Environmental Research 147 (2016) 480–496

Contents lists available at ScienceDirect

Environmental Research

journal homepage: www.elsevier.com/locate/envres

Review article

Exposure to particulate matter in India: A synthesis of findings and future directions

Pallavi Pant [a], Sarath K. Guttikunda [b,c], Richard E. Peltier [a,*]

[a] Department of Environmental Health Sciences, University of Massachusetts, Amherst MA 01003, USA
[b] Institute of Climate Studies, Indian Institute of Technology, Bombay, Mumbai, India
[c] Division of Atmospheric Sciences, Desert Research Institute, 225 Raggio Parkway, Reno, NV 89512, USA

ARTICLE INFO

Article history:
Received 20 January 2016
Received in revised form
1 March 2016
Accepted 5 March 2016

Keywords:
Air pollution
Personal exposure
PM
Microenvironment
India

ABSTRACT

Air pollution poses a critical threat to human health with ambient and household air pollution identified as key health risks in India. While there are many studies investigating concentration, composition, and health effects of air pollution, investigators are only beginning to focus on estimating or measuring personal exposure. Further, the relevance of exposures studies from the developed countries in developing countries is uncertain. This review summarizes existing research on exposure to particulate matter (PM) in India, identifies gaps and offers recommendations for future research. There are a limited number of studies focused on exposure to PM and/or associated health effects in India, but it is evident that levels of exposure are much higher than those reported in developed countries. Most studies have focused on coarse aerosols, with a few studies on fine aerosols. Additionally, most studies have focused on a handful of cities, and there are many unknowns in terms of ambient levels of PM as well as personal exposure. Given the high mortality burden associated with air pollution exposure in India, a deeper understanding of ambient pollutant levels as well as source strengths is crucial, both in urban and rural areas. Further, the attention needs to expand beyond the handful large cities that have been studied in detail.

1. Introduction

Air pollution is a global environmental burden, and has been identified as a significant public health risk. Human exposure to ambient (outdoor) air pollution (AAP) and household air pollution (HAP) are important risk factors for morbidity and mortality, particularly in the developing countries (Lim et al. 2012). Exposure is a broadly used term that can be used to indicate both qualitative and quantitative measures of a pollutant, and can be defined in several ways, although the most generic definition takes into account the pollutant concentration and the amount of time an individual spends in contact with that pollutant (Moschandreas et al. 2002; Morawska et al. 2013). Various approaches for defining and estimating exposure are discussed by Zartarian et al.(2004). Over the last fifty years, a majority of the air pollution research efforts have focused on characterization and speciation of ambient particulate matter (PM), and its sources. While ambient PM concentrations are an important contributor to personal exposure (PE), they often do not correlate with, or are lower than levels of air pollutants individuals are exposed to (Monn et al. 1997; Huang et al. 2012).

Exposure to PM is a globally relevant topic because it has been linked to a number of deleterious health outcomes. Recent work has shown associations with exposures to generation of oxidative stress, and particle toxicity can vary based on chemical constituents (e.g. trace metals and polycyclic aromatic hydrocarbons [PAHs]), which in turn, can trigger inflammation and oxidative stress (Valavanidis et al. 2008; Shah et al. 2013; Kelly 2003). Exposure to PM has been associated with morbidity and mortality due to respiratory, cardiovascular and cerebrovascular diseases

Abbreviations: AAP, Ambient Air Pollution; AQI, Air Quality Index; BC, Black Carbon; CO, Carbon Monoxide; COPD, Chronic Obstructive Pulmonary Disease; CPCB, Central Pollution Control Board; EBC, Exhaled Breath Condensate; ETS, Environmental Tobacco Smoke; FEV, Forced Expiratory Volume; GBD, Global Burden of Disease; HAP, Household Air Pollution; HIA, Health Impact Assessment; I/O, Indoor/Outdoor; LPG, Liquefied Petroleum Gas; LUR, Land Use Regression; NAAQS, National Ambient Air Quality Standards; NCR, National Capital Region; OC, Organic Carbon; PAH, Polycyclic Aromatic Hydrocarbon; PE, Personal Exposure; PM, Particulate Matter; PNC, Particle Number Concentration; PSD, Particle Size Distribution; ROS, Reactive Oxygen Species; RSPM, Respirable Suspended Particulate Matter; SAFAR, System of Air Quality Weather Forecasting and Research; SPM, Suspended Particulate Matter; VOC, Volatile Organic Compound; WHO, World Health Organization

* Corresponding author.
E-mail address: rpeltier@schoolph.umass.edu (R.E. Peltier).

http://dx.doi.org/10.1016/j.envres.2016.03.011
0013-9351/

(Harrison and Yin 2000; Chen and Lippmann 2009; Raaschou-Nielsen et al. 2016). In 2012, outdoor air pollution was classified as a group 1 carcinogen (i.e. carcinogenic to humans) (Benbrahim-Tallaa et al. 2012), and a recent World Health Organization (WHO) estimate suggests that in the same year, more than seven million premature deaths were due to air pollution exposure, with more than 80% being in the Pacific and South Asian regions (WHO 2014).

Several studies have concluded that use of fixed outdoor monitoring stations to estimate PE can lead to exposure misclassification (Huang et al. 2012; Dons et al. 2011; Koistinen et al. 2004), and as a result, the focus is rapidly shifting to direct characterization and quantification of PE to various air pollutants. For example, Hsu et al. (2012) reported personal PM exposures in the United States to be higher than indoor levels and studies in Europe have reported similar results (Broich et al. 2012; Johannesson et al. 2007). However, it is important to note that this may not be true in developing countries where ambient pollution levels are often very high. A recent global study on ambient PM concentrations reported a twofold difference between average population weighted $PM_{2.5}$ (PM with aerodynamic diameter less than 2.5 μm) concentrations in North America (12 $\mu g/m^3$) and Asia (38 $\mu g/m^3$) with the highest levels of $PM_{2.5}$-associated mortality in Asia due to high population densities (Apte et al. 2015). Similarly, intake fractions are also reported to be higher for Asian and Pacific countries compared with developed countries (Apte et al. 2012). Further, the population density levels in the developing countries are higher, which can contribute to higher overall exposure levels. A case in point is India where more than 50% of the population lives in areas where ambient $PM_{2.5}$ levels exceed the annual $PM_{2.5}$ (40 $\mu g/m^3$) Indian National Ambient Air Quality Standard (NAAQS) and less than 0.01% of the population lives in areas that meet the WHO $PM_{2.5}$ guideline of 10 $\mu g/m^3$ (Brauer et al. 2015). This is in stark contrast to Europe, where only 10-14% of the population lives in areas that exceed the European $PM_{2.5}$ guideline value (25 $\mu g/m^3$) (EEA 2014).

Very little is known about exposure to air pollutants in developing countries in Asia, Africa and Latin America where communities are saddled with burgeoning population levels and face rapid growth in urbanization and industrialization. There are several factors that can affect the exposure to PM in developing countries. First, there are several unique sources that are not found in developed countries (e.g. open waste burning, open cooking, use of animal dung as a fuel, use of diesel generator sets), and these sources have not been characterized well. Secondly, there are sources such as solid fuel burning which are significant in terms of indoor as well as ambient air pollution in developing countries, but do not contribute much to exposures in developed countries. Third, humans spend the largest proportion of their time in indoor microenvironments in the developed countries (Ott 1995; Diapouli et al. 2013) but this may not be true for developing countries. Finally, due to high poverty (especially energy poverty) levels and lower quality of life, certain sections of the population have higher susceptibility to negative health effects linked with air pollution. Thus, there is a need for a concerted global effort to generate exposure estimates for developing countries, yet most of our understand of health effects and air pollution are derived from communities that are cleaner and more developed.

The Global Burden of Disease (GBD) study in 2010 was a significant step forward in estimating exposures, and characterizing health risks associated with exposure to air pollution. In India, the GBD study identified both AAP and HAP as key risk factors in terms of disease for the Indian population (Lim et al. 2012), and air pollution, both AAP and HAP has been identified as one of the critical social determinants of health in India (Cowling et al. 2014). Nearly 100,000 premature deaths in India are linked to air pollution exposure (PHFI 2014) and in Delhi (Fig. 1) alone, between 7,350 and 16,200 premature deaths have been attributed to PM exposure (Guttikunda and Goel 2013). For a rapidly developing

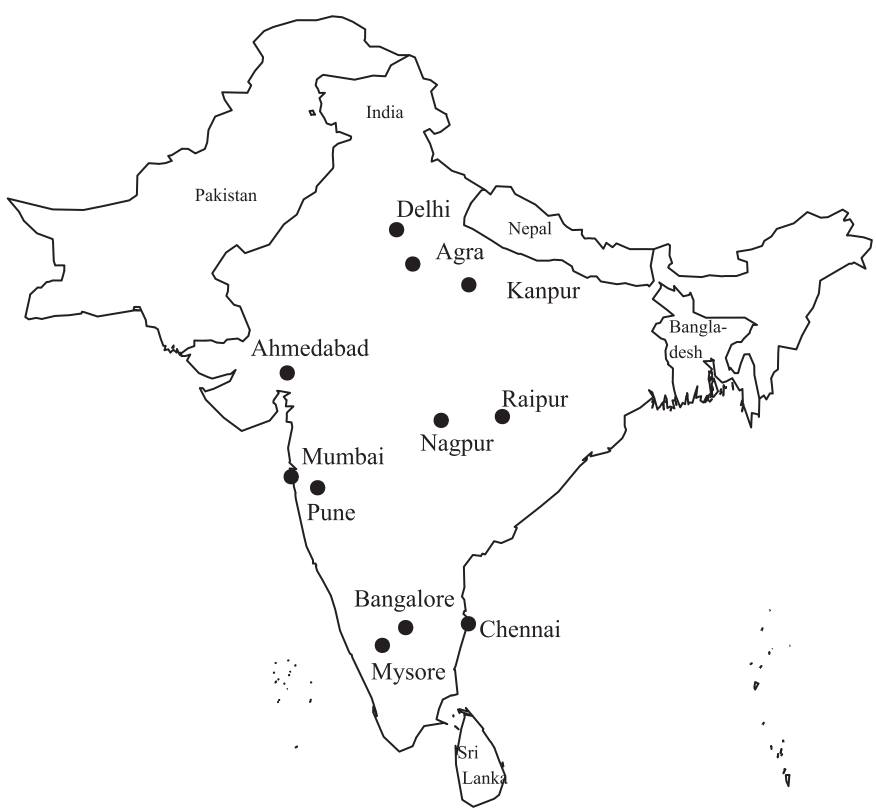

Fig. 1. Location of major Indian cities where indoor and/or personal exposure studies have been conducted, with results published in the literature.

Table 1
Summary of ambient PM concentrations from several cities across India (mass concentrations in μg/m³).

City	Sampling year	Type	PM_{10}	$PM_{2.5}$	PM_1	Reference
Delhi	September 2010-August 2012	Avg ± SD	222 ± 142	130 ± 103		Tiwari et al. (2014)
Delhi	January-December 2007		219 ± 84	97 ± 56		Tiwari et al. (2009)
Agra	May 2010-December 2012			65.4 ± 3.1		(Dubey et al. 2015)
Agra	November 2011-June 2012			60 ± 39		(Bergin et al. 2015)
Kanpur	Winter 2013–14			155.28 ± 28.53		(Singh and Gupta 2016a)
Raipur	January- December 2006		246 ± 68	115 ± 36		(Giri et al. 2013)
Pune	June 2011-May 2012		113.8 ± 51.6	72.3 ± 31.3		Yadav and Satsangi (2013)
Rajim	October- November 2011			200 ± 127		Nirmalkar et al. (2015)
Kolkata			445 ± 210	313 ± 181		Das et al. (2015)
Hyderabad			105.2 ± 28.6	72.6 ± 18		Guttikunda and Kopakka (2014)
Hyderabad	June 2004–May 2005			135.1 ± 37.9		Gummeneni et al. (2011)
Mumbai	2007–2008			69 ± 20		Abba et al. (2012)
Mumbai			138.4 ± 46.4			Herlekar et al. (2012)
Bhubaneshwar	December 2012-February 2013			60.72 ± 20.1[a]		Panda et al. (2015)
Delhi	December 2012-February 2013			186.25 ± 47.46		
Agra	October 2007-March 2009	Range	216–301	112–212	82–143	Massey et al. (2012)
			155–172	91–157	71–134	
Jorhat	January 2007-January 2008			108–143		Baruah and Khare (2010)
Ahmedabad	2005–2008		17–327			Bhaskar and Mehta (2010)
Chennai			49.7–212.6	28.3–105.5		Srimuruganandam and Shiva Nagendra (2012)
Delhi	June 2013-January 2014			58–276[b]		Pant et al. (2015)
Kanpur	July 2008-May 2009				31–199	Chakraborty and Gupta (2010)

[a] 8 h average.
[b] 12 h average.

country of more than a billion people, there is very limited central monitoring data and even less data describing personal exposures although several studies have identified health impacts associated with exposure to air pollution in India.

This review paper focuses primarily on exposure to PM and its components in India although gaseous pollutants are also briefly discussed. Specific questions include:

- What do we understand about exposure to air pollution (indoor and outdoor) in the Indian context?
- How does the exposure to air pollutants vary from other geographical regions?
- What are the key data gaps and how can they be addressed?

Exposure to air pollutants can occur through two major routes-personal exposure to ambient or indoor air pollutants during day-to-day activities (cooking, commuting etc.), and occupational exposure at the workplace. Though HAP has been identified as one of the most important health risk factors globally, it is not discussed in detail in this paper since there has been extensive research on this subject and several reviews have addressed this source in the past (Smith 2000; Balakrishnan et al. 2011; Martin et al. 2013). Additionally, studies focused on occupational exposure are not discussed in this review (Semple et al. 2008; Suresh et al. 2000; Gupta et al. 2011).

The literature search for this review was primarily performed using Web of Knowledge, Science Direct, PubMed and Google Scholar using various combinations of keywords such as “exposure”, “indoor air”, “personal exposure”, “India”, “particulate matter”, “health effects” and literature between 1985 and 2015 has been included in this review. While efforts were made to include all published peer-reviewed literature, we acknowledge that some articles may have been missed in the review and thus not discussed here.

2. Ambient air pollution

PM_{10} concentrations in India often exceed the national air quality standards, and in 2010, 140 out of 176 cities were found to exceed the $PM_{2.5}$ National Ambient Air Quality Standard (NAAQS) standard values (Gargava and Rajagopalan 2015). Key sources of PM in India include vehicles, industries, power plants (coal combustion), dust, construction, biomass combustion (for cooking and heating) and waste burning (Liu et al. 2013; Pant et al. 2015; Srimuruganandam and Shiva Nagendra 2012; Abba et al. 2012a; Villalobos et al. 2015). HAP can also be an important contributor to ambient $PM_{2.5}$ particularly in rural areas where wood/biomass and dung are used for cooking as well as heating (Rehman et al. 2011; Chafe et al. 2014). In addition, there are several unique sources including funeral pyres (Chakrabarty et al. 2014) and festive fireworks and biomass burning (Beig et al. 2013a; Pervez et al. 2015) that are not very well understood. Contribution of specific sources can vary significantly across cities, and meteorological parameters play an important role in determining the ambient concentrations as is evident by the lowest concentrations during the monsoon (August-September) period (Trivedi et al. 2014; Yadav et al. 2014). Further, it is reasonable to assume varying degrees of spatial and temporal variability due to the influence of local activities (Both et al. 2013; Saraswat et al. 2013). Annual PM_{10} and $PM_{2.5}$ concentrations in New Delhi were reported to be 222 ± 142 μg/m³ and 130 ± 103 μg/ m³ (Tiwari et al. 2014) while an earlier study reported annual average concentrations of 219 ± 84 μg/ m³ and 97 ± 56 μg/ m³ (Tiwari et al. 2009). In comparison, lower concentrations have been reported for Chennai where annual PM_{10} and $PM_{2.5}$ concentrations range between 49.7 and 212.6 μg/ m³ and 28.3-105.5 μg/ m³ (Srimuruganandam and Shiva Nagendra 2012). In Mumbai, on the other hand, reported $PM_{2.5}$ concentrations range from 69 ± 20 μg/m³ at a background location to 95 ± 36 μg/m³ at an industrial location (Abba et al. 2012a) and in Hyderabad, average PM_{10} concentration was 135.1 ± 37.9 μg/m³ (Guttikunda and Kopakka 2014). Only a few studies have measured PM_1 concentrations, and in Kanpur, concentrations were reported to be in the range of 31–199 μg/m³ (Chakraborty and Gupta 2010) while in Agra, average concentrations ranged from 112 μg/m³ at a roadside location to 104 μg/m³ at an urban site (Massey et al. 2012). Between 2000 and 2010, $PM_{2.5}$ concentrations have been reported to increase in several Indian cities including Delhi, Mumbai, Kolkata, Patna, Kochi, Amritsar and Jamshedpur (Dey et al. 2012). A summary of reported concentrations is presented in

Table 1.

