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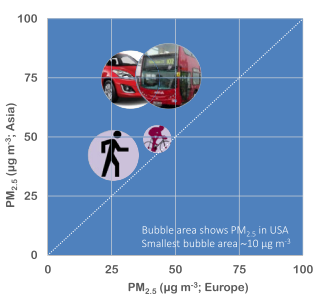
Review article

A review of factors impacting exposure to $PM_{2.5}$, ultrafine particles and black carbon in Asian transport microenvironments

Prashant Kumar[a,b,*], Allison P. Patton[c], John L. Durant[d], H. Christopher Frey[e]

[a] Global Centre for Clean Air Research, Department of Civil and Environmental Engineering, Faculty of Engineering and Physical Sciences (FEPS), University of Surrey, Guildford GU2 7XH, United Kingdom
[b] Department of Civil, Structural & Environmental Engineering, School of Engineering, Trinity College Dublin, Ireland
[c] Health Effects Institute, Boston, MA, USA
[d] Department of Civil & Environmental Engineering, Tufts University, Medford, MA, USA
[e] Department of Civil, Construction, and Environmental Engineering, North Carolina State University, Campus Box 7908, Raleigh, NC 27695-7908, USA

GRAPHICAL ABSTRACT



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ABSTRACT

The World Health Organization estimates 3.7 million deaths in 2012 in low- and middle-income Asian countries due to outdoor air pollution. However, these estimates do not account for the higher exposures of specific particulate matter (PM) components – including fine particles ($PM_{2.5}$), ultrafine particles (UFP) and black carbon (BC) – typical of transport microenvironments (TMEs). With the rapidly growing number of on-road vehicles in Asia, human exposure to PM is an increasing concern. The aim of this review article is to comprehensively assess studies of $PM_{2.5}$, UFP, and BC in Asian TMEs in order to better understand the extent of exposure, the underlying factors leading to exposure, and how Asian exposures compare to those found in Europe and the United States of America (USA). The health impacts of exposure to $PM_{2.5}$, UFP, and BC are described and the key factors that influence personal exposure in TMEs (i.e., walk, cycle, car, and bus) are identified. Instrumentation and measurement methods, exposure modeling techniques, and regulation are reviewed for $PM_{2.5}$, UFP, and BC. Relatively few studies have been carried out in urban Asian TMEs where $PM_{2.5}$, UFP, and BC had generally higher concentrations compared to Europe and USA. Based on available data, $PM_{2.5}$ concentrations while walking were 1.6 and 1.2 times higher in Asian cities (average 42 $\mu g\,m^{-3}$) compared to cities in Europe (26 $\mu g\,m^{-3}$) and the USA (35 $\mu g\,m^{-3}$), respectively. Likewise, average $PM_{2.5}$ concentrations in car (74 $\mu g\,m^{-3}$) and bus (76 $\mu g\,m^{-3}$) modes in Asian cities were approximately two to three times higher than in Europe and American cities. UFP exposures in Asian cities were twice as high for pedestrians and up to ~9-times as high in cars than in cities in Europe or the USA. Asian pedestrians were exposed to ~7-times higher BC concentrations compared with pedestrians in the USA. Stochastic population-based models have yet to be applied widely in Asia

* Corresponding author. Global Centre for Clean Air Research, Department of Civil and Environmental Engineering, Faculty of Engineering and Physical Sciences (FEPS), University of Surrey, Guildford GU2 7XH, United Kingdom.
E-mail addresses: P.Kumar@surrey.ac.uk, Prashant.Kumar@cantab.net (P. Kumar).

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but can be used to quantify inter-individual and inter-regional variability in exposures and to assess the contribution of TMEs to total exposures for multiple pollutants. The review also highlights specific gaps in the Asian TME data set that need to be filled since UFP and BC studies were rare as were studies of pedestrian and cyclist exposure.

1. Introduction

Exposure to air pollution contributes significantly to morbidity and mortality in Asia. About 60% of the world's population lives in Asia (Population Reference Bureau, 2015) and nearly 88% of 3.7 million global excess deaths due to outdoor air pollution exposure in 2012 occurred in low- and middle-income countries (WHO, 2012). In 2013, exposure to ambient fine particulate matter ≤2.5 μm in aerodynamic diameter ($PM_{2.5}$) contributed to an estimated 915,900 deaths in China alone, and of these 137,400 (15%) were attributable to $PM_{2.5}$ from the transport sector (GBD, 2016). Ultrafine particles (UFP; < 0.1 μm in diameter) and individual components of $PM_{2.5}$, such as black carbon (BC; usually < 0.3 μm) also have high concentrations in transportation microenvironments (TMEs) and have been associated with adverse health outcomes (Dons et al., 2011a; Rivas et al., 2017a). UFPs are capable of travelling deep into the lungs (HEI, 2013), and their exposure has been linked to increases in blood biomarkers of cardiovascular effects and to excess mortality (Atkinson et al., 2010; Lane et al., 2016). BC exposure has been linked with cardiorespiratory events (Dons et al., 2012; Janssen et al., 2016; Karanasiou et al., 2014). Elemental carbon (EC) has often been used as a surrogate for BC, although strictly EC and BC are not identical, and are based on operational definitions related to measurement methods (Briggs and Long, 2016). Since UFP and BC have been studied less than PM as a whole, their effects are usually not included in mortality estimates (Kumar et al., 2013c, 2014).

TMEs such as walking, cycling, car, bus, and open-air vehicles (i.e., motorcycles, auto rickshaws) are the most common modes of local transport in the majority of Asian cities (Arphorn et al., 2017; Kumar et al., 2013a, 2015; Rivas et al., 2017a; Zuurbier et al., 2010). Concentrations of traffic-related air pollutants are generally higher in TMEs than other areas because the direct emissions from mobile sources have not been widely dispersed (Berghmans et al., 2009; Colvile et al., 2001; Goel and Kumar, 2014; Patton et al., 2016). In high-traffic areas, congested traffic flow can have higher emissions due to frequent acceleration and deceleration of vehicles (Goel and Kumar, 2015).

People living in Asian, European, and North American cities spend comparable amounts of time (7–10% of the day) in TMEs (Chau et al., 2002; Ragettli et al., 2013; Riediker et al., 2003; Saksena et al., 2007; Wallace and Ott, 2011). Air pollution exposures experienced while commuting contribute up to 30% and 12% of total daily inhaled doses of BC (Dons et al., 2011b) and $PM_{2.5}$ (Fondelli et al., 2008), respectively. Ambient levels of air pollution concentrations in Asian cities have been reported to be relatively higher than European cities (Kumar et al., 2014, 2015). The levels of exposure in Asian TMEs may increase in the future due to increases in numbers of on-road vehicles driven by increasing population and growing economies (Kumar et al., 2013a). For example, the number of road vehicles in Beijing (China) increased from about 1.5 million in 2000 to over 5 million in 2014 (Yang et al., 2015b). Likewise, the number of on-road vehicles in Delhi (India) is expected to increase from 4.74 million in 2010 to 25.6 million by 2030 under a business as usual scenario (Kumar et al., 2011a). Furthermore, the number of motorcycles increased from ~0.14 million in 2001 to 0.24 million in 2004 in Ho Chi Minh City, Vietnam (UNESCAP, 2009).

Passenger travel per capita using light-duty, two- and three-wheeler vehicles, buses, and passenger rail in China is projected to increase from ~2000 passenger-miles in 2012 to over 6000 passenger-miles by 2032, and per capita passenger travel in India is projected to more than double during the same period (IEO, 2016). The rate of private vehicle ownership in China and India, at less than 100 light-duty vehicles per thousand people in 2012, is far below the OECD average of over 550 but is expected to increase by a factor of three over the next three decades (IEO, 2016).

As seen in Table 1, only about half a dozen review articles have been published on pollutant exposures in TMEs, and none of these have focused on air pollutant exposures in TMEs in Asian cities. Because vehicle fleet composition, air pollution control technologies, road designs, and driving behavior are generally different in Asian cities compared to other parts of the world, TME studies performed elsewhere may not be generalizable to Asian cities (Kumar et al., 2015). To address this gap in the literature, this review discusses the underlying factors that impact exposure to $PM_{2.5}$, UFP and BC in TMEs in Asian cities. Our objectives are to (i) critically synthesise the published literature on $PM_{2.5}$, BC and UFP exposures in urban and rural TMEs (i.e., car, bus, cycle, walk, open-air vehicles including motorcycles and auto rickshaw) of Asian environments; (ii) establish a baseline using the available measurements of BC, UFP and $PM_{2.5}$; (iii) compare exposure levels in Asian TMEs to those in other more thoroughly studied areas including Europe and United States of America (USA); and (iv) highlight challenges, research gaps and future directions to increase the amount of information available on Asian TME exposures.

2. Methods and outline

This review covers three pollutants (UFP, BC and $PM_{2.5}$) and the following most common modes of daily commuting in Asian cities: cars, buses, bicycles, motorcycles, auto rickshaws and on/near-road walking. To identify articles on TME studies in general, and Asian TME studies in particular, we used the following keywords: ultrafine particles, $PM_{2.5}$, black carbon, transport microenvironments, walking, bicycling,

Table 1
Summary of relevant review articles considering pollutant exposure in TMEs.

