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Review

Concentration and factors affecting the distribution of phthalates in the air and dust: A global scenario

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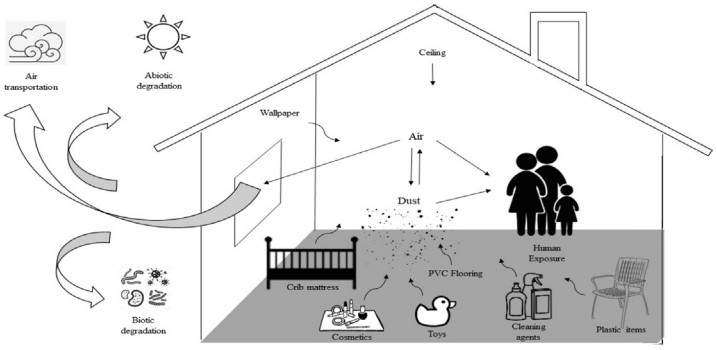
HIGHLIGHTS

- Global phthalate levels in the air and dust along with factors associated with their presence were reviewed.
- Asian countries were found to contain the highest level of phthalates.
- DEHP and DBP were found to be predominant in both air and dust.
- Temperature, air exchange rate and use of PVC materials were found to be strongly associated with presence of phthalates.

GRAPHICAL ABSTRACT

Graphical representation of the occurrence of phthalates in air and dust.



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ABSTRACT

Phthalates are ubiquitously present environmental contaminants. Air and dust are the most important mediums of exposure to phthalates. The present study reviews the presence of phthalates in the air and dust reported from different countries in the last ten years (2007–2017). The phthalate concentrations revealed wide heterogeneity with a mean and median value 6 ± 19 μg/m^3 and 0.5 μg/m^3 respectively in the air and $1.5 \times 10^3 \pm 2.2 \times 10^3$ μg/g and 7.8x10^{2}μg/g respectively in the dust. The highest phthalates levels in the air were reported from India (1.1×10^2 μg/m^3) and in dust from Bulgaria (1.2×10^4 μg/g). Overall higher levels were reported from developing countries as compared to developed countries. Di (2-ethylhexyl) phthalate (DEHP) and Di-*n*-butyl phthalate (DBP) were found to be predominant in both air and dust. Temperature, humidity, air exchange rate, building material and indoor maintenance were reported as the important factors influencing the levels of phthalates in the air and dust. In addition to policy level interventions, reducing the use of phthalate containing materials and controlling the factors which enhance the emission from existing sources can help in reducing human exposure to phthalates.

Contents

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1. Introduction

Phthalates are esters of ortho-phthalic acids (1, 2-benzenedicarboxylic acid), that is produced by the reaction between alcohol and phthalic anhydrite. The hydrocarbon chain derived from alcohols ranging from methanol and ethanol to tridecanol imparts specific properties to each of the phthalates (Fernandez et al., 2012). Phthalate esters can be broadly divided into two groups based on the length of the ester groups' main carbon chain, namely high molecular weight (HWM) phthalates (7–13 carbon atoms) and low molecular weight (LMW) phthalates (3–6 carbon atoms). These are primarily used as plasticizers in polyvinyl chlorides (PVCs) and consumer products such as toys, building material, electronics and medical devices (Sampath et al., 2016). DEP and DMP with <3 carbon atoms are not used as plasticizers and not classified in any group. These are used as solvents and fixatives in fragrances, additives in cosmetics, medical devices, household and personal care products (Katsikantami et al., 2016).

Compared to the long duration of their use in the plasticizer industry, concerns over possible adverse effects of phthalates on human health are relatively recent, dating back to the 1980s (Kamrin and Lansing, 2009). Today, phthalates are known to be reproductive and developmental toxicants (EU, 2005; WHO, 2013). Human bio-monitoring studies have indicated that exposure to phthalates can cause adverse health outcomes like fertility problems, respiratory diseases, childhood obesity and neuropsychological disorders (Katsikantami et al., 2016). Consequently, six phthalates, namely, Di-methylphthalate (DMP), Di-ethylphthalate (DEP), Di-*n*-butyl phthalate (DBP), Butyl-benzyl phthalate (BBP), Di (2-ethylhexyl) phthalate (DEHP), and Di-*n*-octylphthalate (DnOP) have been identified as priority pollutants by the United States Environmental Protection Agency (U.S. EPA) and the European Union (EU). Usage of DEHP, DBP, BBP & DnOP has been limited to ≤0.1% in toys and childcare articles by EU (Directive 2005/84/EC), US (Consumer Product Safety Improvement Act of 2008 (CPSIA, 2008), China (China National Standard GB 6675, 2014), India (BIS, 2011) and Japan (Japan Toy Safety Standard ST-2002 Part 3, 2011). Recently in 2015, DEHP, DBP and BBP were classified as reproductive toxicant category 1B and completely banned for any application without prior approval from EU (ECHA, 2016).

Owing to the strict regulations on the use of phthalates because of human health concerns, there has been a reduction in their production and alternative plasticizers are being manufactured and employed by the plastic industry (Katsikantami et al., 2016). This is clearly visible in declining contribution of phthalates to world consumption of plasticizers from 88% in 2005 to 70% in 2014, forecasted to be 65% in 2019 (IHS Markit, 2015). Bui et al. (2016) reported an increasing trend in use of alternate plasticizers in Sweden over the past decade. However, even with the shift towards the use of alternate non–phthalate plasticizers and higher molecular weight phthalates, the exposure risk from products already containing high levels of phthalates prior to regulation cannot be neglected (Larsson et al., 2010). Large-scale and diverse use of phthalates in consumer products combined with their property of volatilization and leaching have made these chemicals ubiquitously present in the environment in all kinds of media including air, water, dust, sediment and food (Cao, 2010). Phthalates dwell in a state of dynamic flux in the environment, through the processes of emission, transportation, degradation, deposition and exchange between different environmental media. These processes depend upon environmental factors and their physicochemical properties like vapor pressure (Vp), air-water partition coefficient (K_{AW}), octanol-air partition coefficient (K_{OA}), and octanol-water partition coefficient (K_{OW}), organic carbon partition coefficient (K_{OC}) (Teil et al., 2006; Net et al., 2015). The vapor pressure of a particular phthalate and its concentration in a source determines the equilibrium gas-phase concentration of that phthalate, which in turn is a key parameter in influencing source emission rates of phthalates (Liang and Xu, 2014b; Afshari et al., 2004). Octanol-air partition coefficient (K_{OA}), an indicator of hydrophobicity, is another among the key factors which determines partitioning between the gas phase and indoor surfaces/dust (Sukiene et al., 2016). The ability to partition between gas and particle phase leads to a dynamic distribution of phthalates in the air and on surfaces like particles, dust and human skin, which leads to direct exposure through dust ingestion, inhalation and dermal absorption (Bergh et al., 2011; Blanchard et al., 2014). Ingestion of dust and inhalation of indoor air and re-suspended dust are the most important non-food sources of exposure to phthalate esters (Clark et al., 2003; Clausen et al., 2004). In the present review, peer-reviewed literature published in the last ten years i.e. from 2007 to 2017 from different parts of the world reporting phthalate levels in the air and dust was examined. The status of the worldwide presence of six priority phthalates, in two of the most important environmental media viz. air and dust, and the factors affecting their presence are presented and discussed here.

2. Phthalates in the air and dust samples on a global scale

Phthalates are high production volume chemicals that accounted for 70% of the world consumption of plasticizers in 2014. Of this, Asia, Western Europe and the U.S.A accounted for 59%, 14% and 16% respectively of the world plasticizer consumption in 2014 (IHS Markit, 2015). However, phthalate exposure (phthalate metabolite levels in urine) among countries indicates the highest exposure for people living in Europe (2.1×10^2 μg/l) closely followed by USA (2.0×10^2 μg/l) and least in Asia (1.3×10^2 μg/l) (Katsikantami et al., 2016). There exists a discrepancy between trends for industrial consumption and human exposure. Trade might be a plausible explanation, for instance, a large proportion of articles containing phthalates are brought into Sweden via imports (KEMI, 2015). The highest concentrations of phthalates in

articles have been found in the range of 300–461 g/kg for DEHP, 283–345 g/kg for DBP, 150 g/kg for DnOP and 20–33 g/kg for BBP in floorings, shower curtains, gloves, plastic sandals, plastic balls, soap packaging among others (KEMI, 2015).

2.1. Phthalates in the air

Phthalates are distributed worldwide due to atmospheric transport over long distances (Lenoir et al., 2016) with a global presence ranging from remotest regions in the Arctic (Xie et al., 2007) to ant cuticles in isolated rainforests of the Amazon (Lenoir et al., 2016). Phthalates were also reported to be the most abundant SVOC among 58 SVOCs quantified in indoor and outdoor air in Paris (Moreau-Guigon et al., 2016).

2.1.1. Spatial variations in the air

Phthalate levels in the indoor and outdoor air have been studied and reported from several countries. Phthalate levels were found to be higher in indoor air as compared to outdoor air as major sources of phthalates are located indoors and faster degradation takes place in the outdoor environment (Afshari et al., 2004; Rudel and Perovich, 2009; Bergh et al., 2011). Among different locations, concentrations of phthalates were found to be higher in urban sites as compared to suburban/forest sites (Teil et al., 2016). This could be because of anthropogenic activities, increased production and consumption of plastics and environmental conditions leading to volatilization of phthalates that contribute significantly to increased phthalate loading in atmospheric depositions in urban areas (Zeng et al., 2010; Wang et al., 2008).

Phthalates are associated with both gaseous and particulate phases of air. Studies have reported phthalates in either or both phases. Mean concentration of six phthalates reported from different countries and its summary statistics are given in Table 1 (mean of values has been taken for studies reporting levels from multiple sites). The data shows high variability with a mean and median value of 6 ± 19 μg/m^3 and 0.5 μg/m^3 respectively. Worldwide, the highest concentration of six phthalates in indoor and outdoor air was found to be 2.0×10^2 μg/m^3 and 7.3×10^1 μg/m^3 respectively in an industrial area (Okhla) in New Delhi, India (Das et al., 2014), followed by 27.6 μg/m^3 in outdoor air in Hangzhou, China (Zeng et al., 2010). For other countries in Asia, the range varied from 0.8 μg/m^3 (Vietnam) (Tran et al., 2017) to 6.2 μg/m^3 (Japan) (Takeuchi et al., 2014). Relatively lower phthalate levels were reported for developed regions like Europe and North America. Among European countries Sweden was found to have a high phthalate burden in indoor air (2.1 μg/m^3) (Bergh et al., 2011) whereas for other European countries the levels ranged from 0.01 μg/m^3 (Spain) (Aragón et al., 2012) to 0.32 μg/m^3 (France) (Blanchard et al., 2014). Relatively fewer studies have been reported from North America, and phthalate concentrations varied from 0.5 μg/m^3 (U.S.A) (Rudel et al., 2010) to 0.9 μg/m^3 (U.S.A) (Gaspar et al., 2014).

