Review article

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

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A B S T R A C T

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.
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 mis-classification (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 PM2.5 (PM with aerodynamic diameter less than 2.5 μm) concentrations in North America (12 μg/m3) and Asia (38 μg/m3) with the highest levels of PM2.5-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 PM2.5 levels are often very high. A recent global study on ambient PM concentrations reported a twofold difference between average population weighted PM2.5 (PM with aerodynamic diameter less than 2.5 μm) concentrations in North America (12 μg/m3) and Asia (38 μg/m3) with the highest levels of PM2.5-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).

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...
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 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). Annual PM10 and PM2.5 concentrations in New Delhi were reported to be 222 ± 142 g/m³ and 130 ± 103 g/m³ (Tiwari et al. 2014) while in Agra, average concentrations ranged from 112 ± 212 g/m³ to 91 ± 157 g/m³ (Chakraborty and Gupta 2010) in the monsoon period (August–September) of that year.

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

PM10 concentrations in India often exceed the national air quality standards, and in 2010, 140 out of 176 cities were found to exceed the PM2.5 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. 2012). HAP can also be an important contributor to ambient PM2.5, 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 (Chakraborty et al. 2014) and festive fireworks and biomass burning (Beig et al. 2013a; Perozzi 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).

### Table 1

<table>
<thead>
<tr>
<th>City</th>
<th>Sampling year</th>
<th>Type</th>
<th>PM10</th>
<th>PM2.5</th>
<th>PM1</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Delhi</td>
<td>September 2010-August 2012</td>
<td>Avg ± SD</td>
<td>222 ± 142</td>
<td>130 ± 103</td>
<td></td>
<td>Tiwari et al. (2014)</td>
</tr>
<tr>
<td>Delhi</td>
<td>January-December 2007</td>
<td></td>
<td>219 ± 84</td>
<td>97 ± 56</td>
<td></td>
<td>Tiwari et al. (2009)</td>
</tr>
<tr>
<td>Agra</td>
<td>May 2010-December 2012</td>
<td></td>
<td>65.4 ± 3.1</td>
<td>113 ± 36</td>
<td></td>
<td>(Dubeby et al. 2015)</td>
</tr>
<tr>
<td>Agra</td>
<td>November 2011-June 2012</td>
<td></td>
<td>60 ± 39</td>
<td></td>
<td></td>
<td>(Bergin et al. 2015)</td>
</tr>
<tr>
<td>Raipur</td>
<td>Winter 2013–14</td>
<td></td>
<td>153.28 ± 28.53</td>
<td>72.3 ± 31.3</td>
<td></td>
<td>(Singh and Gupta 2016a)</td>
</tr>
<tr>
<td>Raipur</td>
<td>January–December 2006</td>
<td></td>
<td>246 ± 68</td>
<td></td>
<td></td>
<td>(Giri et al. 2013)</td>
</tr>
<tr>
<td>Pune</td>
<td>June 2011-May 2012</td>
<td></td>
<td>113.8 ± 51.6</td>
<td>31.3 ± 188</td>
<td></td>
<td>Yadav and Satansingi (2013)</td>
</tr>
<tr>
<td>Kolkata</td>
<td></td>
<td></td>
<td>200 ± 127</td>
<td></td>
<td></td>
<td>Das et al. (2015)</td>
</tr>
<tr>
<td>Hyderabad</td>
<td></td>
<td></td>
<td>445 ± 210</td>
<td></td>
<td></td>
<td>Guttikunda and Kopakka (2014)</td>
</tr>
<tr>
<td>Hyderabad</td>
<td>June 2004–May 2005</td>
<td></td>
<td>105.2 ± 28.6</td>
<td>72.6 ± 18</td>
<td></td>
<td>(Gummieneni et al. 2011)</td>
</tr>
<tr>
<td>Mumbai</td>
<td></td>
<td></td>
<td>138.4 ± 46.4</td>
<td>60.72 ± 20.1</td>
<td></td>
<td>Panda et al. (2015)</td>
</tr>
<tr>
<td>Bhubaneswar</td>
<td>December 2012-February 2013</td>
<td></td>
<td>327 ± 112</td>
<td>82 ± 143</td>
<td></td>
<td>Massey et al. (2012)</td>
</tr>
<tr>
<td>Kanpur</td>
<td>July 2008-May 2009</td>
<td></td>
<td>31–199</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

*a 8 h average.

