked to changes in sorptive capacities,[59] and thus effects can be significant when large seasonal shifts are expected (e.g., monsoon-impacted areas). The choice of PUF vs XAD should be based on the choice of compound to be sampled and the sampling parameters, but is often due to previous regional practices (e.g., PUF is more commonly used in AAS in Canada and Europe, while XAD is more commonly used in the U.S.). PUF/XAD combinations are seeing increasing global use in recent years (SI Figure S2).

Three types of filters are used in AAS for SVOCs: quartz fiber filters (QFF), glass fiber filters (GFF), and Teflon/Teflon-impregnated filters. GFF and QFF are most common, and are perceived to give comparable results,[61] although specific studies assessing differences between them are limited. The recommendations on filter type instead suggest that choice of filter should consider what, if any, additional analyses will be completed in conjunction with SVOCs (e.g., PM, SO_2, organic/elemental carbon, or metals).[45,62]

While the choice of sampling head determines the upper end of the particle sizes collected on the filter, the lower size cutoff is related to filter choice, but is more ambiguous. Typical GFFs or QFFs have a collection efficiency of >99%, but do not often have a specified fine particle cutoff. Particles <50 nm may pass through the filter and subsequently be trapped by the gas-phase sorbent material.[63] This might be an issue for ultrafine particles (typically <100 nm) which are of concern for human health.[64,65] Although numbers of ultrafine particles are high, their total mass is small and therefore they are challenging to quantify and require adaptations of current instrumentation or new sampler types. There is limited information on how ultrafine particles are distributed within an AAS, and this may contribute to uncertainty in gas-particle partitioning.

Breakthrough. An important consideration in AAS is what volume of air can be collected before the sampling medium experiences "breakthrough", that is, the loss of compounds downstream of the sampling medium. This can be a result of a saturated sampling medium or desorption of compounds from the sampling medium. Sampling design must achieve the right balance between collecting sufficient sample for analysis of lower concentration SVOCs, while not collecting so much volume as to have breakthrough of more volatile SVOCs. In breakthrough estimates or controlled experiments <10% breakthrough is considered an acceptable threshold.[66−68]

The key factors affecting breakthrough are (1) type and geometry of sampling medium, (e.g., PUF vs XAD), (2) target compound properties and concentration, (3) competition for the sampling medium (i.e., from other gas-phase compounds), (4) sample volume, (5) sampler flow rate, (6) temperature, and (7) humidity. Of these, sampling medium, sample volume and sampler flow rate are the most easily controlled, and therefore breakthrough is most often discussed in terms of what sampling medium and sample volume are appropriate for a given class of compounds.

Basic chromatographic relationships can be used to estimate breakthrough,[24,67] but given the number of variables influencing breakthrough, it can be more reliable to conduct small breakthrough experiments with the exact study configurations. Results from early breakthrough studies[67,68] tend to be used as an accepted framework for present studies and little consideration is given to breakthrough in typical sampling set-ups. Despite this, measurements and calculations suggest that in a typical high-volume AAS breakthrough of more volatile compounds (e.g., PCB-28, HCB, HCHs, fluorene) can occur already at <600 m^3.[24,34,59,66,68−70] For example, at tropical temperatures (e.g., around 30°C) a typical high volume AAS (2 PUF plugs, 600 m^3 of air, 24 h) can lose 10−15% of the PCB-28 due to breakthrough (calculations in SI). However, it should be noted that this is also highly temperature dependent: with the same sampling configuration at 5°C the breakthrough volume is >3000 m^3. Many current AAS configurations may experience breakthrough for lower molecular weight SVOCs, leading to systematic underestimation of these compounds; consideration of this is warranted in interpretation of results.

Filter Artifacts. The filter within the AAS (GFF or QFF) can be the site of two sampling artifacts which influence measured concentrations: blowoff and filter adsorption. Blowoff is the volatilization loss of SVOCs from the filter, thereby disproportionately reducing SVOC filter masses, while filter adsorption is the adsorptive partitioning of gaseous compounds onto the filter, increasing the mass of SVOCs on the filter.

This is affected by sampler configurations (e.g., flow rate, sampling time, filter type) and occurs largely under three conditions: (i) when there are large variations in the SVOC concentration in incoming air, (ii) when there are large temperature variations, (iii) when there is very high particle loading to filters. Under these conditions, the gas-particle distribution of SVOCs in the AAS may no longer be in equilibrium with the influent air, and filter blow-off can occur in the case of higher temperatures/lower concentrations, or filter adsorption in the case of lower temperatures/higher concentrations. By using a higher flow rate or longer sampling time, surface-sorbed SVOCs can more easily be stripped from particles than those that are less available for exchange with the atmosphere. In large sample volumes or in areas of high PM, overloading of filters can cause further artifacts. If the filter becomes physically blocked by PM, sampler flow rates can be reduced and thus true sample volumes could be lower, resulting in an underestimate of concentrations.

Filter adsorption and blowoff further vary by compound, depending on partitioning coefficients, ionic interactions, and hydrophilicity.[71,72] For example, significant filter adsorption was identified for perfluorooctanesulfonate (PFOS), and certain perfluoroalkyl carboxylic acids (PFCAs) in a high-volume AAS, resulting in overestimation of particle-associated fractions by up to 80%, while other PFCs did not have significant filter adsorption.[24]

If filter blowoff/adsorption occurs, bulk air concentrations may be correctly characterized, but gas-particle partitioning estimates may be incorrect. This is expected to be most significant for compounds with short-term temporal variations in concentrations[71] (such as CUPs) which may not be in equilibrium between the gas and particle phases, and those with intermediate gas-particle partitioning that vary between largely gas- and largely particle-associated at ambient air temperatures.

Degradation. Degradation of atmospheric SVOCs occurs as a natural process, caused by reaction with atmospheric reactive species, in particular hydroxyl radicals, ozone, and NO_3, or by photolysis (SI Figure S3). Degradation has been found to significantly affect ambient atmospheric concentrations, notably for PAHs through reaction with OH, ozone, and NO_3,[73,74] for PCBs and pesticides through reaction with OH,[75−78] and for PBDEs and novel flame retardants (NFRs) through photolysis.[79−81] While a sampler should provide a snapshot of the atmospheric SVOC concentrations which may be already affected by degradation, there is also the possibility for degradation to continue within the sampler, as atmospheric reactive species are drawn into the sampling medium along with the SVOCs themselves. Within-sampler degradation is an unwanted sampling artifact that can result in underestimation of ambient concentrations.

UV radiation has a minimal direct influence on within-sampler degradation due to the protection provided by the sampler housings, but there is the potential for degradation due to ozone, hydroxyl radicals and other atmospheric reactive species to continue within the sampler. Significant within-sampler degradation has been observed for PAHs both on filters[82,83] and in gas-phase sorbents.[84] Particle-phase PAH concentrations can typically be underestimated by 20−40% due to degradative losses on AAS filters, with underestimates >70% for particularly reactive compounds (e.g., benzo[a]-pyrene).[82,83,85] Gas-phase PAHs may also experience degradation within the sorbent, with losses of up to 50% due to reaction with both OH and ozone.[84,86]

The amount of potential within-sampler degradation depends on sampling conditions, site conditions, and compound, but the range of variability due to these factors is highly uncertain. For example, there are large spatial and temporal ranges in levels of atmospheric reactive species. On a global scale, ozone varies by a factor of 2−3, with higher levels in populated areas,[87] while OH varies by a factor of 6, with higher levels in tropical regions,[75,88] and there is potential for even higher local-scale variability in both ozone and OH.[89] Ozone can vary seasonally by a factor of 2 and diurnally to a lesser extent[87] while OH can vary up 10× both seasonally and diurnally.[75,88,90] Given the direct relationship between the amount of degradative losses and levels of atmospheric reactive species, within-sampler degradation may be significantly higher (up to 100%) if the concentrations of atmospheric reactant species are very high.[84,85] This can then affect the temporal (e.g., summer vs winter) and spatial comparability (urban vs remote) of measured concentrations. For example, estimates based on reaction with ozone[74,91] suggest that measured concentrations at an urban site in summer with 60 ppb ozone would be underreported by 30% due to particle phase reaction with ozone, while at a remote site in winter with 20 ppb ozone, there would be minimal within-sampler degradation. While site and season are very important for degradation, filter and sorbent type, sampling duration, amount and composition of PM, temperature and humidity also affect reactive losses.[74] The length of sampler deployment may have a minor influence on degradation,[74,85] but more characterization of these processes is needed.

Experimental evidence only exists for PAHs, but within-sampler degradation should also be of importance for other reactive SVOCs, according to their susceptibility to degradation in the atmosphere. Gas-phase compound degradation is largely via reaction with the OH radical,[73,86,92] and thus we expect other gas-phase compounds with OH reaction rate constants in the same range as PAHs to also have potential within-sampler degradative losses (SI Table S2). In contrast, on-filter degradation is largely through reaction with ozone,[82,86] and thus degradative losses of particle-associated compounds are important mainly for PAHs (SI Table S3). Based on rate constants and gas-particle partitioning, we estimate that PAHs, CUPs and some NFRs may experience significant within-sampler degradative losses.

Sampling Influences—Passive Air Sampling. Despite the benefits and widespread use of PAS[1] they are still associated with challenges and limitations affecting the comparability of measurements. These are important to recognize and control, but few of them can be fully quantified in a field deployment setting.

Calculation of Air Concentrations. The most relevant limitation of PAS is their lower accuracy compared to AAS.[5] PAS have a semiquantitative nature, since air concentrations can only be derived if an estimated sampling rate (m^3/day) is applied. The sampling rate is specific to the type of sampler (e.g., sorbent, sampler housing) and the environmental conditions (e.g., indoor, outdoor, meteorology), and can vary by orders of magnitude due to these differences.[6,93−96] In theory sampling rates should not vary by compound, but in practice large variations are found due to compound-specific behaviors or uncertainties in the methods of determining sampling rates. For example, measured sampling rates for PUF−PAS span 3 orders of magnitude (e.g., 0.02−20 m^3/day),[32,33,94,95,97−99] and some studies have observed systematic variations in sampling rates according to compound physical-chemical properties.[32,33,93,100] The choice of using one general sampling rate or nongeneric sampling rates is an important current discussion point within the PAS community. The use of nongeneric sampling rates (such as homologue-specific) can "correct" for some of the potential errors a general sampling rate may introduce under certain circumstances. For example, using a general sampling rate of ∼4 m^3/day[36,101] vs a nongeneric sampling rate of 0.3 m^3/day for benzo[a]pyrene[94] results in 13× concentration differences due to the choice of sampling rate alone. However, a general sampling rate for all SVOCs is often used in monitoring networks, as it is unclear whether the corrective potential of nongeneric sampling rates is significant considering the semiquantitative nature of PAS. It may not be a problem when the goal is to study the sum of compounds but may lead to errors if compound-specific concentrations or the SVOC fingerprint is of interest. The PAS community has not yet reached consensus regarding selection of appropriate sampling rates.

The methods for obtaining sampling rates add an additional level of uncertainty to their validity. Sampling rates are obtained from three methods: (1) use of depuration compounds (DCs)/performance reference compounds (PRCs),[102,103] (2) calibration studies with comparison to (i) continuous low volume AAS,[31,33,104] or (ii) intermittent high volume AAS,[18,19,59,105] or (3) modeling exercises[106,107] (e.g., applying a PUF-air partitioning coefficient). Method (i) is considered most accurate, and is particularly advantageous when calibrating compounds with high short-term variability, which could be biased with intermittent high volume AAS depending on the timing of sampling.[31] DCs account for site-specific environmental conditions but do not cover the full-range of compounds, as they are not applicable for particle-associated compounds, and cannot be used for XAD samplers due to their high sorptive capacity.

PAS for SVOCs are used as time-integrated samplers, and sampling rates are only applicable when the sampler is in the linear uptake phase.[41,95] The length of the linear uptake phase varies for compounds within a SVOC class as well as between classes and it is important to keep deployment times within the reported linear time frames for each compound. While the conventional PAS model assumes uniform distribution of a compound within the sorbent,[41,95] recent studies have identified that kinetic resistance within the sorbent material may be limiting and as such, model-based calculations may overestimate the length of the linear uptake phase.[59,100,108] This contention is supported by experimental work.[33,93] For example, penta+hexachlorobenzene and low molecular weight PAHs and PCBs enter a curvilinear uptake after 6−9 weeks,[93] and thus air concentrations from a three-month PUF−PAS deployment may be underestimates. Furthermore, the length of the linear uptake phase may be affected by environmental conditions such as temperature, wind speed and air concentrations, but this is not yet fully characterized.

Sorbent Material. Two sorbent types are commonly used in PAS: PUF and XAD. As with AAS, other sorbents have been used but not applied on a large scale. Their specific characteristics result in different sampling potentials. The PUF sorbent has a lower capacity and higher sampling rate than the XAD sorbent (Table 1).[31] Typical deployment times in established air monitoring networks are months (∼1−4 months) for PUF−PAS[36,101,109] and up to one year for XAD-PAS.[6] As a consequence, data from the two samplers are only comparable if sampling is conducted within the linear uptake phase of each sampler, and time weighted average concentrations from multiple PUF−PAS cover the same sampling period as XAD-PAS.

Another difference between the two sorbents is the type of compounds they can sample. While XAD is solely a gas-phase sampler, PUF is able to accumulate both gas-phase and particle-associated SVOCs, although particles are sampled with a lower accuracy and more variable sampling rates.[93,97] On the other hand, PUF−PAS have shown poorer or inconsistent performance for more volatile and polar SVOCs (e.g., CUPs, PFOS/PFOA), for which XAD perform well.[31,56,59,110] The low sorptive capacity of PUF−PAS for most PFCs[60,110] suggests different partitioning mechanisms for polar and nonpolar compounds. The SIP-PAS has been shown to be more effective for PFCs,[56] CUPs,[37] methyl siloxanes,[111] and phthalates.[112] However, consistency with SIP-PAS may be challenging, as XAD resin can be lost from the SIP disk during sampler deployment, particularly in windy, outdoor conditions,[113] thereby introducing added uncertainties.

PUF disks of different densities (0.021, 0.030, and 0.035 g cm^{-3}) are currently used by different research groups.[36,105,114,115] Although density is included in the theoretical calculation of PUF-air partition coefficients, the comparability of results from different densities is not fully understood. A higher density PUF disk has higher capacity for

SVOCs and thus a longer linear uptake phase. However, kinetic resistance within the PUF also differs with density. This can result in different uptake scenarios, with lower density PUFs having enhanced rates of SVOC transfer into the interior of the disk, whereas higher density PUF has a rapidly filling "surface compartment" and slower filling of the interior of the disk (e.g., a two-phase uptake mechanism).[114] Additionally, particle uptake may be enhanced in a lower-density PUF, which could be a contributing factor in the discrepancies found for particle associated PAHs (0.1 vs 5.0 m^3/day) in different studies.[93,96]

Sampler Housing. PAS sorbents are deployed inside a sampler housing (chamber) to reduce the influence of environmental factors that affect the performance of the PAS, such as wind speed and UV radiation. Studies have indicated that the PAS configuration, including the position of the sampling media (both PUF and XAD) inside the housing as well as the alignment of the housing (tight or open), affects uptake/accumulation and sampling rates.[108,116,117]

The housings used for PUF−PAS consist of two stainless steel bowls fixed on a common vertical axis around the PUF disk. This type of housing may vary slightly in geometry and deployment. Differences in the bowl geometries do not significantly affect uptake[114] but the way the housing is fixed results in different uptake scenarios, especially under windy conditions. A freely hanging sampler housing dampens the variation in external wind speed and results in small variations in sampling rates, while a fixed sampler results in up to 3× higher sampling rates under windy conditions.[115,118] The XAD-PAS housing consists of two parts: an inner stainless steel mesh tube in which the XAD is placed and an outer larger stainless steel cylinder acting as a protective shelter.[6] Differences in housing configuration for XAD-PAS and air movement within the sampler housing also cause significant variation in sampling rates (up to 50%).[108]

Degradation. No study has investigated degradation in PAS in detail. However, it is expected to affect the same compounds and be related to the same environmental variables as in AAS. Kennedy et al.[119] identified photodegradation when PUF−PAS were not shielded by a sampler housing and that this is reduced by the double bowl chamber; typical PAS housings should limit UV exposure. The long deployment times of PAS and thereby long exposures to atmospheric reactive species may exacerbate degradation, both of the SVOCs and the sorbent material itself. The effect on the structure and uptake capacity of the sorbent is unknown. Compound-specific degradation may also be a source of variability in sampler calibrations.

Environmental Factors/Location. The main causes of variability in PAS are environmental variables (e.g., wind speed, temperature, air concentrations) and most refinements to PAS aim to quantify/control these, through calibration of sampling rates, sampler housing design, use of DCs, etc. Despite these efforts, environmental factors continue to lead to bias/errors in estimated air concentrations.

Sampler housings are effective up to wind speeds of 4 m/s as they maintain the air flow within the chamber at less than ~1 m/s, which has a minor effect on sampling rates.[115] At wind speeds >5 m/s the sampler housing cannot sufficiently dampen outdoor winds, resulting in a higher air velocity within the chamber and a rapid increase in sampling rates.[115] This is supported by results from DCs, which have shown much higher sampling rates at windy, coastal, and mountain sites.[36,99,101] For XAD-PAS, sampling rates are also significantly higher under windy conditions.[106,120] In contrast to the PUF−PAS, the XAD-PAS sampling rates double between winds of 0 and 1 m/s, but the effect on sampling rates is less with higher wind speed.[6,120]

Temperature affects PUF and XAD-PAS in similar ways. Temperature controls the gas-particle partitioning of SVOCs, thus affecting the sampling rate (discussed below). However, temperature also affects the sampling medium: for example, higher temperature leads to higher diffusivity and thus higher sampling rate[95] and higher temperature leads to lower sorptive capacity and thus shorter linear uptake phase. The effect of these two factors on the overall sampling rate is complex and varies by compound,[106] but can lead to underestimation of concentrations in warm/tropical conditions if deployment times are the same as in colder/temperate regions. Further complicating matters, variations in temperature also affect ambient air concentrations (e.g., higher temperature leads to higher volatilization and thereby higher air concentrations) which impacts one of the major assumptions of the PAS theory, that the air concentration is constant.

Particle-Associated Compounds. A big question when using PAS for SVOCs is their performance for particle-associated compounds. The XAD-PAS is considered a purely gas-phase sampler while the PUF−PAS also accumulates particle-associated compounds. Current findings are not consistent; some suggest similar PUF−PAS sampling rates for gas- and particle-phase compounds,[33,96,121] others show poorer performance and up to 100× lower sampling rates for the particle-associated compounds.[93,105] The reason for inconsistent results is not known but may be related to site characteristics, PUF types, or analytical differences. High TSP values and high wind speeds may favor the uptake of particle-associated compounds to a level similar to gas-phase compounds. Another possible reason may be differences in PUF density, as mentioned above.[93] The influence of particle-associated compounds on overall sampling rates can be exacerbated at colder temperatures when a higher fraction of SVOCs are associated with particles. The variable results imply large errors and low comparability for particle-associated compounds between sites, and it is difficult to draw a general conclusion that is applicable to all or most of the sites in monitoring networks.

The ability to use PAS for particle-associated compounds is a key need of the SVOC community, as regulated and emerging SVOCs (PAHs, PBDEs, NFRs, PFCs, CUPs) tend to have high particle fractions. The applicability of PAS for these compounds is uncertain, and the subject of ongoing research.

■ CHALLENGES AND LIMITATIONS

Implications for Air Sampling. Given the aforementioned sources of bias and potential error in AAS and PAS, it is clear that each individual measurement is associated with some degree of uncertainty which may be a bigger contribution than the analytical uncertainty. However, more importantly, the uncertainty, accuracy, and precision of the measured values vary widely by sampling technique and sampler configuration, and thus there may be poor comparability between different data sets.

Key issues for comparability are (i) sampling times, sample volumes and frequencies, (ii) PUF−PAS assumptions about particles, (iii) PAS sampling rates, (iv) spatial coverage, and the associated larger range of environmental variables as sampling is expanded to more geographic regions.[2]

Table 2. Percent of Ambient Air Concentration Captured by Given Sampler Configurations[a]

example compounds	vapor pressure (Pa) at 0 °C, 25 °C	log octanol-air partitioning coefficient (K_{OA}) at 0 °C, 25 °C	atm. half-life (hrs) at 25 °C	estimated particle fraction at 0 °C, 25 °C	% of air concentration captured by sampler: high-volume sampler (24 h, 600 m^3) 0 °C	high-volume sampler 25 °C	PUF–PAS (90 day deployment) 0 °C	PUF–PAS 25 °C	XAD-PAS (1 year deployment) 0 °C	XAD-PAS 25 °C
PFOS	0.0973, 0.981	5.23, 4.84	917	0.00, 0.00	70% (1)	6% (1)	1% (3)	1% (3)	50% (3,4)	0% (3,4)
fluorene	0.630, 0.819	8.03, 6.74	15	0.01, 0.00	4% (1, 2)	3% (1, 2)	20% (2,3)	4% (2, 3)	100%	80% (3)
α-HCH	0.105, 0.376	8.37, 7.37	224	0.01, 0.00	70% (1)	20% (1)	60% (3)	20% (3)	100%	90% (3)
PCB-28	0.0314, 0.123	9.28, 8.06	108	0.03, 0.00	100%	100% (1)	90% (3)	50% (3)	100%	100%
chlorpyrifos	0.00399, 0.00536	9.56, 8.41	1.4	0.04, 0.01	0% (2)	0% (2)	80% (2,3)	60% (2, 3)	70% (2)	70% (2)
endosulfan	5.06×10^{-4}, 2.31×10^{-5}	10.58, 9.28	16	0.16, 0.03	90% (2)	90% (2)	80% (2, 3)	80% (2, 3)	80% (2)	80% (2)
PBDE-47	4.91×10^{-7}, 2.19×10^{-5}	12.09, 10.54	128	0.64, 0.15	100%	100%	50% (4)	90% (4)	40% (4)	90% (4)
benzo[a]pyrene	0.00550, 6.26×10^{-6}	12.29, 11.15	2.5	0.97, 0.76	60%	60% (2)	8% (2, 4)	20% (2, 4)	3% (4)	20% (4)
PBDE-209	1.01×10^{-8}, 2.23×10^{-4}	16.51, 15.26	3810	1.00, 1.00	100%	100%	10% (4)	10% (4)	0% (4)	0% (4)

[a] The percentages represent the combined effects of breakthrough/degradation/exceedance of linear uptake phase at 0 and 25°C and 50 ppb ozone. Causes of the losses corresponding to each compound are indicated by the numbers following the percentage. (1) indicates breakthrough, (2) is degradation, (3) is exceedance of the linear uptake phase, (4) is a high particle fraction. Full details on the estimates are given in the SI. References: PCB, HCH, and PAH physical-chemical properties and temperature adjustments are from Beyer et al.[131] and Paasivirta et al.;[132] PBDE physical-chemical properties and temperature adjustments are from Harner and Shoeib,[133] Tittlemeier et al.,[134] and Wang et al.;[135] PFOS, chlorpyrifos and endosulfan physical-chemical properties and temperature adjustments are from Odabasi and Cetin[136] and EpiSuite.[137] Particle-phase fractions were calculated using the methods of Harner and Bidleman[126] using the temperature-adjusted K_{OA} values. Breakthrough was calculated using the equations of Pankow.[67] Length of the linear uptake phase was calculated using the equations of Shoeib and Harner.[95] Degradation was estimated based on field study data from Armstrong et al.,[138] Schauer et al.,[85] Tsapakis and Stephanou,[84] Peters et al.,[139] Menichini,[82] and Goriaux et al.[140] Full details on all assumptions/estimates are provided in the SI.

For example, an intermittent AAS can easily be biased for compounds with short-term temporal concentration variability (e.g., CUPs, combustion-related SVOCs)[31] and a long-term time weighted average concentration from a PAS cannot provide information on episodic events. If intermittent sample timing results in sampling on the same weekday or period of a month, this can be biased by daily, weekly or monthly activity patterns (e.g., weekday traffic). This can lead to an over- or underestimation of concentrations if the sample either coincides with or completely misses a high concentration event. Furthermore, as we have moved to measurement of emerging compounds and lower concentration ranges (pg/m^3 and fg/m^3 levels), sample collection volumes have been pushed higher in order to achieve detection, particularly in remote environments, without thoroughly assessing the associated breakthrough potentials.

As described earlier, there is much uncertainty in how particles behave in PUF−PAS. Comparability issues are introduced because users of PUF−PAS make different assumptions about the fraction of the particle phase sampled by the PUF. The assumptions range from treating PUF−PAS as gas-phase only to assuming 1−10% of particles are sampled,[93,105] to assuming all particles are sampled.[96] If PUF−PAS are treated as bulk samplers when they are only sampling a fraction of particles, measured values may be biased low, particularly for higher molecular weight SVOCs.

The systematic variations in PAS sampling rates by physical-chemical properties identified in some studies,[32,33,93,100] reflect situations where the conventional PAS framework does not apply, such as when volatile compounds enter the curvilinear uptake phase, or when only a fraction of atmospheric particles are sampled. Thus, the use of a general sampling rate may introduce bias in either the low molecular weight or high molecular weight compounds and different compound/congener distributions.[33] It is also not well quantified how much sampling rates differ between different climate zones/seasons, thus considering the global range of average temperatures (SI Figure S4), large biases may be introduced when comparing PAS measurements from very different regions or seasons.

Spatial distributions of sampling networks further bias our global knowledge of air concentrations.[1,2] While coverage is somewhat improved when one also considers case studies, there continue to be areas of the world with limited measurement coverage (Figure 1). For example, if one considers available PBDE data for the period 2003−2013, only 2% of the global land mass is within 100 km of a reported PBDE air concentration, and 0.1% is within 20 km. By population, only 30% of people live within 100 km of a reported PBDE measurement, and only 6% within 20 km. Southeast Asia has some coverage for PBDEs measured by case study, but these largely consist of one-time measurements, thereby limiting temporal information. Additionally, land-based southern hemisphere sampling is largely PAS (SI Figure S1), furthering the spatial bias.

Examples of Bias. To demonstrate the potential combined effects of the aforementioned sampling artifacts and biases, we estimated the potential errors in measured concentrations for three typical sampler configurations: (1) a high-volume active air sampler with GFF or QFF and two PUF plugs, collecting ~600 m^3 of air over 24 h, (2) a PUF−PAS, deployed outdoors for 90 days, and (3) an XAD-PAS, deployed outdoors for one year, and two temperature scenarios: 0 and 25 °C. Estimates are summarized in Table 2 and details are given in SI Table S4. These estimates have significant uncertainty, due to the very different data sources, but they emphasize the large possibility for sampling artifacts to influence measured concentrations.

The estimates suggest combined effects of breakthrough and/or exceedance of the PAS linear uptake phase lead to large underestimates in the reported values of the volatile SVOCs, notably PCB-28, α-HCH, fluorene, PFOS, and by extension, other SVOCs with similar physical-chemical properties. This effect also influences XAD-PAS for more volatile SVOCs (e.g., α-HCH, fluorene, PFOS) at higher temperatures.[59,106] Within-sampler degradation, although not well-characterized, is also estimated to contribute to large losses, particularly for fluorene, benzo[a]pyrene, and chlorpyrifos. Similar degradation is expected for gas-phase compounds with short reactive atmospheric half-lives, which includes all PAHs, many CUPs and some NFRs (SI Table S2). The particle-associated SVOCs are under-sampled by PUF−PAS and not sampled by XAD-PAS. Estimates in Table 2 assume PUF−PAS collect 10% of the total particle fraction, but this may range from 1 to 100%. Thus, reported concentrations may vary between 15 and 100% of the actual concentration, depending on the fraction of particles that is truly collected by the PUF−PAS.

Attention is frequently given to issues of analytical comparability, but sampling comparability affects many compounds which we assume have minimal analytical uncertainties. For example, BDE-209, which has frequently been identified to have difficulties in laboratory analysis,[122] is “correctly” sampled, for example, 100% of the air concentration should be captured by a typical high volume air sampler, whereas PCB-28, a compound that has been consistently included in global monitoring networks and is one of the seven typical indicator PCBs, may experience losses of up to 15% based on breakthrough alone, and thus reported values may be biased low in many situations.

Problematic Chemicals. Many “emerging” SVOCs have different physical-chemical properties compared to the legacy SVOCs and this may introduce challenges in sampling. Current sampling networks and techniques were developed for legacy SVOCs and may not be appropriate for providing representative data on these newer SVOCs. For example, the use of PAS in the future may be more challenging because target compounds are either too volatile (volatile methylsiloxanes, phthalates) or too involatile (CUPs, NFRs) and thus have high uncertainty in conventional PAS. As a consequence of regulatory actions, newer compounds are often designed to be less persistent in the environment, but as a result are also less persistent in samplers (leading to more within-sampler degradation) and thus more difficult to correctly quantify. Degradation is also of concern when measuring nitro- and other substituted PAHs,[123] and existing data on these compounds is likely an underestimate. The lack of data on degradation and the potential importance for many emerging compounds highlights the need for more studies.

Furthermore, we must be aware that chemicals currently measured by the SVOC community are a small fraction of the chemicals in use that may be of environmental and/or human health concern.[124] Adding more chemicals to the lists of analytes without adapting current sampler configurations will introduce more uncertainty in measured concentrations.

RECOMMENDATIONS AND FUTURE DEVELOPMENTS

Recommendations to Enhance Comparability. Given the wide range of potential biases and uncertainties in air sampling identified above, it is clear that careful consideration must be given to sampling approaches. This consideration should first consider the study goals and outcomes. For example, what sampling designs are best suited to address global distributions vs data for human exposure assessments vs source identification, long-term vs short-term trends, and the target SVOCs? The sampler type, frequency, and sampling materials should correspond to the specific question and compounds to be addressed. But ultimately, as the GMP and other initiatives aggregate international data, study design should also aim for comparability with global data sets.

Overall, PAS are best for broad spatial coverage and long-term temporal trends, while AAS are best for short-term temporal trends and coverage of a broader set of compounds. When the intended result of a study is long-term integrated average concentrations (e.g., monthly to yearly scale), and particularly for largely gas-phase compounds, PAS have a clear advantage. The simplicity of PAS (no electricity, cheaper) make them best suited to resolve concentration gradients where levels vary by orders of magnitude. AAS are recommended when absolute concentrations are needed, smaller concentration variations are expected or high time resolution is needed. AAS are the only choice when gas-particle partitioning is important or particle-phase concentrations are expected to be dominant and when short-term trends (hourly to weekly) are of interest. Because congener/compound ratios can be biased in PAS, AAS are best for source identification/fingerprinting techniques. Regulations for human exposure assessment are often based on particle-associated SVOCs and the best sampler choice is a low volume AAS providing continuous gas- and particle-phase concentrations, while PAS are not as well suited. However, current research continues to demonstrate the broad applicability of PAS,[96,97] and, although they have higher uncertainty, particularly for particle-phase compounds, the uncertainty in a temporally integrated PAS could be lower than uncertainty in intermittent AAS for compounds with daily concentration variations.

Sorbent choice should be considered with respect to the length of sampling period and target compounds. In PAS, PUF is recommended for capturing seasonal trends for stable, legacy compounds, while XAD-PAS are recommended for longer term (e.g., yearly, decadal trends)[31] and also to cover emerging gas-phase SVOCs (more polar compounds). In AAS, for best applicability to a broad range of SVOCs and conditions, we recommend a combination of PUF and XAD, as this reduces breakthrough concerns and allows quantification of both polar and nonpolar SVOCs. While PUF and XAD are the current focus of the majority of the SVOC community, new sorbents should be investigated for their applicability to a wider range of compounds.

Although there is a need for standardization in SVOC air measurements, with large regional differences and equipment constraints, implementation of standardization can be challenging. This could be addressed in the long term through the development of international standards for SVOC sampling (e.g., an ISO standard), but in the short term resetting of all sampling campaigns and networks to the same configurations is not practical, nor necessarily useful. Many networks have long-term established techniques and continuation of the same techniques allows for temporal comparisons. Furthermore, universal standardization of techniques would not be appropriate for all regions. For example, sampling in high temperatures tropical regions may require a different sampling approach to low temperature polar regions. However, when different techniques are used it is important for the SVOC community to understand the biases from techniques/configurations/approaches. Instead of implementing identical sampling techniques, data from different sampling configurations may be adjusted using standardization factors. Standardization factors can account for effects of breakthrough, temperature differences, wind, etc., and can be tailored to specific sites/sampler types/seasons/sampling approaches. For example, if degradation was better characterized, a standardization factor could be used to account for the underestimation caused at sites with high levels of reactive trace gases. For cases where large sample volumes are needed, standardization factors can be used to account for losses of the more volatile SVOCs. In fact, this is already done to a certain extent when compound-specific sampling rates are used for PAS; these rates account for nonlinear uptake of volatile SVOCs and lower sampling of particle-associated SVOCs.

