A Comparison of Surface Sampling Techniques for Semi-Volatile Organic Substances in an Electronics Recycling Facility

Clara Jung

A thesis submitted in partial fulfillment of the
requirements for the degree of

Master of Science

University of Washington
2014

Committee:
John C. Kissel
Jeffry H. Shirai
Christopher D. Simpson

Program Authorized to Offer Degree:
School of Public Health –
Department of Environmental & Occupational Health Sciences

TABLE OF CONTENTS

LIST OF FIGURES

Figure Number Page

LIST OF TABLES

Table Number Page

Acknowledgements

There are a number of people without whom this thesis might not have been written, and to those I am greatly indebted.

First and foremost, I would like to thank my committee members for their advice, support and kindness. This thesis would not have been possible without Dr. Kissel's guidance, unsurpassed knowledge of (including, but not limited to) organic film and dermal absorption, and sarcastic (yet witty) comments. The support, friendship, and hours of painstaking edits by Jeff Shirai were truly invaluable to my completion of this degree. I would like to thank Dr. Simpson for being so understanding, patient, and for not ridiculing me (too much) regarding my lack of expertise about the inner workings of GC/MS.

I would like to express my appreciation to the UW DEOHS EHL for all their time and energy spent towards working with a troublesome matrix.

This research would not have been possible without the financial support received from DEOHS Pilot Project, Veritox, Inc., and the Education and Research Center (ERC).

I would like to thank my family for a lifetime of support and my husband James for being so tolerant of me the past two years. And last but not least, I would like to thank God for blessing me with this tremendous opportunity and allowing me to meet so many great faculty members, staff and friends at the UW.

Abstract

Consumer products and building materials contain large numbers of semi-volatile organic compounds (SVOCs), which are consequently routinely detectable in indoor air and dust and in surface wipe samples. Biomarkers of SVOC exposure are commonly found in human blood and urine, and exposures implied by observed biomarker levels often cannot be explained by dietary ingestion and/or inhalation. The apparent missing dose is most likely explained by non-dietary ingestion and/or dermal absorption. However, translation of surface loads to dose remains challenging. Recent studies have determined the presence of an organic layer on indoor surfaces that can serve as a reservoir for SVOCs. Whether SVOCs on surfaces are primarily bound to particulate matter or are dissolved in an oily film might reasonably be expected to influence the mechanism(s) by which humans are exposed to them. The objective of this study is to compare surface sampling by vacuum and solvent-moistened wipes to assess the potential magnitude of the non-particle bound reservoir. Toward that end, an electronic waste recycling facility was selected as a target-rich environment for SVOC surface sampling. Collocated vacuum and wipe samples were collected and analyzed for selected SVOCs including flame retardants and phthalates. The results failed to consistently find higher surface loads by wipe than by vacuum, which may have been attributable to high SVOC loads and limitations of chemical analyses. Therefore, additional testing was performed to further evaluate both sample collection methods and sample extraction efficiencies. Some of the key findings from the *post-hoc* sampling round provided insight regarding the recommended number of sequential wipes and extractions and preferred extraction method for highly contaminated surfaces.

Chapter 1

Introduction

The relationship between contaminated indoor surfaces and human exposure is a potentially important, but generally poorly quantified, aspect of risk assessment. Formal limits on surface contamination in indoor environments are rare. Occupational Safety and Health Administration (OSHA) standards are focused on air contamination and do not provide guidance or criteria for assessing indoor surface contamination. (Some compounds are assigned a “skin” designation indicating potential for absorption through the skin, but there is no OSHA protocol for translating that potential into surface contamination limits.) Building re-entry standards specified in terms of surface contamination have been developed for dioxins and related compounds in the aftermath of building fires (Kim and Hawley, 1985; Doull, 1998). Those standards were developed to protect office workers. In anticipation of possible terrorist attacks, the U.S. Army has developed a more general method for using surface wipe data to set acceptable surface contamination levels in office environments (U.S. Army Center for Health Promotion and Preventive Medicine, 2009). However this methodology has not been widely adopted or accepted. In contrast, air contamination is generally well regulated and studied (Baier, 1985). Surface contamination is likely to be of particular interest in the case of semivolatile organic compounds (SVOCs). Assessments of human exposure to SVOCs typically focus on dietary exposures and inhalation of indoor air and often assume that dermal pathways are negligible (Weschler et al., 2012). However, in a large scale study of children’s exposure, traditional exposure protocols were found to underpredict biomonitoring data for key SVOCs

(Morgan, 2005; Wilson, 200X) for which a mass balance could be attempted. Due to their physical-chemical properties, SVOCs are poorly soluble in air but commonly found on indoor surfaces.

Indoor exposures are a growing concern in the United States as Americans spend approximately 90% of their time indoors (EPA, 2008). In particular, SVOCs have received increasing interest since many of these compounds are listed as high production chemicals (Krol et al., 2011). Some SVOCs are commonly used indoors as additives or solvents in products including plasticizers, flame retardants, pesticides (Glorennec et al., 2011; Little et al., 2012), antioxidants, preservatives, and coalescing agents (Little et al., 2012). Coupled with relatively low turnover rates for indoor air, the low vapor pressures of SVOCs prevent their rapid flushing from indoor environments. Indoor air does, however, serve to redistribute SVOCs from their original sources to all interior surfaces (Xu et al, 2006; Little et al., 2012; Liu et al., 2013). The fate of indoor SVOCs is heavily dependent on their relative partitioning between the gas phase and various compartments such as airborne particles, settled dust, inorganic surfaces, and living organisms (Weschler and Nazaroff, 2010). The fate and transport of SVOCs is also mediated by an organic film that develops on impervious substances via direct deposition or deposition of secondary products of emission (Diamond et al., 2000). This thin film, comprised of a mixture of organic residues and inorganic particles on surfaces, can serve as a reservoir for SVOCs. Whether SVOCs on surfaces are associated with particulate matter or an oily matrix might reasonably be expected to influence their availability to humans. The distinction between SVOCs in the particle phase and in an organic liquid film is therefore central to the understanding and assessment of indoor contamination.

Surface contamination is typically assessed by either wipe or vacuum sampling. Vacuum sampling should collect particle-borne material more effectively than material present in an oily liquid layer. Wipe sampling should collect both types of matrix. Determining how to interpret measurements of surface contamination should result in a better understanding of human exposures to contaminants on surfaces. The specific objective of this study is to collect side-by-side samples of vacuum and surface wipes on vertical and horizontal surfaces to assess differences in measured surface contamination of SVOCs. Such data do not yet exist for organic compounds, so this study will help fill this data gap. The overall goal is to provide insight into the efficacy of each method, and the relative magnitudes of the particulate matter and liquid film reservoirs.

Chapter 2

Background and Significance

Indoor SVOC Surface Contamination

The potential importance of indoor SVOC surface concentrations is exhibited in the Children's Total Exposure to Persistent Pesticides and Other Persistent Organic Pollutants (CTEPP) study (Wilson et al., 2003; Morgan et al., 2004; Morgan et al., 2007; Wilson et al., 2007; Morgan et al., 2008), which investigated the aggregate exposures of preschool children and adult caregivers to a suite of persistent organic compounds, most of which are considered SVOCs. Mass balances could only be attempted for those compounds for which the primary metabolite was measured in the environment or for which the parent compound is excreted as an unmetabolized conjugate. Interestingly, for these compounds (chlorpyrifos, pentachlorophenol, 2,4-dichlorophenoxyacetic acid) predicted exposures based on environmental measurements and standard exposure factors underpredicted doses estimated based on urinary biomonitoring.. Morgan et al. (2005) were able to account for only 30% of the concentration of the urinary metabolite of chlorpyrifos, 3,5,6-trichloro-2-pyridinol (TCP), through the dietary and inhalation routes. Consequently, they state that better measurement methods are required to understand surface contamination to account for dermal and indirect ingestion routes of exposures. Wilson et al. (2007) conjectured if 100% of estimated doses of pentachlorophenol (PCP) were absorbed, the amount of urinary PCP excreted still exceeded the estimated exposures by a factor greater than 10.

The sorptive behavior of SVOCs is well documented (Weschler and Nazaroff, 2008; Weschler and Nazaroff, 2010; Little et al., 2012). Consequently, even if a source of emissions is removed from the room, contaminated surfaces can act as a secondary emitting source of SVOCs (Weschler and Nazaroff, 2008; Van den Eede et al., 2011). Understanding of indoor exposure to SVOCs requires an understanding of sources and sinks, transport, and distribution between the gas phase and other compartments including particles (Shin et al., 2013), inanimate surfaces, and even humans (Weschler and Nazaroff 2010).

Among the omnipresent SVOCs found indoors are phthalates, polybrominated diphenyl ethers (PBDEs), organophosphate flame retardants (OPFRs), and phenols. Table 1 shows a list of the compounds that were targeted in this study. Eight of the nine listed have been given a health ranking related to toxicity and potential for ingestion in house dust (Bonvallot et al., 2010). The rankings were established to prioritize the health effects of the SVOCs in settled dust (dermal pathway was not included due to the absence of toxicity reference values (TRVs) via the dermal route). The rankings were calculated differently depending on the existence of toxicity threshold values:

with toxicity threshold: $\frac{Ci_{dust}}{TRV1_i}$ (eq.1)

without toxicity threshold: $Ci_{dust} \times TRV2_i$ (eq. 2)

where Ci_{dust} is the median concentration of compound *i* in settled house dust, $TRV1_i$ is the toxicity reference value for compound *i*, and $TRV2_i$ is the potency factor for compound *i* when no toxicity reference values were available (Bonvallot et al., do not distinguish between the reference dose (RfD) and slope factors).

Table 1 List of compounds analyzed in the current study

Compound	**Acronym**	**Chemical Family**	**CAS Number**	**Molecular weight (g/mol)** [a]	**Vapor pressure (mmHg) at 25°C** [1]	**Log K_{OW}** [a]	**Health Ranking** [b]
Di-n-butyl phthalate	DBP	phthalate	84-74-2	278.3	2.01E-05	4.9	10
Di-iso-butyl phthalate	DiBP	phthalate	84-69-5	278.3	4.76E-05	4.1	5
di(2-ethylhexyl)-phthalate	DEHP	phthalate	117-81-7	390.6	1.42E-07	7.6	1
tris(1,3-dichloro-2-propyl) phosphate	TDCPP	organophosphate flame retardant	13674-87-8	430.9	2.86e^{-7} [c]	3.7	-
3,3',5,5'-tetrabromo bisphenol A	TBBP-A	phenols	79-94-7	543.9	1.37E-08	5.9	21
2,2',4,4'-tetrabromodiphenyl ether	PBDE-47	polybrominated diphenyl ethers	5436-43-1	485.8	2.20E-06	6.1	33
2,2',4,4',5,-pentabromodiphenyl ether	PBDE-99	polybrominated diphenyl ethers	60348-60-9	564.7	3.50E-07	6.8	12
2,2',4,4',5,5'-hexabromodiphenyl ether	PBDE-153	polybrominated diphenyl ethers	68631-49-2	643.6	5.10E-08	7.4	36
2,2',3,3',4,4',5,5',6,6'-decabromodiphenyl ether	PBDE-209	polybrominated diphenyl ethers	1163-19-5	959.2	4.67E-12	12.1[*]	23

[a] National Library of Medicine, 2011

[b] Ranking of compounds considered to have a toxicity threshold in settled house dust (Bonvallot et al., 2010)

[c] estimated by Bonvallot et al.

\- no ranking provided

Organic Film

A potentially important factor mediating indoor exposures is the presence of organic film on surfaces. The indoor environmental fate of SVOCs is characterized by their physico-chemical properties which include molecular weight, melting point, vapor pressure, Henry's law constant, water solubility, octanol-air partition coefficient and octanol-water partition coefficient. Particularly, the low aqueous solubility and high octanol-water and octanol-air partition coefficients of certain SVOCs signify their tendency to partition to organic matter (Cetin and Odabasi, 2008). Consequently, the organic film found on all impervious surfaces allows SVOCs to accumulate, acting as a potential reservoir for dermal contact. In addition, the organic film reflects the ambient air quality and could be used to evaluate SVOC composition (Butt et al., 2004b; Cetin and Odabasi, 2011). For example, the organic compound reservoir on the film can act similarly to a time-integrated passive sampler to determine gas-phase air concentrations (Butt et al., 2004b).

The film on outdoor surfaces is derived from deposition of vehicle emissions, emissions transformation products with oxidants and other reactive compounds commonly found in urban areas (Law and Diamond, 1998). Organic films occur during the condensation of gas-phase chemicals onto impermeable surfaces, and the condensation produces a sticky layer. Diamond et al., (2000) and Butt et al., (2004a) determined that an organic layer of film collects wet and dry particles and that gas phase compounds deposit onto the surface. The gas phase compounds can further partition into the film depending on the relative fugacity of the air and film. As a result, the film can provide a sorption and desorption medium for gas-phase organic compounds that can expand via particle mixtures and gas-phase condensation (Cetin and

Odabasi, 2011). Diamond et al. (2000) estimated the thickness of window films, comprised of both organic and particulate phases, to be between 29 to 250nm. It has been theorized that an increasing thickness of the organic film is associated with a decreasing fraction of SVOC in the vapor phase (Weschler and Nazaroff, 2008; Krol et al., 2011).

Lui et al (2003) hypothesized that organic films in indoor environments are particularly important since compounds have lower degradation rates compared to outdoors due to breakdown from photolysis and removal by precipitation. Although Lui et al. found that film masses were greater on outdoor windows compared to indoor windows, the percentage of organic matter on indoor windows was greater than on outdoor windows. In addition, furniture and textiles supply large surface areas, which often act as sinks in indoor environments potentially increasing human exposure. These attributes of organic films increase the importance of investigating and characterizing surface contamination levels. One method of characterizing surface contamination is wipe sampling. Some studies have suggested an association between SVOCs in indoor air and wipe samples (Lim et al., 2014). Tables 1 and 2 summarize the results from previous studies of household surface wipe data and workplace surface wipe data, respectively.

Table 2: Household surface wipe concentrations cited in recent literature.

Compound	n	Central tendency		Variability		Solvent[a]	Media	Sample area	Description or Sampling Information	Reference
		Measure	Value (ng/m^2)	Measure	Value (ng/m^2)					
OPFRs										
TDCPP	1	-	290	-	-	-	antistatic screen cleaner	~ 1m^2	computer screen	Marklund et al., 2003
TDCPP	1	-	170	-	-	-	antistatic screen cleaner	~ 1m^2	computer cover	Marklund et al., 2003
PBDEs										
PBDE-47	6	mean	260	range	(1, 102)	dry	gauze pad	100 cm^2	product surfaces	Lim et al., 2014
PBDE-47	135	mean	44	SD	8.1	IA (6ml)	Twillwipes	1600 in^2	indoor window	Clifton et al., 2013
PBDE-47	123	mean	86	SD	10	IA (6ml)	Twillwipes	2304 in^2	floor	Clifton et al., 2013
PBDE-47	15	mean	9	SD	13	DCM	Kimwipes	4160 ± 1765cm^2	indoor window	Cetin and Odabasi, 2011
PBDE-47	5	mean	134	SD	81.7	dry	Whatmann filter paper	100 cm^2	electronics & appliances	Toms et al., 2009b
PBDE-47	5	mean	136	SD	86.2	dry	Whatmann filter paper	100 cm^2	electronics & appliances	Toms et al., 2009b
PBDE-99	6	mean	380	range	(20, 1690)	dry	gauze pad	100 cm^2	product surfaces	Lim et al., 2014
PBDE-99	135	mean	38	SD	6.6	IA (6ml)	Twillwipes	1600 in^2	indoor window	Clifton et al., 2013
PBDE-99	123	mean	76	SD	9.3	IA (6ml)	Twillwipes	2304 in^2	floor	Clifton et al., 2013
PBDE-99	15	mean	34	SD	100	DCM	Kimwipes	4160 ± 1765cm^2	indoor window	Cetin and Odabasi, 2011
PBDE-99	5	mean	70	SD	17.3	dry	Whatmann filter paper	100 cm^2	electronics, appliances	Toms et al., 2009b
PBDE-99	5	mean	100	SD	58.7	dry	Whatmann filter paper	100 cm^2	electronics, appliances	Toms et al., 2009b

Compound	n	Central tendency		Variability		Solvent[a]	Media	Sample area	Description or Sampling Information	Reference
		Measure	Value (ng/m^2)	Measure	Value (ng/m^2)					
PBDE-153	12	mean	110	range	(ND, 630)	dry	gauze pad	100 cm^2	product surface	Lim et al., 2014
PBDE-153	135	mean	4.9	SD	0.6	IA (6ml)	Twillwipes	1600 in^2	home, indoor window	Clifton et al., 2013
PBDE-153	123	mean	14	SD	4.2	IA (6ml)	Twillwipes	2304 in^2	floor	Clifton et al., 2013
PBDE-153	15	mean	10	SD	21	DCM	Kimwipes	4160 ± 1765cm^2	indoor window	Cetin and Odabasi, 2011
PBDE-153	5	mean	6	SD	2.3	dry	Whatmann filter paper	100 cm^2	electronics, appliances	Toms et al., 2009b
PBDE-153	5	mean	2510	SD	5584.5	dry	Whatmann filter paper	100 cm^2	electronics, appliances	Toms et al., 2009b
PBDE-209	12	mean	660	range	(380, 1150)	dry	gauze pad	100 cm2	product surfaces	Lim et al., 2014
PBDE-209	135	mean	0.031	SD	5.3	IA (6ml)	Twillwipes	1600 in^2	indoor window	Clifton et al., 2013
PBDE-209	123	mean	110	SD	52	IA (6ml)	Twillwipes	2304 in^2	floor	Clifton et al., 2013
PBDE-209	15	mean	34	SD	63	DCM	Kimwipes	4160 ± 1765cm^2	indoor window	Cetin and Odabasi, 2011
PBDE-209	5	mean	3860	SD	234.8	dry	Whatmann filter paper	100 cm^2	electronics, appliances	Toms et al., 2009b
PBDE-209	5	mean	2050	SD	3704.6	dry	Whatmann filter paper	100 cm^2	electronics, appliances	Toms et al., 2009b

- not stated in reference
ND non-detect
SD: standard deviation
[a] Solvent: IA: isopropanol alcohol, DCM: dichloromethane

Table 3: Occupational surface wipe concentrations cited in recent literature.

Compound	n	Central Tendency		Variabilty		Solvent [b]	Sampling media	Area sampled	Workplace	Reference
		Measure[a]	Value (ng/m^2)	Measure	Value (ng/m^2)					
Phthalates										
DBP	7	mean	0.001	range	-	dry	gauze pad	-	indoor playground	Kim et al., 2011
DBP	12	mean	0.067	range	(ND, 0.50)	dry	gauze pad	-	playroom	Kim et al., 2011
DBP	6	mean	0.014	range	(ND, 0.03)	dry	gauze pad	-	daycare center	Kim et al., 2011
DBP	7	mean	0.021	range	(ND, 0.07)	dry	gauze pad	-	kindergarten room	Kim et al., 2011
DEHP	7	mean	0.355	range	(0.12, 0.74)	dry	gauze pad	-	indoor playground	Kim et al., 2011
DEHP	12	mean	1.232	range	(0.32, 2.33)	dry	gauze pad	-	playroom	Kim et al., 2011
DEHP	6	mean	1.2	range	(0.22, 2.05)	dry	gauze pad	-	daycare center	Kim et al., 2011
DEHP	7	mean	1.116	range	(0.08, 2.45)	dry	gauze pad	-	kindergarten room	Kim et al., 2011
PBDEs										
PBDE-47	**43**	mean	220	range	(ND, 3130)	dry	gauze pad	100 cm^2	elementary schools	Lim et al., 2014
PBDE-47	**17**	mean	190	range	(ND, 1810)	dry	gauze pad	100 cm^2	private academies	Lim et al., 2014
PBDE-47	**3**	mean	50	range	(20, 80)	dry	gauze pad	100 cm^2	public facilities	Lim et al., 2014
PBDE-47	**31**	GM	350	GSD	31	IA (3ml)	gauze pad	232 cm^2	office	Watkins et al., 2013
PBDE-47	**2**	mean*	250	range	(220, 280)	hexane	Whitman filter paper	100 cm^2	university	Schecter et al., 2005
PBDE-47	**2**	mean*	10850	range	(4400, 17300)	hexane	Whitman filter paper	100 cm^2	university	Schecter et al., 2005
PBDE-47	**5**	GM	11.5	-		IA	Kimwipes	variable	bookstore, homes, urban/industrial, ewaste, rural	Butt et al., 2004
PBDE-99	**43**	mean	150	range	(ND, 1420)	dry	gauze pad	100 cm^2	elementary schools	Lim et al., 2014

Compound	n	Central Tendency		Variabilty		Solvent[b]	Sampling media	Area sampled	Workplace	Reference
		Measure[a]	Value (ng/m^2)	Measure	Value (ng/m^2)					
PBDE-99	**17**	mean	150	range	(ND, 1640)	dry	gauze pad	100 cm^2	private academies	Lim et al., 2014
PBDE-99	**3**	mean	30	range	(20, 40)	dry	gauze pad	100 cm^2	public facilities	Lim et al., 2014
PBDE-99	**31**	GM	210	GSD	33	IA (3ml)	gauze pad	232 cm^2	office	Watkins et al., 2013
PBDE-99	**2**	mean*	380	range	(90, 670)	hexane	Whitman filter paper	100 cm^2	university	Schecter et al., 2005
PBDE-99	**2**	mean*	24600	range	(10200, 39000)	hexane	Whitman filter paper	100 cm^2	university	Schecter et al., 2005
PBDE-99	**5**	GM	17.7	-		IA	Kimwipes	variable	bookstore, homes, urban/industrial, ewaste, rural	Butt et al., 2004
PBDE-153	**43**	mean	190	range	(ND, 261)	dry	gauze pad	100 cm^2	elementary schools	Lim et al., 2014
PBDE-153	**17**	mean	20	range	(ND, 150)	dry	gauze pad	100 cm^2	private academies	Lim et al., 2014
PBDE-153	**3**	mean	220	range	(10, 640)	dry	gauze pad	100 cm^2	public facilities	Lim et al., 2014
PBDE-153	**31**	GM	21	GSD	55	IA (3ml)	gauze pad	232 cm^2	office	Watkins et al., 2013
PBDE-153	**2**	mean*	75	range	(20, 130)	hexane	Whitman filter paper	100 cm^2	university	Schecter et al., 2005
PBDE-153	**2**	mean*	2200	range	(1000, 3400)	hexane	Whitman filter paper	100 cm^2	university	Schecter et al., 2005
PBDE-153	**5**	GM	3.9	-		IA	Kimwipes	variable	bookstore, homes, urban/industrial, ewaste, rural	Butt et al., 2004
PBDE-209	**43**	mean	1770	range	(ND, 12780)	dry	gauze pad	100 cm^2	elementary schools	Lim et al., 2014
PBDE-209	**17**	mean	77	range	(ND, 3360)	dry	gauze pad	100 cm^2	private academies	Lim et al., 2014
PBDE-209	**3**	mean	7570	range	(960, 20340)	dry	gauze pad	100 cm^2	public facilities	Lim et al., 2014

Compound	n	Central Tendency		Variabilty		Solvent [b]	Sampling media	Area sampled	Workplace	Reference
		Measure[a]	Value (ng/m^2)	Measure	Value (ng/m^2)					
PBDE-209	**31**	GM	1250	GSD	38	IA (3ml)	gauze pad	232 cm^2	office	Watkins et al., 2013
PBDE-209	**2**	mean	15215	range	(7330, 23100)	hexane	Whitman filter paper	100 cm^2	university	Schecter et al., 2005
PBDE-209	**2**	mean	51750	range	(21400, 82100)	hexane	Whitman filter paper	100 cm^2	university	Schecter et al., 2005
PBDE-209	**5**	GM	47.9	-		IA	Kimwipes	variable	bookstore, homes, urban/industrial, ewaste, rural	Butt et al., 2004

* central tendency or value of variability was calculated from data provided in literature
- not stated in reference
ND non-detect
SD: standard deviation
GM: geometric mean
GSD: geometric standard deviation
[a] Solvent: IA: isopropanol alcohol, DCM: dichloromethane

Dust

Generally, when investigators attempt to determine associations between indoor exposures to SVOCs, air and/or dust samples are gathered. However, due to the chemical properties of SVOCs, dust samples have some advantages. Because of the lower vapor pressure and higher molecular weights of organophosphate and phthalate esters, these SVOCs are more abundant in dust than in air (Weschler and Nazaroff, 2008; Bergh et al., 2010). Some of the practical benefits of sampling dust versus air are that it may be more time efficient to collect dust samples, compounds with low vapor pressures are likely to have levels that exceed the limit of detection (LOD), and dust may provide a more representative profile of chemical loading over time versus air (Shin et al., 2013).

