

## Polycyclic aromatic hydrocarbons, polychlorinated biphenyls and legacy and current pesticides in indoor environment in Australia – occurrence, sources and exposure risks

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1 Polycyclic aromatic hydrocarbons, polychlorinated biphenyls and legacy and current  
2 pesticides in indoor environment in Australia – occurrence, sources and exposure risks

3

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13 **Highlights**

- 14 A modified PAS was used to simultaneously collect gaseous SVOCs and particles;
- 15 PCB 11 was identified and quantified in indoor environments for the first time;
- 16 PCBs and pesticides were from indoor while PAHs were influenced by outdoor sources;
- 17 Indoor pesticide levels in Australia were among the highest in the world;
- 18 The highest EDI value was estimated for permethrin, mostly via exposure to dust.

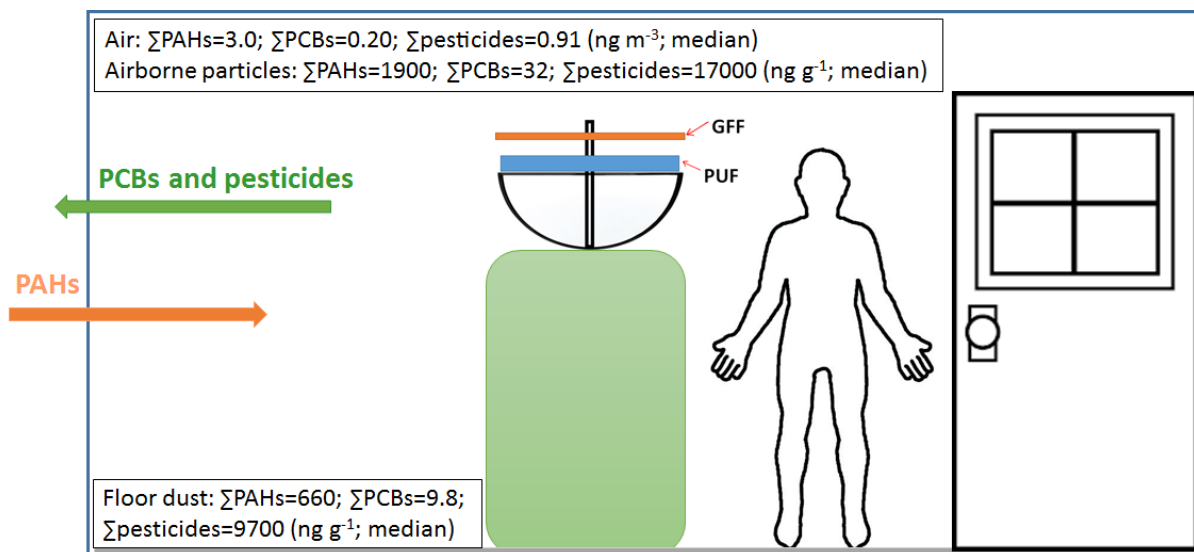
19 **Abstract**

20 Paired indoor air and floor dust samples were collected from residential houses and offices (n = 28) in  
21 two Australian cities in 2015. For the air samples, a modified passive air sampler (PAS) was used to  
22 collect semi-volatile organic compounds (SVOCs) in gaseous phase and airborne particles  
23 simultaneously. Sampling rates (*R*) of the PAS for gaseous SVOCs ranged from 0.69 to 3.4 m<sup>3</sup>  
24 sampler<sup>-1</sup> day<sup>-1</sup>. Out of the 33 analytes, 22, 14 and 17 compounds were detected (above the method  
25 detection limit) in over 50% of air, airborne particles and floor dust samples respectively. The highest  
26 median level in air, airborne particles and floor dust was observed for phenanthrene (2.0 ng m<sup>-3</sup>),  
27 permethrin (8800 ng g<sup>-1</sup>) and permethrin (5100 ng g<sup>-1</sup>) respectively. Amongst polychlorinated  
28 biphenyl (PCB) congeners, with few exceptions, the largest contribution was from 3,3'-  
29 dichlorobiphenyl (PCB11) for both indoor air and floor dust samples. In these houses and offices, the  
30 indoor level of polycyclic aromatic hydrocarbons (PAHs) was mainly influenced by ambient (outdoor)  
31 air. Primary sources of PCBs were from within indoor environments and generally older houses have  
32 higher concentrations in air. Amongst pesticides, hexachlorobenzene in indoor environments appeared  
33 to be due to transfer from outdoor sources whereas chlordanes and pyrethroids were associated with  
34 past and current household application respectively. Compared to data from other countries/regions,  
35 concentrations of chlordanes, chlorpyrifos and pyrethroids in indoor air and dust samples from  
36 Australia were among the highest whereas PCB and PAH levels were among the lowest. The sum of  
37 estimated daily intakes (EDIs) via inhalation and dust contact and ingestion were calculated. The  
38 highest median value of EDI was observed for permethrin at 2.8 (for adults) and 74 ng kg<sup>-1</sup> day<sup>-1</sup> (for  
39 toddlers), which are <0.15 % of the U.S. EPA reference dose.

40 **Keywords**

41 Indoor; semi-volatile organic compounds; passive air sampling; sources; global comparison; daily  
42 intakes

43 Graphical abstract



44  
45

## 46 1. Introduction

47 Indoor environments represent an important source for human exposure to organic chemicals for two  
48 reasons: i) people typically spend 70 – 90% of their time within homes and offices (Lucattini et al., 2018;  
49 Schweizer et al., 2007); and ii) a wide range of organic chemicals were/are used, and produced in and  
50 introduced into indoor environments. Many of these chemicals are semi-volatile organic compounds  
51 (SVOCs). Once emitted into indoor environments, these chemicals can disperse among different  
52 matrices/surfaces such as air, airborne particles and dust resulting in potential chronic exposure via  
53 inhalation, dust ingestion and/or dermal contact (Weschler and Nazaroff, 2008). Indoor environments also  
54 typically have less exposure to UV light and thus fewer photolytic reactions, making these chemicals more  
55 stable therein (Audy et al., 2018).

56 Understanding the levels and sources of these chemicals in indoor environments is key for assessing human  
57 exposure pathways and risks, and implementing control/intervention processes. Comprehensive data for  
58 SVOCs in indoor environments are scarce in Australia and have mostly focused on flame retardants (Harrad  
59 et al., 2016; He et al., 2018; Toms et al., 2009). There is a lack of data for many other SVOCs such as  
60 polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), and legacy and current  
61 pesticides. These chemicals can be introduced into indoor environments from different sources and via  
62 multiple pathways. PAHs can be emitted from cooking and smoking, indoor and outdoor biomass burning  
63 and vehicular emissions. PCBs and legacy pesticides, although banned for many years, are very persistent  
64 and can be emitted into indoor environments from sources such as old but in-use transformers and  
65 capacitors as well as building sealant (PCBs) and previously contaminated surfaces in and surrounding the  
66 building (OCPs). Current pesticides such as organophosphate pesticides and pyrethroids which are used for  
67 pest control in the building may also result in a high level of human exposure in indoor environments.  
68 Some compounds in these chemical groups such as benzo(a)pyrene and some PCB congeners are classified  
69 as carcinogenic to humans by the International Agency for Research on Cancer (IARC, 2015). Some other  
70 chemicals such as pyrethroid pesticides are considered to be endocrine disrupting and linked to various  
71 negative health outcomes (Meeker et al., 2008).

72 Air and dust are two very relevant matrices for investigating the exposure to SVOCs in indoor  
73 environments (Weschler and Nazaroff, 2008). Lucattini et al. (2018) conducted a comprehensive review of  
74 SVOCs in the indoor environment and presented that concentrations of PAHs, PCBs and pesticides in  
75 indoor air and floor dust samples have been reported mostly from countries/regions in the Northern  
76 Hemisphere. Knowledge gaps concluded in the above review also included the difficulty in the  
77 identification of main sources, as well as a limited (and likely underestimated) data of particle-associated  
78 SVOCs. Active air samplers (AAS) can sample both gaseous chemicals and particulates at the same time  
79 but their main drawback for indoor sampling, particularly within residential settings, is the noise emitted  
80 during operation. Therefore, various PAS have been developed as alternatives to AAS and  
81 widely/prevalently used for indoor SVOC sampling since 2004 (Bohlin et al., 2008; Harrad et al., 2006;  
82 Wilford et al., 2004) but primarily sample chemicals in the gaseous phase. Reported in a very limited

83 number of studies, effort has been recently made to designing PAS that can sample SVOCs in the particle-  
84 associated phase in the air in addition to the gaseous phase to achieve comparable monitoring capability as  
85 active air sampling (Abdallah and Harrad, 2010; Tao et al., 2009).