Within India, the largest cities and the Indo-Gangetic basin have been identified as areas with the most severe air pollution (Guttikunda et al. 2014; Ram and Sarin 2011; Dey et al. 2012). Concentrations of PM constituents (e.g. metals, PAHs) have been reported to be very high in some cases, and several toxic metals (including Pb, Cu, Zn and Ba) have been reported to be highly water soluble (Yadav and Satsangi 2013). A number of studies have reported enrichment of elements associated with anthropogenic emissions in PM_{10} and $PM_{2.5}$ (Pant et al. 2015; Shridhar et al. 2010; Sudheer and Rengarajan 2012), and likewise, high PAH concentrations have also been reported in several cities (Sarkar and Khillare 2013; Abba et al. 2012b; Pant et al. 2015; Cheng et al. 2013). Secondary PM is also expected to be an important contributor to total PM concentrations, and while there have been no concerted efforts to quantify primary and secondary PM (Pant and Harrison 2012), a number of recent studies have started addressing primary and secondary aerosol components. Ram and Sarin (2011) attributed between 35 and 45% of total OC to secondary OC (SOC) in Kanpur while Villalobos et al. (2015) attributed 20-68% of $PM_{2.5}$ to SOC. In Delhi, Pant et al. (2015) attributed 33–67% of total OC to SOC across seasons; in Allahabad, Ram and Sarin (2010) estimated nearly 30% of total OC to be secondary in nature and in Agra, Villalobos et al. (2015) SOC was estimated to contribute between 25% and 42%. Several reviews have summarized the levels of air pollution in India including Krishna (2012) which provides an overview of the research on aerosols in India and Guttikunda et al. (2014) who presented a comprehensive review of air pollution sources, trends and characteristics. Pant and Harrison (2012) and Banerjee et al. (2015) discuss receptor modelling studies in India in detail. In 2011, the Central Pollution Control Board (CPCB) released a six-city source apportionment study which was first of its kind, and included characterization of ambient PM as well as sources (CPCB 2010; Patil et al. 2013). However, a majority of source apportionment studies in India have focused on suspended particulate matter or PM_{10} with a limited number of studies on $PM_{2.5}$ and PM_1.

In the last few years, there have been many significant improvements in monitoring of ambient air pollution, and launch of programs such as SAFAR (System of Air Quality Weather Forecasting and Research) is helping improve the understanding of ambient air pollution in India (Beig et al. 2013a; Beig et al. 2013b; Trivedi et al. 2014). However, the data collected as part of such monitoring networks is not available in the public domain, and in the absence of standardized operating protocols, data quality remains a concern. In 2015, the Indian national government launched the national Air Quality Index (AQI) with the aim of providing information about air pollution in a relevant format for general public using a color-coded index (CPCB, 2014). The index is based on several key air pollutants including PM_{10} and $PM_{2.5}$, ozone (O_3), sulfur dioxide (SO_2), ammonia (NH_3), nitrogen dioxide (NO_2), carbon monoxide (CO) and lead (Pb). However, the AQI is also plagued with questions on data quality. The Indian Institute of Tropical Meteorology (IITM) has also designed an AQI (Beig et al. 2010) although the two indices are not inter-comparable.

Further, the Indian indices are not directly comparable to US or EU AQIs due to different pollutant concentration thresholds for AQI calculation. Fig. 2 presents average daily $PM_{2.5}$ concentrations for four Indian cities based on the publically available data collected by the US Consulates.

3. Indoor air pollution

In indoor environments, there are several different sources of exposure to air pollutants including consumer products (cleaning, fresheners, hair sprays, and deodorants), solid fuel combustion (for cooking, heating), cooking, use of incense sticks/candles, use of printers, biological materials (pollen, mites), environmental tobacco smoke (ETS), kerosene lamps, building products (e.g. varnish, adhesives, paints), and other household activities (e.g. cleaning) (Meng et al. 2004; Jones 1999; Colbeck et al. 2010; Wallace 1995; Spengler and Chen 2000; Nazaroff 2004). Infiltration of ambient aerosols and generation of particles due to reactions also contribute to the air pollutant concentrations observed indoors (Diapouli et al. 2013; Long et al. 2000; Morawska et al. 2013). In fact, ambient particles can contribute between 25 and 65% of the total indoor particle load, depending on the characteristics of the house (Meng et al. 2004). Similar to the ambient environment, fine particles are generated from combustion-related activities including cooking, smoking and use of solid fuel for heating or cooking while coarse particles are generated by mechanical activities such as walking, vacuuming, dusting and sweeping. However, some studies have argued that resuspension activities such as walking and dusting are important sources for both fine and coarse particles in the indoor environment (Long et al. 2000). Bioaerosols (including pollen, dander) and human skin also contribute to indoor particle concentrations, and presence of pet animals can also influence the concentration of bioaerosols, and thereby total PM concentration (Chen and Hildemann 2009). Specific toxic species can also be present in high concentrations in indoor dust to which people are exposed every day, and recently Ma and Harrad (2015) reported that indoor concentrations of PAHs have remained constant over the last three

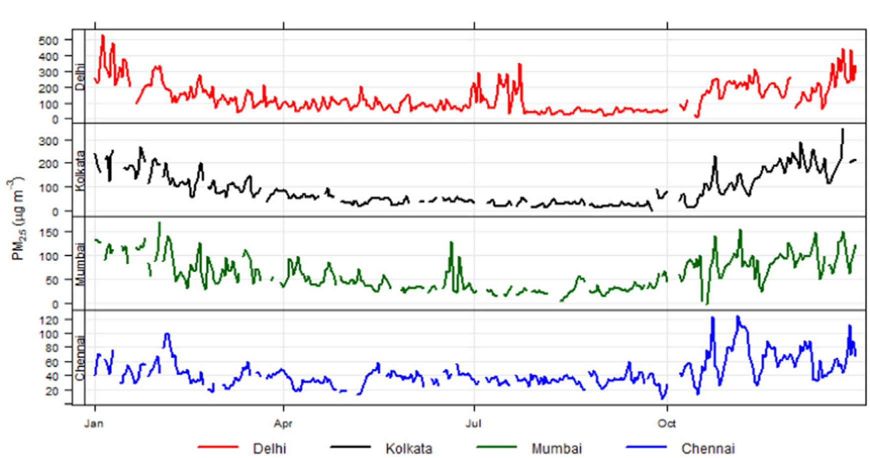

Fig. 2. Daily average $PM_{2.5}$ concentration (in μg/m3) for the year 2014 for select metro cities (USEPA, 2015).

decades, and inhalation, together with diet, are the two major routes of exposure to PAHs associated with indoor dust.

Concentrations observed in indoor microenvironments in India are much higher than levels reported in the literature from other countries, and a wide range of PM concentrations have been reported in literature (20-1000 μg/m^3) with variations between urban and rural areas. In central India (Raipur-Bhilai), Gadkari and Pervez (2008) observed average PM_5 concentrations ranging from 135 to 551 μg/m^3 across houses with different ventilation mechanisms, and in Delhi, Saksena et al. (2007) reported concentrations as high as 350 μg/m^3 (office/shop) and 640 μg/m^3 (bedroom) in indoor environments. In Kanpur, Devi et al. (2009) reported $PM_{2.5}$ concentrations ranging from 22.5 to 165.2 μg/m^3 in indoor microenvironments. In Agra, indoor and outdoor $PM_{2.5}$ concentrations at a roadside site were recorded as 161 ± 62 and 160 ± 62 μg/m^3, while at an urban site, the concentrations were 109 ± 48 and 123 ± 45 μg/m^3 respectively (Massey et al. 2012). For the same size fraction, concentrations ranging from 34 to 60 μg/m^3 in summer and 197-936 μg/m^3 in winter across houses of different sizes and occupancy have been observed in Delhi (Kulshreshtha et al. 2008). In Pune, Satsangi et al. (2014) observed average indoor $PM_{2.5}$ and PM_{10} concentrations of 89.7 ± 43.2 μg/m^3 and 138.2 ± 68.2 μg/m^3 respectively. Compared to these studies, Varshney et al. (2015) reported relatively lower indoor PM concentrations in Agra in urban (45.33 μg/m^3), roadside (36.71 μg/m^3) and rural (71.23 μg/m^3) households, and Habil et al. (2016) reported 123.79 ± 35.32 μg/m^3. Indoor concentrations in hospitality venues (bars, cafés, restaurants etc.) were reported to be in the same range with an average concentration of 207 μg/m^3 although a much wider range (20-748 μg/m^3) of concentrations were recorded (Lee et al. 2010). A summary of reported PM concentrations from India is presented in Table 2.

In comparison, relatively lower concentrations are reported in developed countries. In a study in the USA, median $PM_{2.5}$ concentrations were reported as 14.4 and 15.5 μg/m^3 for indoor and outdoor environments while PE $PM_{2.5}$ level was recorded at 31.4 μg/m^3 (Meng et al. 2004). In Oslo (Norway), median $PM_{2.5}$ concentrations of 5-7 μg/m^3 and 6-8 μg/m^3 were reported for indoor and outdoor environments while in Gothenburg (Sweden), median $PM_{2.5}$ concentrations were reported to be 8.4, 8.6 and 6.4 μg/m^3 for personal, indoor and outdoor air samples while in case of PM_1, the concentrations were 5.4, 6.2 and 5.2 respectively (Lazaridis et al. 2008; Johannesson et al. 2007).

It is clear that indoor as well outdoor concentrations in cites in developing countries are higher than levels reported in developed countries and this has important implications for exposure levels.

3.1. Factors affecting indoor air quality

Contributors to the indoor air quality include the ambient pollutant levels and indoor pollution sources and their strength and a range of factors including air exchange rate, meteorological conditions and rate of particle deposition can determine the extent of pollutant build-up in indoor microenvironments (Cyrys et al. 2004; Hsu et al. 2012; Morawska et al., 2013; Nazaroff, 2004). In particular, smoking (environmental tobacco smoke-ETS) and cooking (both fuel combustion and cooking processes) have been identified as important sources of indoor air pollution (Monn 2001), and AAP is also believed to contribute significantly to indoor pollutant concentrations. Choice of cooking fuel is often linked with a family's socio-economic status, and energy choices can significantly influence indoor air quality. In addition to direct emissions from indoor sources and the contribution of ambient pollution, particle formation can also occur via reactions in the indoor environment which can further deteriorate indoor air quality (Sundell, 2004). Key factors that can influence indoor particle concentrations are discussed below.

3.1.1. Air exchange and filtration

Air filtration rate is an important factor in that context, and indoor/outdoor pollutant ratios are dictated by a complex set of variables including building design, pollution strength and local meteorology (Yocom 1982). Since the level of ambient PM in a majority of Indian cities is quite high, the issue of infiltration of ambient PM, particularly with respect to air exchange rate, ventilation and building design and leakages becomes important in the assessment of indoor PM. Air exchange rate is defined as, "*the sum of the flow rates into a building, divided by the interior volume served*" (Nazaroff, 2004). Breen et al. (2014) have discussed the various methods for air exchange rates and Nazaroff (2004) provides a detailed account of the physical processes associated with particle generation.

There are a limited number of studies reporting both indoor and outdoor concentration of PM. Most buildings in India are

Table 2
Summary of indoor PM concentrations in India (PM mass concentrations in μg/m^3, number concentrations in #/cm^3, U-urban, RO-roadside, R-rural, ME- microenvironment).

References	City	Year	ME / Season	PM_1	$PM_{2.5}$	PM_5	PM_{10}	Particle Number
Prasad et al. (2003), Saksena et al. (2007)	Delhi (U)	February–March 1997	Living Room			480		
			Bedroom			640		
			Workplace			350		
Mönkönnen et al. (2005)	Nagpur (U)	2002	Flat					4400-101,000
Gadkari and Pervez (2007)	Raipur (U)	March–June 2004	Indoor (poorly ventilated)			550.9 ± 27.4		
			Indoor (well ventilated)			134.7 ± 11.23		
Srivastava and Jain (2007)	Delhi (U)	April-June 2000	Laboratory			168.7–196.1		
Kumar et al. (2008)	Delhi	June-July 2004				0.96 ± 0.36		
Kulshrestha et al. (2008)	Delhi (U)	April 2004-February 2005	Indoor (monsoon)	142	189		349	
			Indoor (summer)	31	50		155	
			Indoor (winter)	403	461		654	
Massey et al. (2012)	Agra (RO)	October 2007-March 2009	Annual	111 ± 32	161 ± 62	211 ± 59	247 ± 78	
	Agra (U)		Annual	99 ± 41	109 ± 48	145 ± 62	181 ± 84	
Devi et al. (2013)	Kanpur (U)	September 2009-May 2010	Pre-monsoon Post-monsoon		30.5 ± 2.7 37 ± 26			
Satsangi et al. (2014)	Pune (U)	June 2012-May 2013			89.7 ± 43.2		138.2 ± 68.2	
	Pune (R)				197.5 ± 84.3		287 ± 92	
Habil et al. (2016)	Agra	December 2013-February 2014	Home		123.79 ± 35.32			
			Office		112.28 ± 36.45			

naturally-ventilated which allows infiltration of ambient air (Goyal and Kumar 2013) which indicates that in certain cases, PM concentrations in ambient air can significantly influence indoor concentrations. Another issue of considerable importance is ventilation, and previous studies have highlighted the issue of poor ventilation in India which often results in higher indoor concentrations of pollutants (Gadkari and Pervez 2007; Mönkkönen et al. 2005). For example, PM_5 concentrations were found to be several fold higher in a poorly-ventilated house compared to a well-ventilated house (550.85 ± 27.39 and 134.67 ± 11.28 $\mu g/m^3$ respectively) (Gadkari and Pervez 2007). Similar to this finding, Lee et al. (2010) reported a negative correlation between air exchange and indoor $PM_{2.5}$ concentrations in hospitality venues, indicating higher $PM_{2.5}$ levels for poorly-ventilated buildings.

Calculation of indoor/outdoor (I/O) ratios of pollutants can give an indication regarding the rate of infiltration of particles and the ratios can vary significantly between naturally- and mechanically –ventilated buildings. Several studies have reported I/O ratios in naturally-ventilated buildings for PM, although ratios lower than 1 have been reported for mechanically-ventilated buildings. In New Delhi, for example, I/O ratios were found to range between 0.2 and 3.2 for $PM_{2.5}$ and 0.17 and 2.9 for PM_1 (Goyal and Kumar, 2013) and in Nagpur, Mönkonnen et al. (2005) reported I/O ratios ranging between 0.72 and 1.36 in urban houses. Studies elsewhere have reported ratios values greater than 1 in case of indoor sources; in Australia, median I/O ratios of 0.7 (PM_{10}) and 0.9 ($PM_{2.5}$) were reported for a naturally-ventilated building in the absence of indoor sources, while in the presence of indoor sources, the ratios were observed to be 1.41 and 1.21 respectively (Morawska et al., 2013). It is important to note that the indoor concentrations can also vary due to prevalent meteorological conditions (e.g. temperature, moisture, wind) and this often drives the seasonal variability of pollutant concentrations in indoor microenvironments. Further, specific indoor sources including solid fuel combustion and smoking can lead to higher I/O ratios.

Within indoor microenvironments, use of ceiling fans can influence the rate of resuspension as well as particle deposition in indoor microenvironments (Rosati et al., 2008), while the use of air conditioners can affect the air exchange rate, and therefore the overall indoor concentrations. Correlation between indoor and outdoor PM concentration is often higher in case of summer due to the better air exchange due to open windows and doors (Massey et al., 2012) although in households where air conditioning is used, the infiltration is lower during summer as well, due to the limited air exchange (windows are closed). In Delhi, for example, the contribution of fine particles to PM_{10} was found to be higher in case of air-conditioned buildings (Goyal and Kumar, 2013) although overall particle concentrations tend to be lower in an air-conditioned building (Jai Devi et al., 2009; Ashok et al., 2014).

Although there is limited data on indoor and outdoor PM data, all studies have reported very weak correlation between ambient (outdoor) and indoor PM concentrations (Habil and Taneja, 2011; Kulshreshtha and Khare, 2011).