Studies from the Czech Republic (Růžičková et al., 2015), Greece (Salapasidou et al., 2011), Norway (Rakkestad et al., 2007) and Spain (Salgueiro-González et al., 2013; Aragón et al., 2012) reported phthalate levels only in particulate phase. Pei et al. (2013) noted that sampling of a single phase leads to underestimation of phthalate levels in the air. The non-availability of data for phthalates in gaseous phase could be one of the reasons behind the lower levels reported from these countries. Phthalates were also reported from the Arctic (Xie et al., 2007) in the vapor phase of air despite its remoteness and lack of any direct sources. Their presence has been attributed to atmospheric transport and deposition and lower rates of degradation owing to unfavorable conditions in the Arctic. However, Xie et al. (2007) stated the value to be an underestimation due to possible surface adsorption of phthalates by snow and ice.

2.1.2. Profile variations

The percentage contribution of individual phthalates in both phases of indoor and outdoor air is represented in Fig. 1 (mean of all values in Tables 1 and 2 for each phthalate have been taken for calculating percentage). DEHP was found to be the most dominant phthalate in both indoor and outdoor air with a contribution of 40.6% and 29.4% respectively. The contributions of DMP, DEP, DBP, BBP and DnOP ranged between 9 and 15% in indoor air, while for outdoor air their contributions varied from 6 to 19%. The country-wise composition of phthalates in the air (both gaseous and particle phase) was also found to vary widely (Fig. 2) (mean of values for each country reported in Table 1 has been taken for calculating percentage in Fig. 2). Chi et al. (2017), reported levels of the six phthalates in both phases of outdoor air and found that DEP and DMP dominated the gaseous phase while DEHP was the most dominant phthalate in the particulate phase. Similarly, DEHP and DBP were found to be the most abundant phthalates in the atmospheric particulate matter in other studies as well (Xie et al., 2005; Teil et al., 2006; Wang et al., 2008; Kong et al., 2013). The higher concentrations of DBP and DEHP in the particulate matter can be partly attributed to their widespread use in household products (Weschler, 1980). Additionally, the longer half-life of DBP in the atmosphere as compared to DEHP also contributes to its prevalence (Teil et al., 2006). DEP and DMP are more volatile and more likely to be found in the vapor phase. A decrease in vapor pressure (increase in molecular weight) leads to an increase in the fraction associated with airborne particles and a simultaneous decrease in the fraction present in the gaseous phase (Weschler et al., 2008). With regard to phase distribution, the majority of phthalates were reported to exist in the vapor phase as compared to particulate phase (Wang et al., 2008; Song et al., 2015). However, Chi et al. (2017) found that the contribution of gaseous or particulate phase to the total phthalate concentration varied according to different microenvironments.

2.1.3. Seasonal variations

An increase in temperature increases the emission rates of non-covalently bound phthalates from their polymer matrices resulting in a higher concentration in warmer months (Ma et al., 2014; Zhu et al., 2016). In good agreement, a number of studies have found higher phthalate levels in summer in indoor as well as outdoor particulate matter (Zhang et al., 2014; Ma et al., 2014; Zhu et al., 2016). The mean concentration of phthalates in outdoor particulate matter (PM) in summer months in Shanghai, China (2.1×10^2 ng/m^3 in PM_{10} and 7.2×10^1 ng/m^3 in $PM_{2.5}$) were found to be significantly higher than those in winter months (7.7×10^1 ng/m^3 in PM_{10} and 5.0×10^1 ng/m^3 in $PM_{2.5}$) (Ma et al., 2014). Similarly, Zhu et al. (2016) reported phthalate concentrations in the air of Tianjin, China during summer to be 13.9 ± 4 ng/m^3 and 23.4 ± 12.3 ng/m^3 in PM_{10} and $PM_{2.5}$ as compared to 5.4 ± 4.6 ng/m^3 and 4.2 ± 1.9 ng/m^3 in PM_{10} and $PM_{2.5}$ respectively during winter. Even though phthalate emission shows a direct linear relationship with the increase in temperature; studies have reported contradictory results with respect to seasonal variations of phthalates in the air. Wang et al. (2008) reported higher values in gaseous phase ($1.2 \times 10^2 \pm 8.4$ ng/m^3) and particulate phase (29.8 ± 2.4 ng/m^3) in cold weather (January) as compared to a concentration of 18 ± 1.5 ng/m^3 and 2.4 ± 0.5 ng/m^3 in gaseous and particulate phase respectively during hot weather period (July) in Nanjing, China. Another study from Tianjin, China by Kong et al. (2013) reported phthalate concentrations to be higher in winter than those in spring and summer for the atmospheric particulate matter in seven urban sites. Similarly, 5–10 times higher concentration of phthalates in $PM_{2.5}$ was found in winter (heating season) as compared to summer in different locations in the Czech Republic. The highest difference was 44 times in one of the sites (Růžičková et al., 2015). Location-specific differences in seasonal variations of phthalates were reported by Sampath et al. (2016). The mean concentration of phthalates in urban and agricultural locations was found to be in the order of summer > pre-monsoon > monsoon. A different pattern was observed for suburban and coastal locations, with pre-monsoon concentrations 2–3 times higher than that in summer and monsoon. The air in the monsoon season was found to contain the

Table 1
Mean concentrations (in ng/m^3) of six priority phthalates in air reported from different countries (east to west).

	Country	Sampling location	Phase	n	DMP	DEP	DBP	BBP	DEHP	DnOP	Total	Reference
				Vp	3.08×10^{-3}	2.1×10^{-3}	2.01×10^{-5}	8.25×10^{-6}	1.42×10^{-7}	1.0×10^{-7}		
ASIA	Japan	Homes	G + P	21	19	33	67	22	86	5.5	232.5	Takeuchi et al., 2015[a]
	Japan, Sapporo	Homes, cars, kindergartens, laboratories, offices and hair salons	G + P	40	47.9	60.7	200	<dl	147	n.r.	455.6	Kanazawa et al., 2010[a]
	Japan, Sapporo	Homes	G + P	6	181.17	408.33	3611.67	45.83	1928.3	0.83	6176.13	Takeuchi et al., 2014
	China	Apartment	G	28	1770	340	740	10	730	n.r.	3590	Bu et al., 2016
	China	Bus, subway, taxi, car	G + P	235	6348.7	4627.65	2689.97	3401.55	5289.48	2332.96	24,690.31	Chi et al., 2017
	China, Shanghai	University campus (suburban area)	G + P	77	1.82	0.9	39.67	1.87	318.7	n.r.	362.96	Ma et al., 2014
	China	Office, residential building	G + P	28	610	n.r	1650	n.r	1510	n.r.	3770	Wang et al., 2014
	China, Tianjin	Residential, traffic, industrial, commercial area	P	21 (PM_{10}); p15 ($PM_{2.5}$)	1.42	1.03	21.62	0.23	173.97	1.16	199.43	Kong et al., 2013
	China, Tianjin	Urban	P	16	0.32	0.15	4.34	0.04	18.39	0.16	23.4	Zhu et al., 2016
	China, Tianjin	Homes	G + P	13	368.36	54.52	573.47	0.59	71.7	0.12	1068.76	Zhang et al., 2014
	China, Hangzhou	Homes	G + P	10	1455	2290.2	1938.6	3975	2438	n.r.	12,096.8	Pei et al., 2013
	China, Hangzhou	Office	G + P	10	815.55	1042.75	1099.2	665.33	1125.4	n.d.	4748.23	Song et al., 2015
	China, Guangzhou	Urban, Suburban	G + P	–	425.33	340	2592	115.33	23,736.67	377.33	27,586.66	Zeng et al., 2010
	China, Nanjing, Jianging	Urban, Suburban	G + P	10	6.4	2.3	37.2	2.1	13.85	0.7	62.55	Wang et al., 2008
	China, Xian	Urban	P	15	n.r	n.r	37	17	164	n.r.	218	Li et al., 2014
	Vietnam	Homes, cars, kindergartens, laboratories, offices, and hair salons	G + P	97	39.5	376	133	16.4	187	10.9	762.8	Tran et al., 2017
	India	Urban, Suburban, Coastal, Agricultural site	G	31	7.83	21.58	78.50	14.5	204.92	6	333.33	Sampath et al., 2016
	India	Urban	P	49	0.07	4.52	33.65	n.r	275	n.r.	313.23	Fu et al., 2010
	India	Urban	P	13	n.r	n.r	55	17	689	n.r.	761	Li et al., 2014
	India, New Delhi	Urban	G + P	40	11,101.48	12,759.5	9201	11,659.75	50,764	12,625.25	108,110.98	Das et al., 2014[b]
EUROPE	France	Homes	G + P	30	8.2	158	99.9	3.7	51.5	n.r.	321.3	Blanchard et al., 2014[a]
	France, Several sites	Urban	G + P	6	n.r	3.35	2.25	n.r	27.28	n.r	32.88	Teil et al., 2016
	Greece	Urban- industrial, urban -traffic	P	20	<dl	<dl	1.55	0.5	12.03	<dl	14.08	Salapasidou et al., 2011
	Sweden	Home, day care, work	G + P	30	9.6	1170.33	735.33	21	197.67	n.r.	2133.93	Bergh et al., 2011
	Norway	University, school, kindergarten, dwelling	P	14	n.r	n.r	149	21	24	n.r.	194	Rakkestad et al., 2007
	Czech	Urban	P	–	n.r	3	12.28	n.r	1.69	n.r.	16.97	Růžičková et al., 2015
	Germany	Day care centre	G + P	63	163	208	283	n.r	276	n.r.	930	Fromme et al., 2013
	Spain	Harbour	P	10	0.39	2.14	<dl	0.04	5.91	0.11	8.58	Aragón et al., 2012
	Spain	Urban, suburban, industrial	P	8	0.17	1.02	13.75	n.r.	0.73	n.r.	15.67	Salgueiro-González et al., 2013
	Arctic	–	G + P	–	0.11	0.43	0.43	0.08	0.76	n.r.	1.81	Xie et al., 2007
N. AMERICA	U.S.A, California	Child care facility	G	40	n.r	210	520	100	100	n.r.	930	Gaspar et al., 2014[a]
	U.S.A, California	Homes	G + P	50	n.r	330	140	6.8	68	n.d.	544.8	Rudel et al., 2010[a]
	U.S.A, Albany	Home, office, lab, schools, salons, public places	G + P	60	15.9	445	112	9.25	85.3	<mql	667.45	Tran and Kannan, 2015
Summary statistics		Mean			835.61	858.43	814.31	694.03	2749.16	903.59	6102.25	
		S.D.			2362.25	2475.01	1762.22	2313.18	9574.85	3073.09	19,424.59	
		Median			12.75	60.7	112	10	164	0.7	455.6	
		5th Percentile			<dl	0.26	1.1	<dl	1.32	<dl	11.88	
		95th Percentile			4746.15	3692.67	3058.65	3745.62	12,668.36	4391.42	25,848.85	

n: Sample size; Vp: Vapor pressure (in mm Hg) at 25 °C; G: Gaseous phase, P:Particle phase; n.r.: not reported; n.d.: not detected; mql: method quantification limit; dl: detection limit. Average concentrations of all sites were taken for studies reporting several sites; Values for G and P were added for papers reporting them separately. Statistical summary calculated as n.d. = 0; <dl = 0.

