Chapter 2

Literature Review

The literature review is divided into two parts. The first part examines the factors affecting commuters' exposure to PM, while the second one reviews recent advancements in air quality monitoring. A review of these studies supported the selection of appropriate factors to be considered in the study and the instruments to be used for data collection.

2.1 Factors affecting Commuters' PM Exposure

The commuters' exposure to PM depends on various factors such as pollutant concentrations, nose level of commuters, position of the vehicle, route choice, traffic conditions, and meteorological conditions (Kumar et al., 2018). The meteorology factors consist of atmospheric temperature (AT), relative humidity (RH), season of the year, traveling time (TT), wind direction (WD), and wind speed (WS), while traffic factors consist of position of the vehicle on the road (PVR), off-peak hour traffic (OPHT), peak hour traffic (PHT), traffic count (TC) and travel time or travel mode (TT/TM). The ventilatory factors include facemask, inhalation rate (IR) and inhaled dose (ID), while street configurations consist of land use (LU) and road geometry (RG). Finally, the background or ambient concentration of pollutants is an important factor affecting the on-road pollutant concentration. These factors were studied in commuters' exposure studies around the globe. Some of the important PM exposure studies and the factors considered in those studies are presented in 2.1.

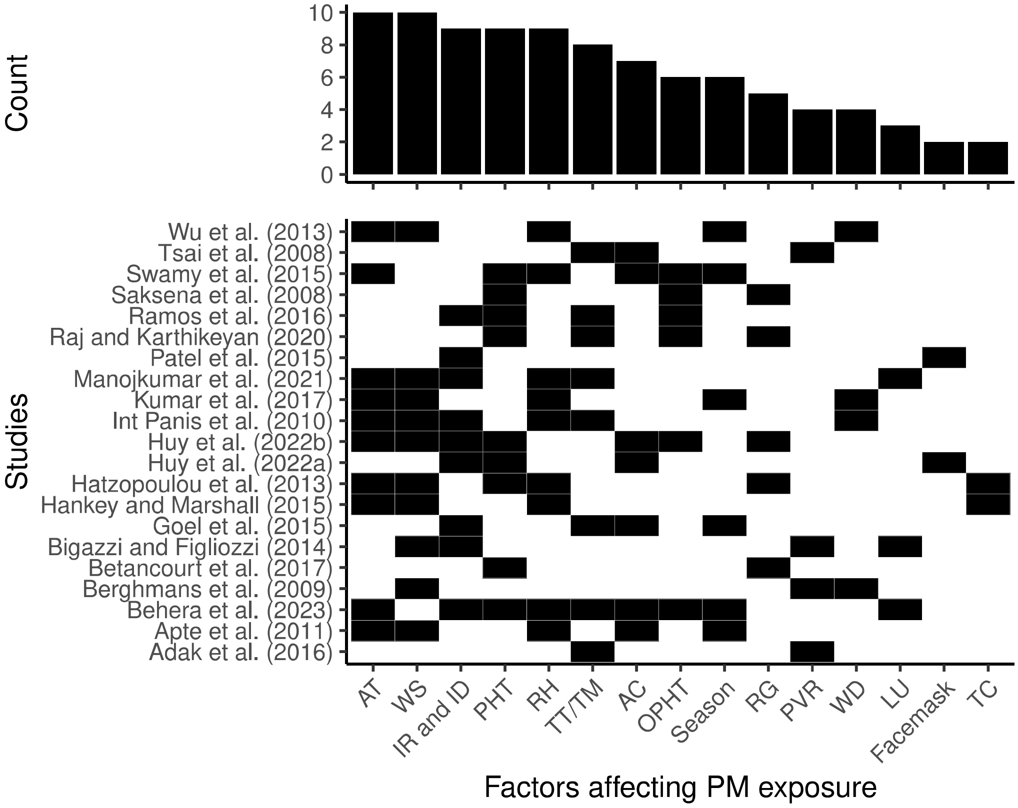

Fig. 2.1 Factors considered in various PM exposure studies around the world.

2.1.1 Meteorology Factors

Atmospheric Temperature (AT)

Panis et al. (2010) found that higher temperatures combined with sunny weather and low relative humidity led to an increase in PM concentrations in different transport modes. In lower temperatures, particles larger than 10 nm may shrink via evaporation to a diameter smaller than 10 nm at higher temperatures (Apte et al., 2011). Thus, as the temperature on the roadway is higher than the outside roadway temperature, the proportion of smaller particles is expected to be higher on the road. However, Huy et al. (2022) discovered that higher AT increases the mixing height, thereby expediting the pollutant dispersion. Due to the higher rate of dispersion of pollutants, the on-road PM exposure decreases (Wu et al., 2013). Behera et al. (2023) and Manojkumar et al. (2021) found the exposure concentration negatively correlated (r = 0.69 and 0.42, respectively) with AT.

Relative Humidity (RH)

The effect of humidity can't be ignored as this is one of the primary factors that significantly affect pollutant concentrations in different transport modes (Panis et al., 2010). Multiple studies confirmed the positive association of humidity with on-road PM levels (Behera et al., 2023; Manojkumar et al., 2021; Swamy et al., 2015; Wu et al., 2013). In Varanasi, the correlation between RH was moderately correlated (r = 0.65) with $PM_{2.5}$ exposure concentration (Behera et al., 2023). In contrast, concentration was found to be weakly correlated with RH in Vellore, India (r = 0.209), Ahmedabad, India (r = 0.47), and Foshan, China (r = 0.35) (Manojkumar et al., 2021; Swamy et al., 2015; Wu et al., 2013).