Authors (year)	Pollutants	Remarks
Han and Naeher (2006)	PM, CO, NO_2, VOCs, and PAHs	Air pollution exposure assessment in the developing world related to traffic and air pollution monitoring methods.
Kaur et al. (2007)	$PM_{2.5}$, UFP, and CO	Personal exposure in TMEs (walk, cycle, bus, car, van, and taxi) and comparison between actual personal exposure measurement and fixed monitoring stations (FMS) for representing community exposure to pollutants.
Knibbs et al. (2011)	UFP	UFP exposure concentrations in transport modes (automobile, bus, cycle, ferry, rail, and walk) and correlation with other pollutants.
Karanasiou et al. (2014)	PM and BC	Personal exposure in European TMEs (cycle, car, bus, and train).
Goel and Kumar (2014)	UFP	Important aspects of concentrations at traffic intersections.
de Nazelle et al. (2017)	UFP, BC, $PM_{2.5}$ and CO	Exposures of active versus passive travel mode users in European environment
This study	$PM_{2.5}$, BC and UFP	Exposure assessment and methods of measurements in Asian versus developed countries

personal cars, buses, auto rickshaws, urban transportation, air pollutants and exposure, Asian cities – in Google Scholar, ISI Web of Science, Scopus, Science Direct and PubMed. This was supplemented by papers included in the reference lists of the traced papers and papers that were already known to us based on previous TME work. We only included articles that were written in the English language, and that were published over the past two decades (1997–2017) in peer-reviewed journals.

Our article is organised as follows: Section 3 provides background on the pollutants covered and their health impacts; key factors that influence personal exposure in TMEs are summarized in Section 4; exposure to $PM_{2.5}$, UFP, and BC in TMEs (walk, cycle, car, bus, auto rickshaw) and comparisons of personal exposures in Asian and non-Asian cities are presented in Section 5; and instrumentation used to measure $PM_{2.5}$, UFP and BC in TME studies and considerations for study design are presented in Sections 6. In addition, exposure modeling techniques are described in Section 7, and relevant air pollution control regulations are described in Section 8. Section 9 contains a summary as well as conclusions and suggestions for future research.

3. Importance of pollutants covered

$PM_{2.5}$, UFP and BC have strong associations with exhaust emissions and have not been widely discussed in the context of Asian TMEs. The primary source of these pollutants in many cities including Delhi and major Chinese cities is engine emissions from the combustion of fuel (diesel and petrol) in motor vehicles (Nagpure et al., 2016; Zhang and Cao, 2015). These emissions are also the most relevant for exposure assessment in TMEs since urban commuters are in relatively close proximity to emission sources during transit. $PM_{2.5}$ exposures have been linked to increased mortality (Di et al., 2017; Heal et al., 2012; Pun et al., 2017), cardiovascular and respiratory morbidity (Clougherty, 2010; Metzger et al., 2004; Peel et al., 2005; Peters, 2005), and reduced birth weight and increased post-neonatal mortality (WHO, 2005b).

In an epidemiologic study for southern China, exposure to $PM_{2.5}$ was associated with mortality from stroke, COPD, ischemic heart disease, lung cancer, and all-cause hospital admissions (Lin et al., 2017). Exposure to $PM_{2.5}$ in Taipei was associated with increased risk of emergency room visits related to respiratory diseases, increased all-cause mortality among the elderly, and increased mortality from circulatory diseases among the general population (Wang and Lin, 2016). In Guangzhou, short-term (daily) and long-term (annual) exposures to $PM_{2.5}$ were associated with mortality, including all-cause, cardiovascular, and respiratory mortality (Lin et al., 2016). Long-term exposure to $PM_{2.5}$ in Hong Kong was associated with elevated cancer risks for all-cancer mortality and cause-specific cancer mortality, including all digestive organs, upper respiratory tract, and accessory digestive organs (Wong et al., 2016).

UFPs are typically quantified in terms of particle number concentration (PNC), and dominate the number but not necessarily the mass of particulate matter in ambient urban environments or in TMEs (Heal et al., 2012; Goel and Kumar, 2014). On-road vehicles are a major source of UFP in urban areas and can contribute up to 90% of the total PNC in polluted roadside environments (Kumar et al., 2010). Ambient UFP number concentrations are typically elevated on and near busy roads, and decay to background within 500 m of the edge of the road (HEI, 2010; Karner et al., 2010; Padró-Martínez et al., 2012; Patton et al., 2014). For example, PNC can be on the order of 10 million particles cm^{-3} close to vehicle tailpipes (Carpentieri et al., 2011) and decrease to the order of 1000 particles cm^{-3} at urban background locations hundreds of meters away (Kumar et al., 2013b). The very small size of UFPs enables them to penetrate deeply into the cardiorespiratory system to the terminal bronchioles and alveoli (Rückerl et al., 2011), bloodstream (Bakand et al., 2012) and body organs such as lung, brain, kidney and liver (Morris et al., 2003). UFP exposure has been linked to increased medication use in asthmatic adults (Von Klot et al., 2002), diminished lung function (McCreanor et al., 2007), and damage DNA of buccal cells (de Almeida et al., 2018). Furthermore, positive associations have been observed between UFP exposure and myocardial ischemia, cardiac arrhythmia, cardiovascular mortality, and blood biomarkers of inflammation (Atkinson et al., 2010; Lane et al., 2016; Stölzel et al., 2007).

Studies in China have also found associations between health indicators or effects and exposure to UFP. For example, exposure to particles of 20 nm–100 nm was found to increase the fraction of exhaled nitric oxide (FeNO), an indicator of acute respiratory inflammation, in a panel of 55 elderly subjects in Shanghai (Han et al., 2016). Another study in Shanghai produced evidence consistent with a hypothesis that autonomic heart function is more associated with UFPs than with larger particles (Sun et al., 2015). In a time-series study in Shenyang, China, exposure to ambient PM smaller than 500 nm in diameter was related to total and cardiovascular mortality but not respiratory mortality (Meng et al., 2013). In Beijing, short-term exposure to sub-micrometer PM was associated with an elevated risk of cardiovascular mortality (Breitner et al., 2011).

BC is produced as a result of inefficient combustion of carbon-rich fuels including coal and oil in power plants, gasoline and diesel fuel in road vehicles, biomass (e.g. wood, agricultural and animal waste, wildfires) (Bond et al., 2007). Transportation is a major source of BC in urban environments where road vehicles have been reported to contribute ~58% of total BC emissions (Reddy and Venkataraman, 2002) and lead to high BC concentrations on and near busy roads (Apte et al., 2011; Dons et al., 2012; Patton et al., 2016; Quincey et al., 2009; Rivas et al., 2017a). Diesel-fueled cars and trucks usually emit more BC per unit of fuel (1.3–3.6 g BC/kg fuel) than rail and shipping (0.34–0.51 g BC/kg fuel) (Bond et al., 2004). As of 2012, the transportation sector was the third largest source of BC emissions in Asia (behind residential and open burning), and it contributed 21% of BC emissions in India. Transportation is expected to become the second largest source of BC in Asia by 2030 (USEPA, 2012b). Short- and long-term exposure to BC is also associated with adverse cardiovascular effects and mortality (Delfino et al., 2010; McCracken et al., 2010; Wilker et al., 2010) as well as respiratory effects (Patel et al., 2013), decreases in childhood cognition (Suglia et al., 2008), and ventricular arrhythmias (Baja et al., 2010; Rich et al., 2005).

Four recent studies based in Shanghai (China) provide evidence for associations between exposure to BC and health effects. Based on a panel study of 33 adults, FeNO was found to be most strongly and significantly associated with the EC fraction of $PM_{2.5}$, whereas other $PM_{2.5}$ components such as organic carbon and various trace ions were not significant (Shi et al., 2016). Specific $PM_{2.5}$ constituents, including EC, were associated with FeNO among COPD patients (Chen et al., 2015). There were significant associations between ambient levels of BC and daily hospital visits, especially for emergency room visits, and the effect for BC was significant even after adjusting for $PM_{2.5}$ or coarse particles ($PM_{2.5\text{-}10}$) (Wang et al., 2013). Ambient BC concentrations were significantly associated with cardiovascular and respiratory mortality (Janssen et al., 2011).

4. Factors impacting exposure in TMEs

Personal exposure in TMEs depends on factors such as the choice of transport mode, traffic flow conditions, the extent of emissions, background concentrations, the position of breathing zone, personal behavior and choice, and meteorological conditions. Exposure during *walking* depends on the individuals themselves – their location relative to road or personal behavior, their height (adults and children have different heights and breathing zones, leading to a different personal exposure) (Garcia-Algar et al., 2015) – and choice of the route while walking (Garcia et al., 2014; Kumar et al., 2018). For instance, some people avoid walking in crowded areas and close to busy roads, traffic,

or smokers while others may not perceive these factors to be important for their personal exposure (Kaur et al., 2005a).

Exposure during *cycling* is influenced by the position of the cyclist relative to traffic, the height of a cyclist's breathing zone relative to tailpipe emissions from vehicles, and whether the bicycle travel lane is protected from motor vehicle traffic (Bigazzi et al., 2016; Dons et al., 2012; Kaur et al., 2007; MacNaughton et al., 2014). Air pollution exposures during cycling also depend on the route and traffic conditions, with higher exposures usually occurring in and near busy traffic and lower exposures in dedicated lanes farther from the main roads (CAFEH, 2015; Karanasiou et al., 2014; MacNaughton et al., 2014). Other factors that are common for most TMEs include traffic volume (Goel and Kumar, 2015; Rakowska et al., 2014) and the number of traffic signals (Karanasiou et al., 2014). For example, pedestrian and cyclist exposures to UFP and BC on high-traffic routes have been reported to be up to 60% higher compared with low-traffic routes (Kaur et al., 2005b; Strak et al., 2010).