[a] Studies reporting median values.

[b] Values for Particulate matter from quartz fiber filter were reported as "dust".

least amount of phthalates due to removal by seasonal precipitation (Sampath et al., 2016). Hydrolysis of phthalates on substrates at high relative humidity could be another reason (Westberg et al., 2009).

Temperature plays a vital role in both volatilization and hydrolysis of phthalates and therefore in determining the amount of phthalates present in the atmosphere. Studies that have found higher phthalate levels in summer have accounted temperature as a sole factor responsible for the seasonal variability of phthalate esters in the atmosphere. The reported increase in winter concentration was explained by decreased photochemical reactions with free radicals as opposed to summers, lower diffusion of pollutants due to steady atmosphere and heavier inversion layer in case of mountain surrounded region (e.g. Nanjing), and burning for heating purposes including coal burning which might be a source of DBP and DEHP and combustion of plastic and waste incineration (Wang et al., 2008; Kong et al., 2013; Růžičková et al., 2015).

2.2. Phthalates in dust

Dust in indoor environments is a sink for semi-volatile compounds like phthalates which settle over a long period of time (Butte and Heinzow, 2002; Abb et al., 2009). Direct transfer due to source-sink contact, abrasion from sources, and diffusion via airborne particles contribute to the presence of phthalates in the dust (Schripp et al., 2010; Sukiene et al., 2016). Studies have reported phthalate levels in indoor dust from different places like homes (Gevao et al., 2013; Ait Bamai et al., 2014; Langer et al., 2010), day care centers/nursery schools (Fromme et al., 2013; Gaspar et al., 2014; Kim et al., 2013) retail stores (Xu et al., 2014), offices (Wang et al., 2014; Tran et al., 2016), and dormitories (He et al., 2015; Li et al., 2016). Phthalates have also been reported in the outdoor dust (Lan et al., 2012; Wang et al., 2014; Škrbić et al., 2016). Phthalate levels in outdoor dust were found to be lower than indoor dust by several orders of magnitude.

2.2.1. Spatial variations in dust

The mean concentration of six phthalates reported from different countries and summary statistics are given in Table 2. The highest concentration of six phthalates in dust was reported from Bulgaria (1.2×10^4 μg/g) (Kolarik et al., 2008), followed by China (4.7×10^3 μg/g) (Bu et al., 2016), South Korea (4.1×10^3 μg/g) (Kim et al., 2013) and Japan (3.5×10^3 μg/g) (Ait Bamai et al., 2014). The lowest mean concentration in dust was reported in Vietnam (29.3 μg/g) (Tran et al., 2016). Phthalate levels reported for developed countries like the U.S.A., Canada and France were nearly four times lower as compared to levels reported for Asian countries, which were among the highest worldwide. Mean and median phthalate concentrations in dust were found to be 1.5 $\times 10^3 \pm 2.2 \times 10^3$ μg/g and 7.8×10^2 μg/g respectively. Large variability was observed in levels of phthalates reported from different countries and various sites within the same country. In addition to differences in sampling and analytical procedures, differences in degree of economic development and consequent consumption patterns and lifestyle could be the primary reason for the observed range of concentrations.

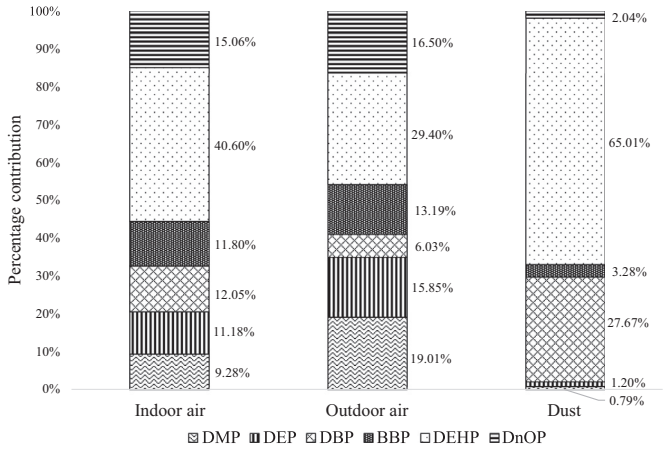

Fig. 1. Percentage contribution of each phthalate in indoor air, outdoor air (for studies reporting both phases i.e. G: gaseous phase + P: particle phase) and dust. (Mean of all values in Tables 1 and 2 for each phthalate have been taken for calculating percentage.)

2.2.2. Profile variations

DEHP was found to be the dominant phthalate in dust in majority of the countries, with the exceptions of Bulgaria, Italy and Greenland, where concentrations of DBP were found to be highest among the six phthalates (Fig. 3) (mean of values for each country reported in Table 2 has been taken for calculating percentage in Fig. 3). Among different sites, childcare centers accounted for the highest amount of DEHP (Bergh et al., 2011; Bi et al., 2015b). The percentage contributions of six phthalates in dust are represented in Fig. 1 (mean of all values in Tables 1 and 2 for each phthalate have been taken for calculating percentage). Overall DEHP accounted for 65% of phthalates in the dust, followed by DBP at 28%, while other phthalates contributed between<1–2%. DEHP and DBP have been found to be predominant in the household dust because of their usage in plastic and PVC products. This combined with their relatively high log [K_{oa}] values (log [K_{oa}] for DEHP and DBP are 12.6 and 8.6 respectively) lead to higher sorption from the atmosphere to aerosol particles and other surfaces (Cousins and Mackay, 2000). Additionally, DEHP is also commercially dominant phthalate covering 50% of the plasticizer market (KEMI, 2015). BBP and DnOP showed considerably lower presence in the dust, while the relatively more volatile DEP and DMP accounted for the least presence in the dust as these are also mainly used in non-PVC products like cosmetics and perfumes.

3. Factors influencing levels of phthalates in indoor environments

Phthalate levels build up over time in indoor environments where their main sources like children's toys, cosmetics, flexible PVC flooring, and cable insulation among others are found.

(Abb et al., 2009; Bornehag et al., 2004; Fromme et al., 2013; Gevao et al., 2013). Several studies have explored the various factors affecting the levels of phthalates in the environment to understand the behavior of phthalates under different conditions and improve strategies to limit exposure (Table 3). These factors can broadly be divided into environmental factors like temperature, humidity, air exchange rate and physical/cultural factors like building materials and characteristics, and indoor maintenance.

3.1. Temperature

Temperature is a strong determinant of the fate of phthalates in indoor environments as phthalate emission and sorption processes depend strongly on temperature. Higher temperature increases emission rates of phthalates from sources (Liang and Xu, 2014a). However, different phthalates are affected differently by temperature (Fujii et al., 2003; Wang et al., 2008). Positive correlations have been found between room temperature and indoor air concentrations of higher vapor pressure phthalates like DEP and DBP (Gaspar et al., 2014). Emissions of DEP, DBP and DEHP from plastic materials were found to increase by several orders (0.9 to 2.8 μg/m^2 h for DEP, 0.8 to 4.5×10^2 μg/m^2 h for DBP, 14 to 1.5×10^3 μg/m^2 h for DEHP) with an increase in incubator temperature from 20 °C to 80 °C (Fujii et al., 2003). Similarly, an increase in temperature from 22 °C to 62 °C led to 238 times increase in steady state concentration of DEHP emissions from PVC flooring (containing 13% DEHP) in Field and Laboratory Emission Cell (FLEC) study (Clausen et al., 2012). In a test house, total airborne concentrations of BBP and DEHP were found to be sensitive to indoor temperatures and their steady-state concentration levels increased by three times with an increase in temperature from 21 °C to 30 °C in a test house (Bi et al., 2015a). However,

Table 2
Mean concentrations (in μg/g) of six priority phthalates in dust reported from different countries (east to west).