3.1.2. Energy and fuel choice

A significant proportion of households in India use traditional fuel sources including wood, coal, cow dung and kerosene (Balakrishnan et al., 2013a; Choi et al., 2014), and factors such as income, education, socio-economic status and age have been reported to influence fuel choice (Andresen et al., 2005; Duflo et al., 2008; Dutta and Banerjee, 2014). The issue of energy choice is also linked to the greater social and environmental justice argument since use of LPG is associated with a higher cost, and the use of solid fuel and/or kerosene is higher in case of poorer neighborhoods. A detailed discussion on the larger political and socio-economic aspects of energy access in India can be found elsewhere.

Household energy use varies significantly between urban and rural areas, and while liquid petroleum gas (LPG) and kerosene use is higher in case of urban areas, biomass (wood, crop residue) is more popular in rural areas. According to the 2011 National Census (Census of India, 2011), while 65% of urban households use LPG, the percentage of rural households using LPG is merely 11.4%. As a result, indoor concentrations are typically higher in houses in rural areas due to poor ventilation and greater reliance on the use of traditional stoves. For example, Satsangi et al. (2014) reported $PM_{2.5}$ concentrations of 197.5 ± 84.3 $\mu g/m^3$ and 89.7 ± 43.2 $\mu g/m^3$ in rural and urban areas in Pune (Maharashtra). Dirty fuels such as biomass, coal and kerosene can also contribute to high concentrations of PAHs, VOCs and metals in the indoor environment (Pandit et al., 2001). HAP is one of the key public health issues in India, particularly in the semi-urban and rural areas where the percentage of households using solid fuel (wood, dung) is very high. HAP, particularly the air pollutants generated due to combustion of wood/biomass has been the subject of several studies, and detailed analyses have been undertaken to assess the role of exposure to wood/biomass smoke in adverse health effects (Balakrishnan et al., 2002; Dherani, 2003; Bhargava et al., 2004; Sukhsohale et al., 2013; Agrawal and Yamamoto, 2015). However, in cases where cleaner cooking fuels (e.g. LPG) are used in rural areas, exposure levels, and associated health impacts such as decrease in lung function are higher in urban areas compared to rural areas (CPCB 2008). As discussed earlier, this review does not focus on HAP in India since the topic has been covered in several other reviews (Smith, 2000; Balakrishnan et al., 2011).

For the kitchen microenvironment, not surprisingly, PM concentrations are typically higher in cases where biomass is used compared with kerosene and LPG (Table 3). However, surprisingly, Kulkarni and Patil (1999) found no correlation between the type of fuel used and the PM_5 levels in the indoor environment. Similar to PM concentrations, the incidence of disease and adverse respiratory symptoms are higher when biomass or kerosene is used compared to LPG (Mönkkönen et al., 2005; Choi et al., 2014).

3.1.3. Environmental tobacco smoke

ETS has been identified as a significant contributor to indoor PM (Broich et al., 2012). India is among the largest consumers of tobacco, and forms of consumption include cigarettes, hookah, cigars, pipes, bidi (naturally cured dry tobacco flakes rolled in tendu leaves (Rahman and Fukui, 2000; Lee et al., 2013)) and other local tobacco products including chillum (Chhabra et al., 2001; Kaur and Prasad, 2011). Type of products consumed vary between households with different incomes, and while bidi is more popular in poorer households, cigarette use is more common in the more affluent households (Rahman and Fukui, 2000; Chhabra et al., 2001). On average, bidi produces a higher amount of nicotine, tar and carbon monoxide (CO) and is associated with a higher incidence of chronic respiratory symptoms compared to use of cigarettes (Rahman and Fukui, 2000; Malson et al., 2001).

Exposures are typically higher in rural areas compared to urban areas, and number of male smokers is several fold higher than the number of female smokers (Garg et al., 2012; Grills et al., 2015). Household ETS exposure has also been associated with childhood asthma (Cheraghi and Salvi, 2009). Southeast Asian countries are reported to have the highest levels of exposure to SHS across the globe (Singh and Lal, 2011). Rahman et al. (2003) found bidi smokers to have a high risk for oral cancer compared to cigarette smokers, although the authors cited limitations to the analysis. In a review paper, Gupta et al. (2002) identified several studies in India that have quantified the effects of exposure to passive smoking in India, and linked passive exposure to ETS with higher incidence of asthma, and reduction in lung function as well as

Table 3
Summary of indoor PM concentrations in India across households with different fuel types (PM mass concentrations in μg/m^3, particle number concentration in #/cm^3).

References	City/ Region	Cooking fuel	Concentrations	$PM_{2.5}$	SPM	Particle Number
Andresen et al. (2005)	Mysore	Kerosene	Personal exposure	123 ± 16		
		LPG		56 ± 7		
		Kerosene	Indoor	109 ± 14		
		LPG		57 ± 7		
Balakrishnan et al. (2002)	Tamil Nadu (rural)	Kerosene	Personal exposure	132 ± 41		
		LPG		83 ± 6		
		Wood		1307 ± 50		
		Agricultural produce		1535 ± 115		
		Kerosene	Indoor	80 ± 4		
		LPG		78 ± 5		
		Wood		847 ± 248		
		Agricultural produce		1327 ± 207		
Saksena et al. (2007)	Delhi	LPG	Kitchen		890	
Mönkönnen et al. (2005)	Nagpur	Kerosene	Indoor	41–146.2		17.5–252
		LPG		27–162.6		11.9–79.3
		Kerosene	Cooking	67.7–576.8		42.9–92
		LPG		29.7–565.2		18.5–125.6

higher incidence of cancer. Indoor smoking can significantly elevate indoor PM concentrations and ETS can contribute to very high indoor concentrations for short periods of time (Jai Devi et al., 2009). In 2003, smoking was banned in workplaces and certain categories of public places in India but despite the ban, compliance is imperfect, and the general population is routinely exposed to SHS (Deshpande et al., 2010). However, a recent study has highlighted that a number of venues sampled in New Delhi were smoke-free (Jackson-Morris et al., 2016). Globally, median air nicotine concentrations of 0.2 and 3.5 μg/m^3 have been reported for non-smoking and smoking entertainment (bar/nightclub) venues (Jones et al., 2012) whereas home air nicotine concentrations of 0.18 and 0.01 μg/m^3 have been reported in households with and without smokers respectively (Wipfli et al., 2008). In India, Kaur and Prasad (2011) analyzed nicotine concentrations in public places, and reported some variability across the cities, but bars and restaurants were found to have the highest nicotine levels (median levels of 0.74-4.53 μg/m^3). Yang et al., (2015) reported variations in the average nicotine concentration in public spaces (0.04-1.53 μg/m^3 and 0.03-3.77 μg/m^3 before and after smoking ban) in Ahmedabad, and detected lower concentrations after the smoking ban (0.06 μg/m^3 vs 0.03 μg/m^3). SHS exposure has been reported to have reduced after the introduction of the smoking ban in public places (Deshpande et al., 2010; Yang et al., 2015). However, hookah bars are exempt from the smoking ban and continue to report very high indoor PM concentrations (800-1300 μg/m^3), causing short-term high PM exposures for both the employees and the customers (Deshpande et al., 2010; Raute et al., 2011).

3.1.4. Other sources

Several studies have conducted source apportionment of indoor aerosols in India and key sources/activities identified as contributors to indoor PM concentrations include cooking, infiltration of ambient air (especially in homes near specific sources such as roads), incense burning, indoor dust, space heating (wood/coal) and ETS (Gadkari and Pervez, 2007; Saksena et al., 2007; Jai Devi et al., 2009; Masih et al., 2012; Kulshrestha et al., 2014; Roy et al., 2009). Similar observations have been reported elsewhere where cooking (stir-frying), use of candles, ETS and use of incense sticks were associated with higher indoor $PM_{2.5}$ concentrations (He et al., 2004; Huang et al., 2014). While ambient pollutant concentrations can influence indoor concentrations to varying extents depending on the location, indoor cooking and biomass combustion can contribute also to ambient PM concentrations. In fact, Rehman et al. (2011) found higher black carbon (BC) concentrations in indoor environments in a rural area in northern India compared to outdoor concentrations, and postulated that HAP and cooking emissions could be driving ambient BC concentrations, particularly in areas where other PM sources are limited.

Cooking has been identified as a significant source of indoor air pollution although the emission strength can vary based on the type of fuel, cooking style (e.g. frying/grilling/roasting etc.) and ventilation in the kitchen. Further, concentrations tend to increase proportionally with the amount of time spent in cooking. Key parameters for exposure due to cooking include the fuel type (solid fuel/kerosene/LPG), proximity to the stove/cooker and the kitchen type (type of ventilation) (Balakrishnan et al., 2002; Wei See et al., 2006; Morawska et al., 2013). In addition, the process of cooking (grilling/frying/steaming etc.) can also generate fine and ultrafine PM which can contribute to indoor air quality, particularly in the absence of proper ventilation. Asian style cooking has been reported to generate more particles compared to Western cooking (Abdullahi et al., 2013) although emissions associated with Indian food preparation are reported to be lower than Chinese or Malay- style cooking (Wei See et al., 2006).

Incense burning is common in Asian and Middle- Eastern countries including India for religious and ritual purposes, and a large number of people burn incense at least once a day (Lin et al., 2008). Recent studies from India have highlighted the contribution of incense burning and other indoor combustion sources to indoor pollutant concentrations (Jai Devi et al., 2009; Dewangan et al., 2013; Kumar et al., 2014). Exposure to incense can post significant health risk since average modal diameter for incense emissions range between 79 and 89 nm, and are rich in cadmium (Cd) and lead (Pb) (Roy et al., 2009). Another indoor source which is restricted to tropical countries is the use of mosquito repellent coils, particularly in summer and monsoon seasons, and have been reported to be rich in heavy metals as well as PAHs (Nandasena et al., 2010; Roy et al., 2009; Dubey et al., 2014). Geographically restricted socio-cultural practices have been associated with high exposures to air pollution elsewhere including the coffee ceremony in Ethiopia (Keil et al. 2010) and incense burning in other parts of Asia (See and Balasubramanian, 2011; Averett, 2014).

Schools represent an indoor environment where children spend a significant portion of their day, and high pollutant levels in schools can lead to higher daily exposures. Several studies have analyzed concentrations of air pollutants in schools where children spend a significant amount of time during the day (Habil and Taneja, 2011; Chithra and Shiva Nagendra, 2012; Habil et al., 2013; Habil et al., 2016). A summary of reported concentrations is presented in Table 4.

Table 4
Summary of PM concentrations in schools (PM in μg/m^3, WD- weekday, WE- weekend).

References	City	Site Type	Season	PM_1	$PM_{2.5}$	PM_{10}	SPM
Goyal and Khare (2009)	Delhi	School building	Winter (WD)	293.3 ± 180.3	359.9 ± 187.7	1181.1 ± 578.3	
			Winter (WE)	328.5 ± 229.6	366.1 ± 251.8	644 ± 617.2	
			Non-winter (WD)	45.8 ± 26.5	71 ± 37.3	410.6 ± 196.4	
			Non-winter (WE)	41.3 ± 36.7	54.6 ± 40.2	133.5 ± 71.2	
Habil and Massey (2011)	Agra	School building (roadside)		259.51 ± 55.62	240.95 ± 152.79	524.76 ± 169.36	
		School building (residential)		67.46 ± 31.12	92.48 ± 45.74	157.80 ± 67.84	
Saksena et al. (2007)	Delhi	School					230
Chithra and Shiva Nagendra (2014)	Chennai	School building (roadside)	Indoor	20 ± 12	36 ± 15	136 ± 60	
			Outdoor	23 ± 14	33 ± 16	76 ± 42	
		School building (background)	Indoor	29.27	37.57	53.46 ± 23.21	
			Outdoor				
Jyethi et al. (2014)	Delhi	School				222.23 ± 30	
Habil et al. (2016)	Agra	School	Outdoor		128.45 ± 37.58		

4. Transport microenvironments

Emissions from road vehicles are of particular interest since they are emitted in vicinity of human activity, and canyon effects can concentrate the pollutant levels, thereby increasing the threat to human health (Colvile et al. 2001). There has been a 700% increase in the number of vehicles on the road between 1990 and 2010 in India, and by 2030, the total number of vehicles is expected to increase four- to five-fold, with older vehicles contributing between 30 and 50% of the total emission loads from the transport sector (Guttikunda and Mohan 2014). Commuting has been identified as a high-exposure activity, and previous studies have shown that traffic microenvironment exposures can constitute a significant fraction to daily exposure (Dons et al. 2011; Karanasiou et al. 2014), and exposure levels can vary based on vehicle type (engine/aftertreatment technology/fuel type) as well as prevalent meteorological conditions, and land use. Based on a monitoring study in Delhi, Apte et al. (2011) concluded that daily traffic microenvironment exposures for commuters are often as high as 24 h exposures in high-income countries. The type of commute (e.g. low activity [car] vs high activity [cycle]) can lead to a variable PM dose, and with a significant proportion of Indian population still relying on walking or cycling for commuting, it is imperative to include exposure as a variable in transport design. Based on a study in India [1997], people spend between 0.34 and 2.06 hours in commuting (Prasad et al. 2003) and although this study is somewhat dated, more recent analyses have reported similar results (Sabapathy et al. 2012; Manoj et al. 2015). Time spent in commuting can vary substantially across cities and regions based on gender, occupation, availability of public transportation, socio-economic class and pattern of urbanization (Manoj et al. 2015) among other factors. In order to reduce uncertainties associated with time spent in commuting, it is imperative that survey-based analyses for time-activity patterns across India.

Several studies have reported commuter exposure to air pollutants, but there seems to be limited consensus in terms of the most polluted transport mode. Across several studies, air-conditioned cars have been found to have lowest PM exposure. In Mumbai, Namdeo et al. (2014) reported higher PM_{10} exposure while travelling in bus (502.78 μg/m^3), and the lowest exposure in air-conditioned cars (68.78 μg/m^3) while in a similar analysis in Bangalore, average PM_{10} exposures for two-wheelers during the peak hour were recorded as 403 μg/m^3. In an older study, PM_5 concentrations as high as 2860 μg/m^3 were reported for two-wheelers (Prasad et al. 2003). In Delhi, Goel et al. (2015a) reported lowest exposures to $PM_{2.5}$ in air-conditioned cars and metro while in Ahmedabad, (Swamy et al. 2015). Contrary to these results, Kumar and Gupta (2016) reported highest in-vehicle concentration of RSPM for cars and lowest values for motorcycles in New Delhi, although it must be noted that a non-air conditioned car with windows rolled down was used for car measurements. Most studies also report significant differences between traffic peak and non-peak PM concentrations.

Only two studies have estimated PNC associated with road traffic (Apte et al. 2011; Kumar et al. 2011), and both estimates are for Delhi. Kumar et al. (2011) estimate average particle number exposures of $\sim 10^4$ cm^{-3} and 10^5 cm^{-3} for ambient and roadside environments in New Delhi. These estimates are higher than results reported elsewhere, and warrant further investigation. A summary is presented in Table 5.

Idling times in Indian cities can be lengthy due to traffic, and as a result, personal exposure levels can enhanced. In Delhi, Goel et al. (2015b) reported idling times between 18% and 37% across vehicle types, and previous studies have reported a several fold increase in in-vehicle concentrations attributable to roadway or near-roadway emissions (Apte et al. 2011). In a unique exposure assessment for the Indian conditions, Namdeo et al. (2014) estimated a PM_{10} dose of 0.99 μg/km for air-conditioned cars and 7.32 μg /km for buses, and Goel et al. (2015a) estimated that the inhaled dose of $PM_{2.5}$ is ~9 times higher for cyclists compared to people in cars.

It is important to mention that India currently has a dual emissions standard, and the fuel quality varies across the country. Much of the work on commuter exposures has been conducted in citied with cleaner fuel (Bharat Stage IV), and it is reasonable to assume that emission patterns, and therefore exposures might vary for other urban and rural areas. Further, a number of urban agglomerations, and a majority of the rural areas do not have public transportation, leading to intermediate public transport solutions including shared cabs and auto-rickshaws, mini-buses and cycle rickshaws. In the last decade, there has been an increase in private vehicle ownership in rural areas, often attributed to rising aspirations, changing demographics and increasing income, and this is likely to further contribute to changes in lifestyles, and thus, time-activity patterns.

5. Personal exposure

Personal exposure to PM depends on several factors include age, gender, time spent in different microenvironments, and susceptibility to air pollutants. Exposure can be also be driven by the occupation and the socio-economic status of an individual. Researchers have used multiple approaches including monitoring of

Table 5
Summary of reported air pollutant concentrations in various transport micro-environments (mass in μg/m^3, particle number in #/cm^3).

