

# Chapter 2

## Indoor Air Pollution

Ian Colbeck and Zaheer Ahmad Nasir

**Abstract** Population exposure to various air pollutants is likely to be higher in the indoor micro-environment than outdoors due to the amount of time people spend there. Consequently, indoor air quality has drawn considerable attention in recent years. There are noticeable differences in the types and strength of air pollution sources across the globe and they are closely linked to socio-economic developments. Typically higher indoor concentrations occur in developing rather than developed countries. The types, concentration, and sources of indoor air pollutants vary considerably from one micro-environment to another. Hence, an understanding of the concentration of pollutants in different micro-environments is of great importance for improving exposure estimates and, in turn, for developing efficient control strategies to reduce human exposure and health risk.

### 2.1 Introduction

We often assume that air pollution is a modern phenomenon, and that it has become worse in recent times. However since the dawn of history, mankind has been burning biological and fossil fuel to produce heat. The walls of caves, inhabited millennia ago, are covered with layers of soot and many of the lungs of mummified bodies from Palaeolithic times have a black tone (McNeill, 2001). Brimblecombe (1987) has suggested that the high incidence of sinusitis in Anglo-Saxon Britain was related to a build-up of smoke in their poorly ventilated huts.

Air pollution problems in ancient Rome appear in many documents (Hughes 1993; Makra and Brimblecombe 2004). As residents of what had become the largest city in the world, ancient Romans were well aware of the problem of air pollution. They called it *gravioris caeli* (heavy heaven) or *infamis aer* (infamous air).

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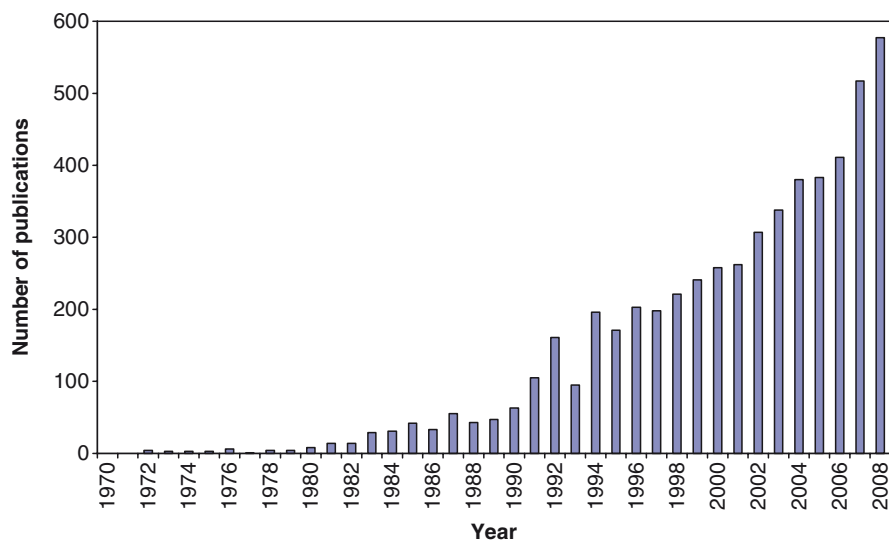
*“The smoke, the wealth, the noise of Rome...”* held no charms for the Roman poet Horace (65 BC–AD 8) who described the blackening of buildings by smoke (Costa 1997).

Indoor air pollution, and in particular particulate matter, was also a significant problem. Animal and vegetable oils were burned to provide artificial light and wood, vegetal materials and animal dung was used to heat their homes. All these materials produced high quantities of soot and toxic gases. Capasso (2000) examined skeletons buried by the volcanic eruptions of Vesuvius and found evidence of inflammation of the pulmonary tract. Histological assessment of the lungs of ancient human mummies has shown that anthracosis was a regular disorder in many ancient societies, including the Egyptian, Peruvian and Aleutian.

In the Bible Leviticus 14, 34–57 indicates that people were aware that residing in damp buildings was dangerous to their health. The remedial action was rather severe:

“and he shall break down the house, the stones of it, and the timber thereof, and all the mortar of the house; and he shall carry them forth out of the city into an unclean place”.

In the early seventeenth century that it was recognised that *“want of ventilation”* resulted in increased rates of infectious disease and in the mid-nineteenth century Griscom (1848) highlighted the impact of poor ventilation on health stating that *“deficient ventilation ... (is) more fatal than all other causes put together.”* In the following years a number of investigations were conducted on the effect of ventilation, carbon dioxide concentrations on disease (Sundell 2004). For example Carnelly et al. (1887) measured CO<sub>2</sub>, mould and bacteria and total organic material in public housing, schools, factories and a hospital in Dundee. They found that concentrations of all these parameters were proportional to number of occupants per room. They went on to recommend a general level of 600 CFU m<sup>-3</sup> as a limit value for human exposure to bioaerosols.



**Fig.2.1** Number of papers published per year since 1970 with ‘indoor air’ as the topic (Bibliographic search of Web of Knowledge [wok.mimas.ac.uk] – accessed 16 January 2009)

It wasn't until the 1960s the papers on indoor air pollution began to appear on a regular basis. The early work was related to radon and tobacco smoke before extending to formaldehyde in the early 1970s, house dust mites and sick building syndrome in the late 1970s, and allergies during the 1990s (Weschler 2009). Figure 2.1 shows the number of papers published each year from 1970 onwards based on a bibliographic search of Web of Science using indoor air as the topic. Although this will not yield every single paper it does give an overall picture of how indoor air pollution is now drawing the attention of scientists from all over the world.

It is evident that, in many circumstances, people are far more exposed to pollution indoors than outdoors. We also tend to believe that the indoor environment is cleaner, more comfortable and healthier on the obvious grounds that the building shelters us from harmful substances in the ambient environment. For this reason a number of quality indication systems in the world give warnings or advice to stay indoors during episodes of poor air quality. However, Chan (2002) pointed out that, the fundamental question is: Is indoor air really cleaner? Is it free from outdoor pollutants?

Indoor aerosol concentrations are associated with both indoor and outdoor sources. The identification of sources and the assessment of their relative contribution can be a complicated process due to the presence of a number of indoor sources, which can vary from building to building. There are also uncertainties associated with estimating the impact of outdoor sources on the indoor environment (Mitchell et al. 2007). There are numerous indoor sources and these, in residential environments, include heating, cooking, cleaning, smoking, the use of a wide variety of consumer products, building materials and furnishings, and the simple act of moving about and stirring up particles. Some of these sources emit pollutants virtually continuously while others are related to specific activities (e.g. cleaning). The relative importance of any single source depends on how much of a given pollutant it emits and how hazardous those emissions are.

A wide range of pollutants have been reported in residential environments and these are summarized in Table 2.1. There are distinct variations in the importance of the different sources in different areas of the world; closely related to level of socioeconomic development.

Outdoor pollutant concentrations may not be reliable indicators of indoor and personal pollutant sources. Lawrence et al. (2005) remarked that assessment of risk to a community resulting from exposure to air pollutants should ideally include measurements of concentration levels of pollutants in all micro-environments where people spend their time.

It is beyond the scope of this chapter to review every single pollutant in every conceivable micro-environment. The reader is referred to Ashmore and Dimitroulopoulou (2009), Fuentes-Leonarte et al. (2009), Morawska (2008), Fullerton et al. (2008), Yu et al. (2008), Mitchell et al. (2007), Ott et al. (2007); Gorny (2004), Salthammer (2009), Burroughs and Hansen (2008) and Pluschke (2004).

This chapter focuses on non-industrial buildings such as homes, schools, offices and transport micro-environments. It considers, in depth, a number of case studies. It should be remembered that the concentration of indoor particles is highly variable and house-specific.

