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REVIEW

World air particulate matter: sources, distribution and health effects

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Abstract Particulate matter (PM) is both a major driver of climate change and a source of toxicity for health. In the upper atmosphere, particulate matter modifies the earth radiation budget, cloud formation and acts as a reaction center for air pollutants. In the lower atmosphere, particulate matter changes atmospheric visibility and alters biogeochemical cycles and meteorology. Most critical effects are observed in ambient air, where particulate matter degrades human health. Here we review the sources, spatial and temporal variability, and toxicity of PM_{10}, the particulate matter having particle sizes 10 micrometers or less in diameter, in world regions. For that we analyzed information from the world wide web and databases from government organizations after the year 2000. Findings show that PM_{10} is a major risk in both developed and developing countries. This risk is more severe in Asian countries compared to Europe and USA, where decreasing trends are recorded during the last two decades. Meteorological factors modify particulate matter variations at local and regional levels. $PM_{2.5}/PM_{10}$ ratio provides information of particulate matter sources under different environment conditions. Crustal matter, road traffic and combustion of fuels are major sources of particulate matter pollution. Health studies indicate that long-term exposure to particulate matter has multiple health effects in people from all age groups. Identification of possible sources and their control with regular epidemiological monitoring could decrease the impact of particulate matter pollution.

Keywords Particulate matter · PM_{10} · Source apportionment · Health effects · Urban · Meteorology

Abbreviations

WHO	World Health Organization
NAAQS	National Ambient Air Quality Standard
CPCB	Central Pollution Control Board
PM	Particulate matter
PM_{10}	Particulate matters of 10 μm size or less
OR	Odds ratio
CI	Confidence interval
IQR	Interquartile range

Introduction

The air quality in major cities of both the developing and developed countries in the world is deteriorating with increase in uncontrolled traffic growth, urban sprawl, increase in urban population, reduction in urban forest and increase in traffic emissions (Kim et al. 2015; Kulshrestha et al. 2009; Pascal et al. 2014; Raaschou-Nielsen et al. 2013; Rashki et al. 2013; Samet et al. 2000; Sharma et al. 2014b; Shen et al. 2010; Von Schneidemesser et al. 2015). PM_{10} (particulate matter having particle size 10 micrometers or less in diameter) is one of the major air pollutants having made up of solid and liquid particles floating in air and is respirable in nature and thus can reach deeper into the respiratory system. Concentrations and toxicity of particulate matter depend on their composition, shape and size of particles, presence of other pollutants and prevailing meteorological factors (Arruti et al. 2012; Clements et al.

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2014; Coronas et al. 2009; Jorquera 2009; Kassomenos et al. 2012; Rashki et al. 2013; Wickramasinghe et al. 2011).

PM_{10} particles in the atmosphere are a threat to all life forms and one of the major indicators of air pollution. After industrialization to the present time, PM_{10} has become one of the major air pollutants in urban, suburban and even in rural and remote regions of the world (Fang and Chang 2010; Koulouri et al. 2008; Kulshrestha et al. 2009; Li et al. 2014; WHO 2016). Most of the urban cities in the world are having PM_{10} levels above the WHO and their countries respective standards (WHO 2016).

Several reports have confirmed the negative impact of PM_{10} on health as congenital heart defects (Agay-Shay et al. 2013), ischemic heart disease (Zhang et al. 2014), respiratory and circulatory mortality (Li et al. 2013), preterm-birth risk (Schifano et al. 2013), mutagenicity and DNA damage (Coronas et al. 2009), fetal growth characteristics and adverse birth outcomes (van den Hooven et al. 2012), cancer risk (Díaz-Robles et al. 2013) and inflammatory responses (Silbajoris et al. 2011).

Apart from the adverse health effects, PM_{10} is also responsible for reducing atmospheric visibility as an important component of smog, reduction of photosynthesis in plants by deposition on leaf surfaces of plants, deposition of minerals and metals in soil, thus altering soil physicochemical properties and also affecting meteorological processes and atmospheric chemistry (Grantz et al. 2003; Lin et al. 2012; Von Schneidemesser et al. 2015).

Several studies have highlighted the severity of particulate matter exposure with loss in terms of years of life (Pascal et al. 2014; Schifano et al. 2013). A health impact assessment study by Keuken et al. (2011) showed an increase in gain in life years saved up to 13 months per person with a decrease in exposure of PM_{10}, which is directly related to decrease in combustion sources of aerosol.

PM_{10} concentrations in the atmosphere is regulated by local sources, dispersion and long rate transport pattern, industrial activity, combustion of fuels, local traffic activity, fire and burning activities, prevailing meteorological conditions, land-use pattern, topography and long-term climate conditions (Jorquera and Barraza 2012; Maenhaut et al. 2016; Von Schneidemesser et al. 2015; Spindler et al. 2013; Toledo et al. 2008). Natural sources of particulate matter like volcanic eruption, dust storm, forest fire and pollen grains also have a significant influence in local and global concentrations of PM_{10}. But the severity of exposure and sources of PM_{10} are significantly different in different regions of the world (Guttikunda et al. 2013; Li et al. 2013; Tiwari et al. 2015; Zhang et al. 2014).

For proper understanding of PM_{10} sources and its effects, it is necessary to analyze the global scenario of particulate matter distribution and its effects to identify special patterns and major health anomalies in different urban, suburban, background and rural environments, especially from Asian and African continents, where studies are limited and have huge local variability. Considering these important points, the objective of this review article is to identify spatial variability in PM_{10} concentrations, sources, meteorological influences, $PM_{2.5}/PM_{10}$ ratio and their health effects based on peer-reviewed articles to provide a current scenario of PM_{10} status and trend to academicians, epidemiologists, urban architecture and policy makers to form guidelines and sustainable mitigation approaches to control particulate matter pollution for improving health quality in different regions of the world.

Literature search

Data of respirable particulate matter or PM_{10} were collected through different search engines with specific keywords such as PM_{10}, respirable particulate matter, health effects of PM_{10}, sources of PM_{10} pollution. For assessment of previous years and recent PM_{10} global status data were obtained through WHO, World Bank and national monitoring networks of China, India, USA and European Union. Only papers published after 2000 were further considered for PM_{10} status and for source apportionment study.

Global status of PM_{10}

Past and present status of PM_{10}

Based on the data of global model of ambient particulates, a 22% reduction in global PM_{10} level has occurred in last two decades, contributed mostly by developed countries (Fig. 1) (Pandey et al. 2006). Russian Federation tops the list with maximum 59.6% reduction with Uzbekistan (59.6%) followed by Greece (53.7%), Ukraine (53.6%), Japan (52.2%) and USA (51.4%) (Pandey et al. 2006). Strong regulation, quality of roads, maintenance of automobiles and planed urban development led to improve the PM levels in developed countries. In Asian countries, China showed 32.5% reduction, whereas only 18% reduction was observed in case of India (Fig. 1). Some countries in Asia and Africa showed significant increases in PM_{10} in last two decades such as Bangladesh (31.4%), Kenya (26.3%), Nepal (24.6%), Senegal (14.1%) and Cambodia (13.3%) (Pandey et al. 2006).

Mongolia is identified as a most polluted country with respect to PM_{10}. Middle East countries like United Arab Emirates, Pakistan and Saudi Arabia have higher levels of particulate pollution. Higher PM values in these regions are mostly due to desert dust events (Jugder et al. 2011;

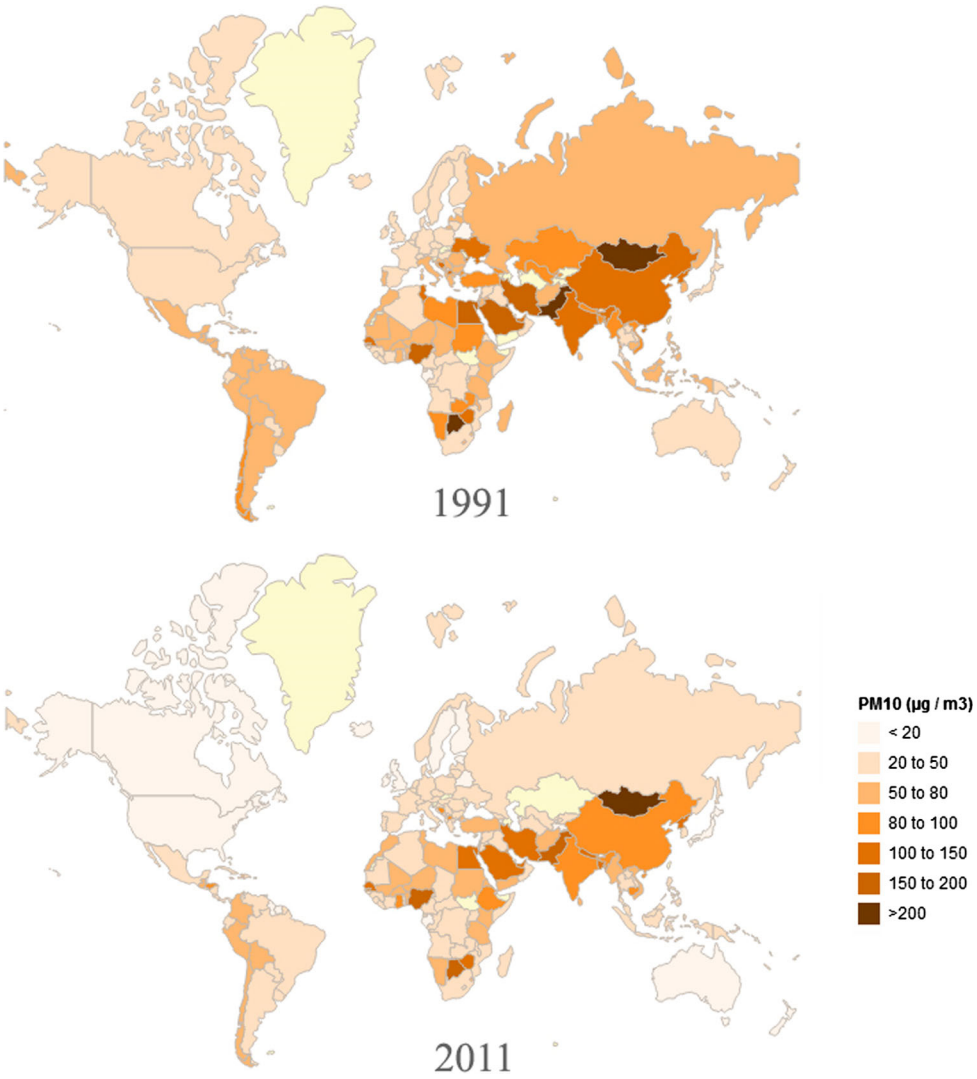

Fig. 1 Global PM_{10} concentrations (µg m^{-3}) in 1991 and 2011. PM_{10}: particulate matter having particle sizes 10 micrometers or less in diameter *Data source* Pandey et al. (2006)

Shahsavani et al. 2012; Wang et al. 2008). Jugder et al. (2011) found hourly maximum PM_{10} level up to 6626 µg m^{-3} during dust storm in Dalanzadgad, Mongolia.

Global PM_{10} levels and exceedances

Asia

PM_{10} status in India The National Ambient Air Monitoring Programme (NAMP) launched by Government of India in 1984 presently monitors air quality in relation to criteria air pollutants at 523 manual monitoring stations located in 215 cities/towns and industrial areas throughout the country (CPCB 2013). Annual average PM_{10} levels of major metropolitan cities during 2011 were analyzed to get an estimation of annual average PM_{10} levels in major metro cities in India mostly the state capitals and fast-growing cities with over one million population (Fig. 2). Delhi, the capital of India is found to be most polluted city with respect to PM_{10} where the level was 11 times higher than the WHO annual mean standard of 20 µg m^{-3} (WHO 2005). WHO study also identified Delhi among the most polluted city in the world (WHO 2016). All major cities in Indo-Gangetic plain (IGP) showed higher PM_{10} levels, and this condition is mostly attributed to high traffic, unplanned urban development, poor maintenance of road and vehicles, meteorological and topographical conditions (Kulshrestha et al. 2009; Sharma et al. 2014b). Among the metropolitan cities, 79% cities exceeded the NAAQS with respect to PM_{10} (CPCB 2013). Both northern and western parts of the country are affected by dust storms, which also enhance the levels of PM_{10} in summer season. Only southern and northeastern cities of India showed relatively lower levels of PM_{10} compared to other parts of the country. Few cities like Kochi and Thiruvananthapuram showed PM_{10} levels below the NAAQS standard of 60 µg m^{-3} (CPCB 2009).