*b 12 h average.
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 PM10 and PM2.5 (Pant et al. 2015; Shridhar et al. 2010; Sudheer and Rengarajan 2012), and likewise, high PM 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 PM2.5 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 PM10 with a limited number of studies on PM2.5 and PM1.

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 PM10 and PM2.5, ozone (O3), sulfur dioxide (SO2), ammonia (NH3), nitrogen dioxide (NO2), 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 PM2.5 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 years.
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/m³) with variations between urban and rural areas. In central India (Raipur-Bhilai), Gadkari and Pervez (2008) observed average PM5 concentrations ranging from 135 to 551 μg/m³ across houses with different ventilation mechanisms, and in Delhi, Saksea et al. (2007) reported concentrations as high as 350 μg/m³ (office/shop) and 640 μg/m³ (bedroom) in indoor environments. In Kanpur, Devi et al. (2009) reported PM2.5 concentrations ranging from 22.5 to 165.2 μg/m³ in indoor microenvironments. In Agra, indoor and outdoor PM2.5 concentrations at a roadside site were recorded as 161 ± 62 and 160 ± 62 μg/m³, while at an urban site, the concentrations were 109 ± 48 and 123 ± 45 μg/m³ respectively (Massey et al. 2012).

For the same size fraction, concentrations ranging from 34 to 60 μg/m³ in summer and 197-936 μg/m³ 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 PM2.5 and PM10 concentrations of 89.7 ± 43.2 μg/m³ and 138.2 ± 68.2 μg/m³ respectively. Compared to these studies, Varshney et al. (2015) reported relatively lower indoor PM concentrations in Agra in urban (45.33 μg/m³), roadside (36.71 μg/m³) and rural (71.23 μg/m³) households, and Habil et al. (2016) reported 123.79 ± 35.32 μg/m³. Indoor concentrations in hospitality venues (bars, cafés, restaurants etc.) were reported to be in the same range with an average concentration of 207 μg/m³ although a much wider range (20–748 μg/m³) 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 PM2.5 concentrations were reported as 14.4 and 15.5 μg/m³ for indoor and outdoor environments while PE PM2.5 level was recorded at 480 μg/m³ for indoor and 640 μg/m³ for outdoor environments. In Gothenburg (Sweden), median PM2.5 concentrations of 5-7 μg/m³ and 6-8 μg/m³ were reported for indoor and outdoor environments while in Gothenburg (Sweden), median PM2.5 concentrations were reported to be 8.4, 8.6 and 6.4 μg/m³ for personal, indoor and outdoor air samples while in case of PM1, the concentrations were 5.4, 6.2 and 5.2 respectively (Lazaridis et al. 2008; Johansson 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/m³, number concentrations in #/cm³, U-urban, RO-roadside, R-rural, ME- microenvironment).