SVOC exposures related to contamination of indoor dust are routinely attributed to inhalation and non-dietary ingestion and, less frequently, to absorption through the skin. Although the general relationship between dust and inhalation has been well studied, there is less information on the impacts of dust associated with incidental ingestion and dermal pathways. Several studies have hypothesized that incidental ingestion of PBDEs in dust is a major route of human exposure (Jones-Otazo et al., 2005; Allen et al., 2008; Stapleton et al., 2008; Muenhor et al., 2010). Specifically, a study by Abdallah et al. (2008), concluded that dust ingestion of TBBP-A constituted for 34% of overall exposure. In contrast dermal absorption from dust is often discounted on the grounds that the additional mass transfer resistance from dust to skin renders the process too inefficient to be important. While there is supporting evidence that dermal absorption can generally be an important exposure pathway for some SVOCs (Lees et al., 1987; VanRooij et al., 1993; Vermeulen et al., 2003; Weschler and Nazaroff, 2012), dust-borne SVOCs on surfaces may be considered available only through ingestion. Therefore, understanding whether surface residues are particle-bound or dissolved in an oily film is potentially important in predicting indoor exposure to SVOCs.

There have been several studies quantifying the total mass of contaminants in dust samples, demonstrating indoor presence of the chemicals analyzed in the current study. Tables 3 and 4 display central tendencies of concentrations of SVOC contamination in house dust and work dust, respectively. As mentioned above, many of the selected compounds are routinely found indoors and have proved to be of concern for humans. For example, several studies have determined a relationship between PBDEs in indoor air and dust, suggesting the importance of assessing dust concentrations (Zhang et al., 2011; Watkins et al., 2013; Lim et al., 2014).

Table 4: Dust concentrations of SVOCs in households cited in recent literature.

Compound	n	Central tendency		Variability		Particle size (µm)	Location $^{a, b}$	Collected by ^{c}	Collection Method	Reference
		Measure	Value (µg/g)	Measure	Value (µg/g)					
Phthalates										
DBP	128	median	16.6	range	(ND, 1670)	-	F	R	handheld vacuum, bag	Bamai et al., 2014
DBP	128	median	34	range	(ND, 1380)	-	A	O	handheld vacuum, bag	Bamai et al., 2014
DBP	10	median	130	range	(17, 260)	-	A	R	vacuum, filter	Bergh et al., 2010
DBP	41	median	22.3	range	(5.1, 549)	-	M	R	vacuum, bag	Kanazawa et al., 2010
DBP	41	median	19.8	range	(1.8, 1476)	-	F	R	vacuum, bag	Kanazawa et al., 2010
DBP	177	GM	7860	95% CI	(6590, 9360)	0.3-10	A	R	vacuum, filter	Kolarik et al., 2008
DBP	346	median	150	mean	226	-	A	R	vacuum, filter	Bornehag et al., 2004
DBP	30	median	47	95th %ile	129.6	fine dust	-	O	vacuum bag	Fromme et al., 2004
DBP	119	mean	9.39	SD	9.7	<150	F, NC	R	vacuum	Morgan et al., 2004
DBP	119	mean	7.1	SD	5.47	<150	F, OH	R	vacuum	Morgan et al., 2004
DBP	119	median	20.1	range	(LOD, 352)	<150	M	R	vacuum, thimble	Rudel et al., 2003
DBP	6	mean	27.4	SD	17.2	<150	M	R	vacuum, thimble	Rudel et al., 2001
DBP	38	mean	100	range	(10, 1030)	0.8 – 8	M	R	vacuum, filter	Oie et al., 1997
DiBP	128	median	3.1	range	(ND, 97.4)	-	F	R	handheld vacuum bag	Bamai et al., 2014
DiBP	128	median	2.5	range	(0.6, 26.6)	-	A	O	handheld vacuum bag	Bamai et al., 2014
DiBP	10	median	4	range	(ND, 18)	-	A	R	vacuum, filter	Bergh et al., 2010
DiBP	41	median	2.4	range	(0.5, 21.8)	-	M	R	vacuum, bag	Kanazawa et al., 2010

Compound	n	Central tendency		Variability		Particle size (μm)	Location [a, b]	Collected by [c]	Collection Method	Reference
		Measure	Value (μg/g)	Measure	Value (μg/g)					
DiBP	41	median	2.9	range	(0.6, 31.1)	-	F	R	vacuum, bag	Kanazawa et al., 2010
DiBP	346	median	45	mean	97	-	A	R	vacuum, filter	Bornehag et al., 2004
DiBP	119	median	1.91	range	(ND, 39.1)	<150	M	R	vacuum, thimble	Rudel et al., 2003
DiBP	6	mean	1.32	SD	0.355	<150	M	R	vacuum, thimble	Rudel et al., 2001
DiBP	38	mean	10	range	(0, 300)	0.8 – 8	M	R	vacuum, filter	Oie et al., 1997
DEHP	128	median	1110	range	(213, 7090)	-	F	R	handheld vacuum bag	Bamai et al., 2014
DEHP	128	median	2290	range	(ND, 44000)	-	A	O	handheld vacuum bag	Bamai et al., 2014
DEHP	10	median	680	range	(130, 3200)	-	A	R	vacuum, filter	Bergh et al., 2010
DEHP	41	median	1200	range	(220, 10200)	-	M	R	vacuum, bag	Kanazawa et al., 2010
DEHP	41	median	880	range	(98.2, 5850)	-	F	R	vacuum, bag	Kanazawa et al., 2010
DEHP	177	GM	960	95% CI	(790, 1170)	0.3-10	A	R	vacuum, filter	Kolarik et al., 2008
DEHP	346	median	770	mean	1310	-	A	R	vacuum, filter	Bornehag et al., 2004
DEHP	30	median	703.4	95th %ile	1542	fine dust	-	O	vacuum bag	Fromme et al., 2004
DEHP	6	mean	315	SD	153	<150	M	R	vacuum, thimble	Rudel et al., 2001
DEHP	38	mean	640	range	(100, 1610)	0.8 – 8	M	R	vacuum, filter	Oie et al., 1997
OPFRs										
TDCPP	45	GM	1.58	95th %ile	47	150	-	O	vacuum bag	Meeker et al., 2013
TDCPP	6	mean	<0.080	range	(<0.080, 0.110)	500	-	R	vacuum, nylon sock	Brommer et al., 2012

Compound	n	Central tendency		Variability		Particle size (µm)	Location [a, b]	Collected by [c]	Collection Method	Reference
		Measure	Value (µg/g)	Measure	Value (µg/g)					
TDCPP	33	median	0.36	95th %ile	0.99	<500	F	R	vacuum, nylon sock	Van de Eede et al., 2011
TDCPP	10	median	10	range	(2.2, 27)	-	A	R	vacuum, filter	Bergh et al., 2010
TDCPP	41	median	22.3	range	(5.8, 127)	-	M	R	vacuum, bag	Kanazawa et al., 2010
TDCPP	41	median	4	range	(ND, 105)	-	F	R	vacuum, bag	Kanazawa et al., 2010
TDCPP	50	median	1.752	90th %ile	20.4	150	-	O	vacuum bag	Meeker and Stapleton, 2010
TDCPP	50	GM	1.89	range	(<.090, 56.090)	150	-	O	vacuum bag	Stapleton et al., 2009
TDCPP	2	mean*	1.49	SD*	0.5	-	F	O	vacuum bag	Marklund et al., 2003
Phenols										
TBBP-A	33	median	0.01	95th %ile	0.09	<500	F	R	vacuum, nylon sock	Van de Eede et al., 2011
TBBP-A	2	mean*	0.505	SD*	0.0212	-	-	O	vacuum bag	Takagami et al., 2009
TBBP-A	45	mean	0.087	SD	0.071	25 - 500	F	R	vacuum, nylon sock	Abdallah et al., 2008
TBBP-A	18	mean	0.146	SD	0.365	500	F	R	vacuum, nylon sock	Geens et al., 2008
PBDEs										
PBDE-47	8	mean	0.032	range	(ND, 0.012)	< 500	F	R	vacuum, bag	Lim et al., 2014
PBDE-47	7	median	0.0007	range	(0.00024, 0.0037)	-	A, suburban	R	straw broom	Tue et al., 2013
PBDE-47	6	median	0.0041	range	(0.00034, 0.018)	-	A, urban	R	straw broom	Tue et al., 2013
PBDE-47	6	mean	0.0021	range	(0.0016, 0.0036)	500	-	R	vacuum, nylon sock	Brommer et al., 2012
PBDE-47	19	median	0.015	range	(.0015, 0.047)	< 500	F	O	vacuum bag	Bjorklund et al., 2011
PBDE-47	18	median	0.038	range	0.0085, 0.250)	< 500	A	R	vacuum, filter	Bjorklund et al., 2011

Compound	n	Central tendency		Variability		Particle size (µm)	Location [a, b]	Collected by [c]	Collection Method	Reference
		Measure	Value (µg/g)	Measure	Value (µg/g)					
PBDE-47	10	median	0.008	range	(0.00473, 0.0585)	<500	M, near ewaste facility	R	brush collection	Zheng et al., 2011
PBDE-47	27	median	0.0073	range	(0.0008, 0.543	<500	M, urban	R	brush collection	Zheng et al., 2011
PBDE-47	10	median	0.0012	range	(ND, 0.00389)	<500	M, rural	R	brush collection	Zheng et al., 2011
PBDE-47	34	median	0.0091	95th %ile	0.173		-	O	vacuum bag	Fromme et al., 2009
PBDE-47	12	mean	0.093	range	(0.0167, 0.0354)	< 2000	-	O	vacuum bag	Johnson-Retrepo and Kannan, 2009
PBDE-47	2	mean*	0.00245	SD*	0.000071	-	-	O	vacuum bag	Takagami et al., 2009
PBDE-47	10	mean	0.091	SD	0.124	< 2000	F	R	vacuum bag	Toms et al., 2009a
PBDE-47	5	mean*	0.0238	SD*	0.0176	< 2000	-	R	vacuum, bag	Toms et al., 2009b
PBDE-47	30	mean	0.015	SD	0.013	25-500	F	R	vacuum, nylon sock	Harrad et al., 2008a
PBDE-47	20	mean	0.81	SD	0.96	> 25	F, US	R	vacuum, nylon sock	Harrad et al., 2008b
PBDE-47	10	mean	0.3	SD	0.27	> 25	F, Canada	R	vacuum, nylon sock	Harrad et al., 2008b
PBDE-47	20	mean	0.036	SD	0.039	> 25	F, New Zealand	O	vacuum, nylon sock	Harrad et al., 2008b
PBDE-47	28	mean	0.02	SD	0.032	> 25	F, UK	R	vacuum, nylon sock	Harrad et al., 2008b
PBDE-47	10	median	<0.014	range	(<0.014, 0.022)	< 2000	Germany	O	vacuum bag	Sjodin et al., 2008
PBDE-47	10	median	0.06	range	(0.020, 1.400)	< 2000	Australia	O	vacuum bag	Sjodin et al., 2008
PBDE-47	10	median	0.022	range	(0.007, 0.180)	< 2000	UK	O	vacuum bag	Sjodin et al., 2008
PBDE-47	10	median	0.43	range	(0.230, 3.000)	< 2000	US	O	vacuum bag	Sjodin et al., 2008
PBDE-47	5	mean	0.051	SD*	0.0614	-	M	R	vacuum, filter	Karlsson et al., 2007
PBDE-47	11	median	0.67	range	(0.24, 14.61)	<125	F	R	vacuum, thimble	Wu et al., 2007

Compound	n	Central tendency		Variability		Particle size (µm)	Location [a, b]	Collected by [c]	Collection Method	Reference
		Measure	Value (µg/g)	Measure	Value (µg/g)					
PBDE-47	9	mean	1.621	median	0.364	< 2000	-	O	vacuum bag	Schecter et al., 2005
PBDE-47	64	median	0.3	95th %ile	2.6	< 150	-	O	vacuum bag	Wilford et al., 2005
PBDE-99	8	mean	0.0142	range	(ND, 0.028)	< 500	F	R	vacuum, bag	Lim et al., 2014
PBDE-99	7	median	0.0005	range	(0.00035, 0.0028)	-	A, suburban	R	straw broom	Tue et al., 2013
PBDE-99	6	median	0.0036	range	0.00098, 0.054)	-	A, urban	R	straw broom	Tue et al., 2013
PBDE-99	6	mean	0.0026	range	(0.0015, 0.0040)	500	-	R	vacuum, nylon sock	Brommer et al., 2012
PBDE-99	19	median	0.013	range	(0.000074, 0.068)	< 500	F	O	vacuum bag	Bjorklund et al., 2011
PBDE-99	18	median	0.025	range	(0.0023, 0.130)	< 500	A	R	vacuum, filter	Bjorklund et al., 2011
PBDE-99	10	median	0.013	range	(0.00717, 0.120)	<500	M, near ewaste facility	R	brush collection	Zheng et al., 2011
PBDE-99	27	median	0.0071	range	(0.00067, 0.842)	<500	M, urban	R	brush collection	Zheng et al., 2011
PBDE-99	10	median	0.0016	range	(ND, 0.00509)	<500	M, rural	R	brush collection	Zheng et al., 2011
PBDE-99	34	median	0.0125	95th %ile	0.228		-	O	vacuum bag	Fromme et al., 2009
PBDE-99	12	mean	0.176	range	(0.0256, 0.644)	< 2000	-	O	vacuum bag	Johnson-Retrepo and Kannan, 2009
PBDE-99	2	mean*	0.0028	SD*	0.00057	-	-	O	vacuum bag	Takagami et al., 2009
PBDE-99	10	mean	0.184	SD	0.256	< 2000	F	R	vacuum	Toms et al., 2009a
PBDE-99	5	mean*	0.0356	SD*	0.0282	< 2000	-	R	vacuum, bag	Toms et al., 2009b
PBDE-99	30	mean	0.036	SD	0.038	25-500	F	R	vacuum, nylon sock	Harrad et al., 2008a
PBDE-99	10	mean	0.51	SD	0.53	> 25	F, Canada	R	vacuum, nylon sock	Harrad et al., 2008b

Compound	n	Central tendency		Variability		Particle size (µm)	Location [a, b]	Collected by [c]	Collection Method	Reference
		Measure	Value (µg/g)	Measure	Value (µg/g)					
PBDE-99	20	mean	0.087	SD	0.1	> 25	F, New Zealand	O	vacuum, nylon sock	Harrad et al., 2008b
PBDE-99	28	mean	0.047	SD	0.074	> 25	F, UK	R	vacuum, nylon sock	Harrad et al., 2008b
PBDE-99	20	mean	1.4	SD	1.6	> 25	F, US	R	vacuum, nylon sock	Harrad et al., 2008b
PBDE-99	10	median	0.01	range	(<0.004, 0.028)	< 2000	Germany	O	vacuum bag	Sjodin et al., 2008
PBDE-99	10	median	0.1	range	(0.026, 3.400)	< 2000	Australia	O	vacuum bag	Sjodin et al., 2008
PBDE-99	10	median	0.028	range	(0.010, 0.300)	< 2000	UK	O	vacuum bag	Sjodin et al., 2008
PBDE-99	10	median	0.88	range	(0.070, 3.700)	< 2000	US	O	vacuum bag	Sjodin et al., 2008
PBDE-99	5	mean	0.079	SD*	0.0673	-	M	R	vacuum, filter	Karlsson et al., 2007
PBDE-99	11	median	1.01	range	(0.29, 14.80)	<125	F	R	vacuum, thimble	Wu et al., 2007
PBDE-99	9	mean	2.295	median	0.612	< 2000	-	O	vacuum bag	Schecter et al., 2005
PBDE-99	64	median	0.43	95th %ile	4.7	< 150	-	O	vacuum bag	Wilford et al., 2005
PBDE-153	8	mean	0.2189	range	(0.080, 0.498)	< 500	F	R	vacuum, bag	Lim et al., 2014
PBDE-153	7	median	0.0003	range	0.00024, 0.0019)	-	A, suburban	R	straw broom	Tue et al., 2013
PBDE-153	6	median	0.0014	range	0.00038, 0.0096)	-	A, urban	R	straw broom	Tue et al., 2013
PBDE-153	19	median	0.0022	range	(0.00012, 0.012)	< 500	F	O	vacuum bag	Bjorklund et al., 2011
PBDE-153	18	median	0.006	range	(0.00096, 0.014)	< 500	A	R	vacuum, filter	Bjorklund et al., 2011
PBDE-153	10	median	0.011	range	(0.00272, 0.0405)	<500	M, near ewaste facility	R	brush collection	Zheng et al., 2011
PBDE-153	27	median	0.0028	range	0.00030, 0.0604)	<500	M, urban	R	brush collection	Zheng et al., 2011
PBDE-153	10	median	0.0011	range	0.00063, 0.00414)	<500	M, rural	R	brush collection	Zheng et al., 2011

Compound	n	Central tendency		Variability		Particle size (µm)	Location [a, b]	Collected by [c]	Collection Method	Reference
		Measure	Value (µg/g)	Measure	Value (µg/g)					
PBDE-153	34	median	0.0027	95th %ile	0.0282		-	O	vacuum bag	Fromme et al., 2009
PBDE-153	12	mean	0.0302	range	(<0.00004, 0.0815)	< 2000	-	O	vacuum bag	Johnson-Retrepo and Kannan, 2009
PBDE-153	2	mean*	0.00201	SD*	0.00225	-	-	O	vacuum bag	Takagami et al., 2009
PBDE-153	10	mean	0.023	SD	0.04	< 2000	F	R	vacuum	Toms et al., 2009a
PBDE-153	5	mean*	0.0068	SD*	0.0045	< 2000	-	R	vacuum, bag	Toms et al., 2009b
PBDE-153	30	mean	0.014	SD	0.025	25-500	F	R	vacuum, nylon sock	Harrad et al., 2008a
PBDE-153	10	mean	0.071	SD	0.084	> 25	F, Canada	R	vacuum, nylon sock	Harrad et al., 2008b
PBDE-153	20	mean	0.0098	SD	0.011	> 25	F, New Zealand	O	vacuum, nylon sock	Harrad et al., 2008b
PBDE-153	28	mean	0.014	SD	0.026	> 25	F, UK	R	vacuum, nylon sock	Harrad et al., 2008b
PBDE-153	20	mean	0.24	SD	0.4	> 25	F, US	R	vacuum, nylon sock	Harrad et al., 2008b
PBDE-153	10	median	<0.006	range	(<0.006, 0.022)	< 2000	Germany	O	vacuum bag	Sjodin et al., 2008
PBDE-153	10	median	0.013	range	(0.005, 0.410)	< 2000	Australia	O	vacuum bag	Sjodin et al., 2008
PBDE-153	10	median	0.005	range	(<0.002, 0.053)	< 2000	UK	O	vacuum bag	Sjodin et al., 2008
PBDE-153	10	median	0.14	range	(0.005, 0.650)	< 2000	US	O	vacuum bag	Sjodin et al., 2008
PBDE-153	5	mean	0.0049	SD*	0.0017	-	M	R	vacuum, filter	Karlsson et al., 2007
PBDE-153	11	median	0.11	range	(ND, 0.56)	<125 um	F	R	vacuum, thimble	Wu et al., 2007
PBDE-153	9	mean	0.199	median	0.0612	< 2000	-	O	vacuum bag	Schecter et al., 2005
PBDE-153	64	median	0.049	95th %ile	0.52	< 150 um	-	O	vacuum bag	Wilford et al., 2005
PBDE-209	8	mean	4.3538	range	(0.222, 26.548)	< 500	F	R	vacuum, bag	Lim et al., 2014

Compound	n	Central tendency		Variability		Particle size (µm)	Location[a, b]	Collected by[c]	Collection Method	Reference
		Measure	Value (µg/g)	Measure	Value (µg/g)					
PBDE-209	7	median	0.089	range	(0.022, 0.550)	-	A, suburban	R	straw broom	Tue et al., 2013
PBDE-209	6	median	0.16	range	(0.023, 0.200)	-	A, urban	R	straw broom	Tue et al., 2013
PBDE-209	6	mean	0.045	range	(0.021, 0.072)	500	-	R	vacuum, nylon sock	Brommer et al., 2012
PBDE-209	19	median	0.28	range	(0.110, 6.600)	< 500	F	O	vacuum bag	Bjorklund et al., 2011
PBDE-209	18	median	0.52	range	(0.190, 9.300)	< 500	A	R	vacuum, filter	Bjorklund et al., 2011
PBDE-209	33	median	0.31	95th %ile	0.92	<500	F	R	vacuum, nylon sock	Van de Eede et al., 2011
PBDE-209	10	median	1.49	range	(0.721, 4.920)	<500	M, near ewaste facility	R	brush collection	Zheng et al., 2011
PBDE-209	27	median	4.04	range	(0.498, 40.500)	<500	M, urban	R	brush collection	Zheng et al., 2011
PBDE-209	10	median	0.67	range	(0.182, 1.680)	<500	M, rural	R	brush collection	Zheng et al., 2011
PBDE-209	34	median	0.312	95th %ile	0.839		-	O	vacuum bag	Fromme et al., 2009
PBDE-209	12	mean	2.81	range	(0.327, 9.210)	< 2000	-	O	vacuum bag	Johnson-Retrepo and Kannan, 2009
PBDE-209	2	mean*	0.39	SD*	0.3253	-	-	O	vacuum bag	Takagami et al., 2009
PBDE-209	10	mean	0.377	SD	0.438	< 2000	F	R	vacuum	Toms et al., 2009a
PBDE-209	5	mean*	0.243	SD*	0.22	< 2000	-	R	vacuum, bag	Toms et al., 2009b
PBDE-209	30	mean	260	SD	580	25-500	F	R	vacuum, nylon sock	Harrad et al., 2008a
PBDE-209		mean	0.67	SD	0.32	> 25	F, Canada	R	vacuum, nylon sock	Harrad et al., 2008b
PBDE-209		mean	45	SD	130	> 25	F, UK	R	vacuum, nylon sock	Harrad et al., 2008b
PBDE-209		mean	1.6	SD	0.96	> 25	F, US	R	vacuum, nylon sock	Harrad et al., 2008b
PBDE-209	45	mean	260	SD	580	25 - 500	F	R	vacuum	Abdallah et al., 2008

Compound	n	Central tendency		Variability		Particle size (µm)	Location [a, b]	Collected by [c]	Collection Method	Reference
		Measure	Value (µg/g)	Measure	Value (µg/g)					
PBDE-209	10	median	0.063	range	(<0.006, 0.410)	< 2000	Germany	O	vacuum bag	Sjodin et al., 2008
PBDE-209	10	median	0.73	range	(0.023, 13.000)	< 2000	Australia	O	vacuum bag	Sjodin et al., 2008
PBDE-209	10	median	10	range	(0.910, 54.000)	< 2000	UK	O	vacuum bag	Sjodin et al., 2008
PBDE-208	10	median	2	range	(0.120, 21.000)	< 2000	US	O	vacuum bag	Sjodin et al., 2008
PBDE-209	5	mean	0.47	SD*	0.63	-	M	R	vacuum, filter	Karlsson et al., 2007
PBDE-209	11	median	ND	range	(ND, 9.6)	<125 um	F	R	vacuum, thimble	Wu et al., 2007
PBDE-209	9	mean	8.567	median	0.665	< 2000	-	O	vaccum bag	Schecter et al., 2005
PBDE-209	64	median	0.63	95th %ile	4.1	< 150 um	-	O	vacuum bag	Wilford et al., 2005

* central tendency or value of variability was calculated from data provided in literature
- not stated in reference
ND non-detect
GM: geometric mean
SD: standard deviation
GSD: geometric standard deviation
CI: confidence interval
%ile: percentile

[a] Location of sample collection: F: floor surfaces, A: above floor surfaces, M: multi-surface (includes floor and above floor surfaces)

[b] Location specified if analyzed at multiple locations in the same study: NC: North Carolina, OH: Ohio, US: United States, UK: United Kingdom

[c] Sample collected By: R: researcher, O: occupant

Table 5: Dust concentrations SVOCs in work environments cited in recent literature.

