

Indoor Sources of Air Pollutants

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ABSTRACT

People spend an average of 90% of their time in indoor environments. There is a long list of indoor sources that can contribute to increased pollutant concentrations, some of them related to human activities (*e.g.* people's movement, cooking, cleaning, smoking), but also to surface chemistry reactions with human skin and building and furniture surfaces. The result of all these emissions is a heterogeneous cocktail of pollutants with varying degrees of toxicity, which makes indoor air quality a complex system. Good characterization of the sources that affect indoor air pollution levels is of major importance for quantifying (and reducing) the associated health risks. This chapter reviews some of the more significant indoor sources that can be found in the most common non-occupational indoor environments.

1 Introduction

Outdoor air pollution has been extensively studied for a considerable amount of time, with the first long-term fixed outdoor monitoring stations being established in the 1960s in the UK.¹ Traditionally, outdoor concentrations have been used in epidemiological studies to evaluate health effects of air pollution.^{2,3} However, people spend an average of 90% of their time indoors^{4,5} and, therefore, indoor environments will contribute significantly to the total daily exposure, as the latter is the product of the pollutant

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concentration (which will vary with time and with space/microenvironment) and the time that a person is in contact with the given pollutant.⁶

Although the indoor environment has been explored much less extensively than the outdoor atmosphere, there has been a focus on characterizing indoor air pollution over the past few decades, and several studies have attempted to understand the sources and the various parameters that impact on indoor air quality (IAQ). However, there are still major challenges to be addressed. It is difficult to characterize air pollution in indoor environments, for two main reasons in particular: (1) people spend time in a wide range of indoor environments (*e.g.* home, office, school, restaurants) and, therefore, are exposed to a wide range of indoor sources of air pollution; home is the indoor environment where people spend most of their time (58–69%), followed by their work environment (28%) (Table 1); (2) real-world measurements of indoor environments require consent and willingness of the owner or the person responsible for the facility to participate in a study, and these may sometimes be difficult to obtain. Air pollution monitoring instruments are usually large and noisy, hence space requirements and disruption may cause unbearable annoyance. Furthermore, high instrumentation costs may limit the monitoring to a single room or area of the building. In recent years, this issue has been overcome by the use of miniaturized instruments or sensors,⁷ although sometimes their use implies a trade-off in precision and/or accuracy and also in the air pollutants that can be measured [*e.g.* it is difficult to measure the chemical composition of particulate matter (PM) with a sensor].

IAQ is affected by a series of sources, including outdoor particulate and gaseous pollutants that infiltrate indoors⁸ and PM and gases emitted indoors either by human activities (*e.g.* cooking, cleaning^{9,10}) or by indoor materials, building surfaces¹¹ and biological surfaces.¹² Indoor activities are often characterized by short events, but with very high pollutant concentrations. A common source in most (probably all) environments is the infiltration of outdoor sources, which can explain a varying range as large as 30–80% of the indoor concentrations of PM_{2.5} (PM with a diameter of <2.5 µm),¹³ depending on many factors such as the building envelope and ventilation settings. In situations with the absence of or very few indoor sources (*e.g.* in homes during sleeping time), indoor concentrations of air pollutants (both gases and PM) usually show similar patterns to outdoor concentrations.^{14,15} The

Table 1 Percentage of time spent at home and in other microenvironments.

Home (%)	Work (%)	Other indoor (%)	Population	Country	Ref.
68.7	18.2		Adults and children	USA	Klepeis <i>et al.</i> (2001) ⁴
58.1	28.0	7.0	Adults	Europe (7 countries)	Schweizer <i>et al.</i> (2007) ⁵
65.4	—	—	Adults and children	Germany	Brasche <i>et al.</i> (2005) ²⁰¹

contributions of outdoor sources to indoor air pollution are covered in a later chapter. Moreover, indoor chemistry is determined by indoor conditions that differ from those outdoors such as sunlight incidence, temperature variability, dispersion and surface-to-volume ratio.¹⁶

Owing to this wealth of sources and chemical reactions, the composition, and thus toxicity, of indoor particles and gaseous compounds are dynamic and very complex.⁶ Various studies have found associations between exposure to indoor air pollution and impaired health. Poor IAQ in non-occupational settings has been associated with several negative health effects, including asthma exacerbation,¹⁷ increased blood pressure¹⁸ and 'sick building' syndrome, a set of non-specific symptoms (*e.g.* headaches, allergy, eye irritation) related to the time spent indoors and that may be partly explained by exposure to indoor air pollutants. In low- and middle-income countries, household air pollution (HAP) from the use of solid fuels in inefficient stoves for cooking or heating is a major issue. According to the Global Burden of Disease study, HAP is the tenth leading global risk factor for deaths [eighth for disability-adjusted life-years (DALYs)] and was responsible for 2.8 million deaths (and 85.6 million DALYs) in 2015.¹⁹

In contrast to ambient air pollutants, only a few countries (*e.g.* China, Portugal and Taiwan) have established standards for indoor air pollutants. Therefore, IAQ is not routinely monitored. However, the World Health Organization (WHO) has proposed some guidelines for some indoor air pollutants²⁰ (Table 2).

Asbestos and radon and its decay products are often the targets of studies aimed at characterizing IAQ. However, asbestos fibres and radioactive particles are not discussed in this chapter, and readers are directed to many comprehensive publications on this topic such as the IARC Monograph on asbestos²¹ and the WHO *Handbook on Indoor Radon*.²²

2 Indoor Sources in Homes

One may expect indoor concentrations of air pollutants to be lower than outdoors as the building may exert some kind of protection against outdoor sources of air pollution (*e.g.* traffic and industrial emissions). However, in addition to possible strong infiltration of some outdoor air pollutants,¹³ there are several routine domestic activities, such as smoking, cooking and cleaning, that constitute important sources of indoor pollution.

2.1 Sleeping

Of all our lifetime activities, sleeping occupies the most time. Time-activity pattern studies and surveys report similar trends in sleeping around the world, with about 8–9 h per day of sleeping on average across all ages.^{23–25} The extent of the time spent sleeping makes the bedroom an important microenvironment to explore.²⁶ Generally, most studies show that indoor air particle number concentrations (PNCs) of ultrafine particles (UFPs; particle

Table 2 Indoor air pollution guidelines.

Pollutant	Guidelines (averaging time)	Ref.
Benzene (C ₆ H ₆)	No safe level of exposure Unit risk for leukaemia is 6×10^{-6} per $1 \mu\text{g m}^{-3}$	WHO (2010) ²⁰
Carbon monoxide (CO)	100 mg m ⁻³ (15 min) 35 mg m ⁻³ (1 h) 10 mg m ⁻³ (8 h) 7 mg m ⁻³ (24 h)	WHO (2010) ²⁰
Formaldehyde (CH ₂ O)	0.1 mg m ⁻³ (30 min)	WHO (2010) ²⁰
Naphthalene (C ₁₀ H ₈)	0.01 mg m ⁻³ (1 year)	WHO (2010) ²⁰
Nitrogen dioxide (NO ₂)	200 $\mu\text{g m}^{-3}$ (1 h) 40 $\mu\text{g m}^{-3}$ (1 year)	WHO (2010) ²⁰
Benzo[<i>a</i>]pyrene (B[<i>a</i>]P; C ₂₀ H ₁₂) as marker of PAH mixture	No threshold determined Unit risk for lung cancer for PAH mixtures is 8.7×10^{-5} per 1 ng m^{-3} of B[<i>a</i>]P	WHO (2010) ²⁰
Radon (Rn)	Recommended reference level of 100 Bq m^{-3} . If this level cannot be reached, the reference level should not exceed 300 Bq m^{-3} (10 mSv year^{-1})	WHO (2010) ²⁰
Trichloroethylene (C ₂ HCl ₃)	No threshold determined Unit risk for cancer is 4.3×10^{-7} per $1 \mu\text{g m}^{-3}$	WHO (2009) ²² WHO (2010) ²⁰
Tetrachloroethylene (C ₂ Cl ₄)	0.25 mg m^{-3}	WHO (2010) ²⁰

size <100 nm) and black carbon (BC) reach their minimum in homes during unoccupied periods and during the night, when few active human activities occur.^{27,28}

In addition to background concentrations of air pollutants from outdoor infiltration or generated previously from indoor activities, the bedroom will contain emissions from furniture and building materials. Of particular interest are soft furnishings such as mattresses and pillows, with which people are in close contact during their sleep. Mattress dust is comprised of a wide range of viruses, organisms (bacteria, fungi) and their allergens and inorganic dust,²⁶ all of which can be resuspended during movement. Moreover, a mattress is a source of a variety of (semi-)volatile organic compounds [(S)VOCs], such as plasticizers and flame retardants, which may volatilize and are known to be endocrine-disrupting chemicals.^{29,30} Little literature is available on human-induced particle resuspension from pillows, mattresses and other bedding items, but it has been reported to be comparable in magnitude to resuspension induced by other human indoor activities, such as walking.³¹

Bedrooms may sometimes be characterized by lower ventilation rates than the other home environments,³² which may hinder the dispersion of indoor-generated pollution and, therefore, favour its accumulation. Few studies are available on investigations of ventilation patterns and the presence of different air pollutants during time spent sleeping,³³ with inconclusive results. Owing to the time spent in this microenvironment, further research is needed to characterize exposure fully.

2.2 Cooking

Cooking activities have also been linked to increased concentrations of PM (and specially PNCs) and gaseous pollutants in indoor home environments. Studies have reported a wide variability between regions, owing to different stove types, fuels, cooking styles and food types.^{9,34}

2.2.1 Type of Fuel. Cooking emissions are of particular importance in developing countries, where populations rely strongly on solid fuels and use inefficient cookstoves (especially in rural areas).³⁵ Around half of the world's households are dependent on solid fuels (*e.g.* wood, crop wastes, animal dung and coal) for cooking.³⁶ The proportion varies across regions, with solid fuels being used in >60% of the households in Africa and Southeast Asia, 46% in the Western Pacific region, 35% in the Eastern Mediterranean area and much less (<20%) in the American continent and Europe.³⁶ Globally, the exposure to emissions from burning solid fuels (for cooking, heating and lighting) was responsible for 2.8 million deaths and 85.6 million DALYs in 2015.¹⁹ Several studies have linked the exposure to these emissions with adverse pregnancy outcomes (*e.g.* low weight at birth, stillbirths),³⁷ respiratory diseases (*e.g.* respiratory tract infection, including tuberculosis, and also aggravation of inflammatory lung conditions such as asthma), cancer,^{38,39} cardiovascular disease (*e.g.* stroke), and other health issues (*e.g.* eye diseases, skin ageing).^{40,41}

Several studies (most of them carried out in India and China) have reported that the person cooking in these regions (mainly women⁴²) are exposed to very high concentrations of cooking fumes/HAP. Fuel type has a significant influence on IAQ and subsequent health effects. Indeed, the significant impact of biomass and coal combustion on indoor air pollutant concentrations has been outlined in several publications.^{43,44} The hierarchy of fuels in order of decreasing PM_{2.5} concentrations reported in kitchens is the following: biomass, coal, kerosene and LPG/electric stoves,⁴⁵⁻⁴⁸ with biomass and coal having much higher emissions of PM_{2.5} than kerosene and LPG/electric. For instance, in a study in southern India, concentrations of respirable PM (PM with an aerodynamic diameter $\leq 4 \mu\text{m}$; PM₄) in households using biomass ranged from 500 to 2000 $\mu\text{g m}^{-3}$ during cooking, with the 24 h personal concentration average being $231 \pm 109 \mu\text{g m}^{-3}$ for the person cooking and $90 \pm 21 \mu\text{g m}^{-3}$ for those not involved.⁴⁹ Moreover, in the same study, people using cleaner fuels (gas or kerosene) had a much

lower 24 h personal average exposure ($82 \pm 39 \mu\text{g m}^{-3}$).⁴⁹ In another study comparing the use of coal, gas and electric stoves for cooking in China, the reported 24 h average concentrations of $\text{PM}_{2.5}$ in the non-heating season were 213, 65 and $55 \mu\text{g m}^{-3}$,⁴⁴ respectively. Although a change of fuel would result in large reductions of PM and other pollutant concentrations, the possibilities for complete replacement with clean fuels may be limited for many years in regions where the fuel supply is unreliable or not affordable.⁵⁰

2.2.2 Type of Stove. The type of stove is also a strong determinant of the emissions and, thus, of indoor concentrations. In several developing countries, traditional cookstoves are often very inefficient, which results in a higher consumption of fuel and larger release of air pollutants.³⁵ The type of stove used in a household depends on the income, with improved stoves being associated with higher income.⁵¹ Interventions to switch current stoves to more efficient types probably represent a more achievable step than changing the fuel type for exposure reduction. Nevertheless, the success of a stove change is also challenging as it has to be accepted by the community and should be supported by the availability of replacement parts.⁵⁰ A meta-analysis of the performance of 'cleaner' stoves (either by adding a chimney or a smoke hood or because of improved stoves) in real-life operation showed large reductions in 24 h average kitchen pollutant concentrations.⁵² For $\text{PM}_{2.5}$, the reduction was 42% (29–50%) for changing to advanced combustion stoves and 83% (64–94%) for changing to ethanol stoves. Regarding CO reductions, the reduction ranged from 39% (11–55%) for solid fuel stoves without chimneys to 82% (75–95%) for ethanol stoves.⁵² Replacement of traditional stoves with improved stoves or adding a chimney resulted in an 87% reduction in NH_3 concentrations⁵³ and a 40% reduction in BC concentrations.⁵⁴ Laboratory tests of improved stoves usually report higher reductions than field studies, which indicates the importance of testing the stoves in real-world conditions.⁵⁴

