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## **Indoor inhalation intake fractions of fine particulate matter: Review of influencing factors**

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## 40 **Abstract**

41 Exposure to fine particulate matter (PM<sub>2.5</sub>) is a major contributor to the global human  
42 disease burden. The indoor environment is of particular importance when considering the  
43 health effects associated with PM<sub>2.5</sub> exposures because people spend the majority of their  
44 time indoors and PM<sub>2.5</sub> exposures per unit mass emitted indoors are two to three orders of  
45 magnitude larger than exposures to outdoor emissions. Variability in indoor PM<sub>2.5</sub> intake  
46 fraction ( $iF_{in,total}$ ), which is defined as the integrated cumulative intake of PM<sub>2.5</sub> per unit  
47 of emission, is driven by a combination of building-specific, human-specific, and  
48 pollutant-specific factors. Due to a limited availability of data characterizing these  
49 factors, however, indoor emissions and intake of PM<sub>2.5</sub> are not commonly considered  
50 when evaluating the environmental performance of product life cycles. With the aim of  
51 addressing this barrier, a literature review was conducted and data characterizing factors  
52 influencing  $iF_{in,total}$  were compiled. In addition to providing data for the calculation of  
53  $iF_{in,total}$  in various indoor environments and for a range geographic regions, this paper  
54 discusses remaining limitations to the incorporation of PM<sub>2.5</sub>-derived health impacts into  
55 life cycle assessments and makes recommendations regarding future research.

56

## 57 **Practical Implications**

58 This paper reviews and summarizes the factors that influence indoor inhalation intake  
59 fraction of fine particulate matter, with a focus on primary particle emissions indoors. It  
60 provides valuable data for the calculation of indoor inhalation intake fraction for a range  
61 of indoor environments and contributes to the effort to incorporate PM<sub>2.5</sub>-derived health  
62 impacts into life cycle assessment.

63

64 **Key words:** fine particulate matter (PM<sub>2.5</sub>), human exposure, indoor air, intake fraction,  
65 life cycle impact assessment (LCIA), ventilation

## 66 **Introduction**

67 Human exposure to fine particulate matter (PM<sub>2.5</sub>) is a major contributor to  
68 disease burden on a global scale (WHO, 2002, 2013). The indoor environment is a  
69 particularly important venue for exposure to PM<sub>2.5</sub> because people spend the majority of

70 their time indoors (Klepeis et al., 2001; Phillips and Moya, 2014 and references therein).  
71 Further, due to the lesser degree of dilution, chemical transformation, and dispersion, as  
72 well as the higher density of occupants indoors, exposures per unit mass of PM<sub>2.5</sub> emitted  
73 indoors are two to three orders of magnitude larger than exposures to emissions to the  
74 outdoor environment (Smith, 1988; Lai et al., 2000; Klepeis and Nazaroff, 2006; Ilacqua  
75 et al., 2007; Nazaroff, 2008). In order to fully assess the impacts associated with all  
76 emission sources of PM<sub>2.5</sub> and to evaluate the life cycle environmental performance of  
77 products and systems (e.g., energy and transport systems, food products and production  
78 systems, and consumer products), there is a need for the incorporation of PM<sub>2.5</sub>  
79 exposures and the associated health effects into Life Cycle Impact Assessments (LCIA),  
80 with a specific need for the consideration of the impacts related to indoor exposures to  
81 PM<sub>2.5</sub> emitted or formed indoors.

82 Due to current limitations in data availability and modeling tools that  
83 systematically combine indoor and outdoor intakes from indoor and outdoor sources, as  
84 well as challenges in consistently linking indoor and outdoor intakes to exposure-  
85 response, indoor sources and related intake of PM<sub>2.5</sub> are currently not considered in  
86 product-related assessments (Humbert et al., 2015). To integrate indoor sources into such  
87 assessment frameworks, there is a need for (1) the identification of factors contributing  
88 substantially to variability in PM<sub>2.5</sub> exposure and an examination of the value of  
89 accounting for this variability when assessing PM<sub>2.5</sub> health impacts, (2) the aggregation  
90 and evaluation of modeling tools and data available for assessing human exposure to  
91 PM<sub>2.5</sub>, and (3) a thorough assessment of the availability of exposure-response functions  
92 (ERFs) and the appropriateness of ERF shape (e.g., linear, non-linear, presence of a  
93 threshold) for a variety of health outcomes (Fantke et al., 2015). With the aim of  
94 addressing these barriers and the lack of a standardized methodology to estimate  
95 exposures and health effects, the United Nations Environment Programme (UNEP)-  
96 Society for Environmental Toxicology and Chemistry (SETAC) Life Cycle Initiative  
97 formed a task force to provide guidance for the assessment of PM<sub>2.5</sub> exposures and  
98 associated health effects (Jolliet et al., 2014; Fantke et al., 2015). Under the framework of  
99 this task force and with input from an international team of experts, this paper constitutes  
100 a first step toward incorporating indoor PM<sub>2.5</sub> exposures into LCIA by characterizing the

101 factors that drive variability in the inhalation intake fraction of  $PM_{2.5}$  derived from indoor  
102 sources.

103 Inhalation intake fraction ( $iF$ ), which is defined as the ratio of mass of a pollutant  
104 inhaled by an exposed human population to the total mass associated with a given source  
105 (Bennett et al., 2002), provides a well-suited metric by which to consider  $PM_{2.5}$  impacts  
106 in the context of LCIA. As an exposure metric,  $iF$  integrates components that are key to  
107 such assessments: (1) it describes source-receptor relationships in a manner that allows  
108 for direct comparisons across emission sources and (2) it can readily be related to  
109 potential toxicity in terms of specific health outcomes when exposure-response  
110 relationships are known (Bennett et al., 2002; Ilacqua et al., 2007; Nazaroff, 2008; Fantke  
111 et al., 2015). Table 1 illustrates the contributions of  $PM_{2.5}$  derived from indoor sources  
112 ( $S_{in}$ ) and outdoor sources ( $S_{out}$ ) to indoor intake, outdoor intake, total intake, and the  
113 intake fraction of  $PM_{2.5}$ . As is described in detail below, this paper reviews the major  
114 factors influencing the inhalation intake fraction of  $PM_{2.5}$  derived from indoor sources  
115 (Table 1, Equation 1). Examples of common indoor sources of  $PM_{2.5}$  include cooking,  
116 household and office appliances, smoking, cleaning, candles, and heating appliances or  
117 stoves. Additional efforts are currently underway within the UNEP-SETAC LCIA  
118 framework to characterize the other aspects of  $PM_{2.5}$  intake and intake fraction shown in  
119 Table 1.

120 Indoor inhalation intake fraction ( $iF_{in,total}$ ) describes the total inhalation intake of  
121  $PM_{2.5}$  (in kg) per unit mass emitted indoors (in kg). Two components contribute to  
122  $iF_{in,total}$  (Table 1, Equation 1): (1) the fraction of  $PM_{2.5}$  emitted or formed indoors that is  
123 taken in via inhalation indoors ( $iF_{in \rightarrow in}$ ) and (2) the fraction of  $PM_{2.5}$  emitted or formed  
124 indoors that is transported outdoors and taken in via inhalation outdoors ( $iF_{in \rightarrow out}$ ).  
125 However, because  $PM_{2.5}$  of indoor origin experiences a greater degree of dispersion and  
126 dilution following transport outdoors and outdoor population density is lower than  
127 indoors,  $iF_{in \rightarrow out}$  is typically three orders of magnitude smaller than  $iF_{in \rightarrow in}$  (Smith,  
128 1988; Lai et al., 2000; Klepeis and Nazaroff, 2006; Ilacqua et al., 2007; Nazaroff, 2008;  
129 Humbert et al., 2011). Thus, in calculations of  $iF_{in,total}$ ,  $iF_{in \rightarrow out}$  can be considered  
130 negligible compared to  $iF_{in \rightarrow in}$ . As a result, this paper focuses on characterizing the major  
131 factors contributing to variability in  $iF_{in \rightarrow in}$ , as this term dominates  $iF_{in,total}$ . While not

132 the main focus, we also note the importance of interactions between pollutants of outdoor  
133 and indoor origin and the influence of outdoor PM<sub>2.5</sub> sources on cumulative indoor intake  
134 (Table1, Equation 2) and briefly discuss the current state of knowledge regarding these  
135 aspects.

136 Nazaroff (2008) divided the factors influencing variability in  $iF_{in \rightarrow in}$  for primary  
137 particles into three categories: (1) factors related to building characteristics (e.g.,  
138 ventilation, airflow, and mixing rates), (2) factors related to occupant characteristics and  
139 behaviors (e.g., inhalation rates and occupancy/activity patterns), and (3) pollutant  
140 dynamics (e.g., first order removal processes and sorptive interactions). That study noted  
141 the need for a “richly constituted tool kit to effectively comprehend the system of the  
142 human health risk associated with products and processes in indoor environments.”  
143 Humbert et al. (2011) provided an initial set of parameters characterizing two archetypal  
144 indoor environments (residences within the United States [U.S.] and mechanically  
145 ventilated offices). Herein, we expand on that effort by developing an inventory of  
146 parameters (i.e., a “tool kit”) to (1) address each of the factors influencing  $iF_{in \rightarrow in}$   
147 discussed by Nazaroff (2008) and (2) allow for the characterization of multiple archetypal  
148 indoor environments (e.g., residences, offices, schools, etc.), covering a broad range of  
149 geographic scales.

150

## 151 **Methods**

152 For each category of factors influencing  $iF_{in \rightarrow in}$  (building, occupant, and pollutant  
153 factors), sub-groups with expertise in that specific field were created within an indoor-air  
154 task force. Literature searches conducted by each sub-group were obtained from Web of  
155 Science, Google Scholar, and/or SCOPUS with search terms representing sources of  
156 variability related to the above-described categories (e.g., “air exchange rate  
157 measurements,” “building ventilation,” “commercial building ventilation rates,”  
158 “inhalation rates,” “indoor particle deposition,” “indoor particle emission rates,” etc.).  
159 When available, review papers were preferentially selected to be included in this review  
160 due to its multidimensional focus. Collected references were then reviewed and compiled  
161 to provide an inventory of data-sources (e.g., peer-reviewed scientific articles and  
162 reports) and data regarding each factor influencing  $iF_{in \rightarrow in}$ . We included key papers (i.e.,

163 those with the most sound experimental/modeling practices, those that provide the  
164 greatest breadth of data, and those that allow for consideration of a range of exposure  
165 scenarios) in the present review and provide data from those papers in the supporting  
166 information (SI). In general, the data compiled include summary statistics (i.e., mean,  
167 standard deviation, geometric mean, geometric standard deviation, percentiles, minimum,  
168 and maximum values) from individual studies conducted under a variety of experimental  
169 conditions and for a range of geographic locations. Where possible, data are categorized  
170 by country/geographic region and specific conditions in order to allow for the selection of  
171 data most relevant to an exposure-scenario of interest. Each factor contributing to  
172 variability in  $iF_{in \rightarrow in}$  is discussed in an individual section below.

173

## 174 **Building Factors**

175 Building-specific factors influencing  $iF_{in \rightarrow in}$  include building volume and  
176 ventilation (Table 1, Equations 1 and 2). Building ventilation is a key parameter in  
177 estimating  $iF_{in \rightarrow in}$ , as it drives the transport, dispersion, and dilution of PM<sub>2.5</sub> emitted  
178 indoors. Indoor ventilation is driven by three processes: (1) leakage through cracks in the  
179 building shell and walls (infiltration/exfiltration), (2) airflow through open windows and  
180 doors (natural ventilation), and (3) mechanical ventilation (i.e., flow driven by fans; Chan  
181 et al., 2005; US EPA, 2011). Infiltration/exfiltration and natural ventilation are driven by  
182 pressure gradients that exist across the building envelope due to indoor-outdoor  
183 temperature differences and wind (US EPA, 2011). Mechanical ventilation systems range  
184 between exhaust- or supply-only systems (e.g., bathroom and kitchen exhaust  
185 fans/hoods), balanced supply and exhaust systems, localized unitary/single-zone systems,  
186 and central/integrated systems (Sippola and Nazaroff, 2002; Brelvi and Seppänen, 2011;  
187 Litiu, 2012). Building ventilation is typically quantified as whole-building/whole-zone air  
188 exchange rates (AERs) [ $\text{h}^{-1}$ ] or, as is common for non-residential/commercial buildings,  
189 volumetric flow rate normalized by building occupancy, volume, or floor area [ $\text{L s}^{-1}$   
190  $\text{person}^{-1}$ ,  $\text{L s}^{-1} \text{m}^{-3}$ ,  $\text{L s}^{-1} \text{m}^{-2}$ ] (Persily, 2015). In the following paragraphs, we review the  
191 body of literature focused on characterizing these building properties and processes in a  
192 range of building archetypes.