	Country	Sampling location	N	DMP	DEP	DBP	BBP	DEHP	DnOP	Total	Reference
ASIA	Japan	Home	19	0.1	0.2	30	4.7	1700	4.6	1739.5	Takeuchi et al., 2015[a]
	Japan, Sapporo	Home(floor, multi-surface)	128	<dl	<dl	50.6	5.9	3400	n.r.	3456.5	Ait Bamai et al., 2014[a]
	Japan, Sapporo	Home(floor, multi-surface)	41	<dl	0.68	42.1	6.6	2080	n.r.	2129.38	Kanazawa et al., 2010[a]
	China	Office/home	28	5.66	n.r.	447.78	n.r.	786.61	n.r.	1240.05	Wang et al., 2014
	China, Guangzhou, Hong Kong	Outdoor, indoor	160	2.39	3.13	9.42	6.8	407.1	4.08	432.92	Wang et al., 2013
	China, Nanjing	Settled house dust	215	0.4	0.9	52.3	2.9	462	1.6	520.1	Zhang et al., 2013
	China, Nanjing	Home, office, PME, dorms	33	1.91	0.22	99.6	0.77	557	n.r.	659.5	He et al., 2015
	China, Guangzhou	Home, office, apartment, road surfaces	50	1.68	1.47	59.38	1.86	1011.5	5.48	1081.37	Lan et al., 2012
	China, several sites	Home	30	10.9	30.6	408.4	1.0	4245	n.r.	4695.9	Bu et al., 2016
	China, several sites	Home	75	0.2	0.4	20.1	0.2	228	0.2	249.1	Guo and Kannan, 2011[a]
	China, several sites	Workplace,home, manufacturing plant, hospital, office, secondary school and shopping mall	78	0.6	1.81	66.67	13.98	828.75	11.04	922.85	Kang et al., 2012[a]
	S. Korea	Nursery school	50	3.1	24.6	66.9	60.7	3950	n.d	4105.3	Kim et al., 2013
	Taiwan	Home	101	0.1	1.0	20.2	1.0	753.3	n.r.	775.6	Hsu et al., 2012[a]
	Vietnam	Home, shop, laboratory, office	46	0.06	0.04	1.24	1.18	26.4	0.42	29.34	Tran et al., 2016
	Saudi Arabia	Floor dust, car dust, AC filter	15	1.33	3.27	56.07	2.1	1203.33	61.07	1327.17	Albar et al., 2017
	Kuwait	Floor dust, AC filter	15	0.2	6.65	4.3	2.75	265	2.9	281.8	Albar et al., 2017
	Kuwait	Home	21	0.03	1.8	45	8.6	2256	14	2325.43	Gevao et al., 2013[a]
EUROPE	France	Home settled dust	30	0.2	2.8	11.9	8.5	289	n.r.	312.4	Blanchard et al., 2014[a]
	France	Home	300	n.r.	3.3	10.8	10.3	341	n.r.	365.4	Mandin et al., 2014[a]
	Bulgaria	Home	177	280	340	9930	340	1050	300	12,240	Kolarik et al., 2008[a]
	Serbia	Street dust	30	0.009	0.006	0.071	0.015	1.22	0.01	1.33	Škrbić et al., 2016
EUROPE	Sweden, Stockholm	Preschool	100	0.61	15	250	24	470	n.r.	759.61	Larsson et al., 2017
	Sweden, Stockholm	Home, work, day care	30	0.27	18.3	156.67	32.33	1493.33	n.r.	1700.9	Bergh et al., 2011
	Sweden, Stockholm	Apartments	62	0.47	14	103	16	449	n.d.	582.47	Luongo and Ostman, 2016[a]
	Italy, Palermo	Home	12	15	31	799	99	304	41	1289	Orecchio et al., 2013
	Denmark	Home, day care	648	n.r.	1.95	26.5	10.35	355	n.r.	393.8	Langer et al., 2010[a]
	Germany, Halle/Saale	Home	30	n.r.	n.r.	87.4	15.2	604	n.r.	706.6	Abb et al., 2009[a]
	Germany	Day care centre	63	0.7	3.4	30	21	1973	n.r.	2028.1	Fromme et al., 2013
	Greenland	Home	43	30	10	790	60	130	20	1040	Kolarik et al., 2007[b]
N. AMERICA	Canada	Home	126	0.12	2	16.8	42.3	462	n.r.	532.22	Kubwabo et al., 2013[a]
	U.S.A, Delaware	Apartment, home, garage, gym, office, store, day care centre	43	0.14	3.6	255	494	637	n.r.	1389.74	Bi et al., 2015b
	U.S.A, California	Child care facility	39	n.r.	1.4	13.7	46.8	172.2	n.r.	234.1	Gaspar et al., 2014[a]
	U.S.A, Pennsylvania	Retail store filter dust	14	<dl	52	213	272	730	88	1355	Xu et al., 2015
	U.S.A, Albany	Home	33	0.08	2	13.1	21.1	304	0.4	340.68	Guo and Kannan, 2011[a]
	U.S.A, California	Apartments, community hall	11	n.r.	n.r.	n.r.	n.r.	386	n.r.	386	Hwang et al., 2008
Statistical summary	Mean		11.88	18.05	417.26	49.51	980.31	30.82	1475.12		
	S.D.		51.01	59.94	1692.85	108.61	1069.37	71.48	2171.72		
	Median		0.34	2.4	51.45	10.3	557	4.34	775.6		
	5th Percentile		<dl	0.03	3.23	0.54	98.92	<dl	172.67		
	95th Percentile		23.25	40.45	793.15	299.2	3565	119.8	4282.48		

N: Sample size; dl: detection limit; n.d.: not detected; n.r: not reported.
Average concentrations of all sites were taken for studies reporting several sites. Statistical summary calculated as n.d. = 0; <dl = 0.
[a] Studies reporting median values.
[b] Studies reporting Geometric mean.

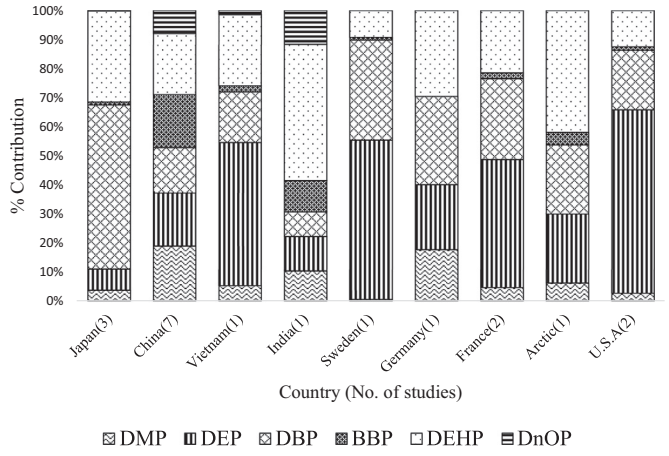

Fig. 2. Percentage contribution of each phthalate in the air (Gaseous + Particle) in different countries. (Mean of values for each country reported in Table 1 has been taken for calculating percentage.)

temperature was not found to influence phthalate concentrations in dust as it was nullified by increased gas-phase concentration and decreased dust/gas sorption partitioning with increasing temperature (Bi et al., 2015a). Migration of DEHP to dust from flooring material was found to increase (from n.d. to $1.8 \times 10^2 \pm 29$ μg/g) with increase in temperature from 25 °C to 50 °C under surface heating conditions for 3 days in FLEC (Jeon et al., 2016). Likewise, an increase in the temperature range from 25 °C to 35 °C of vinyl flooring and crib mattresses increased the gas-phase concentration of phthalates by more than a factor of 10. Temperature influenced phthalate release from crib mattress is a critical exposure factor for infants who remain in close contact with the mattress as body heat leads to increased transfer of phthalates (Liang and Xu, 2014a). The emission rates of DEHP from PVC insulations were found to match that of pure liquid DEHP and was found to be controlled by diffusion at elevated temperatures (about 100 °C) (Ekelund et al., 2008). Such elevated temperatures do not constitute ambient conditions but could become significant exposure factors for products which are heated by convection, like foods microwaved in plastic containers or heated up with the use of electronic items. Surface temperature-dependent increase in phthalates also has important implications for residences with heated flooring systems (Jeon et al., 2016). In cars, left in the sun for long periods, DEHP concentrations can reach levels that are significantly higher than recommended levels (Fujii et al., 2003). However, Sukiene et al. (2016), using deuterium-labeled phthalates as tracers in artificially doped consumer products, found no effect of temperature on phthalate emission. According to the authors, this could be due to the short duration of the experiment or relatively lower temperatures (30 °C) of exposure.

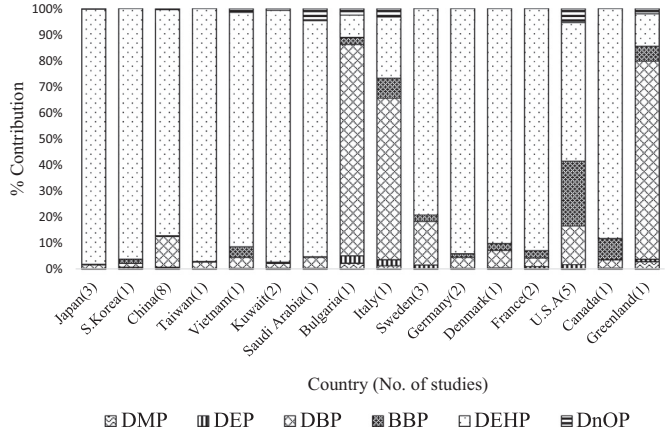

Fig. 3. Percentage contribution of each phthalate in the dust in different countries. (Mean of values for each country reported in Table 2 has been taken for calculating percentage.)

In indoor environments, an increase in temperature leads to an increase in emission of phthalates from sources. Elevated temperatures considered in various studies do not represent ambient temperature but it is important in the case of heating of products containing phthalates. However, the influence of temperature on phthalate emissions in dust requires further investigation.

3.2. Humidity

Humidity variations have been known to enhance PVC degradation by altering its polymer structure (Afshari et al., 2004; Zhang and Chen, 2014). The cracking in PVC coating film leads to decrease in diffusion distance and increase in the area of diffusion, which according to Fick's law, results in the higher emission of DEHP (Hsu et al., 2017). Water leakage history and dampness in buildings has been positively associated with the higher presence of phthalates in the dust (Bornehag et al., 2005; Zhang et al., 2013). However, in the majority of field studies, no significant associations were observed between relative humidity and phthalate concentrations in the air (Gaspar et al., 2014; Bu et al., 2016) and dust (Kolarik et al., 2008; Kim et al., 2009). Likewise, Specific Emission Rate of DEHP from PVC flooring was found to be independent of relative humidity in a one-year FLEC study (Clausen et al., 2007). According to the authors, site occupation in the presence of water vapor and the hydrophobic nature of DEHP decreased sorption of phthalates in the dust.

However, the effect of humidity in the air might be different from humidity in the source material as the moisture content (MC) of materials has been found to influence the migration of DEHP from polymeric materials by accelerating the diffusion of DEHP within polymer (Hsu et al., 2017; Wei et al., 2018). Higher levels (35.3%) of DEHP were found to be released into air and dust from damp plastic wallpapers (MC at 52.3%) as compared to dry wallpapers (MC at 3.6%) in a 15-day closed chamber study (Hsu et al., 2017).

Higher relative humidity also leads to hydrolysis of phthalates which results in a gradual decrease in concentration of phthalates in the source as well as sinks over a long duration (Staples et al., 1997; Westberg et al., 2009; Ekelund et al., 2010). While the effect of relative humidity on the emission of phthalates is still unclear, moisture in the source material has been found to increase the migration of DEHP within the polymer and to air and dust.

3.3. Air exchange rate

Ventilation influences airborne particle concentration, which in turn strongly enhances convective mass transfer, SVOC emission and transport (Liu et al., 2015; Xu et al., 2009). The relationship between air exchange rate and emission rate/concentration of SVOCs have been investigated in several studies in experimental chambers/modelled rooms. Liu et al. (2015) found that increasing the air exchange rate by a factor of three decreased the steady-state concentration of SVOCs by 60%, and consequent human exposure by 57–90% via inhalation and 59–116% via dermal and dust ingestion. Xu and Zhang (2011) found that an increase of the air change rate (from 0.5/h to 1.5/h) reduced the total airborne concentrations of SVOC from vinyl flooring in a modelled room. Typical ventilation systems in commercial buildings are ineffective in controlling concentrations of SVOCs with high Koa (log [Koa] > 12) as they remain attached to particles (Parthasarathy et al., 2012). Among the six phthalates, only DEHP has log [Koa] > 12. In an experimental chamber, the gas-phase concentration of DEHP was reported to be very sensitive to the chamber airflow rate with 50% lower concentration at higher air flow rate of 3×10^3 ml/min as compared to 3×10^2 ml/min. However, a decrease in the gas phase was found to be offset by increased emission rate (Liang and Xu, 2015). Specific emission rate (SER) of DEHP from vinyl flooring was

Table 3
Factors affecting the presence and distribution of phthalates.