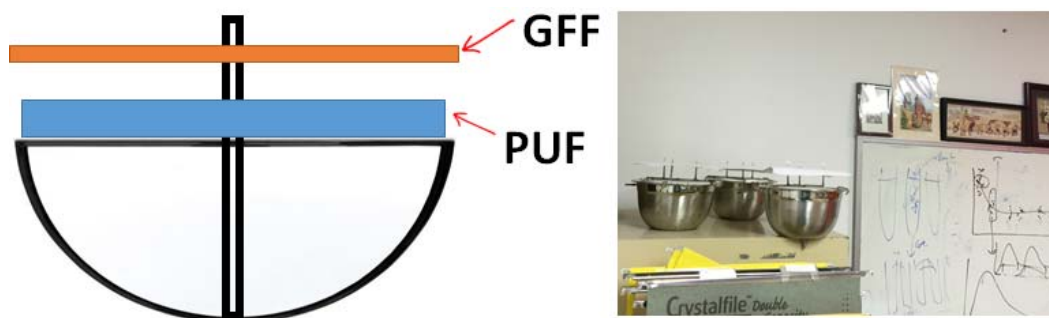
86 In this pilot study, a new modified PAS was used to simultaneously collect gaseous SVOCs and airborne  
87 particles from 28 homes and offices in two Australian cities. Paired dust samples were also collected. The  
88 current study presents an effort as one of the pioneering studies that systematically and simultaneously  
89 monitors multi-class SVOCs in air, particles and dust in indoor environments. To our best knowledge, this  
90 is also the first study in Australia monitoring PAHs, PCBs and pesticides in indoor environments from all  
91 these three matrices. The aim of this study was to investigate the levels, profiles and distribution of PAHs,  
92 PCBs and legacy and current pesticides in indoor air and dust and to evaluate their current major sources  
93 and human exposure risks.

## 94 2. Methodology

### 95 2.1 Sample collection

96 Paired air and dust samples were collected from 14 residential houses and 14 offices in Brisbane and  
97 Canberra, Australia in 2015. Air samples were collected using a modified PAS composed of glass fibre  
98 filter (GFF) and polyurethane foam (PUF), respectively (Fig. 1). The GFF mounted on the top of the  
99 sampler was held by a metal rack to collect airborne particles and a PUF disk was positioned under the GFF  
100 to sample the target chemicals in the gaseous phase as well as potentially the ones associated with fine  
101 (with a diameter  $< 2 \mu\text{m}$ ) particles (Bohlin et al., 2010). Samplers were deployed for 2-3 months at  
102 locations away from direct ventilation (i.e. open window and/or air supply outlet of air conditioners) and  
103 being mounted at a height of 1.5 – 2 m above the ground. After collection, GFF and PUF were retrieved and  
104 wrapped separately in aluminium foil and then stored at  $-20 \text{ }^\circ\text{C}$  until analysis. The study was conducted  
105 with the ethics approval from The University of Queensland (2015000153).

106 Collection of floor dust samples were detailed elsewhere (He et al., 2018). Briefly, they were collected  
107 using a clean nylon sampling sock with a vacuum cleaner and then sieved using a pre-cleaned 1mm-mesh  
108 metal sieve. Typically 0.1 gram of sieved dust was aliquoted from each sample and analysed.



109

110 **Figure 1.** Schematic description and exemplified deployment of the modified PAS used in this study

### 111 2.2 Chemical analysis

112 Solvents, reagents and chemical standards were purchased from multiple suppliers. Details are provided in  
113 Section S1 of the Supplementary Material (SM). The collected GFF, PUF and dust samples were spiked  
114 with 100  $\mu\text{L}$  of an internal standard solution containing multiple deuterated PAHs,  $^{13}\text{C}$ -labelled PCB  
115 congeners and  $^{13}\text{C}$ -labelled pesticides (in isooctane) for quantification purposes (Table S1). Subsequently,  
116 these samples were separately extracted via pressurised liquid extraction (PLE) on a Dionex ASE 350  
117 Accelerated Solvent Extractor (Thermo Fisher Scientific) before purified using a chromatographic column.  
118 Samples were analysed using a Thermo TRACE GC Ultra coupled to a TSQ Quantum XLS triple  
119 quadrupole mass spectrometer equipped with a TriPlus Autosampler. Target analytes include 13 PAH  
120 compounds, namely phenanthrene (Phe), anthracene (Ant), fluoranthene (Flu), pyrene (Pyr),  
121 benzo[a]anthracene+chrysene (BaA+Chr), benzo[b+k]fluoranthene (BbF+BkF), benzo[e]pyrene (BeP),  
122 benzo[a]pyrene (BaP), indeno[1,2,3-cd]pyrene (I123cdP), dibenzo[a,h]anthracene (DahA),  
123 benzo[g,h,i]perylene (BghiP); 8 PCB congeners including PCB 11, 28, 52, 101, 118, 138, 153 and 180; and  
124 12 pesticides including hexachlorobenzene (HCB), heptachlor (HEPT), heptachlor epoxide B (HEPX),  
125 *trans*-chlordane (TC), *cis*-chlordane (CC), *p,p'*-DDE, *p,p'*-DDT, chlorpyrifos (CPF), permethrin (PERM),  
126 cypermethrin (CYPERM), cyfluthrin (CYF), and deltamethrin (DELTAM). Details of chemical analysis are  
127 described in Section S2 in the SM.

### 128 2.3 Quality assurance and quality control (QA/QC)

129 *Sampling rate of the PAS.* To obtain the sampling rate  $R$  ( $\text{m}^3 \text{sampler}^{-1} \text{day}^{-1}$ ) of the PAS, a low-volume  
130 active air sampler (SAICI Technology Co., LTD., LSAM-100; (Wang et al., 2016)) operating at  $0.22 \text{ m}^3 \text{ h}^{-1}$   
131 was used to collect air samples in parallel with the PAS deployed at one of the sampling site (Office #1).  
132 An XAD cartridge (1 g) was used to trap chemicals in the gaseous phase beneath a GFF collecting particle-  
133 associated chemicals. The results from XAD were used to calculate the sampling rate of the PUF for  
134 gaseous chemicals. The chemical concentrations in gaseous phase are reported as  $\text{pg m}^{-3}$  after conversion  
135 and the particle-associated chemicals collected by the GFF of the PAS are reported as  $\text{ng g}^{-1}$  airborne  
136 particles.

137 We assumed a linear-phase sampling condition being maintained for all the chemicals on PAS during the  
138 sampling period (50 – 95 days) (Shoeib and Harner, 2002) and calculated the compound-specific sampling  
139 rate  $R$  from:

$$140 \quad R = \frac{C_{PAS}/t}{C_{AAS}} \quad (1)$$

141 where  $C_{PAS}$  is the amount of chemicals sequestered by the samplers (in  $\text{pg sampler}^{-1}$ ) during the deployment  
142 period  $t$  (days).  $C_{AAS}$  the volumetric concentrations (in  $\text{pg m}^{-3}$ ) of chemicals derived from the sample  
143 collected by the active air sampler. The calculated  $R$  was then applied to each sampler deployed on different  
144 sampling sites using:

$$145 \quad C_{Air} = \frac{C_{PAS}}{R \times t} \quad (2)$$



146 where  $C_{Air}$  is the converted volumetric concentrations in air (in  $\text{pg m}^{-3}$ ). It should be noted that uncertainties  
147 of  $R$  existed, due to the differences in ventilation capacity/rate at different homes and offices. Since efforts  
148 were made to avoid direct ventilation contact as mentioned above, the impact of wind speed and velocity on  
149  $R$  was expected to be minimal.

150 With the active XAD samples, 16 compounds were detected above their respective method detection limit  
151 (MDL) and the sampling rate  $R$  was derived directly from equation (1). For the other chemicals,  $R$  was  
152 estimated based on their molecular weights and structures. Detailed estimation method for specific  
153 compounds and  $R$  value for all the above chemicals (and extended ones) can be found in Table S2. Briefly,  
154  $R$  ranged from 0.69 to 3.4  $\text{m}^3 \text{sampler}^{-1} \text{day}^{-1}$  for the compounds targeted in the current study.