193

194 **Residential Buildings**

195 Residential ventilation rates have been most heavily studied in Europe (Hänninen  
196 et al., 2011; Dimitroulopoulou, 2012 and references therein; Asikainen et al., 2013; Orru  
197 et al., 2014) and North America (Figure 1a) (Clark et al., 2010; Persily et al., 2010; US  
198 EPA, 2011 and references therein; Chen et al., 2012; MacNeil et al., 2012, 2014; El Orch  
199 et al., 2014; Bari et al., 2014; Breen et al., 2014; Persily, 2015). While more limited in  
200 their number and scope, some studies have also been carried out in New Zealand (McNeil  
201 et al., 2012), Asia (Baek et al. 1997; Williams and Eunice, 2013; Huang et al., 2014; Park  
202 et al., 2014; Li and Li, 2015; Shi et al., 2015), Africa, and South America (Williams and  
203 Eunice, 2013 and references therein) (Figure 1a). In addition to those studying the  
204 housing stock in broad geographic regions, some studies have focused on homes with  
205 specific characteristics (e.g., new homes, energy-efficient homes, low-income/public  
206 housing; Zota et al., 2005; US EPA, 2011). A limited number of studies have  
207 characterized ventilation in homes in developing countries (Williams and Eunice, 2013,  
208 L'Orange et al., 2015, and references therein) (Figure 1a). The use of solid fuels for  
209 cooking and heating, particularly in developing countries, is a leading indoor air quality  
210 issue on a global scale, with approximately 4.3 million premature deaths annually  
211 attributed to related pollutant exposures ([www.WHO.int/indoorair/en](http://www.WHO.int/indoorair/en)). As a result, such  
212 measurements for homes in developing countries are very important to the effort to  
213 incorporate the impacts of indoor PM<sub>2.5</sub> exposures into LCIA.

214 The above-described body of work illustrates that there is spatial variability in  
215 residential ventilation with climate, building construction characteristics, home age,  
216 heating, ventilation, and air conditioning (HVAC) system configurations, ventilation  
217 standards and regulations, and residence type (i.e., detached, single family homes,  
218 apartments) (Figure 2a). Temporal heterogeneity in ventilation rates results from  
219 variability in meteorological conditions and human behaviors such as window opening  
220 and mechanical ventilation system usage. The compilation of data characterizing homes  
221 over a broad range of geographic scales, housing types, seasons, and meteorological  
222 conditions is needed because the prevalence of different ventilation systems varies  
223 strongly across these factors. For example, AERs in 100% of both apartments and  
224 detached homes in Bulgaria are driven by infiltration and natural ventilation. On the other

225 hand, 48% of detached homes in Finland have mechanical ventilation systems. This  
226 proportion increases to 72% when considering apartments (Litiu, 2012). To aid in the  
227 selection of representative ventilation parameters when calculating  $iF_{in \rightarrow in}$ , the  
228 ventilation rates and air exchange rate data provided here are categorized by country,  
229 home type, season, and ventilation system where the available data allow for this (Figure  
230 1a and SI). Studies characterizing window-opening behavior and/or mechanical  
231 ventilation system usage and runtime (e.g., Iwashita and Akasaka, 1997; Chao, 2001;  
232 Wallace et al., 2002; Johnson and Long, 2005; US EPA, 2011; Fabi et al., 2012; Marr et  
233 al., 2012; Breen et al., 2014; El Orch et al., 2014; Gorenzenski et al., 2014; Levie et al.,  
234 2014; Persily, 2015; Stephens, 2015) provide needed information for accounting for  
235 temporal and spatial variability in ventilation conditions.

236 Figure 2a summarizes available residential air exchange rate data, with detailed  
237 data provided in the SI. For all residential AER measurements combined, we observed a  
238 median value of  $0.50 \text{ h}^{-1}$  (95% confidence interval [CI] =  $0.08, 8.2 \text{ h}^{-1}$ ) (Figure 2a), which  
239 is slightly higher than the recommended median value of  $0.45 \text{ h}^{-1}$  for homes in the U.S.  
240 provided in the Environmental Protection Agency Exposure Factors Handbook (US EPA  
241 EFH) (US EPA, 2011). This difference can likely be attributed, at least in part, to our  
242 inclusion of a small number measurements from high AER homes in developing  
243 countries, as well as differences in home characteristics and ventilation systems across  
244 nations. While treated as a single distribution above for the purpose of comparison  
245 against the recommended value in the US EPA EFH, residential AERs are likely best  
246 characterized by a bimodal distribution. This is evidenced by differences in the median  
247 AER values for homes in developed and developing countries: median (95% CI) =  $0.48$   
248  $(0.08 \text{ } 2.26) \text{ h}^{-1}$  and  $14.1 (2.0, 61.0) \text{ h}^{-1}$ , respectively.

249 Many of the studies described above in which air exchange and ventilation are  
250 measured also provide data regarding the volume/floor area of the homes studied (Figure  
251 1f). It is important to note that homes included are not necessarily statistically  
252 representative of the housing stock and this influences estimates of both home volume  
253 and ventilation. Population-level data describing home characteristics can also typically  
254 be gathered from census and housing survey databases (e.g., the American Census,  
255 American Housing Survey, Eurostat, and Census India). Recommended values for

256 various housing and building characteristics are also available in reports summarizing  
257 exposure factors in several countries (US EPA, 2011; Phillips and Moya, 2014 and  
258 references therein). Available measurements of residential volumes illustrate their high  
259 variability, both within and across nations, with values ranging from 15 – 1446 m<sup>3</sup>  
260 (median [95% CI] = 247 [41, 971] m<sup>3</sup>) (see SI). The median residential volume for the  
261 studies considered in this work is lower than the recommended value provided in the US  
262 EPA EFH (492 m<sup>3</sup>) (US EPA, 2011), likely illustrating differences in residential volumes  
263 across regions of the world.

264

### 265 **Non-Residential Buildings**

266 Ventilation measurements have been conducted in a range of non-residential  
267 buildings, including retail stores (US EPA, 2011; Zaatari et al., 2014 and references  
268 therein; Dutton et al., 2015), schools, kindergartens, and daycare centers (Coley and  
269 Beisteiner, 2002; Wargocki et al., 2002; Emmerich and Crum, 2006; Mi et al., 2006; Li et  
270 al., 2007; Guo et al., 2008; Santamouris et al., 2008; Brehlih and Seppänen, 2011;  
271 Sundell et al., 2011; Aelenei et al., 2013; Canha et al., 2013) offices (Persily and Gorfain,  
272 2004; Dimitropoulou and Bartzis, 2013), fitness facilities (Zaatari et al., 2014), jails  
273 (Seppänen et al., 1999; Li et al., 2007), and healthcare facilities, hospitals, and nursing  
274 homes (Wargocki et al., 2002, Li et al., 2007 and references therein). Summary statistics  
275 of more than 700 measurements from 17 studies, for example, have been compiled for  
276 retail facilities, bars/restaurants, healthcare facilities, fitness facilities, offices, and  
277 schools (Zaatari et al., 2014). As is true for residential ventilation rates, measurements in  
278 non-residential buildings are more heavily focused in North America and Europe, with a  
279 smaller number of studies also conducted in Asia (Figure 1a). Non-residential AERs are  
280 summarized in Figure 2a, with more detailed information (e.g., categorized by building  
281 type) provided in the SI. We observed a median AER for non-residential buildings of 1.5  
282 h<sup>-1</sup> (95% CI = 0.29, 9.1 h<sup>-1</sup>).

283 The above-described studies again demonstrate geographic variability in  
284 ventilation-system characteristics and the prevalence of mechanically and naturally  
285 ventilated buildings, as well as temporal variability in ventilation with meteorological  
286 conditions, window opening, and HVAC-system operation. For example, 100% of

287 schools and kindergartens are naturally ventilated in Italy, while only 5% and 28% of  
288 kindergartens and schools are naturally ventilated in Finland (Litiu, 2012). Sippola and  
289 Nazaroff (2002) note that single-zone HVAC systems are common in smaller commercial  
290 buildings with floor areas on the order of 150 m<sup>2</sup>, while central systems dominate in  
291 larger buildings (>1000 m<sup>2</sup>) such as malls, university buildings, theaters, and retail  
292 centers.

293     ▪ A small number of studies discuss window-opening and HVAC-system-use  
294 behavior in commercial/non-residential buildings (e.g., Fabi et al., 2012; Roetzel et al.,  
295 2010; Ramos and Stephens, 2014; D’Oca and Hong, 2014; Li et al., 2015; Stephens,  
296 2015). Two recent studies (Bennett et al., 2012; Chan et al., 2014) conducted detailed  
297 measurements of AERs and ventilation rates in thirty seven commercial buildings and  
298 nineteen retail stores, respectively, and provided summary statistics for various building  
299 types (e.g., grocery stores, hardware stores, restaurants, healthcare facilities, and public  
300 assembly spaces) and for varying ventilation conditions (e.g., with doors open/closed,  
301 with and without mechanical ventilation systems in use).

302     As was true for the residential ventilation studies, many of the above-described  
303 studies provide information regarding the characteristics of the buildings studied,  
304 including building volume and/or floor area; however, again, these values are typically  
305 not statistically representative of the full range of non-residential building stock. The  
306 Building Assessment Survey and Evaluation (BASE) Study provides measurements of  
307 building and occupied-space size for 100 randomly selected large office buildings in the  
308 U.S. (Persily and Gorfain, 2004). US EPA (2011) is also a valuable resource for summary  
309 statistics of volume data for buildings with a wide range of uses and sizes (e.g.,  
310 warehouses, shopping malls, schools, and healthcare facilities). As a result of the range of  
311 building uses, commercial building volumes display a large degree of variability, ranging  
312 from 408 to 849,505 m<sup>3</sup> (median [95% CI] = 3,398 [461, 192,554] m<sup>3</sup>) (see SI).

313

### 314 **Inter- and Intra-Zonal Airflows and Mixing**

315     Inter-zonal and intra-zonal airflow and local-scale mixing (i.e., convective and  
316 advective mixing on intra-zonal scales) can be of importance in both residential and non-  
317 residential indoor environments, specifically when considering differences in exposures

318 and  $iF_{in \rightarrow in}$  for building occupants with varying proximities to sources of interest  
319 (Drescher et al., 1995; Nazaroff, 2008). Measurements of inter-zonal and intra-zonal  
320 flows are limited. In addition, these flows vary within and across buildings and depend on  
321 multiple factors including door opening, ventilation conditions, home layout, and  
322 temperature gradients (Klepeis, 2004; McGrath et al., 2014). Thus, selecting a  
323 representative value or sampling from a distribution of measured values when calculating  
324  $iF_{in \rightarrow in}$  is not straightforward. As a result, such flows typically must be modeled for an  
325 exposure scenario of interest.

326 Commonly used models for the estimation of inter-zonal flows include COMIS  
327 (Feustel, 1998) and CONTAM (Walton and Dols, 2010). AER and inter-zonal flows  
328 predicted with CONTAM and/or COMIS have been evaluated against measurements  
329 conducted in more than ten countries and for a variety of building types (Emmerich, 2001  
330 and references therein; Haas et al., 2002; Emmerich et al., 2004). Details regarding the  
331 required inputs and use of these models are available in their respective users' manuals  
332 (Feustel, 1998; Walton and Dols, 2010).

333 Computational fluid dynamics (CFD) has been used to explicitly model airflow and  
334 turbulence on smaller, within-room scales (e.g. Gadgil et al., 2003; Zhang and Chen,  
335 2007; Zhao et al., 2007, 2008). Pragmatically, multi-zone and zonal modeling methods  
336 can be combined by nesting an intra-zonal model within an inter-zonal model (Stewart  
337 and Ren, 2003, 2006; Wang and Chen, 2007), so that a specific room of interest (e.g. the  
338 room with a  $PM_{2.5}$  source) can be divided into several small zones, while other rooms  
339 within the same home/building are treated as larger, well-mixed zones.