References	City	Transport Mode	PM_5	$PM_{2.5}$	PM_{10}	BC	Particle Number
Saksena et al. (2007)	Delhi	Three-wheeler	810				
		Bus	800				
		Car	370				
		Two-wheeler	2860				
Namdeo et al. (2014)	Mumbai	Bus			502.78 ± 157.7		
		A/C Car			68.78 ± 15.8		
		Non A/C Car			310.76 ± 83.8		
		Train			156.18 ± 32.4		
Saksena et al. (2007)	Delhi	Two-wheeler			2860		
		Car					
Apte et al. (2011)	Delhi	Three-wheeler		190		42	280×10^3
Sabapathy et al. (2012)	Bangalore	Two-wheeler			375–403		
		Car			61–253		
		Bus			244–405		
		Walk			182–254		
Swamy et al. (2015)	Ahmedabad	BRTS[a] Non A/C Bus		109–325			
		BRTS A/C Bus		67–226			
		AMTS		122–367			
		Non A/C Car		123–414			
		A/C Car		83–292			
		Three-wheeler		134–385			
		Two-wheeler		114–359			
		Non-motorized		110–347			
Devi et al. (2009)	Kanpur	Railway platform		43.2 ± 5.6			
Kumar and Gupta (2016)	New Delhi	Three-wheeler	168.6		242.9		
		Bus	175.9		255.8		
		Car	225.8		296.9		
		Motorcycle	168.2		229.4		

[a] Bus Rapid Transit System.

PE using personal monitors, or measurements in various micro-environments and use of time-activity diaries. Several reviews have been written on the topic and provide a good overview of the monitoring methodologies (Steinle et al. 2013; Morawska et al. 2013; Nieuwenhuijsen et al. 2014).

PE can vary based on ambient concentrations, but as discussed earlier, use of fixed-site monitors for quantifying exposures can cause exposure misclassification leading to incorrect exposure estimates (Han and Naeher 2006; Dons et al. 2012). Research on PE to air pollutants in India has been very limited, and in most cases, reported data is mostly only available for PM_{10} and gaseous pollutants such as CO, and in some cases, for $PM_{2.5}$ and PM_1. Limited analysis has revealed a lack of correlation between ambient PM levels and the concentrations individual are exposed to, although PE levels and the indoor PM concentrations have been reported to be correlated (Kulkarni and Patil 1999).

While there has been extensive analysis of HAP both in terms of pollutant characteristics and health impacts associated with exposure, datasets on urban microenvironment exposures is poor. A number of studies have focused on the health effects related to exposure to solid fuel combustion (wood/cow dung etc.) with respect to HAP, particularly in women and children. However, it is important to consider that in cities, solid fuels are often used for heating, particularly by men who work outdoors during the day or at night. Additionally, combustion for the purpose of heating is often more toxic, as a wide variety of materials including waste plastics, dry leaves and other waste materials can be burnt. The differences in exposure patterns are driven, in part, by the differences in time-activity patterns which can vary significantly based on location (urban vs. rural), occupation, age and gender. Another important consideration for the analysis of the health effects is the inhaled dose which depends on pollutant concentration as well as lung function and activity level. Of particular importance are exposure patterns for vulnerable groups including the elderly, children and women, and it is important to segregate the time-activity patterns based on degree of urbanization (urban, peri-urban and rural) as well as socio-economic status due to differences in lifestyle and emission sources. In India, there are significant gaps in the characterization of time-activity pattern in India and only a few studies have characterized time-activity patterns, and that too for a limited number of subgroups including students, rural women, workers and the elderly (Saksena et al. 2007; Kulshreshtha et al. 2008; Jai Devi et al. 2009; Shimada and Matsuoka 2011). However, in the last decade, there has been a significant change in lifestyle, particularly in urban and peri-urban areas, and the time-activity patterns established five to ten years ago may no longer be relevant.

A few studies have reported personal exposure concentrations, and they range from 30 to 140 μg/m^3. In a recent analysis using land use regression (LUR) modelling, Saraswat et al. (2016) reported annual average $PM_{2.5}$ exposure ranging between 109 and

Table 6
Summary of reported personal exposure to PM (mass in μg/m^3, particle number in #/cm^3).

References	City	Season	$PM_{2.5}$	PM_1	$PM_{2.5\text{-}10}$	PNC
Jai Devi et al. (2013)	Kanpur	Pre-monsoon	30.5 ± 2.7	12 ± 9	52.9 ± 17.2	17,656 ± 7,014 (PM_1) 177.7 ± 147.4 ($PM_{2.5}$)
		Post-monsoon	37 ± 26	24 ± 21	34.5 ± 20.4	422.8 ± 359.3 ($PM_{2.5}$)
Habil et al. (2016)	Agra	Winter	135.28 ± 44.86 (Home) 140.24 ± 42.21 (School) 125.78 ± 40.24 (Office)			

125 $\mu g/m^3$ for New Delhi. A summary is presented in Table 6.

Important determinants of exposure levels are discussed briefly in the following sections.

5.1. Impact of socio-economic status

Exposure to air pollution and associated health effects are often interlinked with socio-economic status as well as socio-cultural norms. Several studies have also observed higher levels of exposure for households with lower median incomes compared to households with higher median incomes (Kulkarni and Patil 1999; Kathuria and Khan 2007; Khan and Kraemer 2008; Firdaus and Ahmad 2011). Higher incidence of respiratory symptoms has also been associated with socio-economic level, with the decrease in lung function associated with the socio-economic level of the household (CPCB 2008). Poorer households also tend to live in crowded conditions, and co-habitation with smokers, and the use of polluting fuels is much more common (Duflo et al. 2008; Choi et al. 2014). In Mumbai, Kulkarni and Patil (1999) reported an increase of nearly 100 $\mu g/m^3$ of PM in smaller houses compared to larger houses, and Ghosh and Mukherji (2014) reported a higher frequency of negative health effects in children from slum areas compared to non-slum areas. Gupta et al. (2006) also found an association between ETS exposure and socio-economic conditions, with people with higher incomes exposed to lower levels of ETS compared to households with lower income levels. A detailed discussion on the relationship between socio-economic conditions and air pollution exposures is available elsewhere (O'Neill et al. 2003; Jerrett 2009). Kathuria and Khan (2007) have also commented on the inequity in air pollution exposure in the Indian context. Not surprisingly, the relationship between socio-economic conditions and pollution exposure is expected to vary across communities and countries, making it difficult to draw generalizations.

5.2. Urban vs. rural areas

Currently, 31.16% of the population of India is in urban areas while 68.84% reside in rural areas, and between the National Censuses in 2001 and 2011, there has been a drop in the percentage of population residing in rural areas (Census of India 2011). The number of people suffering from respiratory ailments is on the rise in urban agglomerations (Kumar 2011), and health impacts such as reduction in lung function, and increase in incidence of COPD and other cardio-vascular diseases is more common in urban areas compared to rural areas (CPCB 2008). Hospital admissions are also reported to be correlated to air pollutant concentrations, with higher hospital admission rates in areas with higher air pollution (Liu et al. 2013). Within urban areas, highest concentrations of PM are often observed in low-income neighborhoods as has been shown in Mumbai and Bangalore (Abba et al. 2012b; Both et al. 2013). With rapid urbanization, people are living in close proximity to roadways, industrial units and construction sites; differences in exposure patterns can be driven by spatial differences in pollutant concentrations which in turn depend on the source strength in surrounding areas. For example, proximity to roadways can influence the level of exposure with people residing in close proximity of air pollution sources experiencing higher levels of pollution (Kulkarni and Patil 1999; Kathuria and Khan 2007; Padhi and Padhy 2008). Delhi, for example, Kumar and Foster (2009) reported spatial variations within the city, and noted higher concentrated in peripheral regions of the city compared to the center, and this was partly attributed to different regulation across states. High variability of PM concentration has also been observed for other cities in developing countries including Ghana (Dionisio et al. 2010).

Studies have reported a higher incidence of lung function reduction in children residing in urban areas compared to children in rural areas and a positive correlation between ambient PM (PM_{10}) and lung function deficits (Siddique et al. 2010). Other studies have reported higher incidence of restrictive airway disease in children living in industrial areas compared with cleaner areas (Kalappanavar et al. 2012). Higher rates of hypertension and oxidative stress have also been observed for residents of urban areas compared to rural areas (Dutta et al. 2011; Banerjee et al. 2012; Dutta and Ray 2013). Nagaraja et al. (2012) reported higher levels of H_2O_2 in the exhaled breath condensate (EBC) for both smokers and non-smokers in urban areas compared to rural areas, and attributed this to higher levels of exposure to air pollution (traffic and industrial emissions) while in a study in Delhi, researchers reported compromised immunity (increased numbers of natural killer cells, decreased number of CD4+T-helper cells) in case of urban residents exposed to pollution for long-term (>10 years) (Banerjee et al. 2012). A number of studies have reported reduction in lung function (particularly FEV_1-forced expiratory volume) in people residing in areas with high PM levels (Siddique et al. 2011; Kesavachandran et al. 2013) and this is particularly important since FEV_1 has been reported to be an indicator for all-cause mortality as well as cancer mortality based on the assessment of an Indian cohort (Hebert et al. 2010). These findings have immediate relevance since the urban population in India is increasing, and India will have 14% of the total urban population of the world by 2050 coupled with high population density across urban agglomerations (Swerts et al. 2014). Population density has been reported to be an important variable for pollutant concentration (Saraswat et al. 2013), and in a recent World Bank study, $PM_{2.5}$ was reported to increase by 35% with doubling of population density in South Asia (Ellis and Roberts 2016).

In rural areas, combustion of agricultural crop residue (rice/paddy or wheat) is an important source of exposure to air pollutants is the, and concentrations during the burning can be almost twice as high compared to average ambient concentrations (Rajput et al. 2014). Several studies have characterized pollutant emissions associated with this source (Awasthi et al. 2010; Agarwal et al. 2012; Rajput et al. 2014; Singh et al. 2014). $PM_{2.5}$ concentrations as high as 390 $\mu g/m^3$ have been observed during the crop residue burning, and typically, emissions are higher during paddy/rice residue burning compared to wheat residue burning (Rajput et al. 2014). Almost a third of the total emission are in the form of OC in case of paddy residue burning while it is $\sim$ 25% in case of wheat residue burning (Rajput et al. 2014). There is some existing literature on the health effects of exposure to emissions generated from combustion of agricultural crop residues (Agarwal et al. 2012; Agarwal et al. 2013), and a reduction in pulmonary function has been reported, although reduction in some parameters such as peak expiratory flow rate has been observed to recover at the end of the exposure period (Agarwal et al. 2013). In particular, reduction in lung function has been associated with fine particles ($PM_{2.5}$) (Agarwal et al. 2013), which form a majority of the particles emitted during the combustion process. Children have been found to be particularly susceptible to exposure to agricultural residue burning (Awasthi et al. 2010).

Other sources can also vary between urban and rural areas. For example, in urban areas, short but high exposures can occur due to the use of air fresheners, spray-based deodorants and through use of vacuum cleaners, most of which are typically absent in rural areas in India (Broich et al. 2012). A new study in southern India-Tamil Nadu Air Pollution and Health Effects (TAPHE) is assessing exposure to $PM_{2.5}$ in integrated rural-urban cohorts (Balakrishnan et al. 2015).

5.3. Gender

Gender type often determines activity patterns (e.g. time spent indoor vs. outdoor, type and duration of job) (Clougherty 2010; Joon et al. 2009), and exposure to indoor emission sources such as cooking and/or cleaning are often highly variable between men and women. Women are generally more vulnerable to health effects associated with air pollution, and have a disproportionately higher levels of exposure to HAP, particularly in developing countries (Parikh et al. 1999; Shimada and Matsuoka 2011; Buonanno et al. 2014; Dutta and Banerjee 2014; Gordon et al. 2014). Highest respirable PM concentrations were also reported during the cooking process in Delhi (Kulshreshtha and Khare 2011), and across households with different median incomes, women often have higher overall exposures (Kulshreshtha et al. 2008). Studies have provided variable estimates for the time spent by women in kitchen including Behera and Balamugesh (2005) who suggested that women spend an average of four to six hours in kitchens and Jain et al. (2011) who suggested that women can spend as many as ten hours in the kitchen. Being the main cook in the household, women spend a disproportionate amount of time in the kitchen which results in high overall exposure although they often have limited decision-making power in terms of choice of fuel (Duflo et al. 2008; Dutta and Banerjee 2014). It is important to note that these estimates are true for rural areas where most households still rely on solid fuels (wood, crop residue, animal dung), and as a result, women have a very high incidence of exposure to PM in indoor environments, especially when the kitchens are poorly ventilated, and are not separated from the rest of the house. Both the amount of time spent in the kitchen and the exposure levels are expected to be different in urban households using LPG fuel, although a large number of women spend a majority of their time in indoor environments. Dutta and Banerjee (2014) reported higher average exposure for women compared to men both in slum and non-slum locations in Kolkata, and the difference was found to be particularly relevant for people in the lower socio-economic classes. Exposure to ETS (direct or SHS) is another source of PM exposure for women, particularly in poorer households, and the risk of asthma is higher for women who are exposed to HAP as well as ETS.

In terms of gender-related health differences, girls and women are reported to have higher incidence of reduction in lung function, irrespective of the type of area where sampling is conducted (CPCB 2008; Siddique et al. 2010), with the exception of Kumar et al. (2008) found no relation in respiratory health endpoints and sex. Jain et al. (2011) also found that due to implicit socio-cultural systems as well as the notion that COPD is more common in men compared to women, women tend to receive less medical care compared to men.

Most of the studies focused on health effects of air pollution exposure for women have focused on exposure to HAP in rural areas, and there aren't any primary data or estimates for gender-specific exposure to PM in urban areas.

5.4. Climate and meteorology

India is a large country, and different parts of the country experience different weather patterns. As a result, the air pollution trends also vary significantly across the country. For example, reported pollution levels in the Indo-Gangetic basin in northern India are much higher in comparison to the southern India. The onset of winter is usually accompanied by additional combustion sources (heating), particularly in the northern and central parts of the country, and as a result, PM concentrations in winter are several fold higher in winter compared to summer and monsoon (rainy) seasons. For example, Pant et al. (2015), Varshney et al. (2015) and Izhar et al. (2016) reported higher concentrations of $PM_{2.5}$ as well individual chemical components (metals, PAHs) in winter, and Baxla et al. (2009) observed higher PNC during winter. Pollutant concentrations are typically lowest during the monsoon (rainfall) season, when wet deposition of particles reduces ambient PM levels significantly and high winter concentrations are attributed to combined effects of meteorology (lower mixing height, little to no wind etc.) and an increase in emissions (e.g. space heating). Most studies have reported higher PM concentrations in indoor environments in winter compared to summer (Nautiyal et al. 2007; Habil and Taneja 2011; Massey et al. 2012; Masih et al. 2012), which is driven both by the higher ambient pollutant concentrations, but also the lower air exchange rate (closed windows), especially in naturally-ventilated buildings. Higher indoor concentrations in winter have also been reported in other countries (Buonanno et al. 2014).

Exposures can also vary between seasons, and as a result, impact of exposure on health endpoints can vary by season. In Delhi, for example, the highest reduction in lung function was observed in winter when the pollutant concentrations are much higher (CPCB 2008; Siddique et al. 2010). On a 24 h timescale, the exposure levels are usually lower during the night, since there is little indoor activity during this time (Dons et al. 2011; Broich et al. 2012).

6. Health impacts

Household and ambient air pollution feature among the top ten health risk factors in India as per the GBD estimates, and ~1.04 million and 627,000 premature Indian deaths were reported to be associated with HAP and APP respectively (Lim et al. 2012; Balakrishnan et al. 2014). This includes deaths due to a range of diseases including cardiovascular diseases, lower respiratory infections and cerebrovascular diseases. Several studies have assessed the relationship between air pollution levels and all-cause mortality, emergency visits, and respiratory morbidity, either directly as PM concentrations (Kumar et al. 2010; Rajarathnam et al. 2011; Balakrishnan et al. 2013b; Choi et al. 2014; Dholakia et al. 2014) or using other factors such as visibility (Kumar et al. 2010). Balakrishnan et al. (2013b) assessed mortality effects of exposure to PM in Chennai and reported a 0.44% increase in mortality per 10 $\mu g/m^3$ increase in PM_{10}. On the other hand, in Delhi, Rajarathnam et al. (2011) reported an increase of 0.19% in mortality for every 10 $\mu g/m^3$ increase in PM_{10}. Multiple studies have used modelling analysis using global concentration-response functions to assess health impacts associated with PM exposure, and in absence of reliable data, a majority of such studies have used modelled PM data (Guttikunda and Jawahar 2012). In some other cases, measured PM data has been used albeit for short durations (e.g. assessment of health impacts of exposure to air pollutants during the Commonwealth Games (Beig et al. 2013a)). Several other studies have also focused on estimation of health effects associated with specific sources including coal combustion (Guttikunda and Jawahar 2014) and transport-related emissions (Aggarwal and Jain 2015). Interestingly, Dholakia et al. (Dholakia et al. 2014) observed a higher percentage increase in mortality for cleaner cities (e.g. Shimla) compared to more polluted cities (e.g. Ahmedabad), an observation that warrants further investigation. Analysis of satellite data has led to conclusions that a majority of Indians, both in urban and rural areas are exposed to high levels of $PM_{2.5}$ (Dey et al. 2012). A summary of the reported health effects is presented in Table 7.