2.2.3 Ingredients and Type of Cuisine. In addition to pollution generated from the different fuels, food can also emit large amounts of pollutants. Several studies have reported elevated concentrations derived from the cooking of the ingredients themselves,^{55,56} and reported cooking as one of the indoor activities that contribute to PM the most in households in developed countries, especially within the fine and ultrafine particle size range.^{57–60} Cooking was estimated to contribute 62% of the total $\text{PM}_{2.5}$ in eight homes in Hong Kong, with a much lower relative contribution to PM_{10} (17.2%).⁵⁷ The average increase above background concentrations of $\text{PM}_{2.5}$ for cooking activities was estimated to be $51 \mu\text{g m}^{-3}$ in 37 homes in the USA.⁶¹ In Norway, frying a waffle would lead to PM_{10} concentrations 19 times higher (peak hourly concentration of $130 \mu\text{g m}^{-3}$) if the hood was on and 62 times higher ($246 \mu\text{g m}^{-3}$) if the hood was off than background concentrations.⁶² For PNCs, concentrations have been reported to be between 10 and 40 times higher in kitchens after cooking,^{59,63} although

other studies reported increases of many orders of magnitude when comparing non-cooking hours (1220–6200 particles cm^{-3}) and cooking hours (1 400 000 particles cm^{-3}) in Taiwanese homes.⁶⁴

The cooking method affects emissions. Different studies are consistent in reporting the following high-to-low emission hierarchy for different cooking methods: barbequing (due to the use of solid fuels), grilling, deep frying, pan frying, stir frying, boiling and steaming for PM mass (PM_{2.5} and PM₁₀)^{65–69} and most of its chemical constituents [elemental carbon (EC), organic carbon (OC), polycyclic aromatic hydrocarbons (PAHs), aldehydes, metals]^{67,70} and PNC.^{68,71–73} See and Balasubramanian^{67,73} reported the following concentration increments for each cooking method for PNC (0.01–0.5 μm) and PM_{2.5}: 1.3×10^4 particles cm^{-3} and 50.3 $\mu\text{g m}^{-3}$ for steaming, 2.6×10^4 particles cm^{-3} and 66 $\mu\text{g m}^{-3}$ for boiling, 4.7×10^4 particles cm^{-3} and 104.6 $\mu\text{g m}^{-3}$ for stir frying, 6.9×10^4 particles cm^{-3} and 114.6 $\mu\text{g m}^{-3}$ for pan frying and 52.1×10^4 particles cm^{-3} and 174.6 $\mu\text{g m}^{-3}$ for deep frying.

Owing to the combustion processes that take place during cooking, emission and formation of UFPs occur and the hot vapours may also cool and nucleate to form more UFPs.⁷⁴ The majority of particulate emissions generated during the cooking process are within the submicrometre size range (generally the ultrafine range, <100 nm) and include oil droplets, combustion products, steam from the water used to cook and from the ingredients, and condensed organic pollutants.⁹ Organic compounds (*e.g.* fatty acids, diacids, alkanes, aldehydes and PAHs⁷⁰) contribute significantly to cooking emissions, owing to the chemical reactions that sugars and carbohydrates undergo when heated and/or combined with other ingredients.⁹ The ingredients themselves may also cause the rates of emission to vary. Those ingredients with higher fat contents may lead to the highest emissions of PM_{2.5} while being cooked in the same way as ingredients with low fat contents, as they produce more fatty acids. This has been observed for different kinds of meat^{69,70,75} and also for different types of oil.^{58,70,75,76} The cooking temperature also affects particle emissions, with higher emissions being observed at higher oil temperatures.^{58,77,78}

2.3 Cleaning

Cleaning is another common indoor activity that can cause the IAQ to deteriorate. US adults spend on average 20–30 min per day cleaning their homes.⁷⁹ Cleaning tasks are necessary for the removal of unwanted substances, such as dust and biological aerosols, from floor and furniture surfaces. However, such activities may result in the resuspension and redistribution of previously settled particles on the different indoor surfaces, in addition to generating emissions from the cleaning products. The former generally generates coarser particles, thus affecting particle mass concentrations,^{80–82} whereas the latter leads to UFPs, thus influencing PNCs,^{83,84} although mass concentrations may also be increased.^{10,85} Factors that affect the effectiveness of cleaning activities are the device

used (*e.g.* vacuum, broom), the type of floor surface and the cleaning behaviour. Longer periods between cleaning activities increase the floor dust loading (the mass of dust per unit surface area) and, consequently, resuspension of floor dust.⁸⁶ As with cooking, this source leads to increased concentration events that tend to be relatively brief and intermittent, but may generate high peaks of concentration.⁸⁷

Resuspension of particles by mechanical turbulence during cleaning activities such as vacuuming and sweeping affect mostly the coarse fraction of PM (particles between 2.5 and 10 μm ; $\text{PM}_{2.5-10}$). For instance, in a set of carpet vacuuming experiments, researchers observed an increase of $>17 \mu\text{g m}^{-3}$ of PM_{10} , whereas the resuspension of $\text{PM}_{2.5}$ was established as $1.1 \mu\text{g m}^{-3}$.⁸⁰ In another study evaluating typical indoor activities, dry dusting was found to contribute $32 \mu\text{g m}^{-3}$ to indoor $\text{PM}_{2.5}$ concentrations whereas the peak 3 min PM_5 concentrations for vacuuming were reported to be between 81 and 90 $\mu\text{g m}^{-3}$.⁸⁸ In a study by Vu *et al.*⁸⁹ using a vacuum cleaner, the mass concentration peaks for PM_1 , $\text{PM}_{2.5}$ and PM_{10} were 1.5, 22.7 and 75.4 $\mu\text{g m}^{-3}$, respectively. Vu *et al.*⁸⁹ also measured the PNC during vacuum cleaner use and reported that more than 98% of the total PNC was found in the UFP range, with a peak PNC of 9.4×10^4 particles cm^{-3} and a unimodal distribution (mode 19.8 nm). However, rather than resuspension, the major source of UFPs has been attributed to spark discharging taking place at the motor at voltages above 100 V.⁹⁰

Most of the research regarding emissions from cleaning activities has been focused on those from the cleaning products themselves, which have been associated with a wide range of respiratory diseases⁹¹⁻⁹³ and are potential carcinogens and endocrine disruptors.⁷⁹ The wide range of cleaning products results in many different chemical compositions that may include terpenes, chlorine, amines and aldehydes.⁹⁴ These emissions from cleaning products, particularly terpenes (primarily limonene and linalool), can react with ozone (O_3) infiltrated from outdoors and be a significant source of indoor secondary organic aerosols (SOAs).^{16,95} The volatilities of the products of ozone-initiated reactions range from high (*e.g.* formaldehyde) to low (*e.g.* dicarboxylic acids), and the latter are those contributing to the formation of SOAs.¹⁶ High and rapid formation of UFPs can take place even when there is a low concentration of reactants.⁸⁴

Cleaning product emissions were identified as one of the major sources contributing to VOCs in indoor samples in non-smoking homes in Helsinki (Finland), explaining 18% of the variance of VOCs (outdoor traffic emissions also explained 18% of indoor VOCs variance).⁹⁶ In residences in Edmonton (Canada), household products (mostly cleaning products) accounted for 44% of the total VOCs.⁹⁷ An increased dilution of VOC compounds during the warmer seasons in association with higher ventilation was also observed. Cleaning experiments performed using a kitchen limonene-based cleaner soap in an apartment in Prague (Czech Republic)⁸⁹ reported peak UFP concentrations of 1.24×10^5 particles cm^{-3} (size mode of 30.6 nm), with PM mass concentrations for PM_1 , $\text{PM}_{2.5}$ and PM_{10} of 2.5, 10.4 and 22 $\mu\text{g m}^{-3}$,

respectively. It should be noted that higher PNCs and lower PM mass concentrations were reported for the kitchen cleaning product in comparison with the use of a vacuum cleaner in the same study.⁸⁹ In addition to low VOC-emitting products,⁹⁸ further reductions of exposure to VOCs and other pollutants during cleaning activities can be achieved with optimal ventilation.⁹⁷

2.4 Heating

Owing to low temperatures during winter, some regions of the world require heating to maintain a comfortable indoor temperature. In developing countries, most families use solid fuels in inefficient stoves for heating as they do for cooking, hence the magnitude of concentrations and subsequent health effects are similar to those already described for cooking emissions. In developed countries, central heating is often the main system. However, in recent years, wood-burning stoves and fireplaces have increased in popularity, partly owing to their attractiveness (generating a cosy and warm ambience) and also lower cost compared with other heating fuels.⁹⁹ Centrally heated homes in Germany showed lower concentrations of PM, heavy metals and PAHs than homes heated by coal-burning or an open wood-burning fireplaces.¹⁰⁰ In the UK, increased CO and NO₂ concentrations were observed in homes with unflued heating appliances than in homes with flued systems.¹⁰¹ Gustafson *et al.*¹⁰² compared PAH levels between homes using wood-burning appliances (such as residential wood log boilers, stoves and fireplaces) and homes using electrical heating appliances. They observed that the median 24 h indoor levels of benzo[*a*]pyrene were more than four times higher in the wood-burning homes (0.52 ng m⁻³) than in the reference homes (0.12 ng m⁻³). However, for most PAHs, indoor concentrations were lower than ambient concentrations.¹⁰² On the other hand, the use of seven wood-burning fireplace ovens (generally designed in such a way that the fire chamber is airtight) were tested with no significant increase in room concentrations of CO, CO₂, NO, NO₂, total VOCs, formaldehyde and acetaldehyde, but with increases in the concentrations of UFPs and benzene.⁹⁹

2.5 Tobacco Smoking

Tobacco smoking can generate substantial indoor air pollution, specially PM.¹⁰³ Around 4000 constituents have been isolated from the mainstream smoke of cigarettes, distributed between the gas phase and aerosol particles, including 11 compounds (2-naphthylamine, 4-aminobiphenyl, benzene, vinyl chloride, ethylene oxide, arsenic, beryllium, nickel compounds, chromium, cadmium and polonium-210) classified as IARC Group 1 human carcinogens,¹⁰⁴ and many other toxic agents such as carbon monoxide, ammonia, heavy metals, rare earth elements and PAHs.^{105,106} The cigarette burns at a higher temperature during inhalation, leading to more complete

combustion in the mainstream smoke (the smoke drawn through the cigarette and inhaled by the smoker). Hence side-stream smoke contains considerably higher concentrations of many carcinogenic and toxic substances than the mainstream smoke.¹⁰⁷ Exposure to environmental tobacco smoke (ETS) has been linked to an increased risk of several adverse health outcomes, including lung cancer and acute respiratory illness.^{105,108} The WHO estimated that tobacco use (not only exposure to smoke) is responsible for about 6 million deaths across the world each year, including about 600 000 deaths from the effects of second-hand smoke.¹⁰⁹ Thanks to tobacco control policies and population awareness of the negative health effects, the prevalence of tobacco smoking has been decreasing steadily since 2000 (26.9% smokers older than 15 years of age), with a worldwide prevalence (older than 15 years of age) in 2015 estimated to be 20.2%.¹¹⁰

The important implications for public health have led to smoking bans in public spaces, but regulatory approaches have limited efficacy in reducing exposure to ETS in private residences. Indoor PM_{2.5} mass concentrations have been reported to increase in homes with smokers by a factor ranging between 1.6 and 4.0^{103,111,112} in comparison with non-smoking homes, with the absolute differences ranging from 10 to 45 $\mu\text{g m}^{-3}$. The PM₁₀ mass difference between smoking and non-smoking homes is very similar to that observed for PM_{2.5}, indicating that tobacco smoke emissions contribute primarily to fine PM. This is also confirmed by studies reporting the median diameter for smoking emissions to range from 90 to 117 nm^{89,113} and observations that tobacco tracers (Cr, Ni, As, Cd and Pb) are mainly present in the fine PM fraction.¹¹⁴ The increments are even larger for specific components. For instance, Cd, Ce and La concentrations were 5.5, 24.0 and 29.3 times higher, respectively, in smoker than non-smoker households.¹¹⁵ For total VOCs, concentrations have been found to be 1.4–3.0 times higher in smokers' homes.^{112,116} The contribution from ETS seems to impact particle mass concentration more than PNC.⁸⁹ For instance, peak concentrations during smoking increased background home concentrations 4-fold for PM_{2.5} (peak concentration of 79 $\mu\text{g m}^{-3}$) and 1.5-fold for PNC (peak concentration of 26.6×10^3 particles cm^{-3}) in a study carried out in homes in Brisbane (Australia).⁶⁸

In addition to the exposure to second-hand tobacco smoke, exposure to third-hand tobacco smoke, which is the remaining tobacco smoke contamination that persists long (days, weeks) after the cigarette was extinguished, has also been reported. Some ETS constituents, such as the majority of PAHs, phenols, cresols, nicotine, cotinine and tobacco-related nitrosamines, can be affected by sorptive interactions with indoor surfaces^{117–119} and be re-emitted later in time. Consequently, in smokers' homes, the fingerprint from cigarette smoke is detectable in dust and on surfaces 2 months after smokers have moved out of their homes post cleaning for future tenants.¹²⁰ Thus, the third-hand smoke exposure of contaminated dust may contribute to the smoking-attributable burden of disease.