One way standardization factors can be determined, particularly for long-term monitoring networks, is at "supersites" where samplers and sampling approaches from different networks are used simultaneously.[125] Quality assurance techniques such as spiking samplers with labeled compounds should also be used periodically across different networks to further evaluate comparability.

However, standardization factors, once developed, should not be treated as a perfect solution. Better consideration of study design should be given in advance of sampler deployments. There is sufficient theoretical and experimental knowledge to allow researchers to estimate PAS linear uptake phases,[41,95] gas-particle partitioning of target compounds,[126,127] and AAS breakthrough[67] in advance of sampler deployment. A necessary part of sampling campaign should be estimating these effects and structuring the sampling campaign to reduce these artifacts. New monitoring networks and case studies should aim to harmonize, where possible, with existing networks to maximize data comparability.

Recommendations for Future Work. Extensive experimental, modeling and calibration work has and continues to be done on PAS techniques with the goal of improving the accuracy of measured concentrations and better understanding uptake processes. This should be continued. However, limited efforts have been made to improve AAS, despite knowledge of uncertainties from degradation, breakthrough, etc. and this should be an additional focus.

For PAS, there are some obvious improvements needed, particularly in the determination of sampling rates. Currently, sampling rates are determined either with calibration studies or DCs, but sampling rates could be improved by combining these two methods. Sampling rates (determined from calibration studies) could be adjusted according to differential losses of DCs between sampling sites to account for site-specific influences. Further studies should also focus on understanding the performance of PUF−PAS for particle-associated SVOCs under different conditions and for different PUF densities, to better understand why current studies obtain inconsistent results.

For AAS there are also clear improvements needed, particularly in the study of breakthrough and degradation. Current sampling configurations are based on information from breakthrough studies from 1980s and early 1990s[24,67,128] when different SVOCs were the primary concerns and concentrations were often higher. Today, larger volumes are used to detect compounds at the pg/m^3 and fg/m^3 level, particularly in low concentration regions (e.g., polar regions, Figure 2) without a good understanding of the associated impacts (breakthrough, degradation). Since the use of denuders in conjunction with AAS has yielded important information regarding the degradation of PAHs, this avenue should be further investigated to better understand within sampler degradation for PAHs and other reactive SVOCs. As the relationship between ozone levels and degradation appears relatively consistent,[85] it may be possible to estimate SVOC losses based on ozone levels, which could be especially important in locations/times of year when losses are expected to be large. Additionally, we should reconsider the paradigm that AAS provide more accurate results. We have summarized numerous well-known sampling artifacts associated with PAS and AAS, but due to limited experimental evidence, the question of the overall effect of these artifacts on the accuracy of reported concentrations remains a rough estimate (e.g., Table 2). The question of the overall significance of these artifacts on PAS vs AAS could be addressed with a well-designed PAS-AAS comparison.

In addition, as sampling expands to broader global coverage, a more thorough understanding of the effect of environmental conditions/meteorology on both AAS and PAS is needed. Currently, sampler evaluation and testing is largely performed in temperate regions; this may limit applicability in more extreme environments.

There is also room for innovation in sampling techniques. For example, source regions and the influence of specific air masses can be assessed using new directional AAS[129] and PAS.[130]

Finally, the SVOC community must be progressive and adaptable. How can sampling campaigns be designed to produce more comparable data without compromising the data needs of individual networks and case studies? What compounds are most important to continue monitoring, and what compounds should be added to global monitoring? We have established that there are big challenges in comparability between well-known SVOCs such as PAHs, PCBs, and PBDEs. These must be addressed, but the SVOC community should also move forward with air sampling methods that can meet the future challenges of new chemicals. We must also be prepared to address the future global developments, as the key questions in SVOCs shift, e.g., responding to climate change-induced meteorological shifts and new source balances due to melting of glaciers and polar ice caps[1] and the relationship between SVOCs and human health impacts, including consideration of possible synergistic effects.[3]

■ ASSOCIATED CONTENT

Ⓢ Supporting Information

Maps of global distributions of sampling networks and case studies, description of atmospheric degradation processes, atmospheric reactive half-lives for SVOCs, and details of calculations for Table 2. This material is available free of charge via the Internet at http://pubs.acs.org.

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Notes

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■ REFERENCES

(1) Klánová, J.; Harner, T. The challenge of producing reliable results under highly variable conditions and the role of passive air samplers in the Global Monitoring Plan. *Trends Anal. Chem.* **2013**, *46*, 139−149.

(2) Hung, H.; MacLeod, M.; Guardans, R.; Scheringer, M.; Barra, R.; Harner, T.; Zhang, G. Toward the next generation of air quality monitoring: Persistent organic pollutants. *Atmos. Environ.* **2013**, *80*, 591−598.

(3) Klánová, J.; Diamond, M. L.; Jones, K. C.; Lammel, G.; Lohmann, R.; Pirrone, N.; Scheringer, M.; Balducci, C.; Bidleman, T. F.; Bláha, K.; et al. Identifying the research and infrastructure needs for the global assessment of hazardous chemicals ten years after establishing the Stockholm Convention. *Environ. Sci. Technol.* **2011**, *45*, 7617−7619.

(4) Bergman, A.; Heindel, J. J.; Jobling, S.; Kidd, K. A.; Zoeller, R. T. *State of the Science of Endocrine Disrupting Chemicals 2012*; Bergman, A., Heindel, J. J., Jobling, S., Kidd, K. A., Zoeller, R. T., Eds.; World Health Organization: Geneva, 2012.

(5) Harner, T.; Shoeib, M.; Diamond, M. L.; Ikonomou, M. G.; Stern, G. A. Passive sampler derived air concentrations of PBDEs along an urban-rural transect: Spatial and temporal trends. *Chemosphere* **2006**, *64*, 262−267.

(6) Wania, F.; Shen, L.; Lei, Y. D.; Teixiera, C. F.; Muir, D. C. G. Development and calibration of a Resin-Based Passive Sampling System for Monitoring Persistent Organic Pollutants in the Atmosphere. *Environ. Sci. Technol.* **2003**, *37*, 1352−1359.

(7) Zhang, T.; Huang, Y.; Chen, S.-J.; Liu, A.-M.; Xu, P.-J.; Li, N.; Qi, L.; Ren, Y.; Zhou, Z.-G.; Mai, B.-X. PCDD/Fs, PBDD/Fs, and PBDEs in the air of an e-waste recycling area (Taizhou) in China: Current levels, composition profiles, and potential cancer risks. *J. Environ. Monit.* **2012**, *14*, 3156−3163.

(8) Sharma, B. M.; Bharat, G. K.; Tayal, S.; Nizzetto, L.; Cupr, P.; Larssen, T. Environment and human exposure to persistent organic pollutants (POPs) in India: A systematic review of recent and historical data. *Environ. Int.* **2014**, *66*, 48−64.

(9) Leyk, S.; Binder, C. R.; Nuckols, J. R. Spatial modeling of personalized exposure dynamics: The case of pesticide use in small-scale agricultural production landscapes of the developing world. *Int. J. Health Geogr.* **2009**, *8*, 17.

(10) Pozo, K.; Harner, T.; Lee, S. C.; Sinha, R. K.; Sengupta, B.; Loewen, M.; Geethalakshmi, V.; Kannan, K.; Volpi, V. Assessing seasonal and spatial trends of persistent organic pollutants (POPs) in Indian agricultural regions using PUF disk passive air samplers. *Environ. Pollut.* **2011**, *159*, 646−653.

(11) Melymuk, L.; Robson, M.; Helm, P. A.; Diamond, M. L. PCBs, PBDEs, and PAHs in Toronto air: Spatial and seasonal trends and implications for contaminant transport. *Sci. Total Environ.* **2012**, *429*, 272−280.

(12) Hearn, L. K.; Kennedy, K. E.; Hawker, D. W.; Toms, L.-M. L.; Alberts, V.; Müller, J. F. Spatial mapping of city-wide PBDE levels using an exponential decay model. *J. Environ. Monit.* **2012**, *14*, 643−650.

(13) Persoon, C.; Peters, T. M.; Kumar, N.; Hornbuckle, K. C. Spatial distribution of airborne polychlorinated biphenyls in Cleveland, Ohio and Chicago, Illinois. *Environ. Sci. Technol.* **2010**, *44*, 2797−2802.

(14) Simcik, M. F. Air monitoring of persistent organic pollutants in the Great Lakes: IADN vs. AEOLOS. *Environ. Monit. Assess.* **2005**, *100*, 201−216.

(15) Blanchard, P.; Kallweit, D.; Brice, K. A.; Froude, F. A.; Chan, C. H.; Neilson, M.; Holz, J.; Millat, H. A comparison of European and North American atmospheric deposition networks: Polycyclic aromatic hydrocarbons and lindane. *J. Environ. Monit.* **2006**, *8*, 465.

(16) Wu, R.; Backus, S. M.; Basu, I.; Blanchard, P.; Brice, K. A.; Dryfhout-Clark, H.; Fowlie, P.; Hulting, M.; Hites, R. A. Findings from quality assurance activities in the Integrated Atmospheric Deposition Network. *J. Environ. Monit.* **2009**, *11*, 277−296.

(17) Schlabach, M.; Farag-clement, R.; Hung, H.; Kallenborn, R. *AMAP/EMEP/NCP Inter-Laboratory study for POP Analysis*, EMEP/CCC-Report 7/2011; Norwegian Institute for Air Research: Kjeller, Norway, 2010.

(18) Gouin, T.; Harner, T.; Blanchard, P.; Mackay, D. Passive and active air samplers as complementary methods for investigating persistent organic pollutants in the Great Lakes Basin. *Environ. Sci. Technol.* **2005**, *39*, 9115−9122.

(19) Klánová, J.; Kohoutek, J.; Hamplová, L.; Urbanová, P.; Holoubek, I. Passive air sampler as a tool for long-term air pollution monitoring: Part 1. Performance assessment for seasonal and spatial variations. *Environ. Pollut.* **2006**, *144*, 393−405.

(20) Halse, A. K.; Schlabach, M.; Sweetman, A. J.; Jones, K. C.; Breivik, K. Using passive air samplers to assess local sources versus long range atmospheric transport of POPs. *J. Environ. Monit.* **2012**, *14*, 2580−2590.

(21) Dreyer, A.; Shoeib, M.; Fiedler, S.; Barber, J.; Harner, T.; Schramm, K.-W.; Jones, K. C.; Ebinghaus, R. Field intercomparison on the determination of volatile and semivolatile polyfluorinated compounds in air. *Environ. Chem.* **2010**, *7*, 350.

(22) Brandsma, S. H.; de Boer, J.; Leonards, P. E. G.; Cofino, W. P.; Covaci, A. Organophosphorus flame-retardant and plasticizer analysis, including recommendations from the first worldwide interlaboratory study. *TrAC Trends Anal. Chem.* **2013**, *43*, 217−228.

(23) Su, Y.; Hung, H. Inter-laboratory comparison study on measuring semi-volatile organic chemicals in standards and air samples. *Environ. Pollut.* **2010**, *158*, 3365−3371.

(24) Bidleman, T. F.; Simon, C. G.; Burdick, N. F.; You, F. Theoretical plate measurements and collection efficiencies for high-volume air samplers using polyurethane foam. *J. Chromatogr.* **1984**, *301*, 448−453.

(25) Bidleman, T. F.; Olney, C. E. High-volume collection of atmospheric polychlorinated biphenyls. *Bull. Environ. Contam. Toxicol.* **1974**, *11*, 442−450.

(26) Orgill, M. M.; Sehmel, G. A.; Petersen, M. R. Some initial measurements of airborne DDT over Pacific Northwest forests. *Atmos. Environ.* **1976**, *10*, 827−834.

(27) Simon, C. G.; Bidleman, T. F. Sampling airborne PCB with polyurethane foam—Chromatographic approach to determination of collection effeciencies. *Abstr. Pap. Am. Chem. Soc.* **1978**, *176*, 187.

(28) Aas, W.; Breivik, K. *Heavy Metals and POP Measurements, 2010*, EMEP Co-operative Programme for Monitoring and Evaluation of the Long-Range Transmission of Air Pollutants in Europe, EMEP/CCC-Report 3/2012; Norwegian Institute for Air Research: Kjeller, Norway, 2011.

(29) Hung, H.; Kallenborn, R.; Breivik, K.; Su, Y.; Brorström-Lundén, E.; Olafsdottir, K.; Thorlacius, J. M.; Leppänen, S.; Bossi, R.; Skov, H.; et al. Atmospheric monitoring of organic pollutants in the Arctic under the Arctic Monitoring and Assessment Programme (AMAP): 1993−2006. *Sci. Total Environ.* **2010**, *408*, 2854−2873.

(30) Su, Y.; Hung, H.; Sverko, E.; Fellin, P.; Li, H. Multi-year measurements of polybrominated diphenyl ethers (PBDEs) in the Arctic atmosphere. *Atmos. Environ.* **2007**, *41*, 8725−8735.

(31) Hayward, S. J.; Gouin, T.; Wania, F. Comparison of four active and passive sampling techniques for pesticides in air. *Environ. Sci. Technol.* **2010**, *44*, 3410−3416.

(32) Hazrati, S.; Harrad, S. J. Calibration of polyurethane foam (PUF) disk passive air samplers for quantitative measurement of polychlorinated biphenyls (PCBs) and polybrominated diphenyl ethers (PBDEs): Factors influencing sampling rates. *Chemosphere* **2007**, *67*, 448−455.

(33) Melymuk, L.; Robson, M.; Helm, P. A.; Diamond, M. L. Evaluation of passive air sampler calibrations: Selection of sampling rates and implications for the measurement of persistent organic pollutants in air. *Atmos. Environ.* **2011**, *45*, 1867−1875.

(34) Batterman, S. A.; Chen, T.-C.; Chernyak, S. M.; Godwin, C. Design and performance evaluation of a medium flow sampler for airborne brominated flame retardants (BFRs). *J. Environ. Monit.* **2009**, *11*, 858−866.

(35) Aas, W.; Breivik, K. *Heavy metals and POP measurements 2011*; EMEP Co-operative Programme for Monitoring and Evaluation of the Long-Range Transmission of Air Pollutants in Europe. Norwegian Institute for Air Research. Kjeller, Norway, 2011; EMEP/CCC-Report 4/2013.

(36) Pozo, K.; Harner, T.; Wania, F.; Muir, D. C. G.; Jones, K. C.; Barrie, L. A. Toward a global network for persistent organic pollutants in air: Results from the GAPS study. *Environ. Sci. Technol.* **2006**, *40*, 4867−4873.

(37) Koblizková, M.; Genualdi, S.; Lee, S. C.; Harner, T. Application of sorbent impregnated polyurethane foam (SIP) disk passive air samplers for investigating organochlorine pesticides and polybrominated diphenyl ethers at the global scale. *Environ. Sci. Technol.* **2012**, *46*, 391−396.

(38) Farrar, N. J.; Prevedouros, K.; Harner, T.; Sweetman, A. J.; Jones, K. C. Continental scale passive air sampling of persistent organic pollutants using rapidly equilibrating thin films (POGs). *Environ. Pollut.* **2006**, *144*, 423−433.

(39) Schuster, J. K.; Gioia, R.; Breivik, K.; Steinnes, E.; Scheringer, M.; Jones, K. C. Trends in European background air reflect reductions in primary emissions of PCBs and PBDEs. *Environ. Sci. Technol.* **2010**, *44*, 6760−6766.

(40) Mayer, P.; Wania, F.; Wong, C. S. Advancing passive sampling of contaminants in environmental science. *Environ. Sci. Process. Impacts* **2014**, *16*, 366−368.

(41) Bartkow, M. E.; Booij, K.; Kennedy, K. E.; Müller, J. F.; Hawker, D. W. Passive air sampling theory for semivolatile organic compounds. *Chemosphere* **2005**, *60*, 170−176.

(42) Tuduri, L.; Millet, M.; Briand, O.; Montury, M. Passive air sampling of semi-volatile organic compounds. *Trends Anal. Chem.* **2012**, *31*, 38−49.

(43) Bohlin, P.; Jones, K. C.; Strandberg, B. Occupational and indoor air exposure to persistent organic pollutants: A review of passive sampling techniques and needs. *J. Environ. Monit.* **2007**, *9*, 501−509.

(44) Esteve-Turrillas, F. A.; Yusà, V.; Pastor, A.; de la Guardia, M. New perspectives in the use of semipermeable membrane devices as passive samplers. *Talanta* **2008**, *74*, 443−457.

(45) Berg, T.; Dye, C.; Hanssen, J. E.; Krognes, T.; Munthe, J.; Reissell, A.; Schaug, J.; Schmidbauer, N.; Semb, A.; Torseth, K.; et al. *EMEP Manual for Sampling and Analysis*; Norwegian Institute for Air Research: Kjeller, Norway, 2001.

(46) Finlayson-Pitts, B. J.; Pitts, J. N. J. Chapter 9: Particles in the troposphere. In *Chemistry of the Upper and Lower Atmosphere*; Elsevier Inc.: San Diego, 1999; pp 349−435.

(47) Englert, N. Fine particles and human health–a review of epidemiological studies. *Toxicol. Lett.* **2004**, *149*, 235−242.

(48) Lammel, G.; Klánová, J.; Ilić, P.; Kohoutek, J.; Gasić, B.; Kovacić, I.; Škrdlíková, L. Polycyclic aromatic hydrocarbons in air on small spatial and temporal scales − II. Mass size distributions and gas-particle partitioning. *Atmos. Environ.* **2010**, *44*, 5022−5027.

(49) Zhang, B.-Z.; Zhang, K.; Li, S.-M.; Wong, C. S.; Zeng, E. Y. Size-dependent dry deposition of airborne polybrominated diphenyl ethers in urban Guangzhou, China. *Environ. Sci. Technol.* **2012**, *46*, 7207−7214.

(50) Chrysikou, L. P.; Samara, C. A. Seasonal variation of the size distribution of urban particulate matter and associated organic pollutants in the ambient air. *Atmos. Environ.* **2009**, *43*, 4557−4569.

(51) Degrendele, C.; Okonski, K.; Melymuk, L.; Landlová, L.; Kukučka, P.; Cupr, P.; Klánová, J. Size specific distribution of the atmospheric particulate PCDD/Fs, dl-PCBs and PAHs on a seasonal scale: Implications for cancer risks from inhalation. *Atmos. Environ.* **2014**, in press.

(52) Okonski, K.; Degrendele, C.; Melymuk, L.; Landlová, L.; Kukučka, P.; Vojta, Š.; Kohoutek, J.; Klánová, J. The value of size-segregated data: Investigation of the size-fraction distribution of flame retardants on atmospheric aerosols. Submitted.

(53) Lee, J. J.; Huang, K.-L.; Yu, Y. Y.; Chen, M. S. Laboratory retention of vapor-phase PAHs using XAD adsorbents. *Atmos. Environ.* **2004**, *38*, 6185−6193.

(54) Barthel, P.; Thuens, S.; Shunthirasingham, C.; Westgate, J. N.; Wania, F.; Radke, M. Application of XAD-resin based passive air samplers to assess local (roadside) and regional patterns of persistent organic pollutants. *Environ. Pollut.* **2012**, *166*, 218−225.

(55) Sun, P.; Basu, I.; Hites, R. A. Temporal trends of polychlorinated biphenyls in precipitation and air at chicago. *Environ. Sci. Technol.* **2006**, *40*, 1178−1183.

(56) Genualdi, S.; Lee, S. C.; Shoeib, M.; Gawor, A.; Ahrens, L.; Harner, T. Global pilot study of legacy and emergin persistent organic pollutants using sorbent-impregnated polyurethane foam disk passive air samplers. *Environ. Sci. Technol.* **2010**, *44*, 5534−5539.

(57) Ahrens, L.; Harner, T.; Shoeib, M.; Lane, D. A.; Murphy, J. G. Improved characterization of gas-particle partitioning for per- and polyfluoroalkyl substances in the atmosphere using annular diffusion denuder samplers. *Environ. Sci. Technol.* **2012**, *46*, 7199−7206.

(58) Kamprad, I.; Goss, K.-U. Systematic investigation of the sorption properties of polyurethane foams for organic vapors. *Anal. Chem.* **2007**, *79*, 4222−4227.

(59) Hayward, S. J.; Lei, Y. D.; Wania, F. Sorption of a diverse set of organic chemical vapors onto XAD-2 resin: Measurement, prediction and implications for air sampling. *Atmos. Environ.* **2011**, *45*, 296−302.

(60) Shoeib, M.; Harner, T.; Lee, S. C.; Lane, D. A.; Zhu, J. Sorbent-impregnated polyurethane foam disk for passive air sampling of volatile fluorinated chemicals. *Anal. Chem.* **2008**, *80*, 675−682.

(61) Clark, C. R.; Truex, T. J.; Lee, F. S. C.; Salmeen, I. T. Influence of sampling filter type on the mutagenicity of diesel exhaust particulate extracts. *Atmos. Environ.* **1981**, *15*, 397−402.

(62) Brown, A. S.; Yardley, R. E.; Quincey, P. G.; Butterfield, D. M. Studies of the effect of humidity and other factors on some different filter materials used for gravimetric measurements of ambient particulate matter. *Atmos. Environ.* **2006**, *40*, 4670−4678.

(63) Lee, D. S.; Nicholson, K. W. The measurement of atmospheric concentrations and deposition of semi-volatile organic compounds. *Environ. Monit. Assess.* **1994**, *32*, 59−91.

(64) Ruuskanen, J.; Tuch, T.; Ten Brink, H.; Peters, A.; Khlystov, A.; Mirme, A.; Kos, G. P. A.; Brunekreef, B.; Wichmann, H. E.; Buzorius, G.; et al. Concentrations of ultrafine, fine and PM2.5 particles in three European cities. *Atmos. Environ.* **2001**, *35*, 3729−3738.

(65) Cabada, J. C.; Rees, S.; Takahama, S.; Khlystov, A.; Pandis, S. N.; Davidson, C. I.; Robinson, A. L. Mass size distributions and size resolved chemical composition of fine particulate matter at the Pittsburgh supersite. *Atmos. Environ.* **2004**, *38*, 3127−3141.

(66) Hites, R. A.; Basu, I.; Venier, M.; Salamova, A.; Nettesheim, T.; Blanchard, P.; Backus, S. M.; Brice, K. A.; Froude, F. A.; Galarneau, E. *Technical Summary of the Progress of the Integrated Atmospheric Deposition Network (IADN)—2002−2008*; Environment Canada, 2008; p 117.

(67) Pankow, J. F. Overview of the gas phase retention volume behavior of organic compounds on polyurethane foam. *Atmos. Environ.* **1989**, *23*, 1107−1111.

(68) You, F.; Bidleman, T. F. Influence of volatility on the collection of polycyclic aromatic hydrocarbon PAH vapors with polyurethane foam. *Environ. Sci. Technol.* **1984**, *18*, 330−333.

(69) Harrad, S. J.; Hazrati, S.; Ibarra, C. Concentrations of polychlorinated biphenyls in indoor air and polybrominated diphenyl ethers in indoor air and dust in Birmingham, United Kingdom: Implications for human exposure. *Environ. Sci. Technol.* **2006**, *40*, 4633−4638.

(70) Ter Schure, A. F. H.; Larsson, P.; Agrell, C.; Boon, J. P. Atmospheric transport of polybrominated diphenyl ethers and polychlorinated biphenyls to the Baltic Sea. *Environ. Sci. Technol.* **2004**, *38*, 1282−1287.

(71) Cotham, W. E.; Bidleman, T. F. Laboratory investigations of the partitioning of organochlorine compounds between the gas phase and atmospheric aerosols on glass fiber filters. *Environ. Sci. Technol.* **1992**, *26*, 469−478.

(72) Pankow, J. F.; Bidleman, T. F. Effects of temperature, TSP and per cent nonexchangeable material in determining the gas particle partitioning of organic compounds. *Atmos. Environ.* **1991**, *25*, 2241−2249.

(73) Bunce, N. J.; Dryfhout, H. G. Diurnal and seasonal modelling of the tropospheric half-lives of polycyclic aromatic hydrocarbons. *Can. J. Chem.* **1992**, *70*, 1966−1970.

(74) Keyte, I. J.; Harrison, R. M.; Lammel, G. Chemical reactivity and long-range transport potential of polycyclic aromatic hydrocarbons—A review. *Chem. Soc. Rev.* **2013**, *42*, 9333−9391.

(75) Mandalakis, M.; Berresheim, H.; Stephanou, E. G. Direct evidence for destruction of polychlorobiphenyls by OH radicals in the subtropical troposphere. *Environ. Sci. Technol.* **2003**, *37*, 542−547.

(76) Li, R.; Jin, J. Model predictions of toxaphene degradation in the atmosphere over North America. *Environ. Toxicol. Chem.* **2013**, *32*, 2663−2671.

(77) Al Rashidi, M.; Chakir, A.; Roth, E. Heterogeneous oxidation of folpet and dimethomorph by OH radicals: A kinetic and mechanistic study. *Atmos. Environ.* **2014**, *82*, 164−171.

(78) Bai, J.; Yang, W.; Zhang, C.; Zhao, Y.; Gong, C.; Sun, X.; Zhang, Q.; Wang, W. Theoretical study on the OH-initiated atmospheric reaction of 1,1,1-trichloro-2,2-bis(4-chlorophenyl) ethane (DDT). *Atmos. Environ.* **2013**, *67*, 177−183.

(79) Schenker, U.; Soltermann, F.; Scheringer, M.; Hungerbühler, K. Modeling the environmental fate of polybrominated diphenyl ethers (PBDEs): The importance of photolysis for the formation of lighter PBDEs. *Environ. Sci. Technol.* **2008**, *42*, 9244−9249.

(80) Stapleton, H. M.; Dodder, N. G. Photodegradation of decabromodiphenyl ether in house dust by natural sunlight. *Environ. Toxicol. Chem.* **2008**, *27*, 306−312.

(81) Wang, J.; Chen, S.-J.; Nie, X.; Tian, M.; Luo, X.; An, T.-C.; Mai, B. Photolytic degradation of decabromodiphenyl ethane (DBDPE). *Chemosphere* **2012**, *89*, 844−849.

(82) Menichini, E. On-filter degradation of particle-bound benzo-[a]pyrene by ozone during air sampling: A review of the experimental evidence of an artefact. *Chemosphere* **2009**, *77*, 1275−1284.

(83) Liu, K.; Duan, F.; He, K.; Ma, Y.; Cheng, Y. Investigation on sampling artifacts of particle associated PAHs using ozone denuder systems. *Front. Environ. Sci. Eng.* **2013**, *8*, 284−292.

(84) Tsapakis, M.; Stephanou, E. G. Collection of gas and particle semi-volatile organic compounds: Use of an oxidant denuder to minimize polycyclic aromatic hydrocarbons degradation during high-volume air sampling. *Atmos. Environ.* **2003**, *37*, 4935−4944.

(85) Schauer, C.; Niessner, R.; Pöschl, U. Polycyclic aromatic hydrocarbons in urban air particulate matter: Decadal and seasonal trends, chemical degradation, and sampling artifacts. *Environ. Sci. Technol.* **2003**, *37*, 2861−2868.

(86) Finlayson-Pitts, B. J.; Pitts, J. N. J. Chapter 10: Airborne polycyclic aromatic hydrocarbons and their derivatives. In *Chemistry of the Upper and Lower Atmosphere*; Elsevier Inc.: San Diego, 1999; pp 436−546.

(87) Fowler, D.; Amann, M.; Anderson, R.; Ashmore, M. R.; Depledge, M.; Derwent, D.; Grennfelt, P.; Hewitt, C. N.; Hov, O.; Jenkin, M.; et al. Ground-level ozone in the 21st century: Future trends, impacts and policy implications Science Policy. *Sci. Policy Rep. - R. Soc.* **2008**, *15/08*.

(88) Bahm, K.; Khalil, M. A. K. A new model of tropospheric hydroxyl radical concentrations. *Chemosphere* **2004**, *54*, 143−166.

(89) Finlayson-Pitts, B. J.; Pitts, J. N. J. *Chemistry of the Upper and Lower Atmosphere*; Elsevier Inc.: San Diego, 1999.

(90) Bloss, W. J.; Evans, M. J.; Lee, J. D.; Sommariva, R.; Heard, D. E.; Pilling, M. J. The oxidative capacity of the troposphere: Coupling of field measurements of OH and a global chemistry transport model. *Faraday Discuss.* **2005**, *130*, 425.

(91) Perraudin, E.; Budzinski, H.; Villenave, E. Kinetic study of the reactions of ozone with polycyclic aromatic hydrocarbons adsorbed on atmospheric model particles. *J. Atmos. Chem.* **2006**, *56*, 57−82.

(92) Atkinson, R. Atmospheric lifetimes of dibenzo-para-dioxins and dibenzofurans. *Sci. Total Environ.* **1991**, *104*, 17−33.

(93) Bohlin, P.; Audy, O.; Škrdlíková, L.; Kukučka, P.; Vojta, Š.; Přibylová, P.; Prokeš, R.; Čupr, P.; Klánová, J. Evaluation and guidelines for using polyurethane foam (PUF) passive air samplers in double-dome chambers to assess semi volatile organic compounds in non-industrial indoor environments. *Environ. Sci. Process. Impacts* **2014**, *Accepted*, DOI: 10.1039/C4EM00305E.

(94) Bohlin, P.; Audy, O.; Škrdlíková, L.; Kukučka, P.; Přibylová, P.; Prokeš, R.; Vojta, Š.; Klánová, J. Outdoor passive air monitoring of semi volatile organic compounds (SVOCs): A critical evaluation of performance and limitations of polyurethane foam (PUF) disks. *Environ. Sci. Process. Impacts* **2014**, *16*, 433−444.

(95) Shoeib, M.; Harner, T. Characterization and comparison of three passive air samplers for persistent organic pollutants. *Environ. Sci. Technol.* **2002**, *36*, 4142−4151.

(96) Harner, T.; Su, K.; Genualdi, S.; Karpowicz, J.; Ahrens, L.; Mihele, C.; Schuster, J. K.; Charland, J.-P.; Narayan, J. Calibration and application of PUF disk passive air samplers for tracking polycyclic aromatic compounds (PACs). *Atmos. Environ.* **2013**, *75*, 123−128.

(97) Heo, J.; Lee, G. Field-measured uptake rates of PCDDs/Fs and dl-PCBs using PUF-disk passive air samplers in Gyeonggi-do, South Korea. *Sci. Total Environ.* **2014**, DOI: 10.1016/j.scitotenv.2014.03.073.

(98) Mari, M.; Schuhmacher, M.; Feliubadaló, J.; Domingo, J. L. Air concentrations of PCDD/Fs, PCBs and PCNs using active and passive air samplers. *Chemosphere* **2008**, *70*, 1637−1643.

(99) Pozo, K.; Harner, T.; Shoeib, M.; Urrutia, R.; Barra, R.; Parra, O.; Focardi, S. Passive-sampler derived air concentrations of persistent organic pollutants on a north-south transect in Chile. *Environ. Sci. Technol.* **2004**, *38*, 6529−6537.

(100) Zhang, X.; Tsurukawa, M.; Nakano, T.; Lei, Y. D.; Wania, F. Sampling medium side resistance to uptake of semivolatile organic compounds in passive air samplers. *Environ. Sci. Technol.* **2011**, *45*, 10509−10515.

(101) Pozo, K.; Harner, T.; Lee, S. C.; Wania, F.; Muir, D. C. G.; Jones, K. C. Seasonally resolved concentrations of persistent organic pollutants in the global atmosphere from the first year of the GAPS study. *Environ. Sci. Technol.* **2009**, *43*, 796−803.

(102) Persoon, C.; Hornbuckle, K. C. Calculation of passive sampling rates from both native PCBs and depuration compounds in indoor and outdoor environments. *Chemosphere* **2009**, *74*, 917−923.

(103) Santiago, E. C.; Cayetano, M. G. Polycyclic aromatic hydrocarbons in ambient air in the Philippines derived from passive sampler with polyurethane foam disk. *Atmos. Environ.* **2007**, *41*, 4138−4147.

(104) Wilford, B. H.; Harner, T.; Zhu, J.; Shoeib, M.; Jones, K. C. Passive sampling survey of polybrominated diphenyl ether flame retardants in indoor and outdoor air in Ottawa, Canada: Implications for sources and exposure. *Environ. Sci. Technol.* **2004**, *38*, 5312−5318.

(105) Klánová, J.; Cupr, P.; Kohoutek, J.; Harner, T. Assessing the influence of meteorological parameters on the performance of polyurethane foam-based passive air samplers. *Environ. Sci. Technol.* **2008**, *42*, 550−555.