Yamamoto et al. (2014) highlighted the limited number of studies and the lack of detailed and long-term health effects analyses on cardio-vascular disease risk associated with exposure

Table 7
Premature mortality estimates (per year) based on modelling analyses.

City	Pollutant	Premature mortality	References
Delhi	$PM_{2.5}$	7350–16200	(Guttikunda and Goel 2013)
Greater Hyderabad Region	$PM_{2.5}$	3700	(Guttikunda and Kopakka 2014)
Pune	$PM_{2.5}$	3600	(Guttikunda and Jawahar 2012)
Chennai	$PM_{2.5}$	3950	
Ahmedabad	$PM_{2.5}$	4950	
Delhi	Effects caused by TSP, SO_2 and NO_2	~10,500 (~1600-respiratory mortality, ~3500 cardiovascular mortality)	(Gurjar et al. 2010)
Kolkata		~1300 (respiratory mortality)	
Delhi	Nanoparticles	508 per million	(Kumar et al. 2011)
Kerela	Ambient air pollution	38 natural deaths per 100,000 people	Tobollik et al. (2015)
National	Coal-fired power plants	80,000-115,000	(Guttikunda and Jawahar 2014)

to air pollution in India and Pakistan, and an analysis of studies focused on COPD led to similar conclusions (McKay et al. 2012). In a review of health effects of air pollution, Mohanraj and Azeez (2004) also highlighted the lack of epidemiological studies on air pollution. In cases where studies have been conducted, researchers have relied on ambient fixed-site monitoring data, and most of them highlighted the lack of complete and continuous datasets. A common theme across the studies has been the uncertainties associated with exposure impact modelling in the absence of quality assured data from ground monitoring, and together, they present a strong case for a quality- controlled air quality monitoring network across the country.

In India, differences have been reported in lung function tests for individuals between clean and polluted areas, with reduced lung function in polluted areas (Sharma et al. 2004; Kalappanavar et al. 2012). Several studies have estimated health risks associated with specific constituents of PM including elements (Khillare and Sarkar 2012; Kushwaha et al. 2012; Yadav and Satsangi 2013) and PAHs (Abba et al. 2012b; Jyethi et al. 2014) as well as gases. For school kids, inhalation exposures to PAHs was estimated as 283.93 ng/day in case of Delhi with vehicular emissions contributing significantly to this exposure (Jyethi et al. 2014). In Agra, excess cancer risk due to PAH exposure was found to be within acceptable range for inhalation exposure (Dubey et al. 2015), while for metals, lead (Pb) and cadmium (Cd) were found to be associated with carcinogenic risk due to inhalation exposure (Varshney et al. 2015). Research in Kanpur reported inhalation exposure to be lower compared to ingestion and dermal contact for metals, and excess cancer risk was found to be above the acceptable limit for both adults and children (Izhar et al. 2016), and similar results were observed in Delhi where Kushwaha et al. (2012) reported inhalable carcinogenic risk to be above the acceptable limit, and Sarkar and Khillare estimated between 503 and 39,780 excess cancer cases associated with exposure to PAHs in ambient air (Sarkar and Khillare 2013). On the other hand, in Amritsar, cancer risk associated with PM_{10}-PAHs was found to be within the acceptable range (Kaur et al. 2013). Higher molecular weight PAHs have been associated with cancer risk for both adults, and children (Kaur et al. 2013; Singh and Gupta 2016a). Singh and Gupta (2016b) also reported higher risks during foggy days compared to non-foggy data in Kanpur due to higher concentrations of metals.

7. Conclusions

A clear understanding of pollution exposure concentrations and trends, and associated health effects will not only help improve the quality of life of Indian citizens, but also help improve the field of exposure science, especially in the context of developing economies. There are similarities between source types and exposure patterns in South Asia, and data generated in India can be useful in policymaking in other countries in the region.

Based on the available literature, it can be concluded that in the context of a rapidly developing country, the amount of data on air pollution exposure currently available is largely inadequate, and the levels of exposure to PM are almost universally much higher compared to levels reported in Europe and North America. As has also been noted by Balakrishnan et al. (2011), data from the developed countries is not readily translatable in the Indian context due to differences in source types as well as lifestyle. It is also important to note that several of the studies included in this review are from five or more years ago, and due to improvements in technology (e.g. emissions controls, pollution measurement capabilities), and improvement of quality of life (due to increase in family incomes), the observations may no longer represent the exposure for most people. Much of the research has focused on the largest cities (Delhi, Mumbai, Bangalore, Chennai, Kolkata), and there is little ambient or indoor pollution data for smaller cities and towns. Characterization of pollutant concentrations in these urban areas is important since smaller cities and towns often lack public transportation options, and in many cases, fuel quality is poor. Additionally, due to haphazard urban growth, large numbers of people live in close proximity to mobile as well as point pollution sources, increasing the risk of exposure to harmful pollutants.

Several gaps emerge from the review of the literature, and future research should address these in an attempt to improve our understanding of exposure to air pollution in India.

- Most of the available data is focused on coarse PM (PM_{10} or PM_5), and there is little information on the exposure to fine and ultrafine PM. However, the most significant health effects are associated with fine particles, and there is a need for greater focus on characterization of exposure to fine and ultrafine particles, especially in urban microenvironments. Additionally, several unique sources (e.g. waste burning, construction emissions) are poorly characterized, or not characterized at all.
- Time-activity characterization is poor, and the amount of time people spend in various microenvironments is not clear.
- Studies characterizing public health impacts associated with air pollution are limited in number as well as scope. A vast majority of the studies have assessed health effects of air pollution by investigating the relationship between PM and health endpoints such as cardiovascular morbidity, incidence of respiratory diseases or premature mortality while a few others have generated modelling estimates for assessing premature death, and morbidity data. There is very limited data on biomarker-based health assessments (e.g. increase in inflammation/presence of elements/PAHs in urine, assessment of exhaled breath condensate etc.), except in the case of HAP.
- While there have been concerted efforts to characterize AAP

sources in the last ten years, indoor source characterization is poor, particularly in urban areas. There are no detailed analyses on the various indoor sources (apart from solid fuel combustion) in India, and how the concentrations might vary across households with different median income levels. Additionally, there is a lack of detailed emission inventories for various pollutants apart from some recent studies (Guttikunda and Calori 2013; Pandey et al. 2014; Sadavarte and Venkataraman 2014), and this can often results in speculative modelling exercises which may not represent the actual conditions accurately.
- Assessment of air pollution exposure without accounting for variables such as gender and socio-economic status can lead to a skewed understanding of exposures. Thus, future research should be based on a holistic framework which consider scientific as well as social aspects linked to air pollution exposure.

In summary, while much progress has been made in the last few years, much remains to be accomplished. Given the undeniable evidence regarding the relationship between socio-economic status and exposure to air pollution, urban policy initiatives need to account for income disparities, and consideration of pollution exposure needs to be integrated in the design process so as to minimize pollution exposures, particularly in new urban areas. In 2015, Prime Minister Modi launched the smart city initiative ("*a smart city has been described as an economically and environmentally sustainable urban area that serves the twin purposes of good infrastructure and quality of life as well as competitive employment and investment opportunities for its inhabitants*"). It is imperative that the development plans for such cities including consideration for breathable cities with clean air. For example, a comprehensive assessment of exposure levels in different transport modes can also be helpful in making policy decisions, particularly, in cases such as bicycle lane planning, or the choice of fuel for public transport so as to minimize pollutant exposure.

Acknowledgements

The authors would like to acknowledge financial support from Climate and Health Research Network and PP and REP would like to thank EMBARQ-India for financial support (grant number 115-1065).

References

Abba, E.J., Unnikrishnan, S., Kumar, R., 2012a. Chemical characterization and mass closure of fine aerosol for different land use patterns in Mumbai city. AAQR 12, 61–72.
Abba, E.J., Unnikrishnan, S., Kumar, R., Yeole, B., Chowdhury, Z., 2012b. Fine aerosol and PAH carcinogenicity estimation in outdoor environment of Mumbai city, India. Int. J. Environ. Health Res. 22, 134–149.
Abdullahi, K.L., Delgado-Saborit, J.M., Harrison, R.M., 2013. Emissions and indoor concentrations of particulate matter and its specific chemical components from cooking: a review. Atmos. Environ. 71, 260–294.
Agarwal, R., Awasthi, A., Singh, N., Mittal, S.K., Gupta, P.K., 2013. Epidemiological study on healthy subjects affected by Agriculture crop-residue burning episodes and its relation with their pulmonary function tests. Int. J. Environ. health Res. 23, 281–295.
Agarwal, R., Awasthi, A., Singh, N., Gupta, P.K., Mittal, S.K., 2012. Effects of exposure to rice-crop residue burning smoke on pulmonary functions and oxygen saturation level of human beings in Patiala (India). Sci. Total Environ. 429, 161–166.
Aggarwal, P., Jain, S., 2015. Impact of air pollutants from surface transport sources on human health: a modeling and epidemiological approach. Environ. Int. 83, 146–157.
Agrawal, S., Yamamoto, S., 2015. Effect of indoor air pollution from biomass and solid fuel combustion on symptoms of preeclampsia/eclampsia in Indian women. Indoor air 25, 341–352.
Andresen, P.R., Ramachandran, G., Pai, P., Maynard, A., 2005. Women's personal and indoor exposures to PM2.5 in Mysore, India: impact of domestic fuel usage. Atmos. Environ. 39, 5500–5508.
Apte, J.S., Marshall, J.D., Cohen, A.J., Brauer, M., 2015. Addressing global mortality from ambient PM2.5. Environ. Sci. Technol. 49, 8057–8066.
Apte, J.S., Bombrun, E., Marshall, J.D., Nazaroff, W.W., 2012. Global Intraurban intake fractions for primary air pollutants from vehicles and other distributed sources. Environ. Sci. Technol. 46, 3415–3423.
Apte, J.S., Kirchstetter, T.W., Reich, A.H., Deshpande, S.J., Kaushik, G., Chel, A., Marshall, J.D., Nazaroff, W.W., 2011. Concentrations of fine, Ultrafine, and black carbon particles in auto-rickshaws in new Delhi, India. Atmos. Environ. 45, 4470–4480.
Ashok, V., Gupta, T., Dubey, S., Jat, R., 2014. Personal exposure measurement of students to various Microenvironments inside and outside the college campus. Environ. Monit. Assess. 186, 735–750.
Averett, N., 2014. Ritual risk: incense use and cardiovascular mortality. Environ. health Perspect. 122, A336-A336.
Awasthi, A., Singh, N., Mittal, S., Gupta, P.K., Agarwal, R., 2010. Effects of Agriculture crop residue burning on children and young on PFTs in north west India. Sci. Total Environ. 408, 4440–4445.
Balakrishnan, K., Ghosh, S., Ganguli, B., Sambandam, S., Bruce, N., Barnes, D.F., Smith, K.R., 2013a. State and national household concentrations of PM2.5 from solid cookfuel use: results from measurements and modeling in India for estimation of the global burden of disease. Environ. Health 12, 77.
Balakrishnan, K., Cohen, A., Smith, K., 2014. Addressing the burden of disease attributable to air pollution in India: the need to integrate across household and ambient air pollution exposures. Environ. Health Perspect., 122.
Balakrishnan, K., Sambandam, S., Ramaswamy, P., Ghosh, S., Venkatesan, V., Thangavel, G., Mukhopadhyay, K., Johnson, P., Paul, S., Puttaswamy, N., Dhaliwal, R.S., Shukla, D.K., SRU-CAR Team, 2015. Establishing integrated Ruralâ€"urban cohorts to assess air pollution-related health effects in pregnant women, children and adults in southern India: an overview of objectives, design and methods in the Tamil Nadu air pollution and health effects (TAPHE) study. BMJ Open 5, e008090.
Balakrishnan, K., Ganguli, B., Ghosh, S., Sambandam, S., Roy, S., Chatterjee, A., 2013b. A spatially disaggregated time-series analysis of the short-term effects of particulate matter exposure on mortality in Chennai, India. Air Qual., Atmosphere Health 6, 111–121.
Balakrishnan, K., Ramaswamy, P., Sambandam, S., Thangavel, G., Ghosh, S., Johnson, P., Mukhopadhyay, K., Venugopal, V., Thanasekaraan, V., 2011. 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, 4. http://dx.doi.org/10.3402/gha.v4i0.5638.
Balakrishnan, K., Sankar, S., Parikh, J., Padmavathi, R., Srividya, K., Venugopal, V., Prasad, S., Pandey, V.L., 2002. Daily average exposures to respirable particulate matter from combustion of biomass fuels in rural households of southern India. Environ. Health Perspect. 110, 1069–1075.
Banerjee, M., Siddique, S., Mukherjee, S., Roychoudhury, S., Das, P., Ray, M.R., Lahiri, T., 2012. Hematological, immunological, and cardiovascular Changes in individuals residing in a polluted city of India: a study in Delhi. Int. J. Hyg. Environ. Health 215, 306–311.
Banerjee, T., Murari, V., Kumar, M., Raju, M.P., 2015. Source apportionment of Airborne particulates through receptor modeling: indian scenario. Atmos. Res. 164–165, 167–187.
Baxla, S.P., Roy, A.A., Gupta, T., Tripathi, S.N., Bandopadhyaya, R., 2009. Analysis of diurnal and seasonal variation of Submicron outdoor aerosol mass and size distribution in a northern Indian city and its correlation to black carbon. Aerosol Air Quality Res. 9, 458–469.
Behera, D., Balamugesh, T., 2005. Indoor air pollution as a risk factor for lung cancer in women. J. Assoc. Physicians India 53, 190–192.
Beig, G., Ghude, S.D., Deshpande, A., 2010. Scientific Evaluation of Air Quality Standards and Defining Air Quality Index for India. RR-127.
Beig, G., Chate, D.M., Ghude, S.D., Ali, K., Satpute, T., Sahu, S.K., Parkhi, N., Trimbake, H.K., 2013a. Evaluating population exposure to environmental pollutants during Deepavali fireworks displays using air quality measurements of the SAFAR network. Chemosphere 92, 116–124.
Beig, G., Chate, D.M., Ghude, S.D., Mahajan, A.S., Srinivas, R., Ali, K., Sahu, S.K., Parkhi, N., Surendran, D., Trimbake, H.R., 2013b. Quantifying the effect of air quality control measures during the 2010 commonwealth Games at Delhi, India. Atmos. Environ. 80, 455–463.
Benbrahim-Tallaa, L., Baan, R.A., Grosse, Y., Lauby-Secretan, B., El Ghissassi, F., Bouvard, V., Guha, N., Loomis, D., Straif, K., 2012. Carcinogenicity of diesel-engine and gasoline-engine exhausts and some Nitroarenes. Lancet Oncol. 13, 663–664.
Bergin, M.H., Tripathi, S.N., Jai Devi, J., Gupta, T., Mckenzie, M., Rana, K.S., Shafer, M. M., Villalobos, A.M., Schauer, J.J., 2015. The discoloration of the taj Mahal due to particulate carbon and dust deposition. Environ. Sci. Technol. 49, 808–812.
Bhargava, A., Khanna, R.N., Bhargava, S.K., Kumar, S., 2004. Exposure risk to carcinogenic PAHs in indoor-air during biomass combustion whilst cooking in rural India. Atmos. Environ. 38, 4761–4767.
Both, A.F., Westerdahl, D., Fruin, S., Haryanto, B., Marshall, J.D., 2013. Exposure to carbon monoxide, fine particle mass, and Ultrafine particle number in Jakarta, Indonesia: effect of commute mode. Sci. Total Environ. 443, 965–972.
M., Brauer, G., Freedman, J., Frostad, A., van Donkelaar, R.V., Martin, F., Dentener, R., Van Dingenen, K., Estep, H., Amini, J.S., Apte, K., Balakrishnan, L., Barregard, D. M., Broday, V., Feigin, S., Ghosh, P.K., Hopke, L.D., Knibbs, Y., Kokubo, Y., Liu, S., Ma, L., Morawska, J.L., Texcalac Sangrador, G., Shaddick, H.R., Anderson, T., Vos, M.H., Forouzanfar, R.T., Burnett, A., Cohen, 2015. Ambient air pollution exposure