Acronym	n	Central tendency		Variability		Occupational Setting	particle size (µm)	Location [a]	Collected by [b]	Collection Method	Reference
		Measure	Value (µg/g)	Measure	Value (µg/g)						
Phthalates											
DBP	78	mean	118.56	range	(29, 542)	indoor playground	-	F	R	vacuum, filter	Kim et al., 2011
DBP	76	mean	90.75	range	(29, 531)	playroom	-	F	R	vacuum, filter	Kim et al., 2011
DBP	82	mean	107.76	range	(29, 908)	daycare center	-	F	R	vacuum, filter	Kim et al., 2011
DBP	79	mean	116.56	range	(31, 719)	kindergarten room	-	F	R	vacuum, filter	Kim et al., 2011
DBP	10	median	150	range	(38, 560)	day care	-	A	R	industrial vacuum, filter	Bergh et al., 2010
DBP	10	median	100	range	(20, 450)	offices	-	A	R	industrial vacuum, filter	Bergh et al., 2010
DiBP	10	median	3	range	(ND, 32)	day care	-	A	R	industrial vacuum, filter	Bergh et al., 2010
DiBP	10	median	37	range	(ND, 106)	offices	-	A	R	industrial vacuum, filter	Bergh et al., 2010
DEHP	78	mean	389.27	range	(131, 705)	indoor playground	-	F	R	vacuum, filter	Kim et al., 2011
DEHP	76	mean	398.76	range	(222, 673)	playroom	-	F	R	vacuum, filter	Kim et al., 2011
DEHP	82	mean	358.23	range	(222, 673)	daycare center	-	F	R	vacuum, filter	Kim et al., 2011
DEHP	79	mean	375.27	range	(66, 556)	kindergarten room	-	F	R	vacuum, filter	Kim et al., 2011
DEHP	10	median	1600	range	(260, 5800)	day care	-	A	R	industrial vacuum, filter	Bergh et al., 2010
DEHP	10	median	1100	range	(57, 3700)	office	-	A	R	industrial vacuum, filter	Bergh et al., 2010

Acronym	n	Central tendency		Variability		Occupational Setting	particle size (µm)	Location [a]	Collected by [b]	Collection Method	Reference
		Measure	Value (µg/g)	Measure	Value (µg/g)						
OPFRs											
TDCPP	10	mean	0.15	range	(<0.080, 0.290)	offices	500	-	R	vacuum, nylon sock	Brommer et al., 2012
TDCPP	15	median	0.76	95th %ile	19.3	stores	<500	F	R	vacuum, nylon sock	Van de Eede et al., 2011
TDCPP	10	median	9.1	range	(3.9, 150)	day care	-	A	R	industrial vacuum, filter	Bergh et al., 2010
TDCPP	10	median	17	range	(3.3, 91)	office	-	A	R	industrial vacuum, filter	Bergh et al., 2010
TDCPP	12	mean*	133.2	SD*	19.7	public spaces	-	F	H	vacuum bag	Marklund et al., 2003
TDCPP	4	mean*	135.1	SD*	38.4	offices	-	F	H	vacuum bag	Marklund et al., 2003
TDCPP	2	mean*	1.2	SD*	0.02	radio shop	-	F	H	vacuum bag	Marklund et al., 2003
Phenols											
TBBP-A	24	mean	0.049	SD	0.046	offices	25 - 500	F	R	vacuum, nylon sock	Abdallah et al., 2008
TBBP-A	4	mean	0.22	SD	0.14	public spaces	25 - 500	F	R	vacuum, nylon sock	Abdallah et al., 2008
TBBP-A	2	mean*	0.0725	SD*	0.0389	offices	500	F	R	vacuum, nylon sock	Geens et al., 2008
PBDEs											
PBDE-47	24	mean	0.2222	range	(ND, 4.891)	elementary schools	< 500	F	R	vacuum, bag	Lim et al., 2014
PBDE-47	15	mean	0.0462	range	(0.003, 0.260)	private academies	< 500	F	R	vacuum, bag	Lim et al., 2014
PBDE-47	4	mean	0.0194	range	(ND, 0.039)	public facilities	< 500	F	R	vacuum, bag	Lim et al., 2014
PBDE-47	31	GM	0.697	GSD	0.004	office	<500	M	R	vacuum, thimble	Watkins et al., 2013

Acronym	n	Central tendency		Variability		Occupational Setting	particle size (µm)	Location [a]	Collected by [b]	Collection Method	Reference
		Measure	Value (µg/g)	Measure	Value (µg/g)						
PBDE-47	10	mean	0.019	range	(0.0057, 0.050)	office	500	-	R	vacuum, nylon sock	Brommer et al., 2012
PBDE-47	25	median	0.16	range	(0.0069, 1.800)	ewaste storage facility	25-500	F	R	vacuum, nylon sock	Muenhor et al., 2010
PBDE-47	5	mean*	0.0923	SD*	0.0789	office	< 2000	-	R	vacuum bag	Toms et al., 2009b
PBDE-47	18	mean	0.067	SD	0.094	office	25-500	F	R	HVS3	Harrad et al., 2008a
PBDE-99	24	mean	0.2158	range	(ND, 4.566)	elementary schools	< 500	F	R	vacuum, bag	Lim et al., 2014
PBDE-99	15	mean	0.0422	range	(0.005, 0.343)	private academies	< 500	F	R	vacuum, bag	Lim et al., 2014
PBDE-99	4	mean	0.0085	range	(ND, 0.019)	public facilities	< 500	F	R	vacuum, bag	Lim et al., 2014
PBDE-99	31	GM	0.915	GSD	0.006	office	<500	M	R	vacuum, thimble	Watkins et al., 2013
PBDE-99	10	mean	0.0077	range	(0.0033, 0.020)	office	500	-	R	vacuum, nylon sock	Brommer et al., 2012
PBDE-99	25	median	0.38	range	(0.010, 4.600)	ewaste storage facility	25-500	F	R	vacuum, nylon sock	Muenhor et al., 2010
PBDE-99	5	mean*	0.129	SD*	0.113	office	< 2000	-	R	vacuum bag	Toms et al., 2009b
PBDE-99	18	mean	0.12	SD	0.16	office	25-500	F	R	HVS3	Harrad et al., 2008a
PBDE-153	24	mean	0.0298	range	(0.001527, 0.171)	elementary schools	< 500	F	R	vacuum, bag	Lim et al., 2014
PBDE-153	15	mean	0.0326	range	(0.003, 0.098)	private academies	< 500	F	R	vacuum, bag	Lim et al., 2014
PBDE-153	4	mean	0.1024	range	(0.037, 0.199)	public facilities	< 500	F	R	vacuum, bag	Lim et al., 2014
PBDE-153	31	GM	0.138	GSD	0.005	office	<500	M	R	vacuum, thimble	Watkins et al., 2013

Acronym	n	Central tendency		Variability		Occupational Setting	particle size (µm)	Location [a]	Collected by [b]	Collection Method	Reference
		Measure	Value (µg/g)	Measure	Value (µg/g)						
PBDE-153	25	median	0.13	range	(0.0026, 3.300)	ewaste storage facility	25-500	F	R	vacuum, nylon sock	Muenhor et al., 2010
PBDE-153	5	mean*	0.0213	SD*	0.0119	office	< 2000	-	R	vacuum bag	Toms et al., 2009b
PBDE-153	18	mean	0.016	SD	0.023	office	25-500	F	R	HVS3	Harrad et al., 2008a
PBDE-209	24	mean	4.1952	range	(0.201, 44.724)	elementary school	< 500	F	R	vacuum, bag	Lim et al., 2014
PBDE-209	15	mean	6.267	range	(0.525, 31.747)	private academies	< 500	F	R	vacuum, bag	Lim et al., 2014
PBDE-209	4	mean	6.2145	range	(0.574, 11.769)	public facilities	< 500	F	R	vacuum, bag	Lim et al., 2014
PBDE-209	31	GM	4.204	GSD	0.0029	office	<500	M	R	vacuum, thimble	Watkins et al., 2013
PBDE-209	10	mean	0.12	range	(0.028, 0.310)	offices	500	-	R	vacuum, nylon sock	Brommer et al., 2012
PBDE-209	25	median	20	range	(0.250, 250.000)	ewaste storage facility	25-500	F	R	vacuum, nylon sock	Muenhor et al., 2010
PBDE-209	5	mean*	1.0883	SD*	0.0842	office	< 2000	-	R	vacuum bag	Toms et al., 2009b
PBDE-209	28	mean	30	SD	67	offices	25-500	F	R	HVS3	Abdallah et al., 2008
PBDE-209	18	mean	30	SD	67	office	25-500	F	R	HVS3	Harrad et al., 2008a

* central tendency or value of variability was calculated from data provided in literature

- not stated in reference

ND non-detect

GM: geometric mean

GSD: geometric standard deviation

SD: standard deviation

CI: confidence interval

%ile: percentile

[a] Location of sample collection: F: floor surfaces, A: above floor surfaces, M: multi-surface (includes floor and above floor surfaces)

[b] Collected By: R: researcher, O: occupant

Semi-Volatile Organic Compounds (SVOCs)

Widespread human exposure to SVOCs is a direct result of their extensive use in modern society since they are present in a wide variety of chemical compounds. SVOCs have low vapor pressures ranging from 10^{-9} to 10 Pa at 25°C, which are nevertheless sufficient to permit redistribution from their original source to indoor air and then to interior surfaces. Furthermore, SVOCs can persist for long periods of time even after the primary source is removed (Weschler and Nazaroff, 2008). For example, although DDT was banned in the U.S. in the 1970s, it is still found in measureable concentrations in some studies (Krol et al., 2011). The indoor presence of SVOCS is largely due to their being resistant to degradation and to removal by ventilation.

SVOCs constitute a diverse class of pollutants that have become a health concern due to their ubiquitous nature and potential adverse health effects. The major indoor exposure routes of SVOCs are dietary and non-dietary ingestion, inhalation, and dermal absorption. Dietary ingestion can take place subsequent to upstream contamination of the food supply or when SVOCs deposit from air onto food while in storage in an indoor environment. Non-dietary ingestion is typically attributed to hand-to-mouth or object-to-mouth contact. Inhalation may be of gas-phase or particle-bound SVOCs. Dermal exposure can result from direct transfer from air to skin and/or by mechanical transfer of residues from contaminated surfaces to skin. In either case, the initial transfer could be to clothing, which subsequently transmits the SVOC to the skin. Consequently, indoor exposures can plausibly be dependent upon surface contamination levels.

The Center of Disease Control and Prevention (CDC) conducts large-scale longitudinal biomonitoring studies measuring blood and urine samples in the U.S. population for over a hundred SVOCs. Comparison of results from the third National Report on Human Exposure to Environmental Chemicals to the second reveals that exposures to some SVOCs are trending upwards (CDC, 2003; CDC, 2005; Weschler and Nazaroff, 2008). For example, three out of four phthalate metabolites that are routinely detected in urine had increased from subsequent reports. Many SVOCs are linked to serious adverse health effects such as asthma (Shin et al., 2013), endocrine disruption (Wechler and Nazaroff, 2008; Little et al., 2012 Shin et al., 2013), and impacts on reproduction and development of fetuses, infants and children (Bonvallot et al., 2010; Xu and Little, 2013).

Polybrominated Diphenyl Ethers (PBDEs)

PBDEs were introduced in the 1970s and placed into many consumer products such as carpet linings, mattresses, furniture, televisions, computers, and appliances. They were designed to reduce the flammability of products and to increase escape time in the event of a fire. The production of PBDEs has increased since its introduction. For example, from 1992 to 1999, the estimated annual global production increased from 50,000 tons to 200,000 tons (Thomsen et al., 2001). However, due to studies associating adverse health effect with PBDEs, the European Union banned three of thirteen BDE classes - pentabrominated diphenyl ether (penta-BDE) and octabrominated diphenyl ether (octa-BDE) in 2004, and decabrominated diphenyl ether (deca-BDE) in 2008. In Canada, all congeners of PBDEs were banned by 2006. Although the U.S. voluntarily terminated production of penta-BDE and octa-BDEs, deca-BDEs have only been banned in a few states (Schecter et al., 2009; Ni et al., 2013). Despite these bans on various PBDE classes in some parts of the world, there is still a sizeable demand for production of deca-BDEs, which is being fulfilled by China and other Asian countries (Ni et al., 2013). As a result of the changes in the regulations and production trends of PDBEs, the potential environmental and health impacts have continued to receive growing attention.

PBDEs have not been reported to naturally occur in the environment, and their production is assumed to be strictly due to anthropogenic activity. PDBEs are persistent and lipophilic, thus have high potential for bioaccumulation in the environment. Human studies have recently revealed that PBDEs can impact intellectual and behavioral patterns in children exposed in utero (Bjorklund et al., 2012; Ni et. al., 2013), decrease fecundability in women (Johnson-Restrepo and Kannan, 2009; Schecter et al., 2009; Bjorklund et al., 2012; Ni et. al., 2013), increase incidence of cryptorchidism in male infants (Allen et al., 2008; Bjorklund et al., 2012; Ni et. al., 2013), and have detrimental effects in human fetal brain cells (Ni et. al., 2013). These adverse health effects could potentially be associated with the presence of PBDEs in indoor environments.

Organophosphate Flame Retardants (OPFRs)

A result of the limited phase-out of certain PBDEs has created a new health concern from the production of an alternative class of flame retardants – organophosphate flame retardants (OPFRs). Similar to PBDEs, OPFRs are additives in polymers, resins, latexes, and foam and are not chemically bonded to these products. Van den Eede et al (2011) found that OPFRs are released into the environment and can distribute to indoor and outdoor environments by abrasion and volatilization of the contaminated products. Further, the authors investigated OPFRs in Belgian homes and determined their levels in indoor dust are 20 to 30 times greater compared to the levels of PBDEs and hexabromocyclododecanes (HBDCs). This signifies an increased potential for exposure of OPFRs via dust pathways, suggesting a need to better understand the impacts of OPFR contaminated dust. OPFR exposures have been associated with effects on neurodevelopment and growth in developing tissue (Xu and Zhang, 2011), asthma and allergies (Araki et al., 2013), altered hormone levels (Meeker and Stapleton, 2010; Van den Eede et al., 2011), and decreased semen quality in men (Meeker and Stapleton, 2010).

Common OPFR replacements for penta-BDE in polyurethane foam are tris(1,3-dichloro-2-propyl) phosphate (TDCPP) and tris(1-chloro-2-propyl) phosphate (TCPP) (Stapleton et al., 2009). TDCPP was previously used in children's pajamas in the 1970's and early 1980's, but it was phased out of this application when a study determined it readily is absorbed through the skin and is considered to be mutagenic (Gold et al., 1978, Marklund et al., 2003). In addition, chronic exposure to TDCPP has been identified as having carcinogenic properties (Van den Eede et al., 2011). Despite these potential adverse effects, U.S. consumption of TDCPP and triphenyl phosphate (TPP) as a flame retardant additive, plasticizer or lubricant is approximately 4,500 metric tons annually and their application and use is expected to increase (Meeker et al., 2013).

A recent study analyzing TDCPP determined that household dust may be an important source of exposure (Meeker et al., 2013).

Phenols

Phenols are a class of SVOCs consisting of one or more hydroxyl groups attached to an aromatic hydrocarbon group. Some uses of phenols are as nonionic surfactants (Xu and Zhang, 2011), plasticizers (Bonvallot et al, 2010), and an intermediate in organic synthesis (Bonvallot et al, 2010). A common phenolic compound, tetrabromobisphenol-A (TBBP-A), is also considered a brominated flame retardant used to prevent and minimize fire hazards. When TBBP-A is used as a BFR, it is employed as an additive or reactive. If used as an additive in products such as high-impact polystyrene (HIPS) or acrylonitrile-butadiene-styrene (ABS) resins, the TBBP-A is not chemically bonded to the components of the products and can easily leach out of its matrix. Additionally, when TBBP-A is added as a reactive flame retardant in products, the excess of non-polymerized TBBP-A can still be emitted to the environment (Abdallah et al., 2008).

In both applications, TBBP-A is easily emitted into indoor environments and is therefore likely to be associated with suspended particulate matter following release (Sjodin et al., 2001). Consequently, TBBP-A has been identified in dust, food, air, sewages sludge, sediment, and human blood (Abdallah et al., 2008; Geens et al., 2009). The presence of TBBP-A indoors has caused growing concern since it is the most widely used BFR and its global demand is increasing (Abdallah et al., 2008; Eljarrat and Barcelo, 2011).

Phthalates

Phthalates are used as plasticizers in polyvinyl chloride (PVC) plastics, which are utilized in many consumer products. Phthalates can leach, migrate, or evaporate into indoor air, the atmosphere, foodstuffs, surfaces and other materials, because phthalate plasticizers are not chemically bound to PVC (Heudorf et al., 2007). The most commonly used plasticizers used in PVC prior to the 1990's were di-*n*-butyl phthalate (DBP), di-iso-butyl phthalate (DiBP), and di(2-ethylhexyl)-phthalate (DEHP) (Wensing et al., 2005). Due to their toxicological, bioaccumulative, and persistent properties, the U.S. Environmental Protection Agency (EPA) developed a

Phthalates Action Plan (2012), adding eight phthalates (including DBP, DiBP, and DEHP) onto the Concern List in Toxic Substances Control Act (TSCA) section (5)(b)(4) as chemicals that present unreasonable risk of injury and harm to the environment.

Some of the potential adverse health effects associated with phthalates are effects on the development of the male reproductive tract, prenatal mortality, reduced growth and birth weight, and asthma and allergies in children (Heudorf et al., 2007; Berg et al., 2010; Xu and Zhang, 2011). Although phthalate exposures have been traditionally associated with ingestion (i.e., object-to-mouth exposure in children and oily foods stored in contaminated containers), studies have increasingly linked phthalate exposures to contaminated dust (Rudel et al., 2003; Bergh et al., 2011; Bamai et al., 2013). Furthermore, phthalate air concentrations are several orders of magnitude higher indoors than outdoors, signifying the impacts of consumer products in indoor environments (Bergh et al., 2010). These high indoor concentrations can be seen in recent literature for household and work environments (Tables 4 and 5).

Surface Sampling: Wipe and Vacuum Methods

There are various media and materials that can be used in wipe sampling methods. The method most commonly used to gather data for organic films involves wetting a wipe material (e.g., Kimwipes or gauze pads) with a solvent (e.g., dichloromethane or isopropanol alcohol) or by using prepared wipes such as Ghost Wipes™ (for metals only). There have been a few studies evaluating the different wipe testing methods and even fewer for SVOCs. One of the SVOC studies compared the use of cotton cheesecloth and cotton pads for perfluorooctanoate (PFO), and determined comparable analytical results (Botelho et al., 2009). A non-SVOC study compared the removal efficiency for lead oxide using Whatman filter paper and moist paper towels, where both media gave similar and good removal efficiencies (Chavalitnitkul et al., 1984). The sampling surface plays an important role as the rougher and porous surfaces had decreased removal efficiencies (Chavalitnitkul et al., 1984).

Multiple vacuum dust sampling methods have been employed in studies targeting SVOCs, including the use of the high volume small surface sampler (HVS3), vacuuming onto pre-cleaned and pre-weighed filters, cellulose extraction thimbles, nylon socks, and affixed paper bags, extraction of fine dust particles between the inner and outer layers of vacuum bags, and extraction directly from a vacuum bag, container or canister (Wechler and Nazaroff, 2010). When comparing the household vacuum cleaner and HVS3, the results from the two studies determined the SVOC fraction of dust samples (particle size <150 µm) in which material was detectable or not detectable were similar (Colt et al., 1998, 2008). Differences have been observed when the researcher collects samples compared to when the occupant collects the samples. The vacuum samples which were collected by the researchers proved to be systematically higher in contaminant concentrations in dust versus subject collected samples (Allen et al., 2009; Wechler and Nazaroff, 2010). The difference may have been because the researchers focused on areas that were suspected to have higher concentrations of contaminants or because researcher collected samples retained a larger fraction of fine particles. Floor dust and multi-surface non-floor surface dust samples were different only when the surface where collection took place had known SVOC contaminants. For example, DiBP is commonly used in vinyl flooring, and the concentrations were significantly higher in the floor dust samples than the non-floor samples (Kanazawa et al., 2009). These findings suggest that the varying vacuum methods can contribute to inconsistent results.

The only comparative studies that have been conducted to evaluate side-by-side wipe and vacuum collection methods were for lead (Farfel et al., 1994) and beryllium (Dufresne et al., 2011). However, the physical and chemical properties of lead and beryllium are far different from those of SVOCs and thus it would likely be inaccurate to extrapolate results from the metal studies to surface contamination of SVOCs in particular and organic chemicals in general. One study characterized the association between PBDEs in indoor air, floor dust and surface wipes, but did not employ side-by-side sampling methods (Watkins et al., 2013). Watkins et al. found a positive correlation of penta-BDEs between surface wipes and vacuumed dust in newly constructed buildings (Spearman $r = 0.77$, $p = 0.07$) and older buildings with no recent renovations ($r = 0.81$, $p = 0.01$), but no correlations in older buildings with renovations a year prior to sampling ($r = 0.24$, $p = 0.35$). For BDE-209, they determined a significant correlation between surface wipes and office dust ($r = 0.69$, $p = 0.007$) among 14 out of 31 surface wipe samples where data were available. Although the study characterized the relationship in dust and surface wipes, no side-by-side comparison was made. This current study aims to fill this data gap by collecting SVOC concentrations in collocated vacuum and wipe samples. By examining the comparative analysis between vacuum and wipe methods, we hope to determine an effective method for surface sampling of SVOCs.