(106) Armitage, J. M.; Hayward, S. J.; Wania, F. Modeling the uptake of neutral organic chemicals on XAD passive air samplers under variable temperatures, external wind speeds and ambient air concentrations (PAS-SIM). *Environ. Sci. Technol.* **2013**, *47*, 13546−13554.

(107) Petrich, N. T.; Spak, S. N.; Carmichael, G. R.; Hu, D.; Martinez, A.; Hornbuckle, K. C. Simulating and explaining passive air sampling rates for semivolatile compounds on polyurethane foam passive samplers. *Environ. Sci. Technol.* **2013**, *47*, 8591−8598.

(108) Zhang, X.; Wong, C.; Lei, Y. D.; Wania, F. Influence of sampler configuration on the uptake kinetics of a passive air sampler. *Environ. Sci. Technol.* **2012**, *46*, 397−403.

(109) Dušek, L.; Borůvková, J.; Hůlek, R.; Jarkovský, J.; Kubásek, M.; Gregor, J.; Šebková, K.; Hřebíček, J.; Holoubek, I. GENASIS—Global environmental assessment and information system. In *Environmental Software Systems*; Springer, 2011; online.

(110) Ahrens, L.; Shoeib, M.; Harner, T.; Koblizková, M.; Reiner, E. J. Characterization of two passive air samplers for per- and polyfluoroalkyl substances. *Organohalogen Compd.* **2012**, *74*, 339−342.

(111) Genualdi, S.; Harner, T.; Cheng, Y.; Macleod, M.; Hansen, K. M.; van Egmond, R.; Shoeib, M.; Lee, S. C. Global distribution of linear and cyclic volatile methyl siloxanes in air. *Environ. Sci. Technol.* **2011**, *45*, 3349−3354.

(112) Saini, A.; Chaudhuri, S. R.; Goosey, E.; Diamond, M. L.; Shoeib, M. Calibration of passive air sampler for monitoring phthalates in indoor air; *Society for Environmental Toxicology and Chemistry European Annual Meeting*; SETAC: Glasgow, 2013.

(113) Robson, M. *Personal communication*; Toronto, Ontario, Canada, 2014; [email communication with L. Melymuk, 21−4−2014].

(114) Chaemfa, C.; Barber, J. L.; Gocht, T.; Harner, T.; Holoubek, I.; Klánová, J.; Jones, K. C. Field calibration of polyurethane foam (PUF) disk passive air samplers for PCBs and OC pesticides. *Environ. Pollut.* **2008**, *156*, 1290−1297.

(115) Tuduri, L.; Harner, T.; Hung, H. Polyurethane foam (PUF) disks passive air samplers: Wind effect on sampling rates. *Environ. Pollut.* **2006**, *144*, 377−383.

(116) Tao, S.; Liu, Y.; Xu, W.; Lang, C.; Liu, S.; Dou, H.; Liu, W. Calibration of a passive sampler for both gaseous and particulate phase polycyclic aromatic hydrocarbons. *Environ. Sci. Technol.* **2007**, *41*, 568−573.

(117) Abdallah, M. A.-E.; Harrad, S. J. Modification and calibration of a passive air sampler for monitoring vapor and particulate phase brominated flame retardants in indoor air: Application to car interiors. *Environ. Sci. Technol.* **2010**, *44*, 3059−3065.

(118) May, A. A.; Ashman, P.; Huang, J.; Dhaniyala, S.; Holsen, T. M. Evaluation of the polyurethane foam (PUF) disk passive air sampler: Computational modeling and experimental measurements. *Atmos. Environ.* **2011**, *45*, 4354−4359.

(119) Kennedy, K. E.; Hawker, D. W.; Bartkow, M. E.; Carter, S.; Ishikawa, Y.; Müller, J. F. The potential effect of differential ambient and deployment chamber temperatures on PRC derived sampling rates with polyurethane foam (PUF) passive air samplers. *Environ. Pollut.* **2010**, *158*, 142−147.

(120) Zhang, X.; Brown, T. N.; Ansari, A.; Yeun, B.; Kitaoka, K.; Kondo, A.; Lei, Y. D.; Wania, F. Effect of wind on the chemical uptake kinetics of a passive air sampler. *Environ. Sci. Technol.* **2013**, *47*, 7868−7875.

(121) Bohlin, P.; Jones, K. C.; Levin, J.-O.; Lindahl, R.; Strandberg, B. Field evaluation of a passive personal air sampler for screening of PAH exposure in workplaces. *J. Environ. Monit.* **2010**, *12*, 1437−1444.

(122) Covaci, A.; Voorspoels, S.; de Boer, J. Determination of brominated flame retardants, with emphasis on polybrominated diphenyl ethers (PBDEs) in environmental and human samples—A review. *Environ. Int.* **2003**, *29*, 735−756.

(123) Albinet, A.; Leoz-Garziandia, E.; Budzinski, H.; ViIlenave, E. Sampling precautions for the measurement of nitrated polycyclic aromatic hydrocarbons in ambient air. *Atmos. Environ.* **2007**, *41*, 4988−4994.

(124) Muir, D. C. G.; Howard, P. H. Are there other persistent organic pollutants? A challenge for environmental chemists. *Environ. Sci. Technol.* **2006**, *40*, 7157−7166.

(125) Cape, J. N.; Bealey, W.; Coyle, M.; DiMarco, C.; Kentisbeer, J.; Leaver, D.; Malcolm, H.; Nemitz, E.; Simmons, I.; Twigg, M.; et al. *Operation of EMEP "Supersites" in the United Kingdom—Annual Report for 2008*; Centre for Ecology and Hydrology: Edinburgh, 2010; 112 p.

(126) Harner, T.; Bidleman, T. F. Octanol-air partition coefficient for describing particle/gas partitioning of aromatic compounds in urban air. *Environ. Sci. Technol.* **1998**, *32*, 1494−1502.

(127) Pankow, J. An absorption model of gas/particle partitioning of organic compounds in the atmosphere. *Atmos. Environ.* **1994**, *28*, 185−188.

(128) Hart, K. M.; Isabelle, L. M.; Pankow, J. F. High-volume air sampler for particle and gas sampling. 1. Design and gas sampling performance. *Environ. Sci. Technol.* **1992**, *26*, 1048−1052.

(129) Research Centre for Toxic Compounds in the Environment. MonAirNet Project webpage. http://www.monairnet.eu/index-en.php.

(130) Ferranti, E. J. S.; Fryer, M.; Sweetman, A. J.; Solera Garcia, M. A.; Timmis, R. J. Field-testing a new directional passive air sampler for fugitive dust in a complex industrial source environment. *Environ. Sci. Process. Impacts* **2014**, *16*, 159−168.

(131) Beyer, A.; Wania, F.; Gouin, T.; Mackay, D.; Matthies, M. Selecting internally consistent physicochemical properties of organic compounds. *Environ. Toxicol. Chem.* **2002**, *21*, 941−953.

(132) Paasivirta, J.; Sinkkonen, S. I.; Mikkelson, P.; Rantio, T.; Wania, F. Estimation of vapor pressures, solubilities and Henry's law constants of selected persistent organic pollutants as functions of temperature. *Chemosphere* **1999**, *39*, 811−832.

(133) Harner, T.; Shoeib, M. Measurements of octanol-air partition coefficients (KOA) for polybrominated diphenyl ethers (PBDEs): Predicting partitioning in the environment. *J. Chem. Eng. Data* **2002**, *47*, 228−232.

(134) Tittlemier, S. A.; Halldorson, T.; Stern, G. A.; Tomy, G. T. Vapor pressures, aqueous solubilities, and Henry's law constants of some brominated flame retardants. *Environ. Toxicol. Chem.* **2002**, *21*, 1804−1810.

(135) Wang, Z.-Y.; Zeng, X.-L.; Zhai, Z.-C. Prediction of supercooled liquid vapor pressures and n-octanol/air partition coefficients for polybrominated diphenyl ethers by means of molecular descriptors from DFT method. *Sci. Total Environ.* **2008**, *389*, 296−305.

(136) Odabasi, M.; Cetin, B. Determination of octanol-air partition coefficients of organochlorine pesticides (OCPs) as a function of temperature: Application to air-soil exchange. *J. Environ. Manage.* **2012**, *113*, 432−439.

(137) U.S. Environmental Protection Agency. *Estimation Programs Interface Suite (EPI Suite) for Microsoft Windows, v 4.11*, 2012.

(138) Armstrong, J. L.; Fenske, R. A.; Yost, M. G.; Tchong-French, M.; Yu, J. Comparison of polyurethane foam and XAD-2 sampling matrices to measure airborne organophosphorus pesticides and their oxygen analogs in an agricultural community. *Chemosphere* **2013**, *92*, 451−457.

(139) Peters, A. J.; Lane, D. A.; Gundel, L. A.; Northcott, G. L.; Jones, K. C. A comparison of high volume and diffusion denuder samplers for measuring semivolatile organic compounds in the atmosphere. *Environ. Sci. Technol.* **2000**, *34*, 5001−5006.

(140) Goriaux, M.; Jourdain, B.; Temime, B.; Besombes, J.-L.; Marchand, N.; Albinet, A.; Leoz-Garziandia, E.; Wortham, H. Field comparison of particulate PAH measurements using a low-flow denuder device and conventional sampling systems. *Environ. Sci. Technol.* **2006**, *40*, 6398−6404.

APPENDIX 17

Karásková, Pavlína, Garry P. Codling, Lisa Melymuk, and Jana Klánová. 2018. "A Critical Assessment of Passive Air Samplers for Per- and Polyfluoroalkyl Substances." *Atmospheric Environment* 185: 186-95. https://doi.org/10.1016/j.atmosenv.2018.05.030

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A critical assessment of passive air samplers for per- and polyfluoroalkyl substances

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ABSTRACT

Since their inclusion in the Stockholm Convention, there has been a need for global monitoring of perfluorooctane sulfonate (PFOS), its salts and perfluorooctanesulfonyl fluoride (PFOSF), along with other non-listed highly fluorinated compounds. Passive air samplers (PAS) are ideal for geographic coverage of atmospheric monitoring. The most common type of PAS, using polyurethane foam (PUF) as a sorbent, was primarily developed for non-polar semivolatile organic compounds (SVOCs) and are not well-validated for polar substances such as the per- and polyfluoroalkyl substances (PFASs), however, they have been used for some PFASs, particularly PFOS. To evaluate their applicability, PAS were deployed for measurement of PFASs in outdoor and indoor air. Outdoors, two types of PAS, one consisting of PUF and one of XAD-2 resin, were deployed in an 18-week calibration study in parallel with a low-volume active air sampler (LV-AAS) in a suburban area. Indoors, PUF-PAS were similarly deployed over 12 weeks to evaluate their applicability for indoor monitoring. Samples were analysed for perfluoroalkyl carboxylic acids (PFCAs), perfluoroalkyl sulfonates (PFSAs), perfluorooctane sulfonamides (FOSAs), and perfluorooctane sulfonamidoethanols (FOSEs). In outdoor air, 17 out of the 21 PFAS were detected in more than 50% of samples, with a median Σ_{17}PFASs of 18.0 pg m^{-3} while 20 compounds were detected in indoor air with a median concentration Σ_{20}PFASs of 76.6 pg m^{-3} using AAS samplers. PFOS was the most common PFAS in the outdoor air while PFBA was most common indoors. Variability between PAS and AAS was observed and comparing gas phase and particle phase separately or in combination did not account for the variation observed. PUF-PAS may still have a valuable use in PFAS monitoring but more work is needed to identify the applicability of passive samplers for ionic PFAS.

1. Introduction

Per- and polyfluoroalkyl substances (PFASs) are a diverse group of industrial chemicals including perfluoroalkyl carboxylic acids (PFCAs), perfluoroalkyl sulfonates (PFSAs), perfluoorooctane sulfonamides (FOSAs) and perflurooctane sulfonamidoethanols (FOSEs). They have been widely used for over 60 years as surfactants, lubricants, paper and textile coatings, polishes, food packaging, and fire-fighting foams (Prevedouros et al., 2006).

PFASs are known for unique physical and chemical properties, their global distribution in the ocean and the atmosphere, their persistence, bioaccumulation and potential toxicity (Giesy and Kannan, 2002). PFASs have been detected globally in a variety of environmental media (Langer et al., 2010; Shoeib et al., 2010; Gomez et al., 2011; Sun et al., 2011; Wang et al., 2011; Yang et al., 2011; Codling et al., 2014; Hung et al., 2016), biota (Giesy and Kannan, 2002), humans (Wu et al., 2017; Poothong et al., 2017), and have been detected in remote regions due to long-range atmospheric transport of the PFAS and their volatile precursors, (Ahrens et al., 2011a,b), or transport via the aquatic system and marine aerosols (Benskin et al., 2012).

A major concern in human exposure to PFASs is the effect of indoor exposure and its potential to affect vulnerable groups such as pregnant mothers and young children (Shoeib et al., 2005). Moreover, since the listing of PFOS, its salts and their precursors under the Stockholm Convention on Persistent Organic Pollutants (POPs) in 2009, the need for consistent and comparable global monitoring is even greater, as measurement of these compounds is a requirement under the Global Monitoring Plan (GMP) (Klánová and Harner, 2013).

Previous atmospheric measurements of PFASs were often based on the use of high-volume (HV) or low-volume (LV) active air samplers (AAS), usually with polyurethane foam (PUF) and/or polystyrene-divinyl benzene copolymeric resin (XAD) as the gas-phase sorbent (Ahrens et al., 2011a,b). While AAS can provide information about gas-particle partitioning and temporal resolution, AAS has limited ability to

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provide the spatial coverage needed to understand global distributions of PFAS (Koblizkova et al., 2012; Shoeib et al., 2011), particularly in remote areas, and are often not ideal for indoor locations due to the disruptive nature of AAS. In human exposure assessment, personal AAS systems highlight that single location assessment does not reflect actual human exposure patterns (Padilla-Sánchez et al., 2017). For these purposes, passive air samplers (PAS) are an alternative. They are silent, normally far cheaper and require no power supply, so more PAS may be employed at less cost than one AAS.

Various PAS have been developed thus far, primarily for monitoring of non-polar semi-volatile organic compounds (SVOCs) in air (Shoeib and Harner, 2002; Wania et al., 2003), and some PAS have also been used to quantify more polar compounds such as the PFASs. Recently, several studies have reported PFAS levels using conventional PUF-PAS (Chaemfa et al., 2010; Liu et al., 2015), or PUF impregnated with XAD-2 resin (a sorbent impregnated PUF disk, or SIP) (Schuster et al., 2012; Ahrens et al., 2013) or with XAD-2 resin alone (Koblizkova et al., 2012). Reported PFAS concentrations vary between these different PAS methods (Ahrens et al., 2013), and the comparison of reported concentrations could be questionable due to different sampling techniques, sampling media, in addition to differences in the accuracy and precision of analytical methods.

This study evaluates the performance of PAS for four PFAS classes (PFCAs, PFSAs, FOSAs, FOSEs) in outdoor and indoor ambient air by comparison with concentrations determined using AAS. A sampling campaign conducted in 2012 and 2013 used LV-AAS and two types of PAS outdoor (PUF disks and XAD cartridges) and PUF-PAS indoor. The purpose of this survey was to assess PFAS concentrations and distributions in air and evaluate PAS performance through determination of sampler uptake over time. This provides information on how studies using PUF to quantify PFASs can be interpreted, and whether existing sampling infrastructure (e.g., global PAS networks using PUF-PAS) can be easily adapted to include PFAS, given the importance of providing global monitoring data under the GMP. AAS gas- and particle-phase fractions were analysed separately, enabling determination of gas-particle partitioning to complement the interpretation of the PAS performance. We determined sampling rates of individual target PFASs for PUF and XAD-PAS, and make recommendations for monitoring of PFASs in air via PAS.

2. Materials and methods

2.1. Materials

Twenty-one analytical standards (11 PFCAs, 5 PFASs, FOSA, N-MeFOSA, N-EtFOSA, N-MeFOSE and N-EtFOSE) and 13 mass-labelled standards (MPFBA, MPFHxA, MPFOA, M_8PFOA, MPFNA, MPFDA, MPFUnDA, MPFDoDA, MPFHxS, MPFOS, M_8PFOS, dMeFOSA, dMeFOSE) were purchased from Wellington Laboratories Inc. (Guelph, Ontario, Canada). For details, including full IUPAC names see Table S1 in the Supplementary Information. LC/MS-grade methanol was obtained from Biosolve b.v. (Valkenswaard, The Netherlands). HPLC-grade water was from Fisher Scientific (Loughborough, UK). Ammonium acetate, used as an addition to the extraction solvent and mobile phase, was puriss p.a. grade ($\geq 98.0\%$), and was obtained from Fluka (Fluka Chemie GmbH, Buchs, Germany). All chemicals and solvents were used without further purification.

2.2. Air sampling

LV-AAS and PUF-PAS were deployed simultaneously outdoors at a suburban background site on the roof of the Research Centre for Toxic Compounds in the Environment (RECETOX) at Masaryk University in Brno, Czech Republic (49.1782 N, 16.5711 E, Figure S1) between late April and early September 2013 (18 weeks – see Table S2). Thirty polyurethane foam passive samplers (OUT-PUF-PAS) were deployed at the start of the study, consisting of two stainless steel bowls (24 diameter lower bowl and 30 cm diameter upper bowl) surrounding a PUF disk (15 cm in diameter x 1.5 cm thick, with a density of 0.030 g cm^{-3}, type T-3037, Molitan a.s., Czech Republic).

A set of triplicate OUT-PUF-PAS was collected at either 1- or 2-week intervals, generating a total of 10 sets of triplicate OUT-PUF-PAS, each corresponding to a specific exposure time. An LV-AAS (OUT-LV-AAS) (Leckel MVS6, Sven Leckel, Ingenieurbüro GmbH, Germany) was run continuously as a reference sampler to provide weekly time-integrated concentrations of the targeted PFASs. Sampler flow rates were 2.3 $\text{m}^3\,\text{h}^{-1}$ and each sample duration was one week, resulting in an average 373.5 m^3 (Table S2) per sample and generating 18 sets of reference samples. The sampling train consisted of a Whatman® 47 mm ø quartz fiber filter (QFF, GE healthcare, Chicago, USA) to collect the particle phase and PUF/XAD/PUF sandwiches with 15 g of XAD-2 resin as sorbent for gas-phase.

To complement the outdoor PUF-PAS calibration, two other PAS were evaluated. To provide insight into the influence of sorbent and sampler choice on PFAS uptake, three XAD-PAS were also deployed in outdoor air in conjunction with the PUF-PAS calibration. XAD-PAS have a lower uptake rate and typically require a longer deployment time to collect sufficient target compound (e.g., 1 year; Wania et al., 2003), so XAD-PAS were not collected on a weekly basis, but rather only at 70, 98 and 126 days. The XAD-PAS consisted of steel cartridges filled by 10 g of XAD-2 resin (OUT-XAD-PAS) (Wania et al., 2003). Triplicates were not collected as this was only intended to screen the possibility of using XAD-PAS for shorter durations. Additionally, to compare PUF-PAS performance in indoor vs. outdoor air, a similar calibration study in indoor air was evaluated. Indoor air samples (IN) were collected from a university lecture room between late January and late April 2012 (12 weeks – see Table S3). The indoor samples were from a carpeted room in a new building (completed in 2011). The indoor air calibration consisted of PUF-PAS (IN-PUF-PAS) and a reference LV-AAS (IN-LV-AAS). The PUF-PAS were the same double-bowl PUF-PAS as used in outdoor air, and the LV-AAS used PUF as gas-phase sorbent and QFF to collect the particle phase. Triplicate PUF-PAS were collected every 7 days, and the LV-AAS was run continuously, with sampling media changed every week. All samples were wrapped in aluminium foil and stored at −18 C until extraction.

2.3. Extraction and instrumental analysis

Prior to extraction all samples were spiked with labelled standards (M_8PFOA, M_8PFOS) to determine recoveries. All samples except XAD-PAS were extracted with 5 mM ammonium acetate in methanol using a B-811 automated extraction unit (Büchi, Switzerland). PUF/XAD/PUF sandwiches were extracted as one unit. Extraction consisted of 60 min warm Soxhlet followed by 30 min of solvent rinsing and a concentration step to 1 mL under a stream of nitrogen. XAD-PAS were extracted by sonication (3 times with 30 mL MeOH) and concentrated to 1 mL. The concentrated extracts were filtered (nylon membrane, 13 mm diameter and 0.45 μm pore size) and transferred into polypropylene centrifuge tubes (Alpha Laboratories, UK). The filtrate was concentrated to 0.5 mL under a gentle stream of nitrogen and diluted with 0.5 mL 5 mM ammonium acetate in water. Extracts were centrifuged (1800 G, 10 min) and 100 μL transferred to LC minivials for analysis (Labicom, Czech Republic).

Separation, identification and quantification of target PFASs was performed using high performance liquid chromatography (UHPLC) with an Agilent 1290 (Agilent Technologies, Palo, Alto, California, USA) connected to a QTRAP 5500 mass spectrometer (AB Sciex, Foster City, California, USA). Chromatographic separation was performed at 20 °C on a SYNERGI 4 μm Fusion RP 80Ä 50 mm × 2 mm column with a corresponding 4 × 2.00 mm precolumn (Phenomenex, USA). The flow rate was 200 μL min^{-1}, and mobile phases consisted of methanol/5 mM ammonium acetate in water at 55/45 (v/v; A), and methanol (B). The gradient elution used for analyte separation has been reported previously (Karásková et al., 2016), with 10 μL injected onto the column with mass-labelled standards added automatically via the autosampler (1 μL of a 100 ng mL^{-1} solution). The mass spectrometer was operated in electrospray negative ionization mode (ESI-) using two MRM transitions for each compound, except PFBA, N-MeFOSE and N-EtFOSE, at 450 °C and ion voltage 4500 V (see Table S1 for ions and conditions for each compound). Analyst 1.6.1 was used for data integration and evaluation.

The LV-AAS (indoor and outdoor) was considered a reference sampler to monitor air concentrations of PFASs, as well as to calculate average concentrations of each target analyte for PAS calibration.

2.4. QA/QC

Three QA/QC procedures were used: (a) spike/recovery experiment; (b) monitoring of individual sample recoveries; (c) laboratory and field blanks.

Prior to deployment, the PUF and XAD-2 resin were pre-extracted by Soxhlet for 8 h in acetone and 8 h in methanol, dried in a fume chamber and stored in two layers of aluminium foil and sealed in zip-lock bags. After exposure, all matrices were wrapped in aluminium foil, labelled, placed into zip-lock bags and stored at −20 °C until analysis.

The efficiency of the extraction method was tested in a spiking experiment using PUF, QFF and XAD with the addition of native PFASs. Recovery of native analytes measured in spiked matrices varied from 63 to 113% for PUFs, from 61 to 112% for QFFs and from 75 to 108% for XAD-2 resins (detailed in Table S4).

Recovery correction was applied to each sample based on the labelled compounds added prior to extraction. The average percent recovery for those labelled target compounds depended on sampling matrices, ranging from 47 ± 4.6% for M_8PFOA in PUF-PAS to 102 ± 7.3% for M_8PFOS in PUF/XAD/PUF-LV-AAS (detailed in Table S5).

Three PUF-PAS, and 2 QFFs and 2 XAD-2 field blanks were analysed with each set of PUF-PAS and LV-AAS samples. Results from field blanks were used to determine method detection limits (MDLs). MDLs were calculated as the mean blank value plus 3 standard deviations. When compounds were not detected in any blanks the instrumental detection limits (IDLs) were used as MDLs. Samples with masses below MDLs are reported as " < MDL". √2/2*MDL was substituted if appropriate for the statistical analyses being performed (Antweiler, 2015). For data visualization, GraphPad Prism 5 (GraphPad Software, Inc., USA) and Excel 2007 (Microsoft Corporation, USA) were used. The results were exported to the software program SIMCA 14.0 (Umetrics, Sweden) for multivariate data analysis. Principle component analysis (PCA) was used to identify relationships between groups (gas and particle phase, different PAS types, PAS and AAS bulk samples). PCA was performed using 50% variable and observational tolerance with variance scaling.

3. Results and discussion

3.1. Air concentration and composition

All of the 21 target PFASs were detected in at least one OUT-LV-AAS sample. In the sum of gas and particle AAS (OUT_{total}AAS), 17 out of 21 compounds were detected in > 50% of samples, and 13 compounds in > 90% (Table S6). PFUnDA and PFDoDA were detected in < 10% of outdoor samples and are excluded from further interpretation. The composition, concentrations, and temporal trends of the 21 target PFASs in OUT_{total}AAS samples are provided in Fig. 1 and additional data on PFAS concentrations in Table S7.

The range of Σ_{21}PFASs in OUT_{total}AAS samples was 10.4–28.1 pg m^{-3} with a median of 18.0 pg m^{-3}, and dominated by PFOS (28% of total PFASs). Ionic PFASs were dominated by PFOS (median 5.11 pg m^{-3}, range: 1.9–7.68 pg m^{-3}), PFOA (median 2.11 pg m^{-3}, range 1.11–5.47 pg m^{-3}), PFBA (median 1.78 pg m^{-3}, range 0.08–8.95 pg m^{-3}), and PFHpA (median 1.11 pg m^{-3}, range 0.39–1.89 pg m^{-3}). The total outdoor concentrations of ionic PFASs were consistent with previous studies, with PFOS the most abundant compound, followed by PFOA and the shorter chain PFCAs (Barber et al., 2007; Kim and Kannan, 2007; Stock et al., 2007; Dreyer and Ebinghaus, 2009; Weinberg et al., 2011a,b; Liu et al., 2015).

Concentrations of the neutral PFASs, ΣFOSA/Es in OUT_{total}AAS samples, ranged from 1.61 to 11.7 pg m^{-3}, and were dominated by N-MeFOSE (57% of ΣFOSA/Es, 15% of total PFASs) with a median concentration of 1.98 pg m^{-3} and a range of 0.58–7.97 pg m^{-3}. Ahrens et al., 2011a,b has reported a wide range of concentrations of neutral PFASs in the atmosphere: the ΣFOSA/Es near waste treatment sites in Canada ranged from 14.3 to 124 pg m^{-3} while in the Canadian Arctic concentrations of ΣFOSA/E ranged from 0.4 to 21 pg m^{-3}. FOSA/Es concentrations measured in the present study were more consistent with rural and remote locations than dense urban or waste disposal sites (Ahrens et al., 2011a,b; Vierke et al., 2011; Weinberg et al., 2011a,b). Likewise, Barber et al. (2007) reported concentrations of volatile PFASs from Northwest Europe which are one or two orders of magnitude greater than in this study. Comparable concentrations of FOSA/Es were reported by Dreyer and Ebinghaus (2009) during land and sea sampling from Hamburg and into the North Sea. N-MeFOSE is a perfluorooctane sulfonamide that was used primarily in carpets and clothing, however, it should also be noted that the lower concentration observed in this present study may reflect changes in use pattern, as rapid shifts in PFAS use have been observed since some compounds became listed under the Stockholm Convention and others came under greater scrutiny (Butt et al., 2010).

In indoor air all of the 20 PFASs compounds were detected in at least one sample. Eighteen out of 20 compounds were detected in all total gas + particle (IN_{total}AAS) samples (Table S8). PFHxA and PFHpA were detected in < 10% of samples, and only in the particle phase, and are excluded from further discussion.

The median indoor concentrations measured by AAS for PFAS were more than three times greater than outdoor (Fig. 2,Table S8), and were dominated by PFBA (~37% of Σ_{20}PFASs in total gas + particle (IN_{total}AAS) samples) and PFPA (22%), associated primarily with particles. Of the more volatile compounds, N-MeFOSE and N-EtFOSE were dominant, representing 25% and 12% of Σ_{18}PFASs in gas phase, respectively.

The median concentration measured by IN_{total}AAS was 76.6 pg m^{-3} with a range of 41.8–161.6 pg m^{-3}. This is lower than that observed in a study of German schools for neutral PFASs, where the N-EtFOSA median concentration was 243 pg m^{-3}, compared to 0.7 pg m^{-3} in this study (Fromme et al., 2015). However, in that study the PUF/XAD/PUF sandwich was used while in our study PUF only was used, which may have resulted in limited sorptive capacity for gaseous PFASs. In Norway, a survey of air and house dust for PFAS concluded that building age, specific furnishings and clothing play a role in indoor PFAS concentrations (Haug et al., 2011). Thus, the low concentrations of PFAS in our indoor samples may also be due to the recent construction of the building (2011), and new furnishings with traditional PFAS-containing materials no longer in use.

3.2. Passive air sampler calibration

3.2.1. Outdoor PAS

Passive sampler performance was evaluated by calculating equivalent air sample volumes for PUF-PAS and XAD-PAS. The equivalent air volume (V_{eq}, in m^3) sampled by PUF disk was determined based on the ambient air concentration and accumulated mass using equation:

$$V_{eq} = M/C_A \qquad \text{(Eq. 1)}$$

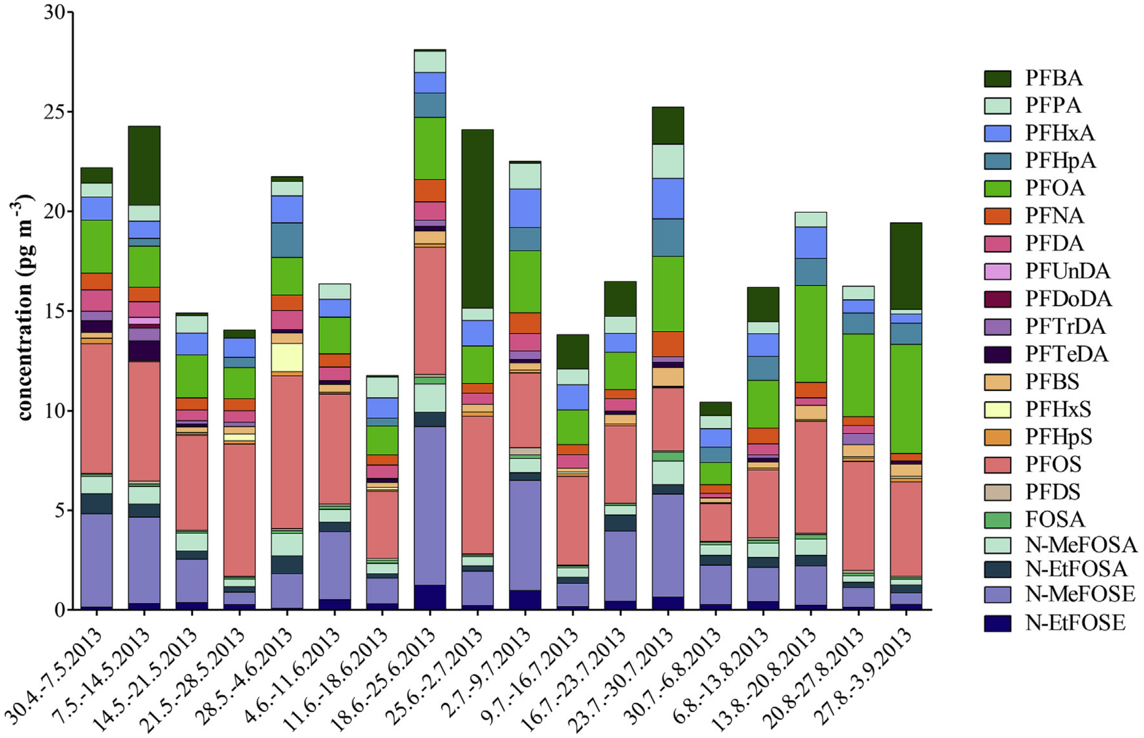

Fig. 1. Gas + particle phase concentrations (pg m^{-3}) of individual PFASs compounds in outdoor air measured by active air sampler (OUT$_{total}$AAS samples) during outdoor PAS calibration.