**Table 2.1** Sources of the main indoor air pollutants (Adapted from CARB 2005)

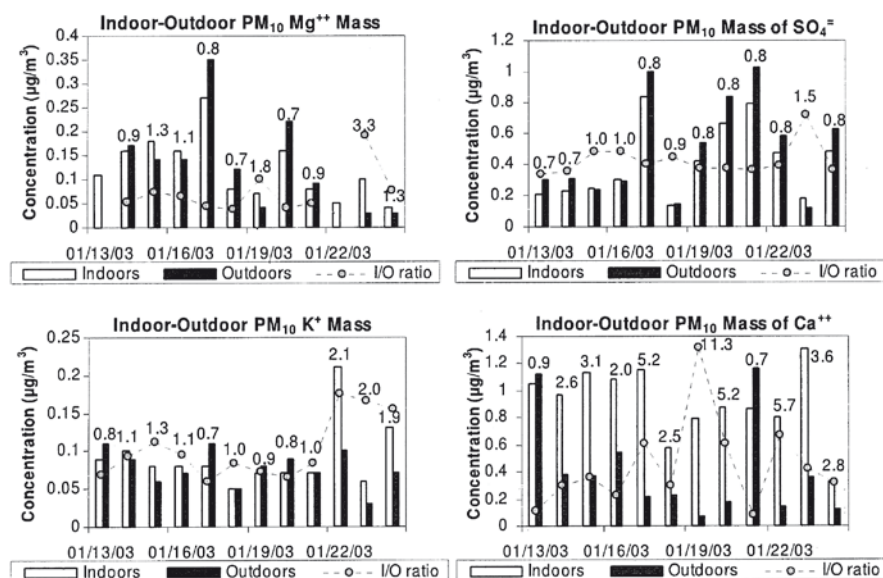
Pollutant	Main sources
Arsenic	Coal combustion
Asbestos	Building materials in older homes released during renovation, naturally occurring in some soils
Biological Agents (bacteria, fungi, viruses, house dust mites, animal dander; cockroaches, microbial VOCs)	House and floor dust; pets; bedding; poorly maintained air-conditioners, humidifiers, dehumidifiers; moist structures or furnishings; insect infestation; building occupants
Carbon monoxide	Unvented or malfunctioning gas appliances, wood stoves, fireplaces, tobacco smoke
Endocrine disruptors (phthalates; DDT, chlordane, heptachlor, <i>o</i> -phenylphenol; PBDEs)	Plastics; pesticides; flame retardants
Environmental tobacco smoke	Cigarettes, cigars and pipes
Formaldehyde, other aldehydes	Composite wood products such as plywood and particleboard; furnishings; wallpaper; paints; combustion appliances; tobacco smoke
Lead	Lead paint chips, contaminated
Nitrogen dioxide	Unvented or malfunctioning gas appliances, other combustion appliances
Organic chemicals (benzene, chloroform, paradichlorobenzene, methylene chloride, perchloroethylene, phthalates, styrene)	Solvents; glues; cleaning agents; pesticides; building materials; paints; treated water; moth repellents; dry-cleaned clothing; air fresheners
Ozone	Infiltration of outdoor air, ozone generating air “purifiers”, office machines
Particulate matter	Cigarettes, wood stoves, fireplaces, cooking, candles, aerosol sprays, house dust
Polycyclic aromatic hydrocarbons	Cigarette smoke, cooking, woodburning
Radon	Uranium-bearing soil under buildings, ground-water, construction materials

## 2.2 Indoor-Outdoor Measurements in Oslo

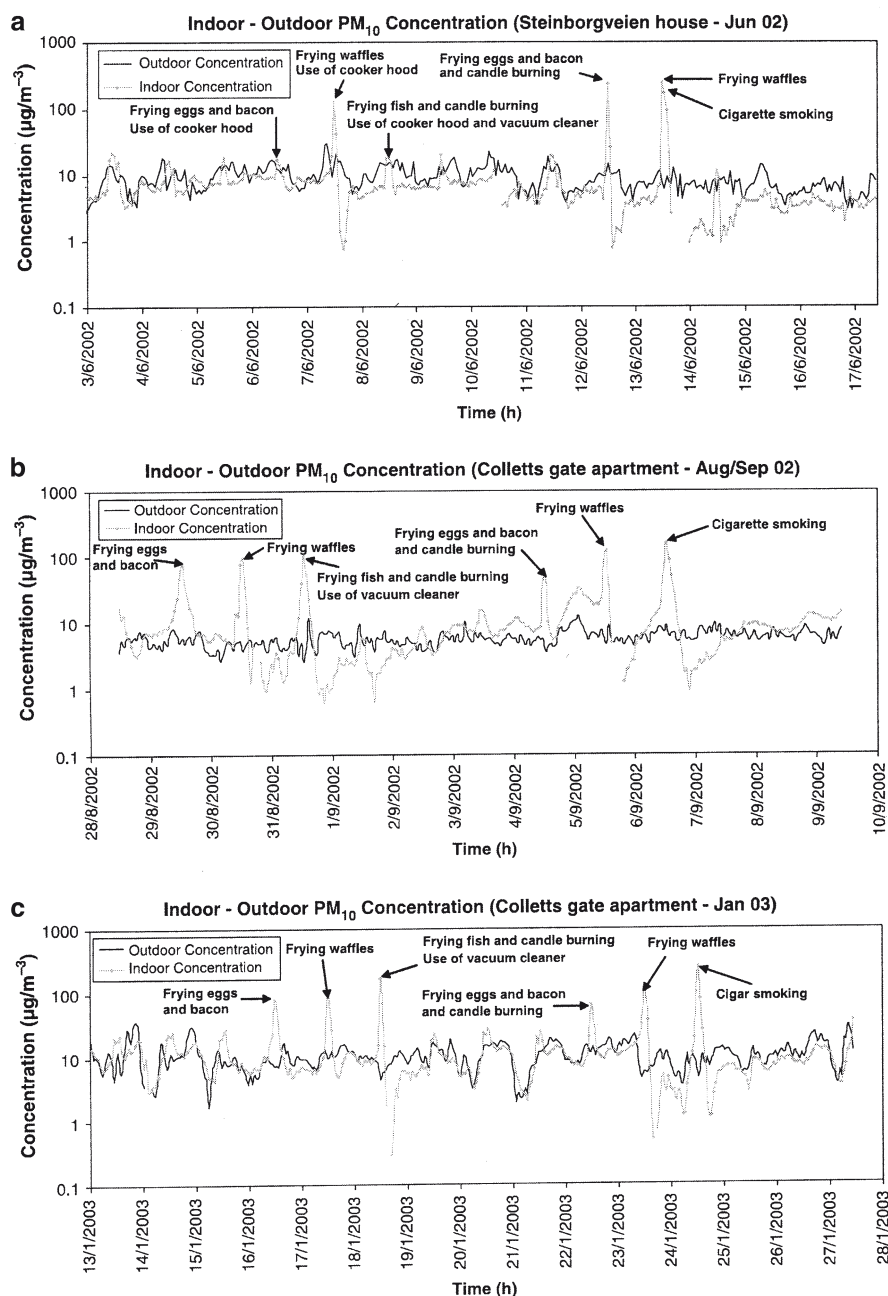
As part of the Urban-Aerosol project (Characterisation of Urban Air Quality – Indoor/Outdoor Particulate Matter Chemical Characteristics and Source-to-Inhaled Dose Relationships) a series of indoor/outdoor measurements were undertaken in Athens, Oslo, London, Hannover, Prague and Milan (Lazaridis et al. 2006). In Oslo indoor-outdoor measurements, during summer and winter periods, were made at two different residential houses (one in the suburbs and the other in the city centre). Apart from the integrated indoor-outdoor particulate matter ( $PM_{10}$ ,  $PM_{2.5}$ , size distribution) and gaseous pollutants ( $O_3$ ,  $NO_x$ , VOC) measurements, infiltration rate evaluation ( $SF_0$ ) and meteorological measurements were also performed along with a detailed chemical speciation and compilation of a daily diary where different indoor activities were registered. In addition continuous measurements for  $PM_{10}$

and  $PM_{2.5}$  using TEOM instruments were performed together with the use of particle number distribution measurements using a Scanning Mobility Particle Sizer (particle size range 10–450 nm), and Aerodynamic Particle Sizer (particle size range 0.7–20  $\mu m$ ) (Lazaridis et al. 2006, 2008).