SVOCs in Electronics Recycling Facilities

According to the Institute of Scrap Recycling, the electronics waste (e-waste) recycling industry has been growing rapidly for the past ten years (Harris, 2013). In January 2009, Washington State passed legislation (SB 6428) to provide a free electronic product recycling program to residents, small businesses, school districts, small governments, special purpose districts and charities, financed by the manufacturers. Since the program was implemented, the E-cycle Program has collected approximately 223 million pounds of electronics as of April 2014, comprised of televisions, monitors, and computers. Increasing attention to potential occupational health hazards has accompanied the growth of the e-waste recycling industry and the processes involved in the dismantling of electronic products.

Studies have already been conducted at e-waste facilities and have suggested that these work sites have increased SVOC exposures compared to the general public. For example, Cahill et al. (2007) measured indoor air PBDE concentrations at an U.S. e-waste facility using an active air sampler and reported an average total (sum over all congeners) concentration of 93,000 to 650,000 pg/m^3 (values dependent on presence/use of a shredder on the day of sampling). Soil samples from e-waste sites in China revealed phthalate concentrations ranging from 12.6 to 46.7 mg/kg (Liu et al., 2009). Biomonitoring studies revealed that the median total PBDE blood level in Swedish e-waste workers was seven times greater than a reference group consisting of hospital workers, white collar workers, residents from nearby regions, and circuit board producers (Schecter et al., 2009). These observations reflect the heavy use of SVOCs as additives and reactants in many electronic products.

The high levels of SVOCs in e-waste materials being brought to recycling locations (Table 6) likely means that such facilities are suitable environments to compare methods of indoor surface sampling. In light of the expected high concentrations of SVOCs (PBDEs in particular) at e-waste facilities, difficulties associated with non-detects should be avoidable.

Table 6: Dust concentrations at ewaste facilities cited in recent literature.

Compound	n	Central tendency		Variability		Particle size (µm)	Location [a, b]	Collected by [c]	Collection Method	Reference
		Measure	Value (µg/g)	Measure	Value (µg/g)					
PBDE-47	10	median	0.0067	range	(0.0018, 0.041)	-	M, TM	R	straw broom	Tue et al., 2013
PBDE-47	10	median	0.16	range	(0.0076, 0.670)	-	M, BD	R	straw broom	Tue et al., 2013
PBDE-47	13	median	0.175	range	(0.0474, 1.230)	<500	M	R	brush collection	Zheng et al., 2011
PBDE-47	5	mean	0.31	range	(0.0698, 0.531)	-	F	-	-	Ma et al., 2009
PBDE-99	10	median	0.0097	range	(0.0025, 0.057)	-	M, TM	R	straw broom	Tue et al., 2013
PBDE-99	10	median	0.15	range	(0.0071, 0.770)	-	M, BD	R	straw broom	Tue et al., 2013
PBDE-99	13	median	0.352	range	(0.0835, 2.510)	<500	M	R	brush collection	Zheng et al., 2011
PBDE-99	5	mean	0.272	range	(0.0242, 0.511)	-	F	-	-	Ma et al., 2009
PBDE-153	10	median	0.0055	range	(0.0015, 0.021)	-	M, TM	R	straw broom	Tue et al., 2013
PBDE-153	10	median	0.056	range	(0.0047, 0.560)	-	M, BD	R	straw broom	Tue et al., 2013
PBDE-153	13	median	0.12	range	(0.0277, 0.378)	<500	M	R	brush collection	Zheng et al., 2011
PBDE-153	5	mean	0.0619	range	(0.0228, 0.107)	-	F	-	-	Ma et al., 2009
PBDE-209	11	mean	42.3	range	(0.910, 320.4)	< 100	EP	R	vacuum, filter	Li et al., 2013
PBDE-209	2	mean	20.6	range	(13.900, 27.400)	< 100	F	R	straw broom	Li et al., 2013
PBDE-209	10	median	0.3	range	(0.030, 1.700)	-	M, TM	R	straw broom	Tue et al., 2013
PBDE-209	10	median	0.86	range	(0.069, 7.900)	-	M, BD	R	straw broom	Tue et al., 2013
PBDE-209	13	median	9.54	range	(1.900, 23.500)	<500	M	R	brush collection	Zheng et al., 2011
PBDE-209	5	mean	29.8	range	(5.560, 80.600)	-	F	-	-	Ma et al., 2009

- not stated in reference

[a] Location of sample collection: F: floor surfaces, A: above floor surfaces, M: multi-surface (includes floor and above floor surfaces), EP: electronic products to be recycled

[b] Location specified if analyzed at multiple locations in the same study: TM: Trang Minh, BD: Bui Dau

[c] Sample collected By: R: researcher, O: occupant

Chapter 3

Methods

Study Design

The goal of the study was to collect horizontal and vertical surface samples containing SVOCs in order to compare paired vacuum-collected and wipe-collected samples. The five side-by-side vertical and horizontal indoor surfaces were established by dividing a demarcated area in half to obtain equal areas for vacuum and dust wipe samples. The sample areas were delineated with masking tape or, if space permitted, by utilizing a 0.25-m^2 template. The five vertical sample sites in the warehouse included a dock door panel, shipping and receiving office window, back panel of the baler, interior of the baler door, and a metal bin used for typical storage. The five horizontal surface sample areas inside the warehouse consisted of the top of a refrigerator, a television screen, and three warehouse floor locations (by the baler, storage bin and near shipping and receiving). The sampling sites were distributed throughout the facility. For the wipe methods, the number of samples collected varied depending on the visual observation of the load on the surface. The vacuum collected a single sock sample for each location (see Table 7).

Table 7: Methods summary for first sample round

Method	Orientation	Number of Locations	Number of Repetitions	Number of Extractions	Total Samples (n)
wipe	horizontal	5	1 - 3	1	11
	vertical	5	1 - 3	1	10
vacuum	horizontal	5	1	1	5
	vertical	5	1	1	5

Sampling Site Description

The sampling site is an electronics waste (e-waste) processing facility participating in the Washington E-Cycle program. The facility is approximately 27,000 square feet, located in the greater Puget Sound area (see Appendix, Figure 16). The major operations include the shipping and receiving process, sorting and manual dismantling of electronic products, and baling plastics. The products recycled at the facility include televisions, monitors, computers, laptops, audio-video equipment, computer peripherals, and other common consumer electronics. Workers at the facility perform mechanical size reduction. For example, televisions and monitors are separated from the plastic casing and the cathode ray tube (CRT) is stripped of metals and wiring. Most other electronics are dismantled in a similar manner by removing all of the components and separating them according to its commodity (plastics, metals, printed circuit boards, etc.). In addition to disassembling the products, plastics are crushed and baled on site. There are 17 employees, comprised of 4 administrative and/or office positions, 2 IT technicians, a driver, a baler operator, 4 sorters, and 5 dismantlers. There is only one 8-hour work shift from 8 am to 5 pm and the facility operates 5 days of the week (Monday through Friday).

Dust Wipe Sampling Methods

The dust sampling method that was used is similar to NIOSH method 9102, except anhydrous isopropanol alcohol (Alfa Aesar, >99.7%, GC Grade) and sterile cotton gauze dressing pads (InvaCare, 4”x4”) were employed. The sampled areas were delineated by a 0.25-m^2 template or masking tape. In the first round of sampling each gauze pad was pre-moistened with 4-mL of isopropanol alcohol and then used to wipe an area using overlapping horizontal and vertical S-patterns to ensure that every space in the demarcated area was wiped twice (see Figure 1). After the area was sampled, the pads were folded with the contaminated side on the interior and placed into a pre-labeled sealable bag (Ziploc®). When multiple wipes were used for a single sample location, the wipes were placed into the same Ziploc® bag. The number of gauze pads used was dependent on the visual observation of surface load. Immediately following completion of sampling, all wipe samples were transported back to the laboratory and placed into a -20°C freezer.

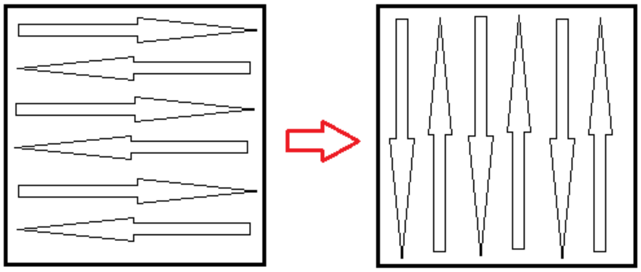

Figure 1: Dust collection schema for vacuum and wipe methods

Vacuum Sampling Methods

The vacuum dust samples were gathered immediately adjacent to the wipe samples using a portable HEPA vacuum cleaner (Atrix International, Inc., Omega Plus SN 44319). Individual samples were collected using a dedicated custom-made 10-inch copper tube (connected to the vacuum cleaner's 6-foot hose) and filter sock (Midwest Filtration X-Cell 100 Dust Sampling Sock). The filter sock was inserted inside the copper tube and held in place with a 45° copper elbow serving as the dust collection point (see Figure 2). Prior to sampling, all copper attachments were cleaned with soap and water, allowed to dry completely, and then rinsed thoroughly with dicholoromethane (BDH, >99.5%). Each set of vacuum implements (*i.e.*, copper tube and elbow, and filter sock) were placed separately into plastic Ziploc® bags and sealed until sampling time.

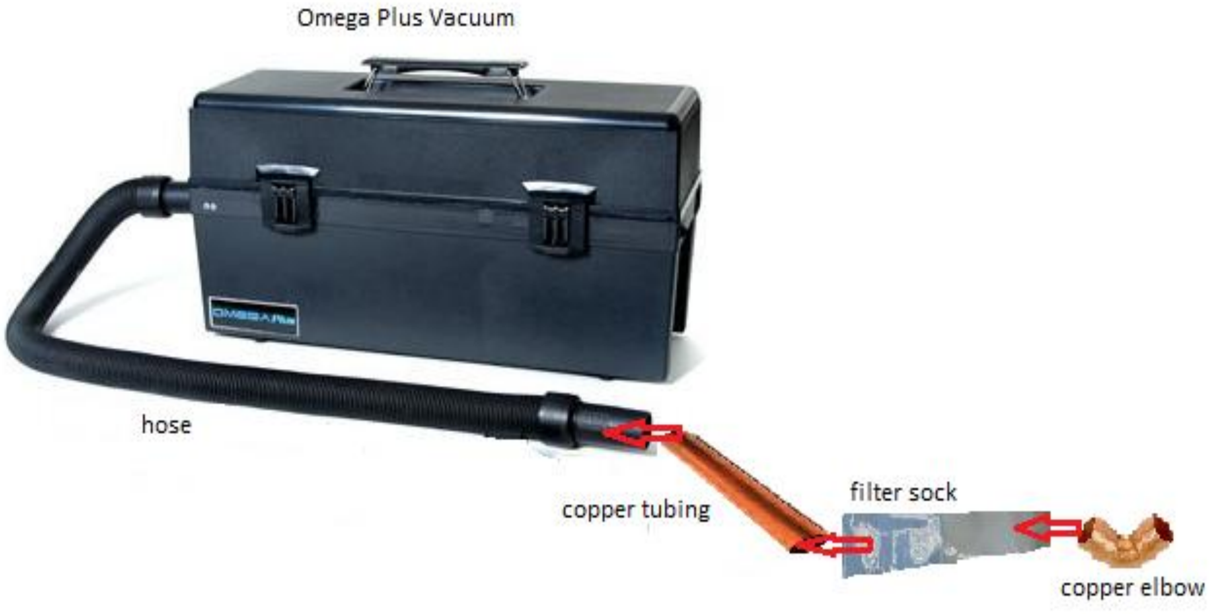

Figure 2: Vacuum sampling device

Vacuuming occurred within the marked area in a sequential manner (left to right, top to bottom, see Figure 1). At completion, the copper elbow was detached and the dust sampling sock was removed carefully and placed upright in a pre-labeled Ziploc® bag to avoid loss of collected material. All filter sock samples were transported back to the laboratory and stored upright in a -20°C freezer until analysis. Copper tubes and elbows were cleaned as described above prior to their reuse.

Laboratory Analysis

All samples were extracted and analyzed at the University of Washington's Environmental Health Laboratory (EHL). The vacuum sock sample extractions were carried out in a microwave reaction system (MARSXpress™ CEM Corporation), with a maximum power output of 1600-W. The microwave-assisted extraction (MAE) method employed was similar to U.S. EPA Method SW846-3546, where microwave energy is used to produce an environment with elevated pressure and temperature in a closed vessel to recover analytes. The vacuum sock samples were prepared by cutting and removing the plastic collar from the filter sock (see Figure 3) and placed into a 75-mL teflon vessel, where 20-mL of acetone/hexane (50/50) was added to immerse the sock into the solvent. The sample was subjected to microwave treatment at a controlled temperature of 110°C (heating ramp-up time was 10 minutes) and held for 10 minutes. A concentrated sample extract was analyzed by using gas chromatography/mass spectrometry (GC/MS) (Agilent 5973N MS, 7890A GC, 7693 Autosampler System) in the negative chemical ionization (NCI) mode using anhydrous ammonia as the reagent gas.

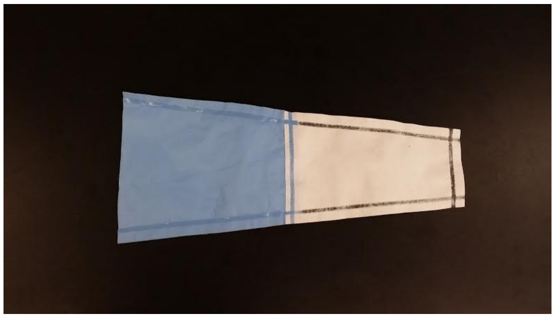

Figure 3: Vacuum sampling sock

The wipe samples were extracted using sonication methods. The wipe samples were individually transferred to a 50-mL polypropylene centrifuge tube and 20-mL of acetone/hexane (50/50) was added in order to completely submerge the wipe into the desorption solvent. In addition, the loose material remaining in the Ziploc® bag containing the wipes was carefully transferred to the tubes. The sample was vortexed for 1 minute and then sonicated for 60 minutes at 40°C. The extracts were reduced in volume to 0.5-mL and then solvent exchanged to toluene with a final volume of 1-mL. Finally, the extracts were filtered (PTFE, 0.45 µm) to remove any suspended particles prior to the GC/MS analysis. The PBDEs, TBBPA, and TDCPP were analyzed by using GC/MS in the NCI mode, whereas the phthalates were quantified by GC/MS in electron ionization (EI) mode.

Data Analysis

Prior to any data analysis, the analytical results for the vacuum and wipe samples were corrected for reagent blank values and matrix blank values, respectively. All data were blank corrected by subtracting the mean of the respective field blanks. In addition, a subset of vacuum and wipe samples had poor response peaks from the GC analysis requiring further correction using internal standard recovery efficiencies.

Contaminant loadings (ng/cm^2) were calculated for each SVOC for wipe and vacuum sock samples from horizontal and vertical surfaces. Ratios of wipe-to-filter sock samples were then computed (separately for horizontal and vertical surface) to observe how collocated wipe and vacuum samples compared:

$$\frac{\frac{wipe\ contaminant\ mass\ (ng)}{wipe\ sample\ area\ (cm^2)}}{\frac{vacuum\ contaminant\ mass\ (ng)}{vacuum\ sample\ area\ (cm^2)}} = \frac{loading_{wipe}}{loading_{vacuum}} = Ratio \qquad \text{(eq. 3)}$$

In the instances where a wipe or filter sock sample was not quantified due to analytical limitations, the ratio was not calculated.

Additionally, statistical analysis was performed using paired t-testing procedures with a statistical significance, alpha (α), set at 0.05. The null hypothesis was that the mean of the two paired samples were equal. The paired t-test analyses were carried out using Microsoft® Excel 2007. Pearson correlation coefficients were calculated to establish associations between continuous measures of vacuum and wipe samples using STATA® (version 11.2). The limited number of paired quantified samples (n = 3 to 5) made it difficult to perform tests that would ensure normality or lognormality for the parametric paired t-tests, so non-parametric Wilcoxon rank sum tests were also performed to evaluate whether the difference of the contaminant loadings from the wipe and sock was due to chance and to conclude if the means of the paired samples were different. The Wilcoxon rank sum tests were performed using Stata® (version 11.2).

Second Sampling Event: Post-hoc Analysis

A second round of sampling was performed *post-hoc* to obtain a better understanding of the sample collection methods and sample analysis techniques. The wipe and vacuum samples were collected using the 0.25-m^2 template from juxtaposed positions on the floor near the southwest end of the warehouse (see Table 8) using methods described above (with the exception that wipe samples utilized 5 ml of anhydrous isopropanol rather than 4 ml).

Two of four quadrants were sampled initially by vacuum. After vacuuming, a single pre-moistened wipe was used to sample remaining material. The vacuum samples were analyzed using two different extraction methods – microwave assisted or sonication - to assess potential differences in extraction efficiencies. The two remaining quadrants were sampled by three sequential wipes without pre-vacuuming. Each sequential wipe sample was bagged and numbered separately to permit evaluation of successive sampling performance. (Only one set of three wipes was subsequently analyzed.) Each of the vacuum and wipe samples that was analyzed was extracted four times (rather than once as in the first round) to assess if a single extraction was adequate and to evaluate the number of extractions required to attain a more representative value for the contaminant mass collected. The first three extractions were performed using equal parts acetone and hexane, and the final extraction used a more aggressive solvent – dichloromethane.

Table 8: Methods summary for second sample round

Method	Orientation	Number of Locations	Number of Repetitions	Extraction Method(s)	Number of Extractions	Total Samples (n)
vacuum	horizontal	2	1	MAE or S [a]	4	8
wipe	horizontal	1	3	S	4	12
		2[*]	1	S	4	8

[*]: One single wipe was used following the vacuum sampling sequence.
a: One sample was extracted using MAE and one sample was extracted using sonication
Extraction Methods: MAE: microwave-assisted extraction, S: sonication

Chapter 4

Results

First Round of Sampling

The contaminant loadings (ng/cm^2) for SVOCs in the vacuum and surface wipe samples, calculated by dividing the measured mass by the area sample, and the means of the loadings are presented in Table 9. Some of the PBDE samples were not quantifiable due to low recovery of the internal standards. Generally, PBDE-209 contaminant loadings in horizontal and vertical surfaces were several orders of magnitude higher relative to other sampled compounds, except TBBP-A for horizontal surfaces. Phthalate data are not included in the reported results because analytical difficulties were so problematic that those results were deemed unreliable.

Table 9. Contaminant loadings for horizontal and vertical surfaces sampled

		Horizontal Surface Loadings (ng/cm2)			Vertical Surface Loadings (ng/cm^2)	
		Vacuum	Wipe		Vacuum	Wipe
PBDE-47	Refrigerator surface	2.3	0.27		0.035	0.01
	Concrete 1	0.83	0.18		0.36	0.036
	Concrete 2	0.48	0.1		0.047	0.025
	Concrete 3	-	0.1		-	0.19
	Glass	0.2	0.08		0.022	0.014
	mean	***0.94***	***0.14***		***0.12***	***0.055***
PBDE-99	Refrigerator surface	0.12	0.46		0.024	0.012
	Concrete 1	-	0.18		0.01	0.057
	Concrete 2	0.009	0.1		0.054	0.052
	Concrete 3	-	0.08		0.004	0.041
	Glass	0.3	0.17		0.02	0.02
	mean	***0.14***	***0.2***		***0.022***	***0.039***
PBDE-153	Refrigerator surface	0.07	0.13		0.007	0.002
	Concrete 1	0.025	0.06		0.01	0.019
	Concrete 2	0.012	0.043		0.015	0.014
	Concrete 3	-	0.05		0.13	0.17
	Glass	0.036	0.042		0.014	0.015
	mean	***0.036***	***0.065***		***0.035***	***0.044***
PBDE-209	Refrigerator surface	1964	104		19.6	1.9
	Concrete 1	2499	66		4432	9.7
	Concrete 2	315	65.8		269	14.7
	Concrete 3	3205	82.1		1876	77.6
	Glass	676	22.3		89.5	16.5
	mean	***1732***	***68***		***1337***	***24.1***
TBBP-A	Refrigerator surface	3	43.8		0.06	0.73
	Concrete 1	43	70.1		4.8	5.1
	Concrete 2	14.4	88.6		0.19	2.4
	Concrete 3	62.2	77.7		7.1	48.8
	Glass	14.2	5.6		0.12	3.8
	mean	***27.4***	***57.2***		***2.5***	***12.2***
TDCPP	Refrigerator surface	2.7	2		0.1	0.05
	Concrete 1	1.5	1.3		0.5	0.28

		Horizontal Surface Loadings (ng/cm2)			Vertical Surface Loadings (ng/cm^2)	
	Concrete 2	1.4	0.81		0.18	0.09
	Concrete 3	1.7	1.1		1.3	0.31
	Glass	0.58	0.44		0.069	0.07
	mean	***1.6***	***1.1***		***0.42***	***0.16***

- : sample not quantifiable due to analytical limitations

Using the contaminant loadings, ratios were calculated (eq. 3) to assess if there was a difference in the mass collected by wipe vs. vacuum (filter sock) and to identify the directionality and magnitude of the difference. If the ratio value was greater than one, it implied that wipes collected more contaminant mass than vacuuming, while a ratio of one signifies no difference between the methods. The mean ratios from the horizontal and vertical samples suggested three compounds supported the hypothesis that surface wipes would have a greater loading than vacuum methods – PBDE-99, PBDE-153 and TBPPA (see Table 10 and 11). Although PBDE-153 had greater loadings in the vertical samples, the result was less conclusive since the ratio was so close to one ($\text{ratio}_{\text{PBDE-153(vertical)}}$ = 1.1).

Table 10: Wipe vs. sock contaminant loading ratios for horizontal samples

Horizontal WIPE/FILTER SOCK contaminant loading ratios						
surface type	**PBDE-47**	**PBDE-99**	**PBDE-153**	**PBDE-209**	**TBBPA**	**TDCPP**
Refrigerator surface	0.12	3.8	1.8	0.053	14.7	0.73
Concrete 1	0.21	--	2.3	0.026	1.6	0.88
Concrete 2	0.20	11.2	3.6	0.21	6.1	0.59
Concrete 3	--	--	--	0.026	1.3	0.61
Glass	0.40	0.57	1.2	0.033	0.39	0.76
mean	**0.23**	**5.2**	**2.2**	**0.069**	**4.8**	**0.71**
standard deviation	**0.12**	**5.5**	**1.0**	**0.079**	**6.0**	**0.12**

- - : samples not quantified due to analytical limitations

Table 11: Wipe/sock contaminant loading ratios for vertical samples

	pbde 47	**pbde 99**	**pbde 153**	**pbde 209**	**TBBPA**	**TDCPP**
Plexiglass	0.30	0.49	0.31	0.10	12.7	0.52
Glass	0.10	5.5	1.9	0.002	1.1	0.56
Painted metal 1	0.52	0.96	0.91	0.055	12.9	0.52
Painted metal 2	--	9.6	1.3	0.041	6.8	0.24
Painted steel	0.63	1.0	1.1	0.18	32.1	0.94
mean	**0.39**	**3.5**	**1.1**	**0.076**	**13.1**	**0.56**
standard deviation	**0.24**	**4.0**	**0.58**	**0.068**	**11.7**	**0.25**

- - : samples not quantified due to analytical limitations

Summary statistics for SVOCs in vacuum and surface wipe samples are presented in Table 12. A two-tailed paired t-test was performed on the wipe and dust samples to determine whether there was a statistically significant ($\alpha = 0.05$) difference in means between the total contaminant loadings of the surface wipe samples compared to the vacuum collected samples.