2.5.1 Electronic Cigarettes. Nowadays, electronic cigarettes (e-cigarettes) are widely used as a form of non-tobacco nicotine as a method to reduce or quit tobacco smoking. This gadget consists of a heating element that atomizes a solution (e-liquid) that usually contains water, nicotine, vegetable glycerine and some flavouring. Little information about the long-term effects of e-cigarettes is available, since they have been in use for only around 15 years. However, some studies have evaluated their emissions, which showed that e-cigarettes also affect PNC and PM mass concentrations, although to a much lesser extent than tobacco smoke. For instance, concentrations for conventional cigarettes were 76, 139 and 158 $\mu\text{g m}^{-3}$ for PM_{10} , $\text{PM}_{2.5}$ and PM_{10} , respectively, whereas those for e-cigarettes without nicotine were 3.5, 7.2 and 8.7 $\mu\text{g m}^{-3}$ (and less than 1 $\mu\text{g m}^{-3}$ in all fractions for e-cigarettes with nicotine).¹²¹ For PNCs, Protano *et al.*¹²² reported that peak concentrations for conventional cigarettes ranged from 1.2×10^5 to 2.9×10^5 particles cm^{-3} whereas for e-cigarettes the peak concentrations reached a maximum of 4.7×10^4 particles cm^{-3} . In measurements carried out at distances of 0.5 and 1 m from an e-cigarette smoker, Volesky *et al.*¹²³ found that $\text{PM}_{2.5}$ background concentrations increased 160- and 103-fold, respectively, with a maximum concentration of 174 $\mu\text{g m}^{-3}$. These increments were much lower for the PNC: 5.2 and 3.0 times the background concentrations at 0.5 and 1 m (maximum PNC: 284 260 particles cm^{-3}). Further research in this field and on the health effects of e-cigarettes is required for future regulations of e-cigarette use in areas where traditional tobacco smoking is not allowed.

2.6 Human Occupancy

Even if not performing a high emitting activity, human occupancy can contribute to IAQ. The movement of people (*e.g.* walking) can resuspend particles previously bound to different surfaces indoors (*e.g.* floors, furniture),¹²⁴ and people's bodies can contribute to airborne levels through desquamation of skin or by reactions of ozone with skin surface lipids and/or personal care products (*e.g.* fragrances).¹²⁵

Resuspension can be an important source of indoor particulate matter compared with other indoor sources.¹²⁴ The magnitude of resuspension varies with particle size, dust loading, surface type, nature and intensity of the activity, ventilation and relative humidity.^{88,126,127} It becomes particularly important for coarser particles, as it increases with particle size in the range 0.7–10 μm . In a laboratory, PM_{10} increased by 84% during walking experiments, whereas no impact on PNC was observed, confirming the minimal effect on UFPs.¹²⁸ Experiments with higher dust loadings were associated with higher PM_{10} concentrations in the room,¹²⁸ which indicates that regular removal of dust (*i.e.* cleaning) would result in an abatement of the resuspended particles by the movement of people inside a room. Carpets were associated with higher resuspension of particles in the range 3–10 μm than hard floorings, whereas no significant difference was observed in the

range 0.4–3 μm .¹²⁹ Tian *et al.*¹²⁹ observed that resuspended particles took about 90 min to settle back again and return to previous background levels. Many other activities, such as moving papers,¹³⁰ making the bed and folding clothes,⁸⁸ can also induce dust resuspension. More energetic events, activities on rugs instead of on wood floors and an increasing number of people performing the action are all associated with the generation of higher PM concentrations.^{82,88,131}

The composition of the resuspended dust depends on the environmental setting, the region and the occupant's practices.¹²⁴ Indoor dust can be constituted of mineral matter, carbonaceous particles, heavy metals, PAHs, pesticides, phthalates, flame retardants and other endocrine disruptors, bacterial and fungal, viruses, dust mites and other animal debris.^{81,132,133} Hence, in addition to health effects from exposure to common air particulates, allergen-related and infectious diseases have been associated with resuspended home dust exposures.¹³⁴

Moreover, the sole presence of a person affects indoor ozone chemistry, as ozone reacts rapidly with lipids of the exposed skin, hair and clothes.¹²⁵ This results in a significant decrease in ozone concentrations (and, thus, of ozone available for other reactions) while increasing the concentrations of airborne compounds with carbonyl, carboxyl or α -hydroxy ketone groups.¹²⁵ Therefore, occupied rooms would usually contain lower levels of SOA derived from ozone–terpene reactions, as the availability of oxidants is reduced owing to the interaction of ozone with the occupant's skin.¹²⁵ In terms of ozone consumption, squalene is the most important individual constituent in skin surface lipids, followed by unsaturated fatty acids.^{125,135} These skin oil reactions can take place in skin flakes present in indoor dust¹⁶ and also in the skin oils transferred to surfaces due to contact.¹³⁶ In addition, personal care products also have an impact on ozone consumption. Traces of soaps, shampoos, perfumes and other personal care products can desorb from a person for several hours after use.¹²⁵ Akin to cleaning products, one of the most abundant compounds that can be found in human bodies is limonene,¹³⁷ which can also be found in scented personal care products.^{10,138}

2.7 Building and Furniture Materials

Materials used for buildings (*e.g.* hardwood, plywood, laminate floorings, adhesives, paints and varnishes), furniture and other products used in homes may also influence IAQ and particularly contribute to VOC concentrations. For example, many materials commonly found indoors have the potential to emit formaldehyde. A noticeable characteristic of indoor environments is the large surface-to-volume ratio. Therefore, surfaces can become an influential factor of indoor air chemistry and quality through numerous mechanisms, such as oxidative reactions, sorption and acid–base chemistry.¹³⁹ Many additives are routinely added to a wide range of materials to enhance their characteristics.^{140,141} For instance, phthalates are used as plasticizers to improve the flexibility of poly(vinyl chloride) products

and brominated flame retardants are used as additives to increase the fire resistance of many materials. Since these additives are often not chemically bound to the polymer matrix, emission into the air can take place,¹⁴² which makes phthalates a ubiquitous and very abundant SVOC indoors.^{140,143} The reactivity of indoor surfaces can vary considerably from highly reactive (such as carpets) to poorly reactive (glass).¹⁴⁴

Material age is also an important characteristic, as emission rates tend to be higher for new materials and decrease with time.^{145,146} Ozone reacts with the available unsaturated bonds in the external surface layer and, with time, ozone uptake from surfaces decreases and so do the emission rates of secondary pollutants. This process has been termed 'ozone aging'.¹⁴⁷ For instance, homes painted within the previous 4 weeks exhibited higher total VOC concentrations than when painted previously.¹⁰¹ Temperature has also been found to affect material emissions. Liang *et al.*¹⁴⁸ observed seasonal variations in the emission rates of formaldehyde from a medium-density fireboard, with the highest concentrations being observed during summer (up to 20 times higher than in winter) rather than on the initial introduction of the material. Nevertheless, the influence of material age was also observed in their study since during the second year the concentrations decreased by 20–65% in the corresponding months.¹⁴⁸ Ventilation plays a major role in reducing exposures to surface-reaction emissions, *e.g.* increasing the air exchange rate from 0.6 to 1.8 h⁻¹ decreased SVOC concentrations (total gas-phase and particle-phase concentrations) by about 60%.¹⁴⁹ Furthermore, the use of 'green' building materials lowers VOC emissions compared with conventional materials. Specifically, materials containing minerals generate the least VOC emissions while exhibiting a fairly high ozone deposition.¹⁵⁰

3 Indoor Sources in Offices and Schools

Offices and schools share common sources that may differ from those in homes or that may be amplified owing to the use and characteristics of these two environments. Children spend a large fraction of their day in schools, whereas a considerable proportion of the adult population (particularly in developed countries) spend substantial hours in offices as part of their working life. Understanding the sources that affect IAQ in both environments is therefore critical to ensure safe and healthy environments.¹⁵¹ Several studies have reported the worst performance of pupils and office workers to be related to lower ventilation rates.^{152,153} Other studies have shown slower cognitive development related to exposure to indoor air pollutants, although the greatest effects were attributed to those related to traffic emissions that infiltrated from outdoors.¹⁵⁴ Moreover, both offices and schools operate primarily during daytime, when outdoor emissions that may infiltrate indoors are at their peak. Both establishments are also characterized by high occupancy rates (especially in schools).¹⁵⁵ One of the main differences between schools and office buildings is that offices rely more often on mechanical ventilation whereas schools are usually naturally ventilated.¹⁵⁵

Of the sources identified in homes, air pollutants derived from human occupancy are the source that will be especially magnified in schools owing to much higher occupancy rates than in the home. Occupancy in offices is often higher than in the home, but to a much lesser extent than in schools. In an office, walking and being around the monitoring equipment explained 24–55% of the variation of the PNC of particles with a diameter between 1 and 25 μm .¹⁵⁶ However, studies carried out in offices often conclude that the main source affecting indoor concentrations is infiltration from outdoor air,^{157–159} although there are a few sources that, in addition to human occupancy, may become important, such as the use of printers/photocopiers. Sources differing from those typical in the home arise in these environments, such as emissions from computers and printers (especially affecting offices and universities) and from the use of chalkboards in schools. Higher contributions from surface reactions may also be present, since the surface-to-volume ratio is generally higher in these environments (*e.g.* a large number of desks) than in the home. Moreover, office workers and schoolchildren would be exposed to negligible concentrations of secondary pollutants generated by cleaning activities if the cleaning is performed after working/school hours (as limonene degradation products are depleted overnight). However, afternoon cleaning would increase the exposure of the cleaning staff as the concentration of ozone is higher in the afternoon than in the early morning.¹⁶⁰

3.1 Human Occupancy and Other Determinants of Indoor School Air Quality

The daytime concentration of PM (and particularly $\text{PM}_{2.5-10}$) is associated with the number of people per hour per day in a classroom, with higher concentrations being observed with increasing numbers of people due to resuspension, clothing fibre emissions and biological emissions.^{124,161,162} In fact, in a study carried out in Pune (India), Jan *et al.*¹⁶³ suggested that the very high concentrations of $\text{PM}_{2.5}$ that they found in classrooms (4–20 times higher than in European schools) were due to possible differences in the building characteristics, but especially to levels of occupancy as they reported 58 students per classroom whereas in Europe the number of students per classroom is around 20–30. Moreover, the strength of resuspension is also dependent on the activity, with more active tasks resulting in higher resuspension,^{131,164} so it is particularly important in primary school classrooms since children tend to be more active at that age than when attending kindergarten or secondary school.^{165,166} Studies carried out in schools have consistently reported high indoor PM mass concentrations (especially of the coarse fraction), sometimes higher than those observed outdoors.^{167–169} Source apportionment was carried out to identify indoor sources in schools in Barcelona (Spain),¹⁷⁰ with a mineral source contributing 17% to $\text{PM}_{2.5}$ ($6.0 \mu\text{g m}^{-3}$). The high contribution of the mineral source in the schools in Barcelona was especially affected by the presence/absence of

sand-filled playgrounds. Those schools with paved playgrounds showed an average indoor concentration of $3.6 \mu\text{g m}^{-3}$ of the mineral source, whereas if the playground was unpaved the average concentration was $9.1 \mu\text{g m}^{-3}$.¹⁷⁰ The mineral components in classrooms were probably introduced into the classroom on children's shoes and clothing and resuspended indoors, hence concentrations are probably also dependent on occupancy ratios. High concentrations of mineral components due to unpaved playgrounds have also been observed in other studies.¹⁷¹