Many cities in India are facing high particulate pollution with values several times higher than both national and

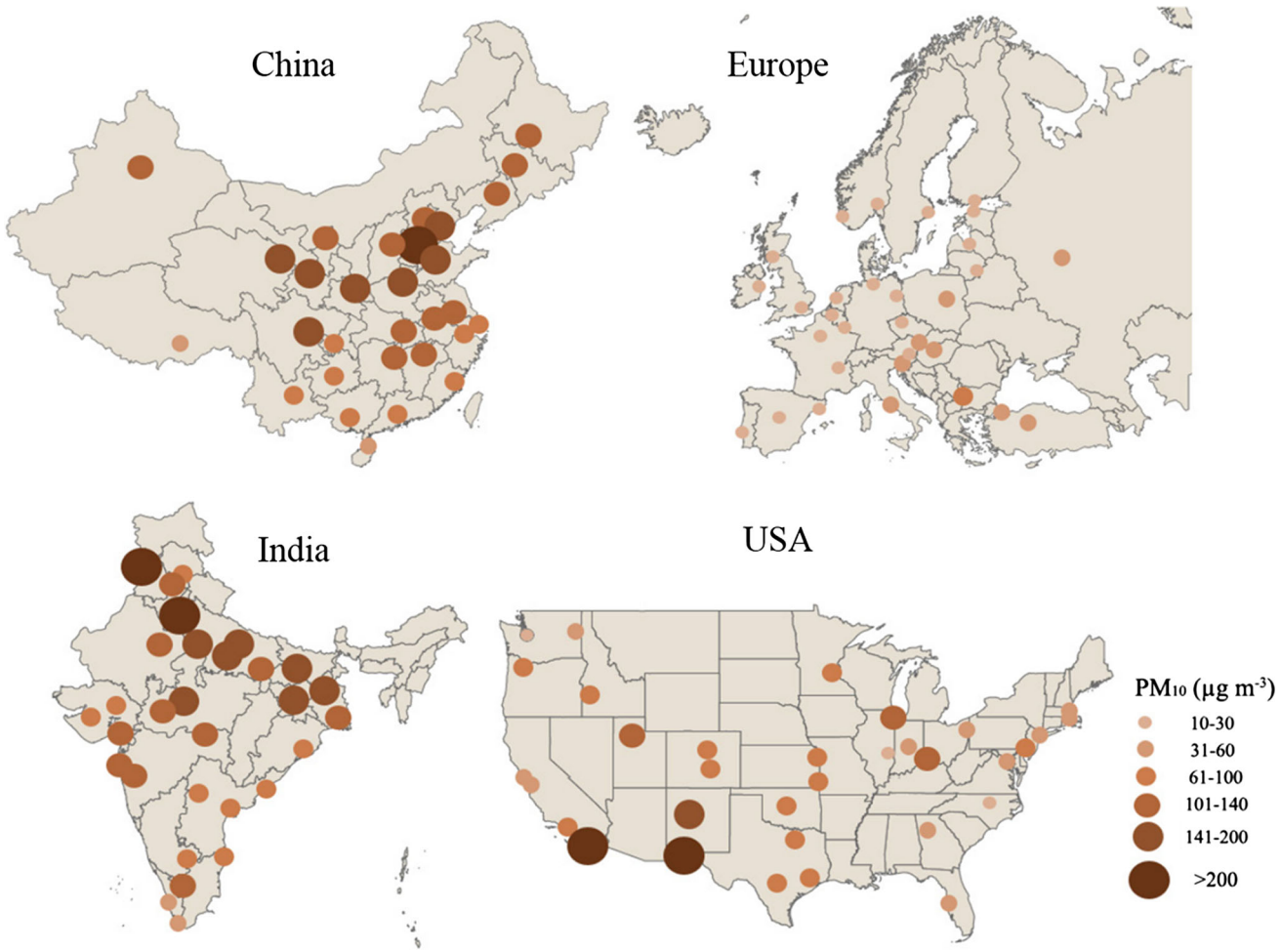

Fig. 2 Mean annual PM_{10} concentrations for year 2013 in China and USA and for 2011 for India and Europe. *Data source* CPCB (2013), CNEM (2013), EEA (2013), US EPA (2015), WHO (2014)

international standards. In semiarid part of India, Kulshrestha et al. (2009) found PM_{10} levels that exceeded 2.5 times the values (60 μg m^{-3} annual average) of National Ambient Air Quality Standard (NAAQS) as specified by Central Pollution Control Board, India (CPCB 2009), and 7.5 times higher than the WHO standard of 20 μg m^{-3} (WHO 2005). Similarly in Raipur, the capital city of Chhattisgarh State, in India, a highly urbanized commercial area with million plus population showed four times higher annual PM_{10} level than the Indian NAAQS standard and 13.5 times the WHO standard along with 100% exceedance in PM_{10} level during the entire monitoring campaign (Deshmukh et al. 2013). In Ahmadabad, an urban location in Western India, annual mean particulate matter value was 2 times higher than the Indian NAAQS and ~7 times higher than the annual mean WHO standard (Sudheer and Rengarajan 2012) (Fig. 3).

Gargava and Rajagopalan (2015) studied spatial variations of PM_{10} in six major cities of India during 2007–2010 and found higher PM_{10} concentrations in North India compared to South India. Mean PM_{10} concentrations during the entire study period at residential, industrial and kerb sites were 98, 137 and 164 μg m^{-3} for Bangalore, 123, 142 and 170 μg m^{-3} for Chennai, 419, 519 and 576 μg m^{-3} for Delhi, 213, 385 and 275 μg $m^{-3}$for Kanpur, 207, 196 and 205 μg m^{-3} for Mumbai and 132, 136 and 195 μg m^{-3} for Pune, respectively. Sharma et al. (2016) evaluated PM_{10} levels in IGP and observed higher mass concentrations of PM_{10} in the middle IGP (Varanasi 206.2 μg m^{-3}) as compared to upper IGP (Delhi 202.3 μg m^{-3}) and lower IGP (Kolkata 171.5 μg m^{-3}). All the values were above the standard of WHO annual mean and NAAQS of India.

Delhi the capital of India and the center of India's urban development is the most studied city related to air pollution in the world (Pandey et al. 2005; Sharma et al. 2014b; Tiwari et al. 2010, 2013, 2015). Most of the studies identified critical condition of air quality in Delhi mostly due to particulate pollution. Tiwari et al. (2015) found that annual mean level of PM_{10} was more than 3.8 times higher than the standard set by NAAQS India and 11.6 times above the

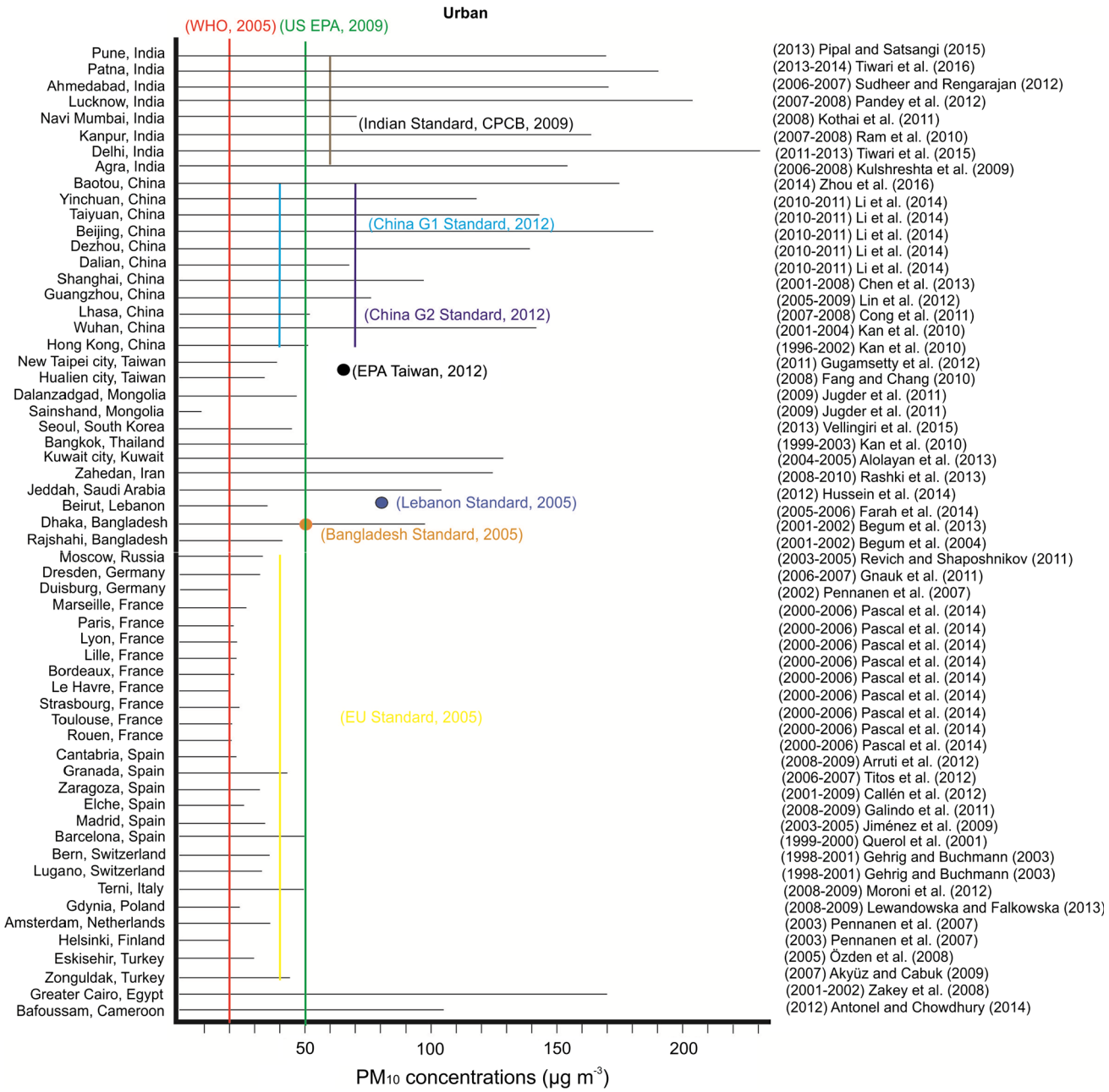

Fig. 3 Variations in PM_{10} concentrations at different urban locations throughout the world. PM_{10}: particulate matter having particle sizes 10 micrometers or less in diameter

annual mean WHO standard. Around heavy traffic sites in Delhi, India, average 24-h PM_{10} concentration in summer was 283.8 μg m^{-3} with a maximum concentration of 592.1 μg m^{-3}, while in winter, the value was 303.9 μg m^{-3} with a maximum concentration of 700.2 μg m^{-3}. These seasonal mean values were 4.7 and 5 times higher than annual mean NAAQS standard of India (Gupta et al. 2017). Such high levels of particulate matter pollution in Delhi as well as in other major cities in India are correlated with rapid increases in motor vehicle population. The numbers of motor vehicles registered in India from 2001 to 2011 have increased up to 158% (Statistical Year Book India 2013). Health situation has worsened in Delhi as 30% of the population suffers from respiratory disorders, which is more than the national average of India (Pandey et al. 2005).

High concentrations of PM_{10} ranging between 42 and 312 μg m^{-3} were reported in urban atmosphere of Kanpur, India, largely due to biomass burning, local emissions and long-range transport (Ram et al. 2010). Kolkata, the third-most populated metropolitan area in India also showed high PM_{10} episodes all over the year with 85 and 70% exceedance of PM_{10} values above NAAQS standards in residential and industrial

areas, respectively (Karar and Gupta 2006). Kothai et al. (2011) observed higher coarse particulate matter concentrations up to 140 μg m^{-3} in the residential area of Navi Mumbai, India, near an industrial site. Apart from urban area, higher levels of PM_{10} are also reported from urban background and even rural areas of India (Fig. 4). Sharma and Maloo (2005) reported mean concentration of 80 μg m^{-3} PM_{10} in urban background site in Kanpur City. Annual mean PM_{10} concentration of 148.4 μg m^{-3} with values ranging between 29.8 and 293 μg m^{-3} was reported from a rural site in Agra by Kulshrestha et al. (2009). These higher values are mostly due to agricultural activities and windblown dust transport.