155 *Sampling and analytical precision.* Duplicated air samplers were deployed in Office #1 for 94 days and  
156 analysed separately to check the precision of sampling and chemical analysis for air samples. Similarly,  
157 dust samples from House #1 and Office #20 were respectively homogenised, aliquoted and analysed in  
158 duplicate to examine the precision of dust analysis. The precision was estimated based on the relative  
159 standard deviation (RSD) between the duplicate samples (i.e. one PUF pair, one GFF pair and two dust  
160 pairs). RSD ranged from 0.47% to 53% and for ~90% of the cases it was  $\leq 30\%$ . Full information is  
161 provided in Table S3 in the SM.

162 *Blank samples and method detection limits (MDLs).* Within each batch of samples analysed (typically 8  
163 samples per batch), a solvent blank, a matrix blank and a field blank were incorporated to check for any  
164 contamination related to analytical instruments, the sample preparation system and transportation and  
165 storage of samples. A pre-cleaned PUF, GFF and 0.1 gram of anhydrous sodium sulphate acted as the  
166 matrix blank for air, airborne particles and floor dust samples respectively. Non-exposed GFFs and PUFs ( $n$   
167 = 6 respectively) that underwent the same procedure of transportation of actual samples were used as the  
168 field blank for air samples. Anhydrous sodium sulphate was sealed in an air-tight jar and underwent the  
169 same procedure of transportation of actual dust samples to act as the field blank ( $n = 4$ ) for dust samples.  
170 MDLs were defined as the average field blank levels plus three times the standard deviation. If the relevant  
171 compounds could not be detected within the field blank samples, MDLs were determined based on half the  
172 instrument detection limits (IDLs). MDLs for the analytes ranged from 0.0012 to 140  $\text{pg m}^{-3}$  in air, from  
173 0.019 to 3400  $\text{ng g}^{-1}$  airborne particles and from 0.0020 to 2500  $\text{ng g}^{-1}$  floor dust (Table 1).

#### 174 **2.4 Estimation of human intake of SVOCs from inhalation, dust ingestion and dermal contact**

175 The equations and parameters used for this estimation were detailed in He et al. (2018) and references  
176 therein. That study collected and analysed indoor air and dust samples from a very similar set of sampling  
177 locations in the same cities with the current study. Briefly, intake estimation was conducted for toddlers and  
178 adults separately, based on a range of parameters/conditions including chemical concentrations ( $\text{pg m}^{-3}$  air  
179 or  $\text{ng g}^{-1}$  dust), fraction of the day spent at workplace and home (dimensionless), respiration rate ( $\text{m}^3 \text{day}^{-1}$ ),  
180 body weight (kg), dust ingestion rate ( $\text{mg day}^{-1}$ ), body surface area ( $\text{cm}^2 \text{day}^{-1}$ ), soil adhered to skin ( $\text{mg cm}^{-2}$ )

181 <sup>2</sup>) and fraction of chemicals absorbed in the skin (dimensionless). Detailed description of these parameters  
182 is provided with Table S6 in the SM.

## 183 **2.5 Statistical analysis**

184 Data analysis was performed using GraphPad Prism 8.0.1. Values lower than the MDL were replaced by ½  
185 MDL. Student's *t* test was used to determine differences of concentrations among samples. Criteria for  
186 significance was set at  $p < 0.05$ .

187

## 188 **3. Results and discussion**

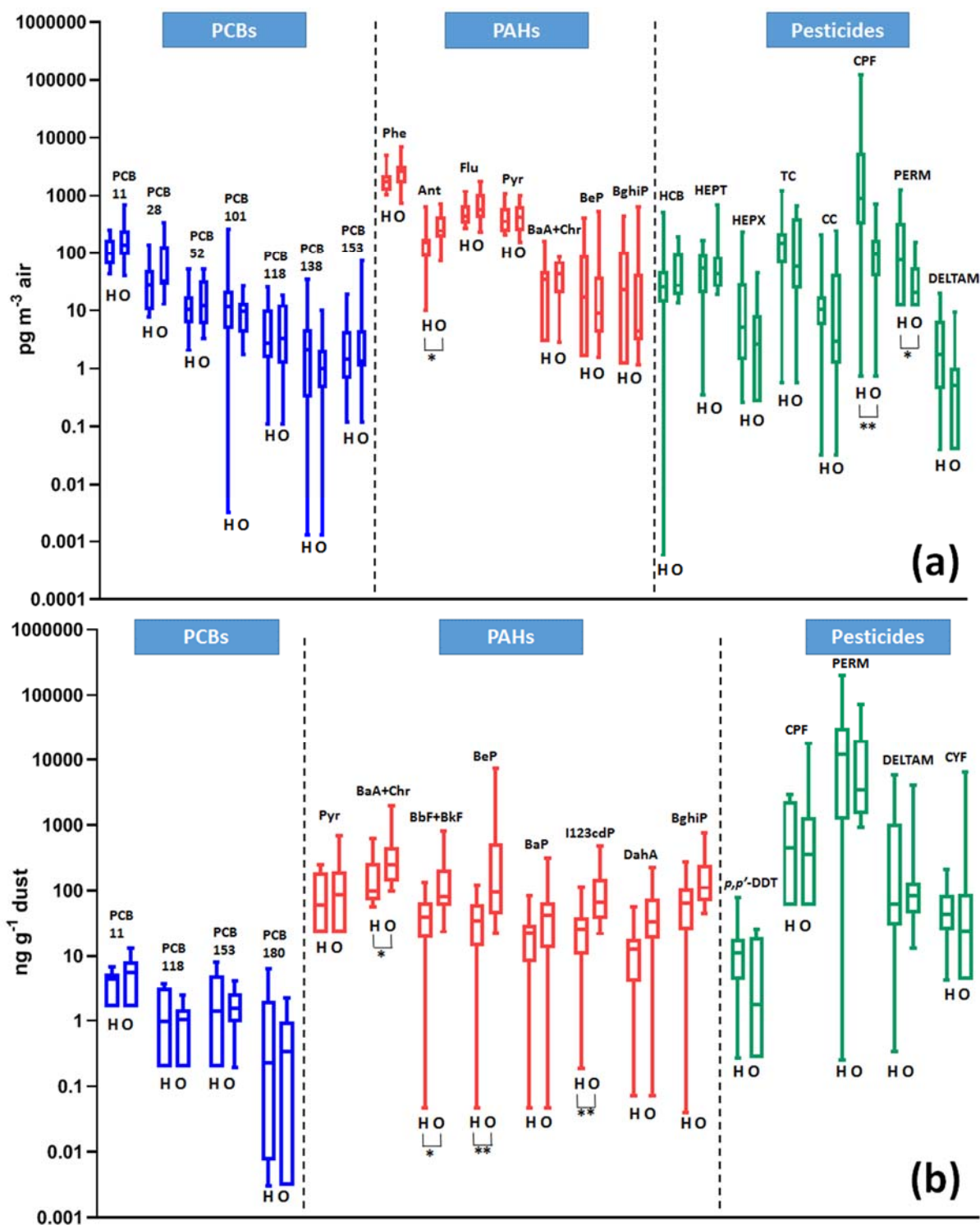
### 189 **3.1 SVOCs in indoor air**

190 Overall, out of the 33 analytes measured, 22 and 14 compounds were detected above the MDL in over 50%  
191 of gaseous and airborne particle samples respectively (Table 1). Amongst these chemicals, the highest  
192 median level in the gaseous phase and airborne particles was observed for phenanthrene (Phe; 2.0 ng m<sup>-3</sup>)  
193 and permethrin (PERM; 8800 ng g<sup>-1</sup>) respectively. Generally, in the same compound group, more volatile  
194 compounds were detected with a higher frequency in the gaseous phase and less volatile chemicals showed  
195 a higher detection frequency in the airborne particles (Table 1). Gas-particle distribution of target chemicals  
196 is discussed in details in Section S3 and Fig. S1 in the SM. In addition, the particle mass (calculated as µg  
197 day<sup>-1</sup>) measured from houses (with a median of 65) was overall higher than the offices (with a median of  
198 34). With the outlier (of data from Office #5) excluded from the office dataset, this difference was  
199 significant (Mann-Whitney test,  $p = 0.014$ ). This reflected an effectiveness of central ventilation systems in  
200 office settings to remove the large particles from the outdoor air being exchanged into indoor environments.  
201 It should be noted that our data for airborne particles are expressed as ng g<sup>-1</sup> particles and thus reflected a  
202 particle mass normalised concentration instead of volumetric levels e.g. ng m<sup>-3</sup>. We recognise this as a  
203 limitation of the comparability of our data with other indoor and outdoor studies. This was due to that our  
204 sampler design/modification collected “all” the airborne particles instead of size-characterised particles e.g.  
205 total suspended particles and there is currently a lack of reliable calibration dataset as we did for the  
206 gaseous SVOC sampling and monitoring (Section 2.3). However, with this limitation in mind, the data for  
207 SVOCs in airborne particles are still valuable. For example, comparing airborne particle data to dust data  
208 (which are reported in the same unit) allows us to estimate SVOC distribution between airborne particles  
209 and floor dust, which assists the assessment of contribution from relevant sources (see later sections).