The concentration of total particles measured outside the city centre apartment was approximately twice that measured in the suburbs. However, the concentration of total particles was higher indoors than outdoors during the winter due to the indoor activities, the operation of a heating system and reduced ventilation for energy saving. On the other hand the indoor concentration was lower than that outdoors in the summer at both sites due to the increased ventilation and the enhanced secondary aerosol formation in the outdoor environment. The concentration of particles in the outdoor environment was attributed to both anthropogenic pollution and natural sources. The results indicated the influence of marine aerosols, crustal aerosols and anthropogenic pollution from both local and remote sources to the ambient aerosol concentrations. Specifically, the concentration of nitrate, chloride, sodium, calcium and magnesium ions was higher during the winter periods due to the enhanced effect of local traffic (direct emissions and resuspension of road side dust) especially in the city centre apartment whereas sulphate and ammonium concentrations were higher during the summer periods due to the enhanced secondary aerosol formation. Furthermore, their concentrations exhibited high daily variation especially during the winter/spring periods due to the episodic strong winds blowing from marine areas (e.g. 13–17 January) due to air masses arriving at the sampling site from western locations (Fig. 2.2). Generally, when air



**Fig. 2.2** Indoor-outdoor variability of inorganic ions,  $PM_{10}$  size fraction, during the January 2003 measurement period at a city centre apartment in Oslo



**Fig. 2.3** Indoor and outdoor PM<sub>10</sub> concentration at the: (a) suburban house (Steinborgveien) during June 2002; (b) city centre apartment (Colletts Gate) during August/September 2002; and (c) during January 2003. The various activities are indicated in the figure (Reproduced with permission)

masses originated from western locations the concentration of marine particles was enhanced, whereas, when the air masses originated from eastern and northern areas the concentration of the secondary and crustal component of aerosols was enhanced. When considering the indoor to outdoor relationship of specific aerosol components it was reported that the chemical concentration of the inorganic aerosol mass was higher outdoors whereas the organic mass was higher indoors, especially during cooking activities (Lazaridis et al. 2008).

Comparing simultaneous measurement of  $PM_{10}$  indoors and outdoors it is evident that specific indoor activities, such as cooking, cleaning and smoking, lead to high indoor concentrations. (Fig. 2.3). Chemical analysis indicated increased indoor concentration of organic carbon particles, sulphate, nitrate, calcium and sodium ions during cooking. The indoor concentration of fine organic carbon particles was approximately 0.65 times the outdoor concentration during days with no activity whereas on days with activities it was approximately five times higher. In addition, the indoor concentration of coarse organic carbon particles was approximately 16 times higher than that outdoors during days with indoor activities (Lazaridis et al. 2006).

## 2.3 Particle Emission Rates

It is evident from the above that both indoor and outdoor sources contribute to and affect the concentration and composition of particles in indoor air. While emission rates for outdoor particle sources are reasonably quantified (Mitra et al. 2002; Zhang and Morawska 2002), ignoring tobacco smoke, there was, until relatively recently, only limited data for particulate emissions from indoor sources. The most important sources include cooking, kerosene heating and wood burning (e.g. Raunemaa et al. 1989; Long et al. 2000; Dennekamp et al. 2001; Sjaastad and Svendsen 2008) while sources such as cleaning, dusting and vacuuming, showering, electric motors, movement of people and gas-to-particle conversion have also been investigated (e.g. Abt et al. 2000a, b; Waring et al. 2008). Secondary formation of ultrafines in large quantities indoors has been observed from chemical reactions of ozone and terpenes (Weschler and Shields 1999, 2003).

The method of cooking can have an impact on the emissions with significant differences between Eastern and Western cultures (He et al. 2004b; Robinson et al. 2006; See and Balasubramanian 2008). The majority of the work on particulate emissions from cooking has considered indoor  $PM_{10}$  or  $PM_{2.5}$  concentrations in houses or restaurants (Lee et al. 2001; Monkkonen et al. 2005; Fortmann et al. 2001). Small particles can be high in number while contributing little to particle mass. Such particles can also penetrate more deeply into the lung and as a result have been a subject of increasing concern. Number concentration measurements have been carried out for various cooking activities (Siegmann and Sattler 1996; Dennekamp et al. 2001; Wallace et al. 2004, 2008; Afshari et al. 2005;

**Table 2.2** Particle emission rates for various indoor activities

Activity	Emission source strength (particles min <sup>-1</sup> )			
	Gehin et al. (2008)	Afshari et al. (2005)	He et al. (2004a)	Wallace et al. (2008)
Heating electric stove	$4.2 \times 10^{11}$	$6.8 \times 10^{11}$	$7.33 \times 10^{11}$	$6 \times 10^{11}$ – $11 \times 10^{12}$
Grilling			$7.34 \times 10^{11}$	
Frying meat/fish	$5.4 \times 10^{12}$	$8.27 \times 10^{11}$	$4.75 \times 10^{11}$	
Cooking (oven) meat/fish	$4.2 \times 10^{11}$		$1.27 \times 10^{11}$	$4 \times 10^{11}$ – $1.1 \times 10^{12}$
Candle	$7.2 \times 10^{10}$	$8.8 \times 10^{10}$		
		$3.65 \times 10^{11}$		
Aerosol spray	$1.8 \times 10^{12}$	$2.34 \times 10^{11}$		
Vacuuming	$1.2 \times 10^{12}$	$3.5 \times 10^{11}$	$9.7 \times 10^{10}$	
Smoking		$3.76 \times 10^{11}$	$1.91 \times 10^{11}$	

Hussein et al. 2005). Burning candle emissions have been reported by Wasson et al. (2002) and Afshari et al. (2005) while Lung and Hu (2003) studied incense combustion. Other sources include a clothes dryer (Wallace 2005) and office equipment (He et al. 2007; Destailats et al. 2007). He et al. (2004a) and Gehin et al. (2008) have looked at a number of activities ranging from cooking to vacuuming. Table 2.2 summarises some of these particle emission rates. It should be remembered that the various emission rates have been based on different methods and cover slightly different size ranges. For instance the work of Gehin et al. (2008) reports emission rate for particles with diameter between 5 nm and 1  $\mu\text{m}$  while that for Wallace et al. (2008) is for the size range 2–64 nm. However it is evident that various indoor activities emit a significant number of particles, the majority of which can be classified as ultrafine.

It is evident from the above that the concentration of indoor air pollution is highly dependent on the resident's activities and lifestyle. Similar houses can exhibit quite different concentrations. Even the same house can experience changes in the diurnal variation as a result of changes in lifestyle. Figure 2.4 shows the diurnal variation in  $\text{PM}_{10}$ ,  $\text{PM}_{2.5}$  and  $\text{PM}_1$  for a typical house in the UK with children of school age. The solid lines represent PM concentrations on non-school days while the dashed lines are for school days. On school days the early morning peak is of short duration and there is little increase around lunch time. When the children return in the afternoon there is a sharp rise in PM concentrations which is not evident on non-school days. It is clear that there is a close relationship between the concentration and human indoor activities. Humans are responsible for their own "personal cloud", that is, exposure to airborne particles resulting from personal activities (e.g. occupation, hobbies) or physical activities (e.g. jogging, vacuum cleaning) (Rodes et al. 1991; McBride et al. 1999; Adgate et al. 2003). For  $\text{PM}_{2.5}$  this cloud is around  $15 \mu\text{g m}^{-3}$  and can be higher for those who live an active outdoor life (Adgate et al. 2003).



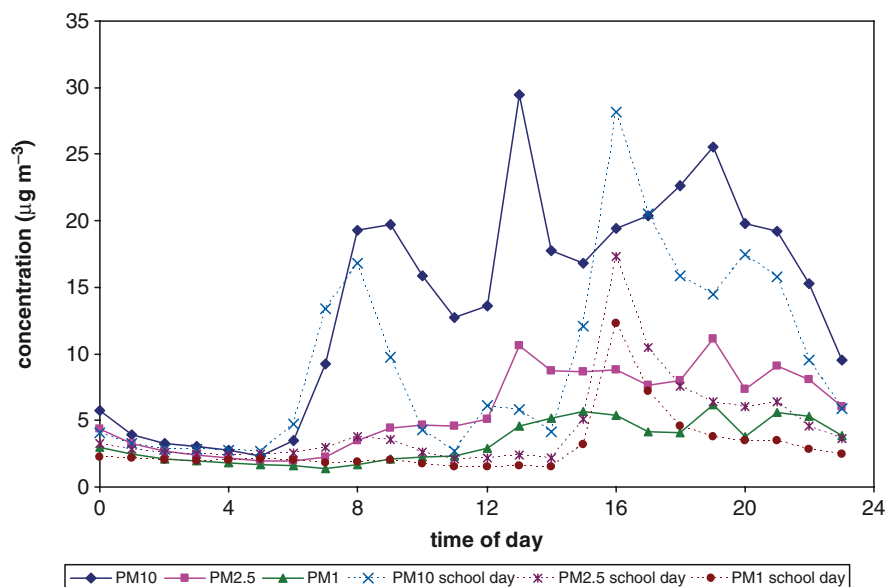


Fig. 2.4 Impact of activities on the diurnal profile of indoor air quality

## 2.4 Bioaerosols

Biological material is present in the atmosphere in the form of pollens, fungal spores, bacteria, viruses, and any fragments from plants and animals. The size scale ranges from about 15–400 nm for viruses, through 0.3–10 µm for bacteria to 1–100 µm for fungal spores, pollen and plant debris. There is great concern about the potential health hazards of indoor bioaerosols to humans, with a special focus on allergenic or toxigenic fungi and their association with indoor air quality. Douwes et al. (2003) concluded that the potential health effects of bioaerosol exposures are diverse including infectious diseases, acute toxic effects, allergies and cancer. There is also a growing body of scientific literature examining the relationship between dampness and mould in buildings and associated health effects (Bornehag et al. 2001; IOM 2004; Mudarri and Fisk 2007).