Panicker et al. (2015) reported PM_{10} concentrations varying from 48 to 149 μg m^{-3} during different months in central Indian city of Jabalpur. The diurnal pattern of PM_{10} in this study showed a bimodal peak with morning peak around 0800–1000 hours and evening peak around 1800–2100 hours, indicating the influence of traffic sources to higher PM_{10} pollution in the city. In Rajnandgaon district, central India, Ambade (2016) evaluated seasonal variations in PM_{10} mass concentrations at urban and rural area and reported higher values during winter season (167 and 153 μg m^{-3}) and lowest values during monsoon season (34 and 32 μg m^{-3}) at both sites. The study also

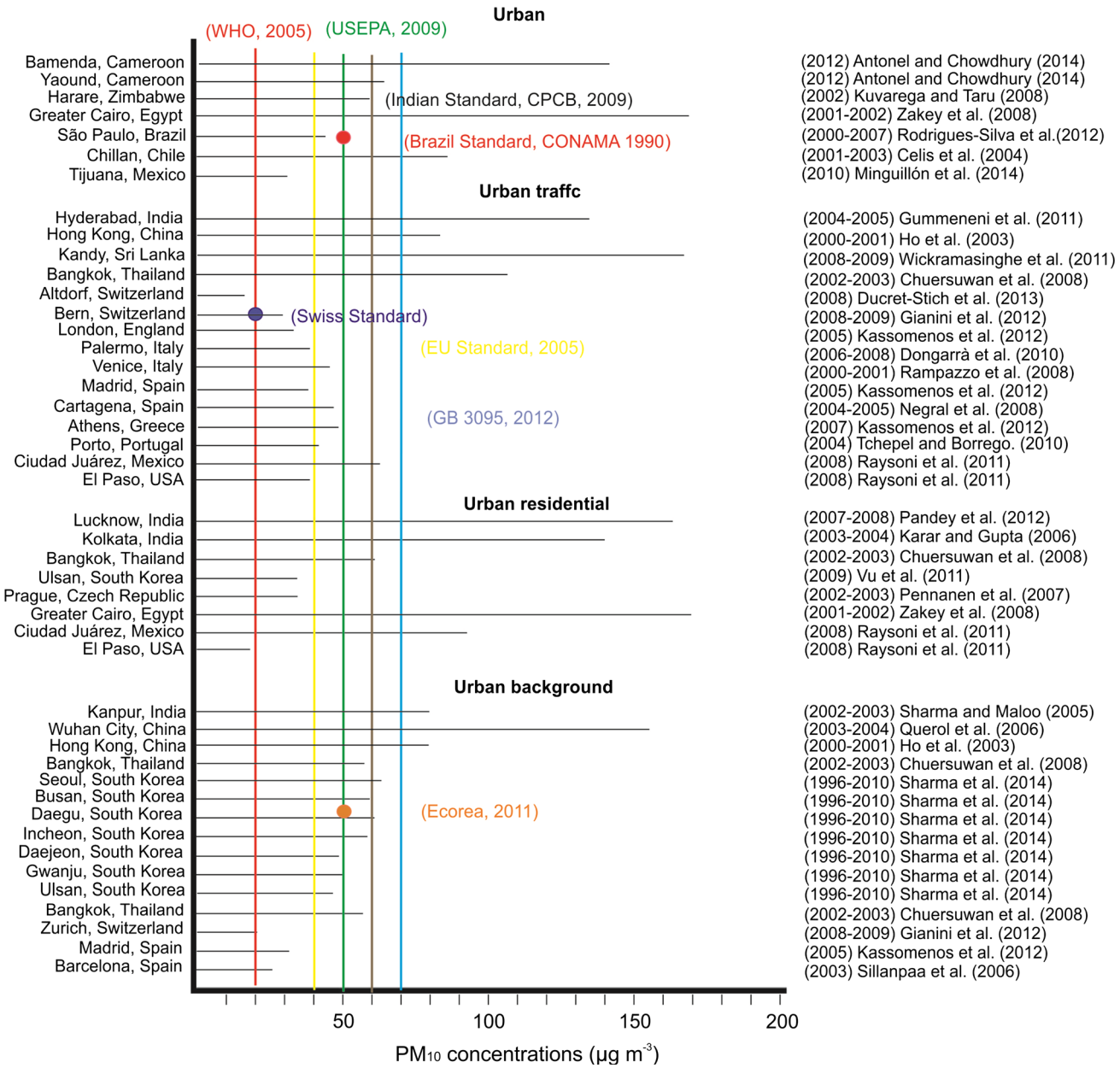

Fig. 4 Variations in PM_{10} concentrations at different urban, urban traffic, urban residential and urban background location throughout the world. PM_{10}: particulate matter having particle sizes 10 micrometers or less in diameter

showed significant particulate pollution at rural site with values above the NAAQS of India during most part of the study period.

PM_{10} status in China Data from China National Environmental Monitoring Center (CNEM 2013) is used to evaluate the spatial variability of PM_{10} concentrations in major cities of China. Overall data showed significant levels of PM_{10} pollution all over China (Fig. 2). Rapid growth and industrialization have resulted in significant changes in land-use pattern in past two decades, leading to higher particulate matter pollution, as a major cause of concern in China (Miao et al. 2013). The annual mean concentrations of PM_{10} in most of the cities exceeded the Chinese Ambient Air Quality Standards (CAAQS), Class I of 40 μg m^{-3} for PM_{10}. Some cities like Shijiazhuang, Jinan and Xian showed several folds higher concentrations compared to other cities, which exceeded the Class III standard (150 μg m^{-3}). Most of the big cities in Northern China like Zhengzhou, Tianjin and Beijing showed year-long high PM_{10} levels, which is also contributed due to severe dust events in that region. Apart from Wuhan and Nanjing in southeast showing frequent PM_{10} levels above 100 μg m^{-3}, other cities are relatively less polluted compared to north. Xian, Chengdu and Lanzhou in the west also showed values above the Chinese Ambient Air Quality Class II standard. Only few cities showed PM_{10} levels below Class II standard such as Haikou in southern and Lhasa in western region. All the major cities in China showed several folds PM_{10} level above the WHO standard.

PM_{10}-exceeding incidences were 30% in Beijing, 23.8% in Chengdu, 12.5% in Shanghai and 7.8% in Guangzhou from 2005 to 2009 (Lin et al. 2012). Li et al. (2012) observed that all the sites in Tianjin, China, exceeded the Class III of Chinese PM_{10} standard (150 μg m^{-3}) with values ranging between 53 and 1024 μg m^{-3}. Monitoring results from 18 sampling sites covering urban, rural village and rural field in Northern China showed that over 40% of the total measurements covering both urban and rural sites exceeded NAAQS with annual mean concentrations of 180,182 and 128 μg m^{-3}, respectively (Li et al. 2014) (Fig. 5). Higher biomass burning and agricultural waste combustion in rural areas result in significant increases in particulate matter pollution in rural areas of China (Li et al. 2014). In industrialized city of Wuhan, China both urban and industrial sites showed exceedances of PM_{10} above Chinese Class III standard (GB 3095 2012) with values ranging between 67 and 413 μg m^{-3} in industrial and 46–379 μg m^{-3} in an urban area (Querol et al. 2006). PM_{10} levels ranged between 12 and 643 μg m^{-3} with mean concentration of 97 μg m^{-3} in an urban area of Shanghai from 2001 to 2008, which was more than Class I and Class II Chinese standards (Chen et al. 2013). Study of roadside ambient air quality by Shen et al. (2010) detected very high concentrations of PM_{10} ranging from 337.9 to 718.0 μg m^{-3} during heavy traffic periods in Xian, China.

Using satellite remote sensing data, Li et al. (2015) reported a decreasing trend of PM_{10} by 0.15 ± 0.23 μg m^{-3} $year^{-1}$ in Pearl River Delta Region from 2001 to 2013. The mean PM_{10} concentration for this period was 56.8 μg m^{-3}, which was 1.4 times higher than annual Chinese Class I standard. Zhongshan, Dongguan and Foshan were identified as the most severely affected areas with PM_{10} values above 70 μg m^{-3} (Li et al. 2015). Xie et al. (2015) assessed PM_{10} status in 31 Chinese provincial capital cities from 2013 to 2014 and found higher PM_{10} concentrations in most part of China. Among the cities Shijiazhuang and Xi'an showed monthly mean PM_{10} concentrations above 300 μg m^{-3}. The values were 15 and 2 times above the annual WHO and Chinese Class III standard, respectively. Lowest concentration was found in Haikou with annual average of 46 μg m^{-3} (Xie et al. 2015). In 11 largest cities of Gansu Province in China, hourly mass PM_{10} concentrations varied from 50 to 70 μg m^{-3} with mean value of 66 μg m^{-3} during the study period from June to August 2015 (Filonchyk et al. 2016).

Zhang et al. (2016) evaluated national-scale PM_{10} concentrations using a satellite-based geographically weighted regression model in China and found PM_{10} mass concentrations varying from 7.67 to 238 μg m^{-3}, with annual mean value of 83.24 μg m^{-3}, which was 4.1 and 2 times higher than annual mean WHO and Chinese Grade I standard, respectively. In Beijing-Tianjin region, annual mean value was above 160 μg m^{-3}. Hainan, Tibet, Yunnan and Heilongjiang were identified as least polluted regions with values below 50 μg m^{-3} (Zhang et al. 2016).

Song et al. (2016) reported PM_{10} concentrations varying from 264.7 to 1066.0 μg m^{-3}, with a mean value of 572.0 μg m^{-3}, which was 3.8 times higher than the annual average PM_{10} mass limit (150 μg/m3) of Class III standard of China during winter season of 2013 in a typical industrial city of Pingdingshan in North China. PM_{10} concentrations varied from 17.2 to 681 μg m^{-3}, with a mean value of 176 μg m^{-3} in six urban sites covering 3 districts in Baotou, China (Zhou et al. 2016). Mean value was 1.17 times higher than Chinese Class III and 8.7 times higher than annual WHO standard.

Other Asian countries In Zahedan, Iran, Rashki et al. (2013) reported that for 90.5% of the days, PM_{10} levels were above the daily EU threshold value of 50 μg m^{-3}. Alolayan et al. (2013) also reported exceedance of PM_{10} level above the WHO daily guidelines on 91% of sampling dates in Kuwait City. In Jeddah, Saudi Arabia, PM_{10} concentrations varied between 17.5 and 1400 μg m^{-3}, with 29 days in the year 2012, the values of PM_{10} exceeded the level of 200 μg m^{-3} (Hussein et al. 2014). PM_{10}