estimation for the global burden of disease 2013. Environmental science & technology.
Breen, M.S., Schultz, B.D., Sohn, M.D., Long, T., Langstaff, J., Williams, R., Isaacs, K., Meng, Q.Y., Stallings, C., Smith, L., 2014. A review of air exchange rate models for air pollution exposure assessments. J. Expo. Sci. Env. Epidemiol. 24, 555–563.
Broich, A., Gerharz, L., Klemm, O., 2012. Personal monitoring of exposure to particulate matter with a high temporal resolution. Environ. Sci. Pollut. Res. 19, 2959–2972.
Buonanno, G., Stabile, L., Morawska, L., 2014. Personal exposure to Ultrafine particles: the influence of time-activity patterns. Sci. Total Environ. 468–469, 903–907.
Census of India, 2011. Rural- Urban Distribution of Population.
Chafe, Z.A., Brauer, M., Klimont, Z., Van Dingenen, R., Mehta, S., Rao, S., Riahi, K., Dentener, F., Smith, K.R., 2014. Household cooking with solid fuels contributes to ambient PM2.5 air pollution and the burden of disease. Environ. Health Perspect. 122, 1314–1320.
Chakrabarty, R.K., Pervez, S., Chow, J.C., Watson, J.G., Dewangan, S., Robles, J., Tian, G., 2014. Funeral Pyres in south Asia: brown carbon aerosol emissions and climate impacts. Environ. Sci. Technol. Lett. 1, 44–48.
Chakraborty, A., Gupta, T., 2010. Chemical characterization and source apportionment of submicron (PM1) aerosol in Kanpur region, India. Aerosol Air Quality Research 10, 433–445.
Chen, L.C., Lippmann, M., 2009. Effects of metals within ambient air particulate matter (PM) on human health. Inhal. Toxicol. 21, 1–31.
Chen, Q., Hildemann, L.M., 2009. Size-resolved concentrations of particulate matter and Bioaerosols inside versus outside of Homes. Aerosol Sci. Technol. 43, 699–713.
Cheng, H., Deng, Z., Chakraborty, P., Liu, D., Zhang, R., Xu, Y., Luo, C., Zhang, G., Li, J., 2013. A comparison study of atmospheric polycyclic aromatic hydrocarbons in three Indian cities using PUF disk passive air samplers. Atmos. Environ. 73, 16–21.
Cheraghi, M., Salvi, S., 2009. Environmental tobacco smoke (ETS) and respiratory health in children. Eur. J. Pedia. 168, 897–905.
Chhabra, S.K., Rajpal, S., Gupta, R., 2001. Patterns of smoking in Delhi and comparison of chronic respiratory morbidity among beedi and cigarette smokers. Indian journal of chest disease and allied. Science 43, 19–26.
Chithra, V.S., Shiva Nagendra, S.M., 2012. Indoor air quality investigations in a naturally ventilated school building located close to an urban roadway in Chennai, India. Build. Environ. 54, 159–167.
Choi, J., Baumgartner, S., Harnden, B.H., Alexander, R.J., Town, G., D'Souza, G., Ramachandran, 2014. Increased risk of respiratory illness associated with kerosene fuel use among women and children in urban bangalore, india. Occupational and environmental medicine.
Clougherty, J.E., 2010. A growing role for gender analysis in air pollution epidemiology. Environ. health Perspect. 118, 167–176.
Colbeck, I., Nasir, Z.A., Ali, Z., 2010. Characteristics of indoor/outdoor particulate pollution in urban and rural residential environment of Pakistan. Indoor Air 20, 40–51.
Colvile, R.N., Hutchinson, E.J., Mindell, J.S., Warren, R.F., 2001. The transport sector as a source of air pollution. Atmos. Environ. 35, 1537–1565.
Cowling, K., Dandona, R., Dandona, L., 2014. Social determinants of health in India: progress and inequities across States. Int. J. Equity Health 13, 1–12.
CPCB, 2014. National Air Quality Index. CUPS/82/2014-15.
CPCB, 2010. Air Quality Monitoring, Emission Inventory and Source Apportionment Study for Indian Cities.
CPCB, 2008. Epidemiological Study on Effect of Air Pollution on Human Health (Adults) in Delhi. EHS/1/2008,.
Cyrys, J., Pitz, M., Bischof, W., Wichmann, H., Heinrich, J., 2004. Relationship between indoor and outdoor levels of fine particle mass, particle number concentrations and black smoke under different ventilation conditions. J. Expo. Anal. Environ. Epidemiol. 14, 275–283.
Das, R., Khezri, B., Srivastava, B., Datta, S., Sikdar, P.K., Webster, R.D., Wang, X., 2015. Trace element composition of $PM_{2.5}$ and PM_{10} from Kolkata – a heavily polluted Indian metropolis. Atmos. Pollut. Res. 6 (5), 742–750.
Deshpande, A., Kudtarkar, P., Dhaware, D., Chowgule, R., 2010. Study of Secondhand smoke levels Pre and Post implementation of the comprehensive smoking ban in Mumbai. Indian J. Commun. Med. 35, 409–413.
Dewangan, S., Chakrabarty, R., Zielinska, B., Pervez, S., 2013. Emission of volatile organic compounds from religious and ritual activities in India. Environ. Monit. Assess. 185, 9279–9286.
Dey, S., Di Girolamo, L., van Donkelaar, A., Tripathi, S.N., Gupta, T., Mohan, M., 2012. Variability of outdoor fine particulate (PM2.5) concentration in the Indian subcontinent: a remote sensing approach. Remote Sens. Environ. 127, 153–161.
Dherani, M., 2003. Indoor air pollution due to biomass fuels in north India. Trans. R. Soc. Trop. Med. Hyg. 97, 626.
Dholakia, H.H., Bhadra, D., Garg, A., 2014. Short term association between ambient air pollution and mortality and modification by temperature in five Indian cities. Atmos. Environ. 99, 168–174.
Diapouli, E., Chaloulakou, A., Koutrakis, P., 2013. Estimating the concentration of indoor particles of outdoor origin: a review. J. Air Waste Manag. Assoc. 63, 1113–1129.
Dionisio, K.L., Rooney, M.S., Arku, R.E., Friedman, A.B., Hughes, A.F., Vallarino, J., Agyei-Mensah, S., Spengler, J.D., Ezzati, M., 2010. Within-neighborhood patterns and sources of particle pollution: mobile monitoring and geographic information system analysis in four communities in accra, Ghana. Environ. Health Perspect. 118, 607–613.
Dons, E., Int Panis, L., Van Poppel, M., Theunis, J., Wets, G., 2012. Personal exposure to black carbon in transport Microenvironments. Atmos. Environ. 55, 392–398.
Dons, E., Int Panis, L., Van Poppel, M., Theunis, J., Willems, H., Torfs, R., Wets, G., 2011. Impact of Time–activity patterns on personal exposure to black carbon. Atmos. Environ. 45, 3594–3602.
Dubey, J., Banerjee, A., Meena, R.K., Maharaj Kumari, K., Lakhani, A., 2014. Characterization of polycyclic aromatic hydrocarbons in emissions of different mosquito coils. Bull. Environ. Conta. Toxicol. 92, 650–654.
Dubey, J., Maharaj Kumari, K., Lakhani, A., 2015. Chemical characteristics and mutagenic activity of PM2.5 at a site in the indo-gangetic plain, India. Ecotoxicol. Environ. Saf. 114, 75–83.
Duflo, E., Greenstone, M., Hanna, R., 2008. Cooking Stoves, indoor air pollution and respiratory health in rural Orissa. Econ. Polit. Wkly., 43.
Dutta, A., Mukherjee, B., Das, D., Banerjee, A., Ray, M.R., 2011. Hypertension with elevated levels of oxidized low-density lipoprotein and Anticardiolipin antibody in the circulation of Premenopausal Indian women chronically exposed to biomass smoke during cooking. Indoor Air 21, 165–176.
Dutta, A., Ray, M., 2013. Increased cardiovascular risk due to systemic inflammatory Changes and enhanced oxidative stress in urban Indian women. Air Qual., Atmosphere Health 6, 501–508.
Dutta, S., Banerjee, S., 2014. Exposure to indoor air pollution & women health: the situation in urban India. Environ. Urbanization Asia 5, 131–145.
EEA, 2014. Air Quality in Europe — 2014 Report. EEA Report No 5/2014.
Ellis, P., Roberts, M., 2016. Leveraging Urbanization in South Asia : Managing Spatial Transformation for Prosperity and Livability. CC BY 3.0 IGO.
Firdaus, G., Ahmad, A., 2011. Indoor air pollution and self-reported diseases? A case study of NCT of Delhi. Indoor Air 21, 410–416.
Gadkari, N.M., Pervez, S., 2007. Source investigation of personal particulates in relation to identify major routes of exposure among urban residentials. Atmos. Environ. 41, 7951–7963.
Gadkari, N., Pervez, S., 2008. Source apportionment of personal exposure of fine particulates among school communities in India. Environ. Monit. Assess. 142, 227–241.
Garg, A., Singh, M.M., Gupta, V.K., Garg, S., Daga, M.K., Saha, R., 2012. Prevalence and correlates of tobacco smoking, awareness of hazards, and quitting behavior among persons aged 30 years or above in a resettlement colony of Delhi, India. Lung India : Official Organ Indian Chest Society 29, 336–340.
Gargava, P., Rajagopalan, V., 2015. Source apportionment studies in six Indian Citiesâ€"drawing broad inferences for urban PM10 reductions. Air Qual., Atmosphere Health, 1–11.
Ghosh, A., Mukherji, A., 2014. Air pollution and respiratory ailments among children in urban India: exploring causality. Econ. Dev. Cult. Chang. 63, 191–222.
Giri, B., Patel, K.S., Jaiswal, N.K., Sharma, S., Ambade, B., Wang, W., Simonich, S.L.M., Simoneit, B.R.T., 2013. Composition and sources of organic tracers in aerosol particles of industrial central India. Atmos. Res. 120–121, 312–324.
Goel, R., Gani, S., Guttikunda, S.K., Wilson, D., Tiwari, G., 2015a. On-road PM2.5 pollution exposure in multiple transport Microenvironments in Delhi. Atmos. Environ. 123, Part A, 129–138.
Goel, R., Guttikunda, S.K., Mohan, D., Tiwari, G., 2015b. Benchmarking vehicle and passenger travel characteristics in Delhi for on-road emissions analysis. Travel Behav. Soc. 2, 88–101.
Gordon, S.B., Bruce, N.G., Grigg, J., Hibberd, P.L., Kurmi, O.P., Lam, K.H., Mortimer, K., Asante, K.P., Balakrishnan, K., Balmes, J., Bar-Zeev, N., Bates, M.N., Breysse, P.N., Buist, S., Chen, Z., Havens, D., Jack, D., Jindal, S., Kan, H., Mehta, S., Moschovis, P., Naeher, L., Patel, A., Perez-Padilla, R., Pope, D., Rylance, J., Semple, S., Martin II, W.J., 2014. Respiratory risks from household air pollution in low and middle income countries. the lancet. Respir. Med. 2, 823–860.
Goyal, R., Kumar, P., 2013. Indoor-outdoor concentrations of particulate matter in nine Microenvironments of a mix-use commercial building in megacity Delhi. Air Qual., Atmosphere Health 6, 747–757.
Grills, N.J., Singh, R., Singh, R., Martin, B.C., 2015. Tobacco usage in Uttarakhand: a dangerous combination of high prevalence, widespread ignorance, and resistance to quitting. BioMed. Res. Int. 2015, 132120.
Gupta, D., Aggarwal, A.N., Jindal, S.K., 2002. Pulmonary effects of passive smoking: the Indian experience. Tob. Induc. Dis. 1, 129–136.
Gupta, S., Mittal, S., Kumar, A., Singh, K.D., 2011. Respiratory effects of air pollutants among Nonsmoking traffic policemen of Patiala, India. Lung India : Official Organ Indian Chest Society 28, 253–257.
Gurjar, B.R., Jain, A., Sharma, A., Agarwal, A., Gupta, P., Nagpure, A.S., Lelieveld, J., 2010. Human health risks in Megacities due to air pollution. Atmos. Environ. 44, 4606–4613.
Guttikunda, S.K., Goel, R., Pant, P., 2014. Nature of air pollution, emission sources, and management in the Indian cities. Atmos. Environ. 95, 501–510.
Guttikunda, S.K., Jawahar, P., 2014. Atmospheric emissions and pollution from the coal-fired thermal Power plants in India. Atmos. Environ. 92, 449–460.
Guttikunda, S.K., Mohan, D., 2014. Re-fueling road transport for better air quality in India. Energy Policy 68, 556–561.
Guttikunda, S.K., Calori, G., 2013. A GIS based emissions inventory at 1 km $\times$ 1 km spatial resolution for air pollution analysis in Delhi, India. Atmos. Environ. 67, 101–111.
Guttikunda, S.K., Goel, R., 2013. Health impacts of particulate pollution in a megacity—Delhi, India. environmental. Development 6, 8–20.
Guttikunda, S.K., Jawahar, P., 2012. Application of SIM-air modeling tools to assess air quality in Indian cities. Atmos. Environ. 62, 551–561.
Guttikunda, S., Kopakka, R., 2014. Source emissions and health impacts of urban air pollution in Hyderabad, India. Air Qual., Atmosphere Health 7, 195–207.