340 Alternatively, Bennett and Furtaw (2004) provide an estimate of a room-to-room  
341 air exchange rate distribution (mean =  $3 \text{ h}^{-1}$ , coefficient of variation = 0.30) based on  
342 measurements conducted under varying ventilation conditions within a single house. Du  
343 et al., (2012) characterized overall and season-specific inter-zonal airflows between  
344 living areas and bedrooms in 126 homes in Detroit, MI as the percentage of room-  
345 specific air exchange attributable to air entering from another zone. Along the same lines,  
346 Hellweg et al. (2009) suggest ranges of values for within-zone mixing factors (0.1 to 1.0)  
347 and inter-zonal air exchange rates ( $3$  to  $30 \text{ m}^3/\text{min}$ ). These are examples of midway  
348 approaches between the typical single, well-mixed compartment assumption and more

349 complex approaches based on CFD. Understanding the influence of smaller-scale flows  
350 on  $iF_{in \rightarrow in}$  is an important area of future research, with a rate coefficient representing the  
351 airflow between zones (including the near-person zone and the rest of an indoor  
352 environment) being a resulting metric of interest for use in LCIA.

353

## 354 **Human Exposure Factors**

### 355 **Inhalation Rate**

356 Inhalation intake fraction is directly related to the inhalation rate ( $IR$ ) of the  
357 subjects or population of interest (Table 1, Equation 1). Inhalation rates vary within and  
358 across individuals with multiple factors including age, sex, body weight, and fitness and  
359 activity levels (Figure 2b) (US EPA, 2011). Studies quantifying  $IR$  are largely based on  
360 relationships between oxygen uptake and consumption, metabolism, and energy  
361 expenditure (US EPA, 2011). Using various methods to quantify energy expenditure and  
362 oxygen consumption, multiple studies have measured  $IR$  for broad, representative  
363 populations (e.g., US EPA, 2011 and references therein; Richardson and Stantec, 2013;  
364 Jang et al., 2014a), while others have focused on specific populations of interest (US  
365 EPA, 2011 and references therein). Recommended values of  $IR$  for the general population  
366 categorized by age, gender, and activity level are available for the U.S. (US EPA, 2011),  
367 Canada (Richardson and Stantec, 2013), and Korea (Figure 1b) (Jang et al., 2014a). As is  
368 discussed below, materials are available to allow for the estimation of  $IR$  for populations  
369 for which such measurements have not been conducted. Specific populations of interest  
370 for which  $IR$  studies have been conducted include children, adults and children with  
371 asthma, and pregnant and lactating adult and adolescent women (US EPA, 2011). Such  
372 studies allow for the consideration of  $iF_{in \rightarrow in}$  for susceptible populations or during  
373 specific periods of susceptibility.

374 Inhalation rates are commonly reported as long-term ( $m^3 \text{ day}^{-1}$ ), or short-term ( $m^3$   
375  $\text{min}^{-1}$ ) rates. The latter allow for distinguishing differences in  $IR$  arising from different  
376 levels of activity. When assessing chronic exposures, long-term  $IR$ s can be utilized to  
377 characterize  $iF_{in \rightarrow in}$ ; however, short-term  $IR$ s are needed when considering acute  
378 exposures or exposures associated with a particular activity (i.e., where the emission is  
379 represented by a pulse rather than a continuous term). Short-term  $IR$ s are generally

380 categorized by age, sex, and intensity of activity (e.g., resting/napping, sedentary, and  
381 light, moderate, and high intensity; Adams, 1993; US EPA, 2011). Some studies are as  
382 specific as to provide activity-level-specific, short-term *IRs* for activities conducted in the  
383 indoor environment (US EPA, 2011).

384 In order to use short-term *IRs* in estimates of  $iF_{in \rightarrow in}$ , information regarding the  
385 fraction of time spent at various activity levels is needed. As is discussed in more detail  
386 below, time-activity patterns have been documented for populations from a wide range of  
387 geographic regions (e.g., Klepines et al., 2001; Statistics Canada, 2011; Jang et al., 2014b;  
388 ExpoFacts [<http://expofacts.jrc.ec.europa.eu/>]; Australian Centre for Human Health Risk  
389 Assessment, 2012) (Figure 1b). US EPA (2011) also provides age-specific estimates of  
390 time spent at various levels of activity intensity. The populations for which short-term *IRs*  
391 have been quantified are limited (US EPA, 2011; Jang et al., 2014b). Time-activity  
392 datasets can be combined with available short-term *IR* to predict *IR* distributions for  
393 populations for which such measurements are not available; however, it must be  
394 acknowledged that there is greater uncertainty in these values. Sensitivity analyses may  
395 be valuable for evaluating the influence of this uncertainty in  $iF_{in \rightarrow in}$ . Several exposure  
396 factor reports detail population demographics and physiological conditions, which can  
397 then be used to generate population-specific long- and short-term *IR* distributions from  
398 available measurements (Phillips and Moya, 2014 and references therein). Figure 2b  
399 summarizes the results of key *IR* studies, with detailed data provided in the SI. Overall,  
400 average *IRs* for children, adults, and all age groups for the data gathered here are slightly  
401 higher than that provided in the US EPA EFH (0.97, 1.20, and 1.09  $\text{m}^3 \text{h}^{-1}$  versus 0.81,  
402 1.04, and 0.92  $\text{m}^3 \text{h}^{-1}$ ). Median values (and 95% CI) of the data provided herein for *IRs*  
403 for children, adults, and all age groups are 0.55 (0.17, 3.40), 0.70 (0.26, 4.47), and 0.66  
404 (0.22, 4.23)  $\text{m}^3 \text{h}^{-1}$ , respectively.

#### 405 **Time-Activity Patterns**

406 In addition to serving as a predictor of activity intensity and *IR*, time-activity data  
407 provide valuable information regarding the time spent indoors and in various indoor  
408 locations. For a given subject, the cumulative intake of  $\text{PM}_{2.5}$  is a function of the time  
409 spent by that subject in various microenvironments (e.g., indoor locations) and the  $\text{PM}_{2.5}$   
410 concentration profiles he or she is exposed to in each of those microenvironments. Thus,

411 the characterization of activity patterns is crucial to estimating  $iF_{in \rightarrow in}$ . Studies  
412 characterizing time-activity patterns generally utilize diaries in which a representative  
413 sample of individuals from the general population record their activities over a 24 or 48  
414 hour period. The Center for Time Use Study at the University of Oxford provides a  
415 database of time-activity diary studies for approximately 100 countries in Africa, Asia,  
416 Australia, Europe, North America, and South America (Fisher and Tucker, 2013). Data  
417 from multiple nations are harmonized to allow for comparison across countries. In  
418 addition to references and links for the studies, where available, this database provides  
419 important information such as temporal scale of the study, sampling and data-collection  
420 methodology, sample size, and response rates. Some studies provide broader information  
421 that is useful for long-term exposure studies (e.g., total time spent indoors and time spent  
422 in the residence; Figures 1c and 2c), while others provide more detailed data, including  
423 time spent in various types of indoor environments (e.g., home, school, retail stores, etc.),  
424 time spent in different rooms within a residence, and time spent engaged in activities of  
425 relevance to specific  $PM_{2.5}$  emissions sources (e.g., cleaning, cooking; Schweizer et al.,  
426 2007; Zhao et al., 2009; US EPA, 2011; Jang et al., 2014b; Matz et al., 2014). Such  
427 studies have demonstrated that time-activity patterns vary with age, gender, location of  
428 residence (e.g., urban versus rural), and various demographic and socioeconomic factors.  
429 Time-activity data are generally categorized by these factors and, thus, activity patterns  
430 can be estimated for a population of interest when demographic information is known.  
431 For the U.S., the Consolidated Human Activity Database (CHAD;  
432 <http://www.epa.gov/heasd/chad.html>) brings together data from various studies, resulting  
433 in several thousand daily diaries that can be used in exposure simulation studies. The  
434 advantage of CHAD over other time-use databases is that it is developed specifically for  
435 exposure studies and certain parameters, such as time spent in indoor microenvironments,  
436 can be more easily distinguished. The Stochastic Human Exposure and Dose Simulation  
437 (SHEDS) Model (Burke et al., 2001), for example, simulates a population representative  
438 of the study populations, as well as their activity patterns, by sampling from input  
439 demographic data and CHAD.

440

## 441 **Occupancy**

442 Also key to determining  $iF_{in \rightarrow in}$  is knowledge regarding the total number of  
443 people occupying a space influenced by indoor  $PM_{2.5}$  emissions (Nazaroff, 2008). Higher  
444 occupancy means a larger number of people in proximity to indoor sources and, thus, a  
445 higher population  $iF_{in \rightarrow in}$ . Several studies provide information regarding household size  
446 and composition, which can be utilized to estimate residential occupancy in calculations  
447 of  $iF_{in \rightarrow in}$  (Figure 1f). The U.S. Census Bureau (USCB), for example, provides  
448 information regarding the number and percentage of homes with household sizes ranging  
449 from one person to seven or more people, as well as demographic data describing  
450 households of varying sizes (USCB, 2010; Vespa et al., 2013). Similar information is  
451 available for the European Union (EU) and individual EU nations from Eurostat (2014).  
452 Bongaarts et al. (2001) presented household size and composition for the developing  
453 countries based on surveys conducted in forty-three nations in the 1990s, but notes that  
454 household-size dynamics can change with increased urbanization and industrialization,  
455 trending toward smaller household sizes (i.e., trending toward the nuclear family). That  
456 study provided data regarding household size and the demographic characteristics of  
457 home occupants for four regions: Asia, Latin America, Near East/North Africa, and Sub-  
458 Saharan Africa (see SI). Drivers of within- and between-nation/region variability are  
459 discussed and include level of development (e.g., gross national product) and residence in  
460 urban versus rural areas. The United Nations Demographic Yearbook is a valuable  
461 reference for identifying and locating household occupancy and characteristic data  
462 collected through national censuses (United Nations, 2013). For non-residential  
463 buildings, US EPA (2011) provides distributions of employee numbers for commercial  
464 buildings with a wide range of uses (SI).

465

## 466 **Pollutant-Specific Factors**

467 Concentrations of  $PM_{2.5}$  and related intake in a given indoor environment or zone  
468 within an indoor environment depend on source emissions rates ( $S_{in}$ ), as well as the  
469 removal mechanisms acting on the particles ( $k_{in}$ ) (Table 1, Equation 2). Such removal  
470 mechanisms include the ventilation and transport processes discussed above, particle

471 deposition, filtration in HVAC-system filters and air cleaners, and, in some cases,  
472 chemical transformations/phase changes (Nazaroff, 2004). AERs and ventilation rates  
473 can be estimated using the data discussed above. In the following paragraphs, we discuss  
474 the data and tools available to take into account other factors influencing indoor  $PM_{2.5}$   
475 concentrations and  $iF_{in \rightarrow in}$ , with a primary focus on  $PM_{2.5}$  emitted directly from indoor  
476 sources.

477

### 478 **Indoor $PM_{2.5}$ Emissions**

479 Multiple studies have characterized total  $PM_{2.5}$  emissions from common indoor  
480 sources and activities such as cooking, cleaning, smoking, use of various home and office  
481 appliances, candles, incense, and insect repellent coils (Figure 1e) (e.g., Jetter et al.,  
482 2002; Liu et al., 2003; Lung and Hu, 2003; Guo et al., 2004; He et al., 2004; Lee and  
483 Wang, 2004; Afshari et al., 2005; Olson and Burke, 2006; He et al., 2007; Evans et al.,  
484 2008; See and Balasubramanian, 2011; Torkmahalleh et al., 2012). Substantial variability  
485 in  $PM_{2.5}$  emission rates has been observed within and across sources (Figures 2e – g). For  
486 example, cooking activities can lead to emission rates as high as  $467 \text{ mg min}^{-1}$  (Olson and  
487 Burke, 2006), while emissions from printers were reported to be  $2.8 \times 10^{-4} \text{ mg min}^{-1}$  (He  
488 et al., 2007). He et al. (2004) observed a median emission rate of  $2.7 \text{ mg min}^{-1}$  for frying  
489 food, while Olson and Burke (2006) reported a value of  $6 \text{ mg min}^{-1}$ . Emission rates for  
490 cooking activities vary with the cooking method (e.g., frying, grilling, baking), with the  
491 type of food or oils used in the cooking process (He et al., 2004; Olson and Burke, 2006;  
492 Torkmahalleh et al., 2012), and with stove type and the source of fuel (e.g., biomass,  
493 coal, gas, electric) (SI) (Jetter and Kariher, 2009; Jetter et al., 2012). The importance of a  
494 given source in terms of its contribution to  $iF_{in \rightarrow in}$  varies with a variety of factors  
495 including the indoor environment under consideration, occupant activities, and time of  
496 day or season. For example, in office environments, appliances (e.g., printers, copy  
497 machines) may contribute substantially to indoor  $PM_{2.5}$  concentrations, while cooking, a  
498 major source in residential environments, is unlikely to be of importance. On the other  
499 hand, cleaning products are likely to be significant sources of  $PM_{2.5}$  in both office and  
500 residential environments.