However, the main contributors to $\text{PM}_{2.5}$ mass concentrations in the schools in Barcelona were a source with a mixture of OC (from skin flakes, cotton fibres and other organic particulates from ozone reactions with skin oil and personal care products) and Ca and Sr from chalk use on blackboards (45% ; $16.0 \mu\text{g m}^{-3}$) and another important source of secondary aerosols characterized by organic compounds and secondary sulfate (11% ; $3.8 \mu\text{g m}^{-3}$). In an occupied classroom, Qian *et al.*¹⁷² reported emission rates per person-hour of 31 mg, 37×10^6 genome copies and 7.3×10^6 genome copies for total PM, bacteria and fungi, respectively. Around 18% of the bacterial emissions came from taxa closely related to the human skin microbiome. Buildings with high human occupancy are also characterized by high levels of airborne bacteria from the skin, nostrils and hair of occupants, which also accumulate in floor dust¹³³ and can lead to an increase in infectious diseases. Different studies carried out by Fox and co-workers^{173,174} in school classrooms confirmed the presence of skin flakes in airborne particles by measuring muramic acid, 3-hydroxy fatty acids and the protein K10 epithelial keratin. The contribution from chalk use on blackboards (which leads to very high concentrations of Ca and Sr) has been reported in numerous studies,^{163,167,168} and the use of markers is associated with increased VOC concentrations.¹⁶² Frequent cleaning and ventilation would help to reduce the concentrations related to occupancy, but care should be taken to avoid exposing children to cleaning-generated pollution (*e.g.* by using low VOC-emitting products and scheduling cleaning after school hours) and to prevent outdoor pollutants infiltrating indoors (*e.g.* by avoiding natural ventilation during peak traffic hours).¹⁵⁴

3.2 Computers, Printers and Photocopiers

Devices such as computers, printers and photocopiers have been reported to be the source of a variety of air pollutants,^{155,175} including PM and ozone, which can initiate reactions leading to secondary pollutants and UFP formation.^{10,98,175} In fact, all investigations into the particle size distributions emanating from different printers and copiers demonstrate the release of particles primarily in the UFP range.^{176,177} Modern printers and photocopiers now release much lower ozone concentrations than older versions,¹⁷⁸ with those emitting ozone being associated with higher PNCs.¹⁷⁹ Most of the studies evaluating printers and photocopiers were chamber experiments, with few data collected in real world-conditions in offices and

schools. Most of the studies agree that, in offices, outdoor sources (*e.g.* traffic emissions) usually represent a major contributor to PNCs in comparison with indoor emissions such as those from printers.^{158,180} McGarry *et al.*¹⁸⁰ reported peak PNC emissions during printing in offices, whereas printer emissions could not be discriminated from background concentrations by Szigeti *et al.*,¹⁵⁸ probably owing to the use of low-emitting printers. Moreover, printers and photocopiers are often installed in dedicated rooms, rather than in the office itself, which means that only the personnel present during the printing or the immediate period after the printing (<30 min) are exposed to elevated PNCs.¹⁸⁰ Computers have been reported to emit little ozone and (S)VOCs (*e.g.* brominated and organophosphate flame retardants)¹⁸¹ in comparison with printers and photocopiers.¹⁷⁵ However, even with low emissions they may become important as they represent a long-term source of (S)VOCs.¹⁷⁵

4 Indoor Sources in Other Microenvironments

Apart from the home and the workplace, people also frequent environments such as restaurants, hairdressers, nail salons and fitness centres. Although visits to such establishments are for relatively short time periods and are not a daily occurrence, they may contribute to the exposure to specific pollutants if very high concentrations are present. From an occupational point of view (which is not the subject of this chapter), people working in these environments are exposed to high concentrations for prolonged periods, but clients are also affected. This section briefly covers a selection of these environments.

4.1 Restaurants

The main source of high levels of air pollutants in a restaurant is cooking emissions. Concentrations of pollutants in restaurants and commercial kitchens were reported to be higher than those observed in homes and domestic kitchens.⁹ This is attributed to the volume of food cooked, longer cooking periods and the cooking methods employed. In California, the mean PNC in restaurants (71 600 particles cm^{-3}) was 12.3 times higher than the average background in homes.¹⁸² In South Korea, higher concentrations of respirable PM ($\text{PM}_{3.5}$) and NO_2 were found in restaurants (159 $\mu\text{g m}^{-3}$ and 56 ppb, respectively) during meal preparation for lunch and dinner than in homes (91 $\mu\text{g m}^{-3}$ and 24 ppb, respectively).¹⁸³ Higher emissions derived from high occupation rates that may increase resuspension and body emissions are also expected, although these emissions have not been the topic of interest when assessing air quality in restaurants.

Nowadays, smoking is forbidden in public spaces and restaurants in many countries. Studies around the world comparing air quality in bars and restaurants before and after the smoking ban agree in indicating considerable reductions of PM mass concentrations (often above 85%). For $\text{PM}_{2.5}$, average

concentrations were reduced from $126 \mu\text{g m}^{-3}$ before the ban to $11.8 \mu\text{g m}^{-3}$ after the ban came into force in 78 restaurants in 14 cities in Michigan (USA),¹⁸⁴ and from 202.17 to $25.53 \mu\text{g m}^{-3}$ in 10 pubs and bars in a city in southern USA.¹⁸⁵ In Barcelona (Spain), PM_{10} concentrations in a cafeteria decreased from 148 to $71 \mu\text{g m}^{-3}$ (52%), with a 100% reduction of some PM_{10} components (Mn, Ga, Nb, Cd, La, Ce, Nd),¹⁸⁶ and the PNC was also reduced from $49\text{--}75 \times 10^3$ to 22×10^3 particles cm^{-3} . Hence the introduction of the smoking ban had a very positive effect on IAQ.

4.2 Hair Salons

Several activities are performed in hair salons that can lead to poor IAQ. Dyeing, drying and other hair care-related activities involve the use of various chemicals, which can release large amounts of organic compounds, including alcohols, glycols, aldehydes and amines.¹⁸⁷ Some of these chemicals (*e.g.* formaldehyde, ammonia, benzenediamines, parabens) are known to be allergenic or potentially carcinogenic.^{187,188} The predominant VOCs found in hair salons are aromatics (*e.g.* toluene, xylene), esters, ketones and terpenes (*e.g.* pinene, limonene).¹⁸⁹ Exposure to these compounds has been associated with respiratory diseases, eye irritation and reproductive disorders.^{190,191} For instance, a study of 50 hair salons in Portugal found an average concentration of total VOCs of 1.4 mg m^{-3} , which is significantly higher than the Portuguese reference concentration of 0.6 mg m^{-3} .¹⁹² In Tokyo (Japan), ammonia concentrations ranged from 0.15 to 0.87 mg m^{-3} , being higher during the chemical waving of hair (perming), as this procedure uses a product containing thioglycolic acid added as the ammonium salt.¹⁹³ Ammonia is usually present in permanent dyes, as it facilitates colour intensification. Average ammonia concentrations of 2.3 ppm in the 50 hair salons in Portugal already mentioned were reported, well below the Portuguese reference value of 20 ppm, although two of the salons reported concentrations above the limit.¹⁹² Good ventilation (preferably mechanical) would help to minimize the exposure of clients and hairdressers to these chemicals in hair salons.

4.3 Nail Salons

The number of nail salons has expanded rapidly over the past two decades.¹⁹⁴ Although exposure in these environments is limited to a relatively small proportion of the population who access nail care services (mainly women), exposures can be very intense, albeit for short periods of time. Recently, a few studies have been performed in nail salons (most of them focusing on the occupational exposure of nail technicians). Manicures and the application of artificial fingernails usually take place with the client's hands resting over a table, and as a consequence the work is performed in close proximity to the breathing zone of both client and technician. Products used in nail salons contain a large array of potentially toxic VOCs in the form

of solvents, pigments, adhesives, plasticizers, resins and acids.¹⁹⁵ For instance, formaldehyde is commonly present in nail products.¹⁹⁶ Exposure to nail products has been associated with spontaneous abortions, skin problems, irritation of respiratory ways, decreased lung function, musculo-skeletal conditions and recurrent headaches.^{194,195} Concentrations of total VOCs and PM_{2.5} in nail salons in Boston (USA) while performing nail services were 11 000 ppb and 24 $\mu\text{g m}^{-3}$, respectively, which decreased to 600 and 10 $\mu\text{g m}^{-3}$ when no services were being performed.¹⁹⁷ Concentrations were uniform in all room areas, with no significantly higher levels when measuring at the nail table. Alaves *et al.*¹⁹⁶ analysed more than 100 different VOCs in nail salons in Salt Lake City (USA) and reported that all concentrations were below the recommendations for occupational exposure (including ethyl methacrylate, the principal ingredient in artificial nail preparations), except for formaldehyde (average concentrations ranging from 0.009 to 0.032 ppm). Nail tables with a downdraught vent are recommended for lowering exposure in the breathing zone for both client and nail technician.¹⁹⁸

4.4 Fitness Centres

Fitness centres are increasingly used by people to increase/maintain their fitness and health. During exercise, the enhanced inhalation rates of participants will increase their respiratory intake of indoor air pollutants. That gyms and fitness centres maintain good air quality is therefore particularly important. The main indoor source of air pollution expected in fitness centres is derived from human occupancy (from skin oil reactions, personal care products and resuspension), which may be elevated due to high levels of physical activity. Few research data are available but, for example, in a study in four fitness centres in Porto (Portugal), PM₁₀ and PM_{2.5} concentrations were much higher than outdoor concentrations, particularly in the smallest centres and during periods when a fitness class was in session.¹⁹⁹ Of note, those centres with mechanical ventilation (including filtration of outdoor air) reported much lower concentrations than those that ventilated the premises by opening windows. The study also reported that PNC (20–1000 nm) concentrations were about 2–3-fold lower in centres with mechanical ventilation (around 3700 particles cm^{-3}) than in those with natural ventilation (6700–9400 particles cm^{-3}), which may be attributed to the entry of particles from outdoor sources. In contrast to what one would expect, Slezakova *et al.*¹⁹⁹ reported that PM_{2.5} contributed around 95% of the PM₁₀, thus indicating that most of the particles were within the fine rather than the coarse size range. Within different indoor sporting environments, the highest levels of PM mass concentrations have been reported for gymnastic halls and climbing centres (up to 4000 $\mu\text{g m}^{-3}$ for PM₁₀), owing to the frequent use of magnesium chalk for drying hands.^{199,200} Further studies are required to better characterize and quantify indoor sources of air pollution in such facilities.

5 Final Remarks

Indoor air pollutants may contribute significantly to personal exposure given the long periods spent in indoor environments. Different activities can increase pollutant concentrations, such as cooking, cleaning, dust resuspension from people's movements, personal organic emissions and many more. Chemical reactions initiated indoors by reactive species such as O₃ may also contribute to poor IAQ. Different toxicities may arise from the wide range of pollutants that can be emitted by indoor sources, including gaseous pollutants (*e.g.* NO_x, CO, O₃), inorganic particles (*e.g.* EC, carbonates, metals) and organic particles (*e.g.* VOCs, PAHs). Although some of these components may be harmless, others may have a significant impact on health. Given the scale of the exposures (because of long periods and/or high concentrations), the need for good characterization of IAQ is essential to quantify the associated health risks.

Abbreviations

BC	Black carbon
DALY	Disability-adjusted life-year
EC	Elemental carbon
ETS	Environmental tobacco smoke
HAP	Household air pollution
IAQ	Indoor air quality
OC	Organic carbon
PAH	Polycyclic aromatic hydrocarbon
PM	Particulate matter
PM _{2.5}	PM with a diameter of <2.5 μm
PNC	Particle number concentration
SOA	Secondary organic aerosol
SVOC	Semi-volatile organic compound
UFP	Ultrafine particle
VOC	Volatile organic compound
WHO	World Health Organization

References

1. UK Air DEFRA, Brief history of Monitoring Networks, <https://uk-air.defra.gov.uk/networks/brief-history> (accessed Sep 19, 2018).
2. D. W. Dockery, C. A. Pope III, X. Xu, J. D. Spengler, J. H. Ware, M. E. Fay, B. G. Ferris and F. E. Speizer, An Association between Air Pollution and Mortality in Six U.S. Cities, *N. Engl. J. Med.*, 1993, **329**(24), 1753–1759.
3. R. W. Atkinson, G. W. Fuller, H. R. Anderson, R. M. Harrison and B. Armstrong, Urban Ambient Particle Metrics and Health. A Time-Series Analysis, *Epidemiol.*, 2010, **21**(4), 501–511.
4. N. E. Klepeis, W. C. Nelson, W. R. Ott, J. P. Robinson, A. M. Tsang, P. Switzer, J. V. Behar, S. C. Hern and W. H. Engelmann, The National