210 **Table 1.** Detection frequency (DF), median and mean concentration and method detection limit (MDL) of PCBs, PAHs and pesticides in indoor air ( $\text{pg m}^{-3}$ ),  
 211 airborne particles ( $\text{ng g}^{-1}$ ) and floor dust ( $\text{ng g}^{-1}$ )

Compounds	Air ( $\text{pg m}^{-3}$ , n = 28)				Airborne particles ( $\text{ng g}^{-1}$ , n = 18)				Floor dust ( $\text{ng g}^{-1}$ , n = 28)			
	DF	Median	Mean $\pm$ SD	MDL	DF	Median	Mean $\pm$ SD	MDL	DF	Median	Mean $\pm$ SD	MDL
Phenanthrene (Phe)	100%	2000	2400 $\pm$ 1500	140	11%			3400	Nil			2500
Anthracene (Ant)	93%	190	260 $\pm$ 180	20	5.6%			280	Nil			230
Fluoranthene (Flu)	100%	500	640 $\pm$ 390	4.3	83%	290	340 $\pm$ 210	100	31%			75
Pyrene (Pyr)	100%	400	460 $\pm$ 250	9.0	89%	520	700 $\pm$ 590	230	62%	140	190 $\pm$ 170	45
Benzo[a]anthracene+chrysene (BaA+Chr)	69%	46	57 $\pm$ 29	5.6	89%	280	360 $\pm$ 210	91	97%	150	310 $\pm$ 410	0.061
Benzo[b+k]fluoranthene (BbF+BkF)	17%			7.7	78%	190	220 $\pm$ 120	54	93%	63	120 $\pm$ 190	0.092
Benzo[e]pyrene (BeP)	72%	24	95 $\pm$ 150	3.1	89%	130	150 $\pm$ 80	25	93%	48	480 $\pm$ 1500	0.092
Benzo[a]pyrene (BaP)	31%			0.10	44%			0.87	83%	30	54 $\pm$ 69	0.092
Indeno[1,2,3-cd]pyrene (I123cdP)	10%			6.3	72%	280	340 $\pm$ 200	46	93%	38	76 $\pm$ 95	0.38
Dibenzo[a,h]anthracene (DahA)	Nil			0.14	28%			1.3	83%	21	41 $\pm$ 50	0.14
Benzo[g,h,i]perylene (BghiP)	69%	35	120 $\pm$ 190	2.3	83%	140	200 $\pm$ 160	26	93%	79	140 $\pm$ 150	0.078
$\Sigma_{13}$ PAHs	NA	3000	4000 $\pm$ 2300		NA	1900	3400 $\pm$ 4600		NA	660	1400 $\pm$ 2000	
PCB 11	100%	110	150 $\pm$ 130	5.2	11%			80	66%	5.6	6.4 $\pm$ 2.5	3.2
PCB 28	100%	31	65 $\pm$ 85	1.4	17%			20	3.4%			5.4
PCB 52	100%	11	17 $\pm$ 14	0.0089	33%			0.17	34%			1.9
PCB 101	97%	11	21 $\pm$ 48	0.0064	50%	10	22 $\pm$ 22	0.11	28%			1.3
PCB 118	86%	3.5	7.2 $\pm$ 6.9	0.22	17%			3.7	69%	1.4	1.7 $\pm$ 1.0	0.39
PCB 138	86%	1.5	4.2 $\pm$ 7.2	0.0026	22%			0.040	24%			1.3
PCB 153	86%	1.8	6.5 $\pm$ 14	0.24	22%			3.7	79%	2.0	2.7 $\pm$ 2.0	0.39
PCB 180	45%			0.094	11%			1.3	69%	0.60	1.3 $\pm$ 1.7	0.0061
$\Sigma_8$ PCBs	NA	200	270 $\pm$ 170		NA	32	60 $\pm$ 110		NA	9.8	12 $\pm$ 9	

Hexachlorobenzene (HCB)	97%	27	63 ± 98	0.0012	78%	11	42 ± 98	0.019	Nil			0.0020
Heptachlor (HEPT)	93%	45	98 ± 150	0.69	33%			8.6	Nil			530
Heptachlor epoxide B (HEPX)	76%	6.0	26 ± 51	0.52	5.6%			6.2	17%			0.18
<i>Trans</i> -chlordane (TC)	93%	120	210 ± 250	1.1	72%	180	2000 ± 5800	29	3.4%			100
<i>Cis</i> -chlordane (CC)	93%	9.1	31 ± 60	0.063	44%			1.1	3.4%			11
<i>p,p'</i> -DDE	31%			2.7	17%			60	48%			1.4
<i>p,p'</i> -DDT	3.4%			27	28%			610	66%	15	17 ± 16	0.54
Chlorpyrifos (CPF)	90%	260	7100 ± 24000	1.5	78%	150	2800 ± 6300	27	69%	610	2000 ± 3800	120
Permethrin (PERM)	62%	84	260 ± 370	23	100%	8800	19000 ± 39000	420	93%	5100	23000 ± 41000	0.51
Cypermethrin (CYPERM)	3.4%			8.9	11%			160	48%			17
Cyfluthrin (CYF)	6.9%			4.6	44%			85	69%	69	400 ± 1400	8.5
Deltamethrin (DELTAM)	76%	1.1	4.2 ± 5.9	0.079	72%	110	1000 ± 2000	1.4	93%	82	660 ± 1300	0.67



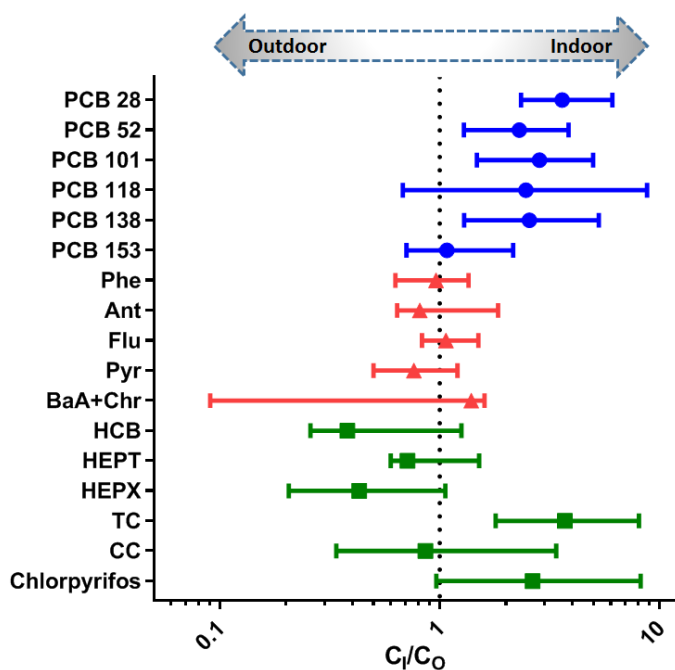
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214 **Figure 2.** Concentrations presented in log scale in air (Figure 2(a); pg m<sup>-3</sup>) and floor dust (Figure 2(b); ng g<sup>-1</sup>) for target compounds in houses (H) and offices (O). Significant difference in concentrations between  
 215 houses and offices for specific compounds was labelled as \* ( $p < 0.05$ ) or \*\* ( $p < 0.01$ ).  
 216

217 **PAHs.** In the gaseous phase, concentration of  $\sum_{13}$  PAHs ranged from 1.3 to 11 ng m<sup>-3</sup> with a median level  
 218 of 3.0 ng m<sup>-3</sup>. No statistically significant difference can be observed for PAH concentrations (log-  
 219 transformed) in air between houses and offices with the only exception of anthracene (Ant) whose  
 220 concentration in the offices was significantly higher than the houses ( $p = 0.034$ ; Fig. 2). A comparison of  
 221 indoor and outdoor concentration in air was conducted to assist the assessment of sources. We used data

222 from Wang et al. (2016) who reported the most up-to-date atmospheric PAH and PCB data for Brisbane  
223 city.