<table>
<thead>
<tr>
<th>References</th>
<th>City</th>
<th>Year</th>
<th>ME/ Season</th>
<th>PM1</th>
<th>PM2.5</th>
<th>PM5</th>
<th>PM10</th>
<th>Particle Number</th>
</tr>
</thead>
<tbody>
<tr>
<td>Prasad et al. (2003), Saksea et al. (2007)</td>
<td>Delhi (U)</td>
<td>February–March 1997</td>
<td>Living Room</td>
<td>480</td>
<td></td>
<td></td>
<td></td>
<td>4400-101,000</td>
</tr>
<tr>
<td>Mönkönnen et al. (2005)</td>
<td>Nagpur (U)</td>
<td>2002</td>
<td>Flat</td>
<td>550.9 ± 27.4</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Kumar et al. (2008)</td>
<td>Delhi (U)</td>
<td>June–July 2004</td>
<td>Indoor (summer)</td>
<td>349</td>
<td>155</td>
<td>654</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Kulshrestha et al. (2008)</td>
<td>Delhi (U)</td>
<td>April 2000–February 2005</td>
<td>Indoor (winter)</td>
<td>654</td>
<td>349</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Devi et al. (2013)</td>
<td>Kanpur (U)</td>
<td>September 2009-May 2010</td>
<td>Annual Indoor (winter)</td>
<td>181 ± 84</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Satsangi et al. (2014)</td>
<td>Pune (U)</td>
<td>June 2012-May 2013</td>
<td>Annual Pre-monsoon Post-monsoon</td>
<td>138.2 ± 68.2</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
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$_{2.5}$ 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 μg/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, Mönkönen 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 μg/m$^3$ and 89.7 ± 43.2 μg/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, Kullarni and Patil (1999) found no correlation between the type of fuel used and the PM$_{10}$ 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
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/m³ have been reported for non-smoking and smoking entertainment (bar/nightclub) venues (Jones et al., 2012) whereas home air nicotine concentrations of non-smoking and smoking entertainment (bar/nightclub) venues without smokers respectively (Wip et al., 2006). 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 (Abdullabhi 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.
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 (Colville 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 PM10 exposure while travelling in bus (502.78 μg/m³), and the lowest exposure in air-conditioned cars (68.78 μg/m³) while in a similar analysis in Bangalore, average PM10 exposures for two-wheelers during the peak hour were recorded as 403 μg/m³. In an older study, PM2.5 concentrations as high as 2860 μg/m³ were reported for two-wheelers (Prasad et al. 2003). In Delhi, Goel et al. (2015a) reported lowest exposures to PM2.5 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 ~10⁴ cm⁻³ and 10⁵ 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 be 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 PM10 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 PM2.5 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 cities 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 including 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>
<thead>
<tr>
<th>References</th>
<th>City</th>
<th>Site Type</th>
<th>Season</th>
<th>PM1</th>
<th>PM2.5</th>
<th>PM10</th>
<th>SPM</th>
</tr>
</thead>
<tbody>
<tr>
<td>Goyal and Khare (2009)</td>
<td>Delhi</td>
<td>School building</td>
<td>Winter (WD)</td>
<td>293.3 ± 180.3</td>
<td>359.9 ± 187.7</td>
<td>1181.1 ± 578.3</td>
<td>644 ± 672.2</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Winter (WE)</td>
<td>328.5 ± 229.6</td>
<td>366.1 ± 251.8</td>
<td>644 ± 672.2</td>
<td>644 ± 672.2</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Non-winter (WD)</td>
<td>45.8 ± 26.5</td>
<td>71 ± 37.3</td>
<td>1181.1 ± 578.3</td>
<td>410.6 ± 196.4</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Non-winter (WE)</td>
<td>41.3 ± 36.7</td>
<td>54.6 ± 40.2</td>
<td>1181.1 ± 578.3</td>
<td>133.3 ± 71.2</td>
</tr>
<tr>
<td>Habil and Massey (2011)</td>
<td>Agra</td>
<td>School building (roadside)</td>
<td>Winter (WD)</td>
<td>259.5 ± 55.62</td>
<td>240.95 ± 152.79</td>
<td>524.76 ± 169.36</td>
<td>157.6 ± 87.94</td>
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<td></td>
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<td></td>
<td>Winter (WE)</td>
<td>67.46 ± 31.12</td>
<td>92.48 ± 45.74</td>
<td>157.6 ± 87.94</td>
<td>230</td>
</tr>
<tr>
<td>Saksena et al. (2007)</td>
<td>Delhi</td>
<td>School building (residential)</td>
<td>Winter (WD)</td>
<td>20 ± 12</td>
<td>36 ± 15</td>
<td>136 ± 60</td>
<td>136 ± 60</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Winter (WE)</td>
<td>23 ± 14</td>
<td>33 ± 16</td>
<td>76 ± 42</td>
<td>76 ± 42</td>
</tr>
<tr>
<td>Chithra and Shiva Nagendra (2014)</td>
<td>Chennai</td>
<td>School building (roadside)</td>
<td>Indoor</td>
<td>29.27</td>
<td>37.57</td>
<td>53.46</td>
<td>53.46</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Outdoor</td>
<td>128.45 ± 37.58</td>
<td>222.23 ± 30</td>
<td>222.23 ± 30</td>
<td>222.23 ± 30</td>
</tr>
<tr>
<td>Jyethi et al. (2014)</td>
<td>Delhi</td>
<td>School building (background)</td>
<td>Winter (WD)</td>
<td>20 ± 12</td>
<td>36 ± 15</td>
<td>136 ± 60</td>
<td>136 ± 60</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Winter (WE)</td>
<td>23 ± 14</td>
<td>33 ± 16</td>
<td>76 ± 42</td>
<td>76 ± 42</td>
</tr>
<tr>
<td>Habil et al. (2016)</td>
<td>Agra</td>
<td>Outdoor</td>
<td>Winter (WD)</td>
<td>20 ± 12</td>
<td>36 ± 15</td>
<td>136 ± 60</td>
<td>136 ± 60</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Winter (WE)</td>
<td>23 ± 14</td>
<td>33 ± 16</td>
<td>76 ± 42</td>
<td>76 ± 42</td>
</tr>
</tbody>
</table>
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 PM10 and gaseous pollutants such as CO, and in some cases, for PM2.5 and PM1. 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/m3. In a recent analysis using land use regression (LUR) modelling, Saraswat et al. (2016) reported annual average PM2.5 exposure ranging between 109 and

### Table 5
Summary of reported air pollutant concentrations in various transport micro-environments (mass in μg/m3, particle number in #/cm3).