- Human Activity Pattern Survey (NHAPS): A Resource for Assessing Exposure to Environmental Pollutants, *J. Exposure Anal. Environ. Epidemiol.*, 2001, **11**(3), 231–252.
5. C. Schweizer, R. D. Edwards, L. Bayer-Oglesby, W. J. Gauderman, V. Ilacqua, M. Juhani Jantunen, H. K. Lai, M. Nieuwenhuijsen and N. Künzli, Indoor Time-Microenvironment-Activity Patterns in Seven Regions of Europe, *J. Exposure Sci. Environ. Epidemiol.*, 2007, **17**(2), 170–181.
 6. L. Morawska, A. Afshari, G. N. Bae, G. Buonanno, C. Y. H. Chao, O. Hänninen, W. Hofmann, C. Isaxon, E. R. Jayaratne and P. Pasanen, *et al.*, Indoor Aerosols: From Personal Exposure to Risk Assessment, *Indoor Air*, 2013, **23**(6), 462–487.
 7. P. Kumar, A. N. Skouloudis, M. Bell, M. Viana, M. C. Carotta, G. Biskos and L. Morawska, Real-Time Sensors for Indoor Air Monitoring and Challenges Ahead in Deploying Them to Urban Buildings, *Sci. Total Environ.*, 2016, **560–561**(April), 150–159.
 8. Q. Y. Meng, B. J. Turpin, L. Korn, C. P. Weisel, M. Morandi, S. Colome, J. J. Zhang, T. Stock, D. Spektor and A. Winer, *et al.*, Influence of Ambient (Outdoor) Sources on Residential Indoor and Personal PM_{2.5} Concentrations: Analyses of RIOPA Data, *J. Exposure Anal. Environ. Epidemiol.*, 2005, **15**(1), 17–28.
 9. K. L. Abdullahi, J. M. Delgado-Saborit and R. M. Harrison, Emissions and Indoor Concentrations of Particulate Matter and Its Specific Chemical Components from Cooking: A Review, *Atmos. Environ.*, 2013, **71**, 260–294.
 10. B. C. Singer, H. Destailats, A. T. Hodgson and W. W. Nazaroff, Cleaning Products and Air Fresheners: Emissions and Resulting Concentrations of Glycol Ethers and Terpenoids, *Indoor Air*, 2006, **16**(3), 179–191.
 11. E. Uhde and T. Salthammer, Impact of Reaction Products from Building Materials and Furnishings on Indoor Air Quality—A Review of Recent Advances in Indoor Chemistry, *Atmos. Environ.*, 2007, **41**(15), 3111–3128.
 12. A. Wisthaler and C. J. Weschler, Reactions of Ozone with Human Skin Lipids: Sources of Carbonyls, Dicarbonyls, and Hydroxycarbonyls in Indoor Air, *Proc. Natl. Acad. Sci., U. S. A.*, 2010, **207**(15), 6568–6575.
 13. C. Chen and B. Zhao, Review of Relationship between Indoor and Outdoor Particles: I/O Ratio, Infiltration Factor and Penetration Factor, *Atmos. Environ.*, 2011, **45**(2), 275–288.
 14. G. Hoek, G. Kos, R. Harrison, J. de Hartog, K. Meliefste, H. ten Brink, K. Katsouyanni, A. Karakatsani, M. Lianou and A. Kotronarou, *et al.*, Indoor-Outdoor Relationships of Particle Number and Mass in Four European Cities, *Atmos. Environ.*, 2008, **42**(1), 156–169.
 15. N. C. Jones, C. A. Thornton, D. Mark and R. M. Harrison, Indoor/Outdoor Relationships of Particulate Matter in Domestic Homes with Roadside, Urban and Rural Locations, *Atmos. Environ.*, 2000, **34**, 2603–2612.
 16. C. J. Weschler, Chemistry in Indoor Environments: 20 Years of Research, *Indoor Air*, 2011, **21**(3), 205–218.

17. R. J. Delfino, Epidemiologic Evidence for Asthma and Exposure to Air Toxics: Linkages between Occupational, Indoor, and Community Air Pollution Research, *Environ. Health Perspect.*, 2002, **110**, 573–589.
18. K. Rumchev, M. Soares, Y. Zhao, C. Reid and R. Huxley, The Association between Indoor Air Quality and Adult Blood Pressure Levels in a High-Income Setting, *Int. J. Environ. Res. Public Health*, 2018, **15**(9), 2026.
19. A. J. Cohen, M. Brauer, R. Burnett, H. R. Anderson, J. Frostad, K. Estep, K. Balakrishnan, B. Brunekreef, L. Dandona and R. Dandona, *et al.*, Estimates and 25-Year Trends of the Global Burden of Disease Attributable to Ambient Air Pollution: An Analysis of Data from the Global Burden of Diseases Study 2015, *Lancet*, 2017, **389**(10082), 1907–1918.
20. WHO, *WHO Guidelines for Indoor Air Quality: Selected Pollutants*, 2010.
21. IARC, *Asbestos (Chrysotile, Amosite, Crocidolite, Tremolite, Actinolite, and Anthophyllite) – IARC Monographs Volume 100C*, 2012.
22. WHO, *Handbook on Indoor Radon – A Public Health Perspective*, 2009.
23. US-EPA, *Exposure Factor Handbook, 2011 Edition*, 2011.
24. OECD, *How Do People in the Asia/Pacific Region Spend Their Time? Society at a Glance, Asia/Pacific 2011*, 2011.
25. Eurostat. Database – Eurostat <http://ec.europa.eu/eurostat/data/database> (accessed Jul 11, 2018).
26. B. E. Boor, M. P. Spilak, J. Laverge, A. Novoselac and Y. Xu, Human Exposure to Indoor Air Pollutants in Sleep Microenvironments: A Literature Review, *Build. Environ.*, 2017, **125**, 528–555.
27. S. Bhangar, N. A. Mullen, S. V. Hering, N. M. Kreisberg and W. W. Nazaroff, Ultrafine Particle Concentrations and Exposures in Seven Residences in Northern California, *Indoor Air*, 2011, **21**(2), 132–144.
28. N. A. Mullen, C. Liu, Y. Zhang, S. Wang and W. W. Nazaroff, Ultrafine Particle Concentrations and Exposures in Four High-Rise Beijing Apartments, *Atmos. Environ.*, 2011, **45**(40), 7574–7582.
29. H. M. Hwang, E. K. Park, T. M. Young and B. D. Hammock, Occurrence of Endocrine-Disrupting Chemicals in Indoor Dust, *Sci. Total Environ.*, 2008, **404**(1), 26–35.
30. C. J. Weschler and W. W. Nazaroff, Semivolatile Organic Compounds in Indoor Environments, *Atmos. Environ.*, 2008, **42**(40), 9018–9040.
31. M. P. Spilak, B. E. Boor, A. Novoselac and R. L. Corsi, Impact of Bedding Arrangements, Pillows, and Blankets on Particle Resuspension in the Sleep Microenvironment, *Build. Environ.*, 2014, **81**, 60–68.
32. C. Dimitroulopoulou, Ventilation in European Dwellings: A Review, *Build. Environ.*, 2012, **47**(1), 109–125.
33. N. Canha, J. Lage, S. Candeias, C. Alves and S. M. Almeida, Indoor Air Quality during Sleep under Different Ventilation Patterns, *Atmos. Pollut. Res.*, 2017, **8**(6), 1132–1142.
34. M. A. Torkmahalleh, S. Gorjinezhad, H. S. Unluevcek and P. K. Hopke, Review of Factors Impacting Emission/Concentration of Cooking Generated Particulate Matter, *Sci. Total Environ.*, 2017, **586**, 1046–1056.

35. M. K. Sidhu, K. Ravindra, S. Mor and S. John, Household Air Pollution from Various Types of Rural Kitchens and Its Exposure Assessment, *Sci. Total Environ*, 2017, **586**, 419–429.
36. S. Bonjour, J. Wolf and M. Lahiff, Solid Fuel Use for Household Cooking: Country and Regional Estimates for 1980 – 2010 Solid Fuel Use for Household Cooking: Country and Regional Estimates, *Environ. Health Perspect.*, 2013, **121**, 784–790.
37. A. K. Amegah, R. Quansah and J. J. K. Jaakkola, Household Air Pollution from Solid Fuel Use and Risk of Adverse Pregnancy Outcomes: A Systematic Review and Meta-Analysis of the Empirical Evidence, *PLoS One*, 2014, **9**(12), 1–23.
38. K. H. Kim, S. A. Jahan and E. Kabir, A Review of Diseases Associated with Household Air Pollution Due to the Use of Biomass Fuels, *J. Hazard. Mater.*, 2011, **192**(2), 425–431.
39. M. Orozco-Levi, J. Garcia-Aymerich, J. Villar, A. Ramírez-Sarmiento, J. M. Antó and J. Gea, Wood Smoke Exposure and Risk of Chronic Obstructive Pulmonary Disease, *Eur. Respir. J.*, 2006, **27**(3), 542–546.
40. M. Li, A. Vierkötter, T. Schikowski, A. Hüls, A. Ding, M. S. Matsui, B. Deng, C. Ma, A. Ren and J. Zhang, *et al.*, Epidemiological Evidence That Indoor Air Pollution from Cooking with Solid Fuels Accelerates Skin Aging in Chinese Women, *J. Dermatol. Sci.*, 2014, **79**(2), 148–154.
41. A. K. Pokhrel, K. R. Smith, A. Khalakdina, A. Deuja and M. N. Bates, Case-Control Study of Indoor Cooking Smoke Exposure and Cataract in Nepal and India, *Int. J. Epidemiol.*, 2005, **34**(3), 702–708.
42. K. F. Austin and M. T. Mejia, Household Air Pollution as a Silent Killer: Women's Status and Solid Fuel Use in Developing Nations, *Popul. Environ.*, 2017, **39**(1), 1–25.
43. K. Balakrishnan, P. Ramaswamy, S. Sambandam, G. Thangavel, S. Ghosh, P. Johnson, K. Mukhopadhyay, V. Venugopal and V. Thanasekaraan, Air Pollution from Household Solid Fuel Combustion in India: An Overview of Exposure and Health Related Information to Inform Health Research Priorities, *Glob. Health Action*, 2011, **4**, 5638.
44. T. Li, S. Cao, D. Fan, Y. Zhang, B. Wang, X. Zhao, B. P. Leaderer, G. Shen, Y. Zhang and X. Duan, Household Concentrations and Personal Exposure of PM_{2.5} among Urban Residents Using Different Cooking Fuels, *Sci. Total Environ*, 2016, **548–549**, 6–12.
45. A. K. Pokhrel, M. N. Bates, J. Acharya, P. Valentiner-Branth, R. K. Chandyo, P. S. Shrestha, A. K. Raut and K. R. Smith, PM_{2.5} in Household Kitchens of Bhaktapur, Nepal, Using Four Different Cooking Fuels, *Atmos. Environ.*, 2015, **113**, 159–168.
46. J. M. Tielsch, J. Katz, R. D. Thulasiraj, C. L. Coles, S. Sheeladevi, E. L. Yanik and L. Rahmathullah, Exposure to Indoor Biomass Fuel and Tobacco Smoke and Risk of Adverse Reproductive Outcomes, Mortality, Respiratory Morbidity and Growth among Newborn Infants in South India, *Int. J. Epidemiol.*, 2009, **38**(5), 1351–1363.

47. S. Wang, W. Wei, D. Li, K. Aunan and J. Hao, Air Pollutants in Rural Homes in Guizhou, China – Concentrations, Speciation, and Size Distribution, *Atmos. Environ.*, 2010, **44**(36), 4575–4581.
48. Q. Li, J. Jiang, S. Wang, K. Rumchev, R. Mead-Hunter, L. Morawska and J. Hao, Impacts of Household Coal and Biomass Combustion on Indoor and Ambient Air Quality in China: Current Status and Implication, *Sci. Total Environ*, 2017, **576**, 347–361.
49. K. Balakrishnan, J. Parikh, S. Sankar, R. Padmavathi, K. Srividya, V. Venugopal, S. Prasad and V. L. Pandey, Daily Average Exposures to Respirable Particulate Matter from Combustion of Biomass Fuels in Rural Households of Southern India, *Environ. Health Perspect.*, 2002, **110**(11), 1069–1075.
50. WHO, *Indoor Air Quality Guidelines: Household Fuel Combustion*, 2014.
51. W. Zhang, Z. Lu, Y. Xu, C. Wang, Y. Gu, H. Xu and D. G. Streets, Black Carbon Emissions from Biomass and Coal in Rural China, *Atmos. Environ.*, 2018, **176**(2017), 158–170.
52. D. Pope, N. Bruce, M. Dherani, K. Jagoe and E. Rehfues, Real-Life Effectiveness of ‘Improved’ Stoves and Clean Fuels in Reducing PM_{2.5} and CO: Systematic Review and Meta-Analysis, *Environ. Int.*, 2017, **101**(2), 7–18.
53. Q. Li, J. Jiang, S. Cai, W. Zhou, S. Wang, L. Duan and J. Hao, Gaseous Ammonia Emissions from Coal and Biomass Combustion in Household Stoves with Different Combustion Efficiencies, *Environ. Sci. Technol. Lett.*, 2016, **3**(3), 98–103.
54. O. S. Patange, N. Ramanathan, I. H. Rehman, S. N. Tripathi, A. Misra, A. Kar, E. Graham, L. Singh, R. Bahadur and V. Ramanathan, Reductions in Indoor Black Carbon Concentrations from Improved Biomass Stoves in Rural India, *Environ. Sci. Technol.*, 2015, **49**(7), 4749–4756.
55. F. Klein, S. M. Platt, N. J. Farren, A. Detournay, E. A. Bruns, C. Bozzetti, K. R. Daellenbach, D. Kilic, N. K. Kumar and S. M. Pieber, *et al.*, Characterization of Gas-Phase Organics Using Proton Transfer Reaction Time-of-Flight Mass Spectrometry: Cooking Emissions, *Environ. Sci. Technol.*, 2016, **50**(3), 1243–1250.
56. E. Saito, N. Tanaka, A. Miyazaki and M. Tsuzaki, Concentration and Particle Size Distribution of Polycyclic Aromatic Hydrocarbons Formed by Thermal Cooking, *Food Chem.*, 2014, **153**, 285–291.
57. C. Y. Chao and E. C. Cheng, Source Apportionment of Indoor PM_{2.5} and PM₁₀ in Homes, *Indoor Built Environ.*, 2002, **11**(1), 27–37.
58. G. Buonanno, L. Morawska and L. Stabile, Particle Emission Factors during Cooking Activities, *Atmos. Environ.*, 2009, **43**(20), 3235–3242.
59. M. P. Wan, C. L. Wu, G. N. Sze To, T. C. Chan and C. Y. H. Chao, Ultrafine Particles, and PM_{2.5} Generated from Cooking in Homes, *Atmos. Environ.*, 2011, **45**(34), 6141–6148.
60. D. Massey, A. Kulshrestha, J. Masih and A. Taneja, Seasonal Trends of PM₁₀, PM_{5.0}, PM_{2.5} & PM_{1.0} in Indoor and Outdoor Environments