Personal exposure while driving in a *car* depends on the (front/back) seat position (Chan and Chung, 2003) and personal behavior (Kaur et al., 2007; Weijers et al., 2004), such as whether the air conditioner is on or air vents and/or windows are open (Fruin et al., 2008; Hudda et al., 2012; Jiao and Frey, 2013; Kumar and Goel, 2016; Patton et al., 2016), the speed of the vehicle (Hudda et al., 2012; Karanasiou et al., 2014) and the distance maintained from preceding vehicles (Goel and Kumar, 2016; Sabin et al., 2004). In-cabin exposures in a vehicle when windows are closed also depend on whether fresh intake air or recirculated cabin air are used with the heating, ventilating, and air conditioning (HVAC) system (Goel and Kumar, 2015; Jiao and Frey, 2013; Joodatnia et al., 2013a).

In *buses*, personal behavior and personal choices such as seat position (front/back, upper/lower deck) affect the level of personal exposure (Choi et al., 2018; Kaur et al., 2007; Yan et al., 2015). Bus commuters tend to have higher personal exposures than pedestrians and cyclists (Kaur et al., 2007; Rivas et al., 2017a, 2017b), possibly because pedestrians and cyclists have more flexibility in their choice of position to avoid congested traffic conditions (Gee and Raper, 1999). Furthermore, in-cabin exposures in buses are affected by infiltration of near-source ambient pollution when doors open at bus stops (Jiao and Frey, 2014). Semi-enclosed bus terminals can also have relatively high levels of particles (Che et al., 2016).

Therefore the choice of transport modes, characteristics of the transportation system (e.g., ventilation type, fuel type, vehicle model (year and design) and ventilation settings (e.g., windows closed/open, AC on/off) are a few dominating factors influencing the personal exposure in TMEs (de Nazelle et al., 2017; Goel and Kumar, 2014; Joodatnia et al., 2013b; Knibbs et al., 2011). Air exchange rates for vehicles with open windows are much higher than for other enclosed environments, leading to exposure to pollutants from surrounding vehicles (Knibbs et al., 2011). Open windows tend to have higher exposure level during the congested traffic conditions because outdoor air from busy traffic can enter the vehicle cabins (Hudda et al., 2012; Kumar and Goel, 2016). Other key factors affecting bus passenger exposures include the higher number of passengers inside the buses, background concentrations of pollutants inside, airtightness of vehicles (old/new), cabin filtration, and frequency of door opening at traffic hotspots and locations of bus stops (Choi et al., 2018; Geiss et al., 2010; Ott et al., 2008; Zuurbier et al., 2010).

Meteorological conditions also influence exposure levels in TMEs. Wind speed has a significant impact on personal exposure as wind helps to dilute the fresh emissions and reduce pollutant concentrations by the time pollutants reach receptors (Adams et al., 2001a; Karanasiou et al., 2014). Not only do concentrations of particulate pollutants near sources decrease faster with higher wind speed (Adams et al., 2001a; Karanasiou et al., 2014; Padró-Martínez et al., 2012), but so do pollutant concentrations in buses and cars (Gómez-Perales et al., 2004; Rivas et al., 2017a). For example, Rivas et al. (2017b) reported a decrease of 2.90 μg m^{-3} of $PM_{2.5}$ concentrations in London buses for every 1 m s^{-1} increase in wind speed. Wind direction is another important factor, which becomes significant when it comes to street canyons. For example, depending on relationships between street canyon height and width (i.e., aspect ratios), a vortex generated within a street canyon can push pollutants from the windward to the leeward side of the street canyon (Kumar et al., 2009, 2011b; Vardoulakis et al., 2003; Garcia et al., 2013) resulting in increased exposure for pedestrians and cyclists (Kaur et al., 2007). Ambient, near-road, and TME pollutant concentrations tend to vary seasonally, with UFP typically highest in cold months when condensation of gases into liquids is greatest and $PM_{2.5}$ is higher in spring and summer due to the secondary aerosol formation (Kaur et al., 2006). However, assessment of the detailed effects of meteorological conditions on air pollution exposure in TMEs is outside of the scope of this review.

5. Exposure to $PM_{2.5}$, UFP, and BC during commuting in different TMEs

5.1. Walking

Walking is one of the most common modes of urban transport. In Asian cities, footpaths are usually flat sidewalks alongside roads. Similar to American cities and unlike many European cities, there are usually no separation barriers in the form of low boundary walks or green vegetation barriers between the road and the footpaths (Abhijith et al., 2017), resulting in exposure of pedestrians to fresh exhaust emissions. A recent study in Singapore reported walking as the worst commuting mode for particle exposure (Tan et al., 2017). Average concentrations of $PM_{2.5}$, UFP, and BC reported for the walking TME are shown in Fig. 1. Pedestrians in Asian cities were exposed to concentrations ranging from ~20 μg m^{-3} on busy roadsides in residential areas in Hong Kong (Yang et al., 2015a) to ~90 μg m^{-3} in Istanbul, Turkey (Onat and Stakeeva, 2013), as seen in Fig. S1. These concentrations were 1.2- to 1.6-times higher than those in European and North American cities (Table 2). Near a busy road in a commercial part

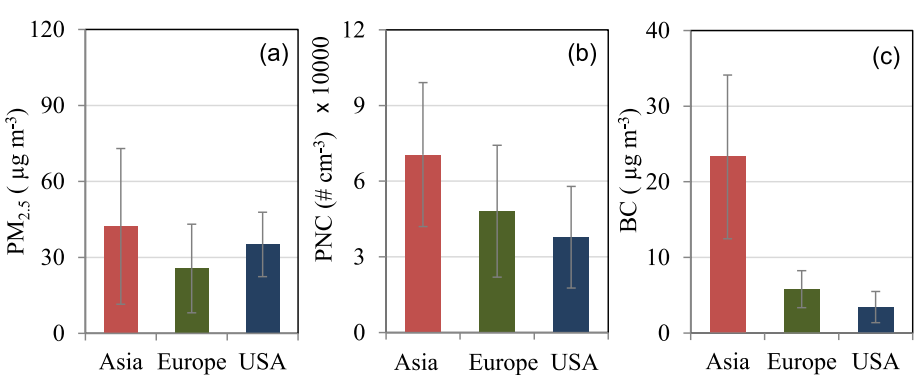

Fig. 1. Concentrations of (a) average $PM_{2.5}$, (b) PNC and (c) BC concentrations while walking along busy roadways in Asian cities (red) compared with those in Europe (green) and the USA (blue). The average values and standard deviations represent the summary of studies presented in Fig. S1 ($PM_{2.5}$), S2 (PNC) and S3 (BC). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

Table 2
Summary of exposure concentrations experienced in various TME studies; *n*, SD and "–" refer to the total number of studies considered, standard deviation and no values available due to limited studies, respectively. Overall mean represents the average concentrations of all individual studies in a given category. The mean values for $PM_{2.5}$, UFP and BC are the average concentrations of all studies (*n*) in a particular TME and the SD is standard deviation of all average values of individual studies.

	$PM_{2.5}$ (μg m^{-3}) Mean ± SD (*n*)[a]	UFP (× 10^4 # cm^{-3}) Mean ± SD (*n*)[b]	BC (μg m^{-3}) Mean ± SD (*n*)[c]
Walk			
Asia	42 ± 31 (5)	6.65 ± 2.89 (7)	23 ± 11 (4)
Europe	26 ± 18 (11)	4.81 ± 2.61 (13)	6 ± 2 (5)
USA	35 ± 13 (10)	3.78 ± 2.01 (3)	3 ± 2 (4)
Overall mean	32 ± 19 (26)	5.29 ± 2.74 (23)	10 ± 11 (13)
Ratio (Asia to Overall mean)	1.3	1.3	2.3
Cycle			
Asia	49 ± 27 (1)	–	–
Europe	43 ± 27 (7)	3.37 ± 2.21 (12)	6 ± 3 (3)
USA	11 ± 5 (1)	3.50 ± 3.77 (1)	3 ± 1 (1)
Overall mean	40 ± 28 (9)	3.38 ± 2.12 (13)	6 ± 3 (4)
Ratio (Asia to Overall mean)	1.2	–	–
Car			
Asia	74 ± 61 (7)	45.7 ± 28.1 (3)	43 ± 12 (1)
Europe	32 ± 30 (15)	6.76 ± 7.89 (17)	14 ± 7 (3)
USA	46 ± 36 (4)	5.29 ± 4.29 (12)	7 ± 3 (1)
Overall average	40 ± 28 (26)	9.86 ± 15.1 (32)	18 ± 15 (5)
Ratio (Asia to Overall mean)	1.9	4.6	2.4
Bus			
Asia	76 ± 62 (5)	3.45 ± 1.34 (2)	10 ± 2 (2)
Europe	47 ± 37 (7)	5.15 ± 3.54 (8)	7 ± 1 (4)
USA	59 ± 59 (2)	5.05 ± 1.93 (11)	3 ± 2 (1)
Overall mean	59 ± 48 (14)	5.10 ± 2.59 (21)	7 ± 3 (7)
Ratio (Asia to Overall mean)	1.3	0.67	1.4
Auto-rickshaw[d]			
Asia	139 ± 86 (2)[e]	29.0 ± 7.70 (1)[f]	47 ± 6 (2)[e]
Motorcycle			
Asia	86.3 ± 55.7 (3)[g]	–	10.4 ± 11.2 (2)[h]
Europe	–	7.3 ± 5.0 (1)[i]	–

[a] These are mean values taken from the studies shown in Table S1 ($PM_{2.5}$).
[b] These are mean values taken from the studies shown in Tables S2 (UFP).
[c] These are mean values taken from the studies shown in Tables S3 (BC), respectively.
[d] No studies available for Europe and USA.
[e] Average of Apte et al. (2011) and Pant et al. (2017).
[f] Apte et al. (2011).
[g] Average of Morales Betancourt et al. (2017), Pant et al. (2017) and Ramos et al. (2016).
[h] Average of Pant et al. (2017) and Jeong and Park (2017).
[i] Grana et al. (2017).

of Hong Kong, pedestrians were exposed to concentrations as high as 4.0×10^4 particles cm^{-3} (Yang et al., 2015a), or as much as 1.8-times the pedestrian UFP exposures in Europe and North America (Fig. S2). Likewise, pedestrians in Hong Kong were exposed to BC concentrations ranging from 15 μg m^{-3} (Yang et al., 2015a) to 39 μg m^{-3} (Rakowska et al., 2014; Yang et al., 2015a), or 3.8 to 7.7 times those in cities in Europe (Dons et al., 2012; Moreno et al., 2015) and the USA (Kinney et al., 2000), as seen in Fig. S3.