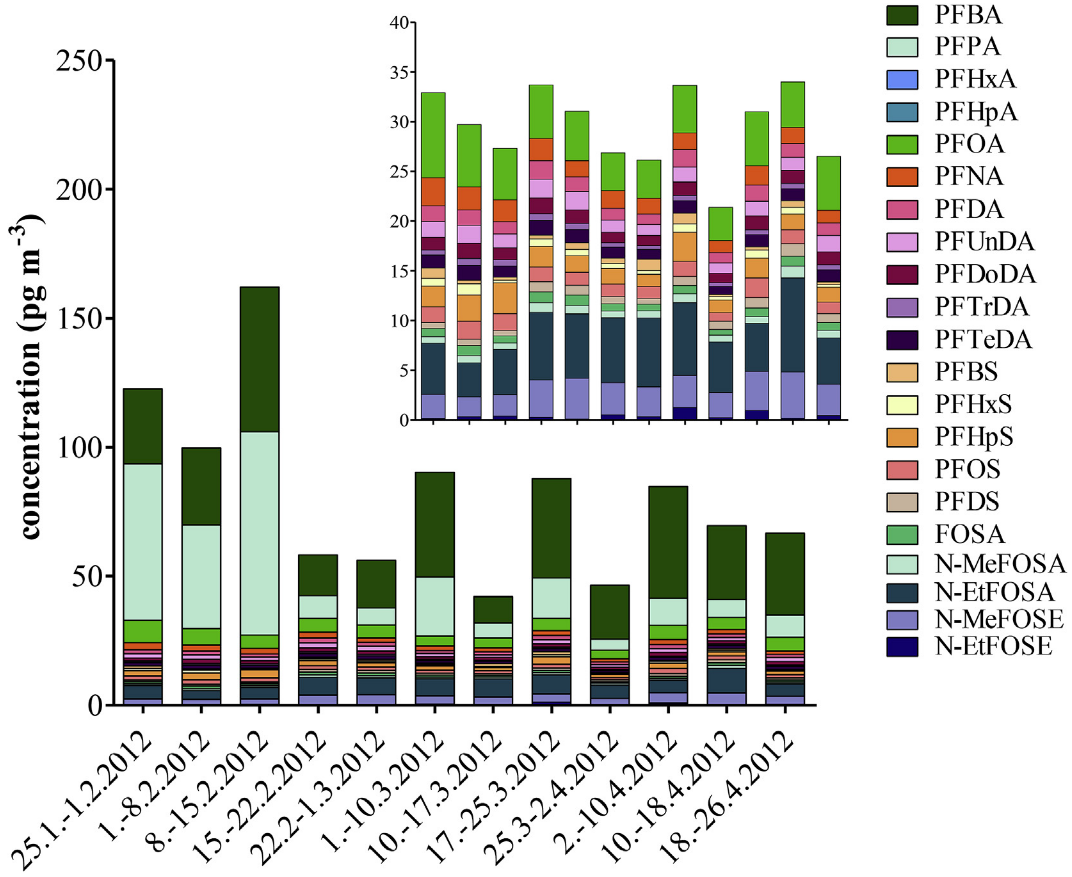

Fig. 2. Gas + particle phase concentrations (pg m^{-3}) of individual target compounds in indoor air measured by active air sampler (IN$_{total}$AAS) during indoor PAS calibration (inset figure excludes PFBA and PFPA).

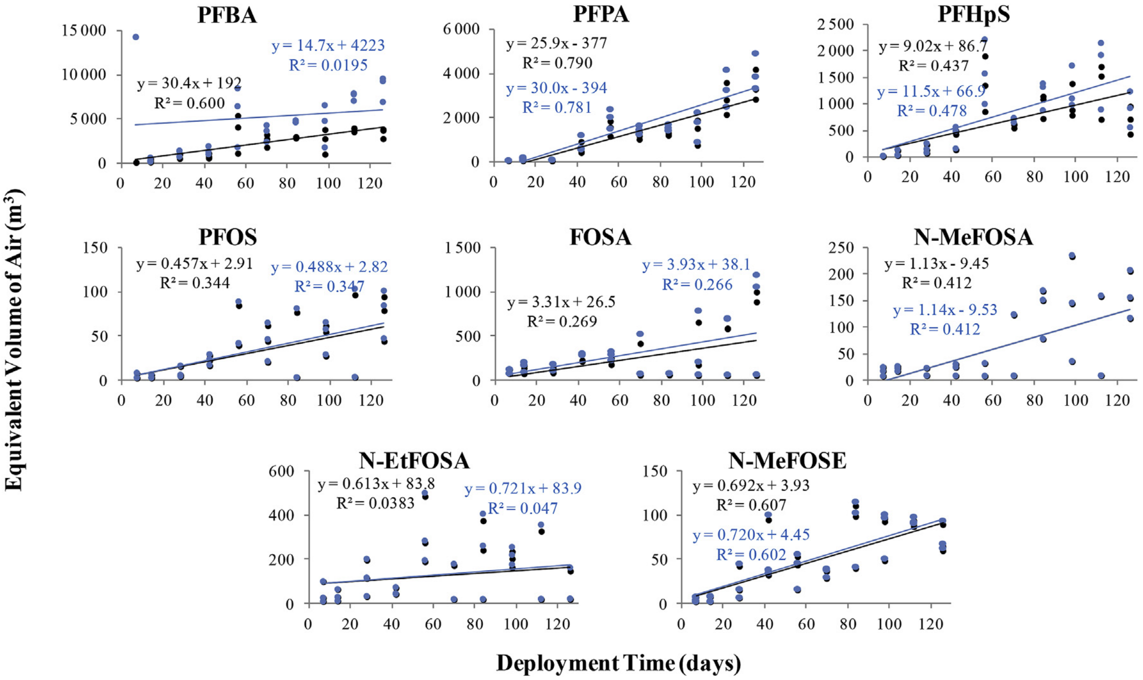

Fig. 3. Equivalent air volume sampled by OUT-PUF-PAS outdoors for PFASs over a 126-day deployment. Black points represent calibration against total air; blue points represent calibration against only gas phase. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

Where M is the mass of compounds accumulated on the PUF disk (pg), and C_A is the concentration of target compounds in ambient air (pg m^{-3}), measured with the reference AAS. The V_{eq} values were then plotted against deployment time to evaluate the uptake characteristics of the PAS. The linearity of uptake to PAS was tested using least squares regression. Calibrations were evaluated using only the gas-phase air concentrations and also using total air concentrations, to provide insight into whether the PAS are sampling gas-phase only or total air.

For air concentrations to be determined from PAS, the sampler ideally should have consistent uptake of a target compound over time, and not equilibrate within the sampler deployment period, i.e., a strong linear relationship between V_{eq} and time with a positive slope. If these criteria are met, the slope of the least squares regression line is the passive air sampling rate (R_s, m^3 day^{-1}) of the PUF disk for the corresponding PFAS (see Table S9). Plots of equivalent air volume and sampling time are shown in Fig. 3.

For comparison, the sampling rate R_s was also determined based on individual time points according to Equation (2), where t is the deployment time. Results for this method are shown in Table S10.

$$R_s = M/C_A \times t \tag{Eq. 2}$$

Sampling rates were not evaluated for compounds detected in ≤50% of total LV-AAS samples (PFUnDA, PFDoDA, PFTrDA and PFHxS).

When calculated V_{eq} values of OUT-PUF-PAS were plotted against deployment time, a linear uptake ($R^2 > 0.5$) was observed for three PFAS (PFBA, PFPA and N-MeFOSE) (Table S9, Fig. 3), suggesting that for these compounds the majority of the variability in the V_{eq} is explained by deployment time, and thus that the PUF-PAS can perform as appropriate passive samplers for these compounds. PFHpS, PFOS, FOSA and N-MeFOSA also had statistically significant linear uptake, but more scatter in the data and poorer fit of the regression. Despite significant correlations for seven of the eight PFASs, the magnitude of the slopes, representing the PAS sampling rates, was different than what is typical of non-polar compounds in outdoor double-bowl PUF-PAS, which typically have sampling rates of 2–6 m^3 day^{-1} (Bohlin et al. 2014a, 2014b). Slopes for regressions with $r^2 > 0.5$ ranged from 0.692 to 30.4 m^3 day^{-1}. For many compounds there was no significant difference between gas-phase and total sampling rates, however for some compounds the sampling rates determined based on total air concentrations rather than gas-phase only were lower (e.g., PFPA, PFHpS), suggesting that the PUF-PAS are capturing some fraction of particle-phase compounds, and thus that they are performing more like total air samplers rather than gas-phase only.

Ahrens et al. (2013) evaluated the performance of PUF-PAS for a range of PFASs; in that calibration, PFSAs were found to have consistent uptake in the PUF-PAS with a sampling rate of approximately 3.9 m^3 day^{-1}; this was within the range of what was observed for PFSAs in our study where sampling rates for PFSAs ranged from 0.45 to 30 m^3 day^{-1}. However, Ahrens et al. did not detect any PFCAs by PUF-PAS; this contrasts with our results where most consistent linear uptake was observed for the PFCAs (PFBA and PFPA, with sampling rates of 30 and 26 m^3 day^{-1}, respectively; Fig. 3). Moreover, Ahrens et al. found rapid equilibration of FOSA, methyl and ethyl FOSAs and methyl and ethyl FOSEs after a few weeks, whereas we observed linear uptake for at least 100 days for N-MeFOSE, more variable but still increasing uptake for FOSA and N-MeFOSA, but no linear uptake for N-EtFOSA. N-MeFOSE is the least volatile of the FOSE/FOSA compounds, which may account for its better performance in the passive sampler (vapour pressure of 0.0004 Pa compared with > 0.002 Pa for other FOSE/FOSA compounds) (Shoeib et al., 2004).

As only three OUT-XAD-PAS were deployed, we did not apply a least squares regression to the plot of V_{eq} vs. deployment time. To obtain an indication of the sampling rates, the sampling rate was determined based on individual time points for sampling using Equation (2). The calculated sampling rates are given in Table S10. We considered the sampling rates reliable if they were consistent between the three samplers, e.g., if the relative standard deviation of the sampling rate was less than 30%.

Mean sampling rates for individual compounds ranged from 0.7 m^3 day^{-1} (for PFOS) to 14 m^3 day^{-1} (for PFBS) (Fig. 4a). The mean sampling rate for all consistent PFASs in the XAD-PAS was higher than the typical sampling rate for XAD-PAS, but still within the range of what has previously been reported. Overall, the XAD-PAS have lower

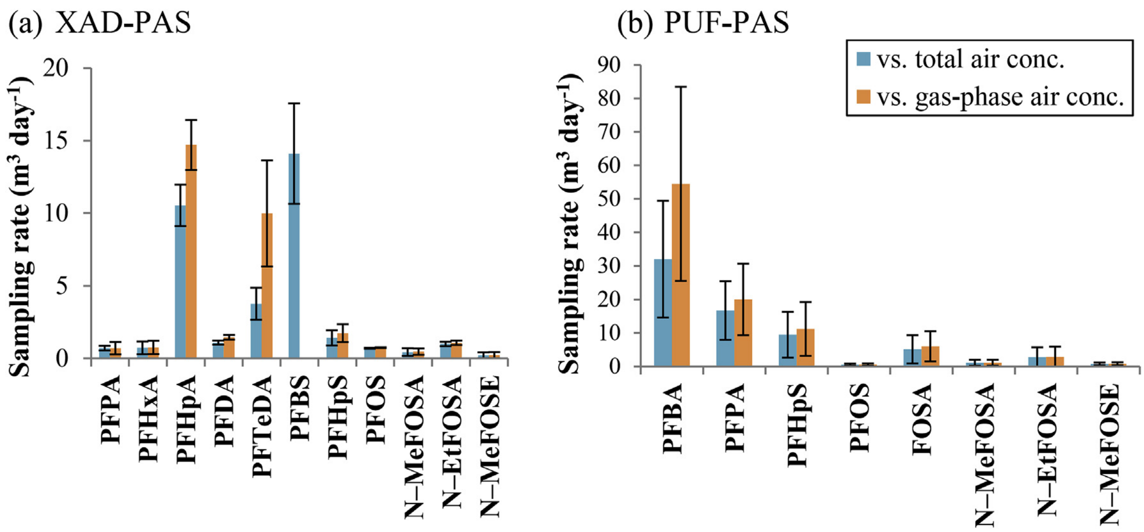

Fig. 4. Comparison of sampling rates for total vs. gas-phase only calibrations for (a) XAD-PAS and (b) PUF-PAS determined according to Equation (2).

variability in sampling rates and more consistent performance when compared with previous calibrations for the non-polar compounds (e.g., PCBs, OCPs).

However, despite the perception that PUF-PAS sample total air, while XAD-PAS sample gas-phase only (Melymuk et al., 2014), for both the XAD-PAS and PUF-PAS, there were no significant differences between the sampling rates obtained using gas-phase only vs. those using total air concentrations for either type of sampler (Student t-test, $\alpha = 0.05$). However, this may be more due to the set of PFASs that were sufficiently detected in both samplers to enable to determination of sampling rates. In both sampler types, at least one compound was detected by the LV-AAS in total air but not sufficiently in gas-phase air (PFBA in PUF-PAS and PFTeDA in XAD-PAS), but was detected in the passive sampler, suggesting some fraction of particle phase air is sampled by both types of PAS.

Overall, there is contrasting evidence for the applicability of PUF-PAS for PFASs in outdoor air; and this suggests that the uptake of PFASs to PUF is not fully understood. The complicating influences could be poor understanding of PFAS gas-particle partitioning, or sampling artifacts influencing the reference AAS as well as the PUF-PAS. While XAD-PAS are perceived to be more appropriate for polar compounds such as PFAS, they may have limited ability to collect particle-phase compounds, and thus may not be ideal for certain less volatile PFASs. However, we have also demonstrated that the XAD-PAS have sufficient uptake rates for the PFAS to be deployed for shorted time periods (e.g., 2–3 months) and thus the samplers may be applicable to assess seasonal trends, at least in non-remote locations.

While the calibration results suggest that PUF-PAS are only appropriate for a limited number of PFASs outdoors, we also considered whether they could be used for qualitative purposes, e.g., detection and identification of PFASs outdoors. We compared the outdoor air profile of PFAS captured by AAS and PAS; this indicated some similarity between samplers but also some notable differences (Fig. 5 and Table S6). The LV-AAS-total and LV-AAS-gas phase profiles were very similar, suggesting that the bulk of PFAS in outdoor air was found in the gas-phase. This supports the appropriateness of XAD-PAS for outdoor air sampling, as it is generally a gas-phase air sampler. PFBA, PFDS and N-EtFOSE were not detected in the XAD-PAS but were small contributions to the gas-phase LV-AAS, while PFTeDA, which was not in the gas-phase LV-AAS, was detected by the XAD-PAS. Beyond these small differences, almost identical sets of PFASs were detected by OUTgas-phase AAS and OUT-XAD-PAS, however the contributions of compounds varied distinctly between the two samplers: PFOS and PFOA had higher contributions in the LV-AAS, while PFHpA and PFBS had higher contributions in the XAD-PAS. While these discrepancies could be rectified in passive sampler calibration by the use of compound-specific sampling rates, if generic sampling rates are used different profiles of PFASs in air would be reported when using active vs. XAD-PAS sampling.

The discrepancies are even larger for the PUF-PAS. The profile in the PUF-PAS is dominated by PFBA and PFPA, which were two of the compounds with the most consistent linear uptake in the sampler. In contrast, PFOA and a number of other PFCAs and PFSAs are not detected by the PUF-PAS, and the contribution of PFOS is much smaller than in either the LV-AAS or XAD-PAS. Thus, PUF-PAS do not seem to be appropriate for detection or quantification of a broad range of PFASs, and may be used for quantification for only a limited number of compounds. We discuss possible reasons for the discrepancy in following sections.

3.2.2. Indoor PUF-PAS

PAS performance in indoor air differs from that in outdoor air because of the more stable conditions (e.g., low air movement thus larger boundary layer in PAS, more stable temperatures) and higher concentrations of many compounds of interest. As noted in section 3.1, the concentrations of PFASs in indoor air were 3× higher than in outdoor air. These effects have conflicting influences on PAS performance. The lower air movement should lead to lower sampling rates indoors, however the higher concentrations and often higher temperatures could decrease the length of the linear uptake phase of samplers, leading to faster equilibration.

We examined the uptake of individual PFASs to the triplicate indoor PUF-PAS, and found large differences in the uptake profiles (Fig. 6). Unlike in outdoor air, very few compounds had increasing concentrations in the PUF-PAS over time. Only PFBA, FOSA, N-MeFOSE and N-EtFOSE had positive slopes; the majority of compounds were detected in the PUF-PAS but had no increase in PUF-PAS concentrations over time, suggesting rapid equilibration in indoor air. We attribute this to the higher temperatures compared to outdoor air in this study and significantly higher indoor air concentrations. These results strongly suggest that PUF-PAS are not appropriate for PFAS indoors due to the high indoor concentrations of PFASs and the low sorptive capacity of PUF for PFASs.

As in outdoor air, we compared the profile of PFASs captured by the indoor LV-AAS and the PUF-PAS (Fig. 5). The INtotalAAS was dominated by PFBA and PFPA, while INgas-phaseAAS also had significant contributions form N-MeFOSE and N-EtFOSE. In contrast, while the PUF-PAS detected almost the same number of compounds as the AAS,

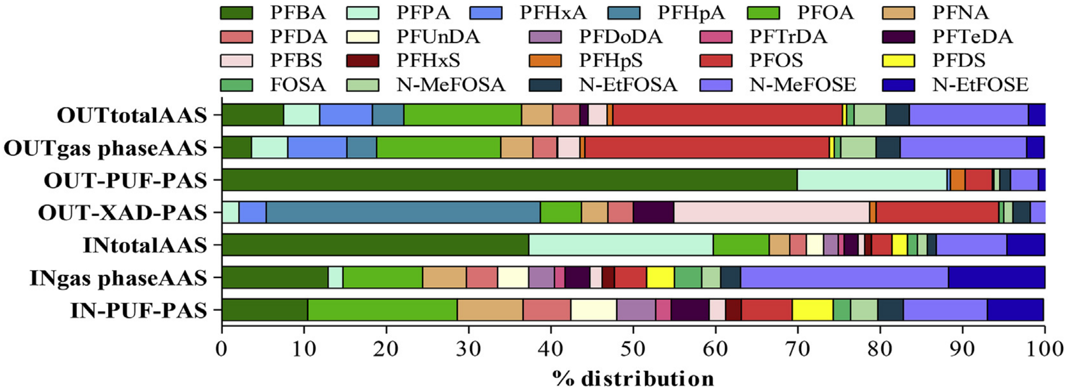

Fig. 5. PFAS distribution in outdoor (OUT) and indoor (IN) air according to different air samplers – showing the average profile measured by LV-AAS, PUF-PAS and XAD-PAS.

the profile again differed. PFOA was the highest percentage contributor, and notably, PFPA was not detected. As with outdoor air, this suggests limited use of PUF-PAS for PFASs, considering the lack of linear uptake for the majority of PFASs indoors, and different overall compound profiles.

3.3. Assessment of factors influencing PAS performance

3.3.1. Equilibration of samplers with air

PAS and AAS were compared by principle component analysis (PCA) for all PFAS (gas and particle phase) in outdoor (Figure S2) and indoor samples (Fig. 7) for the loading and variance. The PCA looks at all inter-correlated quantitative dependent variables and represents these as orthogonal variables known as principle components. The distance between samples indicates the similarity of results. In a loading plot, the distance from 0 indicates the weight to drive the variance, with those at greater distance providing greater weight.

In indoor air, we observed a clear temporal shift with indoor PAS from the 1st sample set after 7 days up to day 21, after which it appears PAS are at equilibrium with the indoor environment, indicated by the clustering in the PCA plot (Fig. 7a). Based on factor loading, the equilibrium of PAS is driven by the uptake of PFPA, PFOA and PFNA, while IN$_{total}$AAS variance is mostly due to concentrations of PFBA that were ~10 times more abundant in the IN$_{total}$AAS, and primarily in the particle phase, with median PFBA concentrations of 28.1 and 2.8 pg m^{-3} in particle and gas phase IN-AAS, respectively. Therefore, a shorter sample interval for PAS exposure may be applicable for indoor PAS use.

In contrast, outdoor samples show greater differences between PAS

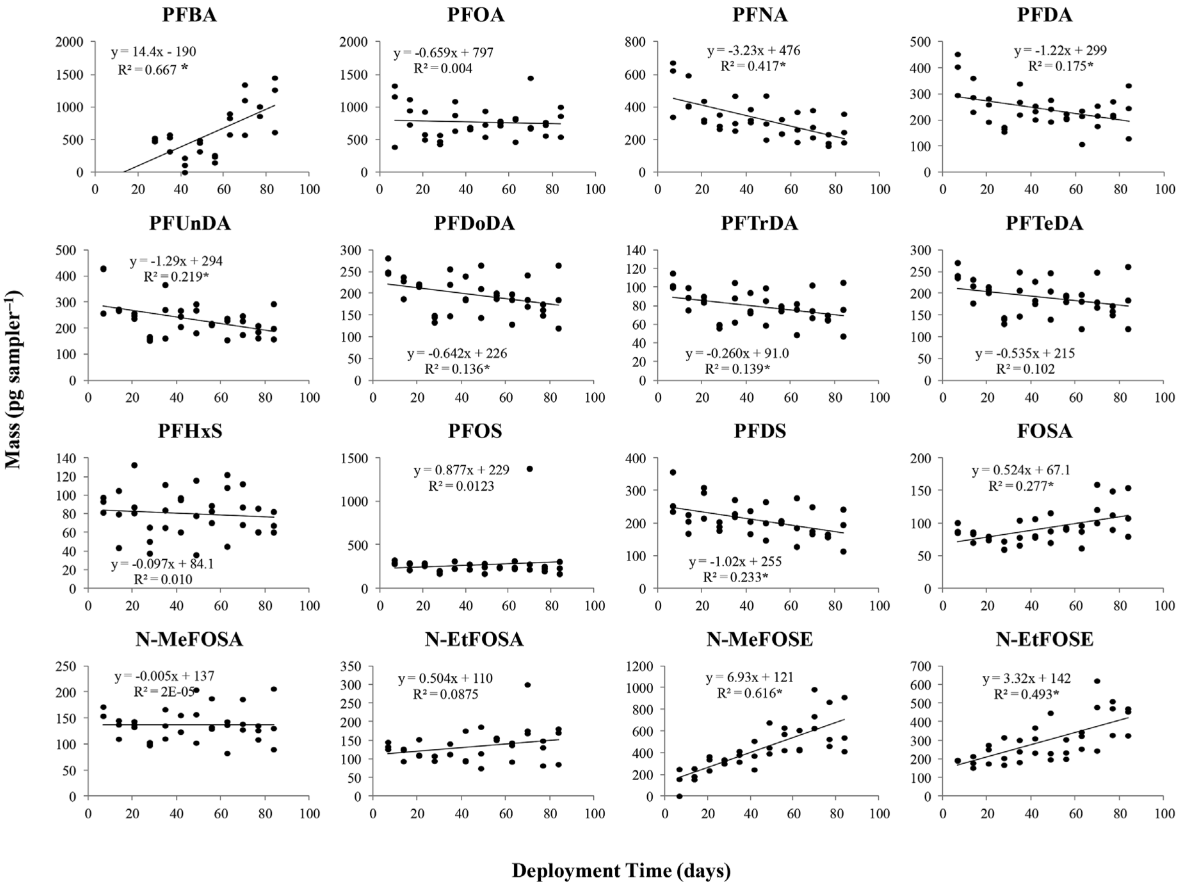

Fig. 6. Uptake to indoor PUF-PAS over 12-week (84-day) sampling period (* indicates significantly non-zero slopes).

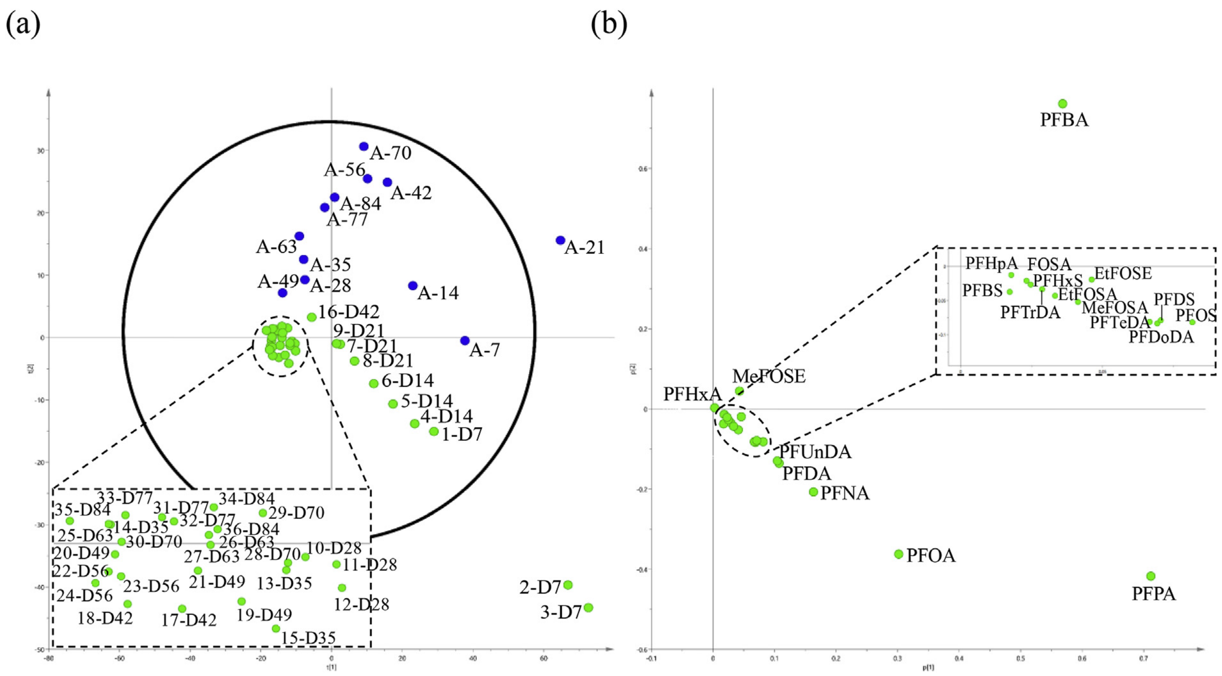

Fig. 7. Principle component analysis (PCA) of indoor PFAS samples using the total air concentration measured by AAS (blue) and total mass captured by PAS (green). The insets on each PCA indicate compounds within the dotted region. Figure (a) shows loading of individual air samples, with AAS labelled with "A–collection day" while PAS are labelled by sample number and the day (D) of collection with 0 as the initial day of PAS deployment and figure (b) shows variance. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

and AAS, and no temporal trend towards equilibrium as was observed for the indoor samples. This supports what is shown in Fig. 3, as the majority of PFASs have increasing uptake over time in outdoor air, while in indoor air the relationship between PUF-PAS mass and time suggests rapid achievement of equilibrium (Fig. 6). Outdoor AAS have low variation and neither gas-phase, particle-phase nor total concentrations compare well with that observed in PUF-PAS (Figure S2). Variance observed in outdoor samples were largely caused by concentrations of PFOS, PFOA and *N*-MeFOSE for AAS and PFBA, PFPA and PFHpA.

3.3.2. Gas-particle partitioning

As one concern in the use of PAS is variable and/or ineffective particle capture by the PAS, the distribution of compounds between the gas and particle phase may make PAS less effective for some PFAS. Furthermore, unlike many POPs such as polychlorinated biphenyls, PFAS are ionisable and it is therefore expected that their partitioning behaviour will also be affected by aqueous aerosols (Kim and Kannan, 2007; Ahrens et al., 2013). In the gas phase, PFAS are in their neutral form while in aqueous solutions they may dissociate to form ionic PFAS. Studies of atmospheric PFAS sometimes focus only on particle- or gas-phase, or they treat the two phases as a single sample. For example, while ionic PFASs have been reported previously (e.g. Barber et al., 2007; Kim and Kannan, 2007; Stock et al., 2007; Dreyer and Ebinghaus, 2009; Weinberg et al., 2011a,b; Liu et al., 2015), their quantification is often limited to only the atmospheric particle phase. Given the physicochemical properties of many PFASs, this is not surprising as many of the ionic PFASs are estimated to be predominantly in the particle phase (Ahrens et al., 2011a,b). However, in this study the particle-bound fraction, assumed to be captured by the QFF in the AAS, and the gas-phase fraction, captured by the PUF/XAD/PUF sandwich, were quantified separately and thus were used for gas-particle partitioning assessment to provide further insight into the observed PAS performance.

Because uncertainties in the sampling of PFAS by PUF sorbent used indoors, we determine gas-particle partitioning only in outdoor samples, where XAD was included as a gas-phase sorbent. For indoor environments, dust has been used as a monitor for particle phase PFAS (Goosey and Harrad, 2011; Padilla-Sánchez and Haug, 2016; Björklund et al., 2009; Reiner et al., 2015), though this does not reflect inhalation exposure. In outdoor air, the most abundant ionic compounds on QFFs were PFBA (24% of Σ_{17}PFASs in particle-phase) with a median of 1.3 pg m^{-3}, PFOS (20%) with a median of 0.3 pg m^{-3} and PFOA (9%) with a median of 0.2 pg m^{-3}, N-MeFOSE was the dominant volatile PFAS (7% of Σ_{17}PFASs in particle-phase).

The particulate associated fraction of individual PFAS groups was calculated using equation:

$$\Phi = C_{\text{particle phase}}/C_{\text{particle+gas–phase}} \qquad \text{(Eq. 3)}$$

The obtained particle-associated fraction (Φ) in outdoor air was found to be statistically lower for PFASs (p = 0.020) and FOSA/Es (p = 0.002) compared to short-chain PFCAs. Statistical differences were also observed between long-chain PFCAs and FOSA/Es (p = 0.008) (Fig. 8).

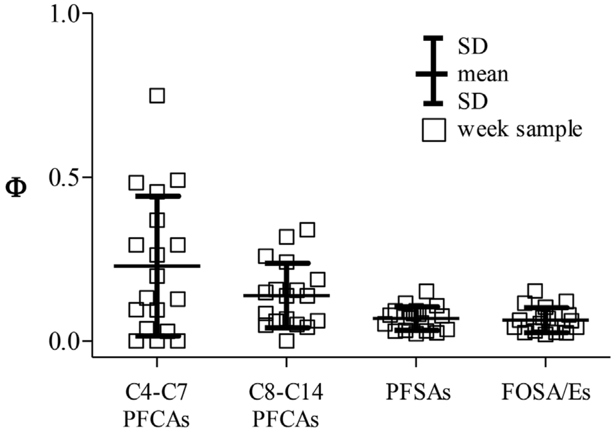

Fig. 8. Particle associated fraction of PFASs groups calculated using Eq. (3) for each individual sampling week in outdoor air samples.

For several compounds, partitioning can be clearly related to their physicochemical properties, with the more volatile compounds such as the FOSA/Es predominantly found in the gas phase (91%), with Φ of 0.06 ± 0.04. In the case of PFCAs, they distribute between both particles and gas phase. Φ had a wide range, between 0.02 (PFHxA) and 0.94 (PFTeDA). PFASs were also primarily in the gas phase ($\Phi = 0.07 \pm 0.04$); except PFHxS, which was predominantly detected on particles, but as PFHxS was detected in less than 50% of all samples, this result may have higher uncertainty. PFOA and PFOS, compounds on which legislative bodies are currently focused, i.e. Stockholm Convention, are predicted to be more volatile and this is confirmed by the findings in this study ($\Phi_{(PFOA)} = 0.08 \pm 0.07$; $\Phi_{(PFOS)} = 0.07 \pm 0.04$).

In general, the majority of target compounds were associated with gas phase of ambient air, which is a useful for PAS, considering the variable or restricted particle-phase uptake (Markovic et al., 2015). This is similar to the findings of Arp and Goss (2009) who determined that the majority of PFCAs would be primarily in the gas-phase and that deposition to filters may be associated with relative humidity, thus those PFCAs measured on filters may result from vapour phase uptake rather than particle deposition. Particle-associated compounds in this study increased with greater compound chain length, in keeping with what has been seen in other studies (Vierke et al., 2011). However, when compared to a high volume AAS sampler, the fraction of particle-bound longer chain PFASs is less in this study than that seen in some other studies (Ahrens et al., 2012). As a range of related factors influence partitioning behaviour of atmospheric compounds between gas and particulate phases, including temperature, humidity, organic matter content of aerosols, etc., the particle-associated fraction may shift noticeably based on ambient conditions and this may influence some of the differences observed in this study.

3.3.3. *Uptake behaviour of PFAS on PUF-PAS*

The poor performance of PUF-PAS for PFAS has been explored previously and there are some theories suggesting that the partitioning of PFAS is different compared to other compounds such as PCBs, OCPs and PBDEs due to the hydrophilic polar features of PFAS. PUF sorptive capacity is typically represented by the PUF-air partition coefficient (log $K_{PUF\text{-}Air}$), estimated based on the octanol-air partition coefficient, (K_{OA}), as described by Shoeib and Harner (2002) for hydrophobic nonpolar POPs (Eq (4)).

$$\log K_{PUF-Air} = 0.6366 \log K_{OA} - 3.1774 \quad \text{(Eq. 4)}$$

Using Eq. (4), log $K_{PUF\text{-}Air}$ was estimated for PFASs. Log K_{OA} values from Lei et al., (2004) for N-EtFOSA, N-MeFOSE, and N-EtFOSE and Kim et al. (2015) for other PFAS were used. Log $K_{PUF\text{-}Air}$ for PFASs ranged from −0.5 for PFBA to 1.7 for PFTeDA (SI Table S11).