Matrix	Study Parameter	Outcome	Reference
Dust	Type of flooring, Temperature	DEHP migration from uncoated residential floor, commercial wax floor; no DEHP migration in UV cured floor; Increase in migration under heated condition.	Jeon et al., 2016
Dust	PVC materials; Building age	No association with PVC/plastic materials, age of building, ventilated duration, number of people.	Li et al., 2016
Dust	Plastic flooring materials; floor care agents	Positive associations between plastic flooring materials and floor care chemical products with phthalate levels.	Bi et al., 2015b
Dust (floor &multi surface)	PVC flooring, PVC wall paper, PVC ceilings	Significant positive association with PVC interior materials and PVC flooring (DEHP); higher DEHP levels in homes with PVC flooring, PVC wallpaper and PVC ceilings; association of phthalates in multi - surface dust with the interior surface materials and floor dust with flooring materials.	Ait Bamai et al., 2014
Dust	PVC flooring area (using XrF analyzer); construction period, indoor maintenance	Significant association with area of PVC flooring (DEHP) and construction period (DEHP); no association with cleaning performance.	Kim et al., 2013
Dust	Flooring material, Floor Wax, Dampness, Humidifier use, Cosmetic products	Significant positive association with solid floor wax(DBP,BBP,DEHP), dampness (BBP, DEHP, DnOP and total phthalate levels), cosmetic and personal care product (DEP), humidifier use (DnOP), and plastic materials.	Zhang et al., 2013
Dust	PVC in flooring and wall paper material, construction period, water leakage	Significant association with PVC flooring and wall paper (DEHP) and construction age (DEHP).	Kim et al., 2009

found to be strongly and positively dependent on the air exchange rate in FLEC (Clausen et al., 2010). Similarly, Xu et al. (2009) found that ventilation rate strongly influenced DEHP emission rates from vinyl flooring in a two-room model. At elevated temperature (100 °C), the evaporation rate of DEHP from pristine DEHP and from plasticized PVC was found to increase with increasing ventilation rate until the constant value was reached at high gas flow rates (Ekelund et al., 2010).

Compared to studies conducted in an experimental chamber, studies in real indoor environments are affected by external factors like individual phthalate's sensitivity to ventilation rate, phthalate concentrations in dust and other building-related factors (Kolarik et al., 2008; Bornehag et al., 2005). Higher DEHP and DBP levels were found in homes that received >4 h of natural ventilation than houses that received 1–4 h ventilation (Zhang et al., 2013). Kim et al. (2013) found significantly higher DBP concentrations in dust sampled from naturally ventilated rooms. Elevated air exchange rates in retail stores slightly raised the concentration of phthalates. This might be because the ventilation system itself served as a source or due to uncertainties associated with a smaller sampling period (4 h as opposed to 48 h) (Xu et al., 2014). High ventilation rates could also lead to higher exposure because of high probability for particles to deposit on surfaces and increased resuspension of the deposited particles because of high air velocity above surfaces (He et al., 2005). Association between ventilation and levels of urinary phthalate metabolite among children in Shanghai was studied and it was observed that natural ventilation was inefficient in reducing levels of urinary phthalate metabolites and might also lead to an increased phthalate exposure (Liao et al., 2018). Kolarik et al. (2008) however reported an increase in DBP concentration with decreasing ventilation rates. Few studies found no statistically significant association between ventilation rate and phthalate levels in the air (Afshari et al., 2004; Gaspar et al., 2014) and dust (Bornehag et al., 2005; Kim et al., 2009).

While air exchange rates influence SVOC concentrations, ventilation systems alone were found to be ineffective in controlling exposure to phthalates as they tend to partition and remain sorbed to particles. Particle filtration in combination with ventilation can be useful for mitigating SVOC exposure (Xu and Zhang, 2011; Ye et al., 2017). Similarly, mechanical ventilation in addition to 'natural ventilation and air cleaner' has been recommended for reducing indoor phthalate exposure levels (Liao et al., 2018).

3.4. Building material and characteristics

Correlations have been found between PVC materials such as floorings and wallpapers used in houses, schools and levels of DEHP and BBP in the dust (Bornehag et al., 2005; Kim et al., 2009; Kim et al., 2013; Ait Bamai et al., 2014). On the contrary, in some other studies, a significant association was not observed between the presence of vinyl flooring and phthalates in the dust (Gaspar et al., 2014; Kolarik et al., 2008) and PVC materials (Li et al., 2016). However, Kolarik et al. (2008) attributed their observation to be influenced by the masking effect of stronger sources like polishing products that were being simultaneously investigated. Direct contact between PVC flooring and dust was found to enhance the emission of DEHP (nearly four times) from PVC by increasing external concentration gradient above the PVC surface in a 68-day experiment in CLIMPAQ (Chamber for Laboratory Investigations of Materials, Pollution, and Air Quality) (Clausen et al., 2004). Similarly, DEHP levels were found to be 10 times higher (2.1×10^3 μg/g v/s 2.1×10^2 μg/g) in dust that was in direct contact with PVC polymer (containing 17% DEHP) as compared to dust that was not in contact with PVC in a 14-day emission test (Schripp et al., 2010). Likewise, BBP concentrations in dust in direct contact with vinyl flooring was found to be about 20–30 times higher than dust on other surfaces (Bi et al., 2015a). Sukiene et al. (2017) also concluded that final SVOC concentration considerably increases in dust that is in direct contact with source due to direct transfer. The emission of DEHP from vinyl flooring is affected by "external" control factors like evaporation at elevated temperatures (Ekelund et al., 2010), partitioning from the material into the gas-phase and convective mass transfer (Clausen et al., 2010), and sorption onto interior surfaces including airborne particles (Liang and Xu, 2015). The presence of airborne particles (ammonium sulfate particles) at concentrations of 1×10^2 to 2.5×10^2 μg/m^3, was found to significantly enhance the emission of DEHP from vinyl flooring and increase its concentration by a factor of three to eight in a test chamber (Benning et al., 2013). Similar results were predicted by Xu et al. (2010) in their model. Schripp et al. (2010) demonstrated that the transfer of DEHP in dust is dependent on the initial concentration of additives in polymer material; DEHP concentration in dust was found to be 4.1×10^2 μg/g and 2.1×10^3 μg/g with direct contact with PVC containing 4% and 17% DEHP respectively after 14 days of incubation. The role of flooring type in DEHP migration into dust was studied by Jeon et al. (2016) in FLEC. DEHP concentrations were found to be $3.8 \times 10^2 \pm 19$ μg/g and $4.8 \times 10^2 \pm 53$ μg/g in dust from the uncoated residential type and wax-coated commercial type PVC flooring respectively, while no migration was reported from UV curing paint coated PVC flooring at room temperature.

Indoor decorating materials, furniture, the presence of plastic materials and carpet coverage have been associated with phthalates in house dust (Abb et al., 2009; Pei et al., 2013; Bi et al., 2015b). Smaller rooms (lesser area), old decoration, crowded items, use of domestic appliances with plastic coating result in the accumulation of phthalates (Øie et al., 1997; Zhang et al., 2014). Higher phthalate concentration in dust has also been attributed to use and leakage from electronic devices (Kang et al., 2012; Ait Bamai et al., 2014), and cosmetics, fragranced products

and personal care products (Zhang et al., 2013; He et al., 2015; Dodson et al., 2017). Absorption by materials other than direct sources, like wood flooring, has been found to influence the emission of phthalates as it acts like a sink for phthalates and also prevents their natural degradation. This was demonstrated by Liang and Xu (2015), who found 2–3 times higher DEHP concentrations in a wood chamber as compared to a stainless steel chamber. Crib mattresses have been reported to act like both, sources and sorptive reservoirs for gas phase phthalates (Boor et al., 2015).

Construction year of buildings has been positively associated with phthalate levels in the dust by several studies (Kang et al., 2012; Ait Bamai et al., 2014; Kim et al., 2013). This may be attributed to higher concentrations of phthalates in older products or higher emission rates as products degrade and newer regulations curbing the use of phthalates (Ait Bamai et al., 2014). Higher DEHP concentrations were found in homes constructed before 1980 in Bulgaria (Kolarik et al., 2008) and children facilities and households in Seoul, South Korea that were >10 years old (Kim et al., 2009). In Sweden, buildings constructed before 1960 were associated with high concentrations of DEHP (Bornehag et al., 2005), while preschools constructed before 1985 were found to have higher levels of both DEHP and DBP as compared to preschools constructed or renovated after 1999 (Balck, 2015). However, preschools built after the year 2000 were found to have higher levels of substitute phthalates and non-phthalate plasticizers, reflecting the shift in trend towards substitute phthalates in the E.U. (Larsson et al., 2017). However, a significant correlation was not observed between phthalate esters concentrations and age of the building (Li et al., 2016). Use of higher levels of phthalates in indoor building material and other products might be responsible for higher concentrations in the indoor environments of buildings constructed before the introduction of non-phthalate plasticizer.

Minimizing the use of plastic products in construction/decoration of the buildings which are regularly occupied by children, might be critical for exposure reduction (Bi et al., 2015b). The sources of phthalates which are used in building materials are more permanent in nature and their removal requires regulatory intervention, while other sources such as plastic materials, foam mattresses are easier to be replaced or removed (Larsson et al., 2017; Ait Bamai et al., 2014).

3.5. Indoor maintenance

Maintaining clean indoor environment has been regarded as a reduction strategy for phthalate contamination (Toda et al., 2004; Ye et al., 2017). Sites with a lower frequency of cleaning/dusting were associated with higher concentrations of phthalates in the air and dust (Zhang et al., 2014; Kolarik et al., 2008). Products used for cleaning and floor care products like polishing wax have been found to be significant sources of phthalates (Bi et al., 2015b; Afshari et al., 2004; Kolarik et al., 2008; Zhang et al., 2013). The highest concentration of DEHP, BBP and DnOP was found in homes with a combination of a low frequency of dusting and the use of polish (Kolarik et al., 2008). Afshari et al. (2004) reported two orders of magnitude higher emission of DBP from polishing wax as compared to other materials like PVC flooring and polyolefine flooring. Floor wax used for solid-wood floors in China was reported to be an important source of DBP, BBP and DnOP in settled house dust (Zhang et al., 2013). The average concentration of phthalates in dust from sites with contracted custodians for cleaning (therefore the routine use of polishing wax and chemical cleaning agents) was reported to be significantly higher than those sites without contracted custodians (Bi et al., 2015b). However, Kim et al. (2013) did not observe any significant association between phthalate levels and cleaning performance in nursery schools.

Presence of particles in the air and dust has been known to increase exposure via re-suspension of dust and increase in emission (Clausen et al., 2004; Benning et al., 2013). Hence, removal of particles and minimizing settled dust by cleaning is regarded as an effective way of reducing exposure to phthalates (Ye et al., 2017; Clausen et al., 2004).