5.2. Cycling

Cycling is a popular means of transport in Asian cities. About half of the households in Asia own at least one bicycle (Oke et al., 2015). Huang et al. (2012) reported that cyclists were exposed to $PM_{2.5}$ concentrations of 49 μg m^{-3} in Beijing, which is comparable to $PM_{2.5}$ levels reported for European cities and higher than those reported in American cities (Fig. 2). No reports of BC or UFP exposures to cyclists in Asian cities were found. Cycling exposures to both BC and UFP were high in studies conducted in Europe (de Nazelle et al., 2012; Dons et al., 2012), and the USA (Hankey and Marshall, 2015; Zuurbier et al., 2010) (Table S1), and could be even higher in Asian cities due to high ambient levels. A recent study by Cepeda et al. (2017) concluded that commuters using motorised transport such as a car, motorcycle or bus lost up to one year more in years of life expectancy compared with cyclists, favoring an active mode of transport even though air pollutant exposures may increase.

5.3. Car

Cars are a popular means of transport in Asian cities. Car models and use of air conditioning in Asia are usually similar to the common practice in Europe or USA. However, fuel and emission standards vary widely across Asia, and standards in some of the most populated countries (i.e., India and China) are less stringent than comparable standards in the USA and Europe (see Section 8). Reported concentrations of $PM_{2.5}$ (Fig. 3), BC (Table 2), and UFP (Fig. 4) were higher in Asian cities than in Europe and USA. The average $PM_{2.5}$ concentration in Asian cars was about twice the concentration in other locations. Apte et al. (2011) reported that $PM_{2.5}$ concentrations in cars in New Delhi were as high as 190 μg m^{-3}, which was about 5-times the average in-car $PM_{2.5}$ concentration across all studies we compared (Fig. 3). In Hong Kong, in-vehicle UFP concentrations were ~8×10^5 cm^{-3} (Both et al., 2013; Kaminsky et al., 2009), or 2- to 4-times higher than cities in Europe (Fig. 4). The only in-car BC measurements available for Asia were made in New Delhi (Apte et al., 2011). The average concentration measured, 43 μg m^{-3}, was 2–5 times in-car concentrations in Europe (de Nazelle et al., 2012; Dons et al., 2012; Zuurbier et al., 2010) and North America (Greenwald et al., 2014; Patton et al., 2016; Weichenthal et al., 2015).

5.4. Bus

Bus routes, particularly for bus rapid transit, are being considered

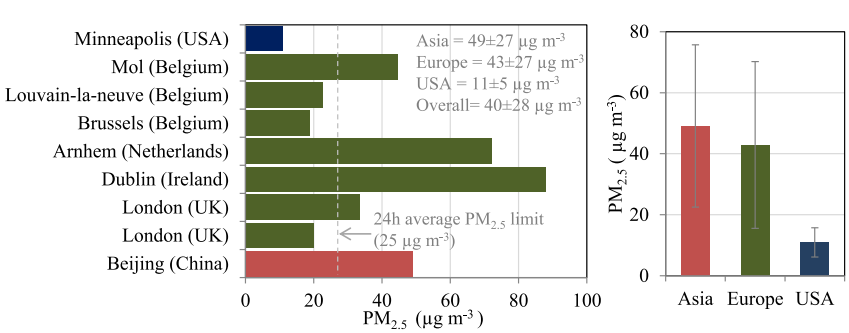

Fig. 2. Average $PM_{2.5}$ concentrations while cycling along busy roadways in Asian cities (red) compared with those in Europe (green) and the USA (blue). Error bars represent the standard deviation of the mean values of studies shown in Table S1. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

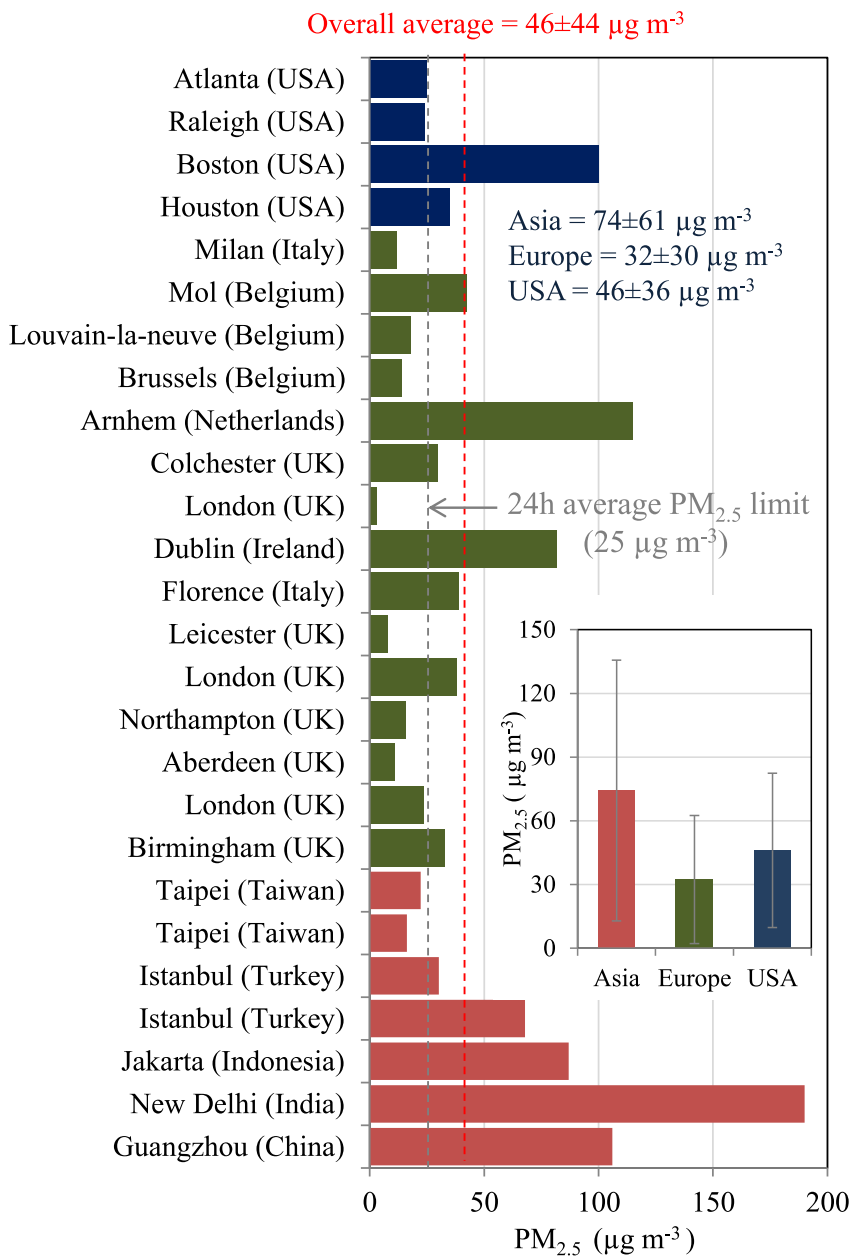

Fig. 3. Average $PM_{2.5}$ concentrations during car journeys in Asian cities (red) compared with those in Europe (green) and the USA (blue). Details of studies are available in Table S1. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

and implemented in Asian cities to decrease traffic congestion (ASIA-LEDS, 2014). While a small fraction of double-decker buses ply on roads of many Asian cities (e.g., Bangalore, Mumbai, Dhaka, Beijing and Singapore), most buses are still single-deck (Ollson and Thynell, 2006; Yang et al., 2015a). Within the bus rapid transit systems of Asian cities, buses are typically standard or articulated vehicles of unreported age and fueled by diesel, condensed or liquified natural gas (CNG/LNG), or hybrid systems (BRT Data, 2018). The literature shows that bus travellers in Asia experience large variability in $PM_{2.5}$ concentrations: from 10 $\mu g\,m^{-3}$ in parts of Hong Kong (Yang et al., 2015a) to 145 $\mu g\,m^{-3}$ in Guangzhou, China (Chan et al., 2002b) (Fig. 5). $PM_{2.5}$ concentrations in buses in Asia were about 1.5-times the $PM_{2.5}$ concentrations in European cities (Adams et al., 2001b; Dennekamp et al., 2002; Fondelli et al., 2008; Kaur et al., 2005b; Spinazzè et al., 2013), and North America (Adar et al., 2007; Gomez-Perales et al., 2004; Gómez-Perales et al., 2005; Levy et al., 2002; Onat and Stakeeva, 2013). Measurements of UFP number concentration (Fig. 6) and BC (Table 2) were more limited for buses in Asian cities; these two pollutants have only been measured in buses in Hong Kong. UFP concentrations in Hong Kong buses ranged from 25,000 particles cm^{-3} in LPG buses to 44,000 particles cm^{-3} in diesel buses and BC concentrations ranged from 7.5 $\mu g\,m^{-3}$ in LPG buses to 11.6 $\mu g\,m^{-3}$ in diesel buses (Yang et al., 2015a). These concentrations were comparable to in-bus concentrations in non-Asian cities (Table 2).