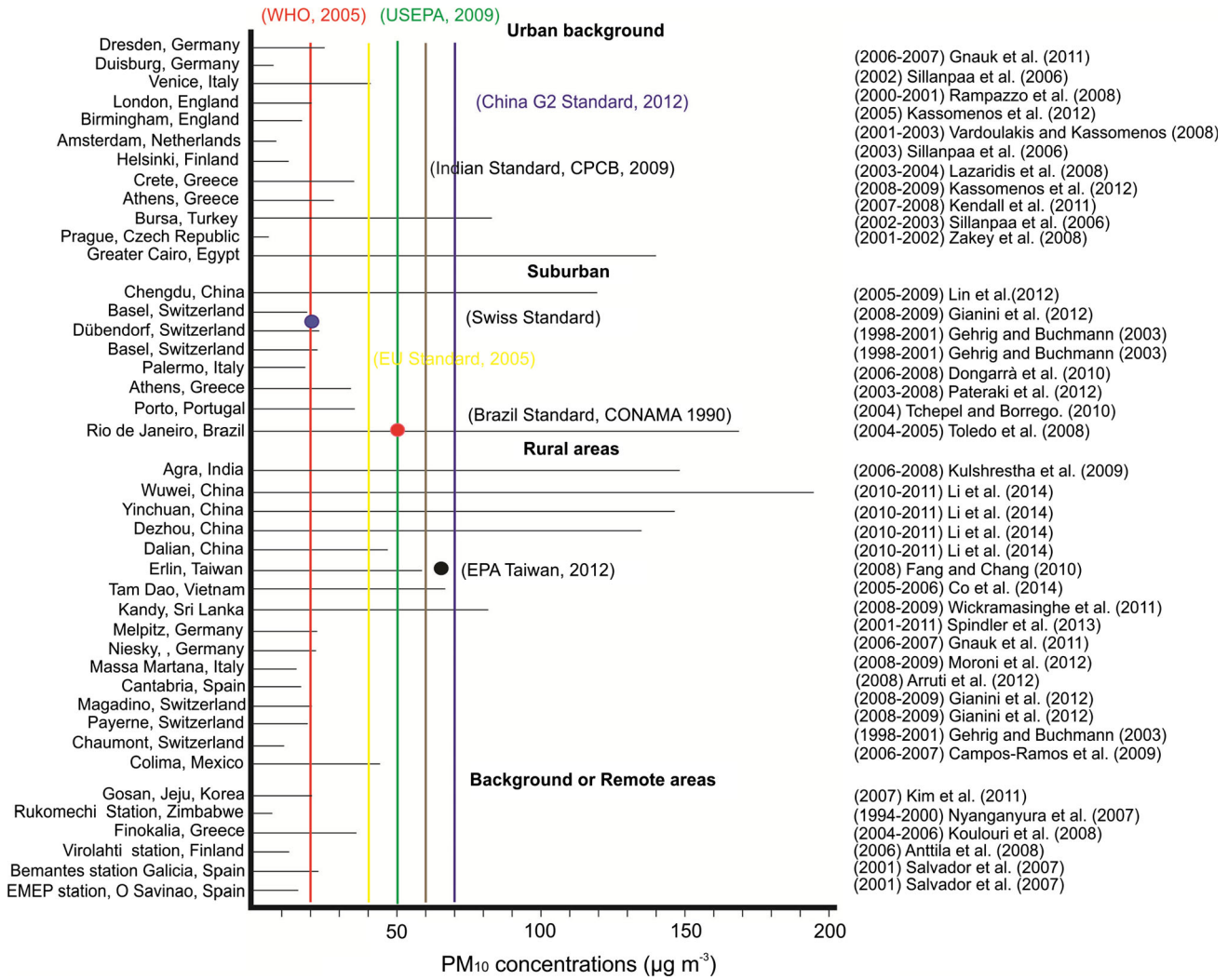

Fig. 5 Variations in PM_{10} concentrations at different urban background, suburban, rural and remote location throughout the world

concentrations ranged between 10.1 and 491 μg m^{-3} with mean value of 97.7 μg m^{-3} in Dhaka, Bangladesh, from 1996 to 2011. These values were exceeded two times from the annual Bangladesh standard and five times of the WHO standard (Begum et al. 2013). The annual mean PM_{10} levels in Bangkok, Thailand, from 1999 to 2003 were 52 μg m^{-3}, which is 2.6 times of WHO standard (Kan et al. 2010). PM_{10} concentrations monitored in seven major cities of Korea from 1996 to 2010 revealed significant decreases in PM_{10} concentration in most of the cities after 2000, although values were still above WHO level and within or just above the Korean Ministry of Environment (KMOE) standard of 50 μg m^{-3} (Sharma et al. 2014a) (Fig. 3).

Europe

Based on 2011 data from European Environment Agency Air Base public air quality database (EEA 2013), PM_{10} levels have been assessed in different major European cities (Fig. 2). Compared to Asian and American cities, Europe seems to have much lower levels of PM_{10}. Most of the European cities were under or just over the EU, PM_{10} standard of 40 μg m^{-3}. Only few cities showed excessive PM_{10} levels as Sofia, the capital and largest city of Bulgaria, which showed 1.5 times the annual EU limit, and Ankara, the second largest city of Turkey. Fuel combustion and vehicular emission sources are major reasons for particulate pollution in Europe (Sillanpaa et al. 2006). Some cities like Helsinki (Finland), Tallinn (Estonia), Stavanger (Norway), Stockholm (Sweden) and Luxembourg have PM_{10} levels below the WHO standard.

In urban areas of Belgrade, Serbia, PM_{10} ranged between 2.8 and 333.8 μg m^{-3} with mean concentration of 68 μg m^{-3}, which is 3 times higher than the WHO and 1.7 times of EU annual limit of 40 μg m^{-3} (Rajšić et al. 2008). In both urban and rural areas in Cantabria region of

northern Spain, PM_{10} levels were below EU regulation for both annual and 24-h limits (Arruti et al. 2012). Negral et al. (2008) found that 36% of days, PM_{10} levels were higher than 50 μg m^{-3} in the historical city of Cartagena, Spain, and ascribed natural (dust transport) and anthropogenic sources for this pollution level in the city. In Athens, Greece, PM_{10} concentrations ranged from 25 to 208 μg m^{-3} near busy roadways which clearly indicated the contribution by diesel vehicles to particulate matter concentrations (Chaloulakou et al. 2005). Pennanen et al. (2007) found PM_{10} levels below the EU standard in six European urban areas, but compared to annual PM_{10} of reference year 2001, the concentrations were 1.2–1.6 times higher in all the cities except Duisburg, Germany. In Barcelona Metropolitan area, the annual mean PM_{10} levels as well as 24-h mean values exceeded the EU standard with 86 daily values exceeding 50 μg m^{-3} concentration during 1999–2000 (Querol et al. 2001). In urban area of Zaragoza, Spain, although mean annual PM_{10} levels were below the EU standard, significant exceedances were recorded in summer and winter (69%) due to long-range transport (Callén et al. 2012). Similarly at an urban site in Granada, Spain, PM_{10} levels were slightly higher than the annual mean value limit (40 μg m^{-3}) of European Directives, but daily levels were exceeded 20 times mainly under African dust events (Titos et al. 2012) (Figs. 3, 4).

PM_{10} monitoring data for year 2015 from 81 cities in Turkey showed significant exceedances of PM_{10} levels from national and international standards. Among 81 cities monitored for PM_{10}, 38 cities showed PM_{10} levels above national standard of 56 μg m^{-3}, whereas values were above EU standard of 40 μg m^{-3} in 62 cities, showing critical condition of particulate matter pollution in Turkey (Güler and Güneri İşçi 2016). Özden et al. (2008) observed significant decrease in particulate matter concentration from 1995 to 2005 in Eskişehir, Turkey, after the start of using natural gas in industry and cooking. In urban area of Gdynia, Poland, PM_{10} levels ranged between 5.4 and 117.1 μg m^{-3}, which exceeded 25 times the daily limit of the European Parliament's CAFÉ, during 2008–2009 (Lewandowska and Falkowska 2013). In Moscow, Russia, PM_{10} levels ranged from 9 to 164 μg m^{-3} with an annual mean of 34 μg m^{-3} well below EU standard (Revich and Shaposhnikov 2010). Kendall et al. (2011) reported PM_{10} levels to be double the EU annual standard of 40 μg m^{-3} in urban background site in Bursa, Turkey. In background or remote areas, PM_{10} concentrations were mostly below the WHO standard (Fig. 5). In four different cities (Limassol, Nicosia, Larnaca and Paphos) of Cyprus, PM_{10} levels were mostly below the WHO 24-h mean value of 50 μg m^{-3}, but overall annual mean value among all the four cities was 1.5 times of the WHO annual mean value of 20 μg m^{-3} (Achilleos et al. 2016).

North America

PM_{10} status in USA United States Environmental Protection Agency (US EPA) monitored air quality around 646 stations in 261 cities in USA. According to US EPA dataset for PM_{10} (second highest 24-h average measurement in the year) for year 2013, most of the Core Based Statistical Area (CBSA) showed PM_{10} levels above the WHO and EPA's health-based national air quality annual standard of 50 μg m^{-3} for PM_{10} (US EPA 2014). Higher PM_{10} concentrations were in the regions of Albuquerque, El Paso, San Diego, Yuma, Chicago and Phoenix (Fig. 2). A 34% decrease in national average PM_{10} concentration has been observed from 1990 to 2013 based on the measurements of 207 sites, whereas monitoring results from 449 sites from 2000 to 2013 showed 30% decrease in national PM_{10} average value (US EPA 2014).

Raysoni et al. (2011) analyzed coarse particulate levels outside school in US-Mexican border and found higher concentrations at school in Ciudad Juárez, Mexico, compared to El Paso, USA. Within El Paso school, twofold higher PM_{10} level was reported near high traffic site compared to school at low traffic site. In rural area of Western Mexico, Campos-Ramos et al. (2009) observed daily average concentration of 44 μg m^{-3} PM_{10} between 2006 and 2007, which was below the WHO permissible daily limit of 50 μg m^{-3} (WHO 2005). Qin et al. (2004) studied weekend/weekday differences of different air pollutants in Southern California from 1995 to 2001. In this study, PM_{10} showed a sharp reduction of 7–32 and 8–28%, respectively, during morning and afternoon hours in weekdays compared to weekend at different sites. In general, PM_{10} levels were higher in weekdays and in early morning hour and evening, indicating that the traffic sources are major contributors to the high particulate matter levels in the cities. Sevimoglu and Rogge (2016) compared PM_{10} concentrations at rural and costal urban area in Southeastern Florida, USA, and found annual mean concentrations of 26.6 and 24.1 μg m^{-3} at rural and urban area, respectively, which were below the US EPA standard.

Annual mean PM_{10} concentrations at different sites in Costa Rica metropolitan area showed maximum concentrations at high traffic commercial area (55 μg m^{-3}) followed by industrial area (52 μg m^{-3}), commercial area (37 μg m^{-3}) and least at residential area (25 μg m^{-3}) (Murillo et al. 2013). Concentrations were well below the Costa Rican standards of 50 μg m^{-3} of PM_{10} at commercial and residential areas (Murillo et al. 2013). Cheng et al. (2000) estimated PM_{10} levels in different sites in Alberta, Canada, with different land uses and found higher average concentrations at rural industrial sites (34.6 μg m^{-3}) followed by rural influenced sites (16.8 μg m^{-3}) and minimum in rural remote sites (8.8 μg m^{-3}). In Pinal County

Arizona, USA, coarse PM concentrations were higher at rural sites (6.3–177.6 μg m^{-3}) with mean value of 66.6 μg m^{-3} compared to urban sites (5.75–78 μg m^{-3}) with mean value of 30.6 μg m^{-3} (Clements et al. 2014), indicating that agricultural activities may also increase local PM_{10} concentrations even in the absence of any industrial or heavy traffic activities (Fig. 5).

Africa

Antonel and Chowdhury (2014) reported PM_{10} levels in three different cities in Cameroon in West Central Africa where highest concentration of PM_{10} was found in Bamenda followed by Bafoussam and Yaound. In Bafoussam City, outskirts were severely affected with high PM_{10} (224 μg m^{-3}), whereas in Bamenda and Yaound market places showed higher PM_{10} levels as 327 and 127 μg m^{-3}, respectively. Levels of PM_{10} in both residential and road sites monitoring sites in Accra, Ghana, were above the WHO limits with annual values ranging from 80 to 108 μg m^{-3} at road site and 57–106 μg m^{-3} at residential site (Dionisio et al. 2010). In a rural (Morogoro) and an urban kerbside site (Dar es Salaam) in Tanzania, mean PM_{10} levels were 27 and 51 μg m^{-3}, respectively (Mkoma et al. 2009). At Rukomechi Research Station, Zimbabwe, a background site in central part of Southern Africa, Nyanganyura et al. (2007) found mean coarse particle concentrations of 7.4 μg m^{-3} during the study period from 1994 to 2000. Authors also found fine PM concentrations of 8.8 μg m^{-3} which was somewhat higher than coarse fraction, but these values were mostly regulated by seasonal effects. In Constantine, Algeria, Terrouche et al. (2016) estimated PM_{10} levels around highway and found levels ranging from 14.5 to 161.8 μg m^{-3} with average concentrations of 80.42 μg m^{-3}. The values were fourfold and twofold higher than the WHO and the EU standards whereas almost equal to Algerian standards of 80 μg m^{-3}. Further, the values exceeded the EU standard in 73% of sampling period (Terrouche et al. 2016). In Durban, South Africa, PM_{10} levels ranged from 24.9 to 99.4 μg m^{-3} with mean concentrations of 57.7 μg m^{-3} (Batterman et al. 2011). Higher PM_{10} levels in African cities are mostly as a result of biomass burning, dusty roads, desert dust and higher population density (Dionisio et al. 2010; Antonel and Chowdhury 2014) (Figs. 3, 4, 5).