4. Conclusions

Several countries in Asia, Europe and North America have reported widely varying levels of phthalates in the air and dust. Consistent with the highest consumption, phthalate levels from countries in Asia particularly China and India were found to be among the highest worldwide. However, considering the importance of phthalates with regard to human health impacts there is a dearth of studies from many countries. Phthalates are associated with both gaseous and particulate phases of air and contribution of either of the phases to the total concentration varies according to the different microenvironment. Some studies have reported phthalates in one of the two phases which might have led to an underestimation of the total concentration in the air in those studies. Indoor air was reported to have higher concentrations of phthalates as compared to the outdoor air. Contradictory results are reported in different studies for the temporal trend of phthalates. This might be due location-specific factors which play a major role in determining the seasonal concentration of phthalates in the air. Among various factors affecting the distribution of phthalates increasing ambient temperature was found to increase the emission of phthalates from sources in indoor environments. Elevated temperatures considered in various studies do not represent ambient temperature but become important in case of heating of for example food in products containing phthalates. However, the influence of temperature on phthalate emissions in dust requires further investigation. Higher relative humidity was reported to increase hydrolysis of phthalates which results in a gradual decrease in concentration of phthalates in the source as well as a sink. However, moisture in the source material has been found to increase the migration of DEHP within the polymer and to air and dust. Ventilation systems alone were reported to be ineffective in controlling exposure to phthalates as they tend to partition and remain sorbed to particles. Particle filtration in combination with ventilation was suggested to be useful for mitigating phthalate exposure. Cleaning of settled dust has been regarded as an effective way of reducing exposure to phthalates, however, cleaning agents like polishing wax has been found to be the source of phthalates. Different microenvironments may contain sources for specific phthalates, leading to a site-specific composition in the air and dust. Variations in the types and sources of phthalates worldwide affect their concentrations in the respective environment. Therefore, mitigation strategies should be based on regional or country-specific studies rather than benchmark studies.

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References

Abb, M., Heinrich, T., Sorkau, E., Lorenz, W., 2009. Phthalates in house dust. Environ. Int. 35, 965–970.

Afshari, A., Gunnarsen, L., Clausen, P.A., Hansen, V., 2004. Emission of phthalates from PVC and other materials. Indoor Air 14, 120–128.

Ait Bamai, Y., Araki, A., Kawai, T., Tsuboi, T., Saito, I., Yoshioka, E., Kanazawa, A., Tajima, S., Shi, C., Tamakoshi, A., Kishi, R., 2014. Associations of phthalate concentrations in floor dust and multi-surface dust with the interior materials in Japanese dwellings. Sci. Total Environ. 468–469, 147–157.

Albar, H.M.S.A., Ali, N., Shahzad, K., Ismail, I.M.I., Rashid, M.I., Wang, W., Ali, L.N., Eqani, S.A.M.A.S., 2017. Phthalate esters in settled dust of different indoor microenvironments; source of non-dietary human exposure. Microchem. J. 132, 227–232.
Aragón, M., Marcé, R.M., Borrull, F., 2012. Determination of phthalates and organophosphate esters in particulated material from harbour air samples by pressurised liquid extraction and gas chromatography-mass spectrometry. Talanta 101, 473–478.
Balck, M., 2015. Phthalates in Preschool Dust: The Relation between Phthalates and Parameters in the Preschool Environment. Master thesis. Uppsala University and Karolinska Institute, Institute for Environmental Medicine.
Benning, J.L., Liu, Z., Tiwari, A., Little, J.C., Marr, L.C., 2013. Characterizing gas-particle interactions of phthalate plasticizer emitted from vinyl flooring. Environ. Sci. Technol. 47, 2696–2703.
Bergh, C., Torgrip, R., Emenius, G., Östman, C., 2011. Organophosphate and phthalate esters in air and settled dust - a multi-location indoor study. Indoor Air 21, 67–76.
Bi, C., Liang, Y., Xu, Y., 2015a. Fate and transport of phthalates in indoor environments and the influence of temperature: a case study in a test house. Environ. Sci. Technol. 49 (16), 9674–9681.
Bi, X.L., Yuan, S.J., Pan, X.J., Winstead, C., Wang, Q.Q., 2015b. Comparison, association, and risk assessment of phthalates in floor dust at different indoor environments in Delaware, USA. J. Environ. Sci. Health A Tox. Hazard. Subst. Environ. Eng. 50, 1428–1439.
BIS, 2011. Safety requirements for toys, part-3: migration of certain elements and phthalates. Available from. http://www.bis.org.in/sf/Jul'11/9873_3A3.pdf, Accessed date: 28 January 2018.
Blanchard, O., Glorennec, P., Mercier, F., Bonvallot, N., Chevrier, C., Ramalho, O., Mandin, C., Le Bot, B., 2014. Semivolatile organic compounds in indoor air and settled dust in 30 French dwellings. Environ. Sci. Technol. 48, 3959–3969.
Boor, B.E., Liang, Y., Crain, N.E., Ja, H., Novoselac, A., Xu, Y., 2015. Identi fi cation of Phthalate and Alternative Plasticizers, Flame Retardants, and Unreacted Isocyanates in Infant Crib Mattress Covers and Foam. doi:https://doi.org/10.1021/acs.estlett.5b00039.
Bornehag, C.G., Sundell, J., Weschler, C.J., Sigsgaard, T., Lundgren, B., Hasselgren, M., Hägerhed-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 (14), 1393–1397.
Bornehag, C.G., Lundgren, B., Weschler, C.J., Sigsgaard, T., Hagerhed-Engman, L., Sundell, J., 2005. Phthalates in indoor dust and their association with building characteristics. Environ. Health Perspect. 113, 1399–1404.
Bu, Z., Zhang, Y., Mmereki, D., Yu, W., Li, B., 2016. Indoor phthalate concentration in residential apartments in Chongqing, China: implications for preschool children's exposure and risk assessment. Atmos. Environ. 127, 34–45.
Bui, T.T., Giovanoulis, G., Palm, A., Magnér, J., Cousins, I.T., De Wit, C.A., 2016. Human exposure, hazard and risk of alternative plasticizers to phthalate esters. Sci. Total Environ. 541, 451–467.
Butte, W., Heinzow, B., 2002. Pollutants in house dust as indicators of indoor contamination. Rev. Environ. Contam. Toxicol. 175, 1–46.
Cao, X.L., 2010. Phthalate esters in foods: sources, occurrence, and analytical methods. Compr. Rev. Food Sci. Food Saf. 9, 21–43.
China National Standard GB 6675, 2014. Safety of toys. Available from. http://www.bureauveritas.com.co/xMuyRtMG/Bulletin_14B-024_Eng.pdf, Accessed date: 29 January 2018.
Chi, C., Xia, M., Zhou, C., Wang, X., Weng, M., Shen, X., 2017. Determination of 15 phthalate esters in air by gas-phase and particle-phase simultaneous sampling. J. Environ. Sci. (China) 55, 137–145.
Cousins, I., Mackay, D., 2000. Correlating the physical chemical properties of phthalate esters using the 'three solubility' approach. Chemosphere 41 (9), 1389–1399.
CPSIA, 2008. The United States Consumer Product Safety Improvement Act. Section 108. Washington, DC.
Clark, K., Cousins, I.T., MacKay, D., 2003. Assessment of critical exposure pathways. Handb. Environ. Chem. 3, 227–262.
Clausen, P.A., Hansen, V., Gunnarsen, L., Afshari, A., Wolkoff, P., 2004. Emission of di-2-ethylhexylphthalate from PVC flooring into air and uptake in dust: emission and sorption experiments in FLEC and CLIMPAQ. Environ. Sci. Technol. 38 (9), 2531–2537.
Clausen, P.A., Xu, Y., Kofoed-Sørensen, V., Little, J.C., Wolkoff, P., 2007. The influence of humidity on the emission of di-(2-ethylhexyl) phthalate (DEHP) from vinyl flooring in the emission cell "FLEC". Atmos. Environ. 41, 3217–3224.
Clausen, P.A., Liu, Z., Xu, Y., Kofoed-Sorensen, V., Little, J.C., 2010. Influence of air flow rate on emission of DEHP from vinyl flooring in the emission cell FLEC: measurement and CFD simulation. Atoms Environ. 44, 2760–2766.
Clausen, P.A., Liu, Z., Kofoed-Sorensen, V., Little, J., Wolkoff, P., 2012. Influence of temperature on the emission of Di-(2-ethylhexyl) phthalate (DEHP) from PVC flooring in the emission cell FLEC. Environ. Sci. Technol. 46 (2), 909–915.
Das, M.T., Ghosh, P., Thakur, I.S., 2014. Intake estimates of phthalate esters for South Delhi population based on exposure media assessment. Environ. Pollut. 189, 118–125.
Dodson, R.E., Udesky, J.O., Colton, M.D., Mccauley, M., Camann, D.E., Yau, A.Y., Adamkiewicz, G., Rudel, R.A., 2017. Chemical exposures in recently renovated low-income housing: influence of building materials and occupant activities. Environ. Int. 109, 114–127.
ECHA, 2016. Information on chemicals. Available from:. http://echa.europa.eu/information-on-chemicals, Accessed date: 14 August 2017.
Ekelund, M., Azhdar, B., Hedenqvist, M.S., Gedde, U.W., 2008. Long-term performance of poly(vinyl chloride) cables, part 2: migration of plasticizer. Polym. Degrad. Stab. 93 (9), 1704— 1710.
Ekelund, M., Azhdar, B., Gedde, U.W., 2010. Evaporative loss kinetics of di (2-ethylhexyl) phthalate (DEHP) from pristine DEHP and plasticized PVC. Polym. Degrad. Stab. 95 (9), 1789–1793.
EU, 2005. Directive 2005/84/EC of the European parliament and of the council. Available from. http://www.ce-mark.com/200584EC.pdf, Accessed date: 8 October 2017.
Fernandez, M.A., Gomara, B., Gonzalez, M.J., 2012. Occurrence of Phthalates and Their Metabolites in the Environment and Human Health Implications. Emerging Organic Contaminants and Human Health, HdbEnvChem. 20. Springer-Verlag, Berlin Heidelberg, pp. 307–336.
Fromme, H., Lahrz, T., Kraft, M., Fembacher, L., Dietrich, S., Sievering, S., Burghardt, R., Schuster, R., Bolte, G., Völkel, W., 2013. Phthalates in German daycare centers: occurrence in air and dust and the excretion of their metabolites by children (LUPE 3). Environ. Int. 61, 64–72.
Fu, P.Q., Kawamura, K., Pavuluri, C.M., Swaminathan, T., Chen, J., 2010. Molecular characterization of urban organic aerosol in tropical India: contributions of primary emissions and secondary photooxidation. Atmos. Chem. Phys. 10, 2663–2689.
Fujii, M., Shinohara, N., Lim, A., Otake, T., Kumagai, K., Yanagisawa, Y., 2003. A study on emission of phthalate esters from plastic materials using a passive flux sampler. Atoms Environ. 37, 5495–5504.
Gaspar, F.W., Castorina, R., Maddalena, R.L., Nishioka, M.G., Mckone, T.E., Bradman, A., 2014. Phthalate exposure and risk assessment in California child care facilities. Environ. Sci. Technol. 48, 7593–7601.
Gevao, B., Al-Ghadban, A.N., Bahloul, M., Uddin, S., Zafar, J., 2013. Phthalates in indoor dust in Kuwait: implications for non-dietary human exposure. Indoor Air 23, 126–133.
Guo, Y., Kannan, K., 2011. Comparative Assessment of Human Exposure to Phthalate Esters from House Dust in China and the United States Comparative Assessment of Human Exposure to Phthalate Esters from House Dust in China and the United States. https://doi.org/10.1021/es2002106.
He, C., Morawska, L., Gilbert, D., 2005. Particle deposition rates in residential houses. Atmos. Environ. 39, 3891–3899.
He, R., Li, Y., Xiang, P., Li, C., Zhou, C., Zhang, S., 2015. Chemosphere organophosphorus flame retardants and phthalate esters in indoor dust from different microenvironments: bioaccessibility and risk assessment. Chemosphere 2010:1–8. https://doi.org/10.1016/j.chemosphere.2015.10.087.
Hsu, N.Y., Lee, C.C., Wang, J.Y., Chang, H.W., Chen, C.Y., Bornehag, C.G., Wu, P.C., Sundell, J., Su, HJ., 2012. Predicted risk of childhood allergy, asthma, and reported symptoms using measured phthalate exposure in dust and urine. Indoor Air 22, 186–199.
Hsu, N.Y., Liu, Y.C., Lee, C.W., Lee, C.C., Su, H.J., 2017. Higher moisture content is associated with greater emissions of DEHP from PVC wallpaper. Environ. Res. 152, 1–6.
Hwang, H.M., Park, E.K., Young, T.M., Hammock, B.D., 2008. Occurrence of endocrine-disrupting chemicals in indoor dust. Sci. Total Environ. 404, 26–35.
IHS Markit, 2015. Plasticizers - Chemical Economics Handbook (CEH) Chem. Econ. Handb. https://www.ihs.com/products/plasticizers-chemical-economics-handbook.html, Accessed date: 4 February 2018.
Japan Toy Safety Standard ST-2002 Part 3, 2011. . Available from. http://www.toyicti.org/PDFs/Jap-PhthReq-08223011.pdf, Accessed date: 29 January 2018.
Jeon, S., Kim, K.T., Choi, K., 2016. Migration of DEHP and DINP into dust from PVC flooring products at different surface temperature. Sci. Total Environ. 547, 441–446.
Kamrin, M.A., Lansing, E., 2009. Phthalate risks, Phthalate Regulation, and Public Health: A Review. :pp. 157–174 https://doi.org/10.1080/10937400902729226.
Kanazawa, A., Saito, I., Araki, A., Takeda, M., Ma, M., Saijo, Y., Kishi, R., 2010. Association between indoor exposure to semi-volatile organic compounds and building-related symptoms among the occupants of residential dwellings. Indoor Air 20, 72–84.
Kang, Y., Man, Y.B., Cheung, K.C., Wong, M.H., 2012. Risk assessment of human exposure to bioaccessible phthalate esters via indoor dust around the Pearl River Delta. Environ. Sci. Technol. 46, 8422–8430.
Katsikantami, I., Sifakis, S., Tzatzarakis, M.N., Vakonaki, E., Kalantzi, O.I., Tsatsakis, A.M., Rizos, A.K., 2016. A global assessment of phthalates burden and related links to health effects. Environ. Int. 97, 212–236.
KEMI, 2015. Phthalates Which Are Toxic for Reproduction and Endocrine-disrupting – Proposals for a Phase-out in Sweden. Report 4/15 Stockholm.
Kim, H.H., Lim, Y.W., Yang, J.Y., Shin, D.C., 2009. Occurrence of phthalates in indoor dust from children's facilities and apartments in Seoul. J. Korea. Soc. Atmos. Environ. 25 (5), 382–391.
Kim, W., Choi, I., Jung, Y., Lee, J., Min, S., Yoon, C., 2013. Phthalate levels in nursery schools and related factors. Environ. Sci. Technol. 47, 12459–12468.
Kong, S., Ji, Y., Liu, L., Chen, L., Zhao, X., Wang, J., Bai, Z., Sun, Z., 2013. Spatial and temporal variation of phthalic acid esters (PAEs) in atmospheric PM_{10} and $PM_{2.5}$ and the influence of ambient temperature in Tianjin, China. Atmos. Environ. 74, 199–208.
Kolarik, B., Naydenov, K., Bornehag, C.G., Sundell, J., 2007. Concentration of phthalate esters found in homes of healthy and allergic children. Proceedings of the Conference on Indoor Air Quality, Ventilation and Energy Conservation in Buildings. Vol. II. IAQVEC, Sendai, Japan, pp. 255–262.
Kolarik, B., Bornehag, C.G., Naydenov, K., Sundell, J., Stavova, P., Nielsen, O.F., 2008. The concentrations of phthalates in settled dust in Bulgarian homes in relation to building characteristic and cleaning habits in the family. Atmos. Environ. 42, 8553–8559.
Kubwabo, C., Rasmussen, P.E., Fan, X., Kosarac, I., Wu, F., Zidek, A., Ku- chta, S.L., 2013. Analysis of selected phthalates in Canadian indoor dust collected using household vacuum and standardized sampling techniques. Indoor Air 23 (6), 506–514.
Lan, Q., Cui, K., Zeng, F., Zhu, F., Liu, H., Chen, H., Ma, Y., Wen, J., Luan, T., Sun, G., Zeng, Z., 2012. Characteristics and assessment of phthalate esters in urban dust in Guangzhou city, China. Environ. Monit. Assess. 184, 4921–4929.
Langer, S., Weschler, C.J., Fischer, A., Bekö, G., Toftum, J., Clausen, G., 2010. Phthalate and PAH concentrations in dust collected from Danish homes and daycare centers. Atmos. Environ. 44 (19), 2294–2301.
Larsson, M., Hagerhed-Engman, L., Kolarik, B., James, P., Lundin, F., Janson, S., Sundell, J., Bornehag, C.G., 2010. PVC - as flooring material - and its association with incident asthma in a Swedish child cohort study. Indoor Air 20, 494–501.
Larsson, K., Lindh, C.H., Jönsson, B.A., Giovanoulis, G., Bibi, M., Bottai, M., Bergström, A., Berglund, M., 2017. Phthalates, non-phthalate plasticizers and bisphenols in Swedish preschool dust in relation to children's exposure. Environ. Int. 102, 114–124.