5.5. Open-air vehicles

Motorcycles and auto-rickshaws are open-air vehicles that are very common in Asian cities. There are only a handful of studies for assessing motorcyclists' exposure in Asia (Li et al., 2017; Pant et al., 2017), Europe (Grana et al., 2017), or elsewhere (Morales Betancourt et al., 2017; Ramos et al., 2016). Some studies have reported that motorcyclists have much higher exposures to UFPs and $PM_{2.5}$ than commuters experience in cars, buses, or trains. In the city of Taipei (Li et al., 2017),

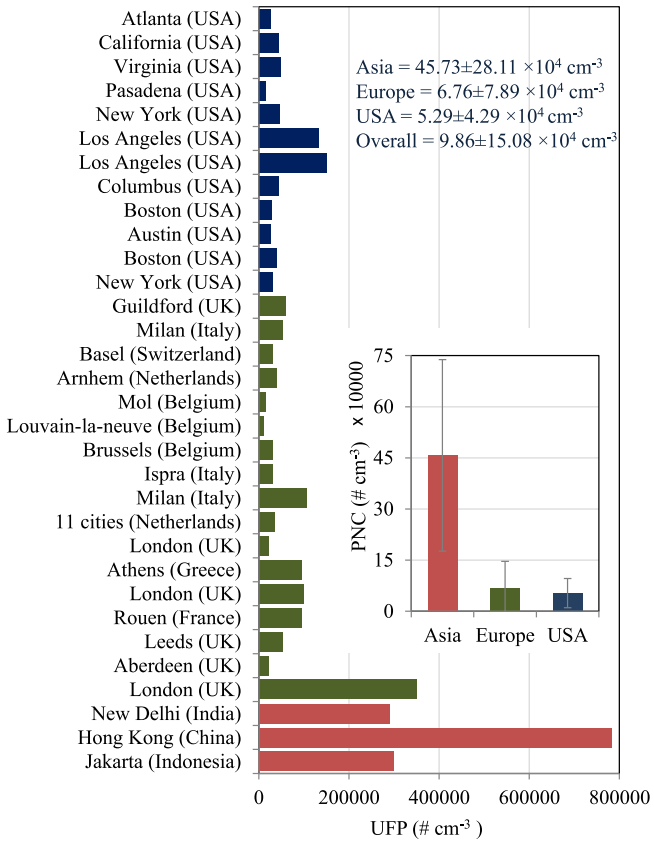

Fig. 4. Average UFP concentrations during car journeys in Asian cities (red) compared with those in Europe (green) and the USA (blue). Details of studies are available in Table S2. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

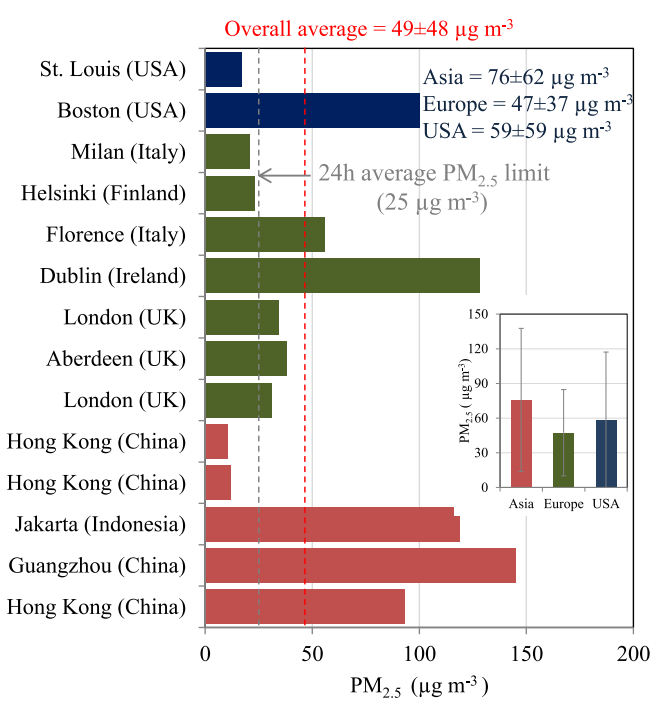

Fig. 5. Average $PM_{2.5}$ concentrations during bus journeys in Asian cities (red) compared with those in Europe (green) and the USA (blue). Details of studies are available in Table S1. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

motorcyclists had shorter commutes but higher overall $PM_{2.5}$ exposures than car or bus commuters, because the $PM_{2.5}$ concentrations breathed by the motorcyclists were much higher than in the other TMEs ($67.5 \pm 31.3\,\mu g\,m^{-3}$ versus $22.1 \pm 9.6\,\mu g\,m^{-3}$ for car and $38.5 \pm 15.6\,\mu g\,m^{-3}$ for bus) (Tsai et al., 2008). For example, the contribution of transport modes to daily integrated UFP concentration was $7.3 \times 10^4\,cm^{-3}$ for motorcycle trips compared with $1.4 \times 10^4\,cm^{-3}$ in underground trains and the contribution of transport modes to daily integrated UFP exposure was ~29% for motorcycle trips compared with 16–21% for car trips and only ~9% for trips on the trains (Grana et al., 2017). Studies for auto-rickshaw exposure in Asian cities are rare (Apte et al., 2011). The findings of these studies suggest that concentrations of $PM_{2.5}$, UFP, and BC were all high and similar to levels in open-window cars, bicycles, or auto-rickshaws, suggesting that additional measurements are needed to better quantify contributions of motorcycles and auto-rickshaws to personal and population-level exposures.

5.6. Comparison of $PM_{2.5}$, UFP, and BC in different TMEs

Measured $PM_{2.5}$ exposures in Asian cities were highest in open-aired vehicles (auto-rickshaw and motorcycle) and lowest for active travel modes (i.e., walking and cycling) (Table 2). However, depending on trip length, cyclists may receive higher total exposures to PM than those experienced by pedestrians due to a combination of higher ventilation (i.e., breathing) rates during more intense exercise and higher concentrations of air pollutants if surrounded by vehicular traffic (Kaur et al., 2006; O'Donoghue et al., 2007). Based on sequential measurements with portable $PM_{2.5}$ monitors in Hong Kong TMEs, concentrations were consistently higher at bus stops, for pedestrians, on the tram, at tram stops, and at bus terminals compared to onboard single-decker buses, double-decker buses, minibuses, or the MTR subway system (Yang et al., 2015a). The comparison among these TMEs was similar on a relative basis for mid-day and evening rush hour. Concentrations experienced by pedestrians were comparable to those measured at roadside fixed-site monitors. The mean concentrations among TMEs varied by more than a factor of two, implying that exposures to commuters can be highly sensitive to mode choice (Che et al., 2016).

Measured UFP concentrations in Asian cities were highest in the walking TME and lowest in the auto-rickshaw TME. Concentrations of UFP experienced by pedestrians in Asian cities were about twice the concentrations in cars and buses. However, since we did not find papers reporting UFP exposure while cycling in Asian cities, we are not able to determine the rank of cycling concentrations among the TME exposures to UFP. BC exposures in Asian cities were highest in auto-rickshaws and cars, and about half as high in pedestrian and bus TMEs. However, similar to UFP, we did not find studies reporting BC exposure while cycling in Asian cities to rank exposures to BC in Asian TMEs.

6. Instruments used for measuring $PM_{2.5}$, UFP and BC in TMEs

Here, we review the monitoring equipment and data collection strategies used in TME studies performed in Asian cities. The results are summarized in Table 3; instruments used in TME studies performed in Europe, America, and Australia are listed in Table S4.

We identified 11 studies that measured $PM_{2.5}$ in TMEs in Asian cities (Table 3). Several different instruments were used in these studies: DustTraks (TSI, Shoreview, MN, USA) were used to measure $PM_{2.5}$ in walking and personal-car TMEs in Jakarta (Indonesia) (Both et al., 2013), in buses in Hong Kong (China) (Chan et al., 2002a), in personal cars and buses in Guangzhou (China) (Apte et al., 2011; Chan et al., 2002b), and in auto rickshaws in New Delhi (India) (Apte et al., 2011). Onat and Stakeeva (2013) used both a Lighthouse optical particle counter and a personal dataRAM (pDR) 1200 (Thermo, Franklin, MA, USA) to measure $PM_{2.5}$ in walking, bus, and personal-car TMEs in Istanbul (Turkey). Pant et al. (2017) used a Thermo pDR 1500 in buses

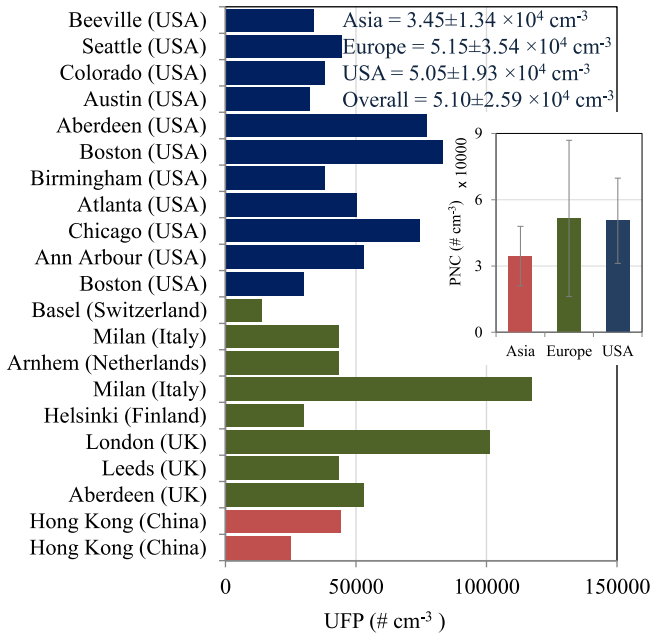

Fig. 6. Average UFP concentrations during bus journeys in Asian cities (red) compared with those in Europe (green) and the USA (blue). Details of studies are available in Table S2. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

and auto-rickshaws in New Delhi. Yang et al. (2015a) used a TSI OPS (model 3330) in walking and bus TMEs in Hong Kong. Tsai et al. (2008) used a GRIMM (model 1.108) in personal cars in Taipei. Gravimetric filtration (i.e., $PM_{2.5}$ in a known volume of air was collected on a filter and the difference in filter weight before and after sampling was determined) was used to quantify $PM_{2.5}$ in personal cars and auto-rickshaws in New Delhi (Saksena et al., 2007).