- of Residential Homes Located in North-Central India, *Build. Environ.*, 2012, 47(1), 223–231.
61. L. Wallace, R. Williams, A. Rea and C. Croghan, Continuous Weeklong Measurements of Personal Exposures and Indoor Concentrations of Fine Particles for 37 Health-Impaired North Carolina Residents for up to Four Seasons, *Atmos. Environ.*, 2006, 40(3), 399–414.
 62. M. Lazaridis, V. Aleksandropoulou, J. Smolik, J. E. Hansen, T. Glytsos, N. Kalogerakis and E. Dahlin, *Physico-Chemical Characterization of Indoor/Outdoor Particulate Matter in Two Residential Houses in Oslo*, Measurements Overview and Physical Properties – Urban-Aerosol Project, Norway, 2006. *Indoor Air* 2006, 16(4), 282–295.
 63. L. A. Wallace, S. J. Emmerich and C. Howard-Reed, Source Strengths of Ultrafine and Fine Particles Due to Cooking with a Gas Stove, *Environ. Sci. Technol.*, 2004, 38(8), 2304–2311.
 64. K. P. Yu, K. R. Yang, Y. C. Chen, J. Y. Gong, Y. P. Chen, H. C. Shih and S. C. Candice Lung, Indoor Air Pollution from Gas Cooking in Five Taiwanese Families, *Build. Environ.*, 2015, 93(P2), 258–266.
 65. N. Canha, J. Lage, C. Galinha, S. Coentro, C. Alves and S. M. Almeida, Impact of Biomass Home Heating, Cooking Styles, and Bread Toasting on the Indoor Air Quality at Portuguese Dwellings: A Case Study, *Atmosphere*, 2018, 9(6), 214.
 66. S. C. Lee, W. Li and L. Y. Chan, Indoor Air Quality at Restaurants with Different Styles of Cooking in Metropolitan Hong Kong, *Sci. Total Environ.*, 2001, 279, 181–193.
 67. S. W. See and R. Balasubramanian, Chemical Characteristics of Fine Particles Emitted from Different Gas Cooking Methods, *Atmos. Environ.*, 2008, 42(39), 8852–8862.
 68. C. He, L. Morawska, J. Hitchins and D. Gilbert, Contribution from Indoor Sources to Particle Number and Mass Concentrations in Residential Houses, *Atmos. Environ.*, 2004, 38(21), 3405–3415.
 69. J. D. McDonald, B. Zielinska, E. M. Fujita, J. C. Sagebiel, J. C. Chow and J. G. Watson, Emissions from Charbroiling and Grilling of Chicken and Beef, *J. Air Waste Manage. Assoc.*, 2003, 53(2), 185–194.
 70. C. Y. Peng, C. H. Lan, P. C. Lin and Y. C. Kuo, Effects of Cooking Method, Cooking Oil, and Food Type on Aldehyde Emissions in Cooking Oil Fumes, *J. Hazard. Mater.*, 2017, 324, 160–167.
 71. J. C. Bordado, J. F. Gomes and P. C. Albuquerque, Exposure to Airborne Ultrafine Particles from Cooking in Portuguese Homes, *J. Air Waste Manage. Assoc.*, 2012, 62(10), 1116–1126.
 72. Q. Zhang, R. H. Gangupomu, D. Ramirez and Y. Zhu, Measurement of Ultrafine Particles and Other Air Pollutants Emitted by Cooking Activities, *Int. J. Environ. Res. Public Health*, 2010, 7(4), 1744–1759.
 73. S. W. See and R. Balasubramanian, Physical Characteristics of Ultrafine Particles Emitted from Different Gas Cooking Methods, *Aerosol Air Qual. Res.*, 2006, 6(1), 82–92.

74. A. C. K. Lai and Y. W. Ho, Spatial Concentration Variation of Cooking-Emitted Particles in a Residential Kitchen, *Build. Environ.*, 2008, **43**(5), 871–876.
75. Z. Xiang, H. Wang, S. Stevanovic, S. Jing, S. Lou, S. Tao, L. Li, J. Liu, M. Yu and L. Wang, Assessing Impacts of Factors on Carbonyl Compounds Emissions Produced from Several Typical Chinese Cooking, *Build. Environ.*, 2017, **125**, 348–355.
76. T. Liu, Z. Wang, D. D. Huang, X. Wang and C. K. Chan, Significant Production of Secondary Organic Aerosol from Emissions of Heated Cooking Oils, *Environ. Sci. Technol. Lett.*, 2018, **5**(1), 32–37.
77. M. A. Torkmahalleh, I. Goldasteh, Y. Zhao, N. M. Udochu, A. Rossner, P. K. Hopke and A. R. Ferro, PM_{2.5} and Ultrafine Particles Emitted during Heating of Commercial Cooking Oils, *Indoor Air*, 2012, **22**(6), 483–491.
78. J. Gao, C. Cao, X. Zhang and Z. Luo, Volume-Based Size Distribution of Accumulation and Coarse Particles (PM_{0.1–10}) from Cooking Fume during Oil Heating, *Build. Environ.*, 2013, **59**, 575–580.
79. R. E. Moran, D. H. Bennett, D. J. Tancredi, X. M. Wu, B. Ritz and I. Hertz-Picciotto, Frequency and Longitudinal Trends of Household Care Product Use, *Atmos. Environ.*, 2012, **55**, 417–424.
80. R. L. Corsi, J. A. Siegel and C. Chiang, Particle Resuspension during the Use of Vacuum Cleaners on Residential Carpet, *J. Occup. Environ. Hyg.*, 2008, **5**(4), 232–238.
81. L. D. Montoya and L. M. Hildemann, Size Distributions and Height Variations of Airborne Particulate Matter and Cat Allergen Indoors Immediately Following Dust-Disturbing Activities, *J. Aerosol Sci.*, 2005, **36**(5–6), 735–749.
82. R. D. Lewis, K. H. Ong, B. Emo, J. Kennedy, J. Kesavan and M. Elliot, Resuspension of House Dust and Allergens during Walking and Vacuum Cleaning, *J. Occup. Environ. Hyg.*, 2018, **15**(3), 235–245.
83. E. Diapouli, A. Chaloulakou and N. Spyrellis, Levels of Ultrafine Particles in Different Microenvironments—Implications to Children Exposure, *Sci. Total Environ.*, 2007, **388**(1–3), 128–136.
84. S. Langer, J. Moldanová, K. Arrhenius, E. Ljungström and L. Ekberg, Ultrafine Particles Produced by Ozone/Limonene Reactions in Indoor Air under Low/Closed Ventilation Conditions, *Atmos. Environ.*, 2008, **42**(18), 4149–4159.
85. G. Sarwar, D. A. Olson, R. L. Corsi and C. J. Weschler, Indoor Fine Particles: The Role of Terpene Emissions from Consumer Products, *J. Air Waste Manage. Assoc.*, 2004, **54**(3), 367–377.
86. D. W. Layton and P. I. Beamer, Migration of Contaminated Soil and Airborne Particulates to Indoor Dust, *Environ. Sci. Technol.*, 2009, **43**(21), 8199–8205.

87. C. M. Long, H. H. Suh and P. Koutrakis, Characterization of Indoor Particle Sources Using Continuous Mass and Size Monitors, *J. Air Waste Manage. Assoc.*, 2000, **50**(7), 1236–1250.
88. A. R. Ferro, R. J. Kopperud and L. M. Hildemann, Source Strengths for Indoor Human Activities That Resuspend Particulate Matter, *Environ. Sci. Technol.*, 2004, **38**(6), 1759–1764.
89. T. V. Vu, J. Ondracek, V. Zdimal, J. Schwarz, J. M. Delgado-Saborit and R. M. Harrison, Physical Properties and Lung Deposition of Particles Emitted from Five Major Indoor Sources, *Air Qual., Atmos. Health*, 2017, **10**(1), 1–14.
90. W. Szymczak, N. Menzel and L. Keck, Emission of Ultrafine Copper Particles by Universal Motors Controlled by Phase Angle Modulation, *J. Aerosol Sci.*, 2007, **38**(5), 520–531.
91. P. Wolkoff, S. T. Larsen, M. Hammer, V. Kofoed-Sørensen, P. A. Clausen and G. D. Nielsen, Human Reference Values for Acute Airway Effects of Five Common Ozone-Initiated Terpene Reaction Products in Indoor Air, *Toxicol. Lett.*, 2013, **216**(1), 54–64.
92. M. Medina-Ramón, J. P. Zock, M. Kogevinas, J. Sunyer, Y. Torralba, A. Borrell, F. Burgos and J. M. Antó, Asthma, Chronic Bronchitis, and Exposure to Irritant Agents in Occupational Domestic Cleaning: A Nested Case-Control Study, *Occup. Environ. Med.*, 2005, **62**(9), 598–606.
93. S. Quirce and P. Barranco, Cleaning Agents and Asthma, *J. Invest. Allergol. Clin. Immunol.*, 2010, **20**(7), 542–550.
94. N. Carslaw, A Mechanistic Study of Limonene Oxidation Products and Pathways Following Cleaning Activities, *Atmos. Environ.*, 2013, **80**, 507–513.
95. C. J. Weschler and H. C. Shields, Indoor Ozone/Terpene Reactions as a Source of Indoor Particles, *Atmos. Environ.*, 1999, **33**, 2301–2312.
96. R. D. Edwards, J. Jurvelin, K. Koistinen, K. Saarela and M. Jantunen, VOC Source Identification from Personal and Residential Indoor, Outdoor and Workplace Microenvironment Samples in EXPOLIS-Helsinki, Finland, *Atmos. Environ.*, 2001, **35**(28), 4829–4841.
97. M. A. Bari, W. B. Kindzierski, A. J. Wheeler, M. È. Héroux and L. A. Wallace, Source Apportionment of Indoor and Outdoor Volatile Organic Compounds at Homes in Edmonton, Canada, *Build. Environ.*, 2015, **90**, 114–124.
98. A. W. Nørgaard, V. Kofoed-Sørensen, C. Mandin, G. Ventura, R. Mabilia, E. Perreca, A. Cattaneo, A. Spinazzè, V. G. Mihucz and T. Szigeti, *et al.*, Ozone-Initiated Terpene Reaction Products in Five European Offices: Replacement of a Floor Cleaning Agent, *Environ. Sci. Technol.*, 2014, **48**(22), 13331–13339.
99. T. Salthammer, T. Schripp, S. Wientzek and M. Wensing, Impact of Operating Wood-Burning Fireplace Ovens on Indoor Air Quality, *Chemosphere*, 2014, **103**, 205–211.
100. H. J. Moriske, M. Drews, G. Ebert, G. Menk, C. Scheller, M. Schöndube and L. Konieczny, Indoor Air Pollution by Different Heating Systems:

- Coal Burning, Open Fireplace and Central Heating, *Toxicol. Lett.*, 1996, **88**(1–3), 349–354.
101. G. J. Raw, S. K. D. Coward, V. M. Brown and D. R. Crump, Exposure to Air Pollutants in English Homes, *J. Exposure Anal. Environ. Epidemiol.*, 2004, **14**, S85–S94.
 102. P. Gustafson, C. Östman and G. Sällsten, Indoor Levels of Polycyclic Aromatic Hydrocarbons in Homes with or without Wood Burning for Heating, *Environ. Sci. Technol.*, 2008, **42**(14), 5074–5080.
 103. L. Wallace, Indoor Particles: A Review, *J. Air Waste Manage. Assoc.*, 1996, **46**(2), 98–126.
 104. IARC, *Tobacco Smoke and Involuntary Smok-Ing. IARCmonographs on the Evaluation of Carcinogenic Risks to Humans*, vol. 83, Lyon, 2004.
 105. R. Brownson, M. Eriksen, R. Davis and K. Warner, Environmental Tobacco Smoke: Health Effects and Policies to Reduce Exposure – ProQuest, *Annu. Rev. Public Health*, 1997, **18**, 163–185.
 106. G. Löfroth, R. M. Burton, L. Forehand, S. K. Hammond, R. L. Seila, R. B. Zweidinger and J. Lewtas, Characterization of Environmental Tobacco Smoke, *Environ. Sci. Technol.*, 1989, **23**(5), 610–614.
 107. G. Löfroth, Environmental Tobacco Smoke: Overview of Chemical Composition and Genotoxic Components, *Mutat. Res. Toxicol.*, 1989, **222**(2), 73–80.
 108. K. Jöckel, H. Pohlabeln, W. Ahrens and M. Krauss, Environmental Tobacco Smoke and Lung Cancer, *Epidemiol.*, 1998, **9**(6), 672–675.
 109. WHO, *WHO Global Report on Trends in Prevalence of Tobacco Smoking 2015*, Geneva, 2015.
 110. WHO, *WHO Global Report on Trends in Prevalence of Tobacco Smoking 2000–2025, 2nd Edn*, Geneva, 2018.
 111. M. Stranger, S. S. Potgieter-Vermaak and R. Van Grieken, Comparative Overview of Indoor Air Quality in Antwerp, Belgium, *Environ. Int.*, 2007, **33**(6), 789–797.
 112. H. K. Lai, M. Kendall, H. Ferrier, I. Lindup, S. Alm, O. Hänninen, M. Jantunen, P. Mathys, R. Colville and M. R. Ashmore, *et al.*, Personal Exposures and Microenvironment Concentrations of PM_{2.5}, VOC, NO₂ and CO in Oxford, UK, *Atmos. Environ.*, 2004, **38**(37), 6399–6410.
 113. D. H. Tsai, Y. H. Wu and C. C. Chan, Comparisons of Commuter's Exposure to Particulate Matters While Using Different Transportation Modes, *Sci. Total Environ.*, 2008, **405**(1–3), 71–77.
 114. K. Slezakova, J. C. M. Pires, F. G. Martins, M. C. Pereira and M. C. Alvim-Ferraz, Identification of Tobacco Smoke Components in Indoor Breathable Particles by SEM-EDS, *Atmos. Environ.*, 2011, **45**(4), 863–872.
 115. A. Böhländt, R. Schierl, J. Diemer, C. Koch, G. Bolte, M. Kiranoglu, H. Fromme and D. Nowak, High Concentrations of Cadmium, Cerium and Lanthanum in Indoor Air Due to Environmental Tobacco Smoke, *Sci. Total Environ.*, 2012, **414**, 738–741.

116. D. L. Heavner, W. T. Morgan and M. W. Ogden, Determination of Volatile Organic Compounds and ETS Apportionment in 49 Homes, *Environ. Int.*, 1995, **21**(1), 3–21.
117. B. C. Singer, A. T. Hodgson, K. S. Guevarra, E. L. Hawley and W. W. Nazaroff, Gas-Phase Organics in Environmental Tobacco Smoke. 1. Effects of Smoking Rate, Ventilation, and Furnishing Level on Emission Factors, *Environ. Sci. Technol.*, 2002, **36**(5), 846–853.
118. B. C. Singer, A. T. Hodgson and W. W. Nazaroff, Gas-Phase Organics in Environmental Tobacco Smoke: 2. Exposure-Relevant Emission Factors and Indirect Exposures from Habitual Smoking, *Atmos. Environ.*, 2003, **37**, 5551–5561.
119. S. F. Schick, K. F. Farraro, C. Perrino, M. Sleiman, G. van de Vossenberg, M. P. Trinh, S. Katharine Hammond, B. M. Jenkins and J. Balmes, Thirdhand Cigarette Smoke in an Experimental Chamber: Evidence of Surface Deposition of Nicotine, Nitrosamines and Polycyclic Aromatic Hydrocarbons and de Novo Formation of NNK, *Tob. Control*, 2014, **23**(2), 152–159.
120. G. E. Matt, P. J. E. Quintana, J. M. Zakarian, A. L. Fortmann, D. A. Chatfield, E. Hoh, A. M. Uribe and M. F. Hovell, When Smokers Move out and Non-Smokers Move in: Residential Thirdhand Smoke Pollution and Exposure, *Tob. Control*, 2011, **20**(1), 1–8.
121. A. A. Ruprecht, C. De Marco, P. Pozzi, E. Munarini, R. Mazza, G. Angellotti, F. Turla and R. Boffi, Comparison between Particulate Matter and Ultrafine Particle Emission by Electronic and Normal Cigarettes in Real-Life Conditions, *Tumori*, 2014, **100**(1), 24–27.
122. C. Protano, M. Manigrasso, P. Avino and M. Vitali, Second-Hand Smoke Generated by Combustion and Electronic Smoking Devices Used in Real Scenarios: Ultrafine Particle Pollution and Age-Related Dose Assessment, *Environ. Int.*, 2017, **107**(May), 190–195.
123. K. D. Volesky, A. Maki, C. Scherf, L. Watson, K. Van Ryswyk, B. Fraser, S. A. Weichenthal, E. Cassol and P. J. Villeneuve, The Influence of Three E-Cigarette Models on Indoor Fine and Ultrafine Particulate Matter Concentrations under Real-World Conditions, *Environ. Pollut.*, 2018, **243**, 882–889.
124. J. Qian, J. Peccia and A. R. Ferro, Walking-Induced Particle Resuspension in Indoor Environments, *Atmos. Environ.*, 2014, **89**, 464–481.
125. C. J. Weschler, Roles of the Human Occupant in Indoor Chemistry, *Indoor Air*, 2016, **26**, 6–24.
126. T. L. Thatcher and D. W. Layton, Deposition, Resuspension, and Penetration of Particles within a Residence, *Atmos. Environ.*, 1995, **29**(13), 1487–1497.
127. J. Qian and A. R. Ferro, Resuspension of Dust Particles in a Chamber and Associated Environmental Factors, *Aerosol Sci. Technol.*, 2008, **42**(7), 566–578.
128. N. Serfozo, S. E. Chatoutsidou and M. Lazaridis, The Effect of Particle Resuspension during Walking Activity to PM10mass and Number

- Concentrations in an Indoor Microenvironment, *Build. Environ.*, 2014, **82**, 180–189.
129. Y. Tian, K. Sul, J. Qian, S. Mondal and A. R. Ferro, A Comparative Study of Walking-Induced Dust Resuspension Using a Consistent Test Mechanism, *Indoor Air*, 2014, **24**(6), 592–603.
 130. E. Karlsson, T. Berglund, M. Strömqvist, M. Nordstrand and I. Fångmark, The Effect of Resuspension Caused by Human Activities on the Indoor Concentration of Biological Aerosols, *J. Aerosol Sci.*, 1999, **30**, S737–S738.
 131. A. R. Ferro, R. J. Kopperud and L. M. Hildemann, Elevated Personal Exposure to Particulate Matter from Human Activities in a Residence, *J. Exposure Anal. Environ. Epidemiol.*, 2004, **14**, S34–S40.
 132. R. M. Maertens, J. Bailey and P. A. White, The Mutagenic Hazards of Settled House Dust: A Review, *Mutat. Res. – Rev. Mutat. Res.*, 2004, **567**(2–3 Spec. Iss.), 401–425.
 133. D. Hospodsky, J. Qian, W. W. Nazaroff, N. Yamamoto, K. Bibby, H. Rismani-Yazdi and J. Peccia, Human Occupancy as a Source of Indoor Airborne Bacteria, *PLoS One*, 2012, **7**(4), e34867.
 134. C. Gomes, J. Freihaut and W. Bahnfleth, Resuspension of Allergen-Containing Particles under Mechanical and Aerodynamic Disturbances from Human Walking, *Atmos. Environ.*, 2007, **41**(25), 5257–5270.
 135. L. S. Pandrangi and G. C. Morrison, Ozone Interactions with Human Hair: Ozone Uptake Rates and Product Formation, *Atmos. Environ.*, 2008, **42**(20), 5079–5089.
 136. Q.-T. Liu, R. Chen, B. E. McCarry, M. L. Diamond and B. Bahavar, Characterization of Polar Organic Compounds in the Organic Film on Indoor and Outdoor Glass Windows, *Environ. Sci. Technol.*, 2003, **37**(11), 2340–2349.
 137. P. Mochalski, K. Unterkofler, H. Hinterhuber and A. Amann, Monitoring of Selected Skin-Borne Volatile Markers of Entrapped Humans by Selective Reagent Ionization Time of Flight Mass Spectrometry in NO^+ mode, *Anal. Chem.*, 2014, **86**(8), 3915–3923.
 138. R. L. Corsi, J. Siegel, A. Karamalegos, H. Simon and G. C. Morrison, Personal Reactive Clouds: Introducing the Concept of near-Head Chemistry, *Atmos. Environ.*, 2007, **41**(15), 3161–3165.
 139. W. W. Nazaroff and A. H. Goldstein, Indoor Chemistry: Research Opportunities and Challenges, *Indoor Air*, 2015, **25**(4), 357–361.
 140. C. G. Bornehag, B. Lundgren, C. J. Weschler, T. Sigsgaard, L. Hagerhed-Engman and J. Sundell, Phthalates in Indoor Dust and Their Association with Building Characteristics, *Environ. Health Perspect.*, 2005, **113**(10), 1399–1404.
 141. I. Liagkouridis, I. T. Cousins and A. P. Cousins, Emissions and Fate of Brominated Flame Retardants in the Indoor Environment: A Critical Review of Modelling Approaches, *Sci. Total Environ.*, 2014, **491–492**, 87–99.
 142. Y. Xu, Z. Liu, J. Park, P. A. Clausen, J. L. Benning and J. C. Little, Measuring and Predicting the Emission Rate of Phthalate Plasticizer

- from Vinyl Flooring in a Specially-Designed Chamber, *Environ. Sci. Technol.*, 2012, **46**(22), 12534–12541.
143. P. A. Clausen, Y. Xu, V. Kofoed-Sørensen, J. C. Little and P. Wolkoff, The Influence of Humidity on the Emission of Di-(2-Ethylhexyl) Phthalate (DEHP) from Vinyl Flooring in the Emission Cell “FLEC.”, *Atmos. Environ.*, 2007, **41**(15), 3217–3224.
144. M. Kruza, A. C. Lewis, G. C. Morrison and N. Carslaw, Impact of Surface Ozone Interactions on Indoor Air Chemistry: A Modeling Study, *Indoor Air*, 2017, **27**(5), 1001–1011.
145. S. Kim, J. A. Kim, H. J. Kim and S. Do Kim, Determination of Formaldehyde and TVOC Emission Factor from Wood-Based Composites by Small Chamber Method, *Polym. Test.*, 2006, **25**(5), 605–614.
146. H. Wang and G. C. Morrison, Ozone-Initiated Secondary Emission Rates of Aldehydes from Indoor Surfaces in Four Homes, *Environ. Sci. Technol.*, 2006, **40**(17), 5263–5268.
147. G. C. Morrison and W. W. Nazaroff, The Rate of Ozone Uptake on Carpet: Mathematical Modeling, *Atmos. Environ.*, 2002, **36**(11), 1749–1756.
148. W. Liang, S. Yang and X. Yang, Long-Term Formaldehyde Emissions from Medium-Density Fiberboard in a Full-Scale Experimental Room: Emission Characteristics and the Effects of Temperature and Humidity, *Environ. Sci. Technol.*, 2015, **49**(17), 10349–10356.
149. C. Liu, Y. Zhang, J. L. Benning and J. C. Little, The Effect of Ventilation on Indoor Exposure to Semivolatile Organic Compounds, *Indoor Air*, 2015, **25**(3), 285–296.
150. Y. H. Cheng, C. C. Lin and S. C. Hsu, Comparison of Conventional and Green Building Materials in Respect of VOC Emissions and Ozone Impact on Secondary Carbonyl Emissions, *Buuld. Environ.*, 2015, **87**, 274–282.
151. T. Salthammer, E. Uhde, T. Schripp, A. Schieweck, L. Morawska, M. Mazaheri, S. Clifford, C. He, G. Buonanno and X. Querol, *et al.*, Children’s Well-Being at Schools: Impact of Climatic Conditions and Air Pollution, *Environ. Int.*, 2016, **94**, 196–210.
152. Z. Bakó-Biró, D. J. Clements-Croome, N. Kochhar, H. B. Awbi and M. J. Williams, Ventilation Rates in Schools and Pupils’ Performance, *Buuld. Environ.*, 2012, **48**(1), 215–223.
153. O. Seppänen, W. J. Fisk and Q. H. Lei, Ventilation and Performance in Office Work, *Indoor Air*, 2006, **16**(1), 28–36.
154. I. Rivas, X. Querol, J. Wright and J. Sunyer, How to Protect School Children from the Neurodevelopmental Harms of Air Pollution by Interventions in the School Environment in the Urban Context, *Environ. Int.*, 2018, **121**(June), 199–206.
155. H. Salonen, T. Salthammer and L. Morawska, Human Exposure to Ozone in School and Office Indoor Environments, *Environ. Int.*, 2018, **119**(July), 503–514.
156. M. Luoma and S. A. Batterman, Characterization of Particulate Emissions from Occupant Activities in Offices, *Indoor Air*, 2001, **11**(1), 35–48.