In comparison, PCBs have log $K_{PUF\text{-}Air}$ from 1.7 for PCB 18 to 3.7 for PCB 156 (Shoeib and Harner, 2002), and PBDEs from 2.7 for BDE 17 to 4.6 for BDE 126 (Harner and Shoeib, 2002). Thus, the estimated $K_{PUF\text{-}Air}$ values suggest weaker partitioning of gaseous PFASs to PUF than for the hydrophobic POPs. Moreover, other studies of the partitioning of PFAS compounds in the atmosphere have indicated that the gas-particle partitioning relationships that have been derived for non-polar compounds do not fit well for PFAS compounds based upon their K_{OA}, e.g., that partitioning to PUF is even weaker than predicted based on Eq. (4) (Shoeib et al., 2008). It has been suggested that the carbon-fluorine chain length, atmospheric moisture content, functional groups and airborne particles may be key factors in the sorption behaviour of PFAS to PUF-PAS (Ahrens et al., 2013).

4. Conclusions and recommendations

The use of XAD-PAS as passive air samplers appears to be a useful tool in the measurement of PFASs but there is still variability between those compounds detected by active air and passive samplers. PUF-PAS may have some limited use in characterizing some of the sulfonate PFAS compounds, but the variable accumulation of many PFASs in this PAS make it a poor media for long-term monitoring. Moreover, the qualitative determination of PFAS profiles is not appropriate with PUF-PAS due to different sorptive capacities for different classes of PFASs. The use of PUF-PAS indoor environments still needs further exploration as equilibrium was reached rapidly and more study on the limitations are needed.

Further study is needed for PUF-PAS applicability for fluorotelomer alcohols (FTOHs) as well as comparison to other atmospheric PAS samplers such as ENV+ (Biotage, Uppsala, Sweden) and SIPs. The use of PAS samplers to predict individual exposure is limited due to the number of environments to which a person is exposed and personal passive samplers may be key to this understanding (Padilla-Sánchez et al., 2017).

In order to fulfil properly global monitoring needs, caution should be given to interpretation of global distributions of PFASs when based on PUF-PAS. Considering the weakly significant uptake rates in the outdoor calibration for PFOS, and better uptake for PFCAs, PUF-PAS may be acceptable to identify large differences in concentration gradients on global scales (e.g., semi-quantitative/order of magnitude differences), however they are not able to provide accurate concentrations for a broad set of PFASs. XAD-PAS are a more reliable option which give profiles that more closely resemble those from AAS, and it may be possible to achieve seasonal resolution with XAD-PAS. Other PAS options (e.g., SIP-PAS) are also useful for seasonal resolution and broader compound coverage. Given the importance of establishing reliable long-term monitoring for PFASs, passive sampling techniques for these compounds should continue to be investigated and optimized.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.atmosenv.2018.05.030.

References

Ahrens, L., Shoeib, M., Del Vento, S., Codling, G., Halsall, C., 2011a. Polyfluoroalkyl compounds in the canadian arctic atmosphere. Environ. Chem. 8 (4), 399–406.

Ahrens, L., Shoeib, M., Harner, T., Lee, S.C., Guo, R., Reiner, E.J., 2011b. Wastewater treatment plant and landfills as sources of polyfluoroalkyl compounds to the atmosphere. Environ. Sci. Technol. 45 (19), 8098–8105.

Ahrens, L., Harner, T., Shoeib, M., Lane, D.A., Murphy, J.G., 2012. Improved characterization of gas–particle partitioning for per- and polyfluoroalkyl substances in the atmosphere using annular diffusion denuder samplers. Environ. Sci. Technol. 46 (13), 7199–7206.

Ahrens, L., Harner, T., Shoeib, M., Koblizkova, M., Reiner, E.J., 2013. Characterization of two passive air samplers for per- and polyfluoroalkyl substances. Environ. Sci. Technol. 47 (24), 14024–14033.

Antweiler, R.C., 2015. Evaluation of statistical treatments of left-censored environmental data using coincident uncensored data sets. II. Group comparisons. Environ. Sci. Technol. 49 (22), 13439–13446.

Arp, H.P.H., Goss, K.-U., 2009. Gas/particle partitioning behavior of perfluorocarboxylic acids with terrestrial aerosols. Environ. Sci. Technol. 43 (22), 8542–8547.

Barber, J.L., Berger, U., Chaemfa, C., Huber, S., Jahnke, A., Temme, C., Jones, K.C., 2007. Analysis of per- and polyfluorinated alkyl substances in air samples from Northwest Europe. J. Environ. Monit. 9 (6), 530–541.

Benskin, J.P., Ahrens, L., Muir, D.C.G., Scott, B.F., Spencer, C., Rosenberg, B., Tomy, G., Kylin, H., Lohmann, R., Martin, J.W., 2012. Manufacturing origin of perfluorooctanoate (PFOA) in atlantic and canadian arctic seawater. Environ. Sci. Technol. 46 (2), 677–685.

Björklund, J.A., Thuresson, K., De Wit, C.A., 2009. Perfluoroalkyl compounds (PFCs) in indoor dust: concentrations, human exposure estimates, and sources. Environ. Sci. Technol. 43 (7), 2276–2281.

Bohlin, P., Audy, O., Skrdlikova, L., Kukucka, P., Pribylova, P., Prokes, R., Vojta, S., Klanova, J., 2014a. Outdoor passive air monitoring of semi volatile organic compounds (SVOCs): a critical evaluation of performance and limitations of polyurethane foam (PUF) disks. Environ. Sci. Process. Impacts 16 (3), 433–444.

Bohlin, P., Audy, O., Skrdlikova, L., Kukucka, P., Vojta, S., Pribylova, P., Prokes, R., Cupr, P., Klanova, J., 2014b. Evaluation and guidelines for using polyurethane foam (PUF) passive air samplers in double-dome chambers to assess semi-volatile organic compounds (SVOCs) in non-industrial indoor environments. Environ. Sci. Process. Impacts 16 (11), 2617–2626.

Butt, C.M., Berger, U., Bossi, R., Tomy, G.T., 2010. Levels and trends of poly- and perfluorinated compounds in the arctic environment. Sci. Total Environ. 408 (15), 2936–2965.

Chaemfa, C., Barber, J.L., Huber, S., Breivik, K., Jones, K.C., 2010. Screening for PFOS and PFOA in European air using passive samplers. J. Environ. Monit. 12 (5), 1100–1109.

Codling, G., Vogt, A., Jones, P.D., Wang, T., Wang, P., Lu, Y.L., Corcoran, M., Bonina, S., Li, A., Sturchio, N.C., Rockne, K.J., Ji, K., Khim, J.-S., Naile, J.E., Giesy, J.P., 2014. Historical trends of inorganic and organic fluorine in sediments of Lake Michigan. Chemosphere 114, 203–209.

Dreyer, A., Ebinghaus, R., 2009. Polyfluorinated compounds in ambient air from ship- and land-based measurements in northern Germany. Atmos. Environ. 43 (8), 1527–1535.

Fromme, H., Dreyer, A., Dietrich, S., Fembacher, L., Lahrz, T., Völkel, W., 2015. Neutral polyfluorinated compounds in indoor air in Germany – the LUPE 4 study. Chemosphere 139, 572–578.

Giesy, J.P., Kannan, K., 2002. Peer reviewed: perfluorochemical surfactants in the environment. Environ. Sci. Technol. 36 (7), 146A–152A.

Gomez, C., Vicente, J., Echavarri-Erasun, B., Porte, C., Lacorte, S., 2011. Occurrence of perfluorinated compounds in water, sediment and mussels from the Cantabrian Sea (North Spain). Mar. Pollut. Bull. 62 (5), 948–955.

Goosey, E., Harrad, S., 2011. Perfluoroalkyl compounds in dust from Asian, Australian, European, and North American homes and UK cars, classrooms, and offices. Environ. Int. 37 (1), 86–92.

Harner, T., Shoeib, M., 2002. Measurements of Octanol−Air partition coefficients (KOA) for polybrominated diphenyl ethers (PBDEs): predicting partitioning in the environment. J. Chem. Eng. Data 47 (2), 228–232.

Haug, L.S., Huber, S., Schlabach, M., Becher, G., Thomsen, C., 2011. Investigation on per- and polyfluorinated compounds in paired samples of house dust and indoor air from Norwegian homes. Environ. Sci. Technol. 45 (19), 7991–7998.

Hung, H., Katsoyiannis, A.A., Brorström-Lundén, E., Olafsdottir, K., Aas, W., Breivik, K., Bohlin-Nizzetto, P., Sigurdsson, A., Hakola, H., Bossi, R., Skov, H., Sverko, E., Barresi, E., Fellin, P., Wilson, S., 2016. Temporal trends of persistent organic Pollutants (POPs) in arctic air: 20 years of monitoring under the arctic monitoring and assessment programme (AMAP). Environ. Pollut. 217, 52–61.

Karásková, P., Venier, M., Melymuk, L., Bečanová, J., Vojta, Š., Prokeš, R., Diamond, M.L., Klánová, J., 2016. Perfluorinated alkyl substances (PFASs) in household dust in Central Europe and North America. Environ. Int. 94, 315–324.

Kim, S.-K., Kannan, K., 2007. Perfluorinated acids in air, rain, snow, surface runoff, and lakes: relative importance of pathways to contamination of urban lakes. Environ. Sci. Technol. 41 (24), 8328–8334.

Kim, M., Li, L.Y., Grace, J.R., Yue, C., 2015. Selecting reliable physicochemical properties of perfluoroalkyl and polyfluoroalkyl substances (PFASs) based on molecular descriptors. Environ. Pollut. (Barking, Essex : 1987) 196, 462–472.

Klánová, J., Harner, T., 2013. The challenge of producing reliable results under highly variable conditions and the role of passive air samplers in the Global Monitoring Plan. Trac. Trends Anal. Chem. 46, 139–149.

Koblizkova, M., Genualdi, S., Lee, S.C., Harner, T., 2012. Application of sorbent impregnated polyurethane foam (SIP) disk passive air samplers for investigating organochlorine pesticides and polybrominated diphenyl ethers at the global scale. Environ. Sci. Technol. 46 (1), 391–396.

Langer, V., Dreyer, A., Ebinghaus, R., 2010. Polyfluorinated compounds in residential and nonresidential indoor air. Environ. Sci. Technol. 44 (21), 8075–8081.

Lei, Y.D., Wania, F., Mathers, D., Mabury, S.A., 2004. Determination of vapor pressures, Octanol−Air, and Water−Air partition coefficients for polyfluorinated sulfonamide, sulfonamidoethanols, and telomer alcohols. J. Chem. Eng. Data 49 (4), 1013–1022.

Liu, B., Zhang, H., Yao, D., Li, J., Xie, L., Wang, X., Wang, Y., Liu, G., Yang, B., 2015. Perfluorinated compounds (PFCs) in the atmosphere of Shenzhen, China: spatial distribution, sources and health risk assessment. Chemosphere 138, 511–518.

Markovic, M.Z., Prokop, S., Staebler, R.M., Liggio, J., Harner, T., 2015. Evaluation of the particle infiltration efficiency of three passive samplers and the PS-1 active air sampler. Atmos. Environ. 112, 289–293.

Melymuk, L., Bohlin, P., Sáňka, O., Pozo, K., Klánová, J., 2014. Current challenges in air sampling of semivolatile organic contaminants: sampling artifacts and their influence on data comparability. Environ. Sci. Technol. 48 (24), 14077–14091.

Padilla-Sánchez, J.A., Haug, L.S., 2016. A fast and sensitive method for the simultaneous analysis of a wide range of per-and polyfluoroalkyl substances in indoor dust using on-line solid phase extraction-ultrahigh performance liquid chromatography-time-of-flight-mass spectrometry. J. Chromatogr. A 1445, 36–45.

Padilla-Sánchez, J.A., Papadopoulou Poothong, E.S., Haug, L.S., 2017. Investigation of the best approach for assessing human exposure to poly- and perfluoroalkyl substances through indoor air. Environ. Sci. Technol. 51, 12836–12843.

Poothong, S., Thomsen, C., Padilla-Sanchez, J.A., Papadopoulou, E., Haug, L.S., 2017. Distribution of novel and well-known poly-and perfluoroalkyl substances (PFASs) in human serum, plasma, and whole blood. Environ. Sci. Technol. 51 (22), 13388–13396.

Prevedouros, K., Cousins, I.T., Buck, R.C., Korzeniowski, S.H., 2006. Sources, fate and transport of perfluorocarboxylates. Environ. Sci. Technol. 40 (1), 32–44.

Reiner, Jessica L., Blaine, Andrea C., Higgins, Christopher P., Huset, Carin, Jenkins, Thomas M., Kwadijk, Christiaan J.A.F., Lange, Cleston C., et al., 2015. Polyfluorinated substances in abiotic standard reference materials. Anal. Bioanal. Chem. 407 (no. 11), 2975–2983.

Schuster, J.K., Gioia, R., Harner, T., Lee, S.C., Breivik, K., Jones, K.C., 2012. Assessment of sorbent impregnated PUF disks (SIPs) for long-term sampling of legacy POPs. J. Environ. Monit. 14 (1), 71–78.

Shoeib, M., Harner, T., 2002. Characterization and comparison of three passive air samplers for persistent organic Pollutants. Environ. Sci. Technol. 36 (19), 4142–4151.

Shoeib, M., Harner, T., Ikonomou, M., Kannan, K., 2004. Indoor and outdoor air concentrations and phase partitioning of perfluoroalkyl sulfonamides and polybrominated diphenyl ethers. Environ. Sci. Technol. 38 (5), 1313–1320.

Shoeib, M., Harner, T., Wilford, B.H., Jones, K.C., Zhu, J., 2005. Perfluorinated sulfonamides in indoor and outdoor air and indoor Dust: occurrence, partitioning, and human exposure. Environ. Sci. Technol. 39 (17), 6599–6606.

Shoeib, M., Harner, T., Lee, S.C., Lane, D., Zhu, J., 2008. Sorbent-impregnated polyurethane foam disk for passive air sampling of volatile fluorinated chemicals. Anal. Chem. 80 (3), 675–682.

Shoeib, M., Vlahos, P., Harner, T., Peters, A., Graustein, M., Narayan, J., 2010. Survey of polyfluorinated chemicals (PFCs) in the atmosphere over the northeast Atlantic Ocean. Atmos. Environ. 44 (24), 2887–2893.

Shoeib, M., Harner, T.M., Webster, G., Lee, S.C., 2011. Indoor sources of poly-and perfluorinated compounds (PFCS) in Vancouver, Canada: implications for human exposure. Environ. Sci. Technol. 45 (19), 7999–8005.

Stock, N.L., Furdui, V.I., Muir, D.C.G., Mabury, S.A., 2007. Perfluoroalkyl contaminants in the canadian arctic: evidence of atmospheric transport and local contamination. Environ. Sci. Technol. 41 (10), 3529–3536.

Sun, H., Li, F., Zhang, T., Zhang, X., He, N., Song, Q., Zhao, L., Sun, L., Sun, T., 2011. Perfluorinated compounds in surface waters and WWTPs in Shenyang, China: mass flows and source analysis. Water Res. 45 (15), 4483–4490.

Vierke, L., Ahrens, L., Shoeib, M., Reiner, E.J., Guo, R., Palm, W.-U., Ebinghaus, R., Harner, T., 2011. Air concentrations and particle–gas partitioning of polyfluoroalkyl compounds at a wastewater treatment plant. Environ. Chem. 8 (4), 363–371.

Wang, T., Lu, Y., Chen, C., Naile, J.E., Khim, J.S., Park, J., Luo, W., Jiao, W., Hu, W., Giesy, J.P., 2011. Perfluorinated compounds in estuarine and coastal areas of north Bohai Sea, China. Mar. Pollut. Bull. 62 (8), 1905–1914.

Wania, F., Shen, L., Lei, Y.D., Teixeira, C., Muir, D.C.G., 2003. Development and calibration of a resin-based passive sampling system for monitoring persistent organic Pollutants in the atmosphere. Environ. Sci. Technol. 37 (7), 1352–1359.

Weinberg, I., Dreyer, A., Ebinghaus, R., 2011a. Landfills as sources of polyfluorinated compounds, polybrominated diphenyl ethers and musk fragrances to ambient air. Atmos. Environ. 45 (4), 935–941.

Weinberg, I., Dreyer, A., Ebinghaus, R., 2011b. Waste water treatment plants as sources of polyfluorinated compounds, polybrominated diphenyl ethers and musk fragrances to ambient air. Environ. Pollut. 159 (1), 125–132.

Wu, M., Sun, R., Wang, M., Liang, H., Ma, S., Han, T., Xia, X., Ma, J., Tang, L., Sun, Y., Xu, G., 2017. Analysis of perfluorinated compounds in human serum from the general population in Shanghai by liquid chromatography-tandem mass spectrometry (LC-MS/MS). Chemosphere 168, 100–105.

Yang, L., Zhu, L., Liu, Z., 2011. Occurrence and partition of perfluorinated compounds in water and sediment from Liao River and Taihu Lake, China. Chemosphere 83 (6), 806–814.

APPENDIX 18

Melymuk, Lisa, Pernilla Bohlin Nizzetto, Tom Harner, Kevin B. White, Xianyu Wang, Maria Yumiko Tominaga, Jun He, Jun Li, Jianmin Ma, Wan-Li Ma, Beatriz H. Aristizábal, Annekatrin Dreyer, Begoña Jiménez, Juan Muñoz-Arnanz, Mustafa Odabasi, Yetkin Dumanoglu, Baris Yaman, Carola Graf, Andrew Sweetman, Jana Klánová. 2021. "Global Intercomparison of Polyurethane Foam Passive Air Samplers Evaluating Sources of Variability in SVOC Measurements." *Environmental Science and Policy* 125: 1-9. https://doi.org/10.1016/j.envsci.2021.08.003

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Global intercomparison of polyurethane foam passive air samplers evaluating sources of variability in SVOC measurements

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ABSTRACT

Polyurethane foam passive air samplers (PUF-PAS) are the most common type of passive air sampler used for a range of semi-volatile organic compounds (SVOCs), including regulated persistent organic pollutants (POPs) and polycyclic aromatic hydrocarbons (PAHs), and emerging contaminants (e.g., novel flame retardants, phthalates, current-use pesticides). Data from PUF-PAS are key indicators of effectiveness of global regulatory actions on SVOCs, such as the Global Monitoring Plan of the Stockholm Convention on Persistent Organic Pollutants. While most PUF-PAS use similar double-dome metal shielding, there is no standardized dome size, shape, or deployment configuration, with many different PUF-PAS designs used in regional and global monitoring. Yet, no information is available on the comparability of data from studies using different PUF-PAS designs. We brought together 12 types of PUF-PAS used by different research groups around the world and deployed them in a multi-part intercomparison to evaluate the variability in reported concentrations introduced by different elements of PAS monitoring. PUF-PAS were deployed for 3 months in outdoor air in Kjeller, Norway in 2015–2016 in three phases to capture (1) the influence of sampler design on data comparability, (2) the influence of analytical variability when samplers are analyzed at different laboratories, and (3) the overall variability in global monitoring data introduced by differences in sampler configurations and analytical methods. Results indicate that while differences in sampler design (in particular, the spacing between the upper and lower sampler bowls) account for up to 50 % differences in masses collected by samplers, the variability introduced by analysis in different laboratories far exceeds this amount, resulting in differences spanning orders of magnitude for POPs and PAHs. The high level of variability due to analysis in different laboratories indicates that current SVOC air sampling data (i.e., not just for PUF-PAS but likely also for active air sampling) are not directly comparable between laboratories/monitoring programs. To support on-going efforts to mobilize more SVOC data to contribute to effectiveness evaluation, intercalibration exercises to account for uncertainties in air sampling, repeated at regular intervals, must be established to ensure analytical comparability and avoid biases in global-scale assessments of SVOCs in air caused by differences in laboratory performance.

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1. Introduction

Long-term global data on atmospheric levels of semi-volatile organic compounds (SVOCs), including polycyclic aromatic hydrocarbons (PAHs) and persistent organic pollutants (POPs), such as polychlorinated biphenyls (PCBs), organochlorine pesticides (OCPs), and polybrominated diphenyl ethers (PBDEs), are a fundamental need in efforts to reduce emissions and minimize human and environmental exposure. This need has been formalized in the requirements of international actions, such as the Stockholm Convention on POPs (Articles 11 and 16) implemented through the Global Monitoring Plan (GMP), the UNECE Convention on Long-Range Transboundary Air Pollution (CLRTAP), and the development of a Global Earth Observation System of Systems (GEOSS) to increase our understanding of global processes and to underpin decision-making through sharing of accessible, high quality interoperable environmental data.

The GMP has a clear policy mandate to collect comparable, harmonized and reliable information on POP levels in core environmental matrices, one of which is ambient air. The Global Observation System for Persistent Organic Pollutants (GOS4POPs) is an initiative within the Group on Earth Observations (GEO) to increase the availability and quality of Earth observation data on POPs, and improve data availability and interoperability across POP monitoring networks, providing support for international conventions on toxic compounds (Stockholm Convention, CLRTAP) and on-going international programs (e.g., GMP, European Monitoring and Evaluation Programme (EMEP)).

Polyurethane foam passive air samplers (PUF-PAS) are widely used in international air monitoring of POPs (Borůvková et al., 2015; Muñoz-Arnanz et al., 2016; Pozo et al., 2009; Wania and Shunthirasingham, 2020) and other semi-volatile organic compounds (SVOCs). The spatial coverage and ease-of-use of PUF-PAS has been crucial in enabling the development of international air monitoring programs such as GAPS (Global Atmospheric Passive Sampling) and MONET (Lee et al., 2007; Muñoz-Arnanz et al., 2018, 2016; Pozo et al., 2006; Přibylová et al., 2012; Rauert et al., 2018; Roscales et al., 2018a; White et al., 2021), and their use in many individual case studies has greatly increased our knowledge of atmospheric levels of SVOCs. Following the entry-into-force of the Stockholm Convention in 2004, the GMP was established to secure monitoring data in core media (ambient air, breast milk, human blood) and became a strong driver for the development of passive air sampling programs to address global data gaps, especially given the simplicity and relatively low cost of passive air samplers (Klánová and Harner, 2013). The first GMP Report (UNEP, 2009) called for improved collaboration within and among regions, and establishment of strategic partnerships with expert laboratories and programs to address the challenges in setting up new POP monitoring programs that can continually adapt to include newly listed POPs. To mobilise such data and ensure their interoperability, we must move towards more harmonised monitoring frameworks with comprehensive datasets. While internal consistency of data within individual programs is necessary to assess long-term trends, the comparability of data among different programs must also be improved so that datasets can be combined for more effective global assessment. However, despite the intended goal of global-scale comparability, differences in analytical methods and sampler configurations between institutes and monitoring programs may affect performance (Holt et al., 2017; Markovic et al., 2015; Roscales et al., 2018b) and decrease the comparability of international monitoring data (Su and Hung, 2010).

The simple design of the PUF-PAS has led to many individually-designed versions around the globe, all following the same original PUF-PAS concept (Shoeib and Harner, 2002) of a PUF disk protected by a metal double-dome housing, but without standardized geometry. In a previous comparison of three samplers, differences in sampler design were found to have no discernable effect on PUF-PAS uptake rates (Chaemfa et al., 2008), however, today the use of PUF-PAS has greatly expanded due to ease of deployment and use, and sampler designs differ to a much greater extent. At least 15 different designs are regularly used, with differences in dome size and shape, placement of the PUF disk relative to the gap between domes, size and density of the PUF disk itself, and deployment practices (i.e., fixed versus freely hanging). In addition, there are clear differences across laboratories in analytical methodology applied to PUF processing and SVOC analysis, which have the potential to lead to large variabilities in reported concentrations (Su et al., 2011; Su and Hung, 2010). Current efforts to harmonize and synthesize global SVOC monitoring combine data collected from different PUF-PAS sampler designs and analyzed in different laboratories (e.g., GMP incorporates PUF-PAS data from five different air monitoring networks globally; Fig. S1), but lack information on how the variability introduced by physical PUF-PAS design parameters compares to the analytical variability between laboratories.

To evaluate the comparability of global SVOC data, we established an international intercomparison in 2015 to evaluate sources of variability in PUF-PAS-generated data. Institutes from 12 countries (Australia, Brazil, Canada, China, Colombia, Czechia, Germany, Mexico, Norway, Spain, Turkey, UK) participated in the intercomparison, covering many of the major research groups using PUF-PAS and including most of the monitoring networks/laboratories that have reported PUF-PAS data to the Stockholm Convention GMP Data Warehouse for the 3rd Global Monitoring Report on Persistent Organic Pollutants. We note that such an exercise is only possible due to the simplicity and small size of the PUF-PAS samplers, whereas a similar effort for active air samplers would not be feasible for logistical reasons. The PUF-PAS intercomparison consisted of three phases to address the following questions:

- o what is the variability introduced by differences in PUF-PAS sampler designs and deployment practices? (Phase 1)
- o what is the variability introduced by differences in analytical methods/performance between laboratories? (Phase 2)
- o what is the overall variability/comparability between PUF-PAS-derived air concentrations for POPs from different programs/laboratories? (Phase 3)

This study evaluates the variability in SVOC measurements across these three phases due to differences in sampler design and laboratory performance to assess the comparability of reported SVOC monitoring data from PUF-PAS across the globe.

2. Methods

Laboratory groups known to routinely use PUF-PAS to quantify SVOCs in air were contacted and invited to join the study. In all, the study included 15 participating research institutes (Table S1) using 12 different PUF-PAS sampler designs (Fig. 1). The institutes supplied their own PUF disks and PAS housings. PUF-PAS designs differed by housing dimensions (dome shape, internal volume, overhang, gap diameter) and/or type of PUF disk (details of the individual designs are given in Table S2). All equipment was kept separated under strict regimes. The PUF disks and housings were pre-cleaned at the Norwegian Institute for Air Research (NILU) before deployment in each phase of the study. All samples in the intercomparison study were deployed at the same site, located in Kjeller, outside Oslo, Norway. The site is semi-rural near grass fields, with a mix of residential, and office buildings at a short distance. Meteorological parameters corresponding with the deployment periods of each study phase are given in Tables S3 and S4. The study was divided into three phases, each addressing a key aspect of monitoring data comparability, as described below.

2.1. Phase 1 – different samplers, same PUFs, same laboratory

The objective of Phase 1 was to isolate and identify the specific influence of different PAS housings on sampler uptake. For Phase 1, 16 passive air samplers (consisting of 12 different designs) were collected from the 15 participating laboratories. Each laboratory's PAS design (different housing and installation parameters, but with identical PUF disks) were deployed simultaneously at the field site in Kjeller, Norway for 80 days from April 1, 2016 to June 13, 2016. Samplers were deployed along a 50 m section of a wire fence at a height of 2 m (Fig S2). This therefore addressed differences in both sampler design and installation parameters, including fixed rigid installations for some samplers and free-swinging installations for others, following the method of the participating laboratory. The PUF disks had a density of 2.70×10^4 g/m^3, mass of 5.9 g, diameter of 14.1 cm, and thickness of 1.4 cm. Average daily ambient temperature during Phase 1 deployment was 9.2 °C (range −4.6 to +28.5 °C) and average wind speed was 2.8 m/s (range 1.3–5.6 m/s) (Table S3). After 80 days, the PUF-PAS were collected, and PUF disks and three field blanks were packed individually in pre-cleaned aluminum foil and shipped to the Trace Analytical Laboratories of RECETOX, Czechia for SVOC analysis.

All PUF disks were analyzed according to accredited analytical methods (ČSN EN ISO 17025: 2018) for 8 PCB congeners (7 indicator PCBs + PCB 11), 12 OCPs (chlorobenzenes, hexachlorocyclohexanes - HCHs, dichlorodiphenyltrichloroethane and associated metabolites - DDX compounds), 29 PAHs and 10 PBDE congeners; compounds are

Fig. 1. Sampler designs used in the study and their basic dimensions. V indicates volume, G indicates the area of the horizontal gap between upper and lower dome, V/G is the ratio of the dome volume to gap area, and overhang is the distance the upper dome extends over the lower dome.

listed in Table S5. Full details on the analytical methods used by RECETOX can be found in Kalina et al. (2017). Recoveries were tracked using deuterated PAHs (d8-naphthalene, d10-phenanthrene, d12-perylene) and non-environmental PCBs (PCB 30, PCB 185) (Table S6). PAH, PCB and OCP masses were adjusted for recoveries based on the closest corresponding recovery standard. PBDEs were quantified by isotope dilution. Method detection limits (MDLs) were determined based on the field blanks; MDL=[avg. mass in field blanks]+3*[standard dev. of field blanks] (Table S7). If a compound was below detection in all field blanks, the instrumental detection limit was taken as the MDL. All results are reported as mass per PUF disk without conversion to air concentration.

2.2. *Phase 2 – same samplers, same PUFs, different laboratories*

The objective of Phase 2 was to identify purely analytical variability between laboratories. Fourteen identical PUF-PAS samplers (Sampler 15 from Fig. 1) were deployed at the Norwegian field site (deployment height 2 m) for 81 days from September 11, 2015 to December 1, 2015. Average daily ambient temperature during deployment was 5.9 °C (range -12.0 to +19.6 °C) and the average wind speed was 1.9 m/s (range 0–4.9 m/s) (Table S4). After 81 days each PUF disk and a corresponding field blank were collected, wrapped in pre-cleaned aluminum foil and sealed in plastic zip-top bags, packed in a padded envelope and sent to the 15 participating laboratories. Participating laboratories were asked to analyze the PUFs according to their in-house methods and report masses for seven PCB congeners, 10 OCPs, 10 PBDE congeners, and 16 PAHs to an Excel template. All results were reported as mass per PUF disk without conversion to air concentration. Details of the individual methods for each laboratory are given in Table S8. Most laboratories used Soxhlet extraction while three laboratories used accelerated solvent extraction (ASE) and one used a Büchi system. Seven different solvent combinations were used, while only three different clean-up methods were used. Not all laboratories reported all sets of compounds, resulting in data for PCBs from 11 laboratories, OCPs from 11 laboratories, PBDEs from 10 laboratories, and PAHs from 9 laboratories. Two laboratories that received PUF samples did not report any results.

2.3. *Phase 3 – different samplers, different PUFs, different laboratories*

The objective of Phase 3 was to identify the full variability in SVOC measurements due to the combined effect of different sampler designs and laboratory analyses. This reflects the “realistic” variability that would occur between different studies/monitoring networks. In this Phase, 14 laboratories sent a PAS housing and PUF disk to NILU, and each laboratory’s own PUF-PAS configuration (considering housing, PUF disk and installation parameters) was deployed at the Norwegian field site (deployment height 2 m), concurrent with Phase 2 from September 17, 2015 to December 3, 2015. After 77 days the PUF disk and a corresponding field blank were collected and shipped with the Phase 2 samples. As with Phase 2, participating laboratories were asked to analyze the PUFs for seven PCB congeners, 10 OCPs, 10 PBDE congeners and 16 PAHs (Table S5) and report results to an Excel template. Laboratories used the same analytical methods as for Phase 2 (Table S8) and reported identical sets of compounds, resulting in records for PCBs from 11 laboratories, OCPs from 11 laboratories, PBDEs from 10 laboratories, and PAHs from 9 laboratories. Two laboratories that received PUF samples did not report any results.

2.4. *Quality assurance/quality control*

Each participating laboratory reported their internal standards, instrumental detection limits, and method detection limits for Phases 2 and 3.

All PUFs were sequentially pre-cleaned by Soxhlet at NILU laboratories with 24 h toluene, 8 h acetone, 8 h hexane, and then dried under vacuum. Field blanks were included in all three phases. Each PUF disk sample sent to participating laboratories was paired with a field blank of the same PUF disk type. PUF disks were only numbered and were not separately identified as field blank or sample. All field blanks were pre-cleaned at the same laboratory (NILU), using the same method. Thus, any variability in levels in the field blanks should be due to contamination during transport or laboratory procedures.

Data received from the Excel template spreadsheets were compiled separately for each compound group. Each sampler was assigned a number code for Phase 1 data (1–16) and each laboratory was assigned a letter code for Phases 2 and 3 data (A—M) to anonymize all results. Inconsistencies or missing values in reported data were addressed individually with participating laboratories. Data handling and statistical evaluation was done through MS Excel and R software.

3. Results and discussion

Twelve different sampler designs were received, differing in dome shape, internal volume, overhang, and gap diameter. Not all domes were hemispherical (some had a straight-sided conical shape), thus individual dome volumes were measured based on the mass of water that could fill each dome. Dome gap dimensions and overhangs were measured for the assembled sampler design, and the surface areas of the gap between upper and lower dome, i.e., the main space for air diffusion into the PAS housing, were calculated assuming circular geometry. All sampler housings also allowed additional diffusion through holes in the bottom of the lower dome, although the number of holes varied from 4 to 8 depending on the sampler. Samplers and associated indicators of sampler geometry are shown in Fig. 1. Internal volumes (V) ranged from 3340 to 6750 cm^3, and surface areas of the gap between upper and lower dome (G) from 75 to 171 cm^2 (Fig. 1). Differences in the PUF disks were smaller, with diameters of 13.2–14.1 cm, thicknesses of 1.1–1.5 cm, and densities of 0.020 to 0.031 g/cm^3. Full dimensions of samplers and PUF disks are given in Table S2.