Based on our literature search, we identified five studies that measured UFP in TMEs in Asian cities (Table 3). In these studies two different UFP monitors were used: the TSI model 3007 and the TSI model 3785. The TSI model 3007 is an isopropanol-based condensation particle counter and the model 3785 is a water-based condensation particle counter. These instruments have been used to measure UFP in walking and personal-car TMEs in Jakarta (Both et al., 2013), and in walking, personal-car and bus TMEs in Hong Kong (Tsai et al., 2008; Yang et al., 2015a; Rakowska et al., 2014; Kaminsky et al., 2009). We identified four Asian TME studies in which BC was measured, and in all of these studies the same monitor was used, the AethLabs microAeth AE51 (AethLabs, San Francisco, CA, USA; Table 3). This instrument was used to measure BC in walking and bus TMEs in Hong Kong (Apte et al., 2011; Yang et al., 2015a), and in buses and auto-rickshaws in New Delhi (Apte et al., 2011; Pant et al., 2017).

As indicated in Table 3, the instruments used to characterize TMEs in Asian cities have differences in terms of measurement frequency, concentration range, particle size resolution (for UFP monitors), clock drift, battery life, and portability. Other considerations not reflected in Table 3, which may impact data quality, include ease of operation, calibration, and measurement artifacts (Kumar et al., 2011c). For example, CPCs – particularly bench-scale instruments – can be prone to flooding when alcohol and water reservoirs are jostled or tilted in moving platforms. When this happens, the data are compromised and the flooded lines and optics need to be purged and dried out before the CPC can be used again. While flooding is less of a problem with field-portable CPCs (e.g., TSI 3007), improvements in portability often come at the expense of measurement sensitivity especially at the lower end of the particle size range (e.g., < 20 nm). Field portable $PM_{2.5}$ monitors that are based on light scattering are relatively easy to operate and can be deployed in moving data-collection platforms because they are less prone to malfunctioning due to tilting (Wang et al., 2009). Nonetheless, one disadvantage to using light-scattering PM monitors is that the optical properties of PM differ with pollution source and particle age. In addition, because optical PM monitors are generally sensitive to ambient temperature and humidity, factory-calibrated monitors must be checked against locally-stationed reference instruments to fully assess their accuracy (Wang et al., 2009). Measurement artifacts are often a problem with the microAeth AE51. This instrument measures the rate of change of light absorption due to the accumulation of black carbon aerosol on a filter; however, the accumulation of BC on the filter decreases the effective optical path resulting in a loading-dependent relationship between the measured absorption and BC mass (specifically, BC concentration is increasingly underestimated as BC load on the filter accumulates). To compensate for this measurement artifact, correction factors must be applied during post-processing of the data (Good et al., 2017).

In the studies listed in Table 3, instruments were deployed fairly consistently for each TME. For example, instruments were either worn by the data collectors or carried in backpacks or suitcases during the characterisation of walking TME. Similarly, instruments were either mounted inside the vehicle or inside backpacks or suitcases carried by the riding data collector during the characterisaton of personal car, bus, and auto rickshaw TMEs. In contrast, the studies in Table 3 show that there were considerable differences in the amounts of data that were collected in different TMEs. For example, many more hours of data were collected in studies characterising personal car and bus TMEs than were in studies of walking TMEs. Also, within each TME, there were differences. For example, in estimating $PM_{2.5}$ exposures in personal cars Saksena et al. (2007) used 0.5–2.2 h of data, Onat and Stakeeva (2013) used 4.4–6.3 h, Tsai et al. (2008) used 7–9 h, Both et al. (2013) used 48–72 h and Chuang et al. (2013) used 360 h of data. Similarly, in characterizing BC exposures in buses, Yang et al. (2015a) used 154 h of measurements while Pant et al. (2017) used 288–576 h of measurements. These differences in study designs – in particular how instruments were deployed in the TME and the amount of data that was collected – has bearing on the overall results of individual TME studies. This is particularly important when synthesising results from multiple studies: care must be taken to distinguish differences that are attributable to the TMEs in question (e.g., walking versus riding in a bus along a busy street during morning rush hour in a city) from those that are attributable to methodological differences in data collection strategies (e.g., instruments that were used, or the duration of the "rush hour" period or the definition of "busy" in terms of vehicles per hour differed between studies).

7. Exposure modeling

Exposure is a function of the frequency, intensity, and duration of contact of a pollutant with the body (EPA, 1992). For air pollution, exposure is typically quantified as the time-weighted concentration experienced by a person moving between microenvironments over the course of a day. Thus, exposure models typically aim to estimate the time-averaged concentration for each exposed individual based on the variation in pollutant concentration over time and space, and the time-activity pattern of each individual (Ott, 1995). Quantification of human exposure to air pollutants is a relatively new field as currently practiced (Ott, 1995), although it traces its origins to related fields such as industrial hygiene, radiation protection, and environmental toxicology (NRC, 2012). Much of the history of quantitative exposure assessment can be traced to work done outside of Asia. However, the concepts and methods are applicable in the Asian context. Furthermore, the concepts embodied in exposure models can be a key motivating factor when designing field measurement studies to collect data on micro-environmental concentrations.

In the last two decades, increased availability of population-based

data regarding activity diaries has been a key factor in enabling the development of a new generation of exposure models that simulate inter-individual variability in exposure. For example, the U.S. EPA developed and has periodically updated the Consolidated Human Activity Database (CHAD) since 2000 (EPA, 2014; Graham and McCurdy, 2004; McCurdy et al., 2000). CHAD is comprised of diaries compiled from multiple studies in which human subjects recorded the amount of time spent in microenvironments on a daily basis. As new technologies have emerged, such as the ubiquitous use of cell phones, there are growing opportunities to develop longitudinal activity databases that can enable more accurate quantification of individual long-term time-location patterns (Breen et al., 2014; NRC, 2012).

Concurrently, data have been developed over the years regarding the relationship between ambient air quality and pollutant concentrations in enclosed microenvironments, such as homes, offices, schools, vehicles, and others. These relationships are typically represented based on mass balance or statistical models. Mass balance models are typically based on quantifying infiltration taking into account air exchange rate,

Table 3
Instruments used for measurement of $PM_{2.5}$, UFP, and BC in transport microenvironment studies in Asian cities.