Habil, M., Massey, D.D., Taneja, A., 2016. Personal and ambient PM2.5 exposure assessment in the city of Agra. Data Brief. 6, 495–502.
Habil, M., Massey, D., Taneja, A., 2013. Exposure of children studying in schools of India to PM levels and metal contamination: sources and their identification. Air Qual., Atmosphere Health 6, 575–587.
Habil, M., Taneja, A., 2011. Children's exposure to indoor particulate matter in naturally ventilated schools in India. Indoor Built Environ. 20, 430–448.
Han, X., Naeher, L.P., 2006. A review of traffic-related air pollution exposure assessment studies in the developing world. Environ. Int. 32, 106–120.
Harrison, R.M., Yin, J., 2000. Particulate matter in the atmosphere: which particle properties are important for its effects on health? Sci. Total Environ. 249, 85–101.
He, C., Morawska, L., Hitchins, J., Gilbert, D., 2004. Contribution from indoor sources to particle number and mass concentrations in residential houses. Atmos. Environ. 38, 3405–3415.
Hebert, J.R., Pednekar, M.S., Gupta, P.C., 2010. Forced expiratory volume predicts all-cause and cancer mortality in Mumbai, India: results from a population-based cohort study. Int. J. Epidemiol. 39, 1619–1627.
Hsu, S., Ito, K., Kendall, M., Lippmann, M., 2012. Factors affecting personal exposure to thoracic and fine particles and their components. J. Expo. Sci. Env. Epidemiol. 22, 439–447.
Huang, Y.-, Chen, H.-, Han, B.-, Liu, C.-, Chuang, H.-, Lin, L.-, Chuang, K.-, 2014. Personal exposure to household particulate matter, household activities and heart rate variability among housewives. PLoS One 9, e89969.
Huang, J., Deng, F., Wu, S., Guo, X., 2012. Comparisons of personal exposure to PM2.5 and CO by different commuting modes in Beijing, China. Sci. Total Environ. 425, 52–59.
Izhar, S., Goel, A., Chakraborty, A., Gupta, T., 2016. Annual trends in occurrence of Submicron particles in ambient air and health risk Posed by particle bound metals. Chemosphere 146, 582–590.
A., Jackson-Morris, K., Bleymann, E., Lyall, F., Aslam, T.S., Bam, I., Chowdhury, E.A., Daouda, M., Espinosa, J., Romo, R.J., Singh, S., Semple, 2016. Low-cost air quality monitoring methods to assess compliance with smoke-free regulations: A multi-center study in six low and middle-income countries. Nicotine & Tobacco Research.
Jai Devi, J., Gupta, T., Tripathi, S.N., Ujinwal, K.K., 2009. Assessment of personal exposure to Inhalable indoor and outdoor particulate matter for student residents of an academic campus (IIT-Kanpur). Inhal. Toxicol. 21, 1208–1222.
Jain, N.K., Thakkar, M.S., Jain, N., Rohan, K.A., Sharma, M., 2011. Chronic obstructive pulmonary disease: does gender really matter? Lung India : Official Organ Indian Chest Society 28, 258–262.
Jerrett, M., 2009. Commentary: global geographies of injustice in traffic-related air pollution exposure. Epidemiology 20, 231–233.
Johannesson, S., Gustafson, P., Molnar, P., Barregard, L., Sallsten, G., 2007. Exposure to fine particles (PM2.5 and PM1) and black smoke in the general population: personal, indoor, and outdoor levels. J. Expo. Sci. Env. Epidemiol. 17, 613–624.
Jones, A.P., 1999. Indoor air quality and health. Atmos. Environ. 33, 4535–4564.
Jones, M.R., Wipfli, H., Shahrir, S., Avila-Tang, E., Samet, J.M., Breysse, P.N., Navas-Acien, A., the FAMRI Bar, S.I., 2012. Secondhand tobacco smoke: an occupational hazard for smoking and non-smoking bar and nightclub employees. Tob. Control 22, 308–314.
Joon, V., Chandra, A., Bhattacharya, M., 2009. Household energy consumption pattern and socio-cultural dimensions associated with it: a case study of rural Haryana, India. Biomass-. Bioenergy 33, 1509–1512.
Jyethi, D.S., Khillare, P.K., Sarkar, S., 2014. Risk assessment of inhalation exposure to polycyclic aromatic hydrocarbons in school children. Environ. Sci. Pollut. Res., 21.
Kalappanavar, N.K., VinodKumar, C.S., Gouli, C., Sanjay, D., Nagendra, K., Basavarajappa, K.G., Patil, R., 2012. Carbon particles in airway macrophage as a surrogate marker in the early detection of lung diseases. the international. J. Occup. Environ. Med. 3, 68–75.
Karanasiou, A., Viana, M., Querol, X., Moreno, T., de Leeuw, F., 2014. Assessment of personal exposure to particulate air pollution during commuting in European cities—Recommendations and policy implications. Sci. Total Environ. 490, 785–797.
Kathuria, V., Khan, N.A., 2007. Vulnerability to air pollution: is there any inequity in exposure? Econ. Polit. Wkly. 42, 3158–3165.
Kaur, J., Prasad, V.M., 2011. Air nicotine monitoring for second hand smoke exposure in public Places in India. Indian journal of community. Medicine 36, 98–103.
Kaur, S., Senthilkumar, K., Verma, V.K., Kumar, B., Kumar, S., Katnoria, J., Sharma, C. S., 2013. Preliminary analysis of polycyclic aromatic hydrocarbons in air particles (PM10) in Amritsar, India: sources, apportionment, and possible risk implications to humans. Arch. Environ. Conta. Toxicol. 65, 382–395.
Keil, C., Kassa, H., Brown, A., Kumie, A., Tefera, W., 2010. Inhalation exposures to particulate matter and carbon monoxide during Ethiopian coffee ceremonies in Addis Ababa: a pilot study. J. Environ. Public Health 2010, 213960.
Kelly, F.J., 2003. Oxidative stress: its role in air pollution and adverse health effects. Occup. Environ. Med. 60, 612–616.
Kesavachandran, C., Pangety, B.S., BIhari, V., Fareed, M., Pathak, M.K., Srivastava, A. K., Mathur, N., 2013. Particulate matter concentration in ambient air and its effects on lung functions among residents in the national capital region, India. Environ. Monit. Assess. 185, 1265–1272.
Khan, M.M.H., Kraemer, A., 2008. Socio-economic factors explain differences in public health-related variables among women in Bangladesh: a cross-sectional study. BMC. Public Health, 8.
Khillare, P.K., Sarkar, S., 2012. Airborne Inhalable metals in residential areas of Delhi, India: distribution, source apportionment and health risks. Atmos. Pollut. Res. 3, 46–54.
Koistinen, K.J., Edwards, R.D., Mathys, P., Ruuskanen, J., Kunzli, N., Jantunen, M.J., 2004. Sources of fine particulate matter in personal exposures and residential indoor, residential outdoor and workplace Microenvironments in the Helsinki phase of the EXPOLIS study. Scand. J. Work, Environ. Health 30, 36–46.
Krishna, R.R., 2012. Current atmospheric aerosol research in India. Curr. Sci. 102, 440–451.
Kulkarni, M., Patil, R., 1999. Monitoring of daily integrated exposure of outdoor workers to respirable particulate matter in an urban region of India. Environ. Monit. Assess. 56, 129–146.
Kulshreshtha, P., Khare, M., 2011. Indoor exploratory analysis of gaseous pollutants and respirable particulate matter at residential Homes of Delhi, India. Atmos. Pollut. Res. 2, 337–350.
Kulshreshtha, P., Khare, M., 2011. Indoor exploratory analysis of gaseous pollutants and respirable particulate matter at residential homes of delhi, india. Atmospheric Pollution Research 2, 337-350.
Kulshrestha, A., Massey, D.D., Masih, J., Taneja, A., 2014. Source characterization of trace elements in indoor environments at urban, rural and roadside sites in a semi arid region of India. AAQR 14, 1738–1751.
Kumar, R., Gupta, N., Kumar, D., Mavi, A.K., Singh, K., Kumar, M., 2014. Monitoring of indoor particulate matter during burning of mosquito coil, incense sticks and Dhoop. Indian. J. Allergy, Asthma Immunol. 28, 68.
R., Kumar, 2011. Transport Sector Dynamics and its Contribution to Urban Health Burden in a Rapidly Growing Metropolitan Areas of India.
Kumar, R., Sharma, S.K., Thakur, J.S., Lakshmi, P.V.M., Sharma, M.K., Singh, T., 2010. Association of air pollution and mortality in the Ludhiana city of India: a time-series study. Indian J. Public Health 54, 98–103.
Kumar, N., Foster, A.D., 2009. Air quality interventions and spatial dynamics of air pollution in Delhi and its surroundings. Int. J. Environ. Waste Manag. 4, 85–111.
Kumar, P., Gupta, N.C., 2016. Commuter exposure to Inhalable, thoracic and Alveolic particles in various Transportation modes in Delhi. Sci. Total Environ. 541, 535–541.
Kumar, P., Gurjar, B.R., Nagpure, A.S., Harrison, R.M., 2011. Preliminary estimates of nanoparticle number emissions from road vehicles in Megacity Delhi and associated health impacts. Environ. Sci. Technol. 45, 5514–5521.
Kumar, R., Nagar, J.K., Kumar, H., Kushwah, A.S., Meena, M., Kumar, P., Raj, N., Singhal, M.K., Gaur, S.N., 2008. Indoor air pollution and respiratory function of children in ashok vihar, Delhi: an exposure-response study. Asia-Pac. J. Public Health 20, 36–48.
Kushwaha, R., Lal, H., Srivastava, A., Jain, V.K., 2012. Human exposure to particulate matter and their risk assessment over Delhi, India. Natl. Acad. Sci. Lett. 35, 497–504.
Lazaridis, M., Aleksandropoulou, V., Hanssen, J.E., Dye, C., Eleftheriadis, K., Katsivela, E., 2008. Inorganic and carbonaceous components in indoor/outdoor particulate matter in two residential houses in Oslo, Norway. J. Air Waste Manag. Assoc. 58, 346–356.
Lee, J., Lim, S., Lee, K., Guo, X., Kamath, R., Yamato, H., Abas, A.L., Nandasena, S., Nafees, A.A., Sathiakumar, N., 2010. Secondhand smoke exposures in indoor public Places in seven Asian countries. Int. J. Hyg. Environ. Health 213, 348–351.
J.T., Lee, S., Agrawal, S., Basu, S.A., Glantz, C., Millett, 2013. Association between smoke-free workplace and second-hand smoke exposure at home in india. Tobacco control.
Lim, S.S., Vos, T., Flaxman, A.D., Danaei, G., Shibuya, K., Adair-Rohani, H., AlMazroa, M.A., Amann, M., Anderson, H.R., Andrews, K.G., Aryee, M., Atkinson, C., Bacchus, L.J., Bahalim, A.N., Balakrishnan, K., Balmes, J., Barker-Collo, S., Baxter, A., Bell, M.L., Blore, J.D., Blyth, F., Bonner, C., Borges, G., Bourne, R., Boussinesq, M., Brauer, M., Brooks, P., Bruce, N.G., Brunekreef, B., Bryan-Hancock, C., Bucello, C., Buchbinder, R., Bull, F., Burnett, R.T., Byers, T.E., Calabria, B., Carapetis, J., Carnahan, E., Chafe, Z., Charlson, F., Chen, H., Chen, J.S., Cheng, A.T., Child, J.C., Cohen, A., Colson, K.E., Cowie, B.C., Darby, S., Darling, S., Davis, A., Degenhardt, L., Dentener, F., Des Jarlais, D.,C., Devries, K., Dherani, M., Ding, E.L., Dorsey, E.R., Driscoll, T., Edmond, K., Ali, S.E., Engell, R.E., Erwin, P.J., Fahimi, S., Falder, G., Farzadfar, F., Ferrari, A., Finucane, M.M., Flaxman, S., Fowkes, F.G.R., Freedman, G., Freeman, M.K., Gakidou, E., Ghosh, S., Giovannucci, E., Gmel, G., Graham, K., Grainger, R., Grant, B., Gunnell, D., Gutierrez, H.R., Hall, W., Hoek, H.W., Hogan, A., Hosgood, H.D., Hoy, D., Hu, H., Hubbell, B.J., Hutchings, S.J., Ibeanusi, S.E., Jacklyn, G.L., Jasrasaria, R., Jonas, J.B., Kan, H., Kanis, J.A., Kassebaum, N., Kawakami, N., Khang, Y., Khatibzadeh, S., Khoo, J., Kok, C., Laden, F., Lalloo, R., Lan, Q., Lathlean, T., Leasher, J.L., Leigh, J., Li, Y., Lin, J.K., Lipshultz, S.E., London, S., Lozano, R., Lu, Y., Mak, J., Malekzadeh, R., Mallinger, L., Marcenes, W., March, L., Marks, R., Martin, R., McGale, P., McGrath, J., Mehta, S., Memish, Z.A., Mensah, G. A., Merriman, T.R., Micha, R., Michaud, C., Mishra, V., Hanafiah, K.M., Mokdad, A. A., Morawska, L., Mozaffarian, D., Murphy, T., Naghavi, M., Neal, B., Nelson, P.K., Nolla, J.M., Norman, R., Olives, C., Omer, S.B., Orchard, J., Osborne, R., Ostro, B., Page, A., Pandey, K.D., Parry, C.D.H., Passmore, E., Patra, J., Pearce, N., Pelizzari, P. M., Petzold, M., Phillips, M.R., Pope, D., Pope, C.A., Powles, J., Rao, M., Razavi, H., Rehfuess, E.A., Rehm, J., Ritz, B., Rivara, F.P., Roberts, T., Robinson, C., Rodriguez-Portales, J., Romieu, I., Room, R., Rosenfeld, L.C., Roy, A., Rushton, L., Salomon, J. A., Sampson, U., Sanchez-Riera, L., Sanman, E., Sapkota, A., Seedat, S., Shi, P., Shield, K., Shivakoti, R., Singh, G.M., Sleet, D.A., Smith, E., Smith, K.R., Stapelberg, N.J.C., Steenland, K., StÃ¶ckl, H., Stovner, L.J., Straif, K., Straney, L., Thurston, G.D., Tran, J.H., Van Dingenen, R., van Donkelaar, A., Veerman, J.L., Vijayakumar, L., Weintraub, R., Weissman, M.M., White, R.A., Whiteford, H., Wiersma, S.T., Wilkinson, J.D., Williams, H.C., Williams, W., Wilson, N., Woolf, A.D., Yip, P.,