224 Generally the concentrations of PAHs measured in these indoor houses and offices were similar to the  
225 outdoor levels in Brisbane (with a median value of indoor/outdoor ratio of 0.76 – 1.4 for different  
226 compounds; Fig. 3). As previously mentioned, indoor environments are a relatively confined space and  
227 potentially have fewer photolytic reactions (Audy et al., 2018), compared to outdoor environments. If any  
228 significant PAH source presents in indoor environments, the indoor concentrations of PAHs are expected to  
229 be (much) higher than the outdoor environments. For example, wood/fuel burning within residential houses  
230 may significantly increase PAH concentrations in homes (Gustafson et al., 2008; Shen et al., 2011), leading  
231 to a summed PAH concentration in indoor air at the level of  $\mu\text{g}/\text{m}^3$  (Ding et al., 2012), 3 orders of  
232 magnitudes higher than that was observed in the current study. Therefore the indoor/outdoor ratio of 0.76 –  
233 1.4 for PAH concentrations in air may indicate the contribution from generic sources for indoor PAH  
234 burdens in air such as exchange from outdoor air, and may also suggest a limited role of cooking and  
235 smoking as sources within these indoor settings involved in the current study. A similar observation for  
236 PAHs has also been reported by Laborie et al., (2016) and Romagnoli et al., (2014). For offices with  
237 controlled ventilation only (i.e. without openable windows), this exchange of PAHs in gaseous phase and  
238 that are associated with fine particles from outdoor to indoor may occur through the central filtration system  
239 (K. Koponen et al., 2001; Lv and Zhu, 2013; Ohura et al., 2004; Weschler and Nazaroff, 2008). In the  
240 airborne particles, concentration of  $\sum_{13}$  PAHs ranged from 480 to 20000  $\text{ng g}^{-1}$  with a median value of 1900  
241  $\text{ng g}^{-1}$ . The highest level of benzo[a]pyrene (BaP) was 100  $\text{ng g}^{-1}$ .



242

243 **Figure 3.** Ratio of concentrations of PCBs (blue dots and lines), PAHs (red triangles and lines) and  
244 pesticides (green squares and lines) in indoor air ( $C_i$ , current study) and outdoor air ( $C_o$ ; Wang et al., 2015,  
245 2016 & 2017) in Brisbane. Median values with 95% confidence interval of the ratio for each chemical are  
246 shown in logarithmic scale.

247 **PCBs.** In the gaseous phase, with a few exceptions, the largest contribution to the sum concentration of  
248 PCBs was from 3,3'-dichlorobiphenyl (PCB 11; Fig. 2). To our knowledge, this is the first time that PCB  
249 11 concentrations are measured and reported in indoor environments. PCB 11 is an impurity that is formed  
250 during the production of yellow pigments that are contemporarily used in multiple consumer products  
251 (Hites, 2018; Hu et al., 2008; Rodenburg et al., 2009; Rodenburg et al., 2010; Shang et al., 2014). This may  
252 explain the relatively steady concentration of PCB 11 in ambient air in North America over the last decade,  
253 which is in contrast to other congeners such as dioxin-like PCBs (Hites, 2018). In the airborne particle  
254 phase, PCB congeners had a low DF (11% – 50%).

255 PCB data from outdoor air were collated from previous studies (Wang et al., 2015&2016) and compared to  
256 their indoor concentrations from the current study. PCB concentrations in indoor air generally exceeded  
257 those in outdoor air (with a median value of indoor/outdoor ratio of 1.1 – 3.6 for different congeners; Fig.  
258 3). This may indicate the contribution from indoor emission sources (Australian Government, 2018a;  
259 Benthe et al., 1992; Kohler et al., 2005). Similar findings were also reported previously (Li et al., 2018;  
260 Marek et al., 2017).

261 Furthermore, linear regression analysis was applied plotting the age of building (converted to number of  
262 Julian days at 1<sup>st</sup> Jan of the year built) and concentrations of PCBs and legacy pesticides in indoor air. Ten  
263 houses with relatively high DF of these chemicals were included with ages ranging from 12 to 50 years.  
264 Overall, an increase in PCB concentrations was observed with older age (Fig. S2), suggesting that older  
265 houses have higher PCB concentrations. It should be noted that, however, no statistical significance could  
266 be observed ( $p = 0.089 - 0.82$ ).

267 PCB concentrations in indoor air from the current study (Table S4) were at the lower end when compared  
268 to other studies in Europe, and North and Central America (Audy et al., 2018; Bohlin et al., 2008),  
269 reflecting the historical usage and emissions of PCBs being generally less in the Southern compared to the  
270 Northern Hemisphere (Breivik et al., 2002). Similar findings were also reported for outdoor ambient air  
271 (Wang et al., 2015).

272 **Pesticides.** In the gaseous phase, various legacy pesticides that have been banned in Australia for 2 – 4  
273 decades including hexachlorobenzene (HCB), heptachlor (HEPT) and chlordanes were detected in >75% of  
274 samples. The highest median concentration was measured for *trans*-chlordane (TC) at 120 pg m<sup>-3</sup> (Fig. 2).  
275 This value was at the higher end compared to data for Europe and Central America (Table S4). This was in  
276 line with the finding from outdoor air (Wang et al., 2015) and might be a reflection of the relatively higher  
277 usage of this pesticide historically in Australia. For the current pesticides, chlorpyrifos (CPF) had a  
278 relatively high concentration ranging from 25 to 120000 pg m<sup>-3</sup> with a median level of 260 pg m<sup>-3</sup> (Fig. 2).

279 In the airborne particle phase, legacy pesticides generally had a lower DF but the highest median level was  
280 still measured for TC (180 ng g<sup>-1</sup>). The current pesticide PERM was detected in all the samples with a much  
281 higher median level of 8800 ng g<sup>-1</sup> mirroring its current use for domestic pest control.

282 Concentrations of gaseous pesticides were consistently higher in houses than in offices, with a statistical  
283 significance observed for CPF and PERM ( $p = 0.0021$  and  $0.023$  respectively for log transformed data; Fig.  
284 2). This may again reflect their use pattern in different environments.

285 *Trans*- and *cis*-chlordane (TC and CC) are components contained in technical chlordane mixtures in a ratio  
286 of approximately 1.2 (Bidleman et al., 2002). Technical chlordane used to be applied for termite control  
287 (Kookana et al., 1998; Radcliffe, 2002) in Australia with registration being cancelled in 1997 (Australian  
288 Government, 2018b). The much higher ratio of TC/CC in the gaseous phase in this study (1.8 – 210) was an  
289 indication of close vicinity to source areas because TC has a higher vapour pressure than CC (Shen and  
290 Wania, 2005). In addition, this range was much wider (and overall higher) than the one for outdoor air  
291 across Australia (0.97 – 4.9; (Wang et al., 2015)). These results suggested that the major source of  
292 chlordanes in indoor environments in Australia maybe the volatilisation from previously treated materials  
293 (Shunthirasingham et al., 2010) within/around houses and offices instead of being from outdoor pathways  
294 (e.g. long-range atmospheric transport (LRAT)). In addition, the highest value of  $C_I/C_O$  was observed for  
295 TC in the pesticide group, which provides a further evidence of indoor-environment associated sources  
296 being dominant for contributing to its burdens in indoor air. The currently used pesticide CPF had a similar  
297 indoor-outdoor concentration ratio range (Fig. 3), reflecting also predominant sources in indoor settings,  
298 particularly the residential houses as discussed above. Wang et al. (2017) also attempted measuring outdoor  
299 concentrations of PERM in Brisbane and the values were all below  $2.1 \text{ pg m}^{-3}$  whereas in this current study  
300 it had a median concentration at  $84 \text{ pg m}^{-3}$ . This indicated the same indoor source dominance scenario for  
301 PERM as for CPF. On the other hand, the lowest ratio of  $C_I/C_O$  was found for HCB, which may mean a  
302 prominent contribution from outdoor air to indoor environments.