The paired t-tests revealed significant differences in the means of PBDE-209 (n = 5; p = 0.036)) and TDCPP (n = 5; p = 0.018) from the horizontal samples only.

The Pearson correlation coefficient results presented in Table 12 reflect a positive correlation between vacuum and wipe; that is, the extent to which higher contaminant mass in vacuum samples is associated with greater surface wipe contaminant mass. A high degree of correlation was observed for five out of six of the SVOCs on the horizontal surfaces, with correlation coefficients ranging from 0.47 to 0.96. The exception was PBDE-99 ($r = 0.048$). However, TDCPP was the only compound for which the correlation was statistically significant ($p < 0.01$). The lack of the statistical significance for the other compounds, in spite of the large correlation coefficients, likely reflects the small sample size for this study. Small correlation coefficients were found for PBDE-99 ($r = 0.10$) and PBDE-209 ($r = 0.14$) for the vertical samples. Whereas the other studied SVOCs exhibited stronger correlations, with correlation coefficients ranging from 0.82 to 1.0. PBDE-153 had the largest coefficient ($r = 1.0$) and was the only compound showing a statistically significant correlation between vertical vacuum and wipe samples.

A non-parametric statistical analysis, the Wilcoxon sum rank test, was also performed since normality of the underlying data was not certain for the paired t-test or Pearson's correlation. The Wilcoxon sum rank test indicated that there was a significant difference in means of the wipe and vacuum samples for horizontal and vertical samples of PBDE-209, vertical samples of TBBPA, and horizontal samples of TDCPP. The only compounds for which a statistically significant difference was found by both the paired t-test and Wilcoxon rank sum test were PBDE-209 and TDCPP (horizontal samples only) and those results indicated lower loading by wipe than by vacuum.

Table 12: Results of statistical tests comparing wipe vs. dust samples

	Horizontal Samples					Vertical Samples			
	Paired t-test	**Pearson's Correlation**		**Wilcoxon Rank Sum Test**		**Paired t-test**	**Pearson's Correlation**		**Wilcoxon Rank Sum Test**
	p-value	*coefficient*	*p-value*	*p-value*		*p-value*	*coefficient*	*p-value*	*p-value*
PBDE-47	0.15	0.87	0.13	0.068		0.3	0.86	0.14	0.068
PBDE-99	0.54	0.048	0.97	0.59		0.3	0.1	0.87	0.59
PBDE-153	0.06	0.90	0.1	0.068		0.34	1.0	0.0002	0.42
PBDE-209	0.036	0.54	0.35	0.043		0.19	0.14	0.82	0.043
TBBPA	0.096	0.47	0.43	0.08		0.29	0.82	0.087	0.043
TDCPP	0.018	0.96	0.0094	0.043		0.22	0.88	0.051	0.08

Post-hoc Sampling Event

The post-hoc sampling round provided insight into the limitations of the results from the first round of sampling. Each of the samples was collected with the 0.25-m^2 sampling frame on the floor of the same facility for comparability. This sampling opportunity was used to further evaluate elements of the field sampling protocol (sequential wipes and post-vacuum wiping) and chemical analyses (sequential extractions and comparison between microwave-assisted and sonication extraction methods). Table 13 displays the total SVOC contaminant masses (blank corrected) for each sample extraction.

Table 13: Contaminant mass for each extraction from vacuum and wipe samples.

	PBDE-47	**PBDE-99**	**PBDE-153**	**PBDE-209**	**TBBPA**	**TDCPP**
Sample Name	*ng*	*ng*	*ng*	*ng*	*ng*	*ng*
V1_ex-1[a]	43.5	59.6	43.0	192491	178445	6931
V1_ex-2[a]	32.4	47.0	55.1	205295	97649	2821
V1_ex-3[a]	67.1	105.1	59.0	60643	86367	1918
V1_ex-4[a]	4.6	8.8	14.2	11081	8173	870.
Total	**147.6**	**220.4**	**171.2**	**469510**	**370635**	**12539**
VW1_ex-1	24.9	32.4	10.8	5948	5351	503
VW1_ex-2	8.3	13.1	7.8	3022	9705	533
VW1_ex-3	4.8	7.7	6.3	1266	1603	68
VW1_ex-4	1.2	2.1	0.0 [c]	156	370	0.7
Total	**39.1**	**55.3**	**25.0**	**10392**	**17028**	**1105**
V2_ex-1	22.4	0.0 [b]	29.8	433843	172598	13813
V2_ex-2	31.5	51.8	44.2	105095	74049	7696
V2_ex-3	11.0	21.2	18.0	25014	13992	417
V2_ex-4	1.1	3.0	5.6	1488	5326	135
Total	**66.0**	**76.1**	**97.6**	**565440**	**265964**	**22062**
VW2_ex-1	11.3	18.2	7.5	9010	5362	513
VW2_ex-2	8.0	12.6	6.4	2326	1609	780
VW2_ex-3	3.8	4.9	0.0 [c]	1258	1566	73
VW2_ex-4	1.2	2.1	0.0 [c]	205	387	0[c]
Total	**24.3**	**37.7**	**13.9**	**12800**	**8924**	**1366**
W1_ex-1	19.5	28.6	34.1	220408	130653	3426
W1_ex-2	12.0	21.5	24.3	71749	18711	1672
W1_ex-3	5.9	11.0	13.4	20028	17970	324
W1_ex-4	2.4	4.2	6.8	7964	3544	288
Total	**39.8**	**65.2**	**78.6**	**320150**	**170877**	**5710**
W2_ex-1	24.4	31.0	19.6	91479	14783	2698
W2_ex-2	8.9	16.3	10.6	16835	10515	192
W2_ex-3	5.1	8.2	8.6	5725	1439	453
W2_ex-4	1.5	2.8	0.0 [c]	1475	1236	50
Total	**39.8**	**58.2**	**38.7**	**115514**	**27973**	**3393**
W3_ex-1	6.7	8.6	11.8	23430	28830	87
W3_ex-2	4.4	8.0	7.4	4597	2815	473
W3_ex-3	2.3	4.4	5.4	2355	2215	464
W3_ex-4	1.1	2.2	4.2	505	746	49
Total	**14.5**	**23.3**	**28.9**	**30887**	**34607**	**1073**

Sample Name: V: vacuum method, VW: wipe after vacuum, W: successive wipe samples from the same location; ex: extraction number

[a] : microwave assisted extraction (all other samples were sonicated)

[b]: unable to integrate target peak due to large unresolved matrix artifact peak

[c]: zero mass due to correction from blank

Sequential Wipes Collection

A total of three surface wipes were collected in one sampling location to evaluate the collection method. When analyzing the compounds separately, the general trend is that after each wipe, the mass decreases (see Figure 4). The two exceptions of this observation are PBDE-47 and TBBP-A.

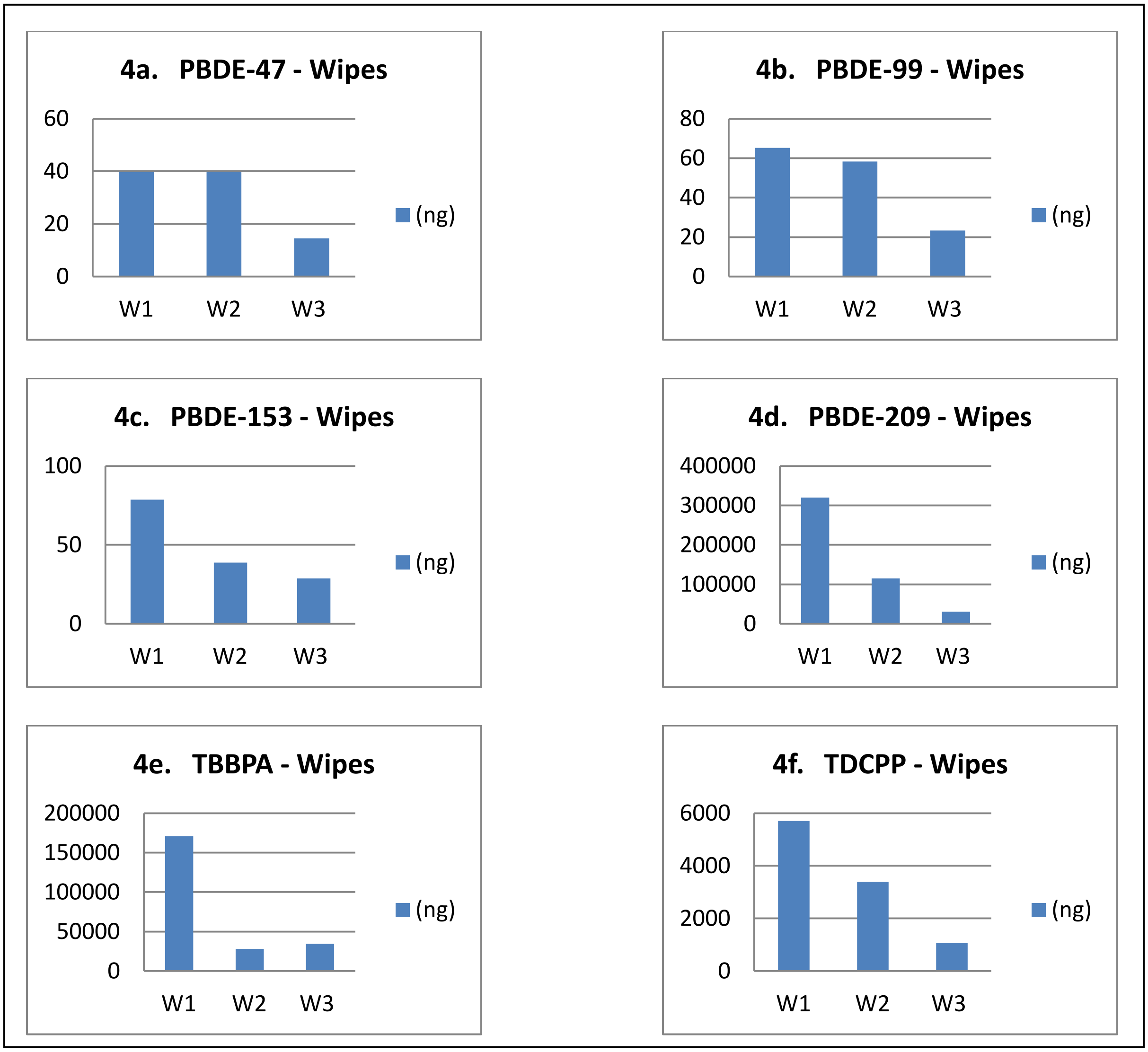

Figure 4 (a-f): SVOC contaminant mass (ng) in successive wipe samples

Post-vacuum Wipes

Surface wipes were acquired immediately following each of the vacuum sample collections. The graphs in Figure 5 and 6 show that when the surfaces were wiped subsequent to the vacuuming, there was still residual mass left on the floor (vacuum and wipe samples shown in Figure 5 underwent microwave and sonication extractions, respectively, while results in Figure 6 all were extracted using sonication methods). The noteworthy compounds displaying substantial remaining masses are PBDE-47 and PBDE-99 in both samples.

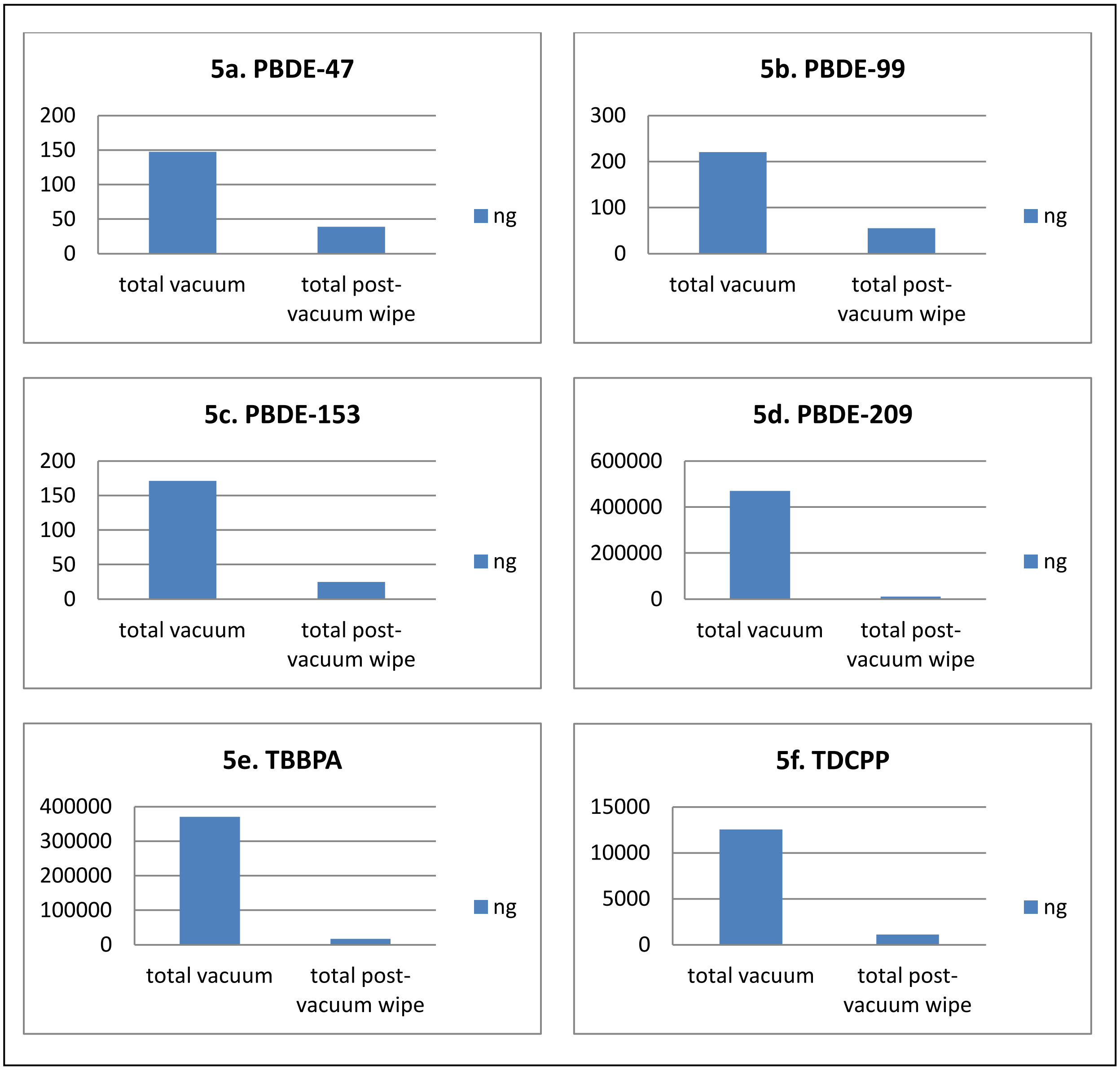

Figure 5 (a-f): Comparison of vacuum and post-vacuum wipe contaminant mass (ng) for V1

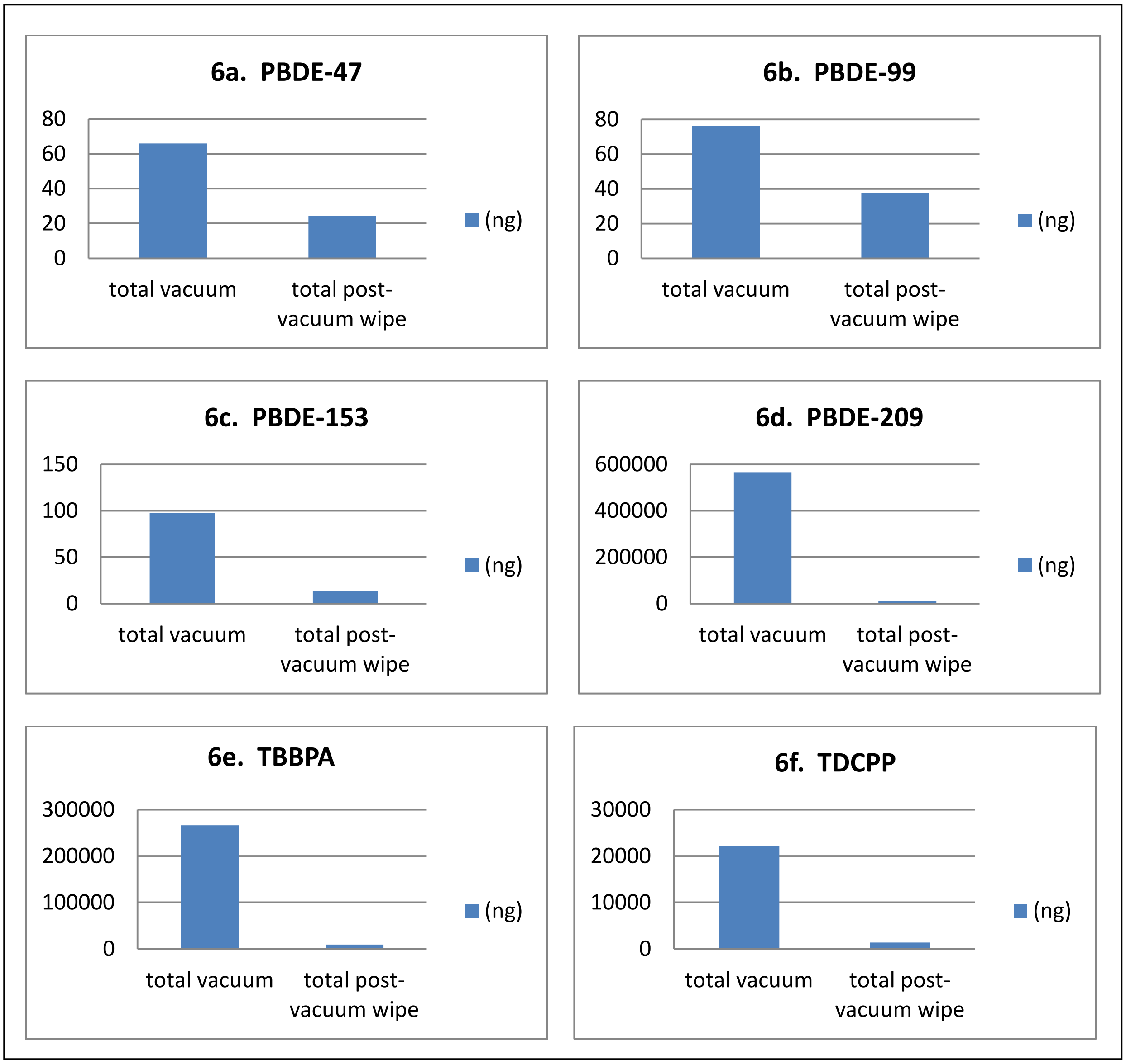

Figure 6 (a-f): Comparison of vacuum and post-vacuum wipe contaminant mass (ng) for V2

Microwave vs. Sonication Extraction

The two vacuum samples (V1 and V2) were extracted using microwave extraction or sonication methods, whereas all the wipe samples were extracted using sonication methods. Figures 7 and 8 demonstrate the different efficiencies of the extraction methods. The compounds with lower masses (< 250-ng) in Figure 7 appear to have greater extraction efficiencies using the microwave extraction methods compared to the sonication methods. Alternatively, when the compounds had larger masses (> 10-µg), the more effective extraction method is not as apparent due to the inconsistencies observed in Figure 8. For example, the microwave extraction was more efficient for TBBP-A, however for PBDE-209 and TDCPP, the favorable method was sonication.

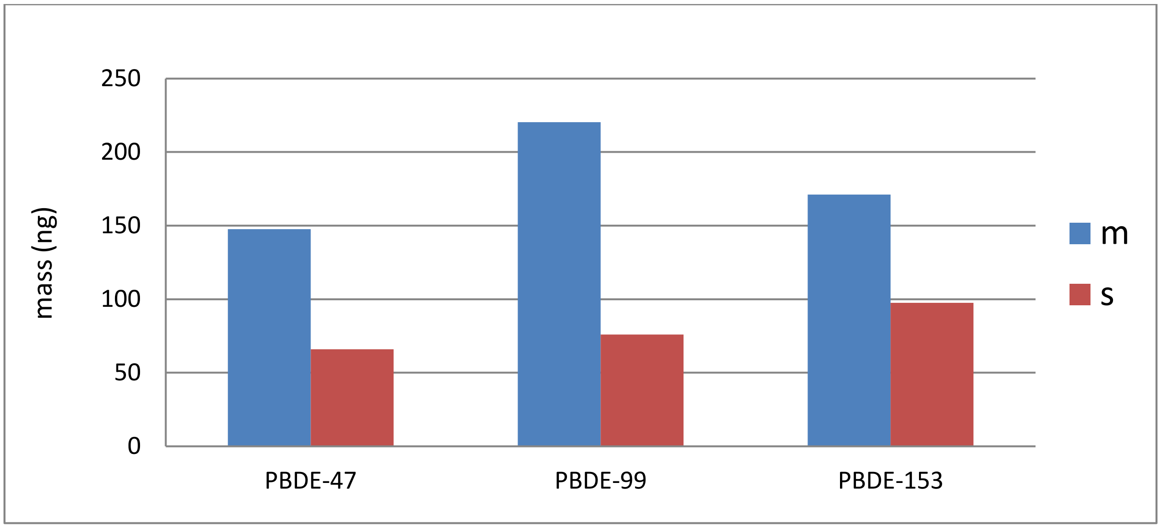

Figure 7: Comparison of microwave and sonication extraction methods for compounds with lower contaminant mass (< 250 ng) in vacuum samples

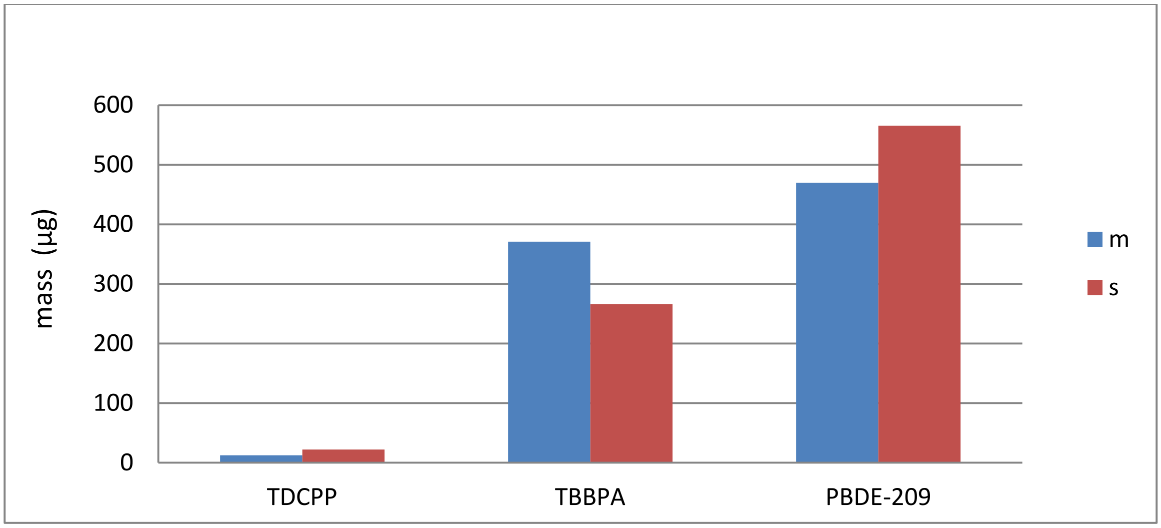

Figure 8: Comparison of microwave and sonication extraction methods for compounds with higher contaminant mass (>10 µg) in vacuum samples

Sequential Extraction: Vacuum and Wipe Samples

The vacuum and wipe samples were extracted four times each. Overall, the microwave methods for the vacuum samples show an expected decline of mass after each succession (e1, e2, e3, and e4). Note that the y-axis scale is logarithmic in Figures 9 to 15. Conversely, PBDE-47 and PBDE-99 increased by almost a factor of two from second extraction to the third (Figure 9). The sonication extracted vacuum and wipe samples show an overall decline of mass (see Figures 10 – 15), but with some exceptions for individual samples and compounds. To further explore irregularities or trends, the internal standards were graphed (see Appendix), however, the examination of the internal standard analytes did not reveal any additional information.