Zielinski, J.M., Lopez, A.D., Murray, C.J.L., Ezzati, M., 2012. A comparative risk assessment of burden of disease and injury attributable to 67 risk factors and risk factor clusters in 21 regions, 1990â€"2010: a systematic analysis for the global burden of disease study 2010. Lancet 380, 2224–2260.
Lin, T., Krishnaswamy, G., Chi, D.S., 2008. Incense smoke: clinical, structural and molecular effects on airway disease. Clin. Mol. Allergy: CMA 6 3-3.
Liu, H., Bartonova, A., Schindler, M., Sharma, M., Behera, S.N., Katiyar, K., Dikshit, O., 2013. Respiratory disease in relation to outdoor air pollution in Kanpur, India. Arch. Environ. Occup. Health 68, 204–217.
Long, C.M., Suh, H.H., Koutrakis, P., 2000. Characterization of indoor particle sources using continuous mass and size monitors. J. Air Waste Manag. Assoc. 50, 1236–1250.
Ma, Y., Harrad, S., 2015. Spatiotemporal analysis and human exposure assessment on polycyclic aromatic hydrocarbons in indoor air, settled house dust, and diet: a review. Environ. Int. 84, 7–16.
Malson, J.L., Sims, K., Murty, R., Pickworth, W.B., 2001. Comparison of the nicotine content of tobacco used in bidis and conventional cigarettes. Tob. Control 10, 181–183.
Manoj, M., Verma, A., Navyatha, M., 2015. Commute travel and its effect on housing tenure choice of males and females living in the urban and rural areas of Bangalore city in India. J. Transp. Geogr. 45, 62–69.
Martin, W.J.I., Glass, R.I., Araj, H., Balbus, J., Collins, F.S., Curtis, S., Diette, G.B., Elwood, W.N., Falk, H., Hibberd, P.L., Keown, S.E.J., Mehta, S., Patrick, E., Rosenbaum, J., Sapkota, A., Tolunay, H.E., Bruce, N.G., 2013. Household air pollution in low- and middle-income countries: health risks and research priorities. PLoS Med. 10, e1001455.
Masih, J., Singhvi, R., Kumar, R., Jain, V.K., Taneja, A., 2012. Seasonal variation and sources of polycyclic aromatic hydrocarbons (PAHs) in indoor and outdoor air in a semi arid tract of northern India. Aerosol Air Quality Res. 12, 515–525.
Massey, D., Kulshrestha, A., Masih, J., Taneja, A., 2012. Seasonal trends of PM10, PM5.0, PM2.5 & PM1.0 in indoor and outdoor environments of residential Homes located in north-central India. Build. Environ.; Int. Workshop Vent., Comf., Health Transp. Veh. 47, 223–231.
McKay, A.J., Mahesh, P.A., Fordham, J.Z., Majeed, A., 2012. Prevalence of COPD in India: a systematic review. Prim. Care Respir. J. 21, 313.
Meng, Q.Y., Turpin, B.J., Korn, L., Weisel, C.P., Morandi, M., Colome, S., Zhang, J., Stock, T., Spektor, D., Winer, A., Zhang, L., Lee, J.H., Giovanetti, R., Cui, W., Kwon, J., Alimokhtari, S., Shendell, D., Jones, J., Farrar, C., Maberti, S., 2004. Influence of ambient (outdoor) sources on residential indoor and personal PM2.5 concentrations: analyses of RIOPA data. J. Expo. Anal. Environ. Epidemiol. 15, 17–28.
Mohanraj, R., Azeez, P.A., 2004. Health effects of Airborne particulate matter and the Indian scenario. Curr. Sci. 87, 741–748.
Mönkkönen, P., Pai, P., Maynard, A., Lehtinen, K.E.J., Hämeri, K., Rechkemmer, P., Ramachandran, G., Prasad, B., Kulmala, M., 2005. Fine particle number and mass concentration measurements in urban Indian households. Sci. Total Environ. 347, 131–147.
Monn, C., Fuchs, A., Högger, D., Junker, M., Kogelschatz, D., Roth, N., Wanner, H.-, 1997. Particulate matter less than 10 μm (PM10) and fine particles less than 2.5 μm (PM2.5): relationships between indoor, outdoor and personal concentrations. Sci. Total Environ. 208, 15–21.
Monn, C., 2001. Exposure assessment of air pollutants: a review on spatial heterogeneity and indoor/outdoor/personal exposure to suspended particulate matter, nitrogen dioxide and ozone. Atmos. Environ. 35, 1–32.
Morawska, L., Afshari, A., Bae, G.N., Buonanno, G., Chao, C.Y.H., Hänninen, O., Hofmann, W., Isaxon, C., Jayaratne, E.R., Pasanen, P., Salthammer, T., Waring, M., Wierzbicka, A., 2013. Indoor aerosols: from personal exposure to risk assessment. Indoor Air 23, 462–487.
Moschandreas, D.J., Watson, J., D'Abreton, P., Scire, J., Zhu, T., Klein, W., Saksena, S., 2002. Chapter three: methodology of exposure modeling. Chemosphere 49, 923–946.
Nagaraja, C., Shashibhushan, B.L., Sagar, Asif, M., Manjunath, P.H., 2012. Hydrogen peroxide in exhaled breath condensate: a clinical study. Lung India : Official Organ Indian Chest Society 29, 123–127.
Namdeo, A., Ballare, S., Job, H., Namdeo, D., 2014. Commuter exposure to air pollution in Newcastle, U.K., and Mumbai, India. J. Hazard., Toxic., Radioact. Waste, A4014004.
Nandasena, Y.L.S., Wickremasinghe, A.R., Sathiakumar, N., 2010. Air pollution and health in Sri Lanka: a review of Epidemiologic studies. BMC. Public Health 10 300-300.
Nautiyal, J., Garg, M.L., Kumar, M.S., Khan, A.A., Thakur, J.S., Kumar, R., 2007. Air pollution and cardiovascular health in mandi-gobindgarh, punjab, India - A pilot study. Int. J. Environ. Res. Public Health 4, 268–282.
Nazaroff, W.W., 2004. Indoor particle dynamics. Indoor Air 14, 175–183.
Nieuwenhuijsen, M.J., Donaire-Gonzalez, D., Foraster, M., Martinez, D., Cisneros, A., 2014. Using personal sensors to assess the Exposome and acute health effects. Int. J. Environ. Res. Public Health 11, 7805–7819.
O'Neill, M.S., Jerrett, M., Kawachi, I., Levy, J.I., Cohen, A.J., Gouveia, N., Wilkinson, P., Fletcher, T., Cifuentes, L., Schwartz, J., 2003. Health, wealth, and air pollution: advancing theory and methods. Environ. Health Perspect. 111, 1861–1870.
Ott, W., 1995. Human exposure assessment: the birth of a new science. J. Expo. Anal. Environ. Epidemiol. 5, 449.
Padhi, B., Padhy, P., 2008. Assessment of intra-urban variability in outdoor air quality and its health risks. Inhal. Toxicol. 20, 973–979.
Pandey, A., Sadavarte, P., Rao, A.B., Venkataraman, C., 2014. Trends in multi-pollutant emissions from a technology-linked inventory for India: II. residential, agricultural and informal industry sectors. Atmos. Environ. 99, 341–352.
Pandit, G.G., Srivastava, P.K., Mohan Rao, A.M., 2001. Monitoring of indoor volatile organic compounds and polycyclic aromatic hydrocarbons arising from kerosene cooking fuel. Sci. Total Environ. 279, 159–165.
Pant, P., Shukla, A., Kohl, S.D., Chow, J.C., Watson, J.G., Harrison, R.M., 2015. Characterization of ambient PM2.5 at a pollution Hotspot in new Delhi, India and inference of sources. Atmos. Environ. 109, 178–189.
Pant, P., Harrison, R.M., 2012. Critical review of receptor modelling for particulate matter: a case study of India. Atmos. Environ. 49, 1–12.
Parikh, J., Smith, K., Laxmi, V., 1999. Indoor air pollution: a reflection on gender bias. Econ. Polit. Wkly. 34, 539–544.
Patil, R.S., Kumar, R., Menon, R., Shah, M.K., Sethi, V., 2013. Development of particulate matter speciation profiles for major sources in six cities in India. Atmos. Res. 132–133, 1–11.
Pervez, S., Chakrabarty, R.K., Dewangan, S., Watson, J.G., Chow, J.C., Matawle, J.L., 2015. Chemical speciation of aerosols and air quality degradation during the festival of lights (Diwali). Atmos. Pollut. Res. 7, 92–99.
PHFI, 2014. Ambient Air Pollution and Public Health: A Call to Action.
Prasad, R.K., Ravi Shankar, V., Saksena, S., 2003. Daily exposure to air pollution in indoor. Outdoor, -Veh. Micro-Environments: A Pilot Study Delhi, 57.
Raaschou-Nielsen, O., Beelen, R., Wang, M., Hoek, G., Andersen, Z.J., Hoffmann, B., Stafoggia, M., Samoli, E., Weinmayr, G., Dimakopoulou, K., Nieuwenhuijsen, M., Xun, W.W., Fischer, P., Eriksen, K.T., Sørensen, M., Tjønneland, A., Ricceri, F., de Hoogh, K., Key, T., Eeftens, M., Peeters, P.H., Bueno-de-Mesquita, H.B., Meliefste, K., Oftedal, B., Schwarze, P.E., Nafstad, P., Galassi, C., Migliore, E., Ranzi, A., Cesaroni, G., Badaloni, C., Forastiere, F., Penell, J., De Faire, U., Korek, M., Pedersen, N., Östenson, C.-, Pershagen, G., Fratiglioni, L., Concin, H., Nagel, G., Jaensch, A., Ineichen, A., Naccarati, A., Katsoulis, M., Trichpoulou, A., Keuken, M., Jedynska, A., Kooter, I.M., Kukkonen, J., Brunekreef, B., Sokhi, R.S., Katsouyanni, K., Vineis, P., 2016. Particulate matter air pollution components and risk for lung cancer. Environ. Int. 87, 66–73.
Rahman, M., Sakamoto, J., Fukui, T., 2003. Bidi smoking and oral cancer: a meta-analysis. Int. J. Cancer 106, 600–604.
Rahman, M., Fukui, T., 2000. Bidi smoking and health. Public health 114, 123–127.
Rajarathnam, U., Sehgal, M., Nairy, S., Patnayak, R.C., Chhabra, S.K., Kilnani, Ragavan, K.V.S., HEI Health, R.C., 2011. Part 2. time-series study on air pollution and mortality in Delhi. Res. Rep. (Health Eff. Inst.), 47–74.
Rajput, P., Sarin, M., Sharma, D., Singh, D., 2014. Characteristics and emission budget of carbonaceous species from post-harvest agricultural-waste burning in source region of the indo-gangetic plain. Tellus B, 66.
Ram, K., Sarin, M.M., 2011. Day–night variability of EC, OC, WSOC and inorganic ions in urban environment of indo-gangetic plain: implications to secondary aerosol formation. Atmos. Environ. 45, 460–468.
Ram, K., Sarin, M.M., 2010. Spatio-temporal variability in atmospheric abundances of EC, OC and WSOC over northern India. J. Aerosol Sci. 41, 88–98.
Raute, L.J., Gupta, P.C., Pednekar, M.S., 2011. Smoking ban and indoor air quality in restaurants in Mumbai, India. Indian. J. Occup. Environ. Med. 15, 68–72.
Rehman, I.H., Ahmed, T., Praveen, P.S., Kar, A., Ramanathan, V., 2011. Black carbon emissions from biomass and fossil fuels in rural India. Atmos. Chem. Phys. 11, 7289–7299.
Rosati, J.A., Thornburg, J., Rodes, C., 2008. Resuspension of particulate matter from carpet due to human activity. Aerosol Sci. Technol. 42, 472–482.
Roy, A.A., Baxla, S.P., Gupta, T., Bandyopadhyaya, R., Tripathi, S.N., 2009. Particles emitted from indoor combustion sources: size distribution measurement and chemical analysis. Inhal. Toxicol. 21, 837–848.
Sabapathy, A., Flachsbart, P.G., Saksena, S., 2012. Commuting patterns of employees in the information technology and traditional manufacturing sectors of Bangalore, India. Transp. Policy 19, 155–166.
Sadavarte, P., Venkataraman, C., 2014. Trends in multi-pollutant emissions from a technology-linked inventory for India: I. Industry and transport sectors. Atmos. Environ. 99, 353–364.
Saksena, S., Prasad, R.K., Shankar, V.R., 2007. Daily exposure to air pollutants in indoor, outdoor and in-vehicle micro-environments: a pilot study in Delhi. Indoor Built Environ. 16, 39–46.
A., Saraswat, M., Kandlikar, M., Brauer, A.K., Srivastava, 2016. PM2.5 population exposure in new delhi using a probabilistic simulation framework. Environmental science & technology.
Saraswat, A., Apte, J.S., Kandlikar, M., Brauer, M., Henderson, S.B., Marshall, J.D., 2013. Spatiotemporal land use regression models of fine, Ultrafine, and black carbon particulate matter in new Delhi, India. Environ. Sci. Technol. 47, 12903–12911.
Sarkar, S., Khillare, P.S., 2013. Profile of PAHs in the Inhalable particulate fraction: source apportionment and associated health risks in a tropical Megacity. Environ. Monit. Assess. 185, 1199–1213.
Satsangi, P.G., Yadav, S., Pipal, A.S., Kumbhar, N., 2014. Characteristics of trace metals in fine (PM2.5) and Inhalable (PM10) particles and its health risk assessment along with in-silico approach in indoor environment of India. Atmos. Environ. 92, 384–393.
See, S.W., Balasubramanian, R., 2011. Characterization of fine particle emissions from incense burning. Build. Environ. 46, 1074–1080.
Semple, S., Green, D.A., McAlpine, G., Cowie, H., Seaton, A., 2008. Exposure to particulate matter on an Indian stone-crushing site. Occup. Environ. Med. 65, 300–305.
Shah, A.S., Langrish, J.P., Nair, H., McAllister, D.A., Hunter, A.L., Donaldson, K., Newby, D.E., Mills, N.L., 2013. Global association of air pollution and heart failure: a systematic review and meta-analysis. Lancet 382, 1039–1048.

Sharma, M., Kumar, V.N., Katiyar, S.K., Sharma, R., Shukla, B.P., Sengupta, B., 2004. Effects of particulate air pollution on the respiratory health of subjects who live in three areas in Kanpur, India. Arch. Environ. Health: Int. J. 59, 348–358.
Shimada, Y., Matsuoka, Y., 2011. Analysis of indoor PM2.5 exposure in Asian countries using time use survey. Sci. Total Environ. 409, 5243–5252.
Shridhar, V., Khillare, P.S., Agarwal, T., Ray, S., 2010. Metallic species in ambient particulate matter at rural and urban location of Delhi. J. Hazard. Mater. 175, 600–607.
Siddique, S., Ray, M., Lahiri, T., 2011. Effects of air pollution on the respiratory health of children: a study in the capital city of India. Air Qual., Atmosphere Health 4, 95–102.
Siddique, S., Banerjee, M., Ray, M., Lahiri, T., 2010. Air pollution and its impact on lung function of children in Delhi, the capital city of India. Water, Air, Soil. Pollut. 212, 89–100.
Singh, A., Rajput, P., Sharma, D., Sarin, M.M., Singh, D., 2014. Black carbon and elemental carbon from Postharvest agricultural-waste burning emissions in the indo-gangetic plain. Adv. Meteorol., 2014.
Singh, R.J., Lal, P.G., 2011. Second-hand smoke: a neglected public health challenge. Indian J. Public Health 55, 192–198.
Singh, D.K., Gupta, T., 2016a. Effect through inhalation on human health of PM1 bound polycyclic aromatic hydrocarbons collected from foggy days in northern part of India. J. Hazard. Mater. 306, 257–268.
Singh, D.K., Gupta, T., 2016b. Source apportionment and risk assessment of PM1 bound trace metals collected during foggy and non-foggy episodes at a representative site in the indo-gangetic plain. Sci. Total Environ. 550, 80–94.
Smith, K.R., 2000. National burden of disease in India from indoor air pollution. Proc. Natl. Acad. Sci. USA 97, 13286–13293.
Spengler, J.D., Chen, Q., 2000. Indoor air quality factors in designing a healthy building. Annu. Rev. Energy Environ. 25, 567–600.
Srimuruganandam, B., Shiva Nagendra, S.M., 2012. Application of positive matrix factorization in characterization of PM10 and PM2.5 emission sources at urban roadside. Chemosphere 88, 120–130.
Steinle, S., Reis, S., Sabel, C.E., 2013. Quantifying human exposure to air pollution: moving from static monitoring to spatio-temporally resolved personal exposure assessment. Sci. Total Environ. 443, 184–193.
Sudheer, A.K., Rengarajan, R., 2012. Atmospheric mineral dust and trace metals over urban environment in western India during winter. Aerosol Air Quality Research 12, 923–933.
Sukhsohale, N., Thakre, U.W., Ughade, S.N., S.S., 2013. Peak expiratory flow rates among women exposed to different cooking fuels in rural India. Southeast Asian J. Trop. Med. Public Health, 44.
Sundell, J., 2004. On the history of indoor air quality and health. Indoor Air 14, 51–58.
Suresh, Y., Sailaja Devi, M.M., Manjari, V., Das, U.N., 2000. Oxidant stress, antioxidants and nitric oxide in traffic police of Hyderabad, India. Environ. Pollut. 109, 321–325.
Swamy, S., Pai, M., Kulshrestha, S., 2015. Commuters' Exposure to PM2.5, Case Study- Ahmedabad . In: TRB 94th Annual Meeting Compendium of Papers.
Swerts, E., Pumain, D., Denis, E., 2014. The future of India's urbanization. Futures 56, 43–52.
Tiwari, S., Srivasatava, A.K., Bisht, D.S., Bano, T., Singh, S., Behura, S., Srivastava, M.K., Chate, D.M., Padmanabhamurty, B., 2009. Black carbon and chemical characteristics of PM10 and PM2.5 at an urban site in north India. J. Atmos. Chem. 62, 193–209.
Tiwari, S., Bisht, D.S., Srivastava, A.K., Pipal, A.S., Taneja, A., Srivastava, M.K., Attri, S. D., 2014. Variability in atmospheric particulates and meteorological effects on their mass concentrations over Delhi, India. Atmos. Res. 145–146, 45–56.
Trivedi, D.K., Ali, K., Beig, G., 2014. Impact of meteorological parameters on the development of fine and coarse particles over Delhi. Sci. Total Environ. 478, 175–183.
Valavanidis, A., Fiotakis, K., Vlachogianni, T., 2008. Airborne particulate matter and human health: toxicological assessment and importance of size and composition of particles for oxidative damage and carcinogenic mechanisms. J. Environ. Sci. Health, Part C. 26, 339–362.
Varshney, P., Saini, R., Taneja, A., 2015. Trace element concentration in fine particulate matter (PM2.5) and their bioavailability in different Microenvironments in Agra, India: a case study. Environ. Geochem. Health, 1–13.
Villalobos, A.M., Amonov, M.O., Shafer, M.M., Devi, J.J., Gupta, T., Tripathi, S.N., Rana, K.S., Mckenzie, M., Bergin, M.H., Schauer, J.J., 2015. Source apportionment of carbonaceous fine particulate matter (PM2.5) in two contrasting cities across the Indo–Gangetic plain. Atmos. Pollut. Res. 6, 398–405.
Wallace, L., 1995. Indoor particles: a review. J. Air Waste Manag. Assoc. 46, 98.
Wei See, S., Karthikeyan, S., Balasubramanian, R., 2006. Health risk assessment of occupational exposure to particulate-phase polycyclic aromatic hydrocarbons associated with Chinese, Malay and Indian cooking. J. Environ. Monit. 8, 369–376.
WHO, 2014. 7 million premature deaths annually linked to air pollution. Accessed April 12 2015. http://www.who.int/mediacentre/news/releases/2014/air-pollution/en/.
Wipfli, H., Avila-Tang, E., Navas-Acien, A., Kim, S., Onicescu, G., Yuan, J., Breysse, P., Samet, J.M., 2008. Secondhand smoke exposure among women and children: evidence from 31 countries. Am J. Public Health 98, 672–679.
Yadav, S., Satsangi, P.G., 2013. Characterization of particulate matter and its related metal Toxicity in an urban location in south west India. Environ. Monit. Assess. 185, 7365–7379.
Yadav, R., Beig, G., Jaaffrey, S.N.A., 2014. The linkages of anthropogenic emissions and meteorology in the rapid increase of particulate matter at a foothill city in the Arawali range of India. Atmos. Environ. 85, 147–151.
Yamamoto, S.S., Phalkey, R., Malik, A.A., 2014. A systematic review of air pollution as a risk factor for cardiovascular disease in south Asia: limited evidence from India and Pakistan. Int. J. Hyg. Environ. Health 217, 133–144.
Yang, J., Modi, B.V., Tamplin, S.A., Aghi, M.B., Dave, P.V., Cohen, J.E., 2015. Air nicotine levels in public Places in Ahmedabad, India: before and after implementation of the smoking ban. Indian journal of community. Indian J. Commu. Med. 40, 27–32.
Yocom, J.E., 1982. A critical review. J. Air Pollut. Control Assoc. 32, 500–520.
Zartarian, V., Bahadori, T., McKone, T., 2004. Adoption of an official ISEA glossary. J. Expo. Anal. Environ. Epidemiol. 15, 1–5.