303 As with PCBs, a decreasing concentration of legacy pesticides was observed with newer houses but none  
304 was statistically significant ( $p = 0.16 - 0.59$ ).

305

### 306 3.2 SVOCs in floor dust

307 Overall, 17 out of 33 compounds were detected above the MDL with a DF of  $> 50\%$ . PERM had the  
308 highest median concentration ( $5100 \text{ ng g}^{-1}$ ) followed by CPF ( $610 \text{ ng g}^{-1}$ ) as shown in Fig. 2.

309 **PAHs.** Heavier PAH compounds were frequently detected in the floor dust samples. The median of BaP  
310 concentration was  $30 \text{ ng g}^{-1}$ , less than half of the value reported in dust samples collected 10 years ago from  
311 houses in the same city (Robertson et al., 2005) (Table S5). Concentrations of PAH compounds in floor  
312 dust from offices were consistently higher than houses. Amongst these, benzo[a]anthracene+chrysene  
313 (BaA+Chr), benzo[b+k]fluoranthene (BbF+BkF), benzo[e]pyrene (BeP) and indeno[1,2,3-cd]pyrene  
314 (I123cdP) were significantly higher (in log-transformed concentrations;  $p = 0.0070 - 0.020$ ) in offices than  
315 houses (Fig. 2). In addition, the ratio of  $C_{BaP}/(C_{BaP} + C_{BeP})$  was significantly higher in offices ( $0.41 \pm$   
316  $0.02$ ) than houses ( $0.37 \pm 0.02$ ) ( $p = 0.049$ ), indicating more contribution from relatively fresher particle  
317 emissions (Oliveira et al., 2011) in the offices. These may suggest that offices were less prone to the



318 influence of outdoor PAHs than homes (Romagnoli et al., 2014) and emission sources for PAHs within the  
319 offices such as operation of office equipment (Destailats et al., 2008) may contribute to their  
320 concentrations in dust. No significant correlation of  $C_{floor\ dust}$  and  $C_{air}$  was observed for any PAH  
321 compound, which indicated a non-equilibrium distribution of these chemicals between air and floor dust  
322 (Harrad et al., 2009; Tao et al., 2016). A detailed discussion is provided in Section S3 in the SM. No  
323 significant correlation was observed between  $C_{floor\ dust}$  and  $C_{airborne\ dust}$  for any target PAH, indicating a  
324 non-equilibrium distribution for these chemicals between airborne particles and floor dust and/or different  
325 source patterns. In 90% of cases the ratio of  $C_{airborne\ dust}/C_{floor\ dust}$  was equal to or higher than 1. This  
326 result reinforced the previously discussed theory that indoor PAH burdens may mainly come from outdoor  
327 contribution from ambient air. Overall the levels of PAHs measured from the current study were 10 – 100  
328 times lower than dust in Nigeria and China but 40-50 times higher than Czech Republic (Table S4)  
329 (Iwegbue et al., 2018; Melymuk et al., 2016; Wang et al., 2013).

330 **PCBs.** Similar to indoor air, PCB 11 still had the highest contribution towards total PCB content with a  
331 median value of 57% but was below the MDL for a few sites. No significant difference in concentration  
332 was observed for any PCB congener between houses and offices (Fig. 2). PCB concentrations from the  
333 current study were at the lower end of the range when compared to other studies in Europe and China  
334 (Table S4). Again, no significant correlation of  $C_{floor\ dust}$  and  $C_{air}$  was observed for any PCB congener,  
335 suggesting on-going emissions in the indoor environment.

336 **Pesticides.** The only legacy pesticide that had a DF of >50% was *p,p'*-DDT with a median concentration at  
337 15 ng g<sup>-1</sup>. In contrast, CPF and pyrethroids such as PERM, deltamethrin (DELTA) and cyfluthrin (CYF)  
338 had a higher DF and 4.5 – 330 times higher median concentration. This reflected the use pattern of these  
339 pesticides, i.e. that DDT has been banned for general use in Australia since 1987 (Radcliffe, 2002) whereas  
340 CPF, PERM, DELTA and CYF are all currently used pesticides (APVMA, 2018). Pesticide  
341 concentrations in dust samples were not significantly different between houses and offices (Fig. 2). It  
342 should be noted, however, that the DF of cypermethrin (CYPERM) was approximately 70% in houses,  
343 compared to approximately 30% in offices. This may indicate the major use of this pyrethroid in domestic  
344 application. Unlike PAHs and PCBs, significant correlations were observed for log-transformed  
345 concentrations of various current pesticides including CPF ( $p = 0.021$ ), PERM ( $p = 0.0063$ ) and DELTA  
346 ( $p = 0.00080$ ) between  $C_{floor\ dust}$  and  $C_{air}$ . This might be a reflection of the indoor use of these pesticides  
347 (e.g. for pest control) and relatively faster equilibrium post application between different phases due to their  
348 relatively low persistence. On the other hand, a significant correlation ( $p = 0.019$ ) was observed for  
349 concentrations of DELTA between  $C_{floor\ dust}$  and  $C_{airborne\ dust}$ , suggesting an equilibrium condition  
350 for this pesticide. Compared to available data from Europe and USA, levels of pyrethroids and CPF from  
351 the current study were higher (Table S4), indicating a relatively higher usage of these pesticides in Australia.

### 352 **3.3 Human exposure estimation**

353 Generally, inhalation is a more important pathway of exposure to more volatile compounds for both adults  
 354 and toddlers (Table 2). For PCB congeners, the highest estimated daily intakes (EDIs) were calculated for  
 355 PCB 11 for both adults (30 pg kg (b.w.)<sup>-1</sup> day<sup>-1</sup>) and toddlers (57 pg kg (b.w.)<sup>-1</sup> day<sup>-1</sup>), mostly via inhalation.  
 356 Pyr had the highest EDI amongst PAHs, namely 180 and 880 pg kg (b.w.)<sup>-1</sup> day<sup>-1</sup> for adults (mostly via  
 357 inhalation and dust contact) and toddlers (mostly via dust contact and ingestion), respectively.

358 For currently used pesticides including CPF and pyrethroids (e.g. PERM and DELTAM), dust contact and  
 359 ingestion contributed > 80% to the sum EDIs for both adults and toddlers (Table 2). This reflected their  
 360 main use patterns (e.g. as active ingredients in pest control products being sprayed onto floor and lower  
 361 wall surfaces) and after which their affinity with indoor dust. Toddlers overall had relatively higher EDIs  
 362 than adults via all routes. The highest ratio of toddler EDI against adult EDI was identified for PERM at 27,  
 363 mostly through dust contact and ingestion. Overall, the sum of EDIs via inhalation, dust contact and  
 364 ingestion from indoor environments from the current study were 3 – 5 orders of magnitudes lower than  
 365 reference dose (RfD; oral), indicating a negligible exposure risk.

366 **Table 2.** Exposure risk estimation for adults and toddlers via inhalation, dermal contact with dust and  
 367 ingestion of dust from indoor environments (median values in pg kg (b.w.)<sup>-1</sup> day<sup>-1</sup> for chemicals with a >50%  
 368 DF only). Values at 5% and 95% were also calculated and shown in Table S6 in the SM.