Lenoir, A., Boulay, R., Dejean, A., Touchard, A., Cuvillier-hot, V., Cuvillier-hot, V., 2016. Phthalate pollution in an Amazonian rainforest. Environ. Sci. Pollut. Res.: 16865–16872 https://doi.org/10.1007/s11356-016-7141-z.

Li, J., Wang, G., Aggarwal, S.G., Huang, Y., Ren, Y., Zhou, B., Singh, K., Gupta, P.K., Cao, J., Zhang, R., 2014. Comparison of abundances, compositions and sources of elements, inorganic ions and organic compounds in atmospheric aerosols from Xi'an and New Delhi, two megacities in China and India. Sci. Total Environ. 476–477: 485–495. https://doi.org/10.1016/j.scitotenv.2014.01.011.

Li, H.L., Song, W.W., Zhang, Z.F., Ma, W.L., Gao, C.J., Li, J., Huo, C.Y., Mohammed, M.O.A., Liu, L.Y., Kannan, K., Li, Y.F., 2016. Phthalates in dormitory and house dust of northern Chinese cities: occurrence, human exposure, and risk assessment. Sci. Total Environ. 565, 496–502.

Liang, Y., Xu, Y., 2014a. Emission of phthalates and phthalate alternatives from vinyl flooring and crib mattress covers: the influence of temperature. Environ. Sci. Technol. 48, 14228–14237.

Liang, Y., Xu, Y., 2014b. Improved method for measuring and characterizing phthalate emissions from building materials and its application to exposure assessment. Environ. Sci. Technol. 48, 4475–4484.

Liang, Y., Xu, Y., 2015. The influence of surface sorption and air flow rate on phthalate emissions from vinyl flooring: measurement and modeling. Atmos. Environ. 103, 147–155.

Liao, C., Liu, W., Zhang, J., Shi, W., Wang, X., Cai, J., Zou, Z., Lu, R., Sun, C., Wang, H., Huang, C., Zhao, Z., 2018. Associations of urinary phthalate metabolites with residential characteristics, lifestyles, and dietary habits among young children in Shanghai, China. Sci. Total Environ. 616-617, 1288–1297.

Liu, C., Zhang, Y., Benning, J.L., Little, J.C., 2015. The effect of ventilation on indoor exposure to semivolatile organic compounds. Indoor Air 25 (3), 285–296.

Luongo, G., Ostman, C., 2016. Organophosphate and phthalate esters in settled dust from apartment buildings in Stockholm. Indoor Air 26, 414–425.

Ma, J., Chen, L. lu, Guo, Y., Wu, Q., Yang, M., Wu, M. hong, Kannan, K., 2014. Phthalate diesters in airborne $PM_{2.5}$ and PM_{10} in a suburban area of Shanghai: seasonal distribution and risk assessment. Sci. Total Environ. 497–498:467–474. https://doi.org/10.1016/j.scitotenv.2014.08.012.

Mandin, C., Mercier, Fabien, Lucas, Jean-Paul, Ramalho, Olivier, Blanchard, Olivier, Bonvallot, Nathalie, Raffy, Gaelle, Gilles, E., Glorennec, Philippe, Le Bot, Barbara, 2014. ECOS-POUSS: A Nationwide Survey of Semi-volatile Organic Compounds in Home Settled Dust. Indoor Air 2014 - 13th International Conference on Indoor Air Quality and Climate pp. 143–148.

Moreau-Guigon, E., Alliot, F., Gasperi, J., Blanchard, M., Teil, M.J., Mandin, C., Chevreuil, M., 2016. Seasonal fate and gas/particle partitioning of semi-volatile organic compounds in indoor and outdoor air. Atmos. Environ. 147, 423–433.

Net, S., Sempere, R., Delmont, A., Paluselli, A., Ouddane, B., 2015. Occurrence, fate, behavior and ecotoxicological state of phthalates in different environmental matrices. Environ. Sci. Technol. 49, 4019–4035.

Øie, L., Hersoug, L.G., Madsen, J.Ø., 1997. Residential exposure to plasticizers and its possible role in the pathogenesis of asthma. Environ. Health Perspect. 105, 972–978.

Orecchio, S., Indelicato, R., Barreca, S., 2013. The distribution of phthalate esters in indoor dust of Palermo (Italy). Environ. Geochem. Health 35, 613–624.

Parthasarathy, S., Chan, W., Fisk, W., Mckone, T., 2012. Modeling Indoor Exposures to VOCs and SVOCs as Ventilation Rates Vary. 10th International Conference of Healthy Building, Brisbane, Australia 10A.5.