Instrument[a]/Characteristics[b]	Study	Study location	TME[c]	Location of monitor[d]
$PM_{2.5}$ Monitors[e]				
GRIMM, model 1.108	Tsai et al. (2008)	Taipei (Taiwan)	PC (~7–9 h), B (~10–13 h)	Carried by data collector
• Measurement frequency: 1–6 s				
• Particle size range: 0.23–20 μm	Chuang et al. (2013)	Taipei (Taiwan)	PC (360 h)	Mounted inside PC
• Concentration range: 0.1–100,000 μg/m^3				
• Battery life: 6–8 h (internal battery)				
• Size: 24 × 13 × 7 cm				
• Weight: 2.4 kg				
• Measurement principle: light scattering				
TSI DustTrak, model 8520 *(Discontinued)*	Chan et al. (2002a)	Hong Kong (China)	B (5–17 h)	Mounted inside B
• Concentration range: 1–1.0 × 10^5 μg/m^3	Both et al. (2013)	Jakarta (Indonesia)	W (12–18 h), PC (48–72 h)	Carried by the data collector
• Battery life: 16 h (internal battery)				
• Size: 22 × 15 × 8.7 cm	Chan et al. (2002b)	Guangzhou (China)	B (16–18 h)	Mounted inside B
• Weight: 1.5 kg (inc. battery)				
• Measurement principle: light scattering	Apte et al. (2011)	New Delhi (India)	AR (180 h)	Mounted inside AR; stationary site
TSI OPS, model 3330	Yang et al. (2015a)	Hong Kong (China)	W (1.9–28 h)[g], B (154 h)	Inside suitcase carried by walking data collector; inside backpack worn by data collector on buses
• Measurement frequency: ≤1 s				
• Concentration range: 0.001–2.75 × 10^5 μg/m^3				
• Battery life: < 10 h (internal battery)				
• Size: 14 × 22 × 22 cm				
• Weight: 2.1 kg (inc. battery)				
• Measurement principle: light scattering				
Thermo pDR 1200	Onat and Stakeeva (2013)	Istanbul (Turkey)	W (3.5–5 h), PC (4.4–6.3 h), B (3–6 h)	Carried by walking data collector; mounted inside PC and B
• Measurement frequency: 1 second–4 hours				
• Particle size range: 0.1–10 μm				
• Concentration range: 1–4.0 × 10^5 μg/m^3				
• Battery life: 20 h (alkaline), 40 h (lithium) (internal battery)				
• Size: 16 × 20 × 6.0 cm				
• Weight: 0.68 kg				
Thermo pDR 1500	Pant et al. (2017)	New Delhi (India)	B (288–576 h), AR (288–576 h),	Inside backpack; was worn by data collectors
• Measurement frequency: 1 second–1 hour				
• Particle size range: 0.1–10 μm				
• Concentration range: 1–4.0 × 10^5 μg/m^3				
• Size: 18.1 × 14.3 × 8.4 cm				
• Weight: 1.2 kg				
Lighthouse Worldwide Solutions OPC Handheld, model 3016	Onat and Stakeeva (2013)	Istanbul (Turkey)	W (3.5–5 h), PC (4.4–6.3 h), B (3–6 h)	Carried by walking data collector; mounted inside PC and B
• Internal battery				
• Size: 22 × 13 × 6.4 cm				
• Weight: 1 kg				
• Measurement method: laser particle counter				
Gravimetric filtration and personal sampling pumps[f]	Saksena et al. (2007)	New Delhi (India)	PC (0.5–2.2 h), AR (1.3–2.2 h)	Mounted inside PC, AR
UFP Monitors				
TSI CPC, model 3007	Yang et al. (2015a)	Hong Kong (China)	W (1.9–28 h)[g], B (154 h)	Inside suitcase carried by walking data collector; inside backpack worn by data collector on buses
• Measurement frequency: 1–300 s				
• Size range: 0.01 – > 1.0 μm				
• Max concentration: 10^6 particles/cm^3	Rakowska et al. (2014)	Hong Kong (China)	W (4 h)	Inside a suitcase carried by the data collector
• Battery life: 5 h at 21 °C (internal battery)				
• Size: 29 × 14 × 14 cm	Both et al. (2013)	Jakarta (Indonesia)	W (12–18 h), PC (48–72 h)	Hand carried by the data collector
• Weight: 1.7 kg				
• Measurement method: light scattering				
TSI CPC, model 3785 *(Discontinued)*	Tsang et al. (2008)	Hong Kong (China)	W (4 h)	Stationary site
• Measurement frequency: 1–3600 s	Kaminsky et al. (2009)	Hong Kong (China)	PC (3 h)	Mounted inside PC
• Size range: 0.005 – > 3 μm				
• Max concentration: 3 × 10^4 particles/cm^3 (single particle counting), 10^4–10^7 particles/cm^3 (photometric counting)				
• Size: 31 × 16 × 28 cm				
• Weight: 5.5 kg				
• Measurement method: light scattering				

(*continued on next page*)

Table 3 (*continued*)

Instrument[a]/Characteristics[b]	Study	Study location	TME[c]	Location of monitor[d]
BC Monitors				
AethLabs Micro-aethalometer, model AE51	Yang et al. (2015a)	Hong Kong (China)	W (1.9–28 h)[g], B (154 h)	Inside suitcase carried by walking data collector; inside backpack worn by data collector on buses
• Measurement frequency: 1–300 s				
• Measurement range: 0–1000 μg/m^3				
• Battery life: 24 h (depending on concentration) (internal battery)	Apte et al. (2011)	New Delhi (India)	AR (180 h)	Mounted inside AR; stationary site
• Size: 18 × 6.6 × 3.8 cm	Rakowska et al. (2014)	Hong Kong (China)	W (4 h)	Inside a suitcase carried by walking data collector
• Weight: 0.28 kg	Pant et al. (2017)	New Delhi (India)	B (288–576 h), AR (288–576 h)	Inside backpack; was worn by data collectors
• Measurement method: light absorption				

[a] Manufacturer and model number. Instruments/articles were only included if sufficient information was provided about the instrument to find a website for it on the internet.

[b] Instrument characteristics were taken from manufacturer's websites. Information included measurement range, particle size, measurement frequency, measurement principle, accuracy and precision. Note that some specifications were not reported by the manufacturers for many instruments (e.g., accuracy and precision). All manufacturer websites were accessed in June and July 2017.

[c] CY = cycling, W = walking, PC = passenger car, B = bus, AR = auto rickshaw; (total number of hours monitored in each TME).

[d] Location of the monitor during data collection.

[e] Huang et al. (2012) measured $PM_{2.5}$ in B and CY TMEs using a portable aerosol spectrometer (model LD-6S, Beijing Green Technology Digital Co., China). Specifications for the monitor could not be found. The instrument was worn by data collectors on buses (18–30 h) and by cyclists (24–25 h).

[f] Measured mass concentration of particles < 5 μm in diameter (d_{50}).

[g] Estimated by multiplying the time a data collector spent waiting for a bus or MTR by the number of successful monitoring trips over the study.

penetration factor, and deposition rates. Statistical methods are typically based on estimating the ratio of indoor-to-outdoor concentrations as a single parameter or as part of a linear regression model in which the intercept term accounts for non-ambient pollution of indoor origin (EPA, 2017). Other key data that enable the development of exposure simulation models include spatially and temporally resolved ambient monitoring data or predictions of air quality models such as the Community Multiscale Air Quality (CMAQ) modeling system, population demographic data at high spatial resolution such as from census data, and data on distributions of other exposure factors such as housing type and age (Baxter et al., 2013a; Burke et al., 2001; EPA, 2017). Much of this work has developed in the U.S. context but the concepts are transferable to other countries. For example, measured ambient and in-vehicle concentrations in multiple transport microenvironments in Hong Kong were used to quantify the ratio of in-vehicle to ambient concentrations (Che et al., 2016).

A variety of exposure models have emerged in the last several decades that are intended to predict inter-individual variability in

Table 4
Description of selected air pollution exposure models (de Bruin et al., 2004; Jensen et al., 2001; Johnson et al., 1999; Klepeis, 2006; Kruize et al., 2003; Law et al., 1997a; Ott et al., 1988; WHO, 2005a).

Model	Description
SHAPE	• Simulation of Human Activity Patterns and Exposure • Detail time-activity model (stay duration in each TME) • Can generate a large variety of possible time-activity patterns • Can assess the changes in the population time-activity in long run, or after particular changes in air quality and social behavior.
APEX	• Air Pollutants Exposure Model • Can be applied at local, urban, consolidated metropolitan • Simulates personal movement (time, space, and exposure of pollutants in different MEs)
HAPEM-MS	• Hazardous Air Pollutant Exposure Model for Mobile Sources • Can assess exposure from highway mobile sources • Can predict inhalation exposure concentrations of air pollutants from outdoor sources, based on ambient concentrations • Can assess risk of screening-level inhalation • Data in ambient air concentration, indoor/outdoor concentration relationship, population, and population activity pattern by calculating routines series • Population activity pattern data is for estimating an expected range of "apparent" inhalation exposure concentration of a group or an individual.
SHEDS	• Stochastic Human Exposure and Dose Simulation Model • Based on stochastic model • Can quantify exposure and dose of human inhalations, multimedia, multi pathway pollutants. • Consider combination of physical/mechanistic algorithms to estimate exposure • Focus on simulation exposures to PM and air toxics, and aggregate exposures.
RISK	• Descendant of EXPOSURE and INDOOR models. • Simulates multi-zone indoor air concentrations, individual exposure, and risk
MCCEM	• Multi Chamber Chemical Exposure Model • Can estimate average and peak indoor air concentrations of chemicals released from products or materials in houses, apartments, townhouses, or other residences • Can estimate inhalation exposures to above-noted chemicals as single day dose, chronic average daily dose or lifetime average daily dose
pNEM/CO	• Probabilistic Exposure Model (pNEM) models • Can estimate frequency distributions of population exposure to carbon monoxide (CO) and the resulting carboxyhemoglobin (COHb) levels
CONTAM	• Multi zone indoor air quality and ventilation analysis model to determine air flows, contaminant concentrations, and personal exposure • Can predict personal exposure of occupants to airborne contaminants for eventual risk assessment
EXPOLIS	• European Population Particle Exposure Model • Can evaluate mean air pollution exposures in different scenarios, population groups, and locations and to understand the factors affecting exposure levels
AirGIS	• Geographic Information System (GIS) based modeling system to estimate traffic-related air pollution for human exposure. • The National Environmental Research Institute in Denmark's model that enables mapping of emission and air pollution exposure.

exposure concentrations taking into account the time that each individual spends among microenvironments over the course of a day, ambient air quality, infiltration of ambient air to enclosed microenvironments, characteristics of enclosed microenvironments such as air exchange rates, and population demographics. For example, the U.S. Environmental Protection Agency has developed two stochastic population-based models: the Stochastic Human Exposure and Dose Simulation (SHEDS) model; and the Air Pollution EXposure model (APEX) (Burke et al., 2001; EPA, 2017). APEX is a successor to the probabilistic NAAQS Exposure Model for carbon monoxide (pNEM/CO) that was one of the first probabilistic exposure models (Law et al., 1997b). A key feature of models such as SHEDS and APEX is that they use probabilistic simulation to "select" a synthetic "individual" from a census database, to find a suitably matched time-location diary, and to estimate the micro-environmental concentrations for the microenvironments in which the simulated individual spends time. The simulation is related to a user-specified number of synthetic individuals. The simulated frequency distributions of exposure to the synthetic individuals have characteristics similar to an actual population in terms of demographics, activity, and exposures. Simulation results can be used to assess exposures over time and for a specified geographic domain based on population means, high-end exposures (e.g., 90th percentile or higher), spatial hotspots for exposure, and so on. APEX has been used to quantify inter-individual variability in exposure to carbon monoxide, nitrogen dioxide, sulfur dioxide, and ozone as part of the scientific review process for the U.S. National Ambient Air Quality Standards (EPA, 2008, 2009b, 2010, 2014). These assessments are often used to determine the proportion of individuals who have exposures greater than a health-based exposure benchmark concentration. APEX has also been considered for use in the scientific review of the PM NAAQS (EPA, 2009a), and has been previously applied to quantify source-to-dose assessment of exposure to PM (Georgopoulos et al., 2005). Predicted exposures using SHEDS, such as for particulate matter, have been found to be statistically associated with adverse outcomes and potentially can reduce exposure errors in epidemiologic studies (Jones et al., 2013; Mannshardt et al., 2013; Özkaynak et al., 2013). The latter are typically based on using fixed-site monitor ambient concentration, rather than individual exposure. Table 4 lists a variety of exposure models, including SHEDS, APEX, and others, that have been developed and applied over the years to assess air pollutant exposures. Models such as SHEDS and APEX have also been used to assess the contribution of transportation microenvironments to daily exposures (Liu and Frey, 2011).