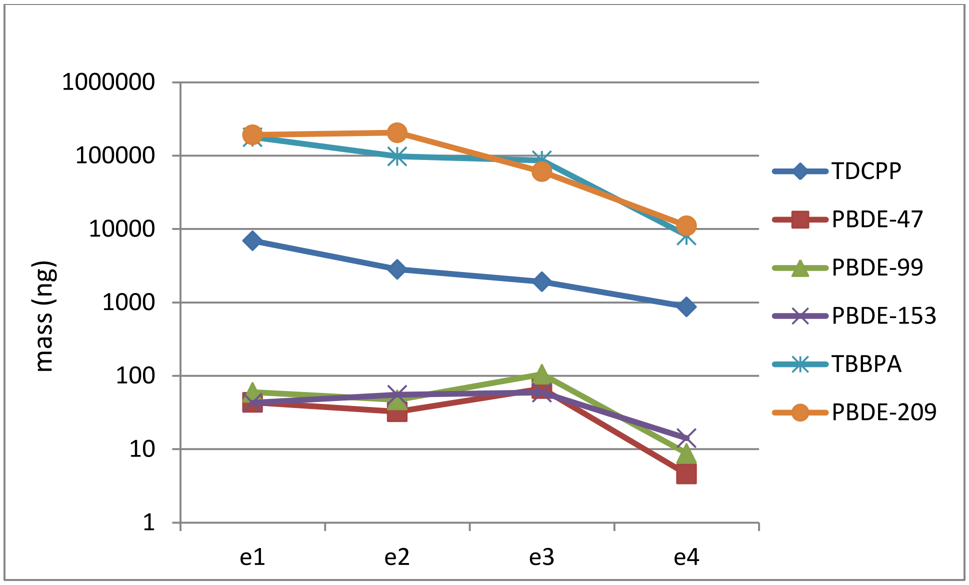

Figure 9: SVOC masses yielded from serial microwave extraction of *post-hoc* vacuum sample V1

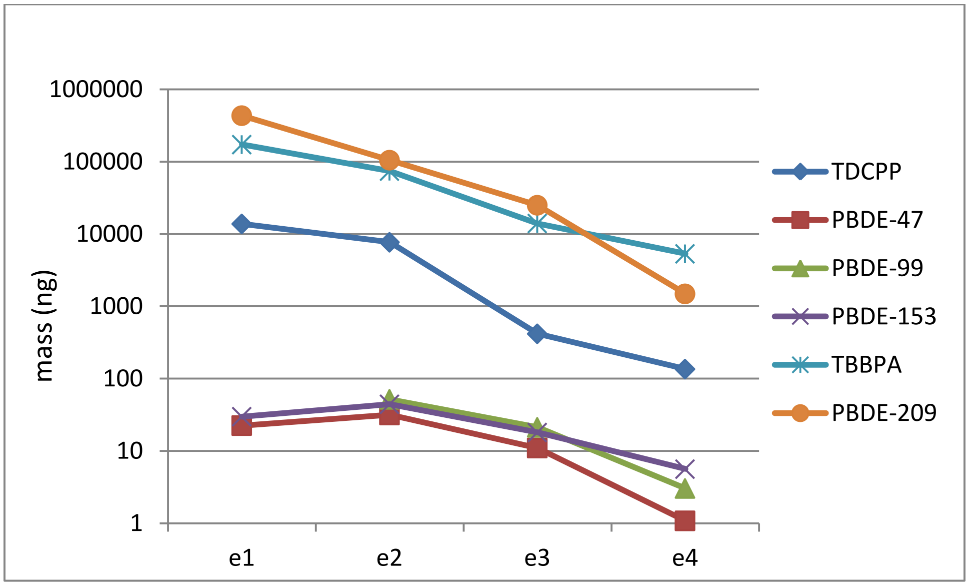

Figure 10: SVOC masses yielded from serial sonication extraction of *post-hoc* vacuum sample V2

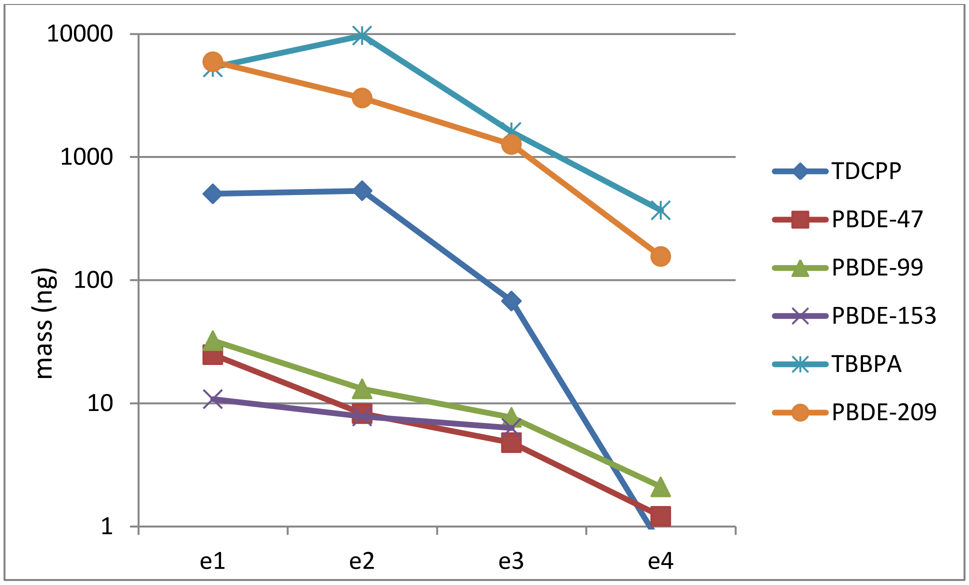

Figure 11: SVOC masses yielded from serial sonication extraction of *post-hoc* post-vacuum wipe sample VW1

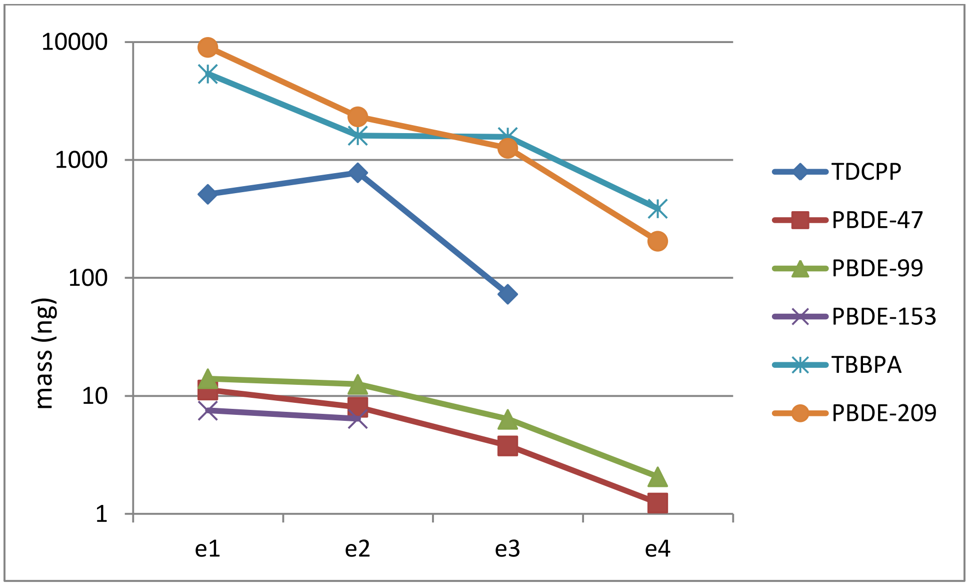

Figure 12: SVOC masses yielded from serial sonication extraction of *post-hoc* post-vacuum wipe sample VW2

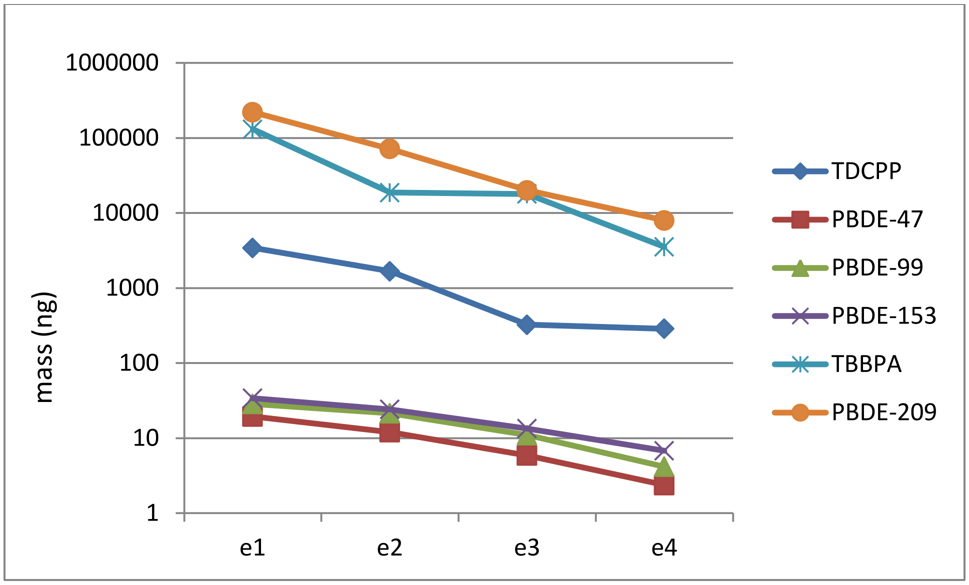

Figure 13: SVOC masses yielded from serial sonication extraction of *post-hoc* wipe sample W1

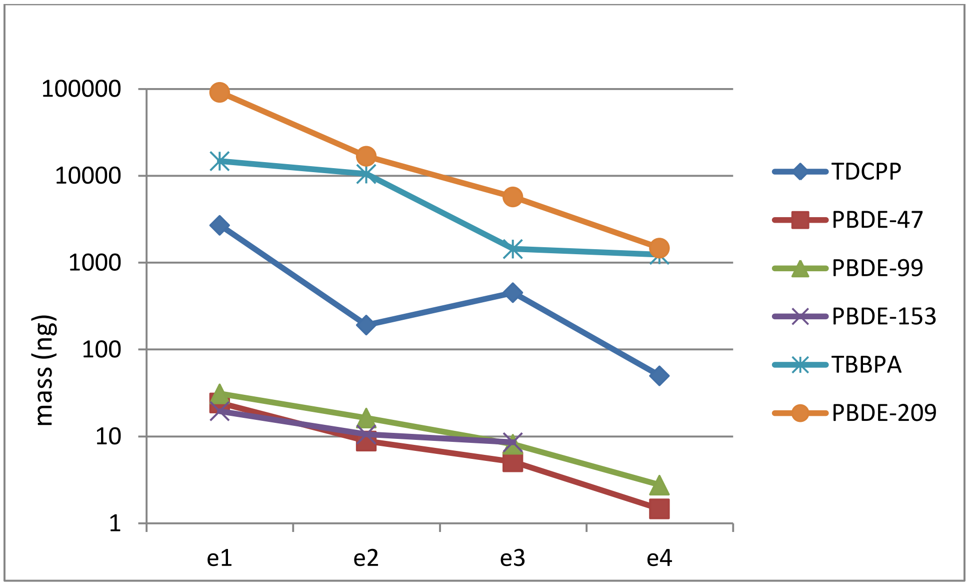

Figure 14: SVOC masses yielded from serial sonication extraction of *post-hoc* wipe sample W2

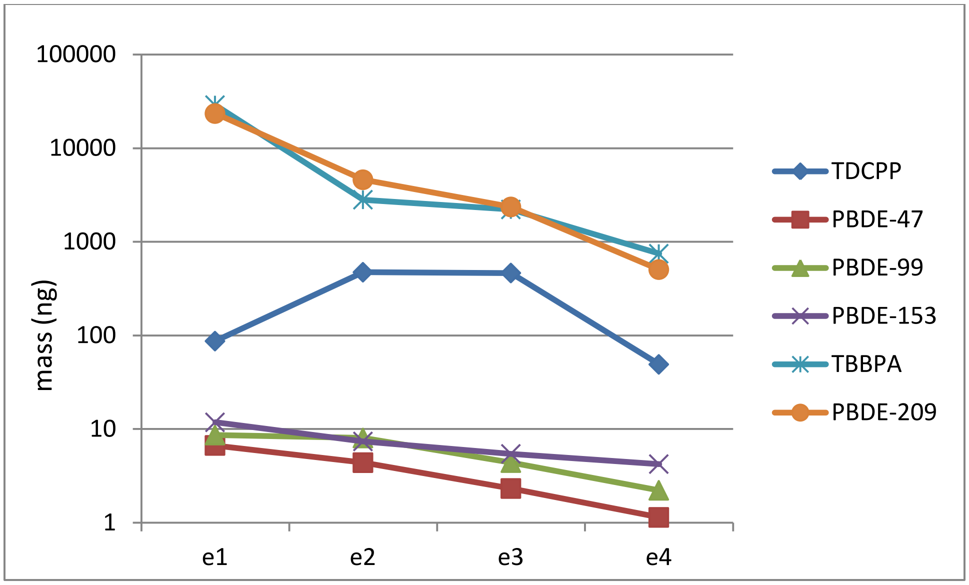

Figure 15: SVOC masses yielded from serial sonication extraction of *post-hoc* wipe sample W3

Chapter 5

Discussion

The data from this study represent measured surface contaminant loadings for four PBDEs and two other flame retardants at an e-waste facility in Washington state as obtained by two alternative methods, wipe and vacuum. The surface samples were also analyzed for three phthalates, but those results are not included here due to disqualifying analytical difficulties. The results of this work are the only known surface sampling data set for collocated SVOC wipe and vacuum samples. A *post-hoc* sampling round was also conducted to provide additional insight into both the sample collection methods and chemical extraction protocols.

Relationship Between Surface Wipes and Vacuum Samples

The results from the study failed to consistently find higher surface loads by wipe than by vacuum. Although there were several wipe and vacuum ratios that concurred with the original hypothesis, overall results did not support the expectation of greater mass recovery by wipe than by vacuum. The paired t-tests and Wilcoxon rank sum test provided few statistically significant results, other than PBDE-209 which had consistent and significant results in both tests (but in the opposite direction expected). The results of the Pearson's correlation determined that 75% (9 of 12 comparisons) of the horizontal and vertical surfaces had relatively large correlations (> 0.47), indicating that higher SVOC loadings in vacuum samples are associated with higher contaminant loadings in surface wipes.

The inconsistencies in the results may be attributable to the constraints from the study design, methods and analytical protocols. The purpose of sampling at an e-waste facility was to ensure a target-rich environment for SVOCs for analytical purposes. However, the total SVOC content of the samples provided many interfering peaks, which the chemical complicated analyses. In addition, the particularly high SVOC contaminant loads in the vacuum samples suggest the possibility of atypically high dust loads relative to the thin organic film expected (previously estimated to be 29 – 250 nm thick). The high contaminant loadings at the facility also very likely resulted in incomplete extraction of the studied SVOCs as the samples were only extracted a single time in the first round of sampling (due to time and financial constraints). In addition to inefficiently extracting the samples, there were two different extraction methods used for samples - the vacuum and wipe samples underwent microwave-assisted and sonication extractions, respectively. The vacuum samples were extracted using the microwave-assisted method because the sampling socks were too tall for the sonication vessel. The wipes could not be extracted using microwave methods to eliminate the potential risk for combustion. For the purposes of comparing methods, it would have been preferable to use the same extraction protocol for both types of samples. Therefore, due to the limitations of the methods, additional testing was performed to further evaluate both sample collection methods and sample extraction efficiencies.

Post-hoc Sampling

The results from the sequential wiping provided information regarding the collection methods, however, it is important to note that only one wipe and two vacuum sampling sequences were conducted. Nevertheless, successive recoveries of mass in the sequential wipe collection series (Figure 4) demonstrate that initial wipe collections may be less than 50% efficient. The overall finding of wipe-to-vacuum ratios not greater than one could be explained if a single vacuum collection is more efficient than a single wipe collection under high load conditions. Several additional factors may have contributed to the irregularities of the wipe results, such as choice and volume of solvent used, inability to normalize the pressure and force of each wipe collection, and inefficiency in collecting contaminant material from cracks and gaps in non-smooth surfaces. These factors were beyond the anticipated scope of work.

In order to assess the efficacy of the vacuum collection methods, post-vacuum wipe samples were collected (Figures 5 and 6). The results from this set of data produced measureable contaminant masses by post-vacuum wipes that were qualitatively consistent with the original hypothesis that significant non particle-bound mass would be found. Particularly, PBDE-47 and PBDE-99 had post-vacuum wipes that comprised 27% and 33% of the total mass collected (vacuum and post-vacuum wipe) for the samples where only sonication extractions were implemented (V2), respectively. This trend is not as obvious for the other compounds, perhaps due to the high loads of the dust samples relative to the organic film. However, each of the post-vacuum wipes had measureable masses ranging from 14 to 12,800 ng (for a 0.25-m^2 sampled surface).

The data obtained in the comparison of microwave and sonication extraction methods (Figures 7 and 8) provide evidence that the two methods may not be interchangeable. The compounds that had relatively lower masses (< 250 ng per sample) showed greater efficiencies using the microwave methods. For example, PBDE-47, PBDE-99 and PBDE-153 were more effectively extracted by the microwave-assisted method by a factor of nearly two or greater when compared to sonication methods. However, as the contaminant masses increased (> 10,000 ng per sample), neither extraction method appeared superior. In light of these inconsistencies, future studies involving the comparison of wipe and vacuum samples could consider further evaluating the influence of extraction method choice or use the microwave-assisted technique as it showed better performance across a wide range of contaminant masses.

The sequential extraction series (Figures 9-15) provided additional information regarding the analytical methods applied. Generally, the extraction series displayed decreasing contaminant mass trends, with a few irregularities. At the site studied here, three to four extractions would have been sufficient to yield the vast majority of the SVOC mass collected in each sample. However, there was still a measureable amount contaminant mass in the final extraction (average percent mass 4%). In future studies at a comparable site, multiple extractions or reduced sample area would be warranted.

Limitations

There were a number of limitations associated with this study. The study design included a small sample size (n = 5) for the paired vacuum and wipe samples on horizontal and vertical surfaces in the first sampling round. When initially designing the study, more samples were envisioned, however, further sampling was not possible due to lack of funding. During the second round of sampling, only a single wipe was collected. Although the *post-hoc* sampling round provided two vacuum samples, they underwent different extraction techniques.

Moreover, it would have been beneficial to sample at more than one e-waste facility but access to additional sites was not feasible.

There remains a high level of uncertainty with regard to the recovery of the contaminant in the vacuum and wipe samples. Only one extraction was performed for the samples taken in the first round of sampling. In the subsequent results of the *post-hoc* analysis, the fourth extraction produced measureable masses, suggesting that a single extraction is not adequate. Furthermore, different extraction methods were utilized for the paired vacuum and wipe samples due to methodological difficulties associated with using the microwave-assisted protocol on the wipe samples. This raises questions regarding direct comparability of the resulting SVOC loadings by wipe and vacuum for each surface sampled.

A disadvantage of conducting this study at an e-waste facility was that the SVOC contaminant loadings were so high as to present challenges to collection and extraction. Results from the first round of sampling undoubtedly reflect both incomplete sample collection and incomplete extraction. This may have also impacted the results as the degree of contamination in the dust phase may have been unusually large in comparison to the thin layer of organic film. The high SVOC loadings present in the sampled matrix also presented a problem when extracting as the samples required repeated extractions.

Finally, no pre-analysis sample clean-up was performed due to budget and time constraints. As a result peak identification was hindered and peak area was subject to interference.

Future Study Recommendations

The outcome of this study provides useful lessons for improved strategies for future studies. Surface loads reported here are useful for range-finding. If future samples are collected at this study site (or another e-waste facility), smaller sampling areas should be considered. Collecting a larger number of samples and sampling multiple e-waste facilities would increase the statistical power and representativeness of the data collected. The use of standardized surfaces (e.g., glass deposition plates) might also improve consistency and reproducibility of results. An air sampling component could be added to provide additional information regarding the gas-phase and particle-bound SVOC in the air, which may shed light on accumulation on surfaces.

The extraction method implemented when comparing paired samples should be uniform for all samples, if possible. If smaller sample areas are used (less mass), microwave-assisted methods may be superior as higher extraction efficiencies using microwave-assisted methods were associated with lower masses. There should be multiple extractions per sample to improve SVOC contaminant recovery from the matrix. It is also strongly recommended that clean-up of sample extracts be performed to reduce sample complexity.

References

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

Ait Bamai, Y., Araki, A., Kawai, T., Tsuboi, T., Saito, I., Yoshioka, E., Kanazawa, A., Tajima, S., Shi, C., Tamakoshi, A. and Kishi, R. 2013. Associations of phthalate concentrations in floor dust and multi-surface dust with the interior materials in Japanese dwellings. Sci Total Environ 468-469c, 147-157.

Allen, J.G., McClean, M.D., Stapleton, H.M., Nelson, J.W. and Webster, T.F. 2007. Personal exposure to polybrominated diphenyl ethers (PBDEs) in residential indoor air. Environ Sci Technol 41, 4574-4579.

Allen, J.G., McClean, M.D., Stapleton, H.M. and Webster, T.F. 2008a. Critical factors in assessing exposure to PBDEs via house dust. Environ Int, United States, pp. 1085-1091.

Allen, J.G., McClean, M.D., Stapleton, H.M. and Webster, T.F. 2008b. Linking PBDEs in house dust to consumer products using X-ray fluorescence. Environ Sci Technol 42, 4222-4228.

Araki, A., Saito, I., Kanazawa, A., Morimoto, K., Nakayama, K., Shibata, E., Tanaka, M., Takigawa, T., Yoshimura, T., Chikara, H., Saijo, Y. and Kishi, R. 2013. Phosphorus flame retardants in indoor dust and their relation to asthma and allergies of inhabitants. Indoor Air.

Baier, E.J. 1985. OSHA Standards do not Include Surface Contamination Criteria or Quantifications for Skin Absorption [1900.1000]. In: O.S.a.H. Administration (Ed), U.S. Department of Labor - OSHA, Washington, D.C.