South America

In São Paulo, Brazil, PM_{10} levels ranged between 37.22 and 50.47 μg m^{-3} with mean value of 44.49 μg m^{-3}, which was below the Brazilian standard of 50 μg m^{-3}, but 2.5 times higher than the WHO annual mean level (Rodrigues-Silva et al. 2012). In suburban region of Rio de Janeiro, Brazil, PM_{10} levels ranged from 71 to 312 μg m^{-3} with mean value of 169 μg m^{-3}, which was approximately 3 times higher than the Brazilian standard (Toledo et al. 2008). Souza et al. (2014) compared urban PM_{10} concentrations between São Paulo and Piracicaba, Brazil, and found higher average concentration at São Paulo (64 μg m^{-3}) compared to Piracicaba (35 μg m^{-3}). Higher concentrations at São Paulo were attributed due to higher industrial activities in this area. Daily PM_{10} average concentrations ranged from ~45 to ~115 μg m^{-3} during highly polluted winter months from 2002 to 2012 in Santiago, Chile (Ragsdale et al. 2013). Jorquera and Barraza (2012) observed PM_{10} levels ranging between 80 and 331 μg m^{-3} with mean value of 161 μg m^{-3}, and the values were well above the national and the WHO standards in an arid industrial region of Antofagasta, a midsize coastal city in Northern Chile. PM_{10} concentrations ranged between 11.1–110 and 8.3–116 μg m^{-3}, respectively, at two rural sites Quillota and Linares in Central Chile (Hedberg et al. 2005). In coastal city of Northern Chile, Tocopilla, PM_{10} concentrations ranged from 48 to 194 μg m^{-3} with mean value of ~90.5 μg m^{-3}, which is almost double the ambient air quality standards for PM_{10} in Chile (Jorquera 2009). PM_{10} levels ranged from 6 to 146 μg m^{-3} with annual mean concentration of 35 μg m^{-3} at different sites in urban area of Buenos Aires, Argentina. The values were above the EU standard for 36 times during the entire study period of 2006–2007 (Arkouli et al. 2010). At two residential areas in Bogota, capital of Colombia, Vargas et al. (2012) found PM_{10} levels as high as 94 μg m^{-3} with mean concentrations of 41 and 52 μg m^{-3} at sites Suba and Kennedy. Average PM_{10} concentrations were almost similar at urban and semi-urban sites in Córdoba City, Argentina, with respective values of 107 and 101 μg m^{-3} and exceedance of WHO standard by fivefold (López et al. 2011).

Australia and New Zealand

Kamruzzaman et al. (2015) studied dispersion pattern of PM_{10} in the city of Adelaide, Australia, from 2006 to 2012 and observed annual mean PM_{10} concentration of 20.139, 18.460, 16.215 and 14.840 μg m^{-3} around busy urban traffic area, residential/light industry area, peri-urban area and residential area respectively. All the values were below the Australian 24-h standard of 50 μg m^{-3}. Chan et al. (2008) compared PM_{10} values around urban and suburban area in four Australian (Melbourne, Sydney, Brisbane and Adelaide) cities during 2003–2004. Annual mean values for urban and suburban areas were 8.84 and 10.37 μg m^{-3} in Melbourne, 11.35 and 9.04 μg m^{-3} in Sydney, 8.21 and 7.15 μg m^{-3} in Brisbane and 12.98 and 12.77 μg m^{-3} in Adelaide. All these values were below the

annual mean WHO standard of 20 μg m^{-3}. A time-series analysis by Roberts (2013) from 1993 to 2007 to assess health impact assessment of PM_{10} found no significant decline in PM_{10} concentrations in Brisbane, Melbourne and Sydney.

Johnston et al. (2011) in Sydney, Australia, compared PM_{10} levels during normal days and dusty days and found mean PM_{10} concentration around 96.8 μg m^{-3} in dusty days which was 1.9 times higher than Australia's 24-h air quality standard whereas in normal days mean value was only 17.8 μg m^{-3}. In a mixed residential-industrial area in Rockela, Brisbane, mean $PM_{2.7–10}$ concentration was 15.8 μg m^{-3}, which was below the both annual WHO and Australia's air quality standards (Chan et al. 2000).

Ancelet et al. (2013) reported PM_{10} concentration of 21.0 μg m^{-3} in the winter season at a rural site in Masterton, New Zealand. The study also found substantial contribution of fine PM (64%) in PM_{10}. Mean PM_{10} concentrations in summer and winter were, respectively, 15.5 and 81.1 μg m^{-3} in Christchurch, 21.8 and 24.4 μg m^{-3} in Dunedin and 10.10 and 59.5 μg m^{-3} in Alexandra, New Zealand, during 2001–2002 (Brown et al. 2005). The higher values in winter were mostly due to higher emissions and meteorological reason. At urban background site in Auckland, New Zealand, PM_{10} concentration varied from 1.90 to 27.9 μg m^{-3} with mean value of 9.9 μg m^{-3} which was below the WHO annual standard (Wang and Shooter 2005).

At a suburban location in Nelson, New Zealand, PM_{10} concentrations varied from 2.0 to 57.0 μg m^{-3} with mean value of 21.0 μg m^{-3} which was marginally above the WHO annual standard (Ancelet et al. 2014). Wilson et al. (2006) monitored PM_{10} levels from ten background monitoring sites during July 2003–June 2004 around Christchurch, New Zealand, and found PM_{10} values ranging between 1.9 and 171.3 μg m^{-3} with mean value of 43.9 μg m^{-3}, which was 2.2 times the annual WHO standard of 20 μg m^{-3}.

PM_{10} source apportionment

Mass concentration of PM does not directly provide the nature of source or its toxicity potential in the environment. Increasing concentrations of PM require identification of the possible sources. Significant differences occur between sources, physical and chemical characteristics of PM_{10} in urban, industrial and rural areas around the world. PM concentration mostly depends upon local factors such as source strength and removal processes.

Gianini et al. (2012) estimated that about 30% of PM_{10} mass at the urban roadside site is generated by local road traffic emissions in Bern, Switzerland. Mineral matter (43%), secondary inorganic aerosol (SIA) (17%), organic matter and elemental carbon (31%) contribute to PM_{10} mass at urban location in Granada, Spain (Titos et al. 2012). Dongarrà et al. (2010) in Palermo, Italy, found that road traffic contributed almost 50% of PM. Callén et al. (2012) quantified that soil resuspension (66%) followed by industrial and traffic emissions (8%), coal combustion (3%), marine component (3%) and heavy-duty vehicles (1%) and unknown sources (19%) to be the contributors of PM_{10} in Zaragoza (Spain) from 2001 to 2009. In contrast, Moreno et al. (2006) in Puertollano, Spain, found PM_{10} to be rich in crustal components (30%). Major sources identified in coarse fractions were vehicular/road dust, secondary/long range, soil/road dust and metallurgy/chemical in Terni, Italy (Moroni et al. 2012). Road traffic specifically emissions from vehicles, suspension of dust due to vehicular movement, tire break wear add significantly to PM_{10} around traffic sites (Mansha et al. 2012; Rahman et al. 2011; Tiwari et al. 2010; Dongarrà et al. 2010).

A monitoring study of urban, rural background site and kerbside in Dresden, Germany, found local sources like crust material, windblown dust, road works and coal burning as the major factors contributing to PM_{10} and its associated components (Gnauk et al. 2011). Diesel exhaust alone contributed to approximately 50% mass to PM_{10} in Athens, Greece (Chaloulakou et al. 2005). In urban units of Rotterdam, Netherlands, primary sources of PM were road traffic emission, resuspension of road dust, tire/engine wear, biomass burning, burning of fuel, construction activities and unpaved road (Keuken et al. 2011). All the sources identified in this study were related to combustion and mechanical processes. Resuspension of soil (78%) along with industrial and traffic emissions (20%) were the major contributors of PM_{10} levels in Zaragoza, Spain (Callén et al. 2012). Alleman et al. (2010) reported higher contribution of road transport (15%) followed by dust resuspension (13%), metallurgical coke plant (12.6%), marine aerosol (12%), crustal dust (11%) and petrochemistry activities (9.2%) to PM_{10} at an urban background station located 2 km away from industries in Dunkirk, France. In most of the rural or background areas of the world, there is a predominant increase in anthropogenic sources of PM emissions while previous reports identified natural sources only as the major contributing factors to PM_{10} (Quass et al. 2013; Hopke 2016).

Long-range transport of PM has significant effect on local air quality as high PM episodes in London were mostly originated from continental Europe (Kassomenos et al. 2012). Similarly, desert dust from North Africa or the Western Mediterranean increased coarse PM levels in Athens and long-range transport from North Africa along with transport from north of Spain and France affected the ambient PM levels in Madrid, Spain (Kassomenos et al.

2012). Atmospheric back trajectory modelling in three European capitals (London, Madrid and Athens) by Kassomenos et al. (2012) suggested significant contribution of long-range transport as a source of coarse particles, in addition to local sources.

Rahman et al. (2011) identified motor vehicle (42.4%), road dust (18.3%), industry (17.6%), two stroke engines (13.1%) and soil (8.5%) as major sources of fine particles in Kuala Lumpur, Malaysia. In urban area of Raipur, India, anthropogenic sources of PM were vehicular traffic, fossil fuel combustion and industrial activities (Deshmukh et al. 2013). In Bangkok metropolitan region, automobile and biomass burning were the most important sources followed by road dust (Chuersuwan et al. 2008). Kaku et al. (2016) found continental dust (35%), road dust (24%), marine dust (11%) and sea salt (12%) as major sources of PM_{10} compared to secondary aerosols (18%) at a costal site in Al Taweela, United Arab Emirates.

In rural village of China, Li et al. (2014) found that coal and biomass fuel combustions for heating and cooking contributed to the high levels of PM_{10}. Source identification at industrial site in Wuhan City, China, by Querol et al. (2006) reported contributions from cement manufacture/construction and demolition/artificial soil resuspension (34%), coal-fired power plant emissions (20%), anthropogenic regional background (16%), steel manufacturing (11%), road traffic (10%) and the metallurgy source (4%) to PM_{10} mass concentration. Major sources of PM_{10} identified in Wuhan, China, were coal consumption by industry and residents, automobiles, road dust and dust from city construction projects (Feng et al. 2011). Activity-based emissions inventory by sector contribution identified major sources as domestic cook stoves (39%), power plants (24%), heat-only boilers (19%), road dust (12%), vehicle exhaust (3.1%), kiosks (1.9%), brick kilns (1.6%) and open waste burning (0.4%) to PM_{10} in Ulaanbaatar, Mongolia (Guttikunda et al. 2013).

Source apportionment study conducted in six Indian cities during 2007–2010 indicated that the carbon, SIA and crustal components constitute ~40–65% of ambient PM_{10} (Gargava and Rajagopalan 2015). Karar and Gupta (2006) in Kolkata, India, observed that higher PM_{10} pollution was contributed due to resuspension of road dust, soil dust, automobiles, traffic and nearby industrial emissions in the industrial area and coal burning, construction activities and emissions from a solid waste dumping in the residential site. PCA revealed soil, sea salt and combustion as main sources for coarse particles at an urban site of Navi Mumbai, India (Kothai et al. 2011). Road dust due to vehicular activities, solid waste incineration and industrial emission were the major sources of PM_{10} at urban site, whereas soil dust due to vehicular emission, construction activities and windblown dust were common sources at rural site in Agra, India (Kulshrestha et al. 2009). Sudheer and Rengarajan (2012) at an urban location in Ahmadabad, India, found mineral dust (43%) to be a major contributor to PM_{10}, whereas resuspended dust (40%), vehicular pollution (22%) and combustion (12%) were the major sources of PM_{10} pollution around traffic site in Hyderabad, India (Gummeneni et al. 2011). Ancelet et al. (2015) found biomass burning as a major source of PM_{10} in Nelson, New Zealand (Fig. 6). In most of the studies conducted in urban centers of the world, predominant source of PM_{10} is traffic along with crustal sources, which is directly correlated with the increment in number of vehicles in the last decade (Chan et al. 2008; Gargava and Rajagopalan 2015).