### 2.4.1 Indoor Concentrations

Bioaerosols have been studied in numerous different regions and settings: schools (Aydogdu et al. 2005), child care centres (Zuraimi and Tham 2008), markets (Narayan et al. 1982), animal feed industry (Hameed et al. 2003), animal sheds

(Rosas et al. 2001), rice mills (Savino and Caretta 1992; Desai and Ghosh 2003), saw mills (Oppliger et al. 2005; Jothish and Nayar 2004), food grain warehouse, bakery and library (Jain 2000), food processing units (Zorman and Jersek 2008), bakeries and flour mills (Musk et al. 1989; Singh and Singh 1994; Awad 2007), hospitals kindergartens, senior care centres and nursing centres (Kim and Kim 2007), social welfare houses (Rolka et al. 2005) and offices (Kalogerakis et al. 2005).

Many measurements carried out on particulate matter ( $PM_{10}$  and  $PM_{2.5}$ ) and their outdoor-indoor relationships automatically include biological particles in the sampling process, although the data does not differentiate between organic and inorganic matter (Conner et al. 2001; Morawska et al. 2001). Measurements taken in Eastern Europe indicated that bacterial levels may be higher indoors when high numbers of people are present, than those outdoors (Goh et al. 2000). Exposure to viable airborne bacteria has also been shown to be considerably higher in the indoor environment, with levels increasing with increased human occupancy (Ambroise et al. 1999). They found this to be in stark contrast to fungal spore concentrations, which were higher outdoors. An early review on fungal exposure was given by Miller (1992). Fungal levels have chiefly been found to be higher outdoors unless the building is ventilated (Parat et al. 1997; Goh et al. 2000; Wu et al. 2000). Living conditions have been shown to affect levels of airborne microbes (Perera et al. 2002), whereby low-income families living in poorer quality accommodation were found to suffer the highest exposure levels. The work implicates the need to comprehensively examine the effects of social class and types of housing on exposure to bioaerosols, themselves requiring accurate assessment.

An evaluation of indoor and outdoor fungal concentration in 1,717 buildings in the United States was presented by Shelton et al. (2002) who found lower concentrations than outdoors. Overall 95% of the buildings had a median indoor and outdoor fungal concentration of less than 1,300 CFU/m<sup>3</sup>, and less than 3,200 CFU/m<sup>3</sup>, respectively. The highest fungal levels were obtained in the fall and summer. The levels of indoor bacteria and fungi in various public places and food processing units were assessed by Zorman and Jersek (2008). The concentration of fungi was significantly higher in food processing units than in public places. In public places, the concentrations of bacteria and fungi were in the range of 0–5,860 CFU/m<sup>3</sup> and 5–3,579 CFU/m<sup>3</sup>, respectively. Whereas, levels ranged from 0 to 3,506 CFU/m<sup>3</sup> for viable bacteria and 22–46,377 CFU/m<sup>3</sup> for fungi in food processing units. Lin and Li (1996) investigated indoor and outdoors fungi in six residences in Taiwan with a two-stage Anderson impactor. They revealed that the number concentrations of indoor total and respirable fungi were in the range of 420–4,200 CFU/m<sup>3</sup> and 250–1,000 CFU/m<sup>3</sup>, respectively.

Monitoring of bacteria and fungi in indoor air at several schools in the city of Edrine, Turkey was carried out by Aydogdu et al. (2005). They reported a positive correlation between the concentration of bacteria and humidity and age of school. A study by Godwin and Batterman (2007) on indoor air quality in 64 Michigan schools revealed that bioaerosol concentrations were <6,500 CFU/m<sup>3</sup> and <4,100 CFU/m<sup>3</sup> for indoor and outdoor, respectively. Bioaerosol exposure in apartments located in high rise buildings in Korea was evaluated by Lee and Jo (2006).

They reported that season, room location in the apartment and floor level influenced the bioaerosol concentration.

To identify bacterial species contaminating working environments Bouillard et al. (2005) conducted sampling in 25 offices. In their study bacterial levels varied from 44–2,511 CFU/m<sup>3</sup> with a median of 277 CFU/m<sup>3</sup>. They concluded that people working in offices can be exposed to large concentrations of airborne bacteria and related endotoxins. The US Environmental Protection Agency carried out a study (The Building Assessment Survey and Evaluation), in 100 large office buildings between 1994 and 1998. The summary of this study, presented by Tsai and Macher (2005), revealed that concentrations varied with a seasonal pattern. Outdoor bacterial concentrations were higher in winter than those indoors: 194 CFU/m<sup>3</sup> compared to 165 CFU/m<sup>3</sup>. However indoor concentrations were higher in summer as compared to those outdoors (116 versus 87 CFU/m<sup>3</sup>) (Tsai et al. 2007).

Haas et al. (2007) carried out a year long study to assess indoor mould in 66 apartments (29 with no visible mould growth and 37 with visible mould growth) in Austria. The median concentrations of viable fungal spores were significantly higher in apartments with mould growth ( $1.5 \times 10^3$  CFU/m<sup>3</sup>) in comparison to those without mould growth ( $2.6 \times 10^2$  CFU/m<sup>3</sup>). Moreover, in flats with no visible mould growth median spore concentrations were significantly higher in summer than winter and spring. On the other hand, there was no significant seasonal difference in the concentration of fungal spores in flats with mould growth, but a ten times higher indoor concentration than outdoor air was obtained. During an investigation on indoor fungal and bacterial aerosols in 60 flats of Upper Silesia, Poland Gorny et al. (1999) reported that levels were below 10<sup>4</sup> CFU/m<sup>3</sup>. In another study in the same region, Pastuszka et al. (2000) showed that levels of bacterial aerosol in homes and offices were 10<sup>3</sup> CFU/m<sup>3</sup> and 10<sup>2</sup> CFU/m<sup>3</sup>. The levels of fungal aerosol, during the winter, in healthy homes ranged from 10–10<sup>2</sup> CFU/m<sup>3</sup> as compared to 10–10<sup>3</sup> CFU/m<sup>3</sup> in mouldy homes. In winter, the levels increased to 10<sup>3</sup> and 10<sup>4</sup> CFU/m<sup>3</sup> in healthy and mouldy homes, respectively.

A study by Reponen et al. (1994) revealed the effect of a range of domestic activities. Most of the activities had a noticeable effect on the spore counts except baking, handling of house plants and vacuum cleaning. A quick decline in the concentration of large size spores was observed after the activity, clearly due to faster gravitational settling. Similar results of short term anthropogenic activity were reported by Brandl et al. (2008) during a study on bioaerosol generation in indoor air of a university hallway. Their results indicated a clear association of presence/absence of people and concentration of bioaerosols. The highest bacterial aerosol concentrations (1,200 CFU/m<sup>3</sup>) were recorded during the presence of students while these levels fell to 200 CFU/m<sup>3</sup> during their absence.

The investigations discussed above demonstrate a wide variation in the concentration of bioaerosols in different microenvironments. The concentration and size distributions not only vary with geographical location but also depend on a wide range of biotic and abiotic factors. According to several studies, the moisture content of building material, relative humidity and temperature (Foarde et al. 1993; Pasanen et al. 2000; Ritschkoff et al. 2000; Viitanen et al. 2000) outdoor concentrations, air

exchange rates (Kulmala et al. 1999), human activities (Buttner and Stetzenbach 1993) and number of people and pets (ACGIH 1999) significantly affect the levels of indoor bioaerosols. Moreover, housing conditions, the activities and life style of occupants considerably contribute to the varying concentrations. These factors fluctuate to a great degree between various housing types, their condition and geographic location. Hence there is need to study the indoor bioaerosols in various types of residential settings.