Baldi, B. and Moore, D.S. 2012. The Practice of Statistics in the Life Sciences, W. H. Freeman and Company, New York.

Bao, L.J., You, J. and Zeng, E.Y. 2011. Sorption of PBDE in low-density polyethylene film: implications for bioavailability of BDE-209. Environ Toxicol Chem 30, 1731-1738.

Batterman, S.A., Chernyak, S., Jia, C., Godwin, C. and Charles, S. 2009. Concentrations and emissions of polybrominated diphenyl ethers from U.S. houses and garages. Environ Sci Technol 43, 2693-2700.

Becker, K., Seiwert, M., Angerer, J., Heger, W., Koch, H.M., Nagorka, R., Rosskamp, E., Schluter, C., Seifert, B. and Ullrich, D. 2004. DEHP metabolites in urine of children and DEHP in house dust. Int J Hyg Environ Health 207, 409-417.

Bendig, P. and Vetter, W. 2010. Photolytical transformation rates of individual polybrominated diphenyl ethers in technical octabromo diphenyl ether (DE-79). Environ Sci Technol 44, 1650-1655.

Bergh, C., Torgrip, R., Emenius, G. and Ostman, C. 2011. Organophosphate and phthalate esters in air and settled dust - a multi-location indoor study. Indoor Air 21, 67-76.

Bjorklund, J.A., Sellstrom, U., de Wit, C.A., Aune, M., Lignell, S. and 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.

Blanchard, O., Mercier, F., Ramalho, O., Mandin, C., Le Bot, B. and Glorennec, P. 2013. Measurements of semi-volatile organic compounds in settled dust: influence of storage temperature and duration. Indoor Air.

Boeniger, M.F. 2003. The significance of skin exposure. Ann Occup Hyg 47, 591-593.

Bonvallot, N., Mandin, C., Mercier, F., Le Bot, B. and Glorennec, P. 2010. Health ranking of ingested semi-volatile organic compounds in house dust: an application to France. Indoor Air 20, 458-472.

Bornehag, C.G., Sundell, J., Weschler, C.J., Sigsgaard, T., Lundgren, B., Hasselgren, M. and Hagerhed-Engman, L. 2004. The association between asthma and allergic symptoms in children and phthalates in house dust: a nested case-control study. Environ Health Perspect 112, 1393-1397.

Botelho, M.A., Kurtz, K., Dawson, B.J. and Kaiser, M.A. 2009. Development and validation of a wipe test method using liquid chromatography with tandem mass spectrometry for the determination of Perfluorooctanoate (PFO) on various surfaces. J Occup Environ Hyg 6, 390-395.

Brommer, S., Harrad, S., Van den Eede, N. and Covaci, A. 2012. Concentrations of organophosphate esters and brominated flame retardants in German indoor dust samples. J Environ Monit 14, 2482-2487.

Butt, C.M., Diamond, M.L., Truong, J., Ikonomou, M.G., Helm, P.A. and Stern, G.A. 2004a. Semivolatile organic compounds in window films from lower Manhattan after the September 11th World Trade Center attacks. Environ Sci Technol 38, 3514-3524.

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

Cahill, T.M., Groskova, D., Charles, M.J., Sanborn, J.R., Denison, M.S. and Baker, L. 2007. Atmospheric concentrations of polybrominated diphenyl ethers at near-source. Environ Sci Technol 41, 6370-6377.

Calfee, M.W., Rose, L.J., Morse, S., Mattorano, D., Clayton, M., Touati, A., Griffin-Gatchalian, N., Slone, C. and McSweeney, N. 2013. Comparative evaluation of vacuum-based surface sampling methods for collection of Bacillus spores. J Microbiol Methods 95, 389-396.

Casas, M., Chevrier, C., Hond, E.D., Fernandez, M.F., Pierik, F., Philippat, C., Slama, R., Toft, G., Vandentorren, S., Wilhelm, M. and Vrijheid, M. 2013. Exposure to brominated flame retardants, perfluorinated compounds, phthalates and phenols in European birth cohorts: ENRIECO evaluation, first human biomonitoring results, and recommendations. Int J Hyg Environ Health 216, 230-242.

CDC. 2003. Second national report on human exposure to environmental chemicals. In: U.S. DHHS, Centers for Disease Control and Prevention, Atlanta, GA.

CDC. 2005. Third national report on human exposure to environmental chemicals. In: U.S. DHHS, Centers for Disease Control and Prevention, Atlanta, GA.

Cetin, B. and Odabasi, M. 2008. Atmospheric concentrations and phase partitioning of polybrominated diphenyl ethers (PBDEs) in Izmir, Turkey. Chemosphere, England, pp. 1067-1078.

Cetin, B. and Odabasi, M. 2011. Polybrominated diphenyl ethers (PBDEs) in indoor and outdoor window organic films in Izmir, Turkey. J Hazard Mater, 2010 Elsevier B.V, Netherlands, pp. 784-791.

Chavalitnitikul, C. and Levin, L. 1984. A laboratory evaluation of wipe testing based on lead oxide surface contamination. Am Ind Hyg Assoc J 45, 311-317.

Clifton, M.S., Wargo, J.P., Weathers, W.S., Colon, M., Bennett, D.H. and Tulve, N.S. 2013. Quantitative analysis of organophosphate and pyrethroid insecticides, pyrethroid transformation products, polybrominated diphenyl ethers and bisphenol A in residential surface wipe samples. J Chromatogr A 1273, 1-11.

Colt, J.S. 1998. Comparison of pesticides and other compounds in carpet dust samples collected from used vacuum cleaner bags and from a high-volume surface sampler. Environ Health Perspect 106, 721-724.

Colt, J.S., Gunier, R.B., Metayer, C., Nishioka, M.G., Bell, E.M., Reynolds, P., Buffler, P.A. and Ward, M.H. 2008. Household vacuum cleaners vs. the high-volume surface sampler for collection of carpet dust samples in epidemiologic studies of children. Environ Health 7, 6.

Cristale, J. and Lacorte, S. 2013. Development and validation of a multiresidue method for the analysis of polybrominated diphenyl ethers, new brominated and organophosphorus flame retardants in sediment, sludge and dust. J Chromatogr A 1305, 267-275.

Cristale, J., Quintana, J., Chaler, R., Ventura, F. and Lacorte, S. 2012. Gas chromatography/mass spectrometry comprehensive analysis of organophosphorus, brominated flame retardants, by-products and formulation intermediates in water. J Chromatogr A 1241, 1-12.

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

Deng, W.J., Zheng, J.S., Bi, X.H., Fu, J.M. and Wong, M.H. 2007. Distribution of PBDEs in air particles from an electronic waste recycling site compared with Guangzhou and Hong Kong, South China. Environ Int 33, 1063-1069.

Destaillats, H., Maddalena, R.L., Singer, B.C., Hodgson, A.T. and McKone, T.E. 2008. Indoor pollutants emitted by office equipment: A review of reported data and information needs. Atmospheric Environment 42, 1371-1388.

Diamond, M.L., Priemer, D.A. and Law, N.L. 2001. Developing a multimedia model of chemical dynamics in an urban area. Chemosphere 44, 1655-1667.

Dorival-Garcia, N., Zafra-Gomez, A., Navalon, A. and Vilchez, J.L. 2012. Analysis of bisphenol A and its chlorinated derivatives in sewage sludge samples. Comparison of the efficiency of three extraction techniques. J Chromatogr A 1253, 1-10.

Doull, J. 1988. Acceptable Levels of Dioxin Contamination in an Office Building Following a Transformer Fire, National Academy Press.

Dufresne, A., Mocanu, T., Viau, S., Perrault, G. and Dion, C. 2011. Efficacy of surface sampling methods for different types of beryllium compounds. J Environ Monit 13, 74-83.

Eljarrat, E. and Barcelo, D. 2011. The Handbook of Environmental Chemistry, Springer, New York.

Farfel, M.R., Lees, P.S., Rohde, C.A., Lim, B.S., Bannon, D. and Chisolm, J.J., Jr. 1994. Comparison of a wipe and a vacuum collection method for the determination of lead in residential dusts. Environ Res, United States, pp. 291-301.

Fromme, H., Korner, W., Shahin, N., Wanner, A., Albrecht, M., Boehmer, S., Parlar, H., Mayer, R., Liebl, B. and Bolte, G. 2009. Human exposure to polybrominated diphenyl ethers (PBDE), as evidenced by data from a duplicate diet study, indoor air, house dust, and biomonitoring in Germany. Environ Int 35, 1125-1135.

Fromme, H., Lahrz, T., Piloty, M., Gebhart, H., Oddoy, A. and Ruden, H. 2004. Occurrence of phthalates and musk fragrances in indoor air and dust from apartments and kindergartens in Berlin (Germany). Indoor Air 14, 188-195.

Geens, T., Roosens, L., Neels, H. and Covaci, A. 2009. Assessment of human exposure to Bisphenol-A, Triclosan and Tetrabromobisphenol-A through indoor dust intake in Belgium. Chemosphere 76, 755-760.

Gold, M.D., Blum, A. and Ames, B.N. 1978. Another flame retardant, tris-(1,3-dichloro-2-propyl)-phosphate, and its expected metabolites are mutagens. Science 200, 785-787.

Gorman Ng, M., de Poot, S., Schmid, K., Cowie, H., Semple, S. and van Tongeren, M. 2013. Properties of liquids and dusts: how do they influence dermal loading during immersion, deposition, and surface contact exposure pathways? Ann Occup Hyg 57, 627-639.

Hardy, M.L. 2005. Comment on: "personal air sampling and analysis of polybrominated diphenyl ethers and other bromine containing compounds at an electronics recycling facility in Sweden" by A. Pettersson-Julander, B. van Bavel, M. Engwall and H. Westberg, JEM, 2004, 6, 874. J Environ Monit 7, 643; author reply 644.

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

Hearn, L.K., Hawker, D.W., Toms, L.M. and Mueller, J.F. 2013. Assessing exposure to polybrominated diphenyl ethers (PBDEs) for workers in the vicinity of a large recycling facility. Ecotoxicol Environ Saf.

Heudorf, U., Mersch-Sundermann, V. and Angerer, J. 2007. Phthalates: toxicology and exposure. Int J Hyg Environ Health 5, 623-34.

Hodge, E.M., Diamond, M.L., McCarry, B.E., Stern, G.A. and Harper, P.A. 2003. Sticky windows: chemical and biological characteristics of the organic film derived from particulate and gas-phase air contaminants found on an urban impervious surface. Arch Environ Contam Toxicol 44, 421-429.

Jakobsson, K., Thuresson, K., Rylander, L., Sjodin, A., Hagmar, L. and Bergman, A. 2002. Exposure to polybrominated diphenyl ethers and tetrabromobisphenol A among computer technicians. Chemosphere, England, pp. 709-716.

Johnson-Restrepo, B. and Kannan, K. 2009. An assessment of sources and pathways of human exposure to polybrominated diphenyl ethers in the United States. Chemosphere 76, 542-548.

Jones-Otazo, H.A., Clarke, J.P., Diamond, M.L., Archbold, J.A., Ferguson, G., Harner, T., Richardson, G.M., Ryan, J.J. and Wilford, B. 2005. Is house dust the missing exposure pathway for PBDEs? An analysis of the urban fate and human exposure to PBDEs. Environ Sci Technol 39, 5121-5130.

Julander, A., Karlsson, M., Hagstrom, K., Ohlson, C.G., Engwall, M., Bryngelsson, I.L., Westberg, H. and van Bavel, B. 2005a. Polybrominated diphenyl ethers--plasma levels and thyroid status of workers at an electronic recycling facility. Int Arch Occup Environ Health 78, 584-592.

Julander, A., Westberg, H., Engwall, M. and van Bavel, B. 2005b. Distribution of brominated flame retardants in different dust fractions in air. Sci Total Environ 350, 151-160.

Julander, A., Westberg, H., Engwall, M. and van Bavel, B. 2005c. Distribution of brominated flame retardants in different dust fractions in air from an electronics recycling facility. Sci Total Environ, Netherlands, pp. 151-160.

Morgan, M., Sheldon, L.S., Croghan C.W., Chuang J.C., Lordo, R.A., Wilson N.K., Lyu C., Brinkman M., Morse, N., Chou Y.L., Hamilton C., Finegold J.K., Hand E. and Gordon S.M. 2004. A Pilot Study of Children's Total Exposure to Persistent Pesticides and Other Persistent Organic Pollutants (CTEPP) In: U.S.E.N.E.R. Laboratory (Ed), Research Triangle Park, NC.

Kanazawa, A. and Kishi, R. 2009. [Potential risk of indoor semivolatile organic compounds indoors to human health]. Nihon Eiseigaku Zasshi 64, 672-682.

Kanazawa, A., Saito, I., Araki, A., Takeda, M., Ma, M., Saijo, Y. and 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.

Karlsson, M., Julander, A., van Bavel, B. and Hardell, L. 2007. Levels of brominated flame retardants in blood in relation to levels in household air and dust. Environ Int 33, 62-69.

Kim, H.H., Yang, J.Y., Kim, S.D., Yang, S.H., Lee, C.S., Shin, D.C. and Lim, Y.W. 2011. Health risks assessment in children for phthalate exposure associated with childcare facilities and indoor playgrounds. Environ Health Toxicol 26, e2011008.

Kim, N.K., Hawley, J.K. and Assessment, N.Y.B.o.T.S. 1985. Re-entry Guidelines, Binghamton State Office Building, Bureau of Toxic Substance Assessment, Division of Environmental Health Assessment, New York State Department of Health.

Kolarik, B., Naydenov, K., Larsson, M., Bornehag, C.G. and Sundell, J. 2008. The association between phthalates in dust and allergic diseases among Bulgarian children. Environ Health Perspect 116, 98-103.

Krol, S., Zabiegala, B. and Namiesnik, J. 2011. Monitoring and analytics of semivolatile organic compounds (SVOCs) in indoor air. Anal Bioanal Chem 400, 1751-1769.

Law, N.L. and Diamond, M.L. 1998. The role of organic films and the effect on hydrophobic organic compounds in urban areas: An hypothesis. Chemosphere 36, 2607-2620.

Lees, P.S., Corn, M. and Breysse, P.N. 1987. Evidence for dermal absorption as the major route of body entry during exposure of transformer maintenance and repairmen to PCBs. Am Ind Hyg Assoc J 48, 257-264.

Lim, Y.W., Kim, H.H., Lee, C.S., Shin, D.C., Chang, Y.S. and Yang, J.Y. 2014. Exposure assessment and health risk of poly-brominated diphenyl ether (PBDE) flame retardants in the indoor environment of elementary school students in Korea. Sci Total Environ.

Little, J.C., Weschler, C.J., Nazaroff, W.W., Liu, Z. and Cohen Hubal, E.A. 2012. Rapid methods to estimate potential exposure to semivolatile organic compounds in the indoor environment. Environ Sci Technol 46, 11171-11178.

Liu, C., Liu, Z., Little, J.C. and Zhang, Y. 2013. Convenient, Rapid and Accurate Measurement of SVOC Emission Characteristics in Experimental Chambers. PLoS One 8, e72445.

Liu, Q.T., Chen, R., McCarry, B.E., Diamond, M.L. and Bahavar, B. 2003. Characterization of polar organic compounds in the organic film on indoor and outdoor glass windows. Environ Sci Technol 37, 2340-2349.

Lopez-Avila, V. and Young, R. 1994. Microwave-assisted extraction of organic compounds from standard reference soils and sediments. Analytical Chemistry 66, 10.

Lopez-Avila, V., Young, R., Benedicto, J., Ho, P. and Kim, R. 1995. Extraction of organic pollutants from solid samples using microwave energy. Anal. Chem. 67, 7.

Lorber, M. 2008. Exposure of Americans to polybrominated diphenyl ethers. J Expo Sci Environ Epidemiol 18, 2-19.

Ma, J., Addink, R., Yun, S., Cheng, J., Wang, W. and Kannan, K. 2009. Polybrominated dibenzo-p-dioxins/ dibenzofurans and polybrominated diphenyl ethers in soil, vegetation, workshop-floor dust, and electronic shredder residue from an electronic waste recycling facility and in soils from a chemical industrial complex in eastern China. Environ Sci Technol 43, 7350-7356.

Ma, T.T., Christie, P., Luo, Y.M. and Teng, Y. 2013. Phthalate esters contamination in soil and plants on agricultural land near an electronic waste recycling site. Environ Geochem Health 35, 465-476.

Marklund, A., Andersson, B. and Haglund, P. 2003. Screening of organophosphorus compounds and their distribution in various indoor environments. Chemosphere 53, 1137-1146.

Medicine, U.S.A.C.f.H.P.a.P. 2009. Health Risk Assessment Methods and Screening Levels for Evaluating Office Worker Exposures to Contaminants on Indoor Surfaces Using Surface Wipe Data. In: E.H.R.A. Program (Ed), MD.

Meeker, J.D., Cooper, E.M., Stapleton, H.M. and Hauser, R. 2013. Urinary metabolites of organophosphate flame retardants: temporal variability and correlations with house dust concentrations. Environ Health Perspect 121, 580-585.

Meeker, J.D. and 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.

Morgan, M.K., Sheldon, L.S., Croghan, C.W., Jones, P.A., Chuang, J.C. and Wilson, N.K. 2007. An observational study of 127 preschool children at their homes and daycare centers in Ohio: environmental pathways to cis- and trans-permethrin exposure. Environ Res 104, 266-274.

Morgan, M.K., Sheldon, L.S., Thomas, K.W., Egeghy, P.P., Croghan, C.W., Jones, P.A., Chuang, J.C. and Wilson, N.K. 2008. Adult and children's exposure to 2,4-D from multiple sources and pathways. J Expo Sci Environ Epidemiol 18, 486-494.

Morose, G. 2006. An Overview of alternatives to tetrabromobisphenol-A (TBBPA) and hexabromocyclododecane (HBCD). Lowell Center for Sustainable Production: University of Massachusetts Lowell, Lowell, MA, p. 26.

Muenhor, D., Harrad, S., Ali, N. and Covaci, A. 2010. Brominated flame retardants (BFRs) in air and dust from electronic waste storage facilities in Thailand. Environ Int, 2010 Elsevier Ltd, Netherlands, pp. 690-698.

Nazaroff, W.W. 2011. Toward Rapid Exposure Assessment. SVOCs in the Indoor Environment, Research Triangle Park, NC.

Ni, K., Lu, Y., Wang, T., Kannan, K., Gosens, J., Xu, L., Li, Q., Wang, L. and Liu, S. A review of human exposure to polybrominated diphenyl ethers (PBDEs) in China. International Journal of Hygiene and Environmental Health.

OSHA. (undated). Evaluation Guidelines for Surface Sampling Methods. In: U.S. Dept. of Labor (Ed), Utah.

OSHA. 2008. OSHA Technical Manual. In: U.S. Dept. of Labor (Ed), Washington, DC.

Pettersson-Julander, A., van Bavel, B., Engwall, M. and Westberg, H. 2004. Personal air sampling and analysis of polybrominated diphenyl ethers and other bromine containing compounds at an electronic recycling facility in Sweden. J Environ Monit 6, 874-880.

Punin Crespo, M.O., Cam, D., Gagni, S., Lombardi, N. and Lage Yusty, M.A. 2006. Extraction of hydrocarbons from seaweed samples using sonication and microwave-assisted extraction: a comparative study. J Chromatogr Sci 44, 615-618.

Qu, W., Bi, X., Sheng, G., Lu, S., Fu, J., Yuan, J. and Li, L. 2007. Exposure to polybrominated diphenyl ethers among workers at an electronic waste dismantling region in Guangdong, China. Environ Int, United States, pp. 1029-1034.

Ramsey, F.L. and Schafer, D.W. 2013. The Statistical Sleuth: A Course in Methods Data Analysis, Brooks / Cole, Boston.

Rosenberg, C., Hameila, M., Tornaeus, J., Sakkinen, K., Puttonen, K., Korpi, A., Kiilunen, M., Linnainmaa, M. and Hesso, A. 2011. Exposure to flame retardants in electronics recycling sites. Ann Occup Hyg 55, 658-665.

Rudel, R.A., Brody, J.G., Spengler, J.D., Vallarino, J., Geno, P.W., Sun, G. and Yau, A. 2001. Identification of selected hormonally active agents and animal mammary carcinogens in commercial and residential air and dust samples. J Air Waste Manag Assoc 51, 499-513.

Rudel, R.A., Camann, D.E., Spengler, J.D., Korn, L.R. and Brody, J.G. 2003. Phthalates, alkylphenols, pesticides, polybrominated diphenyl ethers, and other endocrine-disrupting compounds in indoor air and dust. Environ Sci Technol 37, 4543-4553.

Santillo, D., Labunska, I., Davidson, H., Johnston, P., Strutt, M. and Knowles, O. 2003. Consuming Chemicals: Hazardous chemicals in house dust as an indicator of chemical exposure in the home. In: G.R. Laboratories (Ed), Technical Note, UK.

Schecter, A., Colacino, J.A., Harris, T.R., Shah, N. and Brummitt, S.I. 2009. A Newly Recognized Occupational Hazard for US Electronic Recycling Facility Workers: Polybrominated Diphenyl Ethers. Journal of Environmental and Occupation Medicine 51, 6.

Schecter, A., Papke, O., Joseph, J.E. and Tung, K.C. 2005. Polybrominated diphenyl ethers (PBDEs) in U.S. computers and domestic carpet vacuuming: possible sources of human exposure. J Toxicol Environ Health A 68, 501-513.

Sjodin, A., Carlsson, H., Thuresson, K., Sjolin, S., Bergman, A. and Ostman, C. 2001. Flame retardants in indoor air at an electronics recycling plant and at other. Environ Sci Technol 35, 448-454.

Sjodin, A., Papke, O., McGahee, E., Focant, J.F., Jones, R.S., Pless-Mulloli, T., Toms, L.M., Herrmann, T., Muller, J., Needham, L.L. and Patterson, D.G., Jr. 2008. Concentration of polybrominated diphenyl ethers (PBDEs) in household dust from various countries. Chemosphere 73, S131-136.

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

Stapleton, H.M., Dodder, N.G., Offenberg, J.H., Schantz, M.M. and Wise, S.A. 2005. Polybrominated diphenyl ethers in house dust and clothes dryer lint. Environ Sci Technol 39, 925-931.

Stapleton, H.M., Kelly, S.M., Allen, J.G., McClean, M.D. and Webster, T.F. 2008b. Measurement of Polybrominated Diphenyl Ethers on Hand Wipes: Estimating Exposure from Hand-to-Mouth Contact. Environmental Science & Technology 42, 6.

Stuart, H., Ibarra, C., Abdallah, M.A., Boon, R., Neels, H. and Covaci, A. 2008. 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.

Takigami, H., Suzuki, G., Hirai, Y. and Sakai, S. 2008. Transfer of brominated flame retardants from components into dust inside television cabinets. Chemosphere 73, 161-169.

Takigami, H., Suzuki, G., Hirai, Y. and Sakai, S. 2009. Brominated flame retardants and other polyhalogenated compounds in indoor air and dust from two houses in Japan. Chemosphere 76, 270-277.

Thomsen, C., Lundanes, E. and Becher, G. 2001. Brominated flame retardants in plasma samples from three different occupational groups in Norway. J Environ Monit 3, 366-370.

Thuresson, K., Bergman, K., Rothenbacher, K., Herrmann, T., Sjolin, S., Hagmar, L., Papke, O. and Jakobsson, K. 2006. Polybrominated diphenyl ether exposure to electronics recycling workers--a follow up study. Chemosphere 64, 1855-1861.