Johnston et al. (2011) reported major contribution of crustal matter (24%), sea salt (16%), organics (9.1%) and soot (3.2%) in PM_{10} in Sydney, Australia. Vehicular dust was found to be the major contributor (37–64%) to PM_{10} mass in four Australian cities (Chan et al. 2008). Chan et al. (2000) found significant portion of crustal fraction and sea salt in PM_{10} in the coastal area of Brisbane, Australia. Similarly, Minguillón et al. (2014) also reported higher contribution of sea salt and mineral matter in urban areas of Tijuana, Mexico. In Córdoba City, Argentina, traffic was found to be a major contributor to PM_{10} mass along with urban dust (López et al. 2011). Ancelet et al. (2014) reported sea salt (49%) along with aged sea salt (17%), windblown soil (16%), nitrate (12%) and road dust (6%) as the major component of PM_{10} around suburb of Nelson, New Zealand.

The crustal matter (19%), coal combustion (31.6%), vehicle exhaust and abrasion (7.4%), local burning (6.3%), weathering of waste dumps (9.8%) and industrial metal smelting (25.9%) were identified as a major contributor to PM10 in industrial city of Pingdingshan, China (Song et al. 2016). Meta-analysis results from published record in Europe showed six major sources of PM as secondary inorganic aerosol, traffic, biomass burning, sea/road salt, resuspension of crustal/mineral dust and industrial point sources (Belis et al. 2013).

Different source apportionment studies revealed a strong contribution of crustal sources as mineral dust, road and soil dust followed by organic matter mostly emitted through biomass burning and traffic as the major sources of PM_{10}. In coastal areas, sea salt has a strong influence on local PM_{10} contribution. Apart from these sources, industrial, agricultural, long-range transport, dust storm and other anthropogenic sources strongly influence PM_{10} levels (Fig. 6).

PM_{10} source apportionment technology trend

To estimate the contribution of various sources to ambient PM concentrations, several statistical techniques were employed such as positive matrix factorization (PMF)

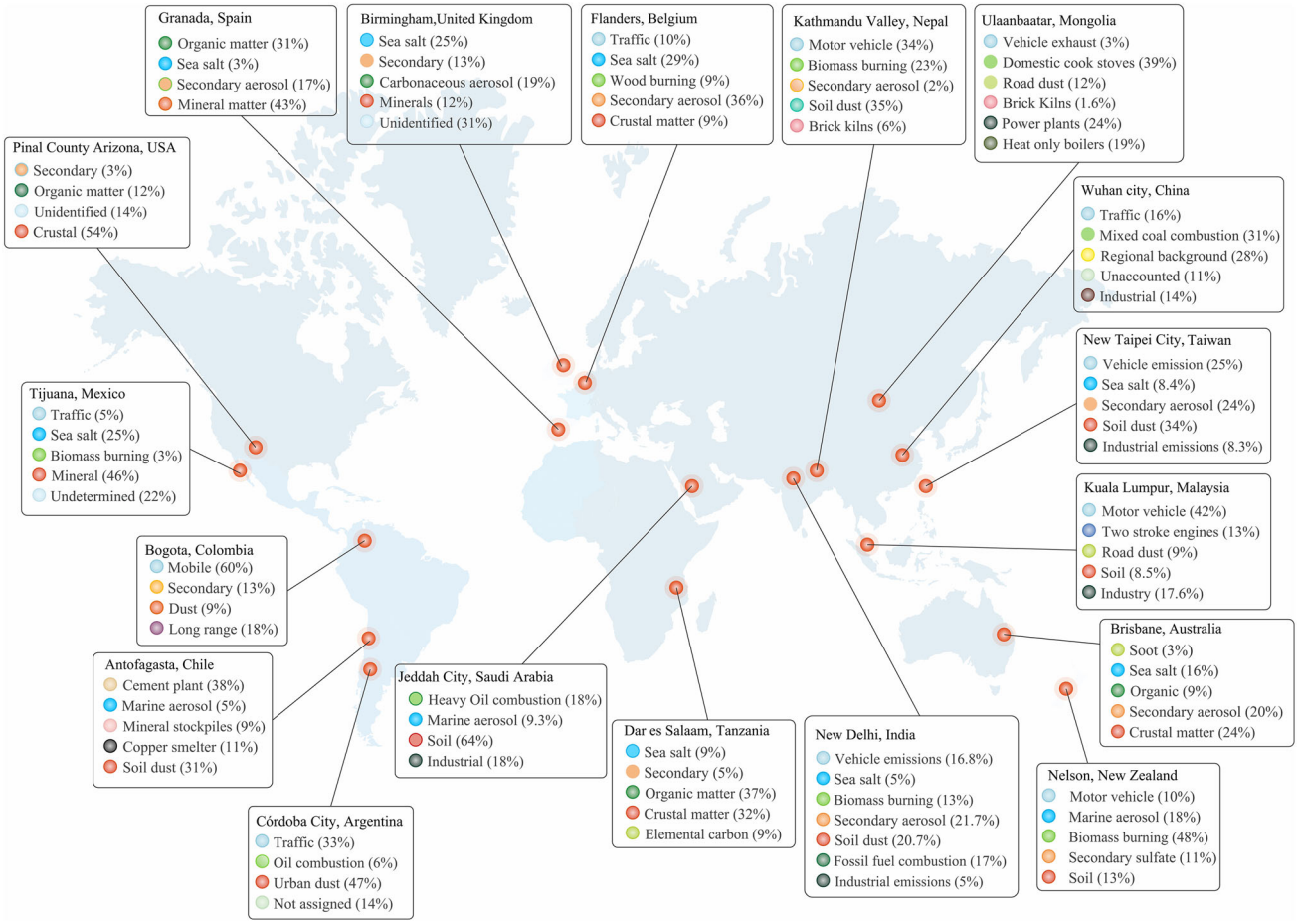

Fig. 6 Sources of PM_{10} in selected cities of the world. Source profiles are given in percentage with respect to different sources [Flanders, Belgium (Maenhaut et al. 2016); Nelson, New Zealand (Ancelet et al. 2015); Kathmandu Valley, Nepal (Kim et al. 2015); Bogota, Colombia (Vargas et al. 2012); Dar es Salaam, Tanzania (Mkoma et al. 2009); Córdoba City, Argentina (López et al. 2011); Antofagasta, Chile (Jorquera and Barraza 2012); Pinal County Arizona, USA (Clements et al. 2014); Wuhan City, China (Querol et al. 2006); Granada, Spain (Titos et al. 2012); Kuala Lumpur, Malaysia (Rahman et al. 2011); Ulaanbaatar, Mongolia (Guttikunda et al. 2013); New Delhi, India (Sharma et al. 2014b); New Taipei City, Taiwan (Gugamsetty et al. 2012); Brisbane, Australia (Chan et al. 2000); Tijuana, Mexico (Minguillón et al. 2014); Jeddah City, Saudi Arabia (Khodeir et al. 2012); Birmingham, UK (Taiwo 2016)]

(Alolayan et al. 2013; Mansha et al. 2012; Rahman et al. 2011), principal component analysis (Negral et al. 2008; Querol et al. 2006; Tiwari et al. 2010) and EPA chemical mass balance model (Gummeneni et al. 2011; Olson et al. 2008).

Quass et al. (2013) reviewed methodologies and results of approaches used for the source apportionment of particulate matter in Germany and found significant variations in source apportionment techniques over the past 20 years with higher investigations of rural and hot-spot areas. The most commonly used approaches were mass closure/tracer-based approaches, statistical receptor models (PCA and PMF), basic Lenschow approach and dispersion and chemical transport models. In Europe, dispersion models (41% Lagrangian, 59% Eulerian and 35% Gaussian) and trajectory models (41%) were mostly used as transboundary contribution of PM_{10} to natural PM (Fragkou et al. 2012).

Belis et al. (2013) observed a shift in receptor modelling from PCA, enrichment factors and classical factor analysis to advance model like PMF based on the review of 272 records of source contribution estimates of PM_{10} mass concentrations during the period of 2000–2012 in Europe.

Hopke (2016) reviewed the global historical perspective of source apportionment technology development and different methodologies, application and advancement. PCA and factor analysis were the earliest source apportionment methods followed by atmospheric mass balance model, target transformation factor analysis and Unmix, whereas positive matrix factorization by EPA is the most recent and used source identification tool (Hopke 2016). Methods using local wind data such as conditional probability function, nonparametric regression, nonparametric wind regression, sustained wind incidence are suggested for PM_{10} source apportionment (Hopke 2016).

Use of back trajectory analysis has significantly increased, whereas use of chemical mass balance (CMB) is limited (Fragkou et al. 2012; Quass et al. 2013; Hopke 2016) Fragkou et al. (2012) reviewed the current trends of PM source apportionment in Europe and found PCA and back trajectory analysis as the most common methods for PM_{10} source identification. It is clear from several source apportionment studies from past to present that on the global scale, natural emissions of particulate matter from sea spray, wildfires, wind erosion and volcanoes are estimated to exceed by far the emissions by anthropogenic activities (Quass et al. 2013; Hopke 2016).

Meteorological influence on PM_{10}

Meteorological factors play important role in the dispersion and consequent concentrations of particulate matter. Deshmukh et al. (2013) in Raipur City, India, found negative correlation between ambient temperature and wind velocity with PM and clear seasonal variations in PM with higher values during winter and lower in monsoon season due to precipitation. Desert dust aerosols have significant impact in local air quality as Rashki et al. (2013) found high summer and low winter concentration of PM_{10} in Zahedan, Iran, because of frequent dust storms in summer season. Zakey et al. (2008) reported that coarse particles from dust storm contribute significantly to PM_{10} in few summer months although higher concentrations were observed in winter than summer. Several fold higher PM_{10} levels were observed in Dalanzadgad, Zamyn-Uud and Dalanzadgad, Mongolia, during dust storm events (Jugder et al. 2011).

Long-range transport affects the local air quality as observed by Spindler et al. (2013) when higher winter season concentration of PM_{10} in Melpitz and the surrounding East and Northeast German lowlands were attributed to long-range transport of polluted air masses from east. Similarly in Athens, high coarse PM levels were reported due to air masses originating from either North Africa or the Western Mediterranean contributed mostly by desert dust or sea salt (Kassomenos et al. 2012). Negral et al. (2008) in Cartagena, Spain, reported sudden increase in PM_{10} mass due to African dust storms. Tiwari et al. (2010) also found a higher concentration of PM_{10} in New Delhi, India, during pre-monsoon as a consequence of mineral dust transported from Thar Desert.

Time-series analysis has shown that PM levels have a seasonal trend with high concentrations during winter season (Dongarrà et al. 2010; Galindo et al. 2011; Kothai et al. 2011; Li et al. 2014; Moroni et al. 2012). Several factors affect PM concentrations during winter season such as lower temperature, calm weather conditions, temperature inversion, reduction in wet scavenging and lower mixing height that reduce and limit PM dispersion and dispersal (Karar and Gupta 2006; Joseph et al. 2012; Kothai et al. 2011). During summer season, both high wind speed and large mixing height help dispersion of pollutants from the atmosphere (Karar and Gupta 2006; Tiwari et al. 2010). Emission sources like coal consumption and biomass burning also add to increase in PM concentrations during winter months apart from prevailing meteorological conditions (Guttikunda et al. 2013). Strong association between variations in PM_{10} with season was reported in Zahedan, Iran, by Rashki et al. (2013). PM_{10} data from seven major cities in Korea from 1996 to 2010 revealed maximum concentration during spring followed by winter, fall and least in summer (Sharma et al. 2014a). Seasonal variations in PM_{10} concentrations are a universal phenomenon in all the regions of the world. Only local emissions or street canyon effects can alter this relationship (Quass et al. 2013; Hopke 2016).

Pakalidou et al. (2013) reported increase in mean PM_{10} concentrations by 7 and 4 $\mu g\ m^{-3}$ during heat wave days, respectively, in urban background and urban traffic stations due to formation of more secondary particles.