3.1. *Phase 1 – different samplers, same PUFs, same laboratory*

The objective of Phase 1 was to isolate and identify the influence of different sampler designs on sampler uptake by comparing different housings fitted with identical PUF disks, deployed simultaneously at the same site, and analyzed in a single laboratory. The differences in the masses of SVOCs sampled by each PAS should therefore give insight into the variability in uptake introduced only by sampler geometry (dome sizes, gap and overhang) and deployment (e.g., fixed vs. free swinging).

Compounds that were below detection limits in 50 % or more of the PAS were excluded from further interpretation. This resulted in the exclusion of eight of 29 PAHs (naphthalene, biphenyl, acenaphthylene, cyclopenta(cd)pyrene, perylene, dibenzo(ah)anthracene, dibenzo(ac) anthracene, and anthanthrene), five of 12 OCPs (β- and δ-HCH, and o,p’-DDD, p,p’-DDD and o,p’-DDE), two of seven PCBs (PCB-138, PCB-180), and six of ten PBDEs (BDE-66, BDE-85, BDE-99, BDE-100, BDE-153, BDE-154). Any results <MDL for the remaining compounds were substituted with 0.5*MDL for statistical analysis. For PAHs and PCBs, this resulted in substitution of 3% of records, and no substitutions for OCPs; for PBDEs the substitution was required for 37 % of records, thus there is higher uncertainty in the statistical analysis of the PBDEs than for the other compound groups.

We evaluated the variability in the individual samplers by normalizing masses measured per compound to the median masses for all samplers (Fig. 2a). Compounds were mostly within a relatively narrow range, spanning 519–939 ng/PUF for Σ_{21}PAHs (Table S9), 4.86–12.3 ng/PUF for Σ_6PCBs (Table S10), 0.80–2.21 for Σ_6DDXs (Table S11), 2.54–5.50 ng/PUF for Σ_4HCHs (Table S11), and 59.5–109 pg/PUF for Σ_9PBDEs (excluding BDE-209; range for BDE-209 was < MDL–1300 pg/PUF) (Table S12). Individual compound masses were within one order of magnitude of the median for all of the sampler designs, with the exception of one record of BDE-209 (Fig. 2a). In general, the variability increased with increasing molecular weight of compounds, e.g., high molecular weight PAHs and PBDEs had the highest variability.

Five of the samplers (Samplers 1–5) were identical configurations of the commercially available Tisch sampler design (TE-200, Tisch Environmental, Cleves, OH). We considered these samplers as replicates and used them to assess the typical range of variability between identical co-deployed samplers (i.e., due to environmental conditions and laboratory uncertainty rather than differences in sampler configuration). These Tisch replicates were normally distributed (Shapiro-Wilks test, α = 0.05, except for p,p’-DDE, and BDE-47 and 209), so means and standard deviations were used to evaluate their distribution. The relative standard deviation (RSD) of the five replicates ranged from 1.98 % for acenaphthene to 55.6 % for BDE-183 (Table S13). The highest RSDs were observed for the higher molecular weight compounds, i.e. 5- and 7-ring PAHs and BDEs 183 and 209 (average 22.7 % vs. 8.9 % for all other compounds), which likely reflects two possible effects on higher molecular weight compounds: (1) higher analytical uncertainty and (2) variable sampler uptake of particulates. Lower ambient levels of higher molecular weight compounds may lead to greater measurement uncertainties as MDLs are approached. The larger variability in the uptake of particle-bound compounds to PUF-PAS has been extensively discussed in other publications (e.g., Holt et al., 2017; Markovic et al., 2015). However, we note that the Tisch sampler is reported as having high particle infiltration in Markovic et al. (2015), suggesting that even when particle infiltration is high, there remains higher variability/uncertainty for particle uptake than gaseous compounds.

The RSD determined from the Tisch samplers was assumed to represent the typical uncertainty in a sampler due to environmental variability and laboratory uncertainty and was therefore used to flag cases when the variability between PAS was beyond the range of typical differences between identical samplers. Upper and lower boundaries were calculated per compound as the median of all sampler masses ±3 times the percent uncertainty (Tables S9-S12). We identify the specific cases and compounds where variability exceeded these thresholds.

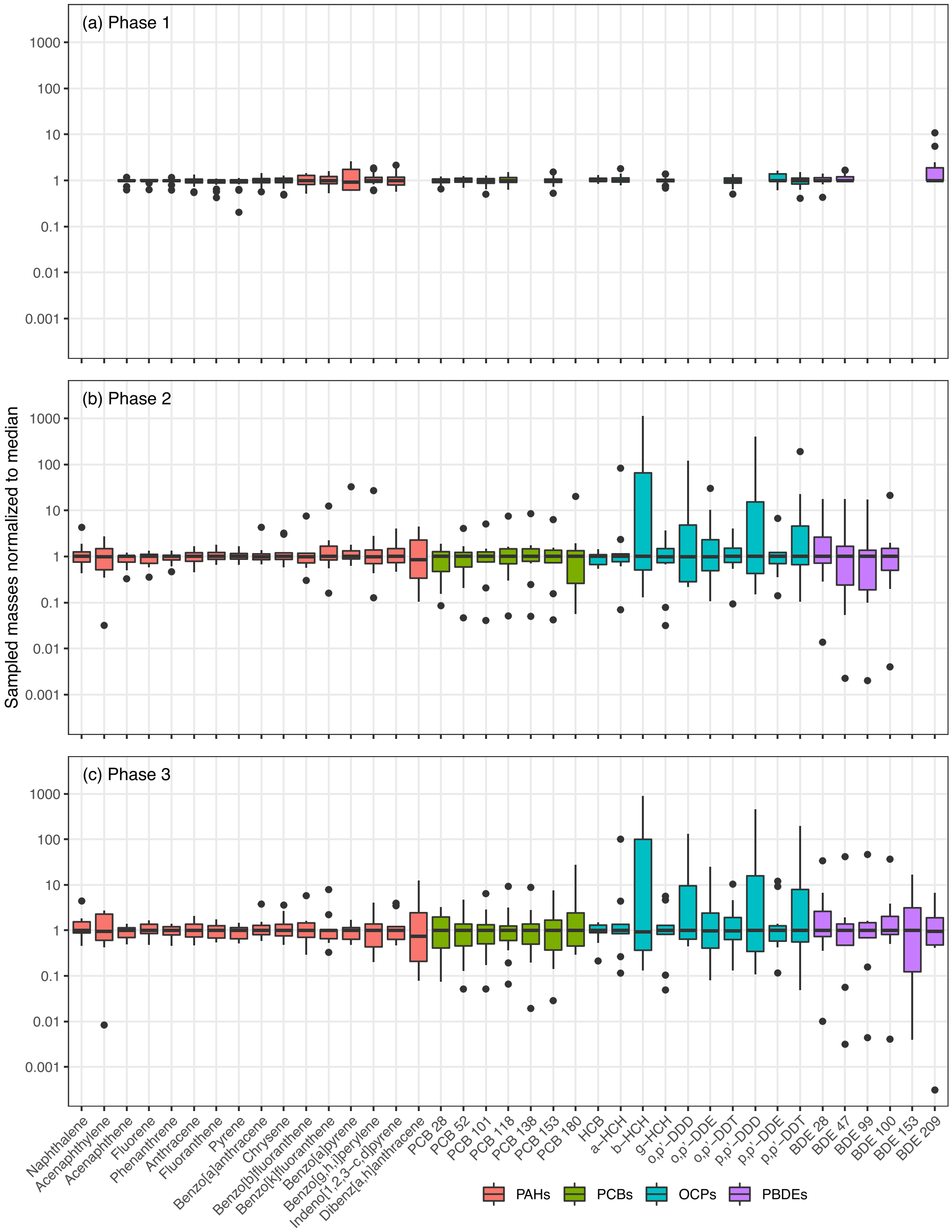

Fig. 2. Variability in analytes detected by PUF-PAS due to differences in (a) sampler geometry and installation (Phase 1, n = 16), (b) analytical methods between laboratories (Phase 2, n = 13), and (c) combined sampler and analytical differences (Phase 3, n = 13). Boxes represent the 25th to 75th percentiles, with the median (50th percentile) as a horizontal black line. Individual SVOC levels were normalized to the median. Whiskers represent ±1.5 times the interquartile range (IQR) with individual points indicating outliers.

No single sampler design had all results within the acceptable range (median±3xRSD of Tisch samplers), however most samplers had only a few compounds outside of this range (Tables S9-S12), with the Tisch samplers (Samplers 1–5) demonstrating the least variability across all compound groups (Fig. 3a). A few samplers had more substantial deviations from the median values: Sampler 6 measured 24 % lower total SVOC masses than the median, with OCPs particularly low; Sampler 13 recorded 38 % lower total SVOC masses, with OCPs and PCBs particularly low (Fig. 3a); Sampler 8 recorded 11 % higher total SVOC masses (Fig. 3a). BDE-209 was substantially more variable than other compounds, with most cases < MDL and five samplers with notably high values.

We also examined the correlation between the measured masses and the sampler geometry parameters. The strongest correlation was with the overhang distance (i.e. the overlap between the top and bottom domes), with a significant negative relationship between overhang distance and mass of SVOCs collected by the PUF (Spearman ρ of -0.796, $p < 0.01$ for correlation between overhang distance and total SVOCs), e.g., larger overhang is correlated with lower mass. This suggests reduced airflow to the inside of the sampler housing due to overhang, and consequent lower uptake of compounds, especially particle-bound compounds (Markovic et al., 2015). Despite large ranges in the other sampler geometries (e.g., sampler volumes ranging from 3340 to 6750 cm^3), no other sampler geometry parameters had significant correlations with the mass of SVOCs collected.

While individual differences were generally small, small systematic errors across many individual compounds can lead to significant differences in compound group totals, e.g., indicators such as ΣEPA-16 PAHs and Σ_7PCBs, which are frequently applied in policy and effectiveness evaluation. For example, ΣEPA-16 PAHs is 482 ng/PUF in the lowest reporting sampler (Sampler 13) vs. 873 ng/PUF in the highest (Sampler 10), and for Σ_7PCB the mass collected was 2440 pg/PUF vs. 5120 pg/PUF in Samplers 13 vs. 8. While these are the extremes of the set, it does suggest that uncertainties introduced by differences in sampler configuration can account for up to 50 % variation in reported sample masses. Moreover, it is known that at sites with meteorological

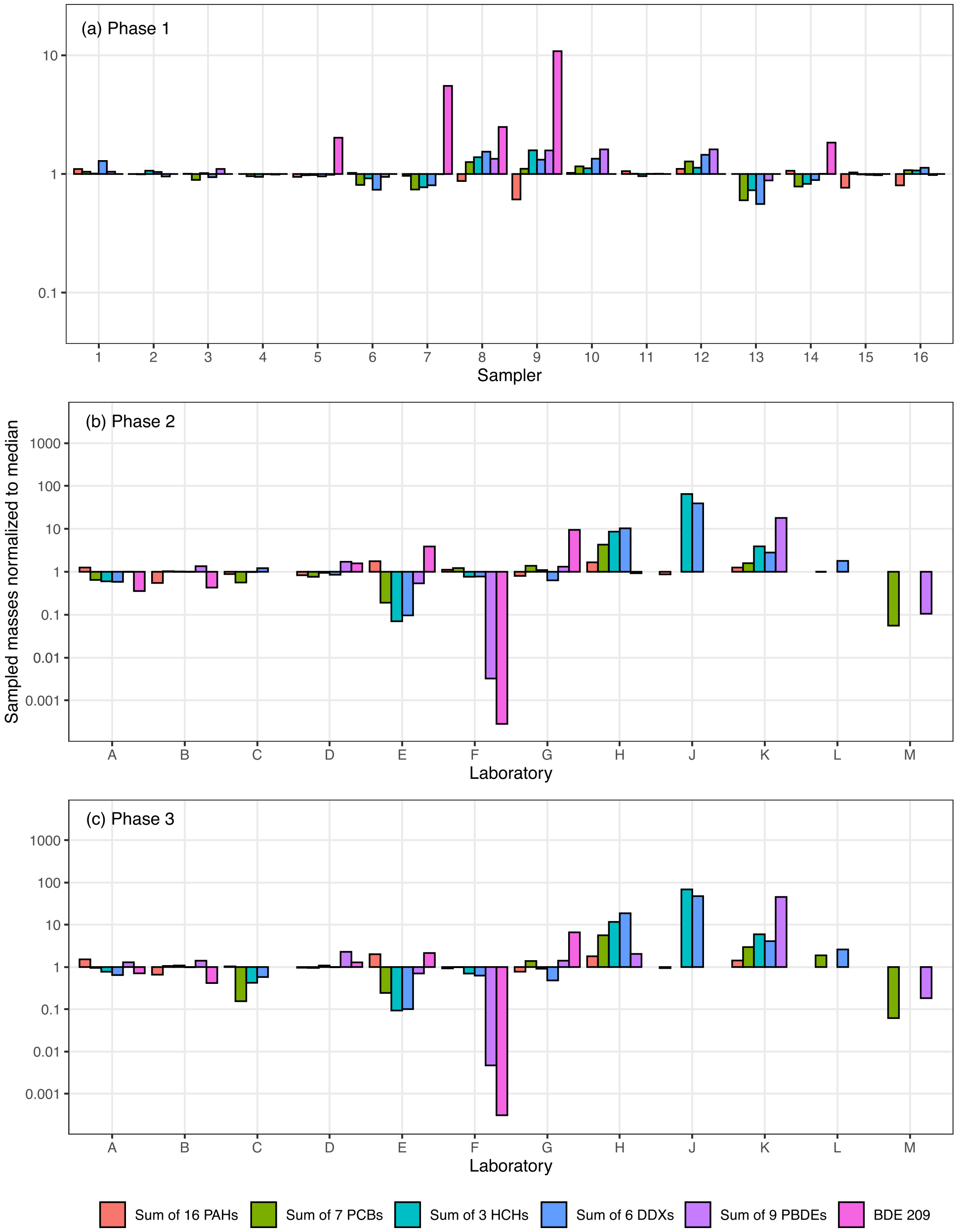

Fig. 3. Reported SVOC masses normalized to median for (a) Phase 1, (b) Phase 2, and (c) Phase 3. Phase 1 shows variability between different PUF-PAS sampler designs (1–16) analyzed at a single laboratory; Phase 2 shows variability between co-deployed, identical PUF-PAS samplers analyzed in 13 different laboratories (A–M); and Phase 3 shows differences in co-deployed different samplers analyzed in different laboratories (Labs A-M). Samplers 1–5 in Phase 1 are identical Tisch TE-200 samplers. Note the smaller y-axis scale in Phase 1 compared to Phases 2 and 3.

extremes (e.g., coastal sites with very high wind speeds, polar sites) the protective effect of the sampler housing on PAS uptake rates is variable, and the effects of differences between sampler configurations could be exacerbated, leading to variation beyond what was seen in this test case.

3.2. *Phase 2 – same samplers, same PUFs, different laboratories*

Results for Phase 2 indicate the extent of variability between SVOC measurements introduced by differences in transport, laboratory handling and analysis of PUF disks. Major differences were identified in two aspects: field blank contamination and sample masses.

Field blank masses varied over 2 orders of magnitude, spanning 187–8260 pg/PUF for Σ_7PCBs, <3–640 pg/PUF for HCB, 50–35600 pg/PUF for Σ_3HCHs, <50–98400 pg/PUF for Σ_6DDXs, 17.9–1670 ng/PUF for Σ_{16}PAHs, and <25–3250 pg/PUF for Σ_{10}PBDEs (Table S14). The 2–3 order of magnitude range in field blanks suggests either errors in analysis and/or large variations in contamination during transport and processing, with the implication being that choices regarding blank treatment can have a large impact on reported values. We also note that in three cases, masses of PBDEs and DDXs reported in field blanks exceeded those reported in the samples (Fig. S2). PBDEs and other flame retardants are often identified as a particular challenge for analysis in PUF samples due to their prevalence in equipment and electronics, and low ambient levels at many locations, including the NILU site in this study.

Despite the significant contribution of blanks to total samples for a few laboratories (laboratories A, J, K; Fig. S1), we did not further adjust samples for blanks as we had only one field blank per laboratory, and in practice many laboratories use blanks only as a quality control in long-term monitoring, rather than for data adjustment.

Phase 2 sample results clearly indicate that the variability due to laboratory analysis is much higher than that introduced by sampler geometry identified in Phase 1. The Σ_{16}PAH reported spanned 1830–5870 ng/PUF, a substantially larger span than that observed in Phase 1 (Table S15). Variations were even higher for POPs, with reported values spanning 3 or more orders of magnitude: Σ_3HCHs spanning 283–263000 pg/PUF, Σ_6DDX from 171 to 70100 pg/PUF (Table S16), Σ_7PCBs from 378 to 29300 pg/PUF (Table S17) and Σ_{10}PBDEs from 0.35 to 1950 pg/PUF (Table S18).

We explored whether the choice not to adjust sample masses for field blank contamination led to such large ranges, but the effect was limited. For example, when blanks were subtracted from measured values Σ_7PCBs ranged 378–23000 pg/PUF, Σ_3HCHs ranged 243–227000 pg/PUF, Σ_6DDX ranged 140–14200 pg/PUF, and the range for PBDEs did not change, suggesting that the large variability in their reported masses is independent of differences in blank contamination.

As with Phase 1, we assessed the variability by normalizing sampled SVOC masses to the median of the whole set of reported masses (Fig. 2b). It is clear from the range of the normalized data, with some compounds covering a range of ~0.1–1000 around the median, that much greater differences in reported masses are introduced by laboratory analysis than by differences in sampler design (where the range of normalized masses was much less than 0.1–10 times the median, Fig. 2a). We note that some of the compounds with the largest ranges in Phase 2 (e.g., PCB 180, dibenzo[a,h]anthracene, β-HCH) were below detection in Phase 1 in all samples, which inherently suggests that the variability for these compounds is smaller than the LOD. Of the compounds detected in both phases, all compounds except chrysene had at least a 2x higher span in reported values in Phase 2 compared with Phase 1, and the span of values was more than 50x higher for BDE-28, BDE-47 and p,p'-DDT.

This substantial increase in variability in Phase 2 compared with Phase 1 indicates that, due to the uncertainty introduced by analysis in different laboratories, close to 50 % of laboratories report SVOC masses with order of magnitude differences from a consensus values, i.e., median of all laboratories (Fig. 3b). We compared the deviations from consensus values with the analytical methods used across the laboratories (Table S8) but did not find any relationship with the basic parameters of extraction method, solvent, clean-up or instrumental analysis and the laboratory performance. However, it is clear from the difference between Phases 1 and 2 that the sample processing and analysis are an important contributor to the differences in the reported values between laboratories. We are unable to identify specific method influences on performance due to the large differences in methods used across the 12 laboratories (e.g., eight different extraction methods), which prevents us from making generalizations.

The differences introduced by analytical methods are higher than what has been typically identified by global interlaboratory evaluations on POPs, e.g., the UNEP-supported Bi-ennial Global Interlaboratory Assessments on POPs (Nilsson et al., 2014; van Bavel et al., 2012; van der Veen et al., 2017; Van Leeuwen et al., 2013). However, the Bi-ennial Global Interlaboratory Assessments on POPs included injection-ready test mixtures and environmental matrices high in organic matter or lipid content, or else spiked air samples (Fiedler et al., 2020, 2017; Nilsson et al., 2014), with relatively high concentrations. These concentrations were higher than what are found in typical rural/remote air samples, and many interlaboratory evaluations have reported lower precision and accuracy at lower concentrations (Melymuk et al., 2018, 2015; Su and Hung, 2010). The low concentrations in our samples collected from semi-rural Norway likely contributed to the poorer performance in our study. Further, our study captured the full scope of variability in sample processing, as laboratories received the PUF material, and were required to extract and purify the samples. This contrasts with the Bi-ennial Global Interlaboratory Assessments on POPs, where laboratories analyzed air extracts not requiring further clean-up (van der Veen et al., 2017); the variability introduced from extraction and clean-up of complex matrices is a large contributor to the differences between laboratories (Abalos et al., 2013; Melymuk et al., 2018; Su and Hung, 2010; Van Leeuwen et al., 2013).

If these differences reflect a recurring deviation by certain laboratories, this suggests that international comparisons of SVOC data could be highly biased when analyses are performed at different laboratories. This does not impact internal reporting within individual laboratories, therefore comparability within individual laboratory research studies and monitoring programs, (e.g., to assess regional spatial differences or temporal changes), should still be valid. Yet when comparisons involve merging data from multiple studies/monitoring networks, (e.g., for model comparisons), there may be significant biases introduced in the interpretation on a global level. This may also apply to data reported from active samplers, which typically follow similar methods of laboratory analysis.

3.3. *Phase 3 – different samplers, different PUFs, different laboratories*

Phase 3 gives an indication of overall variability, encompassing differences between sampler configurations identified in Phase 1 and between laboratory analysis identified in Phase 2. As with Phase 2, large differences were observed in both field blanks and reported sample masses. Field blanks had a similar distribution and range to those of Phase 2 (Table S19) and in two cases the blank mass exceeded that of the sample (Fig. S3). As in Phase 2, no blank adjustment was applied to the reported sample masses.

Phase 3 showed variability in sample masses comparable to that of Phase 2. The Σ_{16}PAH ranged 1640–5010 ng/PUF (Table S20), a similar range to that observed in Phase 2. As in Phase 2, variations were higher for POPs, with ranges spanning >3 orders of magnitude: from 247 to 182000 pg/PUF for ΣHCH, 123–58200 pg/PUF for ΣDDX (Table S21), 283–25900 pg/PUF for Σ_7PCBs (Table S22), and 0.44–3110 pg/PUF for ΣPBDEs (Table S23). As with Phases 1 and 2, we assessed the variability by normalizing masses to the median of the whole set of reported masses (Fig. 2c). The range of the normalized data, covering ~0.001–1000 around the median, was similar to Phase 2 (Fig. 2b), and much greater than Phase 1 (Fig. 2a).

Phase 3 combined three factors affecting variability (1) sampler design, (2) sample transport and laboratory analysis, and (3) differences in PUF disk parameters (size, density; Table S2). While we did not have a separate study phase to isolate the influence of differences in PUF disks, the similarity of results from Phases 2 and 3 suggest that this is minimal, and likely comparable to or less than the variability introduced by sampler design. This is supported by previous work identifying 40–60 % differences in particle uptake and distribution in high density vs. low density PUF disks (Chaemfa et al., 2009), which was a larger range in density than that seen within our study.

As in Phase 2, a large fraction of laboratories reported masses that differed by more than one order of magnitude from the median (Fig. 3c), suggesting that, when PUF-PAS data are reported by different laboratories using different samplers, ~50 % of laboratories are reporting values outside the boundaries of acceptable uncertainty, which can create a major challenge in the global comparability of SVOC data from PUF-PAS monitoring.

The similarity in the ranges and collected masses between Phases 2 and 3 suggests the major influence on differences in comparability between laboratories is sample transport, processing and analytical variation, and this source of variability overwhelms any smaller differences due to sampler design identified in Phase 1.

4. Conclusions and implications

Phase 1 of this international intercomparison revealed that variations in the double-dome PAS housings used by different research groups contributes relatively little to uncertainties in sampled masses of PCBs, PAHs, PBDEs, and OCPs for a 3-month deployment, with differences in reported masses due to PUF-PAS sampler configurations not exceeding 50 %. Any difference in uptake between samplers appears related to differences in airflow into the sampler housing due to the amount of overhang of the upper dome over the lower.

Phase 2 of the study, which assessed uncertainty associated with laboratory performance (i.e., each laboratory analyzing the same type of sample), showed a substantial increase in uncertainty. Reported masses varied by an order of magnitude or greater, with ~10× differences between individual masses and medians for many compounds, as well as a few more extreme outliers (Fig. 2). Phase 3 of the study confirmed the results of Phase 2; similar ranges of reported values and large deviations from medians for some individual laboratories/compounds show that the major influence on comparability between laboratories is sample processing and analytical variation, and this source of variability overwhelms any smaller differences due to sampler design and differences in PUF disk size and density. Considering only the three laboratories that currently report PUF-PAS data to the Stockholm Convention GMP, the variability is lower, but there still exist order-of-magnitude differences in reported masses of Σ_7PCBs, HCB and Σ_6DDX.

The high level of uncertainty observed in Phases 2 and 3 of this study indicates that current global air monitoring data are not directly comparable between different laboratories/monitoring programs for most of the SVOCs included in this study. However, this does not mean that the data are not internally consistent (i.e., within a program using a single laboratory) for deriving valid spatial and temporal trends. Yet on a global scale, it is clear that merging data from multiple laboratories must be done with caution. With current levels of uncertainty, it is not feasible to compare results between laboratories/monitoring programs without prior assurance of comparability in reported data. It is also clear that the uncertainty is not limited to passive sampling but also pertains to active air sampling results if similar extraction, clean-up and analytical procedures are followed. This uncertainty may be due to several factors including instrumental methods, sample processing procedures, potential laboratory contamination due to solvents and other sources, and differences in analytical standards (not assessed in the current study). Additional uncertainties will also arise if concentrations are adjusted to volumetric units (e.g., pg/m^3), since slightly different conventions may be used among laboratories for estimating effective air sample volumes. These uncertainties can be resolved through ongoing participation in intercalibration exercises and adoption of best practices. The high variability between laboratories means that a crucial part of any efforts to integrate and evaluate global spatial patterns of POPs in air must require and implement intercalibration to assess and account for uncertainties, repeated at regular intervals for both active and passive air sampling. Some examples of such international repeated intercalibration exercises already exist, most notably the AMAP/EMEP intercomparison exercises (Schlabach et al., 2012; Tkatcheva et al., 2013). Establishing such actions as a part of research infrastructure is even more necessary given the efforts to mobilize data from additional monitoring networks to contribute to global data repositories.

Effectiveness evaluation of international SVOC actions (Stockholm Convention, CLRTAP) relies on the provision of high-quality air monitoring data. PUF-PAS are a valuable tool to provide this information, particularly given that the slight differences in PUF-PAS sampler designs do not greatly affect data comparability. Yet without establishment of frameworks to ensure analytical comparability, our understanding of global spatial POP distributions will be biased by the regional differences in analytical performance.

CRediT authorship contribution statement

Lisa Melymuk – Conceptualization, Methodology, Writing, Formal Analysis; **Pernilla Bohlin Nizzetto–** Conceptualization, Methodology, Resources, Review & Editing; **Tom Harner** – Conceptualization, Resources, Review & Editing, Supervision; **Kevin B. White** – Formal Analysis, Visualization, Review & Editing; **Xianyu Wang** – Investigation, Resources, Review & Editing; **Maria Yumiko Tominaga** – Investigation, Resources, Review & Editing; **Jun He** – Investigation, Resources, Review & Editing; **Jun Li –** Investigation, Resources, Review & Editing; **Jianmin Ma** – Investigation, Resources, Review & Editing; **Wan-Li Ma** – Investigation, Resources, Review & Editing; **Beatriz H. Aristizábal** – Investigation, Resources, Review & Editing; **Annekatrin Dreyer** – Investigation, Resources, Review & Editing; **Begoña Jiménez** – Investigation, Resources, Review & Editing; **Juan Muñoz-Arnanz** – Investigation, Resources, Review & Editing; **Mustafa Odabasi** – Investigation, Resources, Review & Editing; **Yetkin Dumanoglu** – Investigation, Resources, Review & Editing; **Baris Yaman** – Investigation, Resources, Review & Editing; **Carola Graf** – Investigation, Resources, Review & Editing; **Andrew Sweetman –** Investigation, Resources, Review & Editing; **Jana Klánová** – Resources, Supervision, Review & Editing.

Declaration of Competing Interest

The authors report no declarations of interest.

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Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:https://doi.org/10.1016/j.envsci.2021.08.003.

References

Abalos, M., Abad, E., Van Leeuwen, S.P.J., De Boer, J., Lindström, G., Van Bavel, B., Fiedler, H., 2013. Results for PCDD/PCDF and dl-PCBs in the first round of UNEPs biennial global interlaboratory assessment on persistent organic pollutants. TrAC - Trends Anal. Chem. 46, 98–109. https://doi.org/10.1016/j.trac.2012.11.003.

Borůvková, J., Gregor, J., Šebková, K., Bednářová, Z., Kalina, J., Hůlek, R., Dušek, L., Holoubek, I., Klánová, J., 2015. GENASIS – Global Environmental Assessment and Information System [Online] [WWW Document]. Masaryk Univ., 2015 [cit. 2017-06-19]. Available http//www.genasis.cz. Version 2.0. ISSN 1805-3181.

Chaemfa, C., Barber, J.L., Gocht, T., Harner, T., Holoubek, I., Klánová, J., Jones, K.C., 2008. Field calibration of polyurethane foam (PUF) disk passive air samplers for PCBs and OC pesticides. Environ. Pollut. 156, 1290–1297.

Chaemfa, C., Wild, E., Davison, B., Barber, J.L., Jones, K.C., 2009. A study of aerosol entrapment and the influence of wind speed, chamber design and foam density on polyurethane foam passive air samplers used for persistent organic pollutants. J. Environ. Monit. 11, 1135–1139.

Fiedler, H., van der Veen, I., De Boer, J., 2017. Bi-ennial Global Interlaboratory Assessment on Persistent Organic Pollutants – Third Round 2016/2017. Geneva.

Fiedler, H., van der Veen, I., de Boer, J., 2020. Global interlaboratory assessments of perfluoroalkyl substances under the Stockholm Convention on persistent organic pollutants. TrAC - Trends Anal. Chem. 124, 115459 https://doi.org/10.1016/j.trac.2019.03.023.

Holt, E., Bohlin-Nizzetto, P., Borůvková, J., Harner, T., Kalina, J., Melymuk, L., Klánová, J., 2017. Using long-term air monitoring of semi-volatile organic compounds to evaluate the uncertainty in polyurethane-disk passive sampler-derived air concentrations. Environ. Pollut. 220, 1100–1111. https://doi.org/10.1016/j.envpol.2016.11.030.

Kalina, J., Scheringer, M., Borůvková, J., Kukučka, P., Přibylová, P., Bohlin-Nizzetto, P., Klánová, J., 2017. Passive air samplers as a tool for assessing long-term trends in atmospheric concentrations of semivolatile organic compounds. Environ. Sci. Technol. 51, 7047–7054. https://doi.org/10.1021/acs.est.7b02319.

Klánová, J., Harner, T., 2013. The challenge of producing reliable results under highly variable conditions and the role of passive air samplers in the Global Monitoring Plan. Trends Analyt. Chem. 46, 139–149.

Lee, S.C., Harner, T., Pozo, K., Shoeib, M., Wania, F., Muir, D.C.G., Barrie, L.A., Jones, K. C., 2007. Polychlorinated naphthalenes in the Global Atmospheric Passive sampling (GAPS) study. Environ. Sci. Technol. 41, 2680–2687.

Markovic, M.Z., Prokop, S., Staebler, R.M., Liggio, J., Harner, T., 2015. Evaluation of the particle infiltration efficiency of three passive samplers and the PS-1 active air sampler. Atmos. Environ. 112, 289–293.

Melymuk, L., Goosey, E., Riddell, N., Diamond, M.L., 2015. Interlaboratory study of novel halogenated flame retardants: INTERFLAB. Anal. Bioanal. Chem. 407, 6759–6769. https://doi.org/10.1007/s00216-015-8843-7.

Melymuk, L., Diamond, M.L., Riddell, N., Wan, Y., Vojta, Š., Chittim, B., 2018. Challenges in the analysis of novel flame retardants in indoor dust: results of the INTERFLAB 2 interlaboratory evaluation. Environ. Sci. Technol. 52, 9295–9303. https://doi.org/10.1021/acs.est.8b02715.

Muñoz-Arnanz, J., Roscales, J.L., Ros, M., Vicente, A., Jiménez, B., 2016. Towards the implementation of the Stockholm Convention in Spain: five-year monitoring (2008–2013) of POPs in air based on passive sampling. Environ. Pollut. 217, 107–113. https://doi.org/10.1016/j.envpol.2016.01.052.

Muñoz-Arnanz, J., Roscales, J.L., Vicente, A., Ros, M., Barrios, L., Morales, L., Abad, E., Jiménez, B., 2018. Assessment of POPs in air from Spain using passive sampling from 2008 to 2015. Part II: Spatial and temporal observations of PCDD/Fs and dl-PCBs. Sci. Total Environ. 634, 1669–1679. https://doi.org/10.1016/j.scitotenv.2018.04.164.