501 The influence of specific PM<sub>2.5</sub> sources on  $iF_{in \rightarrow in}$  also varies geographically.  
502 Solid fuel combustion, for example, is a particularly important source of indoor PM<sub>2.5</sub>  
503 emissions in the developing world. As noted above, the effects of indoor exposures to  
504 solid fuel combustion emissions are a major global environmental health concern  
505 ([www.who.int/indoorair/en](http://www.who.int/indoorair/en)). As a result, controlled laboratory studies and field  
506 measurements have been undertaken to characterize PM<sub>2.5</sub> emissions from various cook  
507 stoves and fuel sources (Habib et al., 2008; Edwards et al., 2014 and references therein).  
508 It is important to note, however, that there is evidence that emissions rates measured in a  
509 laboratory setting differ from those in the field (Edwards et al., 2014) and future efforts  
510 are more focused on characterizing emissions in actual household settings. In addition to  
511 emissions, data regarding the percentage of households using solid fuels and geographic  
512 differences in fuel and stove use are available for estimating  $iF_{in \rightarrow in}$  associated with solid  
513 fuel use (Rehfuess et al., 2006; Bonjour et al., 2013;  
514 [www.who.int/indoorair/health\\_impacts/he\\_database/en](http://www.who.int/indoorair/health_impacts/he_database/en); see SI).

515 As is discussed in more detail below, particle loss rates vary with particle size  
516 and, thus, information regarding the size distributions of particles emitted from specific  
517 sources is useful for calculating  $iF_{in \rightarrow in}$ . Recent work has provided particle size  
518 distributions and/or size-resolved emissions rates for a range of common indoor activities  
519 or sources including cooking (Li and Hopke, 1993; Abt et al., 2000; Long et al., 2000;  
520 Wallace et al., 2004; Hussein et al., 2006; Ogueli et al. 2006; Wallace, 2006), cleaning  
521 (Kleeman et al., 1999; Abt et al., 2000; Long et al., 2000; Ogueli et al. 2006; Gehin et al.,  
522 2008), candles, incense, and aroma lamps (Li and Hopke, 1993; Kleeman et al., 1999;  
523 Hussein et al., 2006; Wallace, 2006; Gehin et al., 2008), smoking (Li and Hopke, 1993;  
524 Nazaroff, 2004; Hussein et al., 2006;), cook-stove use in developing countries and  
525 residential wood combustion (Kleeman et al., 1999; Hays et al., 2003; Armendriz-Arnez  
526 et al., 2010; Shen et al., 2011), fuel-combustion lamps and appliances (Wallace, 2006;  
527 Apple et al., 2010), personal care products/appliances (e.g., hairspray, blow dryer)  
528 (Hussein et al., 2006), and printers (Gehin et al., 2008; Wang et al., 2012; Stephens et al.,  
529 2013).

530

531 **Particle Losses: Deposition**

532 Particle deposition describes all particle losses driven by Brownian diffusion,  
533 gravitational settling, interception, and impaction. Brownian diffusion dominates particle  
534 losses for particles with diameters smaller than about 0.1  $\mu\text{m}$  (ultrafine particles [UFP]),  
535 while for larger particles, interception, impaction, and gravitational settling are the  
536 dominant loss processes (Finlayson-Pitts and Pitts, 2000). As a result, deposition loss rate  
537 coefficients ( $k_{\text{dep}}$  [ $\text{h}^{-1}$ ]) vary with particle size (Ozkaynak et al., 1997; Long et al., 2001;  
538 Riley et al., 2002; Nazaroff, 2004; Hering et al., 2007). Multiple studies have measured  
539 particle-size resolved values of  $k_{\text{dep}}$  or indoor particle decay rates (i.e., the sum of all  
540 loss mechanisms) (e.g., Thatcher and Layton, 1995; Ozkaynak et al., 1997; Abt et al.,  
541 2000; Long et al., 2001; Howard-Reed et al., 2003; Thatcher et al., 2003; Ferro et al.,  
542 2004; He et al., 2005; Sarnat et al., 2006; Meng et al., 2007; Stephens and Siegel, 2013).  
543 These studies have been conducted under a range of sampling and building ventilation  
544 conditions. In addition to their particle size dependence,  $k_{\text{dep}}$  values vary with airflow  
545 conditions and indoor environment surface-to-volume ratios driven by the presence of  
546 furnishings and carpets (Lai, 2002; Thatcher et al., 2002; Howard-Reed et al., 2003;  
547 Nazaroff, 2004). For example, Thatcher et al. (2002) demonstrated that  $k_{\text{dep}}$  could vary  
548 by as much as a factor of 2.6 across different surface-to-volume (i.e., room-furnishing)  
549 scenarios and by as much as a factor of 2.4 with different values of airflow speed. Zhang  
550 et al. (2014) brings attention to the fact that variability in  $k_{\text{dep}}$  to surfaces with varying  
551 orientations (e.g., horizontal versus vertical surfaces) can influence indoor  $\text{PM}_{2.5}$   
552 concentrations and  $iF_{\text{in} \rightarrow \text{in}}$ . That study provides vertical- and horizontal-surface  
553 deposition rates for particles in two broad  $\text{PM}_{2.5}$  size classes.

554 Measurements conducted under various conditions have been combined and fit  
555 with a polynomial regression that describes  $k_{\text{dep}}$  as a function of particle size (Riley et  
556 al., 2002; Nazaroff, 2004). This fit does not take into account variability with ventilation  
557 conditions, room turbulence, surface-to-volume ratios, or room surface orientations;  
558 however, Hodas et al. (2014) found that indoor concentrations of ambient  $\text{PM}_{2.5}$  modeled  
559 using  $k_{\text{dep}}$  values selected with this regression curve were well-correlated with measured  
560 indoor  $\text{PM}_{2.5}$ . El Orch et al. (2014) combined measurement data from multiple studies to

561 predict particle-size-resolved  $k_{\text{dep}}$  values, fit a curve describing  $k_{\text{dep}}$  as a function of  
562 particle diameter, and developed a method to account for increased indoor airflow speeds  
563 when windows are open. In those circumstances, values of  $k_{\text{dep}}$  selected from curves  
564 describing depositional loss rates as a function of particle size (e.g., using Monte Carlo  
565 methods to sample from a particle size distribution) can be multiplied by 1.7 for windows  
566 open a large amount and by 1.23 when windows are open a small amount. In addition, a  
567 small number of studies have quantified deposition or decay rates for total PM<sub>2.5</sub> (Figures  
568 1d, 2d) (Ozkaynak et al., 1997; He et al., 2005; Olson and Burke, 2006; Wallace et al.,  
569 2013). Such information can be useful in circumstances in which particle size distribution  
570 data are not available.

571

### 572 **Particle Losses: Filtration**

573 For homes with HVAC systems, particle losses will also be related to HVAC  
574 system recirculation rates and filter removal efficiencies. Several studies have measured  
575 size-resolved particle filtration efficiencies for various filters commonly found in  
576 residential and commercial HVAC systems (Hanley et al., 1994; Stephens et al., 2011;  
577 Stephens and Siegel, 2012b, 2013; Azimi et al., 2014). Stephens et al. (2011) also studied  
578 recirculation rates in residential and light-commercial HVAC systems. El Orch et al.  
579 (2014) extended this type of analysis to provide size-resolved filtration efficiencies for  
580 five classifications of filters, as well as estimates of the prevalence of these filter  
581 categories in homes. Waring and Siegel (2008) and Stephens and Siegel (2013)  
582 considered the influence of not only filtration, but also losses to heat exchangers and  
583 ducts within HVAC systems. Similarly, Sippola and Nazaroff (2002) reviewed studies of  
584 particle deposition in HVAC system ducts. Such losses are likely to be of particular  
585 importance in schools and commercial buildings. Filtration and fractional loss curves  
586 generated from such measurements have been used in many studies to estimate particle  
587 removal efficiencies as a function of particle size (Riley et al., 2002; Hodas et al., 2012,  
588 2014).

589 HVAC-system air recirculation rates are also key parameters in  
590 characterizing filtration rates. Recommended values for HVAC recirculation rates in  
591 residences (El Orch et al., 2014; Stephens et al., 2011; Stephens, 2015) and in non-

592 residential buildings (Sundell et al., 1994; Weschler et al., 1996; Zuraimi et al., 2007 and  
593 references therein; Fadeyi et al., 2009) are available from a limited number of studies.  
594 Note also that the fraction of air that is recirculated in HVAC systems displays large  
595 spatial variability. Zuraimi et al. (2007), for example, state that 90% of air in conditioned  
596 office buildings in the U.S. and Singapore is recirculated. In some countries (e.g.,  
597 Denmark and Germany), however, all mechanical ventilation systems must be single-pass  
598 (i.e., no air is recirculated). Similarly, HVAC system runtimes directly govern whether or  
599 not a system is in operation and filtering particles at a given point in time, but like  
600 recirculation rates, measurements are limited (Thornburg et al., 2004; Stephens et al.,  
601 2011).

602 The prevalence of central air and heating systems is commonly documented in  
603 housing and energy surveys. US EPA (2011), for example, provides information  
604 regarding the prevalence of central heating and cooling systems in residential and  
605 commercial buildings. It is important to note, however, that the prevalence of central and  
606 recirculating HVAC systems is highly variable both within and across nations and  
607 geographic regions. The importance of collecting data regarding the heating and cooling  
608 systems (or lack thereof) present in households on a global scale has recently been  
609 highlighted (United Nations, 2008).

610

### 611 **Particle Resuspension**

612 The resuspension of particles that have deposited on surfaces in indoor  
613 environments can also influence indoor  $PM_{2.5}$  concentrations and  $iF_{in \rightarrow in}$  (Ferro et al.,  
614 2004; Liroy, 2006, and references therein). While typically considered to be an important  
615 determinant of exposures to particles larger than  $PM_{2.5}$ , Ferro et al. (2004) found that  
616 resuspension can result in the equivalent of a  $PM_{2.5}$  source strength ranging from 0.03 to  
617  $0.5 \text{ mg min}^{-1}$ . The prevalence and magnitude of resuspension are dependent on the  
618 activities of building occupants, specifically cleaning (e.g., dusting, vacuuming) and  
619 active movement (e.g., walking, dancing, playing) (Ferro et al., 2004; Liroy, 2006). Thus,  
620 the influence of resuspension on  $iF_{in \rightarrow in}$  is expected to vary temporally and spatially.

621

## 622 Transformation: Phase Changes and Indoor Chemistry

623 Phase changes and chemical transformation can lead to both increases and  
624 decreases in indoor  $PM_{2.5}$  concentrations. The partitioning of semivolatile organic  
625 compounds (SVOCs) between the gas and particle phases, for example, is dependent on  
626 indoor air temperature and the availability of particle-phase organic matter for sorption  
627 (Pankow, 1994). Thus, the extent to which a given indoor source of SVOCs contributes  
628 to  $iF_{in \rightarrow in}$  will depend on the fraction of emissions from that source found in the particle  
629 phase, which, in turn, is dependent on the conditions of the indoor environment (i.e.,  
630 temperature, organic  $PM_{2.5}$  concentrations). Examples of indoor sources of SVOCs that  
631 display this behavior include environmental tobacco smoke, flame retardants, plasticizers,  
632 and pesticides (Liang and Pankow, 1996; Gurunathan et al., 1998; Bennett and Furtaw,  
633 2004; Lioy, 2006; Weschler and Nazaroff, 2008 and references therein). Estimating shifts  
634 in partitioning requires knowledge regarding volatility and partitioning coefficients of  
635 chemical species commonly found indoors, as well as the development of simplified  
636 models to predict SVOC partitioning in indoor air. This is an active area of research  
637 (Weschler and Nazaroff, 2008, 2010; Weschler, 2011; Hodas and Turpin, 2014; Liu et  
638 al., 2014); however, further work is needed to characterize semi-volatile species of indoor  
639 origin before this process can be consistently incorporated into estimates of  $iF_{in \rightarrow in}$ .