<table>
<thead>
<tr>
<th>References</th>
<th>City</th>
<th>Transport Mode</th>
<th>PM10</th>
<th>PM2.5</th>
<th>PM10</th>
<th>BC</th>
<th>Particle Number</th>
</tr>
</thead>
<tbody>
<tr>
<td>Saksena et al. (2007)</td>
<td>Delhi</td>
<td>Three-wheeler</td>
<td>810</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Bus</td>
<td>800</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Namdeo et al. (2014)</td>
<td>Mumbai</td>
<td>Bus</td>
<td></td>
<td>500.78 ± 157.7</td>
<td>68.78 ± 15.8</td>
<td>156.18 ± 32.4</td>
<td></td>
</tr>
<tr>
<td>Saksena et al. (2007)</td>
<td>Delhi</td>
<td>A/C Car</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sabapathy et al. (2012)</td>
<td>Bangalore</td>
<td>Two-wheeler</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Devi et al. (2009)</td>
<td>Kanpur</td>
<td>Three-wheeler</td>
<td></td>
<td>168.6</td>
<td>242.9</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Kumar and Gupta (2016)</td>
<td>New Delhi</td>
<td>Car</td>
<td>175.9</td>
<td>255.8</td>
<td></td>
<td></td>
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</tr>
<tr>
<td>Kumar and Gupta (2016)</td>
<td>New Delhi</td>
<td>Bus</td>
<td>225.8</td>
<td>296.9</td>
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</tr>
<tr>
<td>Kumar and Gupta (2016)</td>
<td>New Delhi</td>
<td>Motorcycle</td>
<td>168.2</td>
<td>229.4</td>
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</tr>
</tbody>
</table>

*Bus Rapid Transit System.*

### Table 6
Summary of reported personal exposure to PM (mass in μg/m3, particle number in #/cm3).

<table>
<thead>
<tr>
<th>References</th>
<th>City</th>
<th>Season</th>
<th>PM2.5</th>
<th>PM1</th>
<th>PM2.5–10</th>
<th>PNC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Jai Devi et al. (2013)</td>
<td>Kanpur</td>
<td>Pre-monsoon</td>
<td>30.5 ± 2.7</td>
<td>12 ± 9</td>
<td>52.9 ± 17.2</td>
<td>17,656 ± 7,014 (PM1)</td>
</tr>
<tr>
<td>Habil et al. (2016)</td>
<td>Agra</td>
<td>Post-monsoon</td>
<td>37 ± 26</td>
<td>24 ± 21</td>
<td>34.5 ± 20.4</td>
<td>177.7 ± 147.4 (PM2.5)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Winter</td>
<td>135.28 ± 44.86 (Home)</td>
<td></td>
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<td>422.8 ± 359.3 (PM2.5)</td>
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<td></td>
<td></td>
<td></td>
<td>140.24 ± 42.21 (School)</td>
<td></td>
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<td></td>
<td></td>
<td></td>
<td>125.78 ± 40.24 (Office)</td>
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</table>
125 μg/m³ 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 μg/m³ 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 concentrations 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 (PM10) 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 H2O2 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 FEV1-forced expiratory volume) in people residing in areas with high PM levels (Siddique et al. 2011; Kesava Chandran et al. 2013) and this is particularly important since FEV1 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, PM2.5 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). PM2.5 concentrations as high as 390 μg/m³ 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 ~ 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 (PM2.5) (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 (Broych et al. 2012). A new study in southern India-Tamil Nadu Air Pollution and Health Effects (TAPHIE) is assessing exposure to PM2.5 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 the 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 (Duffy 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) and Izhar et al. (2016) reported higher concentrations of PM$_{2.5}$ as well individual chemical components (metals, PAHs) in winter, and Baixla et al. (2009) observed higher PM 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; Bröich 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 µg/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 µg/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 cardiovascular disease risk associated with exposure to PM.
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; Kalapannavar 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 PM10–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 (PM10 or PM2.5), 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 may 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 result 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 considers 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 take into account for income disparities, and consideration of pollution exposure needs to be integrated in the design process so as to minimize pollution exposure, 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 include 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.

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