In Kolkata, India, winter to monsoon ratio for PM_{10} was higher than summer to monsoon ratio at both residential and industrial sites (Karar and Gupta 2006). The higher values observed for winter to monsoon ratio may be due to increase in PM formation and emission during winter. Maximum PM_{10} pollution occurred during winter (39%) followed by spring (30%), fall (26%) and least in summer (5%) season in Wuhan, China (Feng et al. 2011). Callén et al. (2012) in Zaragoza, Spain, observed a significant influence of meteorology in exceedance of PM_{10}. Pateraki et al. (2012) found a negative correlation between PM and relative humidity, whereas temperature showed a positive correlation at a suburban area of Athens. Galindo et al. (2011) in Elche, Spain, found positive correlations between coarse PM with temperature and solar radiation ($r = 0.60$ and 0.70, respectively). During summer season in Klang Valley, Malaysia, Juneng et al. (2011) found local meteorological factors such as surface air temperature, local humidity and local wind as major factors determining PM_{10} concentrations. Barmpadimos et al. (2011) found that PM_{10} concentrations are affected by boundary layer depth in all the seasons.

Hussein et al. (2014) reported wind as a major factor controlling PM levels in ambient air of Jeddah, Saudi Arabia, as wind brings more dust to the city from desert. It was also found that other variables like temperature, pressure and relative humidity are also governed by wind direction and wind sector. On the other hand, Chaloulakou et al. (2005) in Athens, Greece, observed negative correlation between PM_{10} levels and wind velocity and found

exceedance of 100 μg m^{-3} PM_{10} concentration in 86% of the sampling days where average wind velocity was below 2 m s^{-1}. Tiwari et al. (2016) observed a negative correlation between wind speeds and PM_{10} in both winter (−0.48) and post-monsoon (−0.32) seasons in urban atmosphere of Patna, India. Chaloulakou et al. (2005) observed 17.7% reduction in PM_{10} level during rainy days in Athens, Greece. Li et al. (2014) suggested that combustion during winter season, sands storms in spring and rain events in summer mostly influence PM_{10} levels in 18 sites across Northern China. Kassomenos et al. (2012) found positive correlation between coarse particulate matter with temperature and negative correlation with relative humidity and precipitation in three European capitals (London, Madrid and Athens).

Barmpadimos et al. (2011) studied the influence of meteorology on PM_{10} concentrations in 13 air quality stations of Switzerland from 1991 to 2008 and found a negative relationship between PM_{10} and wind gust, yesterday precipitation and convective boundary layer depth, whereas afternoon sunshine duration and afternoon temperature were positively related. Small precipitation rate was found to be strongly reducing the concentrations of PM_{10} in Switzerland (Barmpadimos et al. 2011). Rashki et al. (2013) found solar heating and vertical mixing of pollutants to be the major governing factor behind decrease in PM_{10} levels at noon and early afternoon hours. In Beijing metropolitan region, Tian et al. (2014) found atmospheric pressure as the most influencing factor followed by relative humidity and wind speed in regulating PM_{10} level, although effects were different in each season.

Guerra et al. (2006) found statistically significant effect of wind direction on PM_{10} concentrations in Southeast Kansas, USA. Highest PM_{10} concentrations were recorded on days with predominant southern winds, whereas lower concentrations were recorded with predominant northern wind direction or from other directions. Fiddes et al. (2016) assessed relationship between synoptic weather evolution and climate drivers with winter air PM_{10} levels in New Zealand. Higher exceedances in PM_{10} were observed on days with weaker westerly winds as strong wind from west disperses the particulate matter. Synoptic weather conditions such as southwesterlies wind over the equatorial area and cyclonic flow associated with typhoon activities were found to be influencing PM_{10} concentrations over the Klang Valley (Juneng et al. 2011).

$PM_{2.5}/PM_{10}$ ratio

Different emission sources produce particles of different sizes. Identification of these particles can be useful in identification of source/origin. $PM_{2.5}/PM_{10}$ ratio gives useful information about the sources as natural or anthropogenic. Anthropogenic sources are known to produce more fine particles as a result of traffic emissions or burning activities resulting in higher $PM_{2.5}/PM_{10}$ ratio, whereas natural sources as windblown or road dust mostly have higher contribution of coarse particles resulting in a lower value (Zakey et al. 2008; Querol et al. 2001; Kulshrestha et al. 2009; Spindler et al. 2013).

Zakey et al. (2008) reported lower ratio in residential area (0.32) compared to urban area (0.59) in Greater Cairo, Egypt. Similar results were also observed in urban area of Tianjin, China, and Raipur City, India, by Li et al. (2012) and Deshmukh et al. (2013), respectively, with ratio of 0.53 and 0.54, but significant variation was recorded by Deshmukh et al. (2013), as they found ratio between 0.14 and 0.74, and these variations were mostly due to seasonal changes and anthropogenic activity with higher values in winter season. Seasonal variation in this ratio was more distinct in New Delhi, India, where ratio was lowest during summer month of June (0.18) and maximum in winter month of February (0.86) with average value of 0.48. In an urban atmosphere in Raipur City, ratio was higher at winter (0.61) compared to summer (0.44) as a result of increase in combustion activities. In urban area of Pune, India, Pipal and Gursumeeran Satsangi (2015) reported average $PM_{2.5}/PM_{10}$ ratio of 0.64 with values ranging from 0.51 to 0.78. $PM_{2.5}/PM_{10}$ ratio showed slight variations with traffic in Delhi, India, with values of 0.53 and 0.47 at high and less traffic site, respectively (Tiwari et al. 2015). Hussein et al. (2014) found a slightly lower ratio (0.39) in urban area of Jeddah, Saudi Arabia, whereas in Dhaka, Bangladesh, values were between 0.3 and 0.5 (Begum et al. 2013). Sharma and Maloo (2005) found higher ratio at a control site (0.74) followed by commercial (0.56) and least at residential site (0.45) in Kanpur, India. Several studies in urban environment found $PM_{2.5}/PM_{10}$ ratio above 0.6 indicating significance contribution of fine particles to PM_{10} (Antonel and Chowdhury 2014; Chuersuwan et al. 2008; Dongarrà et al. 2010; Kendall et al. 2011; Kulshrestha et al. 2009).

Combustion sources (traffic, biomass burning and industrial processes) emit more fine particles, whereas mechanical processes (crushing, gridding and construction activities) contribute to coarse fraction of particulate matter. So $PM_{2.5}/PM_{10}$ ratio can depict the actual sources of pollution in spatial–temporal studies. It is well known that the combustion sources generally increase the values of $PM_{2.5}/PM_{10}$ ratio.

At a rural site in Spain, Arruti et al. (2012) found ratio of 0.54, whereas Kulshrestha et al. (2009) found slightly higher ratio of 0.61 at a rural site in Agra, India, and suggested the influence of diesel generator, construction and windblown dust for this higher concentration of $PM_{2.5}$.

Moroni et al. (2015) reported average $PM_{2.5}/PM_{10}$ ratio of 0.77 at a rural background station of Monte Martano in Central Italy, during the study campaign but a significant decrease in the ratio was observed during Saharan dust episodes with average value of 0.48. Secondary aerosol formation significantly affects thus ratios which are mostly dependent upon emission sources (Munir 2017). The variations in the ratio are directly correlated with land-use pattern as Munir (2017) found lower value of 0.40 at rural background site. Similar results were observed by Arruti et al. (2012) at a rural site in Cantabria region of Spain.

In a long-term study at a rural site in Melpitz, Germany, Spindler et al. (2013) found an increase in the ratio from 0.71 in 1995 to 0.84 in 2012 and implicated the role of long-range transport of fine particles. Moroni et al. (2012) found lower ratio during Saharan dust intrusions (0.49) and higher value (0.80) during industrial dust intrusions from Eastern Europe at Terni basin in Central Italy. Querol et al. (2001) found that $PM_{2.5}/PM_{10}$ ratio varied from 0.60 to 0.65 at urban kerbside in Barcelona, Spain, during the traffic hours. Dionisio et al. (2010) observed lower ratio at traffic site compared to residential site at Accra, Ghana. Gehrig and Buchmann (2003) found higher contribution of $PM_{2.5}$ level in PM_{10} in Switzerland with a ratio of 0.76.

Munir (2017) reported considerable variations in $PM_{2.5}/PM_{10}$ ratio ranging from 0.4 to 0.8 with median value of 0.65 in UK during 2010–2014 and attributed these variations to seasonal changes and sources around the monitoring sites. Diurnal cycle and variations in weekly cycle also alter the $PM_{2.5}/PM_{10}$ ratio (Gehrig and Buchmann 2003; Pipal and Gursumeeran Satsangi 2015; Munir 2017). Munir (2017) observed higher ratio in early morning (before 0600 hours) and in evening hours (after 1800 hours) and lower around the midday (about 1000–1400 hours). Higher average ratios were observed on Saturday and Sunday and lowest on Thursday in UK (Munir 2017). At four urban sites in Taiwan, Sun et al. (2003) found no significant influence of traffic and rainfall for the seasonal variations in $PM_{2.5}/PM_{10-2.5}$ ratios. Ratios were slightly higher in weekend compared to weekdays owing to higher vehicular emissions during the weekends. Daytime ratio was slightly higher compared to nighttime, which was again correlated with higher traffic emissions during the daytime (Sun et al. 2003).

Meteorological variables also influenced the ratio as wind speed, wind direction, precipitation, relative humidity and temperature affect particulate matter emissions and formation. High wind speed and rainfall lead to greater reduction in large particles, leading to increase in the ratio (Munir 2017). Sun et al. (2003) attributed meteorological conditions (rain fall, solar radiation and atmospheric stability), emissions from traffic and combustion sources and photochemical reactions to be responsible for increase in $PM_{2.5}/PM_{10-2.5}$ ratio.

Based on the literature surveyed we found significant variations in $PM_{2.5}/PM_{10}$ ratio in different regions as well as in different land-use pattern of the world. Values ranged from 0.32 to 0.78 in most of the urban areas in India with mean ratio of 0.50, whereas from 0.55 to 0.61 with mean value of 0.55 in China and East Asian cities. Ratio was comparatively low in Middle Eastern cities (0.25–0.41) which may be due to more contribution of desert dust to PM_{10}. In European cities, values were typically ranged between 0.37 and 0.74. When different land-use patterns were compared, values ranged from 0.37 to 0.74, 0.44 to 0.78 and 0.4 to 0.75, respectively, at urban traffic, urban background and rural or remote areas of the world. These variations were mostly due to local factors, meteorology, and measurement site and nearby sources. These trends suggest that a more uniform criteria and larger database are required to better identify the sources and different size fraction of particulate matter.

Health effects of PM_{10}

Primary target of particulate pollution is mostly associated with respiratory ailment (Anenberg et al. 2010; Ding et al. 2014; Gauderman et al. 2004; Gehring et al. 2013; Pope III et al. 2002), but in recent time important risks of PM_{10} has also been identified as low birth weight (LBW) (Dadvand et al. 2014), fetal growth characteristics and preterm birth (van den Hooven et al. 2012), DNA damage and mutagenic activity (Coronas et al. 2009), congenital heart defects (Agay-Shay et al. 2013), ischemic heart disease (Zhang et al. 2014), inflammatory responses (Silbajoris et al. 2011), infant mortality (Son et al. 2011), oxidative stress (Kim et al. 2012) and atherosclerosis (Tonne et al. 2012).

Meta-analysis of 17 European cohort studies from 2008 to 2011 showed statistically significant association with risk for lung cancer and PM_{10} (Raaschou-Nielsen et al. 2013). The study also highlighted that HR (health risk) for lung cancer of 1.09 (0.99–1.21) was associated with increase in road traffic of 4000 vehicles per km per day within 100 m of the residence.

Mutagenic activity and DNA damage due to PM_{10} in people (age 18–40 years) downwind from an oil refinery were reported in Esteio, Brazil, in 2006 (Coronas et al. 2009). Increased DNA damage in lymphocytes and positive responses for mutagenicity were detected in all samples, indicating the complexity and carcinogenic nature of particulate matter. Effects of particulate matter on cultured cells have shown that different components of particulate matter or their combination can produce marked changes in cellular level (Øvrevik et al. 2009). Silbajoris et al. (2011) experimented with exposure of Mexicali PM_{10} suspension to cultured HAECs resulted in increase in p65 binding to

the genomic IL-8 promoter in HAEC cells, indicating that PM_{10} in ambient air can induce inflammatory responses. Díaz-Robles et al. (2013) identified diesel particulate matter (DPM) to possess maximum cancer risk among priority mobile source air toxics in USA and also found higher concentrations in urban and rural areas around Southeastern USA, Chicago, Indianapolis, Atlanta, Nashville and Birmingham. These areas were marked as most susceptible with highest cancer risk from diesel particulate matter (Table 1).