640 The formation of secondary organic aerosols (SOA) from reactions between  
641 oxidants and gas-phase compounds emitted indoors can also substantially influence  $PM_{2.5}$   
642 concentrations and  $iF_{in \rightarrow in}$  (Weschler and Shields, 1999; Long et al., 2000; Wainman et  
643 al., 2000; Weschler, 2006, 2011; Waring and Siegel, 2010, 2013; Waring et al., 2011;  
644 Waring, 2014). Most work in this area has focused on reactions between terpenoids  
645 emitted from air fresheners, cleaning products, and scented personal care products and  
646 ozone (Nazaroff and Weschler, 2004; Singer et al., 2006; Weschler, 2006; Waring et al.,  
647 2011; Weschler, 2011; Waring and Siegel, 2010, 2013). Such studies have demonstrated  
648 that indoor SOA formation varies with multiple factors including the chemicals present in  
649 indoor air, relative humidity, time of day, season, indoor ventilation conditions and  
650 HVAC system use, indoor surface area and surface materials, and geographic location  
651 (Waring and Siegel, 2010; Weschler, 2011; Waring and Siegel, 2013; Youseffi and  
652 Waring, 2014). Indoor sources of ozone include photocopiers, laser printers, and

653 electrostatic air cleaners; however, the majority of ozone present indoors is the result of  
654 transport from the outdoor environment (Weschler, 2000). SOA generated through  
655 reactions between VOCs of indoor origin and ozone of outdoor origin illustrates one  
656 mechanism through which interactions between indoor- and outdoor-generated pollutants  
657 can influence the intake of PM<sub>2.5</sub> attributable, at least in part, to indoor sources. This  
658 complication of separating outdoor- and indoor-source contributions to the intake of  
659 PM<sub>2.5</sub> in indoor environments is discussed further in the next section.

660

### 661 **Influence of outdoor-generated pollutants on cumulative indoor intake of PM<sub>2.5</sub>**

662 The cumulative intake of PM<sub>2.5</sub> that occurs indoors is influenced by both indoor  
663 and outdoor PM<sub>2.5</sub> sources (Table 1, Equation 2) and depends on (1) primary emissions of  
664 PM<sub>2.5</sub> from indoor sources, (2) the formation of secondary PM<sub>2.5</sub> from precursors of  
665 indoor origin, (3) the transport of outdoor-generated PM<sub>2.5</sub> into the indoor environment,  
666 and (4) interactions between pollutants of indoor and outdoor origin. This latter factor  
667 includes SOA formation through reactions of indoor-emitted volatile organic compounds  
668 (VOCs) and outdoor-generated oxidants, as well as the partitioning of outdoor-generated  
669 gas-phase SVOCs to particulate matter of indoor origin and/or the partitioning of gas-  
670 phase SVOCs emitted by indoor sources to outdoor-generated particles that have  
671 infiltrated indoors. Prior sections focused on factors (1) and (2). Below, we briefly  
672 explore the current state of knowledge regarding interactions between pollutants of  
673 outdoor and indoor origin and the influence of outdoor PM<sub>2.5</sub> sources on cumulative  
674 indoor intake.

675 Outdoor-generated PM<sub>2.5</sub> (ambient PM<sub>2.5</sub>) that penetrates into and persists in the  
676 indoor environment is a major source of indoor PM<sub>2.5</sub>. Multiple studies have quantified  
677 the fraction of ambient PM<sub>2.5</sub> found in indoor air ( $f_{\text{out} \rightarrow \text{in}}$ ) (Chen and Zhao, 2011 and  
678 references therein; Diapouli et al., 2013 and references therein). These studies have  
679 demonstrated that there is substantial between- and within-home variability in  $f_{\text{out} \rightarrow \text{in}}$   
680 (Ozkaynak et al., 1997; Ott et al., 2000; Meng et al., 2005; Weisel et al., 2005; Polidori et  
681 al., 2006; Allen et al., 2012; MacNeil et al., 2012; Hänninen et al., 2013; Kearny et al.,  
682 2014), illustrating the difficulty in utilizing measured values of  $f_{\text{out} \rightarrow \text{in}}$  to estimate  
683 contributions of ambient PM<sub>2.5</sub> to cumulative indoor intake. In addition, most studies are

684 limited in their geographic and temporal scope and cannot be generalized to a broader  
685 population of homes. Two exceptions are the studies conducted by Hänninen et al. (2011)  
686 and El Orch et al. (2014). Estimates of  $f_{\text{out} \rightarrow \text{in}}$  for homes in ten European countries  
687 sampled as part of six studies were aggregated and summary statistics of  $f_{\text{out} \rightarrow \text{in}}$  were  
688 provided for various climatic regions of Europe (Northern, Central, and Southern Europe)  
689 and by season (Hänninen et al. 2011). El Orch et al. (2014) conducted a detailed  
690 modeling study in which particle-size-resolved distributions of  $f_{\text{out} \rightarrow \text{in}}$  for single-family  
691 homes in the U.S. were calculated.

692 For a given exposure scenario,  $f_{\text{out} \rightarrow \text{in}}$  can also be calculated using a mass  
693 balance model in which indoor ambient  $\text{PM}_{2.5}$  concentrations are described as function of  
694 AER, the efficiency with which particles penetrate across the building envelope, particle  
695 deposition, filtration in HVAC-system filters and air cleaners, and, for semivolatile  
696 species, phase changes in indoor air (e.g., Hering et al., 2007; Hodas et al., 2012, 2014).  
697 Similarly, these physical and chemical processes also govern the outdoor transport of  
698 indoor-generated  $\text{PM}_{2.5}$  and, thus,  $iF_{\text{in} \rightarrow \text{out}}$  and  $iF_{\text{in}, \text{total}}$  (see Table 1). While the  
699 contributions of  $iF_{\text{in} \rightarrow \text{out}}$  to  $iF_{\text{in}, \text{total}}$  are typically negligible compared to that of  $iF_{\text{in} \rightarrow \text{in}}$ ,  
700 there is evidence that solid fuel combustion in household cook stoves can contribute  
701 substantially to ambient  $\text{PM}_{2.5}$  concentrations in some regions (e.g., India, China) (Chafe  
702 et al., 2014).

703 The data given above provide inputs to predict AER, deposition, and filtration.  
704 Chen and Zhao (2011) provide a detailed review of penetration efficiency measurements  
705 and modeling strategies. While the focus of previous work has mostly been on the  
706 penetration of ambient  $\text{PM}_{2.5}$  into the indoor environments, results of these studies can  
707 also be used to estimate penetration of indoor-generated particles between separated  
708 indoor zones/rooms. Tools are also available to account for evaporative losses of  
709 ammonium nitrate (Lunden et al., 2003; Hering et al., 2007), and the development of  
710 modeling tools to predict the gas-particle partitioning of SVOCs (of both indoor and  
711 outdoor origin) in indoor air is an active area of ongoing research (Weschler and  
712 Nazaroff, 2008, 2010; Weschler, 2011; Hodas and Turpin, 2014; Liu et al., 2014).

713 Because the availability of organic matter for sorption influences the gas-particle  
714 partitioning of SVOCs, there is the potential for the indoor formation of particles that are

715 only present due to interactions between SVOCs of indoor and outdoor origin. For  
716 example, gas-phase SVOCs emitted indoors can sorb to indoor particulate matter of  
717 outdoor origin that has penetrated into the home (Lioy, 2006; Weschler and Nazaroff,  
718 2008). Similarly, incoming organics from outdoors can shift from the gas phase toward  
719 the particle phase as they sorb to particulate organic matter emitted by indoor sources  
720 (Naumova et al., 2003; Polidori et al., 2006; Weschler and Nazaroff, 2008; Shi and Zhao,  
721 2012; Hodas and Turpin, 2014). The result is the formation of PM<sub>2.5</sub> that is in part, but  
722 not fully, attributable to indoor sources. Such interactions between pollutants of indoor  
723 and outdoor origin highlight the difficulty in fully separating the contributions of indoor  
724 and outdoor PM<sub>2.5</sub> sources to the intake of PM<sub>2.5</sub>.

725 The formation of SOA from reactions between indoor-generated VOCs and  
726 oxidants (e.g., ozone) of outdoor origin is another example of the ways in which outdoor-  
727 generated pollutants can influence the intake of PM<sub>2.5</sub> associated with indoor sources.  
728 Contributions of secondary particulate matter derived from well-characterized inorganic  
729 systems to outdoor *iF* have previously been accounted for using chemical transport  
730 models (e.g., Levy et al., 2003; Greco et al., 2007). The data and modeling tools available  
731 to include indoor secondary particulate matter (specifically, SOA) formation in estimates  
732 of indoor PM<sub>2.5</sub> exposures continue to improve. Waring (2014) presented a mechanistic  
733 model to calculate time-averaged indoor SOA concentrations formed as a result of the  
734 oxidation of reactive organic gases by ozone and the hydroxyl radical. Distributions of  
735 model inputs for 66 reactive organic gases relevant to the indoor environment (Weisel et  
736 al., 2005; Turpin et al., 2007) are provided in that work. In addition, a linear regression  
737 model describing SOA concentrations as a function of AER, indoor concentrations of  
738 outdoor-generated ozone and organic aerosols, indoor organic aerosol emission rates,  
739 particle and ozone deposition rates, temperature, and emission rates of reactive organic  
740 gases described the majority of variability in SOA concentrations calculated using the  
741 more complex mechanistic SOA model described above ( $R^2 = 0.88$ ; Waring, 2014). Ji  
742 and Zhao (2015) demonstrated that the extent to which indoor SOA formation impacts  
743 indoor concentrations of PM<sub>2.5</sub> varies geographically, with SOA comprising 6 to 30% of  
744 indoor PM<sub>2.5</sub> mass for the U.S. homes included in the Waring (2014) study, but less than  
745 3% of PM<sub>2.5</sub> mass for homes in Beijing. Accounting for SOA formation indoors is an

746 active and quickly advancing area of research and is crucial for ensuring that the full  
747 impact of specific products, activities, and processes can be taken into account in LCIA.

748

## 749 **Discussion**

### 750 **Applications in Life Cycle Impact Assessment**

751 The data provided in this review constitute a first step in addressing key questions  
752 and current challenges previously identified for the incorporation of health effects  
753 associated with indoor PM<sub>2.5</sub> emissions into LCIA (Hellweg et al., 2009; Fantke et al.,  
754 2015; Humbert et al., 2015). Specifically, this review allows for the characterization of a  
755 range of exposure-scenario archetypes, both in terms of indoor setting (e.g., residence,  
756 office) and in geographic location, aids in the identification of the major factors  
757 influencing  $iF_{in \rightarrow in}$  and potential spatial and temporal variability in the importance of  
758 these key factors, and allows for the assessment of the level of detail and scope needed  
759 when developing exposure-scenario archetypes for use in LCIA.

760 In an ongoing effort, the UNEP-SETAC task force on PM<sub>2.5</sub> health effects will  
761 utilize the data provided in this review to build a quantitative assessment framework for  
762 consistently combining and evaluating indoor and outdoor intake fractions from PM<sub>2.5</sub>  
763 sources for application in LCIA. Complementary work is currently focusing on (1)  
764 conducting a quantitative assessment of potential variability in  $iF_{in \rightarrow in}$  (e.g., across  
765 exposure scenarios and geographic regions), as well as the sensitivity of calculations of  
766  $iF_{in \rightarrow in}$  to heterogeneity in the input parameters reviewed here, (2) the evaluation of state-  
767 of-the-art modeling tools available to predict indoor and outdoor intake fractions in the  
768 context of suitability for use in LCIA, and (3) the consistent incorporation of various  
769 shapes of ERFs (Fantke et al., 2015). Together, these efforts will aid in the development  
770 of a standardized methodology by which to estimate exposures and will contribute to the  
771 effort to include PM<sub>2.5</sub>-related health effects in LCIA.