Nilsson, H., van Bavel, B., van der Veen, I., 2014. United Nations Environmental Programme, Bi-ennial Global Interlaboratory Assessment on Persistent Organic Pollutants – Second Round 2012/2013. Geneva, Switzerland.

Pozo, K., Harner, T., Wania, F., Muir, D.C.G., Jones, K.C., Barrie, L.A., 2006. Toward a global network for persistent organic pollutants in air: results from the GAPS study. Environ. Sci. Technol. 40, 4867–4873.

Pozo, K., Harner, T., Lee, S.C., Wania, F., Muir, D.C.G., Jones, K.C., 2009. Seasonally resolved concentrations of persistent organic pollutants in the global atmosphere from the first year of the GAPS study. Environ. Sci. Technol. 43, 796–803. https://doi.org/10.1021/es802106a.

Přibylová, P., Kares, R., Borůvková, J., Cupr, P., Prokeš, R., Kohoutek, J., Holoubek, I., Klánová, J., 2012. Levels of persistent organic pollutants and polycyclic aromatic hydrocarbons in ambient air of Central and Eastern Europe. Atmos. Pollut. Res. 3, 494–505.

Rauert, C., Harner, T., Schuster, J.K., Eng, A., Fillmann, G., Castillo, L.E., Fentanes, O., Villa Ibarra, M., Miglioranza, K.S.B., Moreno Rivadeneira, I., Pozo, K., Aristizábal Zuluaga, B.H., 2018. Atmospheric concentrations of new persistent organic pollutants and emerging chemicals of concern in the group of Latin America and Caribbean (GRULAC) region. Environ. Sci. Technol. 52, 7240–7249. https://doi.org/10.1021/acs.est.8b00995.

Roscales, J.L., Muñoz-Arnanz, J., Ros, M., Vicente, A., Barrios, L., Jiménez, B., 2018a. Assessment of POPs in air from Spain using passive sampling from 2008 to 2015. Part I: Spatial and temporal observations of PBDEs. Sci. Total Environ. 634, 1657–1668. https://doi.org/10.1016/j.scitotenv.2018.03.043.

Roscales, J.L., Muñoz-Arnanz, J., Ros, M., Vicente, A., Jiménez, B., 2018b. Does the number of field blanks influence reported air POP concentrations in monitoring programs based on PUF-PAS? Organohalogen Compd. 80, 509–512.

Schlabach, M., Farag-clement, R., Hung, H., Kallenborn, R., 2012. AMAP/EMEP/NCP Inter-laboratory Study for POP Analysis 2010. EMEP Co-operative Programme for Monitoring and Evaluation of the Long-range Transmission of Air Pollutants in Europe.

Shoeib, M., Harner, T., 2002. Characterization and comparison of three passive air samplers for persistent organic pollutants. Environ. Sci. Technol. 36, 4142–4151.

Su, Y., Hung, H., 2010. Inter-laboratory comparison study on measuring semi-volatile organic chemicals in standards and air samples. Environ. Pollut. 158, 3365–3371. https://doi.org/10.1016/j.envpol.2010.07.041.

Su, Y., Hung, H., Stern, G., Sverko, E., Lao, R., Barresi, E., Rosenberg, B., Fellin, P., Li, H., Xiao, H., 2011. Bias from two analytical laboratories involved in a long-term air monitoring program measuring organic pollutants in the Arctic: a quality assurance/quality control assessment. J. Environ. Monit. 13, 3111–3118. https://doi.org/10.1039/c1em10513b.

Tkatcheva, V., Ali, B., Reiner, E.J., 2013. Northern Contaminants Interlaboratory Quality Assurance Program (NCP III – Phase 7) [WWW Document]. URL. www.aadnc-aandc.gc.ca/eng/1100100035611/1100100035612.

UNEP, 2009. Stockholm Convention Global Monitoring Plan for Persistent Organic Pollutants: First Global Monitoring Report. Geneva.

van Bavel, B., van Leeuwen, S., de Boer, J., 2012. Bi-ennial Global Interlaboratory Assessment on Persistent Organic Pollutants – First Round 2010/2011. Geneva, Switzerland.

van der Veen, I., Fiedler, H., de Boer, J., 2017. Bi-ennial Global Interlaboratory Assessment on Persistent Organic Pollutants – Third Round 2016/2017. Organohalogen Compd. 79, 575–578.

Van Leeuwen, S.P.J., De Boer, J., Van Leeuwen, S.P.J., Van Bavel, B., 2013. First worldwide UNEP interlaboratory study on persistent organic pollutants (POPs), with data on polychlorinated biphenyls and organochlorine pesticides. TrAC - Trends Anal. Chem. 46, 110–117. https://doi.org/10.1016/j.trac.2012.12.020.

Wania, F., Shunthirasingham, C., 2020. Passive air sampling for semi-volatile organic chemicals. Environ. Sci. Process. Impacts 22, 1925–2002. https://doi.org/10.1039/D0EM00194E.

White, K.B., Kalina, J., Scheringer, M., Přibylová, P., Kukučka, P., Kohoutek, J., Prokeš, R., Klánová, J., 2021. Temporal trends of persistent organic pollutants across Africa after a decade of MONET passive air sampling. Environ. Sci. Technol. 55 (14), 9413–9424. https://doi.org/10.1021/acs.est.0c03575.

APPENDIX 19

Jílková, Simona, Lisa Melymuk, Šimon Vojta, Martina Vykoukalová, Pernilla Bohlin-Nizzetto, and Jana Klánová. 2018. "Small-Scale Spatial Variability of Flame Retardants in Indoor Dust and Implications for Dust Sampling." *Chemosphere* 206: 132-41. https://doi.org/10.1016/J.CHEMOSPHERE.2018.04.146

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Small-scale spatial variability of flame retardants in indoor dust and implications for dust sampling

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HIGHLIGHTS

- Flame retardant levels in indoor dust varied significantly between and within rooms.
- Up to 1000-fold differences exist in flame retardant levels within the same room.
- Levels of hexabromobenzene were elevated in computer room dust.
- Composite dust samples are recommended due to within-room spatial variability.

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ABSTRACT

Indoor dust is often used to evaluate levels of organic compounds indoors, particularly for compounds with indoor sources, such as flame retardants (FRs). Yet there are uncertainties about the type of information that can be obtained from indoor dust. This study reports detailed dust sampling to assess spatial variability in indoor dust concentrations, the relationship between FR sources and dust, and the implications when interpreting dust concentrations. Multiple dust samples were collected from a range of surface types in three large rooms: a residential flat, a university seminar room, and a university computer room. Samples were analysed for polybrominated diphenyl ethers (PBDEs), novel halogenated flame retardants (NFRs) and organophosphate esters (OPEs).

FR levels in dust varied significantly between and within rooms. Levels typically ranged over one order of magnitude within a room, and up to four orders of magnitude for a few OPEs. The spatial distribution of FRs related (in some cases) to proximity to sources, surface properties, and dust surface loadings. Differences also existed between surface and floor dusts, e.g., the contribution of TBOEP to $\sum$OPEs was higher in floor than surface dust, which has implications for human exposure assessment; adults typically have more contact with elevated surfaces, while young children have greater contact with floor surfaces. Overall, significant spatial heterogeneity exists in indoor dust, even in seemingly homogeneous indoor spaces, thus hampering comparability between studies and locations when single samples are collected. Composite samples are strongly recommended to limit the influence of spatial heterogeneity.

1. Introduction

Flame retardants (FRs) are chemicals used in electronics, furniture, and building materials to reduce flammability in order to meet fire safety regulations (de Wit, 2002). Two types of organic FRs are commonly used: (1) halogenated flame retardants (HFRs) and (2) organophosphate esters (OPEs). HFRs consist of brominated and chlorinated flame retardants (Bergman et al., 2012), including polybrominated diphenyl ethers (PBDEs) and so-called “novel” halogenated flame retardants (NFRs) used as replacements for banned PBDEs (Betts, 2008). OPEs are used as both FRs and plasticizers and, as with the NFRs, OPE production also increased after restrictions on PBDEs (van der Veen and de Boer, 2012).

Due to the nature of FR use, high concentrations are found in

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indoor environments, and differences in indoor levels have been noted between countries (Harrad et al., 2008b; Sjödin et al., 2008; Venier et al., 2016; Vykoukalová et al., 2017) or within countries (Karlsson et al., 2007; Stapleton et al., 2008); between types of buildings (Ali et al., 2011; Cequier et al., 2014; Harrad et al., 2008a); as well as between rooms within a building (Al-Omran and Harrad, 2018; Kuang et al., 2016; Muenhor and Harrad, 2012; Stapleton et al., 2008). Comparisons of different rooms in the same buildings have identified that measured concentrations are influenced by the presence of furniture and electronics. For example, Muenhor and Harrad (2012) suggested that where and when an indoor dust sample is taken exerts a substantial influence on the level of contamination detected, and relates to the proximity to potential sources such as electronics and carpet. Harrad et al. (2009) found decreasing concentrations of hexabromocyclododecane with increasing distance from a TV. Al-Omran and Harrad (2018) identified consistently higher FR concentrations in dust from elevated surfaces than from floors, and differences in dusts from two areas of the same rooms. Kuang et al. (2016) identified higher levels of PBDEs and NFRs in living rooms and bedrooms than in kitchens, however, this could not be directly linked to any source. Rather, it was suggested that environmental factors (e.g., higher moisture in kitchen) lead to lower dust concentrations in kitchens. Thus, a key question that arises considering all of the indoor spatial differences between and within buildings is how much of the variation is due to real differences in the indoor environments, and how much is an artifact of the choice of sampling location within a given room. Moreover, in broad indoor studies covering many locations, single dust samples are often considered representative of whole-room or even whole-building conditions, however, it is not clear whether this assumption is valid.

To address this question, the main objectives of this study were (1) to identify the differences in FR profiles and levels obtained by two sampling methods (wet wipes and vacuuming), (2) to identify the range of concentrations within individual rooms and thus identify what effect the choice of sampling location may have on reported concentrations, and (3) to determine the extent to which concentration levels are influenced by room type and proximity to room elements (electronics, furnishings, different usage of the space, etc.), and identify whether greater heterogeneity in room furnishings and use leads to greater heterogeneity in indoor dust. These objectives were addressed by detailed dust sampling (>9 samples per room) in three indoor environments.

2. Methods

2.1. Sampling strategy

Three indoor environments were chosen for sampling. The first was a residential flat, where the entrance, living room and kitchen were sampled in March 2014. The second and third were university classrooms—one classroom without computers (seminar room - SR) and one with computers (computer room - CR), sampled in July 2014. The seminar and computer rooms were the same size and shape and located in the same university building, differing only by building storey and room equipment. The classrooms are normally vacuumed 3 times/week and desks are wiped 4—5 times/week, but sampling took place after the conclusion of the academic year and there was no regular activity in the lecture room in the two weeks prior to sampling. Both locations were not vacuumed/dusted for two weeks prior to sampling.

In the flat, eight vacuum samples were collected from floors, which included the entrance area, living, dining room, kitchen, and from the sofa. The flat was not cleaned in the week prior to sample collection. Ten wipe samples were collected from horizontal surfaces, including electronics, a chair, and kitchen furnishings, and one vertical surface (i.e. windows). Samples were also collected from areas with high dust accumulation (F4, F6, and F14 in Fig. 1), i.e., places with infrequent cleaning, such as under a sofa and on an inaccessible window sill. In the classrooms, four vacuum samples were collected from the carpet and five wipe samples were collected from desks in both SR and CR. Additionally, six wipe samples were collected from monitors (all-in-one PCs) and keyboards in CR. The sampling locations are shown in Fig. 1 and more details are given in Table S1 in the Supplementary Data.

2.1.1. Sampling

Two dust sampling methods were used — wet wipe sampling for settled dust on smooth elevated surfaces and a vacuum cleaner with sock insert for floor dust and dust from the sofa.

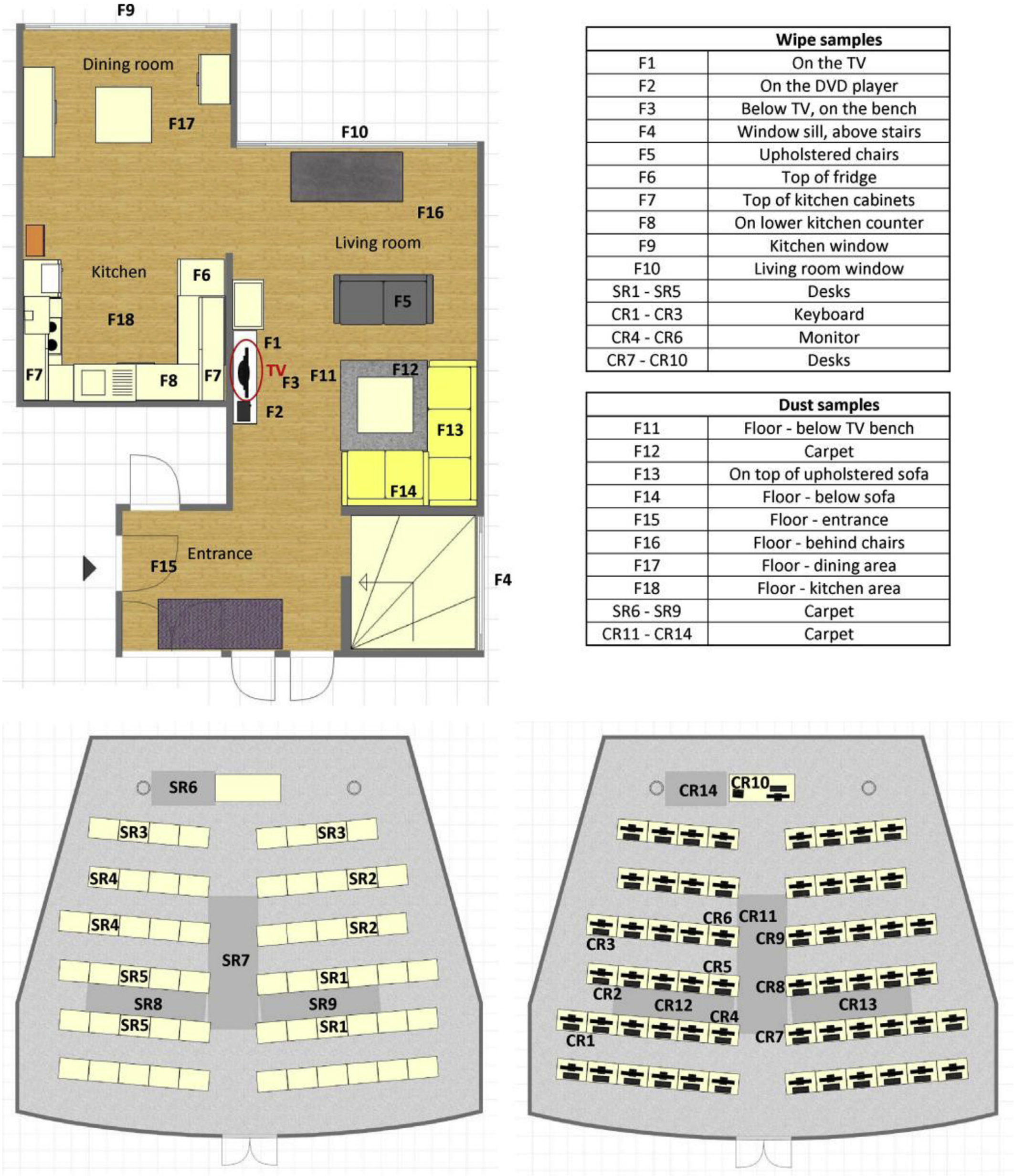
The wet wipe samples were collected using laboratory kimwipes. The kimwipes were pre-cleaned via soxhlet extraction in dichloromethane (DCM) for 8 h. At the site, the kimwipes were moistened by approximately 2—3 ml of propan-2-ol and were used to wipe each target surface. Depending on surface area and amount of dust, 1—5 wipes were used for each surface, and combined into one sample per surface. The area of the sampled surface was measured. Wipes were packed in aluminium foil and sealed in a plastic bag. Wipe samples were transported to the laboratory in a cooler box and stored at −18 °C until processing.

Dust samples from floors and fabric surfaces (e.g., sofa) were collected using a household vacuum cleaner with polyester sock inserts. Socks were precleaned via soxhlet extraction in DCM for 8 h, and before sampling and between samples, the vacuum nozzle and tube were cleaned with propan-2-ol. To collect each sample, a polyester sock was inserted into the front of the vacuum tube and held in place by the vacuum nozzle. Each target surface was vacuumed and its area measured. The sock was removed from the vacuum cleaner, sealed by a plastic cable tie, packed in aluminium foil and sealed in a plastic bag. Socks were transported to the laboratory in a cooler box and stored at −18 °C until processing.

2.2. Sample extraction and clean-up

The wipes were extracted using automated warm Soxhlet extraction in a Büchi B-811 automatic extractor, with DCM as the extraction solvent. Before extraction, samples were spiked with isotopically labelled (^{13}C) internal standards, including triphenyl phosphate, BDE 28, 47, 99, 100, 153, 154, 183 and 209, HBB, PBBZ, syn- and anti-DDC-CO, BTBPE, and DBDPE (all from Wellington Laboratories, Inc., Guelph, Ontario, Canada, except DDC-CO from Cambridge Isotope Laboratories, Inc., Tewksbury, MA, USA). The soxhlet extraction was cycled for 40 min, followed by 20 min where solvent was diverted to concentrate the extract. The extract was then concentrated to less than 10 ml and quantitatively transferred to a vial. Then the extract was split 3:7 by weight (30% and 70% aliquots) and each fraction was concentrated to a volume of 1—2 ml by nitrogen flow.

Vacuum-collected dust samples were extracted via sonication. Before extraction, samples were weighed and sieved with a 500 μm sieve (Newark Wire Cloth Company, USA) to remove coarse particles (e.g., hair, large fibres). Approximately 100 mg of the sieved dust was used for extraction. The sock was rinsed with 20 ml of 1:1 hexane:acetone (v/v) and this volume was added to the accurately weighed dust sample and the mass difference between unrinsed sock and rinsed sock was included to the mass of extracted dust. Before sonication, the sample was spiked with the same set of internal standards as the wipe samples. The sample and 20 ml of hexane:acetone were sonicated for 10 min, then the sample was allowed to settle for 20 min and the solvent supernatant was



Wipe samples	
F1	On the TV
F2	On the DVD player
F3	Below TV, on the bench
F4	Window sill, above stairs
F5	Upholstered chairs
F6	Top of fridge
F7	Top of kitchen cabinets
F8	On lower kitchen counter
F9	Kitchen window
F10	Living room window
SR1 - SR5	Desks
CR1 - CR3	Keyboard
CR4 - CR6	Monitor
CR7 - CR10	Desks

Dust samples	
F11	Floor - below TV bench
F12	Carpet
F13	On top of upholstered sofa
F14	Floor - below sofa
F15	Floor - entrance
F16	Floor - behind chairs
F17	Floor - dining area
F18	Floor - kitchen area
SR6 - SR9	Carpet
CR11 - CR14	Carpet

Fig. 1. Sampling locations in flat (F1–F18), seminar room (SR1–SR9) and computer room (CR1–CR14).

transferred to a vial. An additional 10 ml of solvent was added to the sample and it was sonicated again, allowed to settle and transferred to the same vial. This step was repeated once more. The extracts were split 3:7 (30% and 70% aliquots) by weight and each part was concentrated to a volume of 1–2 ml by nitrogen flow.

The 70% aliquot of the extract was cleaned on a sulfuric acid modified silica column (1 cm activated silica, 5 g silica modified by sulfuric acid and 1 cm cleaned silica), eluted with 30 ml of 1:1 hexane:DCM (v/v). The 30% fraction was cleaned on a column of 5 g activated silica and 1 cm sodium sulphate. This aliquot was further fractionated by elution with 20 ml DCM (Fraction 1) followed by 20 ml of 7:3 acetone:DCM (v/v) (Fraction 2). Six of the Fraction 1 samples and seven of the 70% aliquots required additional clean up (see Table S2).

After the column chromatography, 40 μl nonane was added to the samples, and they were further concentrated and quantitatively transferred to gas chromatography vials, and recovery standards (BDEs 77 and 138) were added. The 70% aliquot was used for PBDE analysis, the 30% F1 fraction for NFR analysis, and the 30% F2 fraction for OPE analysis.

2.3. *Instrumental analysis*

Samples were analysed for 10 PBDEs: BDE 28, BDE 47, BDE 66, BDE 85, BDE 99, BDE 100, BDE 153, BDE 154, BDE 183 and BDE 209; 19 NFRs: TBP-AE, TBP-BAE, TBX, DDC–CO–MA, PBEB, PBT, TBP-DBPE, HBB, DBHCTD, EH-TBB, BTBPE, syn- and anti-DDC-CO, BEH-TEBP, DBDPE, α-, and β-, γ-, and δ-DBE-DBCH, α- and β-TBCO, PBBZ, TBCT, PBB-Acr; and 17 OPEs: TIBP, TNBP, TCEP, TCIPP, DBPP, BDPP, TDCIPP, TPHP, EHDPP, TBOEP, CDP, TEHP, o-, m- and *p*-TMPP, TIPPP, TDMPP, TDBPP, TBPP. The full compound names and CAS numbers are given in Table S3.

Details of the instrumental analysis can be found in the Supplementary Material.

2.4. Quality assurance/quality control (QA/QC)

Average recoveries for spike-recovery tests are reported in Table S4 and sample recoveries in Table S5. Field blanks (kimwipes and vacuum bags) were transported to the field site, manipulated as per the samples (e.g., wipes were exposed to ambient air but not directly to sampled materials, polyester sock inserts were inserted to the vacuum nozzle, but without the vacuum cleaner being on), and then treated as per the samples for the analytical process. Six vacuum sock field blanks and six kimwipe field blanks were collected, three of each from the flat and the classrooms. Solvent blanks were also analysed: six from Büchi extraction and six from ultrasonic extraction. Analysis of the blanks demonstrated that the majority of blank contamination was coming from the field blanks, so field blanks were used to determine method detection limits (MDLs). Solvent and field blanks are reported in Table S6. MDLs were calculated based on the average of the field blank plus three times the standard deviation of the blanks (Table S7). For compounds that were not detected in the blanks, the instrument detection limit was used as the MDL. Values < MDL were recorded as such, and values > MDL were subtracted based on the average of the blank of the corresponding matrix. Mass fraction was calculated using sample mass (nanograms of analyte divided by grams of sample mass) and area density was calculated using sample area (nanograms of analyte divided by square meters of area). For the purposes of statistical analyses, values below detection were replaced by $\sqrt{2}/2$*MDL (Antweiler, 2015). Compounds with low detection (generally <60% detection frequency in a given matrix/room) were not used for statistical analysis.

Statistical analyses were performed in Statistica 12, MS Office Excel 2010 and GraphPad Prism (Version 5).

3. Results and discussion

3.1. Detection and general composition of FRs

Eleven compounds: TBX, DDC—CO—MA, TBCO, TBCT, PBB-Acr, DBPP, BDPP, *o*-TMPP, TDMPP, TDBPP, and TBPP, were not detected, or infrequently detected at low levels with no spatial patterns and will not be discussed further. In the following text we focus on 10 PBDEs, 14 NFRs and 12 OPEs which were broadly detected (Table S3). Hereafter, $\sum$DDC-CO refers to the sum of the syn- and anti-isomers, $\sum$DBE-DBCH refers to the sum of the α-, β-, γ-, and δ-isomers and $\sum$TMPP to the sum of m- and *p*-TMPP.

The FRs in all samples were dominated by OPEs, which were on average 96% of all FRs, but a number of individual NFRs and BDE 209 were also significant contributors (Fig. 2; for higher resolution of NFRs and PBDEs see Fig. S1). TCIPP was detected in every sample; it was the highest contributor in the flat samples (74% of $\sum_{12}$OPEs) and, along with TBOEP, was one of the two highest contributors in classrooms. TCIPP is frequently used in polyurethane foams, as well as textiles and plastics, while TBOEP is used in floor finish products (Brandsma et al., 2014). BDE-209 was the highest PBDE, contributing 78%, 91% and 97% to $\sum_{10}$PBDEs in the flat, seminar room and computer room, respectively. For the NFRs, BEH-TEBP and DDC-CO were predominant in most samples, however, more differences between rooms/samples were observed, e.g., contributions from HBB to $\sum_{14}$NFRs ranged from <1% in the flat and seminar room to 8% in the computer room, while $\sum$DBE-DBCH contributed 20% to $\sum_{14}$NFRs in the flat, and less than 1% in SR and CR. BEH-TEBP and DDC-CO are widely used FRs in polymer materials such as polyurethane foam, wire and cable insulation, PVC, polypropylene, ABS, and epoxy resins (Covaci et al., 2011; Xian et al., 2011). DBE-DBCH is used in home insulation, wire coatings, fabrics and appliances; and HBB is used largely on the Asia market in textiles, electronics, and textiles (Covaci et al., 2011). These differences and possible explanations will be discussed further in the following sections.

Measured mass fractions and area densities of all FRs in individual samples are given in Tables S8-S9. Summary statistics are given in Table 1.

3.2. Differences according to sampling method

The two sampling methods (wet wipes and vacuumed dust) gave a wide range of mass fractions and area densities. As dust mass fractions could not be obtained from the wet wipe samples, the differences according to sampling method can only be compared in terms of area density and percentage composition profiles.

3.2.1. Comparison of area density

The area densities of FRs were highly dependent on the dust surface loading, i.e., the mass of dust per unit area. The highest area density (ng/m^2) of FRs was observed in wipe samples from areas of long-term dust accumulation in the flat. This category represents areas of less frequent cleaning (e.g., top of kitchen cabinets, inaccessible windowsill) where dust is more likely to accumulate for a longer time period, and the dust surface loading is higher. The area densities in these areas were an order of magnitude higher than the area densities from surfaces which were used/disturbed/cleaned on a daily basis, and they are likely not representative of typical indoor human exposure. Moreover, compounds that were below detection limits in other samples can often be detected in these types of samples; for example, PBEB was detected only in this type of sample from the flat, so sampling these types of surfaces could be good strategy to obtain detectable levels of the broadest set of FRs, if an objective is merely to identify their presence indoors, e.g., when screening for what FRs are in use.

Window wipe samples had lower area densities (~pg/m^2) than other samples (~ng/m^2) from the flat. The windows are a vertical surface with low dust surface loading, as opposed to the other sampled surfaces, which were largely horizontal, suggesting that differences in gravitational settling of dust on surfaces strongly influences the measured levels. Vertical surfaces will have little or no contribution from settled dust, and will mostly reflect levels due to partitioning or impaction. Also, analytes on windows may be more susceptible to photodegradation.

The influence of the dust surface loading can be most easily identified from the floor dust samples, as for these samples we could determine the mass of dust per area of floor (Fig. S2). The floor below the sofa (location F14) was the dustiest place in the flat, with 3.98 g dust/m^2 while the area below the TV table (location F11) had the lowest dust loading (0.103 g dust/m^2), suggesting more frequent use and cleaning. The area density of FRs was much higher at F14 than F11: $\sum_{10}$PBDEs was 405 ng/m^2 at F14 vs 33.2 ng/m^2 at F11, $\sum_{14}$NFRs was 138 ng/m^2 at F14 vs. 33.4 ng/m^2 at F11, and $\sum_{12}$OPEs was 319 ng/m^2 at F14 vs 7770 ng/m^2 at F11. Moreover, the area density was strongly correlated with the dust surface loading (Spearman R's of 0.81–0.98, Table S10), with the exception of compounds which we hypothesize to have direct nearby sources, such as BEH-TEBP in sofa upholstery. No such correlation was observed for mass fraction (ng/g). The lack of relationship with mass fraction contrasts with the results of Al-Omran and Harrad (2018), who identified dilution of PBDEs and BEH-TEBP in dust in areas of high dust loading. The absence of a dilution effect in the flat suggests that, except in cases of directly adjacent sources (e.g, BEH-TEBP from sofa), there are sufficient FR sources and sufficient equilibration of the system to load all dust with similar levels of FRs and therefore dust loading is the main control on area density of FRs in dust.

In the classrooms, the locations at the fronts of the rooms (SR6

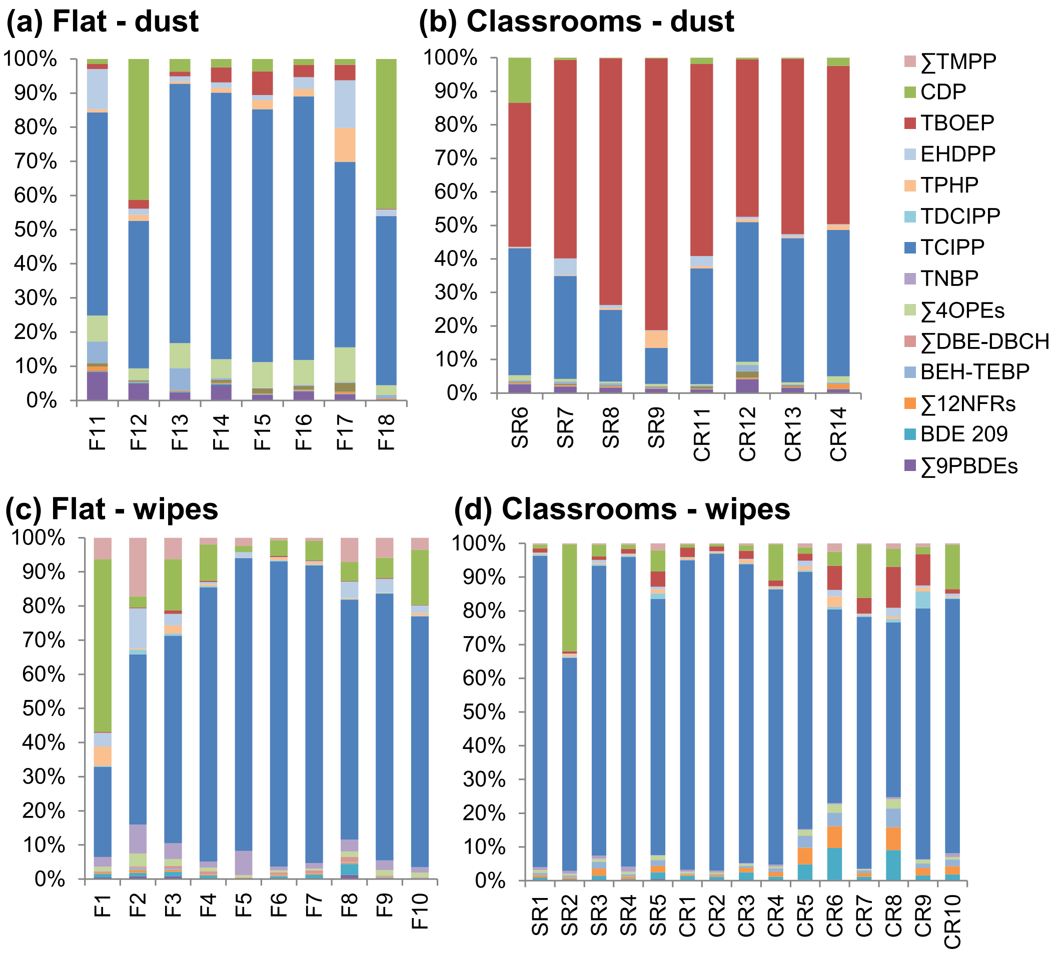


Fig. 2. Percentage composition of FRs in (a) flat and (b) classroom floor dust and (c) flat and (d) classroom wipes. Samples are labelled according to the sampling locations in Fig. 1. For dust samples, compounds with mass fraction >500 ng/g in at least one sample are shown individually, and $\sum_7$OPEs, $\sum_{12}$NFRs, and $\sum_9$PBDEs include all remaining compounds. For wipe samples, compounds with area density >100 ng/m^2 (or >100 ng/sample for electronics) in at least one sample are shown individually, and $\sum_4$OPEs, $\sum_{12}$NFRs, and $\sum_9$PBDEs include all remaining compounds. For better resolution of HFRs see Fig. S1.

and CR14) had the lowest dust loadings, likely reflecting less use of these areas of the room. However, overall the dust surface loading in the classrooms was much more homogeneous than in the flat (0.062–0.209 g/m^2 in classrooms vs. 0.103–3.98 g/m^2 in the flat) due to the greater diversity of activities in the flat, and less systematic cleaning in homes than in institutional buildings.