Pei, X.Q., Song, M., Guo, M., Mo, F.F., Shen, X.Y., 2013. Concentration and risk assessment of phthalates present in indoor air from newly decorated apartments. Atmos. Environ. 68, 17–23.

Rakkestad, K.E., Dye, C.J., Yttri, K.E., Holme, J.A., Hongslo, J.K., Schwarze, P.E., Becher, R., 2007. Phthalate levels in Norwegian indoor air related to particle size fraction. J. Environ. Monit. 9, 1419–1425.

Rudel, R.A., Perovich, L.J., 2009. Endocrine disrupting chemicals in indoor and outdoor air. Atmos. Environ. 43 (1), 170−181.

Rudel, R.A., Dodson, R.E., Perovich, L.J., Morello-Frosch, R., Camann, D.E., Zuniga, M.M., Yau, A.Y., Just, A.C., Brody, J.G., 2010. Semivolatile endocrine-disrupting compounds in paired indoor and outdoor air in two northern California communities. Environ. Sci. Technol. 44, 6583–6590.

Růžičková, J., Raclavská, H., Raclavský, K., Juchelková, D., 2015. Phthalates in $PM_{2.5}$ airborne particles in the Moravian-Silesian Region, Czech Republic. Perspect. Sci. 178–183.

Sampath, S., Selvaraj, K.K., Shanmugam, G., Krishnamoorthy, V., Chakraborty, P., Ramaswamy, B.R., 2016. Evaluating spatial distribution and seasonal variation of phthalates using passive air sampling in southern India. Environ. Pollut.:1–11 https://doi.org/10.1016/j.envpol.2016.12.003.

Salapasidou, M., Samara, C., Voutsa, D., 2011. Endocrine disrupting compounds in the atmosphere of the urban area of Thessaloniki, Greece. Atmos. Environ. 45, 3720–3729.

Schripp, T., Fauck, C., Salthammer, T., 2010. Chamber studies on mass-transfer of di(2-ethylhexyl)phthalate (DEHP) and di-n-butylphthalate (DnBP) from emission sources into house dust. Atmos. Environ. 44, 2840–2845.

Salgueiro-González, N., De Alda, M.L., Muniategui-Lorenzo, S., Prada-Rodríguez, D., Barceló, D., 2013. Determination of 13 estrogenic endocrine disrupting compounds in atmospheric particulate matter by pressurised liquid extraction and liquid chromatography-tandem mass spectrometry. Anal. Bioanal. Chem. 405, 8913–8923.

Škrbić, B.D., Ji, Y., Dstrokeurišić-Mladenović, N., Zhao, J., 2016. Occurrence of the phthalate esters in soil and street dust samples from the Novi Sad city area, Serbia, and the influence on the children's and adults' exposure. J. Hazard. Mater. 312, 272–279.

Song, M., Chi, C., Guo, M., Wang, X., Cheng, L., Shen, X., 2015. Pollution levels and characteristics of phthalate esters in indoor air of offices. J. Environ. Sci. (China) 28, 157–162.

Staples, C.A., Peterson, D.R., Parkerton, T.F., Adams, W.J., 1997. The environmental fate of phthalate esters, a literature review. Chemosphere 23 (4), 667–749.

Sukiene, V., Gerecke, A.C., Park, Y.M., Zennegg, M., Bakker, M.I., Delmaar, C.J.E., Hungerbühler, K., Von Goetz, N., 2016. Tracking SVOCs' transfer from products to indoor air and settled dust with deuterium-labeled substances. Environ. Sci. Technol. 50, 4296–4303.

Sukiene, V., Von Goetz, N., Gerecke, A.C., Bakker, I., Delmaar, C.J.E., Hungerbuehler, K., 2017. Direct and air-mediated transfer of labeled SVOCs from indoor sources to dust. Environ. Sci. Technol. 51 (6), 3269–3277.

Takeuchi, S., Kojima, H., Saito, I., Jin, K., Kobayashi, S., Tanaka-Kagawa, T., Jinno, H., 2014. Detection of 34 plasticizers and 25 flame retardants in indoor air from houses in Sapporo, Japan. Sci. Total Environ. 491–492, 28–33.

Takeuchi, S., Tanaka-Kagawa, T., Saito, I., Kojima, H., Jin, K., Satoh, M., Kobayashi, S., Jinno, H., 2015. Differential determination of plasticizers and organophosphorus flame retardants in residential indoor air in Japan. Environ. Sci. Pollut. Res. https://doi.org/10.1007/s11356-015-4858-z.

Teil, M.J., Blanchard, M., Chevreuil, M., 2006. Atmospheric fate of phthalate esters in an urban area (Paris-France). Sci. Total Environ. 354, 212–223.

Teil, M.-J., Moreau-Guigon, E., Blanchard, M., Alliot, F., Gasperi, J., Cladière, M., Mandin, C., Moukhtar, S., Chevreuil, M., 2016. Endocrine disrupting compounds in gaseous and particulate outdoor air phases according to environmental factors. Chemosphere 146, 94–104.

Toda, H., Sako, K., Yagome, Y., Nakamura, T., 2004. Simultaneous determination of phosphate esters and phthalate esters in clean room air and indoor air by gas chromatography-mass spectrometry. Anal. Chim. Acta 519, 213–218.

Tran, M., Kannan, K., 2015. Occurrence of phthalate diesters in particulate and vapor phases in indoor air and implications for human exposure in Albany, New York, USA. Arch. Environ. Contam. Toxicol. 68, 489–499.

Tran, T.M., Minh, T.B., Kumosani, T.A., Kannan, K., 2016. Occurrence of phthalate diesters (phthalates), p-hydroxybenzoic acid esters (parabens), bisphenol A diglycidyl ether (BADGE) and their derivatives in indoor dust from Vietnam: implications for exposure. Chemosphere 144, 1553–1559.

Tran, M., Thi, H., Binh, T., Kannan, K., 2017. Occurrence of phthalate diesters in indoor air from several Northern cities in Vietnam, and its implication for human exposure. Sci. Total Environ. 601–602, 1695–1701.

Wang, P., Wang, S.L., Fan, C.Q., 2008. Atmospheric distribution of particulate- and gas-phase phthalic esters (PAEs) in a Metropolitan City, Nanjing, East China. Chemosphere 72, 1567–1572.

Wang, W., Wu, F.Y., Huang, M.J., Kang, Y., Cheung, K.C., Wong, M.H., 2013. Size fraction effect on phthalate esters accumulation, bioaccessibility and in vitro cytotoxicity of indoor/outdoor dust, and risk assessment of human exposure. J. Hazard. Mater. 261, 753–762.

Wang, X., Tao, W., Xu, Y., Feng, J., Wang, F., 2014. Indoor phthalate concentration and exposure in residential and office buildings in Xi'an, China. Atmos. Environ. 87, 146–152.

Wei, W., Mandin, C., Ramalho, O., 2018. Chemosphere influence of indoor environmental factors on mass transfer parameters and concentrations of semi-volatile organic compounds. Chemosphere 195, 223–235.

Weschler, C.J., 1980. Characterization of selected organics in size-fractionated indoor aerosols. Environ. Sci. Technol. 14, 428–431.

Weschler, C.J., Salthammer, T., Fromme, H., 2008. Partitioning of phthalates among the gas phase, airborne particles and settled dust in indoor environments. Atmos. Environ. 42, 1449–1460.

Westberg, Å., Momcilovic, D., Björk, F., Karlsson, S., 2009. Quality assessment of building products by the micro-scale headspace vial (MHV) method and HS- SPME for monitoring the emission of hydrolysis products from phthalates. Polym. Degrad. Stab. 94, 914–920.

WHO, 2013. World Health Organization, and United Nations Environment Programme (UNEP). State of the Science: Endocrine Disrupting Chemicals-2012. Summary for Decision Makers.

Xie, Z., Ebinghaus, R., Temme, C., Caba, A., Ruck, W., 2005. Atmospheric concentrations and air-sea exchanges of phthalates in the North Sea (German Bight). Atmos. Environ. 39, 3209–3219.

Xie, Z., Ebinghaus, R., Temme, C., Lohmann, R., Caba, A., Ruck, W., 2007. Occurrence and air-sea exchange of phthalates in the Arctic. Environ. Sci. Technol. 41, 4555–4560.

Xu, Y., Zhang, J.S., 2011. Understanding SVOCs. ASHRAE J. 53 (12), 121–125.

Xu, Y., Hubal, E.A.C., Clausen, P.A., Little, J.C., 2009. Phthalate plasticizer emitted from vinyl flooring: a mechanistic analysis. Environ. Sci. Technol. 43 (7), 2374–2380.

Xu, Y., Cohen Hubal, E.A., Little, J.C., 2010. Predicting residential exposure to phthalate plasticizer emitted from vinyl flooring: sensitivity, uncertainty, and implications for biomonitoring. Environ. Health Perspect. 118, 253–258.

Xu, Y., Liang, Y., Urquidi, J.R., Siegel, J.A., 2014. Phthalates and polybrominated diphenyl ethers in retail stores. Atmos. Environ. 87, 53–64.

Xu, Y., Liang, Y., Urquidi, J.R., Siegel, J., 2015. Semi-volatile organic compounds in heating, ventilation, and air-conditioning filter dust in retail stores. Indoor Air 25 (1), 79–91.

Ye, W., Zhang, X., Gao, J., Cao, G., Zhou, X., Su, X., 2017. Indoor air pollutants, ventilation rate determinants and potential control strategies in Chinese dwellings: a literature review. Sci. Total Environ. 586, 696–729.

Zeng, F., Lin, Y., Cui, K., Wen, J., Ma, Y., Chen, H., Zhu, F., Ma, Z., Zeng, Z., 2010. Atmospheric deposition of phthalate esters in a subtropical city. Atmos. Environ. 44, 834–840.

Zhang, X., Chen, Z., 2014. Observing phthalate leaching from plasticized polymer films at the molecular level. Langmuir 30 (17), 4933–4944.

Zhang, Q., Lu, X.M., Zhang, X.L., Sun, Y.G., Zhu, D.M., Wang, B.L., Zhao, R.Z., Zhang, Z.D., 2013. Levels of phthalate esters in settled house dust from urban dwellings with young children in Nanjing, China. Atmos. Environ. 69, 258–264.

Zhang, L., Wang, F., Ji, Y., Jiao, J., Zou, D., Liu, L., Shan, C., Bai, Z., Sun, Z., 2014. Phthalate esters (PAEs) in indoor PM10/PM2.5 and human exposure to PAEs via inhalation of indoor air in Tianjin, China. Atmos. Environ. 85, 139–146.

Zhu, Z., Ji, Y., Zhang, S., Zhao, J., Zhao, J., 2016. Phthalate ester concentrations, sources, and risks in the ambient air of Tianjin, China. Aerosol Air Qual. Res. 16, 2294–2301.