772 Key to assessing PM<sub>2.5</sub>-related health effects over the life cycle of products is the  
773 ability to evaluate the range of potential human exposure associated with a given particle  
774 emissions source. Previous work has illustrated the potential magnitude of spatial and  
775 temporal variability in  $iF_{in \rightarrow in}$ . Humbert et al. (2011), for example, estimates that typical  
776 values of  $iF_{in \rightarrow in}$  range between approximately  $10^{-3}$  and  $10^{-2}$  kg intake at the population

777 scale per kg emitted indoors. Klepeis and Nazaroff (2006) found that  $iF_{in \rightarrow in}$  for  
778 environmental tobacco smoke varied between  $6.6 \times 10^{-4}$  and  $2.6 \times 10^{-3}$  kg intake per kg  
779 emitted within a single simulated home depending on multiple factors including home  
780 ventilation conditions and occupant activity patterns. Thus, while a single recommended  
781 value meant to characterize a needed modelling parameter is valuable for providing an  
782 estimate of the magnitude of  $iF_{in \rightarrow in}$  (e.g., a single AER value meant to represent typical  
783 housing the U.S.), distributions or ranges describing these input parameters are crucial.  
784 Such distributions allow for the evaluation of the central tendencies of  $iF_{in \rightarrow in}$ , as well as  
785 the extremes, thereby acknowledging the variability in population exposure patterns,  
786 housing aspects, and indoor air chemistry. By aggregating the results of multiple studies,  
787 the present review provides a broader picture of the range of potential values for a given  
788 parameter influencing indoor concentrations of  $PM_{2.5}$  and allows for the consideration of  
789 a range of archetypal indoor environments. It is important to note that these values vary  
790 temporally and spatially with multiple factors, as discussed in the individual sections  
791 above, and parameters are not available to describe all exposure scenarios and geographic  
792 regions. Thus, understanding the full range of input parameters also allows for the  
793 consideration of uncertainty in  $iF_{in \rightarrow in}$  for  $PM_{2.5}$ .

794 Depending on the design of the selected modelling framework, not all of the  
795 factors potentially contributing to variability in  $iF_{in \rightarrow in}$  will necessarily be considered in  
796 LCIA. For example, Hellweg et al. (2009) suggested that the representation of the indoor  
797 environment as a single, well-mixed compartment provides the most effective way to  
798 incorporate indoor  $PM_{2.5}$  exposures into LCIA. On the other hand, in regards to assessing  
799 exposure to individual VOCs from cleaning products, Earnest and Corsi (2013) propose  
800 the use of a two-zone model in which the near-person/near-source region and the rest of  
801 the indoor environment are treated as discrete zones. LCIA often follows approaches  
802 based on archetypes to account for differences in exposure scenarios or geographic  
803 regions. Thus, the parameters that will be of the greatest importance are those that  
804 account for geographic variability in more general housing and building characteristics  
805 (e.g., volume, whole-building air exchange and ventilation), indoor-environment  
806 occupancy, and the prevalence of specific indoor sources (e.g., cooking and heating  
807 appliances). Parameters that provide a higher level of detail (e.g., activity-specific

808 breathing rates, local-scale flows), however, will be valuable to higher tier assessments of  
809 indoor air quality and epidemiologic studies that aim to characterize indoor PM<sub>2.5</sub>  
810 exposures for specific conditions in a well-characterized environment.

811

### 812 **Remaining Limitations and Recommendations for Future Research**

813 One contributor to limitations in the availability and scope of data like those  
814 reviewed here is the fact that the studies carried out to collect the data are expensive and  
815 work intensive. As a result, they tend to be carried out in infrequent, intensive campaigns.  
816 As noted above, for example, many AER studies are not representative of the full range  
817 of housing stock, even for the nations or cities in which they were carried out. Values are  
818 more limited or non-existent in some developing countries and are biased towards U.S.  
819 and European studies. We suggest that there is a need for studies on AER in developing  
820 countries, particularly in rural regions where biomass is used for cooking in homes.

821 Another issue constraining the representativeness of the data is the potential for  
822 changes with time. While some values are not expected to vary temporally (i.e., *IR*,  
823 although the activity levels driving them may change), others change on timescales faster  
824 than the studies characterizing them are carried out. Bongaarts et al. (2001), for example,  
825 noted the tendency for household size to converge towards the nuclear family in rapidly  
826 industrializing and urbanizing regions. Similarly, there is the potential for changes in  
827 human activity patterns with increased access to media, suggesting a need for updated  
828 human activity pattern data. Housing construction practices change with advancing  
829 technology and materials development, as well as with recent pushes toward energy  
830 efficiency. Urban growth (e.g., Seto and Fragikas, 2005; Xiao et al., 2006; Schneider and  
831 Woodcock, 2008) may make the lack of data characterizing AERs in apartments and  
832 multi-family residences a major issue in both developing and developed countries. New  
833 techniques utilizing 3D imaging sensors to evaluate building/room size and leakage  
834 characteristics show promise in increasing data availability for leaky buildings (e.g., in  
835 developing countries), airtight, energy efficient buildings, and multifamily residences  
836 (Gong and Caldas, 2008) and should be a consideration in future work in this area.  
837 Finally, while the principles driving pollutant dynamics will not change with time,  
838 emission rates, particle size distributions, and particle composition may change with

839 technology. Cynthia et al. (2007), for example, reported a 35% decrease in  $PM_{2.5}$   
840 exposures with the introduction of a higher-efficiency cook stove in an intervention study  
841 in rural Mexico. As a result of these ever-changing factors, a continued effort to  
842 undertake such studies and to expand their temporal and spatial scope is key to ensuring  
843 that the impacts associated with specific products and emission sources can be fully  
844 assessed in the context of LCIA.

845     ■ We also recommend that future efforts focus on a number of key research areas.  
846 First, there is a need for a more widespread and detailed characterization of inter- and  
847 intra-zonal airflows and the factors that influence them for a range of residence types,  
848 commercial buildings, and occupational settings to derive useful information for higher  
849 tier assessments of indoor air quality. Such characterizations would be useful in  
850 addressing proximity-to-source issues. Of particular importance may be the development  
851 of a set of archetypal building layouts that describe a range of building types, so that  
852 these highly variable flows can be modelled for a given exposure scenario with tools such  
853 as COMIS and CONTAM. For applications in LCIA, a simple two-zone model might be  
854 more suitable as more complex approaches might lack data and consistency across indoor  
855 and outdoor emission situations. As noted above, there are large geographic differences  
856 in the heating and cooling systems present in households and other indoor environments  
857 on a global scale. Documenting these differences and the related impacts on indoor  
858 particle dynamics is an important area of future work. Finally, there is a need for more  
859 research aimed at obtaining a thorough understanding of interactions between indoor- and  
860 outdoor-generated pollutants and the formation of SOA in indoor air. Key to this is the  
861 development of accurate simplified models that can easily be applied in LCIA. The  
862 regression model developed by Waring (2014) to predict indoor SOA formation based on  
863 a small number of key parameters provides an example of the type of modeling tools that  
864 will advance predictions of  $iF_{in \rightarrow in}$  for  $PM_{2.5}$  in this context.

865

## 866 CONCLUSIONS

867     The present paper reviews and compiles the results of studies exploring the main  
868 factors influencing indoor  $PM_{2.5}$  concentrations and associated  $iF_{in \rightarrow in}$ , with an emphasis  
869 on primary indoor  $PM_{2.5}$  emissions. Specifically, we focus on factors related to building

870 characteristics, occupant characteristics and behaviors, and pollutant properties and  
871 dynamics. The key studies and data sources discussed herein comprise a tool kit of  
872 exposure-modelling parameters that can be used to estimate the central tendencies and  
873 potential ranges of  $iF_{in \rightarrow in}$ . A follow-up effort will utilize the data provided in the present  
874 review to build a framework to consistently integrate indoor and outdoor exposures to  
875  $PM_{2.5}$  emitted by indoor and outdoor sources. Combined, the present review and the  
876 follow-up work contribute to the effort to consistently include  $PM_{2.5}$ -derived health  
877 effects in LCIA. Continued efforts to characterize the factors influencing indoor  $PM_{2.5}$   
878 concentrations will ensure that impacts associated with specific products and emission  
879 sources can be fully assessed in LCIA and other comparative human exposure and impact  
880 assessment frameworks.

881

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<b>Emission</b> [kg <sub>emitted</sub> /h]	<b>Indoor intake</b> [kg <sub>intake</sub> /h]	<b>Outdoor intake</b> [kg <sub>intake</sub> /h]	<b>Total intake</b> [kg <sub>intake</sub> /h]	<b>Intake fraction</b> [kg <sub>intake</sub> /h per kg <sub>emitted</sub> /h]
Indoor (PM <sub>2.5</sub> or precursor) emissions  $S_{in}$	Indoor intake due to indoor emissions  $\frac{iF_{in \rightarrow in}}{\left(\frac{IR_{in} \times n_{in}}{V_{in} \times k_{in}}\right)} \times S_{in}$	Outdoor intake due to indoor emissions  $\frac{iF_{in \rightarrow out}}{\left(iF_{out, total} \times f_{in \rightarrow out}\right)} \times S_{in}$	Total intake due to indoor emissions  $iF_{in, total} \times S_{in}$	Total intake due to indoor emissions per unit of indoor emissions  $iF_{in, total} = \frac{iF_{in \rightarrow in}}{\left(\frac{IR_{in} \times n_{in}}{V_{in} \times k_{in}}\right) + \frac{iF_{in \rightarrow out}}{\left(iF_{out, total} \times f_{in \rightarrow out}\right)}} \quad (\text{eq. 1})$

Outdoor (PM <sub>2.5</sub> or precursor) emissions $S_{out}$	Indoor intake due to outdoor emissions $\frac{iF_{out \rightarrow in}}{(iF_{in, total} \times f_{out \rightarrow in})} \times S_{out}$	Outdoor intake due to outdoor emissions $\frac{iF_{out \rightarrow out}}{\left(\frac{IR_{out} \times n_{out}}{V_{out} \times k_{out}}\right)} \times S_{out}$	Total intake due to outdoor emission $iF_{out, total} \times S_{out}$	Total intake due to outdoor emission per unit of outdoor emissions $iF_{out, total} = \frac{iF_{out \rightarrow out}}{\left(\frac{IR_{out} \times n_{out}}{V_{out} \times k_{out}}\right)} + \frac{iF_{out \rightarrow in}}{(iF_{in, total} \times f_{out \rightarrow in})}$
	Cumulative indoor intake due to indoor and outdoor emissions $\frac{iF_{in \rightarrow in}}{\left(\frac{IR_{in} \times n_{in}}{V_{in} \times k_{in}}\right)} \times S_{in} + \frac{iF_{out \rightarrow in}}{(iF_{in, total} \times f_{out \rightarrow in})} \times S_{out}$ (eq. 2)	Cumulative outdoor intake due to indoor and outdoor emissions $iF_{out \rightarrow out} \times S_{out} + iF_{in \rightarrow out} \times S_{in}$	Cumulative intake due to indoor and outdoor emissions $iF_{in, total} \times S_{in} + iF_{out, total} \times S_{out}$	