### 2.4.2 Size Distribution

Most studies have focused on the total concentration of bioaerosols. However, particle size is critical with regard to their fate in the air and their deposition in the human respiratory system.

Table 2.3 shows the geometric mean (GM) and geometric standard deviation (GSD) of total viable fungal and bacterial aerosol in various size fractions for three

**Table 2.3** Geometric mean (GM) and geometric standard deviation (GSD) of total viable fungal and bacterial aerosol and in various size fractions for three different housing types

Housing Type I	Bacteria		Fungi	
	GM(CFU/m <sup>3</sup> )	GSD	GM(CFU/m <sup>3</sup> )	GSD
Total	1,557	1.5	925	2.9
7 µm & above	279	1.3	176	1.8
4.7–7 µm	277	1.7	241	2.3
3.3–4.7 µm	434	2.6	192	5.8
2.1–3.3 µm	240	1.3	82	9.5
1.1–2.1 µm	212	1.5	110	2.0
0.65–1.1 µm	19	2.5	15	1.9
<i>Housing Type II</i>				
Total	2,403	2.3	813	3.6
7 µm & above	471	3.0	63	1.6
4.7–7 µm	382	2.3	136	2.2
3.3–4.7 µm	451	2.7	232	3.7
2.1–3.3 µm	536	2.2	182	8.4
1.1–2.1 µm	355	3.5	100	2.1
0.65–1.1 µm	64	1.3	7	1.1
<i>Housing Type III</i>				
Total	5,036	2.5	2,124	1.38
7 µm & above	199	1.3	257	4.8
4.7–7 µm	311	1.1	294	4.2
3.3–4.7 µm	207	2.8	429	1.7
2.1–3.3 µm	774	4.2	581	2.2
1.1–2.1 µm	2,228	1.6	140	1.5
0.65–1.1 µm	659	11.1	7	1.0

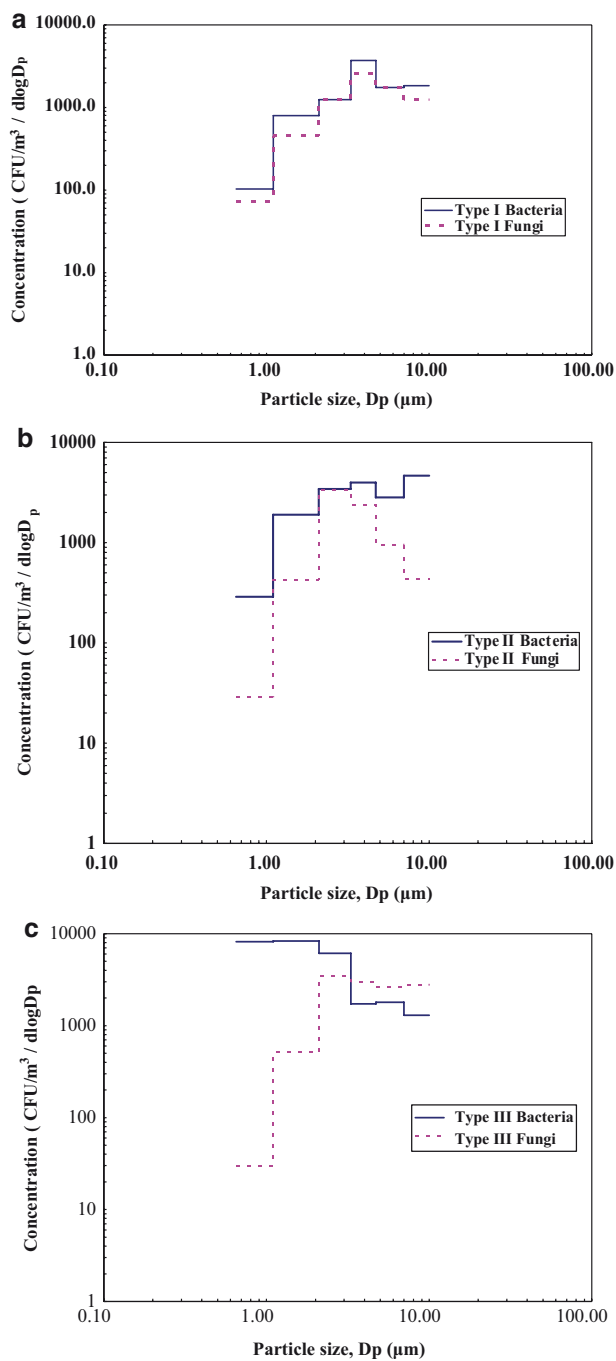
different housing types. Measurement were made with an Anderson six-stage viable particle sampler in different types of residential houses: a single room in shared accommodation (Type I), single bedroom flat in three storey buildings (Type II) and two or more bedroom houses (Type III). All the accommodation was 40–50 years old and no major repairs had been undertaken over the previous year. Type I houses were occupied by one person whereas types II and III were occupied by two to three and four to six people, respectively. It is evident that the concentrations of both bacterial and fungal aerosol were almost double in type III housing accommodation as compared to types I and II. This reflects the effects of number of occupants, their activities and possibly building construction and design on indoor bioaerosol levels. The concentrations of indoor bioaerosol in this study could be due to the season as many studies have reported higher indoor levels in summer in non-mouldy houses (Pastuszka et al. 2000; Ren et al. 1999; Lee and Jo 2006; Shelton et al. 2002) due to migration of fungal spores from outdoors.

Figure 2.5 shows the size distribution of fungal and bacterial spores for the various housing types. The viable bacterial concentration exhibits different size distributions in all the housing types. However, more than 60% of viable bacteria were of  $<4.7\ \mu\text{m}$  in housing type I and II. While in housing type III almost 88% were in the size fraction  $<4.7\ \mu\text{m}$ .

The differences in the size distribution of fungal spores among different housing conditions reflect the different species composition or the different ages of the spores. The aerodynamic sizes of the freshly released spores are larger than those which have been airborne for a longer time (Reponen et al. 1994). Dehydration, agglomeration and relative humidity of surrounding air (Pasanen et al. 1991; Reponen et al. 1996; Ren et al. 2001) are among other factors affecting the size of spores. In the naturally ventilated buildings hygroscopic growth of bioaerosols by condensation or water absorption, influences the kinetics of aerosols (Liao et al. 2004).

## 2.5 Indoor Air Quality in Developing Countries

In developing countries, population explosion along with urbanization and growing industrialization has resulted in dense urban centres with poor air quality. However, around 60% of the people in developing countries continue to live in rural areas and so the micro-environment with the greatest contribution to global person-time is the rural indoor environment. In developing countries, the most significant issue for indoor air quality is exposure to pollutants released during combustion of solid fuels, including biomass (wood, dung and crop residues) or coal (mainly in China), used for cooking and heating. Worldwide, more than three billion people, largely in developing countries, rely on biomass fuels for their domestic energy needs (WHO 2006). A number of different chemical substances are emitted when biomass is burnt including carbon monoxide, nitrogen dioxide, particulate matter, polycyclic aromatic hydrocarbons, benzene and formaldehyde (Smith 1987;



**Fig. 2.5** Size distribution of bacterial and fungal spores in (a) type I houses, (b) type II houses and (c) type III houses

De Koning et al. 1985; Mudway et al. 2005; Naeher et al. 2007). Combustion of coal in addition to the above pollutants may release sulphur dioxide, arsenic and fluorine (Finkelman et al. 1999). Due to the variety of fuel types, together with ventilation rates and combustion temperatures, a variation in emission patterns is expected. There is strong evidence that smoke from biofuels can cause acute lower respiratory infection in childhood (WHO 2006; Smith et al. 2000; Ezzati and Kammen 2001). A recent report on national burden of diseases from indoor air pollution by the World Health Organization (2007) confirms the linkage between indoor air pollution due to solid fuels and different diseases, including acute and chronic respiratory diseases, tuberculosis, asthma, and cardiovascular disease and prenatal health outcomes. In most of cases indoor air pollution disproportionately affects women and children who spend most time near the domestic hearth. Indoor air pollution is responsible for more than 1.6 million annual deaths and 2.7% of global burden of diseases (WHO 2006) and indoor air pollution from solid fuel use is the tenth biggest threat to public health WHO (2007).