Differences in dust loading are not frequently considered when reporting and interpreting indoor dust, as dust is often reported in mass fraction (e.g., ng/g), however, there is a question of how appropriate mass fraction units are to human exposure estimates. Dust ingestion and dermal transfer from dust are highly uncertain (Abdallah et al., 2015; Lioy et al., 2002; Wilson et al., 2013), and some exposure pathways, e.g., particularly dermal exposure, may be more appropriately evaluated using area density (Lioy et al., 2002). Moreover, infant and toddler exposure to compounds through mouthing of objects and ingestion of dust may be significantly higher if the dust surface loading is higher. Thus, dust loading on surfaces should be an important consideration in sampling, both for detection of compounds and evaluations of exposure.

3.2.2. Comparison of profiles

In the flat, vacuumed dust and wipe samples yielded similar profiles of FRs (Fig. 2). TCIPP was the primary contributor in most dust and wipe samples, comprising 11.0–96.0% of the total OPEs, and CDP was also a significant contributor in both dust and wipes (0.1–51.9% of total OPEs). However ∑TMPP had a higher content in wipes than in floor dust samples (on average 2.7% in wipes vs. 0.2% in dust), suggesting either differences in the surfaces sampled by wipes vs. by vacuum or differences in the ability of vacuum vs. wipes to collect dust (e.g., collecting slightly different fractions of dust).

In the classrooms, the FR profiles had more distinct differences between vacuumed dusts and wipes. Most notably, TBOEP was the major contributor to floor dust samples, while in the wipe samples TCIPP was the highest contributor. The area densities of TBOEP (Fig. 3b) also reflect these differences. In the classrooms, the distinct differences in the profiles are more likely due to differences in the sampled material rather than the sampling technique, and suggest a source of TBOEP in the carpeted floor, or the trapping of specific materials in the carpet that are not trapped by the elevated surfaces, e.g., the carpet captures differently sized particles than other surfaces.

3.3. Differences between rooms

Average sample compositions by room were compared to identify the influence of different furnishings and equipment

Table 1
Area density and mass fraction in flat, seminar and computer room. The shaded rows (floors) refer to the same samples, but with different metrics (mass fraction and area density). Red numbers indicate samples with the relative standard deviation > 100%, indicating a large range in concentrations within the same room.

	Sample type and location	n	Σ_9PBDEs		BDE 209		Σ_{14}NFRs		Σ_{12}OPEs	
			Mean ±std. dev.	Median (range)	Mean ±std. dev.	Median (range)	Mean ±std. dev.	Median (range)	Mean ±std. dev.	Median (range)
Flat	Floor [ng/g]	7	18.0± 11.4	13.5 (5.19-36.0)	270± 223	184 (47.2-575)	206± 171	129 (79.6-567)	8730± 5420	7260 (3480-19100)
	Floor [ng/m^2]	7	5.99± 11.6	0.898 (0.058-31.7)	75.9± 135	19.1 (0.759-373)	31.3± 48.2	14.1 (1.38-138)	1680± 2750	390 (38.9-7770)
	Surfaces [ng/m^2]	9	13.5± 15.3	12.0 (0.360-49.0)	68.9± 80.8	20.5 (0.385-229)	223± 274	99.5 (1.02-684)	6840± 10400	1410 (225-32000)
Seminar room	Floor [ng/g]	4	18.4± 5.24	17.6 (13.5-24.9)	617± 162	617 (448-785)	380± 106	357 (283-523)	31700 ±3030	31500 (28500-35300)
	Floor [ng/m^2]	4	1.25± 0.727	1.21 (0.413-2.18)	36.9± 8.98	39.4 (24.0-44.8)	24.5± 10.6	26.7 (9.78-34.7)	2080± 896	2300 (871-2860)
	Surfaces [ng/m^2]	5	0.310± 0.180	0.263 (0.181-0.617)	1.86± 0.458	1.77 (1.29-2.52)	4.35± 1.96	4.40 (1.85-6.25)	247± 125	247 (118-394)
Computer room	Floor [ng/g]	4	20.6± 10.7	16.9 (12.6-36.0)	691± 450	551 (319-1350)	686± 447	546 (315-1340)	34400 ±9860	31200 (26600-48700)
	Floor [ng/m^2]	4	1.34± 1.11	1.02 (0.378-2.94)	46.7± 43.6	33.8 (9.52-110)	44.3± 43.5	26.2 (16.1-109)	2040± 870	2280 (795-2790)
	Surfaces [ng/m^2]	4	0.191± 0.048	0.192 (0.130-0.247)	5.43± 0.939	5.41 (4.47-6.47)	8.22± 2.37	8.36 (5.85-10.3)	270± 155	299 (56.9-426)
Electronics	Monitor in CR [ng/monitor]	3	0.144± 0.042	0.141 (0.103-0.187)	3.88± 1.22	3.64 (2.80-5.21)	3.97± 0.739	4.00 (3.22-4.70)	138± 155	52.5 (43.8-317)
	Keyboard in CR [ng/keyboard]	3	0.384± 0.140	0.464 (0.223-0.466)	8.44± 0.762	8.55 (7.63-9.14)	3.94± 0.475	3.84 (3.52-4.45)	574± 201	687 (342-694)
	Electronics in flat [ng/sample]	2	2.42	(1.71-3.12)	4.76	(2.41-7.11)	12.4	(9.37-15.5)	417	(209-625)

within the same building, and differences in buildings and room types within the same region. The floor dust samples were compared by mass fraction (ng/g) and wipe samples were compared by area density (ng/m^2).

3.3.1. *Seminar vs computer room*

The seminar room (SR) and computer room (CR) are identical in size, shape, location, and building materials, differing only by one floor, and by the equipment in the rooms. Thus any significant differences between the two rooms can be attributed to room equipment, and most likely, to the presence of computers in CR compared with SR.

Six FRs had differences in detection frequencies between CR and SR, and of these HBB, EH-TBB, TBP-BAE, TBP-AE, and DBDPE had higher detection frequencies in CR than SR. Only TNBP was detected in 4 of 5 samples in SR and only in 1 of 4 in CR. The most notable difference between the two rooms was for HBB, which was detected in all floor dust and wipe samples in CR, but only in two wipe samples in SR and in none of the floor dust samples. Previous air sampling suggested computer operation as a source of HBB emissions in this room (Vojta et al., 2017).

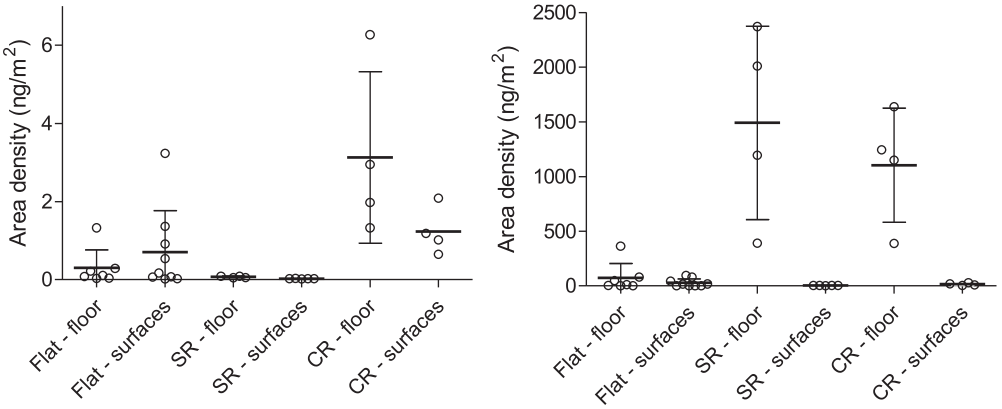
Considering compounds with consistent detection in both rooms, we compared dust mass fractions for floor dust samples and area densities for wipe samples from SR and CR with a Mann-Whitney *U* test (Tables S11 and S12). BDE 209, BTBPE and PBBZ had significantly higher mean area densities in CR than in SR, while TCIPP had significantly higher mean floor dust mass fraction in CR than in SR and TEHP higher mean in SR. The combination of detection frequencies and U-tests strongly suggest a source of HBB in CR that is not present in SR, supporting previous air sampling (Vojta et al., 2017), and possible differences in sources of TBP-AE, EH-TBB, BTBPE, DBDPE, TNBP, and TCIPP.

3.3.2. *Classrooms vs flat*

Greater differences between the flat and classrooms were expected, given the differences in type and age of building materials, furniture (desks and computers in the university building and kitchen equipment, sofa, and TV in the flat), and flooring (carpet in classrooms, parquet and small rug in flat). Firstly, the suite of detected FRs differed between the two sites. Five compounds were detected in the classrooms but not in the flat: PBT, EHDPP, *p*-TMPP, and TIPPP. $\sum$DDC-CO was the only compound that was detected in the flat but not in the classrooms.

For compounds with consistent detection in both flat and classrooms, we again compared dust mass fractions for floor dust and area densities for wipe samples with a Mann-Whitney *U* test (Table S13 and S14). The mean mass fractions of BDE 209, TCIPP, DBDPE, and TBOEP were higher in the classrooms. Conversely,

(a) HBB (b) TBOEP



(c) TCIPP (d) DBDPE

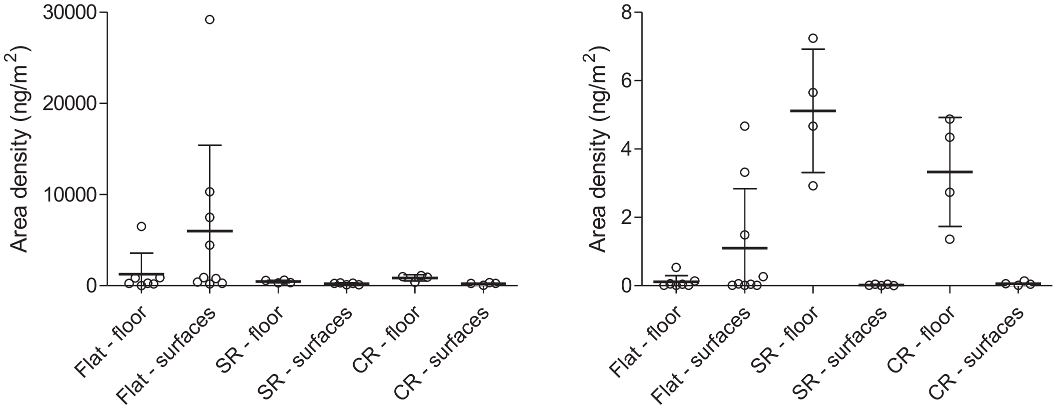


Fig. 3. Area density of (a) HBB, (b) TBOEP, (c) TCIPP and (d) DBDPE in flat and classroom floor dust and wipe samples. Horizontal lines show mean and whiskers show standard deviation.

BTBPE, ∑DBE-DBCH, TIBP and TNBP were higher in the flat. In wipes, 12 compounds were significantly higher in the flat (Table S14).

Fig. 3 shows room comparisons for HBB, TBOEP, TCIPP, and DBDPE based on area densities (ng/m^2) of all samples. TBOEP and DBDPE had on average higher levels in the classrooms than the flat, but it is apparent from Fig. 3 that this was only due to the elevated levels in floor dust, not on surfaces. This suggests the carpet as a likely source of TBOEP and possibly DBDPE. Interestingly, TBOEP is mostly used in floor polish (Brandsma et al., 2014; Cristale et al., 2016; van der Veen and de Boer, 2012), and these rooms were fully carpeted, however it is also used in rigid and flexible polyurethane foam (Cristale et al., 2016; van der Veen and de Boer, 2012), and in plastics and rubbers (van der Veen and de Boer, 2012). DBDPE has similar usage to BDE 209: in textiles, polypropylene and other polymeric materials (Covaci et al., 2011). Thus, the high levels of these compounds on classroom floors may be due to carpet materials or carpet backing, and with releases exacerbated abrasion of the carpet material during use and vacuuming. However, DBDPE also had higher area density in the flat in surface wipe samples than in floor dust from the flat or surface wipes from the classrooms. While TCIPP had a higher mass fraction (ng/g) in the classrooms, the higher area density (ng/m^2) of TCIPP was in the flat (Fig. 3c), and the elevated levels were particularly notable in the wipe samples. This difference suggests differences in the fractions of TCIPP on individual particles, e.g., small particles with high levels of TCIPP in the classrooms, such as from building-specific abrasive sources (floor or wall materials, insulation, thermoplastics and polyurethane foam) (Kemmlein et al., 2003; Liesewitz, 2000).

3.4. *Spatial variability within rooms*

To further address the objective of the study, we examined the small-scale spatial differences in the rooms, and based on these, identified how dust sampling can be adjusted to either capture or minimize these differences, as befits the study objectives.

Area densities and mass fractions in the rooms typically ranged over one order of magnitude in different samples from the same room. The magnitude of small-scale spatial variation is shown for selected FRs through room maps of the flat and CR (Fig. 5) according to area density, mass fraction, and percentage composition. More details are given in Table 1 and in Tables S8 and S9. Much of the heterogeneity in the area densities and sample composition does not appear to be directly linked to obvious sources, such as 5× higher area densities of ∑DDC-CO in one side of the computer room compared with the other (CR12 vs. CR13), or 3–5-fold variation in individual PBDE congeners, even amongst the hard floor samples from the flat.

However, in some cases, we can relate the spatial differences to sources like electronics or foam in the sofa according to the spatial distribution in the room. In the flat, the contribution of BDE 209 to $\sum_{10}$PBDEs decreased with increasing distance from the TV by 13% (Fig. 4a), which supported the hypothesis of electronics as a source of BDE 209. BEH-TEBP had a similar profile to BDE 209, with an even larger decrease; from 74% to 5% of $\sum_{14}$NFRs (Fig. 4b), except in the kitchen, where the contribution was 70%, which could indicate another source in this area. BEH-TEBP is often used in combination with EH-TBB in Firemaster 550 as an FR in foams with a ratio of 1:4 BEH-TEBP:EH-TBB, but also independently in applications such as cable insulation (Covaci et al., 2011). In the flat, the area density of BEH-TEBP was one order of magnitude higher than EH-TBB, suggesting independent usage in electronics rather than as Firemaster 550 in foams, which agrees with the dependence on distance from the electronics. However, the trends in BDE 209 and BEH-TEBP were only observable in the floor dust; wipe samples did not show any pattern. This could be due to additional confounding factors that challenge the interpretation of wipe samples, such as debromination of selected FRs via photodegradation on the windows, or higher temperatures on electronics surfaces selectively degrading certain compounds. Moreover, all floor areas had lower levels of BEH-TEBP (whether in ng/m^2 and ng/g) than the sofa surface (Fig. 5a–c), which could lead to underestimation of exposure via dust if only floor dust is measured. The decreasing percentage contribution of BEH-TEBP to $\sum_{14}$NFRs with increasing distance from the TV shown in Fig. 4 is also visible in Fig. 5c. However, the sofa (F13), armchairs (F5), carpet (F12) and floor below sofa (F14) dust have high contributions of BEH-TEBP, which suggests foam in upholstered furniture as another source.

TCIPP and also HBB displayed strong spatial gradients that may be related to source proximity (Fig. 5). For TCIPP, areas of long term dust accumulation (F4, F6) had the highest area densities (Fig. 5d), as seen for BEH-TEBP. Also similarly to BEH-TEBP, the sofa (F13) and floor below sofa (F14) had high area densities, but in contrast to BEH-TEBP, the TV and DVD (F1 and F2) had low area densities of TCIPP. This suggests spatial distributions influenced by the use of TCIPP in foams (Stapleton et al., 2009), but not in electronics. TCIPP on the kitchen floor (F18) had a high mass fraction but low area density compared with other locations. The dust loading is low on the kitchen floor, but the small amount of particles that are there have a high mass of TCIPP. This contrasts with location F14 (below sofa), where the comparison of area density and mass fraction suggests there is a high particle loading but with lower amounts of TCIPP on the particles. This demonstrates how the combined effects of room use/cleaning patterns and proximity to sources complicate interpretation of indoor dust levels.

HBB in the computer room had the highest values on the floor in the corner of the room (CR14) in all units (area density–ng/m^2; mass fraction–ng/g and percentage content - %) (Fig. 5g–i). This could be due to more limited tracking of particles from outside to the front of the room (like soil from shoes), and thus higher contribution of within-room particle sources with elevated levels of HBB. The comparison between SR and CR (section 3.3.1) identified consistently higher levels of HBB in CR than in SR, suggesting computers as a source, however from the room maps the source is not as clear. The keyboards and monitors (CR1–CR6) had a wide range of percentage content of HBB (Fig. 5i), and thus this does not clearly indicate either the PCs or keyboards as HBB sources. We also expected greater homogeneity in floor dust in terms of both ng/m^2 and ng/g, as the carpet is uniform throughout the room; however, nearby samples, e.g., CR11–CR13, had more than 2× differences in levels. This might be caused by room ventilation patterns (windows are opened only on left side of room), but no reason is clear.

4. Conclusions and recommendations

To evaluate small scale spatial differences in indoor dust, two types of indoor environments were evaluated: a residential flat and two classrooms (one with and one without computers); and two types of sampling methods were chosen—vacuum and wipe samples. Distinct differences between the flat and classrooms were observed for 12 compounds: BDE 209, $\sum$DDC-CO, $\sum$DBE-DBCH, DBDPE, TIBP, TNBP, TCIPP, PBT, EHDPP, TBOEP, *p*-TMPP and TIPPP. Comparison of seminar room and computer room levels suggested a computer-related source of HBB. Individual rooms demonstrated significant within room variability in FR levels in dust, both in mass fractions (ng/g) and area densities (ng/m^2). Levels typically ranged over one order of magnitude within a room, with larger ranges for the OPEs than PBDEs. Spatial variability in the flat was expected due to the heterogeneity in furniture and room usage, and cleaning frequencies. Spatial patterns suggested the TV, sofa and upholstered chairs, and possibly kitchen appliances as possible sources of FRs. Smaller spatial differences were expected in classroom samples, as furnishing and room use are more homogenous, but we observed differences between nearby locations (same carpet, same sampled method) with no clear explanation.

When there are very strong differences in sources between rooms (e.g., HBB between computer and seminar room) then the difference can be observed regardless of where the samples are taken, but more subtle differences are overwhelmed by selection of sampling location. For example, within the flat, if single samples were collected to represent the whole room (as is typical practice when comparing levels across many buildings), significant bias could be introduced in the reported values. The mean level of BEH-TEBP in the flat is 195 ng/g, but if we collected a single sample from

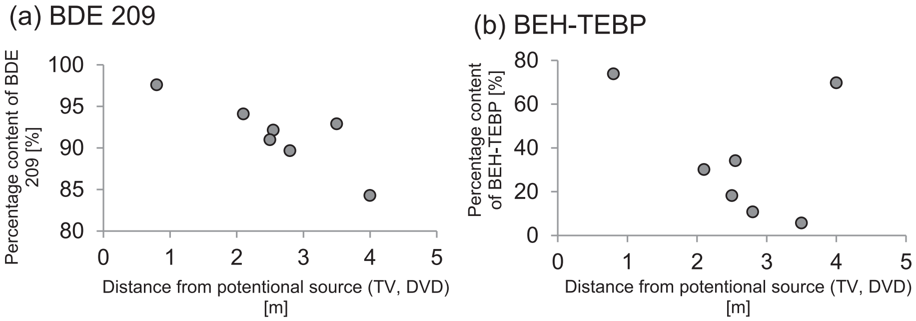


Fig. 4. Spatial variability of (a) BDE 209 as a fraction of $\sum_{10}$PBDEs and (b) BEH-TEBP as a fraction of $\sum$NFRs in floor dust relative to distance from a potential indoor source.

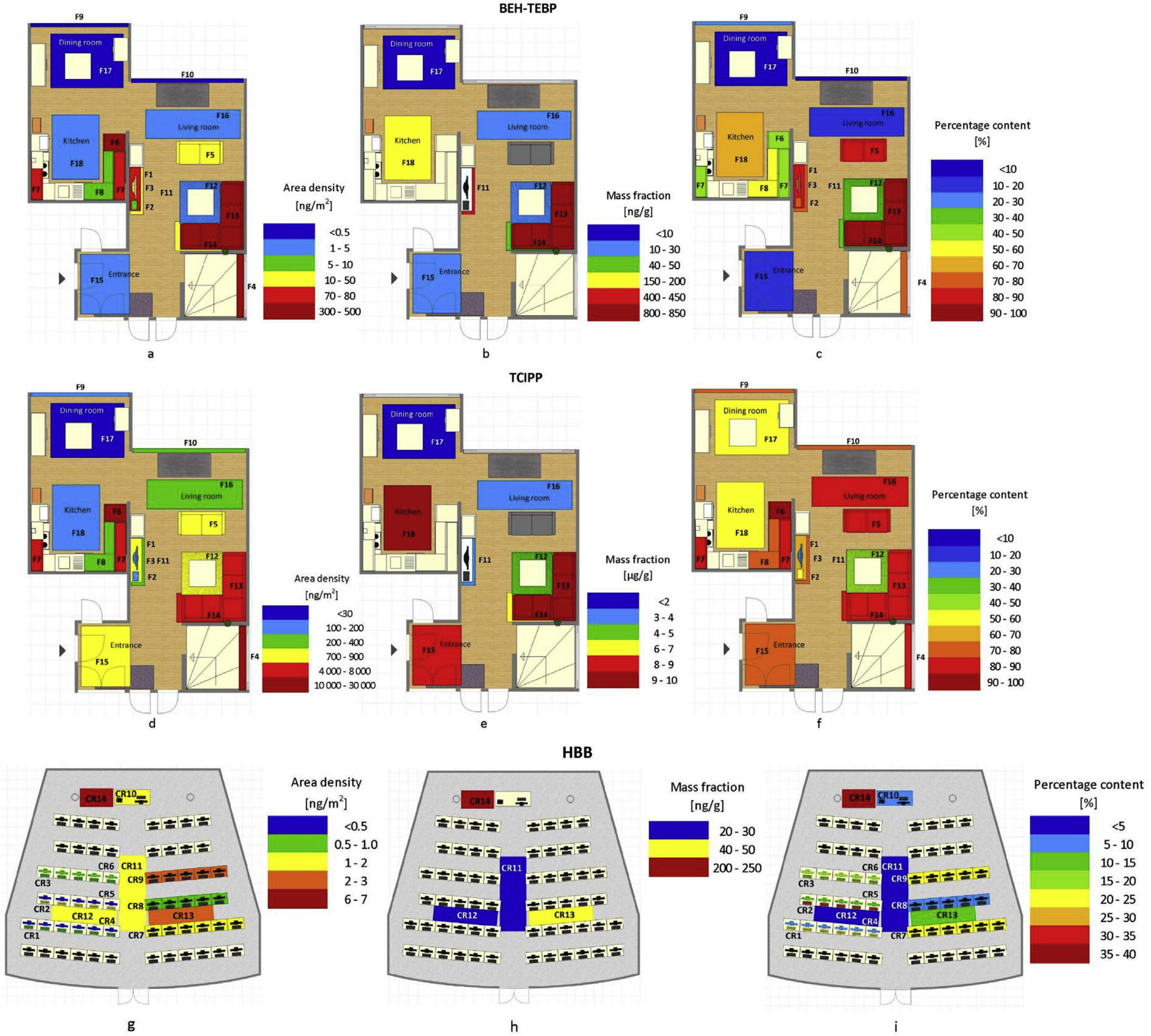


Fig. 5. Maps of area densities, mass fraction and percentage content of a-c) BEH-TEBP and d-f) TCIPP in flat and g-i) HBB in CR.

an area in the dining room (e.g., F17) we would report only 7 ng/g; while based on the area around the TV table (e.g., F11) we would measure 418 ng/g, e.g., 60-fold differences in reported levels based only on location of floor sampling within the room. If extrapolated to estimate human exposure to dust in the home, this would thus result in 60-fold differences in exposure to BEH-TEBP simply due to the location of sample collection within the home.

Moreover, this only considers the difference between floor samples; clear differences also exist between surfaces and floors (e.g., TBOEP in classroom desks vs. floor), which could lead to over- or underestimation of exposure due to usage of the location (i.e. children play more on carpet, but adults have more contact with raised surfaces, e.g., desks). Additionally, we observed that the range of surface dust loading influences levels; as a result, area density and mass fraction do not correlate in many cases, and thus careful consideration should be given to selection of the dust metric most appropriate to the study goals.

The strategy of the sampling depends on the goal of the research. If simply detecting a wide range FRs is a goal, it is ideal to sample in areas with high dust loadings, such as areas of long term dust accumulation. However, this strategy would be inappropriate for exposure assessment; rather, dust exposure is best assessed with a focus on most frequently used areas of an indoor environment, and this should be given careful consideration because there is significant spatial heterogeneity, even in seemingly homogeneous indoor spaces. When single samples are collected, composite samples are strongly recommended to limit the influence of spatial heterogeneity. Dust is an accessible and useful matrix to evaluate indoor concentrations of compounds of concern, and a necessary component of exposure assessment, however, to obtain coherent and comparable data it is necessary to make appropriate choices in sampling method, sampled area and metric for sample interpretation.

Acknowledgements

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Appendix A. Supplementary data

Supplementary data related to this article can be found at https://doi.org/10.1016/j.chemosphere.2018.04.146.

References

Abdallah, M.A.-E., Pawar, G., Harrad, S.J., 2015. Evaluation of 3D-human skin equivalents for assessment of human dermal absorption of some brominated flame retardants. Environ. Int. 84, 64–70. https://doi.org/10.1016/j.envint.2015.07.015.

Al-Omran, L.S., Harrad, S.J., 2018. Within-room and within-home spatial and temporal variability in concentrations of legacy and "novel" brominated flame retardants in indoor dust. Chemosphere 193, 1105–1112. https://doi.org/10.1016/j.chemosphere.2017.02.096.

Ali, N., Harrad, S.J., Goosey, E.R., Neels, H., Covaci, A., 2011. "Novel" brominated flame retardants in Belgian and UK indoor dust: implications for human exposure. Chemosphere 83, 1360–1365. https://doi.org/10.1016/j.chemosphere.2011.02.078.

Antweiler, R.C., 2015. Evaluation of statistical treatments of left-censored environmental data using coincident uncensored data sets. II. Group comparisons. Environ. Sci. Technol. 49, 13439–13446.

Bergman, A., Rydén, A., Law, R.J., de Boer, J., Covaci, A., Alaee, M., Birnbaum, L.S., Petreas, M.X., Rose, M., Sakai, S., Van den Eede, N., van der Veen, I., 2012. A novel abbreviation standard for organobromine, organochlorine and organophosphorus flame retardants and some characteristics of the chemicals. Environ. Bar Int. 49, 57–82.

Betts, K.S., 2008. New flame retardants detected in indoor and outdoor environments. Environ. News. https://doi.org/10.1021/es801070p.

Brandsma, S.H., de Boer, J., van Velzen, M.J.M., Leonards, P.E.G., 2014. Organophosphorus flame retardants (PFRs) and plasticizers in house and car dust and the influence of electronic equipment. Chemosphere 116, 3–9.

Cequier, E., Ionas, A.C., Covaci, A., Marcé, R.M., Becher, G., Thomsen, C., 2014. Occurrence of a broad range of legacy and emerging flame retardants in indoor environments in Norway. Environ. Sci. Technol. 48, 6827–6835. https://doi.org/10.1021/es500516u.

Covaci, A., Harrad, S.J., Abdallah, M.A.-E., Ali, N., Law, R.J., Herzke, D., de Wit, C.A., 2011. Novel brominated flame retardants: a review of their analysis, environmental fate and behaviour. Environ. Int. 37, 532–556.

Cristale, J., Hurtado, A., Gómez-Canela, C., Lacorte, S., 2016. Occurrence and sources of brominated and organophosphorus flame retardants in dust from different indoor environments in Barcelona. Spain. Environ. Res. 149, 66–76. https://doi.org/10.1016/j.envres.2016.05.001.

de Wit, C.A., 2002. An overview of brominated flame retardants in the environment. Chemosphere 46, 583–624.

Harrad, S.J., Abdallah, M.A.-E., Covaci, A., 2009. Causes of variability in concentrations and diastereomer patterns of hexabromocyclododecanes in indoor dust. Environ. Int. 35, 573–579. https://doi.org/10.1016/j.envint.2008.10.005.

Harrad, S.J., Ibarra, C., Abdallah, M.A.-E., Boon, R., Neels, H., Covaci, A., 2008a. Concentrations of brominated flame retardants in dust from United Kingdom cars, homes, and offices: causes of variability and implications for human exposure. Environ. Int. 34, 1170–1175. https://doi.org/10.1016/j.envint.2008.05.001.

Harrad, S.J., Ibarra, C., Diamond, M.L., Melymuk, L., Robson, M., Douwes, J., Roosens, L., Dirtu, A.C., Covaci, A., 2008b. Polybrominated diphenyl ethers in domestic indoor dust from Canada, New Zealand, United Kingdom and United States. Environ. Int. 34. https://doi.org/10.1016/j.envint.2007.08.008.

Karlsson, M., Julander, A., van Bavel, B., Hardell, L., 2007. Levels of brominated flame retardants in blood in relation to levels in household air and dust. Environ. Int. 33, 62–69. https://doi.org/10.1016/j.envint.2006.06.025.

Kemmlein, S., Hahn, O., Jann, O., 2003. Emissions of organophosphate and brominated flame retardants from selected consumer products and building materials. Atmos. Environ. 37, 5485–5493. https://doi.org/10.1016/j.atmosenv.2003.09.025.

Kuang, J., Ma, Y., Harrad, S.J., 2016. Concentrations of "legacy" and novel brominated flame retardants in matched samples of UK kitchen and living room/bedroom dust. Chemosphere 149, 224–230. https://doi.org/10.1016/j.chemosphere.2016.01.092.

Liesewitz, A., 2000. Substituting Environmentally Relevant Flame Retardants: Assessment Fundamentals.

Lioy, P.J., Freeman, N.C.G., Millette, J.R., 2002. Dust: a metric for use in residential and building exposure assessment and source characterization. Environ. Health Perspect. 110, 969–983.

Muenhor, D., Harrad, S.J., 2012. Within-room and within-building temporal and spatial variations in concentrations of polybrominated diphenyl ethers (PBDEs) in indoor dust. Environ. Int. 47, 23–27. https://doi.org/10.1016/j.envint.2012.06.001.

Sjödin, A., Päpke, O., McGahee, E., Focant, J.-F., Jones, R.S., Pless-Mulloli, T., Toms, L.-M.L., Herrmann, T., Müller, J.F., Needham, L.L., Patterson, D.G., 2008. Concentration of polybrominated diphenyl ethers (PBDEs) in household dust from various countries. Chemosphere 73, S131–S136.

Stapleton, H.M., Allen, J.G., Kelly, S.M., Konstantinov, A., Klosterhaus, S.L., Watkins, D.J., McClean, M.D., Webster, T.F., 2008. Alternate and new brominated flame retardants detected in U.S. house dust. Environ. Sci. Technol. 42, 6910–6916.

Stapleton, H.M., Klosterhaus, S.L., Eagle, S., Fuh, J., Meeker, J.D., Blum, A., Webster, T.F., 2009. Detection of organophosphate flame retardants in furniture foam and U.S. house dust. Environ. Sci. Technol. 43, 7490–7495.

van der Veen, I., de Boer, J., 2012. Phosphorus flame retardants: properties, production, environmental occurrence, toxicity and analysis. Chemosphere 88, 1119–1153. https://doi.org/10.1016/j.chemosphere.2012.03.067.

Venier, M., Audy, O., Vojta, Š., Bečanová, J., Romanak, K., Melymuk, L., Krátká, M., Kukučka, P., Okeme, J.O., Saini, A., Diamond, M.L., Klánová, J., 2016. Brominated flame retardants in indoor environment - comparative study of indoor contamination from three countries. Environ. Int. 94, 150–160. https://doi.org/10.1016/j.envint.2016.04.029.

Vojta, Š., Melymuk, L., Klánová, J., 2017. Changes in flame retardant and legacy contaminant concentrations in indoor air during building construction, furnishing, and use. Environ. Sci. Technol. 51, 11891–11899. https://doi.org/10.1021/acs.est.7b03245.

Vykoukalová, M., Venier, M., Vojta, Š., Melymuk, L., Bečanová, J., Romanak, K., Prokeš, R., Okeme, J.O.J.O., Saini, A., Diamond, M.L., Klánová, J., 2017. Organophosphate esters flame retardants in the indoor environment. Environ. Int. 106, 97–104. https://doi.org/10.1016/j.envint.2017.05.020.

Wilson, R., Jones-Otazo, H.A., Petrovic, S., Mitchell, I., Bonvalot, Y., Williams, D.T., Richardson, G.M., 2013. Revisiting dust and soil ingestion rates based on hand-to-mouth transfer. Hum. Ecol. Risk Assess. An Int. J. 19, 158–188. https://doi.org/10.1080/10807039.2012.685807.

Xian, Q., Siddique, S., Li, T., Feng, Y.-L., Takser, L., Zhu, J., 2011. Sources and environmental behavior of dechlorane plus—a review. Environ. Int. 37, 1273–1284.