<b>Emission</b> [kg <sub>emitted</sub> /h]	<b>Indoor intake</b> [kg <sub>intake</sub> /h]	<b>Outdoor intake</b> [kg <sub>intake</sub> /h]	<b>Total intake</b> [kg <sub>intake</sub> /h]	<b>Intake fraction</b> [kg <sub>intake</sub> /h per kg <sub>emitted</sub> /h]
Indoor (PM <sub>2.5</sub> or precursor) emissions  S <sub>in</sub>	Indoor intake due to indoor emissions  $\overbrace{\left(\frac{iR_{in} \times n_{in}}{V_{in} \times k_{in}}\right)}^{iF_{in \rightarrow in}} \times S_{in}$	Outdoor intake due to indoor emissions  $\overbrace{\left(iF_{out, total} \times f_{in \rightarrow out}\right)}^{iF_{in \rightarrow out}} \times S_{in}$	Total intake due to indoor emissions  iF <sub>in, total</sub> × S <sub>in</sub>	Total intake due to indoor emissions per unit of indoor emissions  $iF_{in, total} = \overbrace{\left(\frac{iR_{in} \times n_{in}}{V_{in} \times k_{in}}\right)}^{iF_{in \rightarrow in}} + \overbrace{\left(iF_{out, total} \times f_{in \rightarrow out}\right)}^{iF_{in \rightarrow out}}$ (eq. 1)
Outdoor (PM <sub>2.5</sub> or precursor) emissions  S <sub>out</sub>	Indoor intake due to outdoor emissions  $\overbrace{\left(iF_{in, total} \times f_{out \rightarrow in}\right)}^{iF_{out \rightarrow in}} \times S_{out}$	Outdoor intake due to outdoor emissions  $\overbrace{\left(\frac{iR_{out} \times n_{out}}{V_{out} \times k_{out}}\right)}^{iF_{out \rightarrow out}} \times S_{out}$	Total intake due to outdoor emission  iF <sub>out, total</sub> × S <sub>out</sub>	Total intake due to outdoor emission per unit of outdoor emissions  $iF_{out, total} = \overbrace{\left(\frac{iR_{out} \times n_{out}}{V_{out} \times k_{out}}\right)}^{iF_{out \rightarrow out}} + \overbrace{\left(iF_{in, total} \times f_{out \rightarrow in}\right)}^{iF_{out \rightarrow in}}$
	Cumulative indoor intake due to indoor and outdoor emissions  $\overbrace{\left(\frac{iR_{in} \times n_{in}}{V_{in} \times k_{in}}\right)}^{iF_{in \rightarrow in}} \times S_{in} + \overbrace{\left(iF_{in, total} \times f_{out \rightarrow in}\right)}^{iF_{out \rightarrow in}} \times S_{out}$	Cumulative outdoor intake due to indoor and outdoor emissions  iF <sub>out → out</sub> × S <sub>out</sub> +	Cumulative intake due to indoor and outdoor emissions	

	(eq. 2)	$iF_{in \rightarrow out} \times S_{in}$	$iF_{in,total} \times S_{in}$ $+ iF_{out,total}$ $\times S_{out}$
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<b>Emission</b> [kg <sub>emitted</sub> /h]	<b>Indoor intake</b> [kg <sub>intake</sub> /h]	<b>Outdoor intake</b> [kg <sub>intake</sub> /h]	<b>Total intake</b> [kg <sub>intake</sub> /h]	<b>Intake fraction</b> [kg <sub>intake</sub> /h per kg <sub>emitted</sub> /h]
Indoor (PM <sub>2.5</sub> or precursor) emissions  S <sub>in</sub>	<b>Indoor intake due to indoor emissions</b>  $\overbrace{\left(\frac{iF_{in \rightarrow in}}{\left(\frac{IR_{in} \times n_{in}}{V_{in} \times k_{in}}\right)}\right)} \times S_{in}$	Outdoor intake due to indoor emissions  $\overbrace{\left(iF_{in \rightarrow out}\right)} \times S_{in}$	Total intake due to indoor emissions  $iF_{in,total} \times S_{in}$	Total intake due to indoor emissions per unit of indoor emissions  $iF_{in,total} = \overbrace{\left(\frac{iF_{in \rightarrow in}}{\left(\frac{IR_{in} \times n_{in}}{V_{in} \times k_{in}}\right)}\right)} + \overbrace{\left(iF_{out,total} \times f_{in \rightarrow out}\right)}$  (eq. 1)
Outdoor (PM <sub>2.5</sub> or precursor) emissions  S <sub>out</sub>	Indoor intake due to outdoor emissions  $\overbrace{\left(iF_{in,total} \times f_{out \rightarrow in}\right)} \times S_{out}$	Outdoor intake due to outdoor emissions  $\overbrace{\left(\frac{iF_{out \rightarrow out}}{\left(\frac{IR_{out} \times n_{out}}{V_{out} \times k_{out}}\right)}\right)} \times S_{out}$	Total intake due to outdoor emission  $iF_{out,total} \times S_{out}$	Total intake due to outdoor emission per unit of outdoor emissions  $iF_{out,total} = \overbrace{\left(\frac{iF_{out \rightarrow out}}{\left(\frac{IR_{out} \times n_{out}}{V_{out} \times k_{out}}\right)}\right)} + \overbrace{\left(iF_{in,total} \times f_{out \rightarrow in}\right)}$

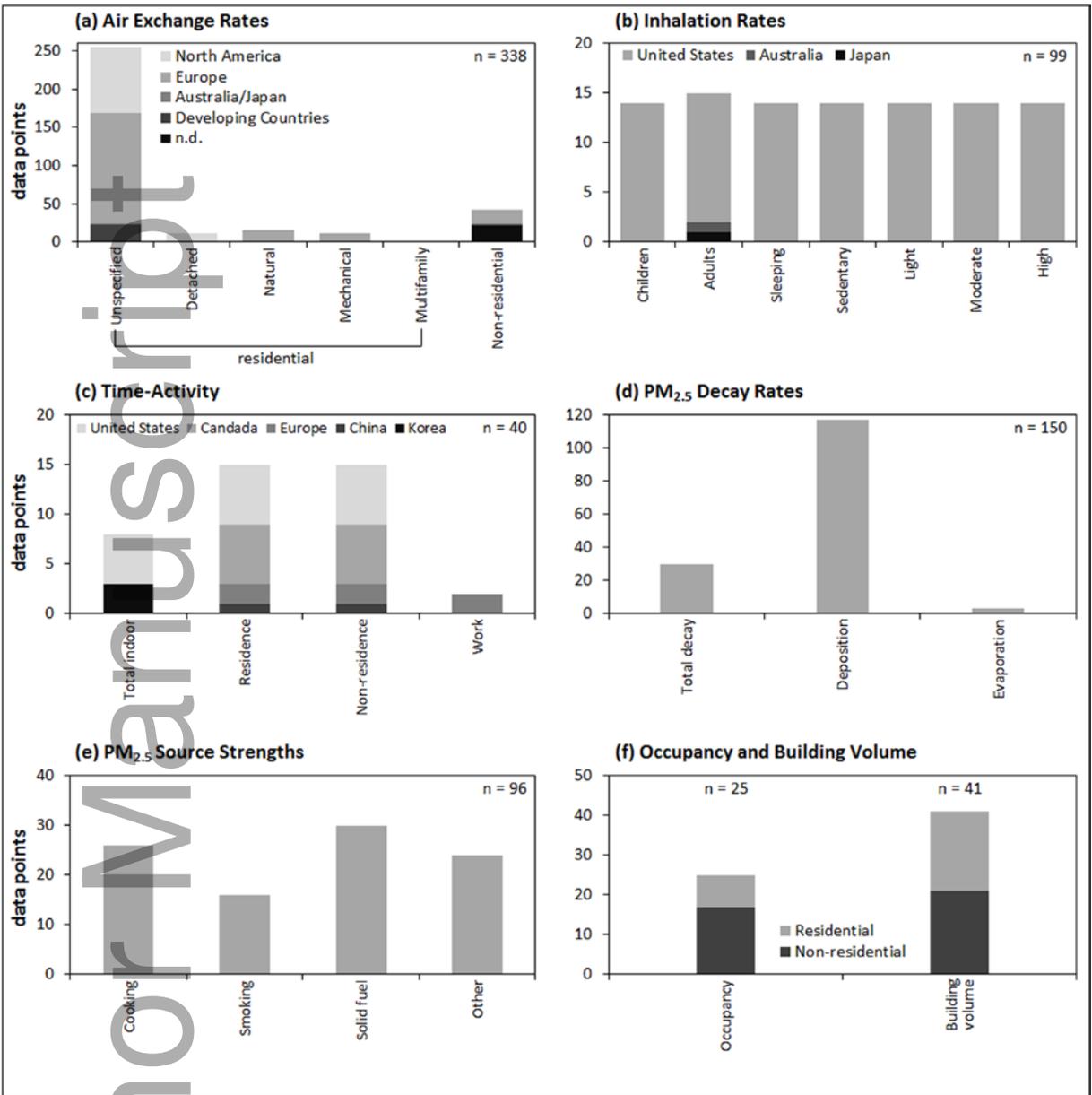
Cumulative indoor intake due to indoor and outdoor emissions	Cumulative outdoor intake due to indoor and outdoor emissions	Cumulative intake due to indoor and outdoor emissions
$\frac{iF_{in \rightarrow in}}{\left(\frac{IR_{in} \times n_{in}}{V_{in} \times k_{in}}\right)} \times S_{in} + \frac{iF_{out \rightarrow in}}{(iF_{in, total} \times f_{out \rightarrow in})} \times S_{out}$ <p style="text-align: center;">(eq. 2)</p>	$iF_{out \rightarrow out} \times S_{out} + iF_{in \rightarrow out} \times S_{in}$	$iF_{in, total} \times S_{in} + iF_{out, total} \times S_{out}$

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1522 Table 1. Matrix illustrating the contributions of PM<sub>2.5</sub> derived from indoor and outdoor sources to indoor intake, outdoor intake, total  
 1523 intake, and intake fraction of PM<sub>2.5</sub>. Aspects discussed in this paper are highlighted in grey and specific areas of focus are in red.

1524 Abbreviations:  $S_{in}$  or  $S_{out}$ , indoor or outdoor PM<sub>2.5</sub> source emission rate;  $iF_{in \rightarrow in}$ , fraction of PM<sub>2.5</sub> emitted/formed indoors that is  
 1525 taken in via inhalation indoors;  $iF_{in \rightarrow out}$ , fraction of PM<sub>2.5</sub> emitted/formed indoors that is transported outdoors and taken in via  
 1526 inhalation outdoors;  $iF_{out \rightarrow out}$ , fraction of PM<sub>2.5</sub> emitted/formed outdoors that is taken in via inhalation outdoors;  $iF_{out \rightarrow in}$ , fraction of  
 1527 PM<sub>2.5</sub> emitted/formed outdoors that is transported indoors and taken in via inhalation indoors;  $IR_{in}$  or  $IR_{out}$ , individual inhalation rate  
 1528 indoors or outdoors [ $m^3_{inhalated}/h$ ];  $n_{in}$  or  $n_{out}$ , number of exposed persons in an indoor or outdoor location;  $V_{in}$  or  $V_{out}$ , volume of  
 1529 indoor or outdoor location [ $m^3$ ];  $k_{in}$  or  $k_{out}$ , total indoor or outdoor particle removal rate attributable to all loss mechanisms (e.g., air  
 1530 exchange, particle deposition) [ $h^{-1}$ ];  $iF_{in, total}$ , total indoor inhalation intake fraction;  $iF_{out, total}$ , total outdoor inhalation intake fraction;  
 1531  $f_{in \rightarrow out}$ , fraction of indoor-generated (emitted/formed) PM<sub>2.5</sub> transported outdoors,  $f_{out \rightarrow in}$ , fraction of outdoor-generated  
 1532 (emitted/formed) PM<sub>2.5</sub> transported indoors. Note that there is no cumulative intake fraction.