157. G. Sangiorgi, L. Ferrero, B. S. Ferrini, C. Lo Porto, M. G. Perrone, R. Zangrando, A. Gambaro, Z. Lazzati and E. Bolzacchini, Indoor Airborne Particle Sources and Semi-Volatile Partitioning Effect of Outdoor Fine PM in Offices, *Atmos. Environ.*, 2013, **65**, 205–214.
158. T. Szigeti, C. Dunster, A. Cattaneo, A. Spinazzè, C. Mandin, E. Le Ponner, E. de Oliveira Fernandes, G. Ventura, D. E. Saraga and I. A. Sakellaris, *et al.*, Spatial and Temporal Variation of Particulate Matter Characteristics within Office Buildings – The OFFICAIR Study, *Sci. Total Environ*, 2017, **587–588**, 59–67.
159. L. Morawska, G. A. Ayoko, G. N. Bae, G. Buonanno, C. Y. H. Chao, S. Clifford, S. C. Fu, O. Hänninen, C. He and C. Isaxon, *et al.*, Airborne Particles in Indoor Environment of Homes, Schools, Offices and Aged Care Facilities: The Main Routes of Exposure, *Environ. Int.*, 2017, **108**(August), 75–83.
160. A. C. Terry, N. Carslaw, M. Ashmore, S. Dimitroulopoulou and D. C. Carslaw, Occupant Exposure to Indoor Air Pollutants in Modern European Offices: An Integrated Modelling Approach, *Atmos. Environ.*, 2014, **82**, 9–16.
161. M. Braniš, P. Rezáčová and M. Domasová, The Effect of Outdoor Air and Indoor Human Activity on Mass Concentrations of PM₁₀, PM_{2.5}, and PM₁ in a Classroom, *Environ. Res.*, 2005, **99**(2), 143–149.
162. P. V. Dorizas, M.-N. Assimakopoulos, C. Helmis and M. Santamouris, An Integrated Evaluation Study of the Ventilation Rate, the Exposure and the Indoor Air Quality in Naturally Ventilated Classrooms in the Mediterranean Region during Spring, *Sci. Total Environ*, 2015, **502C**, 557–570.
163. R. Jan, R. Roy, S. Yadav and P. G. Satsangi, Exposure Assessment of Children to Particulate Matter and Gaseous Species in School Environments of Pune, *India. Build. Environ.*, 2017, **111**, 207–217.
164. S. M. Almeida, N. Canha, A. Silva, M. D. C. Freitas, P. Pegas, C. Alves, M. Evtugina and C. A. Pio, Children Exposure to Atmospheric Particles in Indoor of Lisbon Primary Schools, *Atmos. Environ.*, 2011, **45**(40), 7594–7599.
165. H. Fromme, D. Twardella, S. Dietrich, D. Heitmann, R. Schierl, B. Liebl and H. Rüdén, Particulate Matter in the Indoor Air of Classrooms—exploratory Results from Munich and Surrounding Area, *Atmos. Environ.*, 2007, **41**(4), 854–866.
166. P. T. B. S. Branco, M. C. M. Alvim-Ferraz, F. G. Martins and S. I. V. Sousa, Indoor Air Quality in Urban Nurseries at Porto City: Particulate Matter Assessment, *Atmos. Environ.*, 2014, **84**, 133–143.
167. P. Blondeau, V. Iordache, O. Poupard, D. Genin and F. Allard, Relationship between Outdoor and Indoor Air Quality in Eight French Schools, *Indoor Air*, 2005, **15**(1), 2–12.
168. I. Rivas, M. Viana, T. Moreno, M. Pandolfi, F. Amato, C. Reche, L. Bouso, M. Álvarez-Pedrerol, A. Alastuey and J. Sunyer, *et al.*,

- Child Exposure to Indoor and Outdoor Air Pollutants in Schools in Barcelona, Spain, *Environ. Int.*, 2014, **69**, 200–212.
169. L. Zhang, C. Guo, X. Jia, H. Xu, M. Pan, D. Xu, X. Shen, J. Zhang, J. Tan and H. Qian, *et al.*, Personal Exposure Measurements of School-Children to Fine Particulate Matter (PM 2.5) in Winter of 2013, Shanghai, China, *PLoS One*, 2018, **13**, 1–17.
170. F. Amato, I. Rivas, M. Viana, T. Moreno, L. Bouso, C. Reche, M. Alvarez-Pedrerol, A. Alastuey, J. Sunyer and X. Querol, Sources of Indoor and Outdoor PM2.5 Concentrations in Primary Schools, *Sci. Total Environ*, 2014, **490**, 757–765.
171. G. Kalaiarasan, R. M. Balakrishnan, N. A. Sethunath and S. Manoharan, Source Apportionment of PM2.5 Particles: Influence of Outdoor Particles on Indoor Environment of Schools Using Chemical Mass Balance, *Aerosol Air Qual. Res.*, 2017, **17**(2), 616–625.
172. J. Qian, D. Hospodsky, N. Yamamoto, W. W. Nazaroff and J. Peccia, Size-Resolved Emission Rates of Airborne Bacteria and Fungi in an Occupied Classroom, *Indoor Air*, 2012, **22**(4), 339–351.
173. K. Fox, E. Castanha, A. Fox, C. Feigley and D. Salzberg, Human K10 Epithelial Keratin Is the Most Abundant Protein in Airborne Dust of Both Occupied and Unoccupied School Rooms, *J. Environ. Monit.*, 2008, **10**(1), 55–59.
174. A. Fox, W. Harley, C. Feigley, D. Salzberg, C. Toole, A. Sebastian and L. Larsson, Large Particles Are Responsible for Elevated Bacterial Marker Levels in School Air upon Occupation, *J. Environ. Monit.*, 2005, **7**(5), 450–456.
175. H. Destailats, R. L. Maddalena, B. C. Singer, A. T. Hodgson and T. E. McKone, Indoor Pollutants Emitted by Office Equipment: A Review of Reported Data and Information Needs, *Atmos. Environ.*, 2008, **42**(7), 1371–1388.
176. M. Wensing, G. Pinz, M. Bednarek, T. Schripp, E. Uhde and T. Salthammer, Particle Measurement of Hardcopy Devices, in *Proceedings of the Healthy Building 2006 Conference*, Lisbon, Portugal, 2006, vol. II, pp. 461–464.
177. C. He, L. Morawska and L. Taplin, Particle Emission Characteristics of Office Printers, *Environ. Sci. Technol.*, 2007, **41**(17), 6039–6045.
178. L. Morawska, C. He, G. Johnson, R. Jayaratne, T. Salthammer, H. Wang, E. Uhde, T. Bostrom, R. Modini and G. Ayoko, *et al.*, An Investigation into the Characteristics and Formation Mechanisms of Particles Originating from the Operation of Laser Printers, *Environ. Sci. Technol.*, 2009, **43**, 1015–1022.
179. C. W. Lee and D. J. Hsu, Measurements of Fine and Ultrafine Particles Formation in Photocopy Centers in Taiwan, *Atmos. Environ.*, 2007, **41**(31), 6598–6609.
180. P. McGarry, L. Morawska, C. He, R. Jayaratne, M. Falk, Q. Tran and H. Wang, Exposure to Particles from Laser Printers Operating within Office Workplaces, *Environ. Sci. Technol.*, 2011, **45**(15), 6444–6452.

181. S. Kemmlein, O. Hahn and O. Jann, Emissions of Organophosphate and Brominated Flame Retardants from Selected Consumer Products and Building Materials, *Atmos. Environ.*, 2003, 37(39–40), 5485–5493.
182. W. R. Ott, L. A. Wallace, J. M. McAteer and L. M. Hildemann, Fine and Ultrafine Particle Exposures on 73 Trips by Car to 65 Non-Smoking Restaurants in the San Francisco Bay Area, *Indoor Air*, 2017, 27(1), 205–217.
183. S. O. Baek, Y. S. Kim and R. Perry, Indoor Air Quality in Homes, Offices and Restaurants in Korean Urban Areas – Indoor/Outdoor Relationships, *Atmos. Environ.*, 1997, 31(4), 529–544.
184. F. Shamo, T. Wilson, J. Kiley and J. Repace, Assessing the Effect of Michigan’s Smoke-Free Law on Air Quality inside Restaurants and Casinos: A before-and-after Observational Study, *BMJ Open*, 2015, 5(7), 1–6.
185. R. D. Williams and J. L. Evans, Continued Reduction of Particulate Matter in Bars Six Months After Adoption of a Smoke-Free Ordinance, *J. Environ. Health*, 2018, 81(1), 8–15.
186. J. Pey, B. L. van Drooge, A. Ripoll, T. Moreno, J. O. Grimalt, X. Querol and A. Alastuey, An Evaluation of Mass, Number Concentration, Chemical Composition and Types of Particles in a Cafeteria before and after the Passage of an Antismoking Law, *Particuology*, 2013, 11(5), 527–532.
187. D. E. Rollison, K. J. Helzlsouer and S. M. Pinney, Personal Hair Dye Use and Cancer: A Systematic Literature Review and Evaluation of Exposure Assessment in Studies Published since 1992, *J. Toxicol. Environ. Health, Part B*, 2006, 9, 413–439.
188. K. Golka, P. Heitmann, F. Gieseler, J. Hodzic, N. Masche, H. M. Bolt and F. Geller, Elevated Bladder Cancer Risk Due to Colorants – A Statewide Case-Control Study in North Rhine-Westphalia, Germany, *J. Toxicol. Environ. Health – Part A Curr. Issues*, 2008, 71(13–14), 851–855.
189. A. Tsigonia, A. Lagoudi, S. Chandrinou, A. Linos, N. Evlogias and E. C. Alexopoulos, Indoor Air in Beauty Salons and Occupational Health Exposure of Cosmetologists to Chemical Substances, *Int. J. Environ. Res. Public Health*, 2010, 7(1), 314–324.
190. E. Ronda, B. E. Hollund and B. E. Moen, Airborne Exposure to Chemical Substances in Hairdresser Salons, *Environ. Monit. Assess.*, 2009, 153(1–4), 83–93.
191. A. Mandiracioglu, S. Kose, A. Gozaydin, M. Turken and L. Kuzucu, Occupational Health Risks of Barbers and Coiffeurs in Izmir, *Indian J. Occup. Environ. Med.*, 2009, 13(2), 92.
192. A. Mendes, J. Madureira, P. Neves, C. Carvalhais, B. Laffon and J. P. Teixeira, Chemical Exposure and Occupational Symptoms among Portuguese Hairdressers, *J. Toxicol. Environ. Health – Part A Curr. Issues*, 2011, 74(15–16), 993–1000.
193. D. Oikawa, W. Takeuchi, S. Murata, K. Takahashi and Y. Sekine, Measurement of Concentrations of Thioglycolic Acid, Dithiodiglycolic

- Acid and Ammonia in Indoor Air of a Beauty Salon, *J. Occup. Health*, 2012, **54**(5), 370–375.
194. C. Roelofs and T. Do, Exposure Assessment in Nail Salons: An Indoor Air Approach, *ISRN Public Health*, 2012, **2012**, 1–7.
195. C. Roelofs, L. S. Azaroff, C. Holcroft, H. Nguyen and T. Doan, Results from a Community-Based Occupational Health Survey of Vietnamese-American Nail Salon Workers, *J. Immigr. Minor. Health*, 2008, **10**(4), 353–361.
196. V. M. Alaves, D. K. Sleeth, M. S. Thiese and R. R. Larson, Characterization of Indoor Air Contaminants in a Randomly Selected Set of Commercial Nail Salons in Salt Lake County, Utah, USA, *Int. J. Environ. Health Res.*, 2013, **23**(5), 419–433.
197. L. J. Goldin, L. Ansher, A. Berlin, J. Cheng, D. Kanopkin, A. Khazan, M. Kisivuli, M. Lortie, E. B. Peterson and L. Pohl, *et al.*, Indoor Air Quality Survey of Nail Salons in Boston, *J. Immigr. Minor. Health*, 2014, **16**(3), 508–514.
198. D. A. Marlow, T. Looney and S. Reutman, *In-Depth Survey Report: An Evaluation of Local Exhaust Ventilation Systems for Controlling Hazardous Exposures in Nail Salons*, Cincinnati, OH, USA, 2012.
199. K. Slezakova, C. Peixoto, M. Oliveira, C. Delerue-Matos, M. Pereira, C. do and S. Morais, Indoor Particulate Pollution in Fitness Centres with Emphasis on Ultrafine Particles, *Environ. Pollut.*, 2018, **233**, 180–193.
200. S. Weinbruch, T. Dirsch, K. Kandler, M. Ebert, G. Heimbürger and F. Hohenwarter, Reducing Dust Exposure in Indoor Climbing Gyms, *J. Environ. Monit.*, 2012, **14**(8), 2114–2120.
201. S. Brasche and W. Bischof, Daily Time Spent Indoors in German Homes – Baseline Data for the Assessment of Indoor Exposure of German Occupants, *Int. J. Hyg. Environ. Health*, 2005, **208**(4), 247–253.