Toms, L.M., Bartkow, M.E., Symons, R., Paepke, O. and Mueller, J.F. 2009a. Assessment of polybrominated diphenyl ethers (PBDEs) in samples collected from indoor environments in South East Queensland, Australia. Chemosphere 76, 173-178.

Toms, L.M., Hearn, L., Kennedy, K., Harden, F., Bartkow, M., Temme, C. and Mueller, J.F. 2009b. Concentrations of polybrominated diphenyl ethers (PBDEs) in matched samples of human milk, dust and indoor air. Environ Int, Netherlands, pp. 864-869.

Toms, L.M., Hearn, L., Kennedy, K., Harden, F., Bartkow, M., Temme, C. and Mueller, J.F. 2009c. Concentrations of polybrominated diphenyl ethers (PBDEs) in matched samples of human milk, dust and indoor air. Environ Int 35, 864-869.

Trudel, D., Scheringer, M., von Goetz, N. and Hungerbuhler, K. 2011. Total consumer exposure to polybrominated diphenyl ethers in North America and Europe. Environ Sci Technol 45, 2391-2397.

Tu, C. and Prest, H.F. 2005. Determination by gas chromatography in combination with mass spectrometry detector (GC–MS) in electronic ionization mode (EI) with selected-ion monitoring (SIM) acquisition method (GC–MS (EI–SIM)) has been carried out., Aligent Technologies.

Tue, N.M., Takahashi, S., Suzuki, G., Isobe, T., Viet, P.H., Kobara, Y., Seike, N., Zhang, G., Sudaryanto, A. and Tanabe, S. 2013. Contamination of indoor dust and air by polychlorinated biphenyls and brominated flame retardants and relevance of non-dietary exposure in Vietnamese informal e-waste recycling sites. Environ Int 51, 160-167.

Van den Eede, N., Dirtu, A.C., Neels, H. and 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. and de Boer, J. 2012. Phosphorus flame retardants: properties, production, environmental occurrence, toxicity and analysis. Chemosphere 88, 1119-1153.

Vermeulen, R., Bos, R.P., Pertijs, J. and Kromhout, H. 2003. Exposure related mutagens in urine of rubber workers associated with inhalable particulate and dermal exposure. Occup Environ Med 60, 97-103.

Vetter, W., Recke, R., Symons, R. and Pyecroft, S. 2008. Determination of polybrominated biphenyls in Tasmanian devils (Sarcophilus harrisii) by gas chromatography coupled to electron capture negative ion tandem mass spectrometry or electron ionization high-resolution mass spectrometry. Rapid Commun Mass Spectrom 22, 4165-4170.

Wang, J., Chen, S.J., Tian, M., Ma, Y.J., Luo, X.J. and Mai, B.X. 2010a. [Polybrominated diphenyl ethers (PBDEs) in indoor and outdoor dust from an electronic waste (e-waste) recycling area in South China: contamination and human exposure]. Huan Jing Ke Xue 31, 173-178.

Wang, J., Ma, Y.J., Chen, S.J., Tian, M., Luo, X.J. and Mai, B.X. 2010b. Brominated flame retardants in house dust from e-waste recycling and urban areas in South China: implications on human exposure. Environ Int 36, 535-541.

Watkins, D.J., McClean, M.D., Fraser, A.J., Weinberg, J., Stapleton, H.M. and Webster, T.F. 2013. Associations between PBDEs in office air, dust, and surface wipes. Environ Int 59, 124-132.

Webster, T.F., Watkins, D.J., Courtney, W., J, F.A., Heiger-Bernays, W., Stapleton, H.M. and McClean, M.D. PentaBDE Alternatives in Homes, Offices and Cars. Boston University School of Public Health, Durham, NC.

Wensing, M., Uhde, E. and Salthammer, T. 2005. Plastics additives in the indoor environment--flame retardants and plasticizers. Sci Total Environ 339, 19-40.

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

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

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

Weschler, C.J., Salthammer, T. and Fromme, H. 2008. Partitioning of phthalates among the gas phase, airborne particles and settle dust in indoor environments. Atmospheric Environment 42, 1449-1460.

Wilford, B.H., Shoeib, M., Harner, T., Zhu, J. and Jones, K.C. 2005. Polybrominated diphenyl ethers in indoor dust in Ottawa, Canada: implications for sources and exposure. Environ Sci Technol 39, 7027-7035.

Wilson, N.K., Chuang, J.C., Iachan, R., Lyu, C., Gordon, S.M., Morgan, M.K., Ozkaynak, H. and Sheldon, L.S. 2004. Design and sampling methodology for a large study of preschool children's aggregate exposures to persistent organic pollutants in their everyday environments. J Expo Anal Environ Epidemiol 14, 260-274.

Wilson, N.K., Chuang, J.C., Lyu, C., Menton, R. and Morgan, M.K. 2003. Aggregate exposures of nine preschool children to persistent organic pollutants at day care and at home. J Expo Anal Environ Epidemiol 13, 187-202.

Wilson, N.K., Chuang, J.C., Morgan, M.K., Lordo, R.A. and Sheldon, L.S. 2007. An observational study of the potential exposures of preschool children to pentachlorophenol, bisphenol-A, and nonylphenol at home and daycare. Environ Res 103, 9-20.

Wu, N., Herrmann, T., Paepke, O., Tickner, J., Hale, R., Harvey, L.E., La Guardia, M., McClean, M.D. and Webster, T.F. 2007. Human exposure to PBDEs: associations of PBDE body burdens with food consumption and house dust concentrations. Environ Sci Technol 41, 1584-1589.

Xu, Y. and Little, J.C. 2006. Predicting emissions of SVOCs from polymeric materials and their interaction with airborne particles. Environ Sci Technol 40, 456-461.

Xu, Y. and Zhang, J. 2011. Understanding SVOCs. ASHRAE Journal 53, 5.

Yamaguchi, C. and Lee, W.Y. 2010. A cost effective, sensitive, and environmentally friendly sample preparation method for determination of polycyclic aromatic hydrocarbons in solid samples. J Chromatogr A 1217, 6816-6823.

Zheng, J., Luo, X.J., Yuan, J.G., Wang, J., Wang, Y.T., Chen, S.J., Mai, B.X. and Yang, Z.Y. 2011. Levels and sources of brominated flame retardants in human hair from urban, e-waste, and rural areas in South China. Environ Pollut 159, 3706-3713.

Appendix

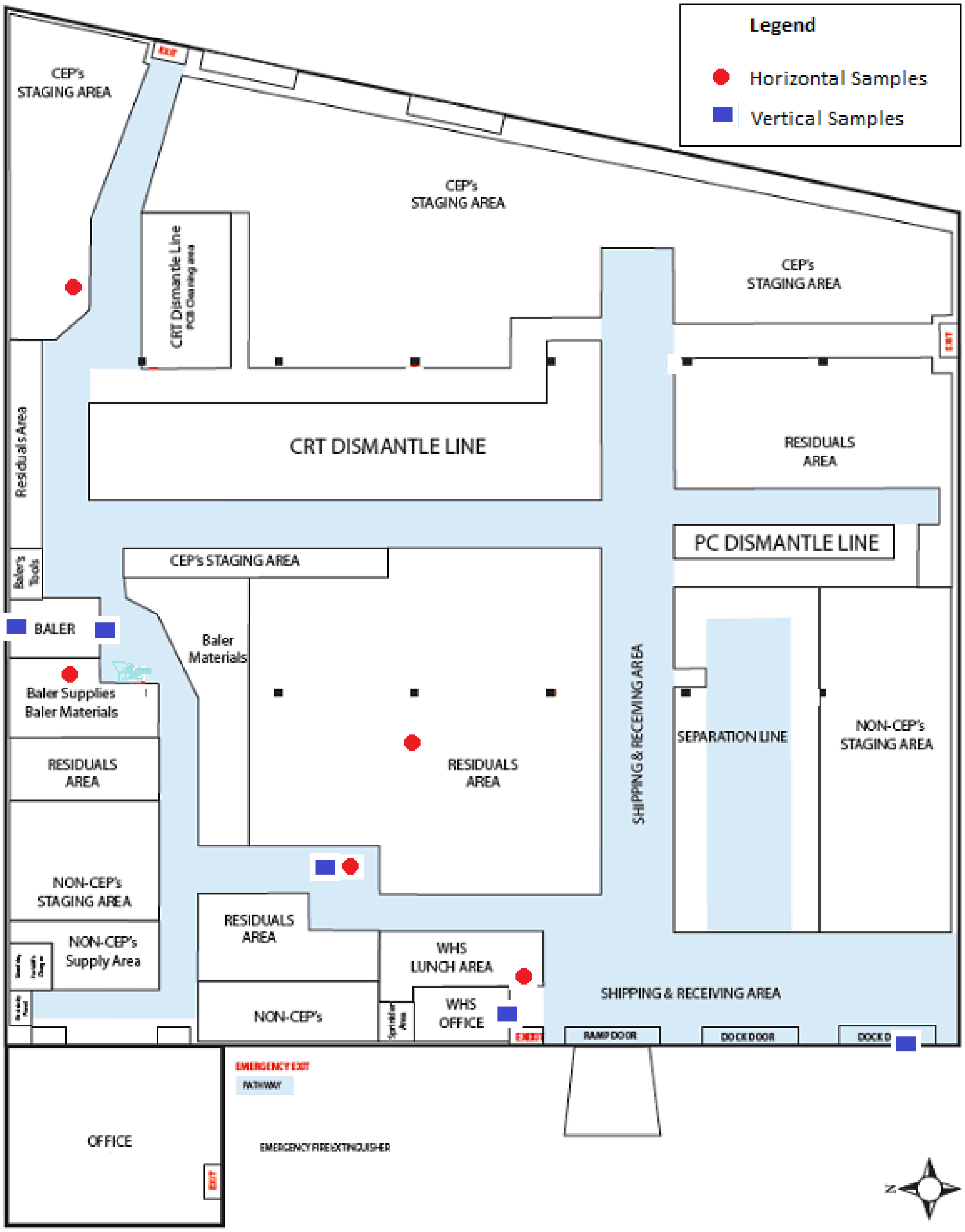

Figure 16: Facility Map

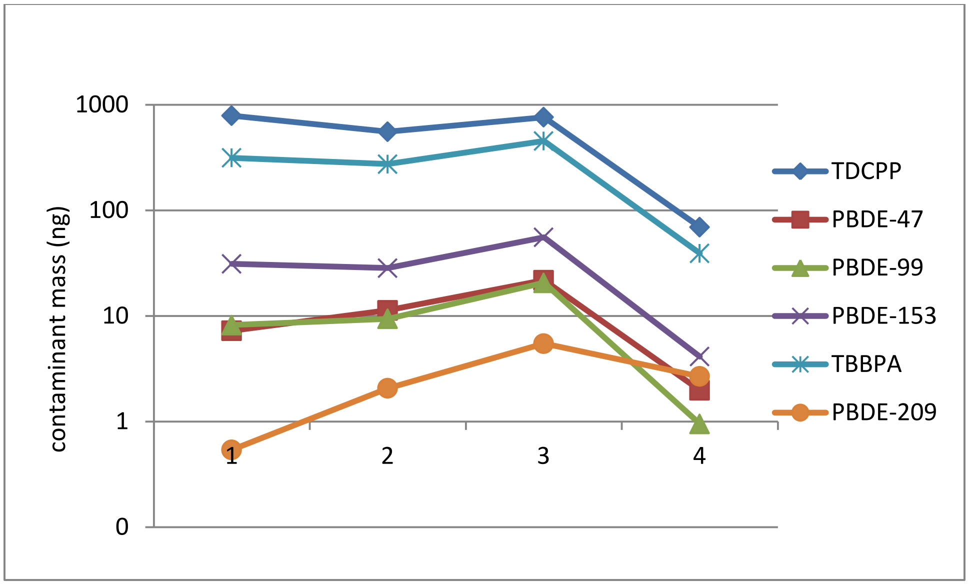

Figure 17: SVOC masses yielded from serial microwave extraction of internal standards for vacuum sample V1

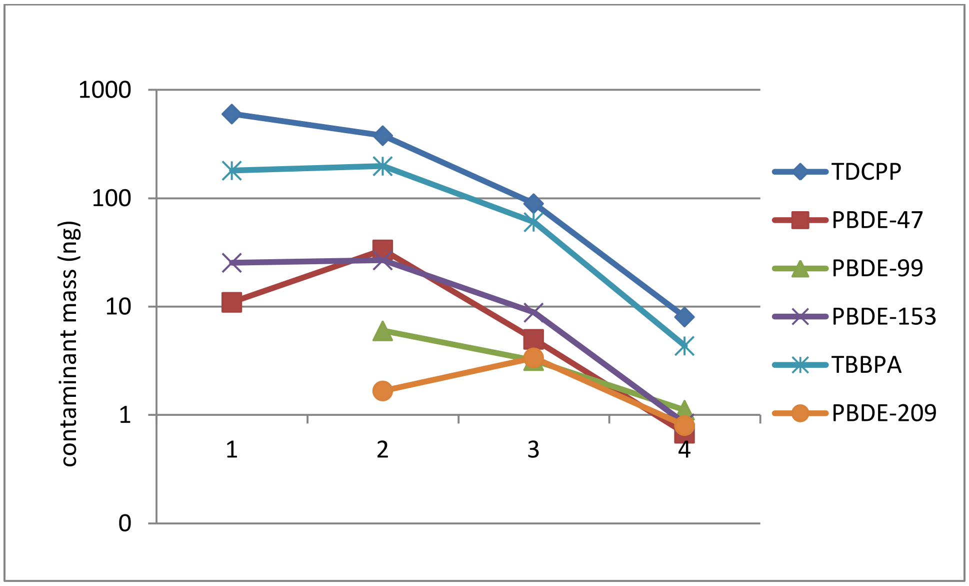

Figure 18: SVOC masses yielded from serial sonication extraction of internal standards for vacuum sample V2

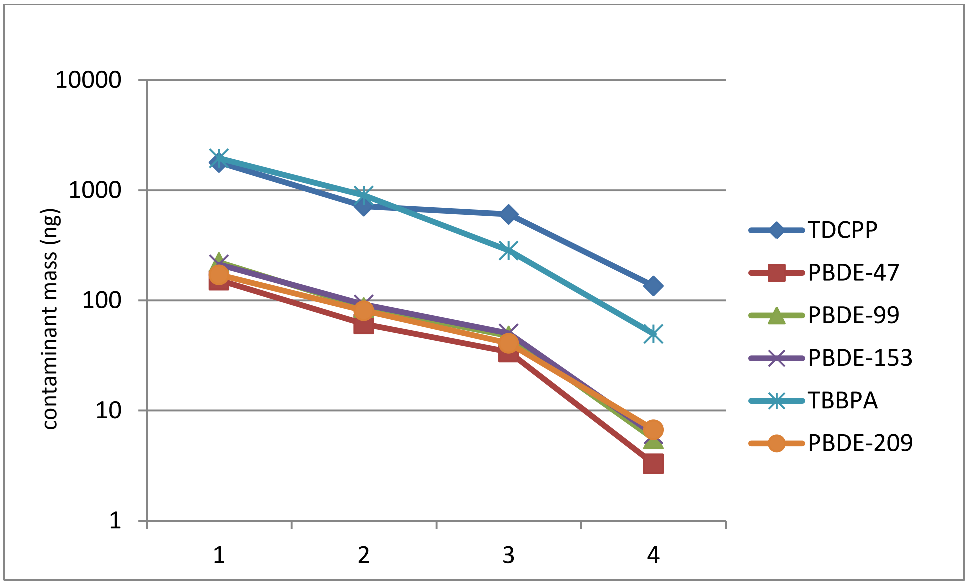

Figure 19: SVOC masses yielded from serial sonication extraction of internal standards for *post-hoc* post-vacuum wipe sample VW1

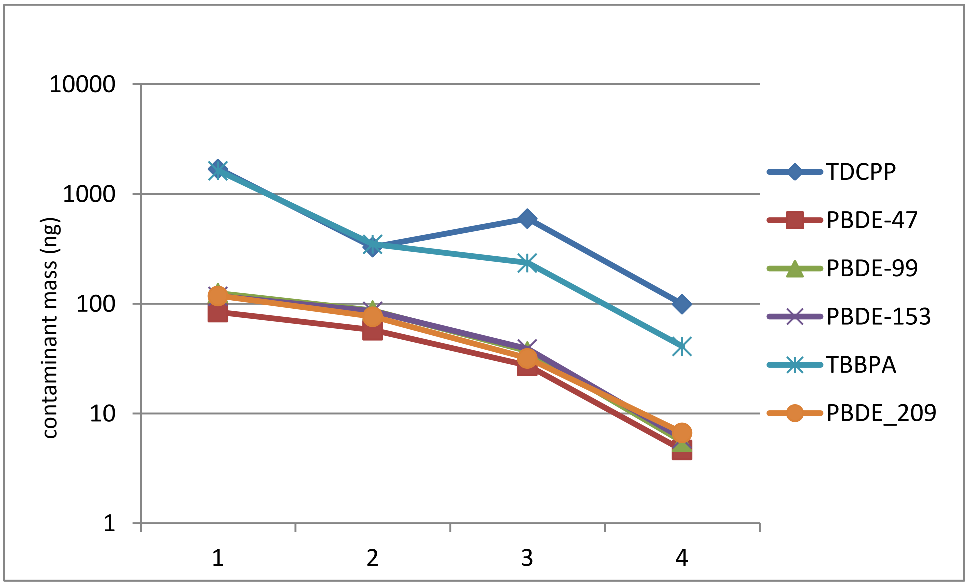

Figure 20: SVOC masses yielded from serial sonication extraction of internal standards for *post-hoc* post-vacuum wipe sample VW2

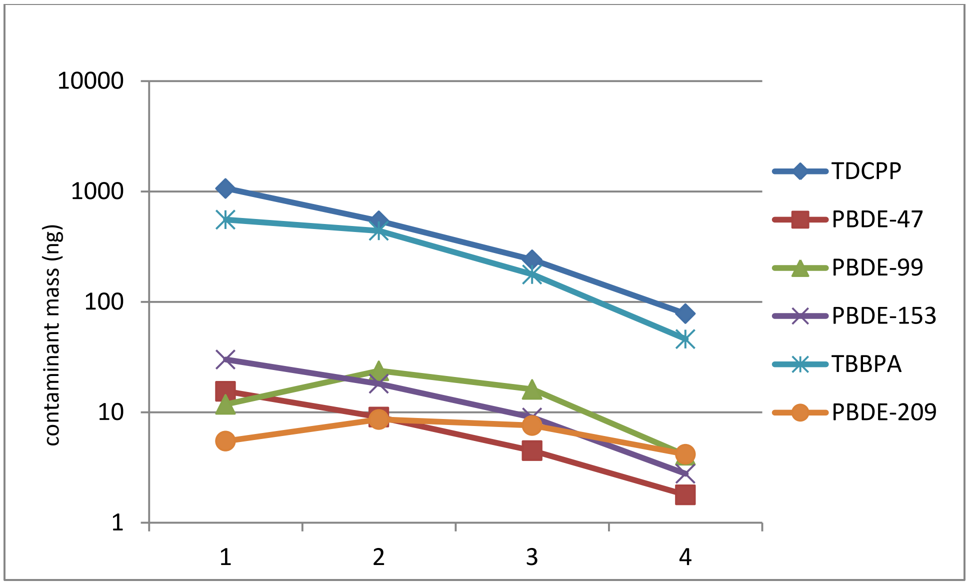

Figure 21: SVOC masses yielded from serial sonication extraction of internal standards for wipe sample W1

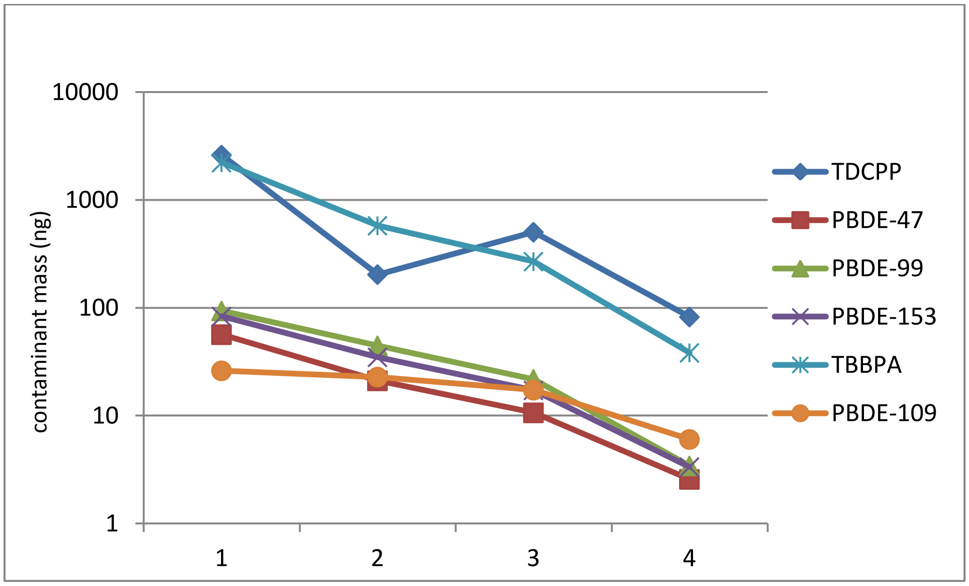

Figure 22: SVOC masses yielded from serial sonication extraction of internal standards for wipe sample W2

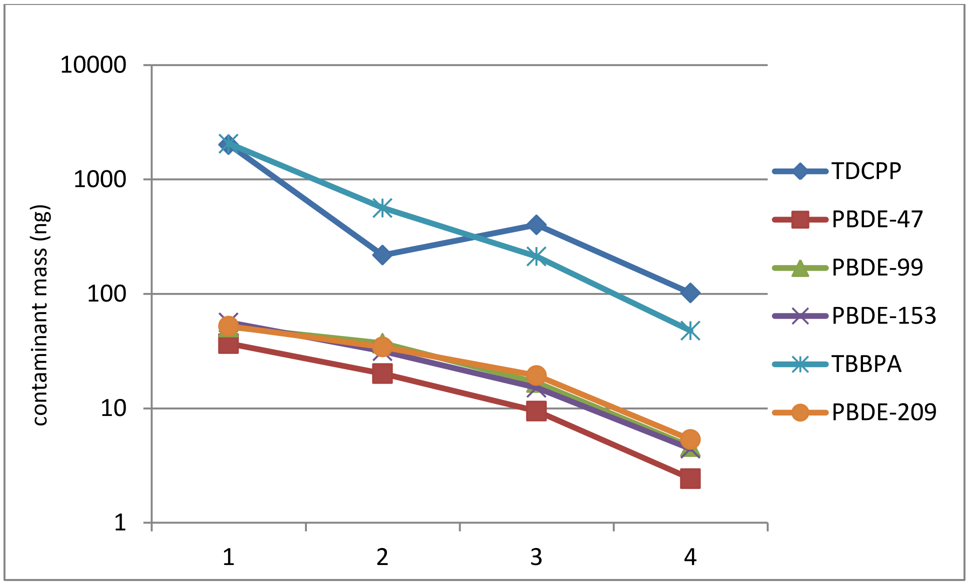

Figure 23: SVOC masses yielded from serial sonication extraction of internal standards for wipe sample W3