Population-based cohort study among 7772 pregnant women in the Netherlands showed that PM_{10} exposure in the third and fourth quartiles were positively associated with preterm birth (van den Hooven et al. 2012). Schifano et al. (2013) also find delayed and prolonged effect of PM_{10} exposure on preterm-birth risk in birth cohort consisted of 132,691 births, from 2001 to 2010 in Rome, Italy. In birth cohort study by Son et al. (2011) in Seoul, Korea, total mortality risks were 1.65 (95% CI 1.18–2.31), respiratory mortality risks were 6.20 (95% CI 1.50–25.66) and sudden infant death syndrome risks were 1.15 (95% CI 0.38–3.48) per IQR (interquartile range) increase in PM_{10} for associations between long-term exposure during pregnancy and end of eligibility for outcome at 1 year of age to different particle sizes and infant mortality. Israeli registry-based birth cohort study from Tel Aviv region concluded that increased exposure to PM_{10} during 3–8 weeks of pregnancy is significantly associated with an increased risk for multiple congenital heart defects (Agay-Shay et al. 2013). Dadvand et al. (2014) found significant association between increased risks of term LBW with maternal residential proximity to major roads on 6438 singleton term birth cohort study in Barcelona, Spain, and identified PM_{10} as a major contributor to increase in term LBW (Table 1).

Many studies have reported the relationship between short-term and long-term exposure to PM_{10} concentrations and mortality. Time-series analysis of mortality effects from particulate matter size fractions in Beijing, China, found significant associations of daily mortality with PM_{10} (Li et al. 2013). Meng et al. (2013) in Shenyang, China, found increase in adverse health effects with reduction in particle size. Short-term effects of ambient particles on cardiovascular and respiratory mortality study in 29 European cities found that increase of 10 μg m^{-3} PM_{10} was associated with increases of 0.76% (95% CI 0.47–1.05) in cardiovascular deaths and 0.58% (0.21–0.95%) in respiratory deaths. A long-term exposure study in four Chinese cities by Zhang et al. (2014) reported 10 μg m^{-3} increase in PM_{10} was associated with the relative risk ratios for all-cause mortality 1.24 (95% CI 1.22–1.27), cerebrovascular disease mortality 1.23 (95% CI 1.18–1.28), cardiovascular disease mortality 1.23 (95% CI 1.19–1.26) and heart failure disease mortality 1.11 (95% CI 1.05–1.17) suggesting significant association between long-term exposure to PM_{10} and cardiovascular mortality. Pascal et al. (2014) reported short-term associations between PM_{10} and mortality in nine French cities and found 10 μg m^{-3} increase in daily PM_{10} levels was associated with a 0.2% [−0.5; 0.9] increase in non-accidental mortality, but importantly these effects were realized even at concentrations within the EU annual regulation, and close to the WHO guideline values. Fine particulate air pollution and mortality study in 20 US cities showed that death from cardiovascular and respiratory causes had higher association with PM_{10} than rate of death from all causes (Samet et al. 2000).

Improvement in air quality has certain health benefits as health impact assessment study in the city of Rotterdam, Netherland, showed that decrease in PM_{10} levels from 1985 to 2008 resulted in a gain in life of an average 13 months per person (Keuken et al. 2011). Table 1 presents the summary of studies examining the association between PM_{10} and health effects.

Conclusion

A global status and trend of PM_{10} indicated a critical situation of PM_{10} levels in most of the developing countries of the world with higher exceedances in large urban centers of all major cities. PM_{10} values were shown to be declining in Europe and USA, whereas in most Asian countries values were still critical, especially in India and China. Complex relationships were observed between PM_{10} sources in different regions of the world, but, crustal matter, vehicular or traffic emissions and biomass burning were the major sources identified for most of the studies. Apart from anthropogenic sources, dust storms have severe effect on PM_{10} variability in most of the continents. Meteorological factors such as wind speed, temperature, relative humidity play significant role in seasonal pattern of PM_{10}. $PM_{2.5}$/PM_{10} ratio is a useful marker for assessment of pollution sources and particulate matter distribution. Health studies showed a negative relationship between exposure of PM_{10} and health status. Birth anomalies, loss of life years and increase in cardiovascular and respiratory diseases are more prevalent health effects in areas with higher PM_{10}. $PM_{2.5}$ contributes a significant portion of PM_{10} and its toxicity which mostly depends upon its chemical components at lower size fraction, so it is important to associate $PM_{2.5}$ and its chemical components in future studies to better assess the status and negative effects of PM_{10}. By reducing the vehicular emissions, improvement in urban planning, promoting green infrastructure and implementation of strict particulate matter standards, the declining health quality due to particulate matter pollution can be improved.

Table 1 Summary of studies examining the association between PM_{10} and health impacts in different regions of the world. PM_{10}: particulate matter having particle sizes 10 micrometers or less in diameter

Study	Data collection period	Location	Sample size	Age	Method/model	Result	Findings	References
PM mass concentration and carotid intima-media thickness	2003–2005	London, England	2348		Generalized linear regression models	Increase of 5.2 μg m^{-3} in PM_{10} was associated with a 5.0% (95% CI 1.9–8.3) increase in intima-media thickness	Significant association between PM mass concentration with carotid intima-media thickness	Tonne et al. (2012)
Long-term exposure to ambient air pollution and lung cancer incidence	2008–2011	9 European countries	312,944		Cox regression models	Significant association between risk for lung cancer and PM_{10} [HR 1.22 (95% CI 1.03–1.45) per 10 μg m^{-3}] and 1.51 (1.10–2.08) for adenocarcinoma of the lung	Significant association between long-term exposure to ambient air pollution and lung cancer	Raaschou-Nielsen et al. (2013)
Mothers' residential proximity to major roads and term low birth weight (lBW)	2001–2005	Barcelona, Spain	6438	30	Generalized additive models	Living within 200 m of major roads was associated with a 46% increase in term lBW risk	Increased risk of term lBW associated with proximity to major roads	Dadvand et al. (2014)
Fine particulate air pollution and mortality	1987–1994	20 US cities	>50 million		Two-stage log-linear regression model	The estimated increase in the relative rate of death from all causes was 0.51% (95% PI 0.07–0.93) and 0.68% (95% PI 0.20–1.16) for cardiovascular and respiratory causes for 10 μg m^{-3} increase in the PM_{10}	Fine particulate air pollution has a significant effect on risk of death from all causes and from cardiovascular and respiratory illnesses	Samet et al. (2000)
Short-term associations between PM_{10} and mortality	2000–2006	9 French cities	12 million	15–74, >74	Poisson regression model	For each 10 μg m^{-3} increase in PM_{10} results in increase of (+0.8% 95% CI [0.2; 1.5] on all-ages non-accidental mortality)	Significant association between short-term impacts of PM_{10} on mortality	Pascal et al. (2014)

Table 1 continued

Study	Data collection period	Location	Sample size	Age	Method/model	Result	Findings	References
Exposure to PM and signalling events involved in expression of the inflammatory gene interleukin-8 (IL-8) in human airway epithelial cells (HAECs)	2005–2006	Mexicali, Mexico			Cultured HAECs exposed to Mexicali PM suspended in media for 0–4 h and analyzed by ChIP assay (chromatin immunoprecipitation) followed by RT-PCR	Twofold to eightfold increase in IL-8 mRNA expression relative to controls with PM exposure. ChIP assays showed a threefold increase in binding of the p65 (RelA) NF-κB isoform to the IL-8 promoter sequence	Exposure to PM_{10} in ambient air can induce inflammatory responses by activating specific signalling mechanisms through an NF-κB-dependent signalling mechanism	Silbajoris et al. (2011)
Assessment of air toxic-related health risks due to different emission scenarios	2003	Nashville, USA			Multi-scale Air Quality model (Models-3/CMAQ)	Diesel particulate matter (DPM) poses a maximum cancer risk that is 4.2 times higher than the combined total cancer risk from all other priority mobile source air toxics	Significant contribution of DPM to health risk	Díaz-Robles et al. (2013)
Maternal air pollution exposure with fetal growth characteristics and adverse birth outcomes	2001–2005	Rotterdam, Netherlands	7772	30.4	Multivariate linear regression models and mixed-effects models	PM_{10} level was associated with fetal head circumference in the third trimester [–0.18 mm, 95% CI −0.24, −0.12 mm. The third and fourth quartiles of PM_{10} exposure were associated with preterm birth (OR 1.40, 95% CI 1.03, 1.89; and OR 1.32; 95% CI 0.96, 1.79) relative to the first quartile]	Inverse association between maternal PM_{10} exposure with fetal growth and with weight at birth. Positive association between elevated PM_{10} exposure with preterm birth and size for gestational age at birth	van den Hooven et al. (2012)

Table 1 continued

Study	Data collection period	Location	Sample size	Age	Method/model	Result	Findings	References
Association between genotoxic effects on people residing and working downwind from an oil refinery with mutagenic activity of airborne PM_{10}	2006	Esteio, Brazil	74	18–40	Peripheral blood and buccal mucosa cells of healthy men recruits were evaluated using comet assay and the micronucleus assay	Mutagenic positive response was shown by all PM_{10} organic extracts. Exposed group showed significantly higher DNA damage in lymphocytes than the reference group	Under the influence of oil refinery PM_{10} samples showed significant mutagenicity and people exposed in this area have higher primary DNA damage	Coronas et al. (2009)
Association between short-term exposure to high and low temperatures and air pollution on preterm birth	2001–2010	Rome, Italy	132,691	11–55	Time-series approach	PM_{10} showed significant effect on preterm-birth risk at a lag period of 12–22 days during the warm season (+0.69%; 95% CI 0.23–1.15, for 1 μg m^{-3} increase of pollutant)	Significant association between short-term effect of heat and a more delayed and prolonged effect of PM_{10} exposure on preterm-birth risk	Schifano et al. (2013)
Health impact of PM_{10}	1985–2008	Rotterdam, Netherlands	500,000	18–64	URBIS (Urban environmental Information System) model	Gain of life per person on average of 13 months for PM_{10}	Reduction of combustion aerosol resulted in a reduction in health impact of PM_{10}	Keuken et al. (2011)
Time-series analysis of mortality effects from PM size fractions	2005–2009	Beijing, China			Poisson generalized additive models	10 μg m^{-3} increase in PM_{10} is associated with a 0.15% (95% CI 0.04–0.22), 0.08% (95% CI 0.01–0.18) and 0.44% (95% CI 0.12–0.63) increase in non-accidental, respiratory and circulatory mortality, respectively	Significant associations between daily mortality with PM_{10} and combined effect of temperature and particulate pollution may have serious health effects	Li et al. (2013)

Table 1 continued

Study	Data collection period	Location	Sample size	Age	Method/model	Result	Findings	References
Cardiovascular mortality of long-term exposure to high level concentrations of inhalable particulate pollution	1998–2009	Tianjin, Shenyang, Taiyuan and Rizhao, China	39,054	44.29 ± 13.95	Cox proportional hazards regression models	The relative risk ratios were maximum for ischemic heart disease mortality 1.37 (95% CI 1.28–1.47) followed by of all-cause mortality 1.24 (95% CI 1.22–1.27), cerebrovascular disease mortality 1.23 (95% CI 1.18–1.28), cardiovascular disease mortality 1.23 (95% CI 1.19–1.26) and heart failure disease mortality 1.11 (95% CI 1.05–1.17) with 10 μg m^{-3} increase in PM_{10}	Increase in mortality from cardiovascular disease, especially from ischemic heart disease with long-term exposure to PM_{10} and several factors affecting this interaction	Zhang et al. (2014)
Gestational exposure to air pollution and risk of congenital heart defects	2000–2006	Tel Aviv, Israel	135,527	<21–41≤	Spatiotemporal methodology of weekly inverse distance weighting in birth cohort	PM_{10} was associated with multiple congenital heart defects (adjusted OR 1.05, 95% CI 1.01–1.10 for 10 μg m^{-3} increment)	Increase in PM_{10} exposure significantly increases the risk for multiple congenital heart defects	Agay-Shay et al. (2013)

IQR interquartile range, *CI* confidence interval, *HR* hazard ratio, *OR* odds ratio

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