	Inhalation		Dust contact		Dust ingestion		Sum		RfD (oral) (USEPA, 2018)
	Adult	Toddler	Adult	Toddler	Adult	Toddler	Adult	Toddler	
PCB 11	26	31	4.0	11	0.18	15	30	57	NA
PCB 28	7.1	8.9	NA	NA	NA	NA	NA	NA	NA
PCB 52	2.6	3.4	NA	NA	NA	NA	NA	NA	NA
PCB 101	2.4	4.2	NA	NA	NA	NA	NA	NA	NA
PCB 118	0.80	0.97	0.99	4.8	0.044	6.4	1.8	12	NA
PCB 153	0.41	0.84	1.4	6.9	0.064	9.2	1.9	17	NA
PCB 138	0.35	0.98	NA	NA	NA	NA	NA	NA	NA
PCB 180	NA	0.16	0.43	1.2	0.019	1.6	NA	3.0	NA
∑ <sub>8</sub> PCBs	40	51	6.8	24	0.31	32	47	110	4.5×10 <sup>4*</sup>
Phe	450	570	NA	NA	NA	NA	NA	NA	NA
Ant	43	52	NA	NA	NA	NA	NA	NA	3×10 <sup>8</sup>
Flu	120	140	NA	NA	NA	NA	NA	NA	4×10 <sup>7</sup>
Pyr	91	120	89	310	4.3	450	180	880	3×10 <sup>7</sup>
BaA+Chr	11	15	99	230	4.8	330	110	580	NA
BbF+BkF	NA	NA	42	93	2.0	130	NA	NA	NA
BeP	5.4	12	32	80	1.5	110	39	200	NA
BaP	NA	NA	20	52	0.95	74	NA	NA	3×10 <sup>5</sup>
I123cdP	NA	NA	25	60	1.2	85	NA	NA	NA
DahA	NA	NA	14	32	0.66	45	NA	NA	NA
BghiP	8.0	16	52	150	2.5	220	63	390	NA
∑ <sub>13</sub> PAHs	730	930	370	1000	18	1400	1100	3300	NA
HCB	6.2	8.9	NA	NA	NA	NA	NA	NA	8×10 <sup>5</sup>
HEPT	10	24	NA	NA	NA	NA	NA	NA	5×10 <sup>5</sup>
HEPX	1.4	2.9	NA	NA	NA	NA	NA	NA	1.3×10 <sup>4</sup>
TC	27	48	NA	NA	NA	NA	NA	NA	5×10 <sup>5</sup>
CC	2.1	3.6	NA	NA	NA	NA	NA	NA	5×10 <sup>5</sup>
<i>p,p'</i> -DDE	NA	NA	NA	1.4	NA	8.5	NA	NA	NA
<i>p,p'</i> -DDT	NA	NA	2.3	6.6	0.48	41	NA	NA	5×10 <sup>5</sup>
CPF	60	310	310	1400	20	2500	390	4200	NA <sup>#</sup>
PERM	19	78	2600	26000	160	48000	2800	74000	5×10 <sup>7</sup>

CYPERM	NA	NA	NA	470	NA	880	NA	NA	NA
CYF	NA	NA	35	96	2.2	180	NA	NA	NA
DELTAM	0.26	0.71	41	140	2.6	270	44	410	NA

\* Averaged from the values for Arochlor 1016 and 1254; # RfD value for chlorpyrifos was estimated at  $10 \times 10^9$  but removed from the assessment system in 2011

369  
370

371

#### 372 4. Conclusions

373 The chemical cocktail of indoor environments consists of both legacy and emerging organic contaminants,  
374 seeing a ubiquitous detection of PCBs, PAHs and pesticides in Australian houses and offices. Their sources,  
375 however, are different – PCBs and some legacy (e.g. chlordane) and most of current pesticides are mainly  
376 from indoor sources while PAHs and some other legacy pesticides (e.g. HCB) are mainly from outdoor  
377 transportation. Although compared to other countries/regions, chlordane, chlorpyrifos and pyrethoid levels  
378 in Australian indoor environments are among the highest, human exposure risk remains negligible. Future  
379 studies are warranted to further improve the design of PAS collecting size-characterised particles and/or  
380 presenting calibration datasets for normalising airborne particle data to volumetric values e.g.  $\text{ng m}^{-3}$  to  
381 improve the comparability with other indoor and outdoor studies. Overall, simultaneous collection and  
382 analysis of gaseous phase, airborne particles and floor dust are an effective approach to estimate indoor  
383 burdens of SVOCs and evaluate potential sources.

384

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391

#### 392 Appendix A. Supplementary Material

393

#### 394 References

- 395 Abdallah MAE, Harrad S. Modification and Calibration of a Passive Air Sampler for Monitoring Vapor and  
396 Particulate Phase Brominated Flame Retardants in Indoor Air: Application to Car Interiors.  
397 Environmental Science & Technology 2010; 44: 3059-3065.
- 398 APVMA. APVMA PubCRIS database search. <https://portal.apvma.gov.au/pubcris>, 2018.
- 399 Audy O, Melymuk L, Venier M, Vojta S, Becanova J, Romanak K, et al. PCBs and organochlorine pesticides  
400 in indoor environments - A comparison of indoor contamination in Canada and Czech Republic.  
401 Chemosphere 2018; 206: 622-631.
- 402 Australian Government. Polychlorinated Biphenyls (PCBs).  
403 <http://www.npi.gov.au/resource/polychlorinated-biphenyls-pcbs>, 2018a.

404 Australian Government. Stockholm Convention on Persistent Organic Pollutants (POPs).  
405 <http://www.environment.gov.au/protection/chemicals-management/pops#action>, 2018b.

406 Benthe C, Heinzow B, Jessen H, Mohr S, Rotard W. Polychlorinated biphenyls. Indoor air contamination  
407 due to thiokol-rubber sealants in an office building. *Chemosphere* 1992; 25: 1481-1486.

408 Bidleman TF, Jantunen LM, Helm PA, Brorström-Lundén E, Juntto S. Chlordane enantiomers and temporal  
409 trends of chlordane isomers in Arctic air. *Environmental science & technology* 2002; 36: 539-544.

410 Bohlin P, Jones KC, Strandberg B. Field Evaluation of Polyurethane Foam Passive Air Samplers to Assess  
411 Airborne PAHs in Occupational Environments. *Environmental Science & Technology* 2010; 44: 749-  
412 754.

413 Bohlin P, Jones KC, Tovalin H, Strandberg B. Observations on persistent organic pollutants in indoor and  
414 outdoor air using passive polyurethane foam samplers. *Atmospheric Environment* 2008; 42: 7234-  
415 7241.

416 Breivik K, Sweetman A, Pacyna JM, Jones KC. Towards a global historical emission inventory for selected  
417 PCB congeners—a mass balance approach: 2. Emissions. *Science of the Total Environment* 2002;  
418 290: 199-224.

419 Destailats H, Maddalena RL, Singer BC, Hodgson AT, McKone TE. Indoor pollutants emitted by office  
420 equipment: A review of reported data and information needs. *Atmospheric Environment* 2008; 42:  
421 1371-1388.

422 Ding J, Zhong J, Yang Y, Li B, Shen G, Su Y, et al. Occurrence and exposure to polycyclic aromatic  
423 hydrocarbons and their derivatives in a rural Chinese home through biomass fuelled cooking.  
424 *Environmental Pollution* 2012; 169: 160-166.

425 Gustafson P, Östman C, Sällsten G. Indoor Levels of Polycyclic Aromatic Hydrocarbons in Homes with or  
426 without Wood Burning for Heating. *Environmental Science & Technology* 2008; 42: 5074-5080.

427 Harrad S, Abdallah MA-E, Covaci A. Causes of variability in concentrations and diastereomer patterns of  
428 hexabromocyclododecanes in indoor dust. *Environment International* 2009; 35: 573-579.

429 Harrad S, Brommer S, Mueller JF. Concentrations of organophosphate flame retardants in dust from cars,  
430 homes, and offices: an international comparison. *Emerging Contaminants* 2016; 2: 66-72.

431 Harrad S, Hazrati S, Ibarra C. Concentrations of polychlorinated biphenyls in indoor air and  
432 polybrominated diphenyl ethers in indoor air and dust in Birmingham, United Kingdom:  
433 Implications for human exposure. *Environmental Science & Technology* 2006; 40: 4633-4638.

434 He C, Wang X, Thai P, Baduel C, Gallen C, Banks A, et al. Organophosphate and brominated flame  
435 retardants in Australian indoor environments: Levels, sources, and preliminary assessment of  
436 human exposure. *Environmental Pollution* 2018; 235: 670-679.

437 Hites RA. Atmospheric Concentrations of PCB-11 Near the Great Lakes Have Not Decreased Since 2004.  
438 *Environmental Science & Technology Letters* 2018; 5: 131-135.

439 Hu D, Martinez A, Hornbuckle KC. Discovery of non-aro-chlor PCB (3, 3' -dichlorobiphenyl) in Chicago air.  
440 *Environmental science & technology* 2008; 42: 7873-7877.

441 IARC. Agents Classified by the IARC Monographs. World Health Organization.  
442 <http://monographs.iarc.fr/ENG/Classification/>. The International Agency for Research on Cancer.,  
443 2015.

444 Iwegbue CMA, Iteku-Atata E-OC, Odali EW, Egobueze FE, Tesi GO, Nwajei GE, et al. Distribution, Sources  
445 and Health Risks of Polycyclic Aromatic Hydrocarbons (PAHs) in Household Dusts from Rural,  
446 Semi-urban and Urban Areas in the Niger Delta, Nigeria. *Exposure and Health* 2018.