Studies on indoor air pollution from solid fuels have been conducted in various developing countries in recent years including Mexico (Zuk et al. 2007), Philippines (Saksena et al. 2007), China (Fischer and Koshland 2007; Mestl et al. 2007), Zimbabwe (Rumchev et al. 2007), Bangladesh (Dasgupta et al. 2006), India (Balakrishnan et al. 2002, 2004), Costa Rica (Park and Lee 2003), Bolivia (Albalak et al. 1999) and Kenya (Boleij et al. 1989). Indoor PM concentrations in a number of developing countries are shown in Table 2.4.

Figure 2.6 shows the mass concentration of particulate matter during cooking, using biomass fuels, at a rural location in Pakistan. A large variation in concentration is evident. Particulate levels increase rapidly during cooking and decrease quickly after cooking. Over a period of 1 week, the daily levels of  $PM_{10}$ ,  $PM_{2.5}$  and  $PM_1$  during cooking ranged from 1,991  $\mu\text{g}/\text{m}^3$  to 7,881  $\mu\text{g}/\text{m}^3$ , 1,531  $\mu\text{g}/\text{m}^3$  to 2,664  $\mu\text{g}/\text{m}^3$  and 1,430  $\mu\text{g}/\text{m}^3$  to 2,396  $\mu\text{g}/\text{m}^3$ , respectively. Generally a wide variation in concentration of particulate matter is observed among different kitchens and even within the same kitchen during different episodes of cooking. The variation primarily depends on the quality (dryness) of biomass fuel used, duration of cooking, degree of incomplete combustion and ventilation. Ezzati and Kammen (2002) have shown that a typical 24-h average concentration of  $PM_{10}$  in homes using biofuels may range from 200 to 5,000  $\mu\text{g}/\text{m}^3$  or more throughout the year. Figure 2.7 indicates the high intensity emissions that commonly occur when using biomass fuels. The data, from a rural site in Kenya, show that the mean  $PM_{10}$  measurement near the fire was 1,250  $\mu\text{g}/\text{m}^3$  – yet levels actually peaked at over 50,000  $\mu\text{g}/\text{m}^3$  (Ezzati et al. 2000a, b). Emissions in the kitchen can vary from day to day and from season to season, due to the moisture content and density of the fuel, the amount of airflow, the type of food being cooked and any changes in the stove or fuel used.

Many improved cookstove projects currently exist worldwide, ranging from local non-governmental organization projects to nationwide initiatives (WHO 2008; Granderson et al. 2009). Methods to reduce indoor air pollution from biomass use fall into four general categories: behavioural modifications to reduce

**Table 2.4** Indoor particulate concentrations in a number of developing countries (GM: geometric mean)

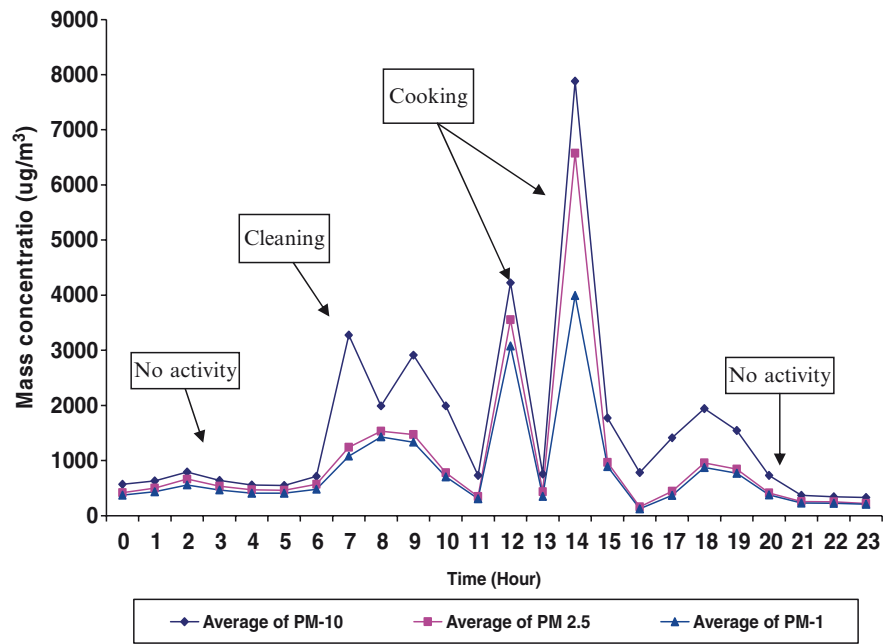
Location	Averaging time and size fraction	Fuel	Concentration ( $\mu\text{g m}^{-3}$ )
Nepal (Pandey et al. 1990)	Cooking period $\text{PM}_{2.5}$	Wood/crop residues	8,200 (traditional stoves) 3,000 improved stoves
Garhwal, India (Saksena et al. 1992)	Cooking period TSP 24 h exposure TSP	Wood/shrubs	4,500 (GM) 700–1,690 (winter) 250–1,130 (summer)
Pune, India (Smith et al. 1994)	12–24 h $\text{PM}_{10}$	Wood	2,000 (area) 1,100 (personal)
Mozambique (Ellegard 1996)	Cooking period $\text{PM}_{10}$	Wood	1,200
Bolivia (Albalak et al. 1999)	6 h $\text{PM}_{10}$	Dung	1,830 (GM, indoor kitchens) 280 (GM, outdoor kitchens)
Kenya (Ezzati et al. 2000a, b)	Daily average exposure $\text{PM}_{10}$	Mixed	1,000–4,800
Tamil Nadu, India (Balakrishnan et al. 2002)	Cooking period $\text{PM}_4$ Daily average exposure $\text{PM}_4$	Wood/agricultural waste Wood/agricultural waste	1,307–1,535 (GM, personal) 172–226
La Victoria, Guatemala (Albalak et al. 2001)	24 h $\text{PM}_{3.5}$	Wood	1,560 (GM, traditional stove) 250 (GM, improved stove) 850 (GM, LPG/open fire)
La Victoria, Guatemala (Bruce et al. 2004)	24 h $\text{PM}_{3.5}$	Wood and crop residue	1,019 (GM, traditional stove) 351 (GM, improved stove)
Andhra Pradesh, India (Balakrishnan et al. 2004)	24 h $\text{PM}_4$ Daily average exposure $\text{PM}_4$	Wood/dung/agricultural waste Wood/dung/agricultural waste	297–666 (kitchen) 215–357 (living area)
Bangladesh (Dasgupta et al. 2004)	24 h $\text{PM}_{10}$	Wood/dung/agricultural waste	196–264 (personal) 60–1,165 (area)
Zimbabwe (Rumchev et al. 2007)	4 h $\text{PM}_4$	Wood	230–7,330
China (Edwards et al. 2007)	24 h $\text{PM}_4$	Wood Crop residues Coal	164 282–456 142–289

(continued)



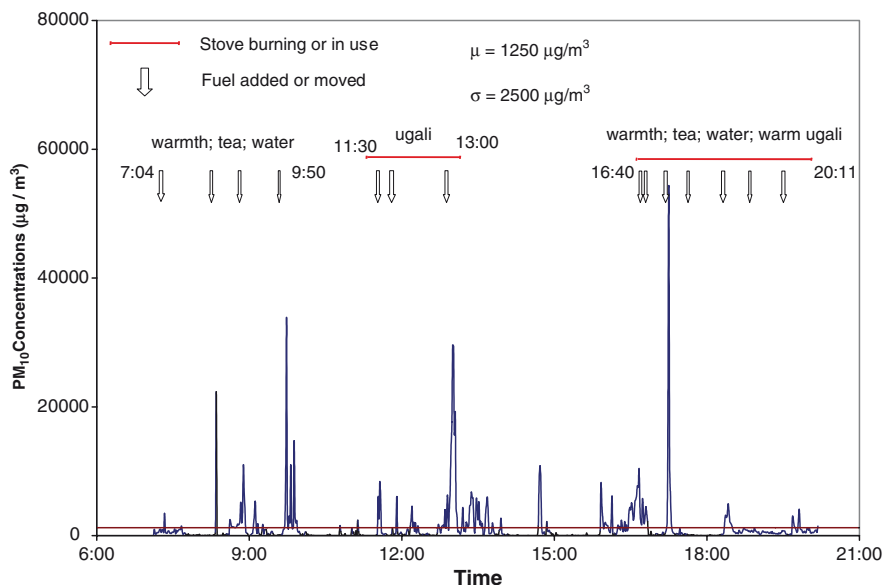
**Table 2.4** (continued)