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1535 **Figure 1.** Frequency plot illustrating the number of data points (i.e., measured or modeled value

1536 or summary statistic from a distribution of measurements describing the parameter of interest)

1537 gathered from the literature for the primary factors influencing indoor inhalation intake fraction

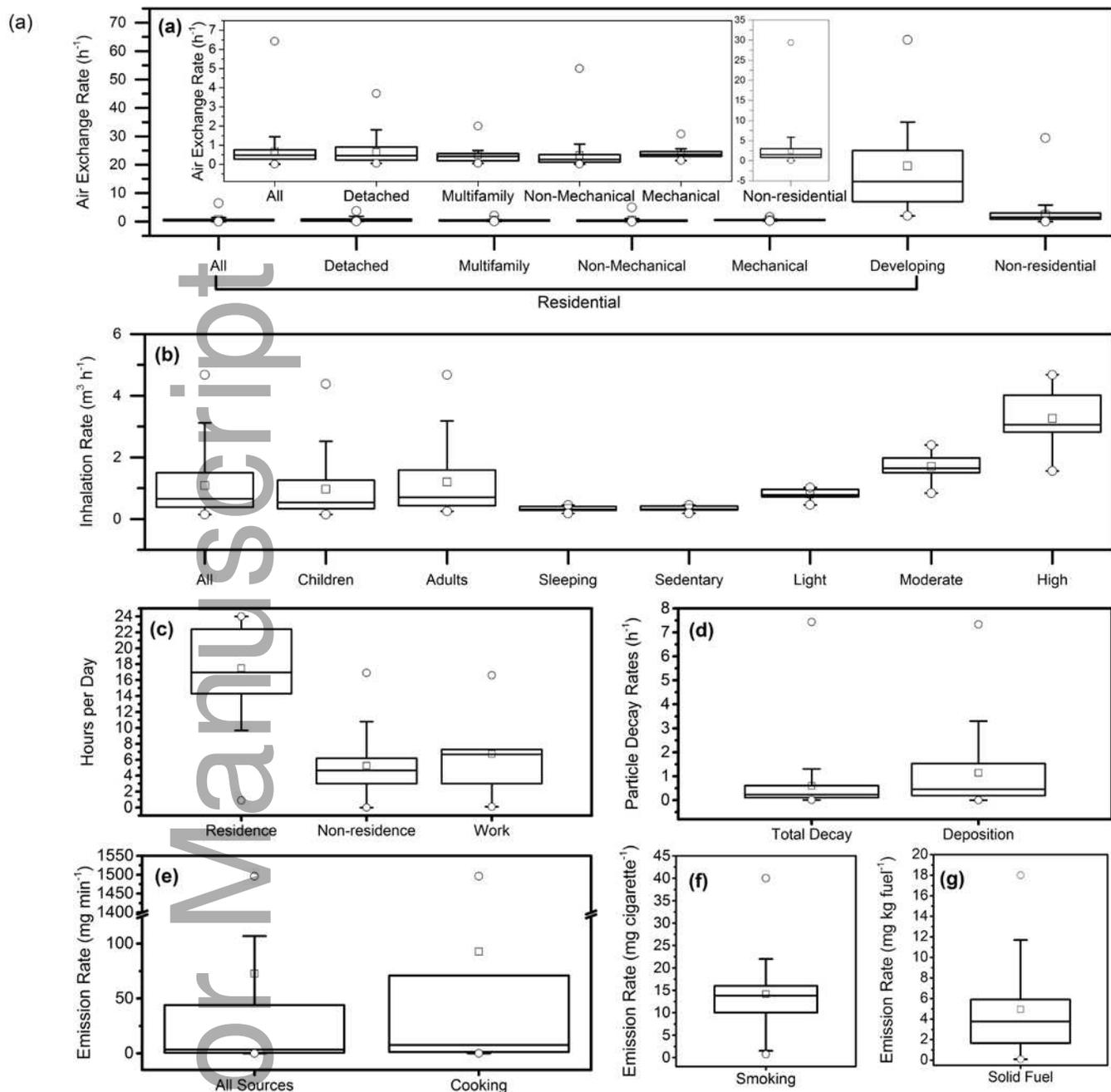
1538 of PM<sub>2.5</sub>: (a) air exchange rates, (b) inhalation rates, (c) time-activity factors, (d) particle decay

1539 rates, (e) indoor PM<sub>2.5</sub> source strengths, and (f) occupancy and building volume. (a) Air exchange

1540 rates are shown for detached/single-family homes (“Detached”), multifamily homes

1541 (“Multifamily”), homes without mechanical ventilation (i.e., infiltration and natural ventilation)

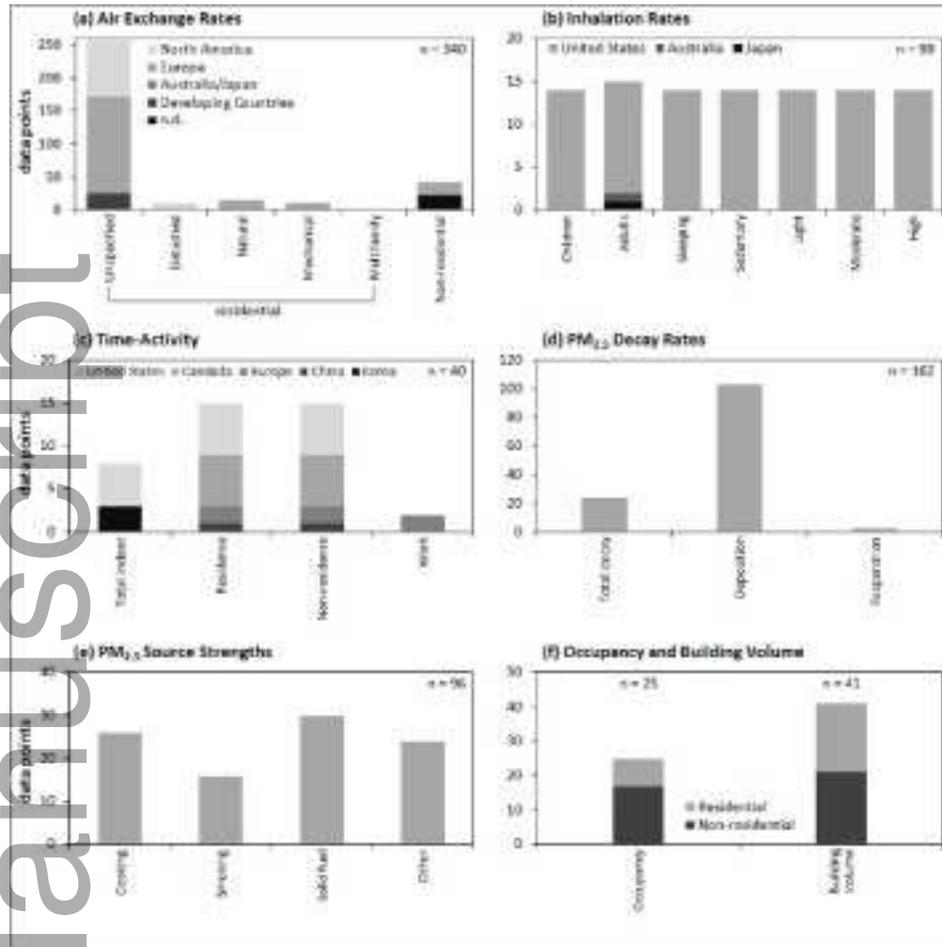
1542 (“Non-Mechanical”), mechanically ventilated homes (“Mechanical”), homes in developing  
1543 countries (“Developing”), residential buildings for which the above-described characteristics  
1544 have not been specified (“Unspecified”), and non-residential buildings (“Non-residential”). (b)  
1545 Inhalation rates are for adults, children, and by activity level (sleeping, sedentary, light,  
1546 moderate, and high). (c) Time-activity factors include total hours spent indoors (“Total  
1547 Indoors”), in the residence (“Residence”), in other indoor locations (“Non-residence”), and at  
1548 work (“Work”) per day. (d) Particle decay rates are for all particle loss mechanisms combined  
1549 (“Total Decay”) and for losses driven only by deposition. (e) Indoor PM<sub>2.5</sub> emission source  
1550 strengths include cooking, smoking, solid fuel combustion, and other indoor sources. (f)  
1551 Occupancy and building volume data are categorized by residential and non-residential indoor  
1552 environments. Where possible, data are categorized by country/geographic region (Not  
1553 determined (“n.d.”) means that geographic region is unspecified). Studies included here have  
1554 primarily been conducted in North America and Europe (a,b,c). In addition, there are disparities  
1555 in the types of indoor environments studied in previous work, with the majority of studies  
1556 focusing on residential environments and a smaller number of studies considering industrial and  
1557 commercial buildings.



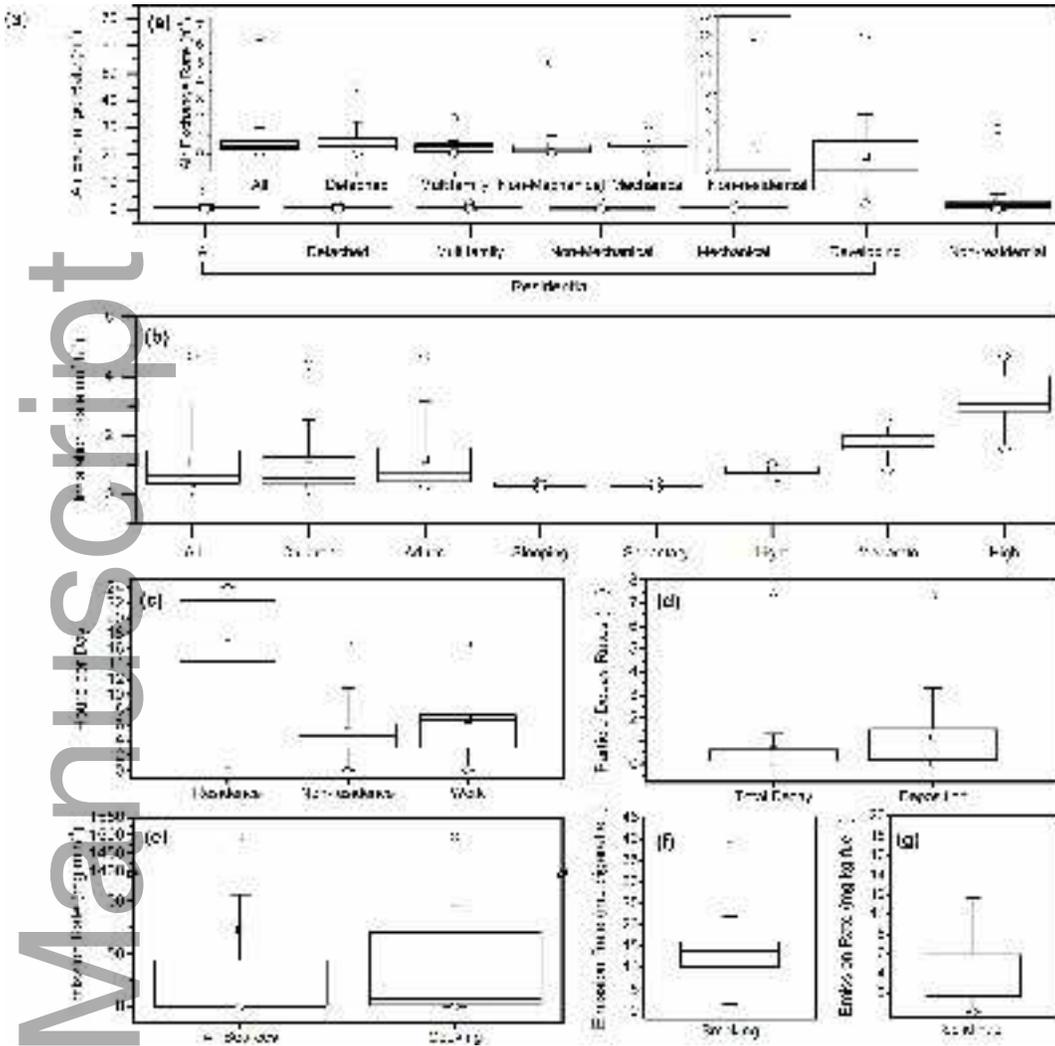
1558  
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 1560 **Figure 2.** Summary of measured or modeled values describing the parameter of interest for (a)  
 1561 building air exchange rates, (b) inhalation rates, (c) time activity factors, (d) particle decay rates,  
 1562 and (e) – (g) indoor PM<sub>2.5</sub> source strengths reported in the literature. For all plots, the boxes  
 1563 indicate the 25<sup>th</sup> percentile, median, and 75<sup>th</sup> percentile. Minimum and maximum values are  
 1564 indicated with circles and mean values are indicated with squares. (a) Air exchange rates shown  
 1565 are for all homes combined (excluding homes in developing nations) (“All”) and separately for  
 1566 detached/single-family homes (“Detached”), multifamily homes (“Multifamily”), homes without

1567 mechanical ventilation (i.e., infiltration and natural ventilation) (“Non-Mechanical”),  
1568 mechanically ventilated homes (“Mechanical”), homes in developing countries (“Developing”),  
1569 and non-residential buildings (“Non-residential”). (b) Inhalation rates are for all measurements  
1570 combined (“All”), and separately for adults ( $> 21$  years), children ( $\leq 21$  years), and activity level  
1571 (sleeping, sedentary, light, moderate, and high). (c) Time-activity factors include hours per day  
1572 spent in the residence (“Residence”), in other indoor locations (“Non-residence”), and at work  
1573 (“Work”). (d) Particle decay rates are given for all particle loss mechanisms combined (“Total  
1574 Decay”) and for losses driven only by deposition. (e) Source emissions are given for common  
1575 indoor PM<sub>2.5</sub> sources including cooking, cleaning, smoking, and various appliances combined,  
1576 excluding the combustion of solid fuels (“All Sources”). (e), (f), and (g) Source emissions are  
1577 also illustrated for cooking, smoking, and solid fuel combustion separately. The total number of  
1578 observations for each parameter is shown in Figure 1 and all underlying data are provided in the  
1579 SI.

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