447 K. Koponen I, Asmi A, Keronen P, Puhto K, Kulmala M. Indoor air measurement campaign in Helsinki,  
448 Finland 1999 – the effect of outdoor air pollution on indoor air. *Atmospheric Environment* 2001;  
449 35: 1465-1477.

450 Kohler M, Tremp J, Zennegg M, Seiler C, Minder-Kohler S, Beck M, et al. Joint Sealants: An Overlooked  
451 Diffuse Source of Polychlorinated Biphenyls in Buildings. *Environmental Science & Technology*  
452 2005; 39: 1967-1973.

453 Kookana RS, Baskaran S, Naidu R. Pesticide fate and behaviour in Australian soils in relation to  
454 contamination and management of soil and water: a review. *Soil Research* 1998; 36: 715-764.

455 Laborie S, Moreau-Guigon E, Alliot F, Desportes A, Oziol L, Chevreuril M. A new analytical protocol for the  
456 determination of 62 endocrine-disrupting compounds in indoor air. *Talanta* 2016; 147: 132-141.

457 Li L, Arnot JA, Wania F. Towards a systematic understanding of the dynamic fate of polychlorinated  
458 biphenyls in indoor, urban and rural environments. *Environment International* 2018; 117: 57-68.

459 Lucattini L, Poma G, Covaci A, de Boer J, Lamoree MH, Leonards PEG. A review of semi-volatile organic  
460 compounds (SVOCs) in the indoor environment: occurrence in consumer products, indoor air and  
461 dust. *Chemosphere* 2018; 201: 466-482.

462 Lv J, Zhu L. Effect of central ventilation and air conditioner system on the concentration and health risk  
463 from airborne polycyclic aromatic hydrocarbons. *Journal of Environmental Sciences* 2013; 25: 531-  
464 536.

465 Marek RF, Thorne PS, Herkert NJ, Awad AM, Hornbuckle KC. Airborne PCBs and OH-PCBs Inside and  
466 Outside Urban and Rural U.S. Schools. *Environmental Science & Technology* 2017; 51: 7853-7860.

467 Meeker JD, Barr DB, Hauser R. Human semen quality and sperm DNA damage in relation to urinary  
468 metabolites of pyrethroid insecticides. *Human Reproduction* 2008; 23: 1932-1940.

469 Melymuk L, Bohlin-Nizzetto P, Vojta Š, Krátká M, Kukučka P, Audy O, et al. Distribution of legacy and  
470 emerging semivolatile organic compounds in five indoor matrices in a residential environment.  
471 *Chemosphere* 2016; 153: 179-186.

472 Ohura T, Amagai T, Fusaya M, Matsushita H. Polycyclic Aromatic Hydrocarbons in Indoor and Outdoor  
473 Environments and Factors Affecting Their Concentrations. *Environmental Science & Technology*  
474 2004; 38: 77-83.

475 Oliveira C, Martins N, Tavares J, Pio C, Cerqueira M, Matos M, et al. Size distribution of polycyclic aromatic  
476 hydrocarbons in a roadway tunnel in Lisbon, Portugal. *Chemosphere* 2011; 83: 1588-1596.

477 Radcliffe JC. Pesticide use in Australia: Australian Academy of Technological Sciences and Engineering,  
478 2002.

479 Robertson ST, Ayoko GA, Duigu JR. Elemental and polycyclic aromatic hydrocarbon compositions of house  
480 dust in Brisbane, Australia. 2005.

481 Rodenburg LA, Guo J, Du S, Cavallo GJ. Evidence for unique and ubiquitous environmental sources of 3, 3'  
482 -dichlorobiphenyl (PCB 11). *Environmental science & technology* 2009; 44: 2816-2821.

483 Rodenburg LA, Guo J, Du S, Cavallo GJ. Evidence for Unique and Ubiquitous Environmental Sources of 3,3'  
484 -Dichlorobiphenyl (PCB 11). *Environmental Science & Technology* 2010; 44: 2816-2821.

485 Romagnoli P, Balducci C, Perilli M, Gherardi M, Gordiani A, Gariazzo C, et al. Indoor PAHs at schools,  
486 homes and offices in Rome, Italy. *Atmospheric environment* 2014; 92: 51-59.

487 Schweizer C, Edwards RD, Bayer-Oglesby L, Gauderman WJ, Ilacqua V, Jantunen MJ, et al. Indoor time-  
488 microenvironment-activity patterns in seven regions of Europe. *Journal of Exposure Science and*  
489 *Environmental Epidemiology* 2007; 17: 170.

490 Shang H, Li Y, Wang T, Wang P, Zhang H, Zhang Q, et al. The presence of polychlorinated biphenyls in  
491 yellow pigment products in China with emphasis on 3,3' -dichlorobiphenyl (PCB 11). *Chemosphere*  
492 2014; 98: 44-50.

493 Shen L, Wania F. Compilation, evaluation, and selection of physical- chemical property data for  
494 organochlorine pesticides. *Journal of Chemical & Engineering Data* 2005; 50: 742-768.

495 Shen G, Wang W, Yang Y, Ding J, Xue M, Min Y, et al. Emissions of PAHs from Indoor Crop Residue Burning  
496 in a Typical Rural Stove: Emission Factors, Size Distributions, and Gas-Particle Partitioning.  
497 *Environmental Science & Technology* 2011; 45: 1206-1212.

498 Shoeib M, Harner T. Characterization and comparison of three passive air samplers for persistent organic  
499 pollutants. *Environmental Science & Technology* 2002; 36: 4142-4151.

500 Shunthirasingham C, Oyiliagu CE, Cao X, Gouin T, Wania F, Lee S-C, et al. Spatial and temporal pattern of  
501 pesticides in the global atmosphere. *Journal of Environmental Monitoring* 2010; 12: 1650-1657.

502 Tao F, Abdallah MA-E, Harrad S. Emerging and Legacy Flame Retardants in UK Indoor Air and Dust:  
503 Evidence for Replacement of PBDEs by Emerging Flame Retardants? *Environmental Science &*  
504 *Technology* 2016; 50: 13052-13061.

505 Tao S, Cao J, Wang WT, Zhao JY, Wang W, Wang ZH, et al. A Passive Sampler with Improved Performance  
506 for Collecting Gaseous and Particulate Phase Polycyclic Aromatic Hydrocarbons in Air.  
507 *Environmental Science & Technology* 2009; 43: 4124-4129.

508 Toms L-ML, Bartkow ME, Symons R, Paepke O, Mueller JF. Assessment of polybrominated diphenyl ethers  
509 (PBDEs) in samples collected from indoor environments in South East Queensland, Australia.  
510 *Chemosphere* 2009; 76: 173-178.

511 USEPA. Integrated Risk Information System. <https://www.epa.gov/iris>, 2018.

512 Wang W, Wu F, Zheng J, Wong MH. Risk assessments of PAHs and Hg exposure via settled house dust and  
513 street dust, linking with their correlations in human hair. *Journal of Hazardous Materials* 2013;  
514 263: 627-637.

515 Wang X, Kennedy K, Powell J, Keywood M, Gillett R, Thai P, et al. Spatial distribution of selected persistent  
516 organic pollutants (POPs) in Australia's atmosphere. *Environmental Science: Processes & Impacts*  
517 2015; 17: 525-532.

518 Wang X, Thai PK, Li Y, Li Q, Wainwright D, Hawker DW, et al. Changes in atmospheric concentrations of  
519 polycyclic aromatic hydrocarbons and polychlorinated biphenyls between the 1990s and 2010s in  
520 an Australian city and the role of bushfires as a source. *Environmental pollution* 2016; 213: 223-  
521 231.

522 Wang X, Thai PK, Mallet M, Desservettaz M, Hawker DW, Keywood M, et al. Emissions of selected  
523 semivolatile organic chemicals from forest and savannah fires. *Environmental Science &*  
524 *Technology* 2017; 51: 1293-1302.

525 Weschler CJ, Nazaroff WW. Semivolatile organic compounds in indoor environments. *Atmospheric*  
526 *Environment* 2008; 42: 9018-9040.

527 Wilford BH, Harner T, Zhu JP, Shoeib M, Jones KC. Passive sampling survey of polybrominated diphenyl  
528 ether flame retardants in indoor and outdoor air in Ottawa, Canada: Implications for sources and  
529 exposure. *Environmental Science & Technology* 2004; 38: 5312-5318.