Location	Averaging time and size fraction	Fuel	Concentration ( $\mu\text{g m}^{-3}$ )
Shenyang, China (Jiang and Bell 2008a,b)	14 h $\text{PM}_{10}$	Crop residue	100 mean 14–1,571 range
Gansu, China (Jin et al. 2005)	24 h $\text{PM}_4$	Wood and crop residue	518 spring mean 661 winter mean
Chak NO.35/2.L and Bhaun, Pakistan (Colbeck et al. 2010)	1 h $\text{PM}_{10}$	Wood/dung	1,581 mean 141–8,555 range
	1 h $\text{PM}_{2.5}$		1,169 mean 23–5,953 range
	1 h $\text{PM}_1$		913 mean 13–3,449 range
Costa Rica (Park and Lee 2003)	24 h $\text{PM}_{10}$	Wood	132 mean 500–18,900 (peak range)
	24 h $\text{PM}_{2.5}$		44 mean 310–8,170 (peak range)



**Fig. 2.6** Mass concentration of  $\text{PM}_{10}$ ,  $\text{PM}_{2.5}$  and  $\text{PM}_1$  in a kitchen using biomass fuel at rural site in Pakistan

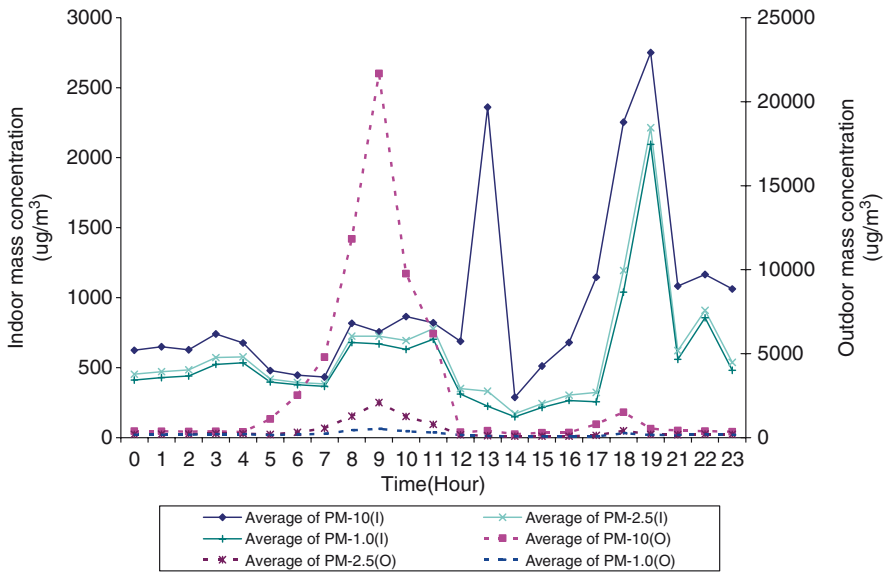
exposure; household changes to improve ventilation; improvements to cooking stoves; and interventions to enable people to use higher-quality, lower-emission liquid or gaseous fuels (Desai et al. 2004). For example switching from wood, dung



**Fig. 2.7**  $PM_{10}$  concentrations, at a distance and height of 0.5 m, in a household using a three-stone stove in rural Kenya. The horizontal line represents the mean concentration for the day. As seen, mean concentration is a poor indicator of the patterns of exposure. Ugali is a common Kenyan food made from maize or sorghum flour. Reproduced with permission from Environmental Health Perspectives (Ezzati and Kammen 2002)

or charcoal to more efficient modern fuels, such as kerosene, liquefied petroleum gas and biogas, brings about the largest reductions (Parikh et al. 2001). The installation of chimneys or smoke hoods can reduce the concentration of respirable particles by up to 80% (Practical Action 2004; McCracken et al. 2007). Changing cooking behaviours are unlikely to bring about such large reductions as other interventions but are important supporting measures. Where cleaner fuels such as gas are introduced  $NO_2$  is likely to become more important as a pollutant in kitchens.

It is not only kitchens that can experience high levels of particulate matter. For example a living room at a site in rural Pakistan experienced  $PM_{10}$  levels, up to  $21,673 \mu g/m^3$  as a result of sweeping. Cleaning outside results in even higher levels as typical courtyards are dry and devoid of any grass/vegetation. In general, during most of the day indoor concentrations were higher than those outdoors (Fig. 2.8) and suggests an indoor source of fine particulates. Social gatherings take place during the evening and, in this example, up to eight smokers were in the room. One out of every two to three middle-aged men in Pakistan smoke cigarettes (Ahmad et al. 2005). During smoking there is a sharp rise in fine particulates and the indoor-outdoor ratio for  $PM_{10}$  rose to a maximum of 12.95.



**Fig. 2.8** Hourly average of indoor and outdoor mass concentration for PM<sub>10</sub>, PM<sub>2.5</sub> and PM<sub>1</sub> in a living room at a rural site in Pakistan. I = Indoors; O = Outdoor

**2.6 Transport Micro-environments**

Exposure research frequently uses the concept of micro-environments to investigate the levels of pollutants humans are exposed to. Transport micro-environments represent are typically ones in which humans are potentially exposed to high pollutant concentrations on a regular basis.

The advancement in transport modes in the last few decades has resulted in increased mobility of people and a large proportion of the working population spend a significant time commuting. For example, in the UK average journey length by car has risen from 11.5 to 13.5 km between 1999 and 2004 and the proportion of journeys made by cars has increased from 57% to 63% (National Statistics 2006). In Great Britain, 40% of all the journeys comprise of short trips (<3.3 km) and almost 40% of these are carried out by car as compared to 55% on foot. In London alone 30% of trips are made by cars (National Statistics 2006).

A number of studies have attempted to quantify exposure to traffic related pollutants and also to relate the exposure to the travel mode. Research on the exposure of car drivers dates back to the 1960s (e.g. deBruin 1967). The majority of studies have focused on cars although, recently, results other transport modes, such as buses, trains, underground railways, bicycling or walking have been published. Table 2.5 summarises a number of studies which have investigated

**Table 2.5** Comparison of particulate matter in various transport micro-environments (Adapted from Nasir and Colbeck 2009)

Reference	Location	PM size fraction	Mode of transport	Mean concentrations $\mu\text{g}/\text{m}^3$
Nasir and Colbeck (2009)	Colchester, UK	$\text{PM}_{10}(\text{PM}_{2.5})$ ( $\text{PM}_1$ )	In car – morning	(22)(9)(6)
			In car – evening	(21)(8)(5)
			AC train – peak	(44)(14)(12)
			AC train – off peak	(21)(6)(4)
			Non-AC train – peak	(95)(30)(19)
			Non-AC train – off peak	(95)(14)(6)
Gulliver and Briggs (2004)	Northampton, UK	$\text{PM}_{10}$	In car	43
		$\text{PM}_{2.5}$	Walking	15
		$\text{PM}_1$		7
		$\text{PM}_{10}$		38
		$\text{PM}_{2.5}$		15
Gulliver and Briggs (2007)	Leicester, UK	$\text{PM}_1$		7
		$\text{PM}_{10}$	In car	24
		TSP – $\text{PM}_{10}$	Walking	18
		$\text{PM}_{10}$ – $\text{PM}_{2.5}$		15
		$\text{PM}_{2.5}$ - $\text{PM}_1$		8
		$\text{PM}_1$		5
		$\text{PM}_{10}$		35
		TSP – $\text{PM}_{10}$		19
		$\text{PM}_{10}$ - $\text{PM}_{2.5}$		22
		$\text{PM}_{2.5}$ - $\text{PM}_1$		10
Pfeifer et al. (1999)	London, UK	$\text{PM}_1$		3
Kaur et al. (2005)	London, UK	$\text{PM}_{2.5}$	Taxi	33
			Waking	27
			Cycling	33
			Bus	34
			Car	38
Briggs et al. (2006)	London, UK	$\text{PM}_{10}$ - $\text{PM}_{2.5}$ $\text{PM}_{2.5}$ - $\text{PM}_1$ $\text{PM}_1$ $\text{PM}_{10}$ - $\text{PM}_{2.5}$ $\text{PM}_{2.5}$ - $\text{PM}_1$ $\text{PM}_1$	Taxi	41
			In Car	5.87
			Walking	3.01
				1.82
				27
Adams et al. (2001)	London, UK	$\text{PM}_{2.5}$ Summer $\text{PM}_{2.5}$ Winter		6
				3
			Bicycle	34
			Bus	39
			Car	37
			Tube – above ground	29
			Bicycle	23
			Bus	38
			Car	33
			Tube – above ground	–