Accurate quantification of exposure factors is critical to developing stronger associations between exposure and metrics of adverse effects in epidemiologic studies, particularly for exposure factors that may differ between communities. For example, mean in-vehicle commuting distance and time were significant effect modifiers for respiratory and non-accident mortality (Baxter et al., 2013b). The effect was negative, implying that longer commutes are associated with lower adverse outcomes. This counter-intuitive finding might be because persons with longer commutes spend more time in suburban areas with better air quality than a central business district because in-cabin exposure concentrations can be much lower than on-road concentrations, or combinations of these. Differences in population demographics may also be important. A recent exposure modeling study for $PM_{2.5}$ in New Delhi focused on population mobility and transport-related exposures using a probabilistic framework that included a land use regression component and a zonal model to simulate home and work locations and commuting between them (Saraswat et al., 2016).

8. Regulatory implications

There are no specific regulations to control air quality within TMEs. Given that the ambient concentrations are affected by road transport emissions that penetrate into TMEs, it is important to bring relevant legislation and incentives to control emissions for human health protection (Colvile et al., 2001).

Various regulatory bodies have established regulatory values for ambient $PM_{2.5}$ concentrations (Table S1) but the parallel values for the BC and UFP are currently non-existent (Kumar et al., 2010, 2016). For example, the United States Environmental Protection Agency (US EPA) issued the first $PM_{2.5}$ standards of 15 μg m^{-3} (annual mean) and 65 μg m^{-3} (24-h mean) in 1997 based on the results of numerous health studies (USEPA, 2012). These standard values were renewed in 2006 with a 35 μg m^{-3} 24-h mean standard, and also with tightening of the annual mean standard from 15 μg m^{-3} to 12 μg m^{-3} in 2012. In the European Union (EU), $PM_{2.5}$ standards were first promulgated in 2008 as 25 μg m^{-3} (annual mean) (AQEG, 2012) as opposed to corresponding values of 10 μg m^{-3} by World Health Organization (WHO) guidelines (WHO, 2006), as seen in Fig. 7. While the WHO provides guideline values for 24-h mean as 25 μg m^{-3}, the EU does not have the corresponding values.

Many Asian countries have air quality standards; however, these are often directly adopted from US EPA, WHO or EU regulations or adjusted according to local conditions such as the traffic fleet, fleet types, geography or meteorology (Table S5). Nonetheless, there are still a number of developing countries in Asia (e.g. Afghanistan, Bhutan) where such regulations are unavailable (CAI-Asia, 2010). A number of Asian countries such as Thailand, India, Hong Kong, and Vietnam have national ambient air quality standards (NAAQS) in place; however, in most cases, these standards are usually for PM_{10}; a few countries also regulate $PM_{2.5}$. As expected, ambient air quality regulations for UFP and BC do not exist despite toxicological evidence showing adverse health impacts on human health, more than $PM_{2.5}$ or PM_{10}.

It is interesting to note the differences in $PM_{2.5}$ regulations between Asian countries and those in Europe or the USA. The allowable ambient $PM_{2.5}$ concentration limits are usually up to 3 times higher in Asian countries. The relatively relaxed standards in Asian cities could be seen as an underliining reason for elevated exposures in Asian TMEs. The air quality regulations are meant to control ambient pollutant concentrations; these do not actually represent the human exposure that varies substantially between different TMEs, as discussed in Section 5. While there is a substantial drive to measure the personal exposure of individuals within urban environments, there seem to be no plans worldwide to develop TME-specific regulations or compliance, at least

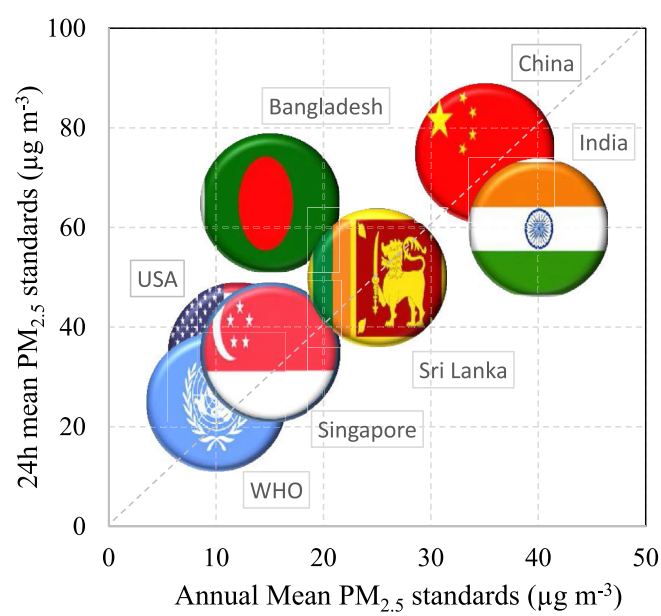

Fig. 7. Ambient regulations of $PM_{2.5}$ concentrations in some of the Asian countries, Europe and USA.

in public transport systems such as buses and taxis.

9. Summary, conclusions and future work

We reviewed exposures and concentrations of $PM_{2.5}$, UFP and BC in Asian TME environments and compared them against those in European, North American, and cities elsewhere. Some $PM_{2.5}$ exposure studies in Asian TMEs were found but similar studies for UFP and BC were rare. Asian studies of exposures of pedestrians, three-wheelers and motorcycles to PM were limited, and studies of cyclist exposures were even rarer. These observations clearly highlight the need for additional studies in Asian TMEs, especially for UFP and BC during walking and cycling. Asia is the largest continent, with ~60% of the total world population of whom nearly half live in urban areas (World Population, 2017). The exposures to air pollutants in TMEs in Asian cities are generally much higher than elsewhere in the world. Therefore, personal exposure data in TMEs of Asian cities are needed to develop exposure mitigation strategies.

The following conclusions are drawn:

- In general, concentrations of $PM_{2.5}$, BC, and UFP were higher by up to about 5, 7 and 9-times, respectively, in Asian TMEs (walk, cycle, car, and bus) than the corresponding TMEs in Europe or USA. In Asian cities, concentrations of $PM_{2.5}$ in open-air vehicles (motorcycles and auto-rickshaws) were 1.8–3.3 times higher than in walk, cycle, car, or bus TMEs.
- The instruments used in the Asian TME studies were generally the same as those used in European TMEs and elsewhere. There was good uniformity among the studies in how the instruments were used to characterize individual TMEs. Nonetheless, the amount of data that was collected varied considerably between studies; therefore, care must be taken in comparing and contrasting the results of different studies.
- Stochastic population-based exposure models have not been widely used in Asia but are well demonstrated in the USA, including extensive applications to a scientific review of ambient air quality standards. Such models, calibrated with adequate data regarding population demographics, time-location diaries, spatial and temporal variability in ambient concentration, and infiltration of ambient pollution to enclosed environments including vehicle cabins, can aid in identifying and prioritizing key sources of exposure and in assessing exposures that exceed health-related benchmarks. These models require probabilistic data on factors affecting exposure (e.g., infiltration factors).
- The regulations for ambient $PM_{2.5}$ are non-existent for some Asian countries while many countries adopt US EPA, WHO or EU regulations as a basis to set their own standards. Both annual and 24-h mean ambient air quality standard values are usually higher, by up to three times, in Asian countries compared to WHO guidelines. The consistently high concentrations observed in Asian TMEs could be seen as a reflection of these relaxed standards. The regulations for ambient UFP and BC are non-existent in Asian countries, which is also the case for European Union and elsewhere. Since personal exposure in TMEs is very different compared to those measured in ambient urban environments, this also leaves a question whether TME-specific regulations, especially for public transports, are needed or whether they should be the same as ambient regulations and enforced in amended form.

Our review highlighted several research gaps. For example, the limited research available for Asian TMEs usually focuses on urban areas. About half of the population of developing countries resides in rural environments (World Population, 2017), which are not always cleaner than urban areas because of factors such as the use of inferior quality fuels, biomass and crop residue burning, and cooking using biofuels (e.g., wood and cow-dung) (Kumar et al., 2013a, 2015). Thus, exposure assessment studies encompassing rural to semi-rural to urban environments will help in establishing exposure profiles in diverse Asian environments. Future work should also consider the seasonal and climate differences between Asian versus European cities as well as a relative wealth of Asian cities and neighbourhoods therein. The differences in exposure between well-regulated versus poorly-regulated countries or areas and the specific regions that are missing (poorly represented) should be explored by future studies.

Conflicts of interest

The authors declare no conflict of interest.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.atmosenv.2018.05.046.

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