(continued)

**Table 2.5** (continued)

Reference	Location	PM size fraction	Mode of transport	Mean concentrations $\mu\text{g}/\text{m}^3$
Gee and Raper (1999)	Manchester, UK	$\text{PM}_{10}$	Bus	338
			Bicycle	54
Seaton et al. (2005)	London, UK	$\text{PM}_{2.5}$	Tube (in cab)	170
Bevan et al. (1991)	Southampton, UK	$\text{PM}_{3.5}$	Bicycle	135
Praml and Schierl (2000)	Munich, Germany	$\text{PM}_{10}$	Bus	153
Fondelli et al. (2008)	Florence, Italy	$\text{PM}_{2.5}$	Bus	56
			Taxi	39
Invernizzi et al. (2004)	Italy	$\text{PM}_{2.5}$	Railway – smoking	250
			Railway – nonsmoking	15
Branis (2006)	Prague, Czech Republic	$\text{PM}_{10}$	Metro	114
Leutwyler et al. (2002)	Zurich, Switzerland	$\text{PM}_{10}$	Railway – smoking	975
			Railway – nonsmoking	209
Chillrud et al. (2004)	New York, USA	$\text{PM}_{2.5}$	Subway	62
Wohnrschimmel et al. (2008)	Mexico City	$\text{PM}_{10}$ ( $\text{PM}_{2.5}$ )	Minibus	201(155)
			Bus	212(146)
			Metrobus	188(112)
Gomez-Perales et al. (2004)	Mexico City	$\text{PM}_{2.5}$	Minibus	68
			Bus	71
			Metrobus	61
Chan et al. (2002b)	Guangzhou, China	$\text{PM}_{10}$ ( $\text{PM}_{2.5}$ )	Non A/C bus	184 (145)
			A/C bus	125 (101)
			Non A/C taxi	140 (106)
			A/C taxi	88(73)
Chan et al. (2002a)	Hong Kong	$\text{PM}_{10}$ ( $\text{PM}_{2.5}$ )	Railway	50 (39)
			Tram	175(109)
			Non AC bus	112(93)
			AC bus	74(51)
			AC taxi	58
Park et al. (2008)	Korea	$\text{PM}_{10}$	High speed train	50.5
			Low speed train (A)(B)(C)	(69)(70)(83)
Kwon et al. (2008)	Korea	$\text{PM}_{10}$	Subway carriage	142
Lewne et al. (2006)	Stockholm, Sweden	$\text{PM}_{10}$	Lorry driver	57
			Bus driver	44
			Taxi driver	26
Riediker et al. (2003)	USA	$\text{PM}_{2.5}$	In car	24

(continued)

**Table 2.5** (continued)

Reference	Location	PM size fraction	Mode of transport	Mean concentrations $\mu\text{g}/\text{m}^3$
McNabola et al. (2008)	Dublin, Ireland	$\text{PM}_{2.5}$	In car (Route 1) (Route 2)	82(88)
			Bus (Route 1) (Route 2)	128(103)
			Cyclist (Route 1) (Route 2)	88(71)
			Pedestrian(Route 1) (Route 2)	63(46)
			Bus and lorry drivers	161
Han et al. (2005)	Trujillo, Peru	$\text{PM}_{2.5}$	Bus and lorry drivers	161

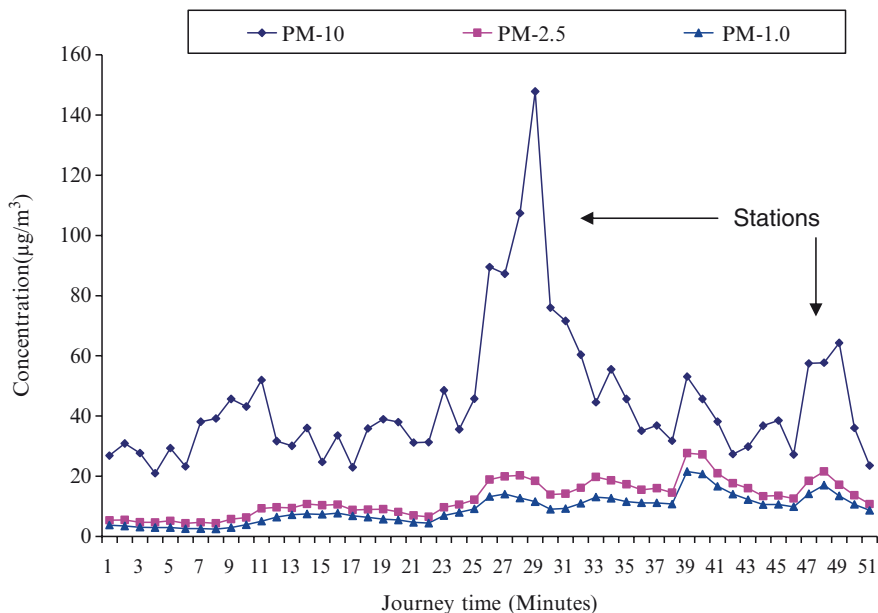
particulate matter concentrations for various modes of transport depends. It should also be noted that the exposure to pollutants does not only depend on travel mode but also varies significantly for different types of road layout and location, vehicle speed, vehicle design and ambient concentrations (Briggs et al. 2008). These studies give an indication of the exposure of individuals on particular journeys. However the results may only be of limited use in estimating exposure for other journeys.

As for a car, pollutant exposure on trains can vary with the time travel and type of train. Nasir and Colbeck (2009) have recently shown that most of particulate matter in trains was in the  $\text{PM}_{10}$  size fraction and this was generally derived from resuspension from the seating areas of the coaches. High concentrations of particulate matter during peak time reflected the contribution of the number of passengers.  $\text{PM}_{10}$  concentrations were well above those for  $\text{PM}_{2.5}$  and  $\text{PM}_1$  in both peak and off-peak journeys. Generally, the highest concentrations were recorded during the stoppage of train at the station (Fig. 2.9). Particulate matter concentrations during off peak journeys were typically half those during peak time.

## 2.7 Summary

Characterisation of indoor air pollutants is complex and requires consideration of the outdoor concentrations of the pollutants, the extent of filtering imposed by the building as air passes from outdoors to indoors, ventilation level of the building, indoor pollution sources, adsorption/desorption and chemical reactions.

There are numerous indoor sources of air pollutants. Many relate to combustion processes and others relate to human activities. Less obvious are those relating to building materials (e.g. composite woods) and products used indoors (e.g. carpets). Wallace (2009) has shown how emissions from these sources have changed over the past 50 years with a decrease in levels of “known” carcinogens and an increase in exposures to suspected endocrine disruptors. The variability of pollutants indoors is high and may be house specific. What is evident is that many



**Fig. 2.9** Typical concentration profile of  $PM_{10}$ ,  $PM_{2.5}$  and  $PM_1$  in air-conditioned train coaches

sociological factors are at play when considering exposure to indoor air pollution. Often the concentrations of air pollutants can be higher inside the dwelling of the poor. In developing countries women and their young children are likely to spend more time in the kitchen and so are at greatest risk from exposure of high levels of pollution from biofuels.

The indoor environment can be subdivided into different micro-environments (e.g. school, transport, restaurant and residential) and each may have a different source of indoor pollution.

There is a growing awareness of the importance of the indoor environment on health and exposure to indoor air pollution has been given higher attention in policy making (Harrison 2002). Several countries including Germany, Norway and Poland, have already established target concentrations for various indoor pollutants, and the UK has issued guidance on indoor air pollutants that includes numerical standards for nitrogen dioxide, carbon monoxide, formaldehyde, benzene and benzo(a)pyrene. Australia has adopted the approach of identifying indicators of good air quality rather than defining quantitative limits. The WHO Air Quality Guidelines 2005 recommended the development of guidelines specific for indoor air quality. These guidelines will cover three groups of issue: specific pollutants, biological agents and indoor combustion products.

On a global scale, the bulk of exposure to air pollution is experienced indoors, as most people spend most of their time there. Indoor concentrations are a complex interaction of various factors such as outdoor concentrations, indoor sources and

sinks, pollutants, depletion, filtration and ventilation. The types, concentration, characteristics and sources of different air pollutants differ both in outdoor and indoor air. The scientific literature offers a broad database and case studies, which can be consulted to perform the necessary estimation of a real risk given in the particular environment.

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