Season of the Year

The season of the year plays a crucial role in determining commuters' exposure to particulate concentrations. Generally, higher PM concentration is observed during winter due to relatively lower atmospheric mixing height, which restricts the dispersion of the pollutants (Tiwari et al., 2013). Thus, high outlying PM levels were observed during the winter season (Sharma et al., 2013).

However, pollution levels for commuters in various Indian cities, such as Varanasi, Delhi, and Ahmedabad, showed significant variations across seasons (Behera et al., 2023; Goel et al., 2015; Kumar et al., 2017; Swamy et al., 2015). For the same transport mode, the pollutant concentration decreased from winter to summer season in Delhi (Goel et al., 2015). Another exposure study in Delhi found that $PM_{2.5}$ levels were highest during winter, followed by summer and post-monsoon seasons, with the lowest levels being during the monsoon season (Kumar et al., 2017). An exposure study in Varanasi revealed that the highest $PM_{2.5}$ and PM_{10} exposure occurred during winter (205 and 324 µg m^{-3}), followed by spring (118 and 213 µg m^{-3}) and summer (87.0 and 191 µg m^{-3}) (Behera et al., 2023). An exposure study in Ahmedabad found $PM_{2.5}$ levels were significantly higher in winter (359 µg m^{-3}) compared to the monsoon (165 µg m^{-3}), with summer (165 µg m^{-3}) falling in between the two (Swamy et al., 2015). Similarly, the exposure study by Wu et al. (2013) in Foshan (China) found that $PM_{2.5}$ levels were significantly higher in the spring (84.1 µg m^{-3}) compared to the summer (59.6 µg m^{-3}). In general, PM exposure is highest in winter, followed by spring, summer, and monsoon, in that order.

Wind Speed and Wind Direction (WS and WD)

Calm wind weakens the dispersion of pollutants, resulting in increased concentrations (Goel and Kumar, 2015; Karar and Gupta, 2007; Tiwari et al., 2013). Wu et al. (2013) identified wind speed and wind direction as significant factors of on-road pollution exposure, contributing to 6.8 and 12.4% of the variability in PM concentration, respectively. However, the study found that wind speed was moderately negatively correlated (0.38 to 0.49) with the commuters' exposure to fine particulate concentration in various modes such as AC taxis, metro, bicycles and motorcycles, except bus (0.29), non-AC taxis (0.14) and walking (0.32). Manojkumar et al. (2021) observed a weak correlation (-0.19) between on-road pm exposure and wind speed. In addition, Huy et al. (2022) found that wind speed had less impact ($p = 0.26$) on-road $PM_{2.5}$ variation. Although we did not observe a significant correlation between PM exposure concentration and wind speed, we can conclude that higher exposure concentrations were observed during lower wind speeds based on the observed trend in these studies.

2.1.2 Traffic Factors

Position of Vehicle on the Road (PVR)

PM exposure for commuters is affected by whether the vehicle is in motion or idling in the traffic. These vehicle states highly depend on the vehicles' position on the road. The study by Tsai et al. (2008) found that the idling periods for motorcycles and buses are dependent on traffic intersections, whereas the extra idling period was noticed for buses during passengers alighting and boarding at bus stops. For motorcyclists, PM ($PM_{2.5}$ and PM_{10}) exposure during idling (56.7 and 103.9 µg m^{-3}) is higher than during the driving period (53.3 and 98.8 µg m^{-3}). The study by Adak et al. (2016) found that the primary reason for the higher idle time for shared auto-rickshaws is that they stop at various locations to pick up passengers. Idling time for motorcycles was longer than for cars despite having smaller widths since motorcyclists generally avoid rows of continuously moving cars. PM exposure for cyclists is generally low for dedicated bicycle lanes that are away from the center of the road (Berghmans et al., 2009; Bigazzi and Figliozzi, 2014).

Peak Hour Traffic and Off-Peak Hour Traffic (PHT and OPHT)

PM concentration is affected by the traffic condition whether it is peak hour traffic (PHT) or off-peak hour traffic (OPHT). These are also called rush hour traffic and non-rush hour traffic, respectively. Various studies found that PM exposure in PHT was higher than OPHT (Huy et al., 2022; Ramos et al., 2016; Saksena et al., 2008; Swamy et al., 2015). In contrast, Raj and Karthikeyan (2020) found that PM exposure in Chennai was the highest in OPHT. The exposure level increased from the morning peak hour, reaching its maximum at off-peak hour and subsequently decreasing to a minimum value during the evening peak hours. Behera et al. (2023) observed that PM was higher in PHT during spring and summer, while it was higher in OPHT during winter.

Travel Time or Travel Mode (TT/TM)

If PM concentration remains constant over a road stretch, we can assert that commuters' exposure would be directly proportional to travel time. Commuters that use motorcycles and cycles have the shortest travel time compared to other transport modes. This is because motorcycles have greater maneuverability than others (Goel et al., 2015), while commuters generally use cycles for short trips and, therefore, experience short travel time and low exposure. Motorcyclists in Vellore (India) experienced the lowest PM concentration compared to an auto-rickshaw, car and bus commuters due to shorter travel times (Manojkumar et al., 2021). In contrast, despite having the shortest travel time, motorcycle commuters in Hanoi (Vietnam) experienced the highest PM concentrations, primarily due to the open nature of this mode of transportation (Tsai et al., 2008). Similarly, despite having 50% less traveling time than cars and buses, motorcyclists in Chennai were exposed to higher overall mean $PM_{2.5}$ exposure concentration (Raj and Karthikeyan, 2020). Therefore, route planners should assist commuters in choosing alternative routes with the shortest travel times (Ramos et al., 2016).

Traffic Count (TC)

Traffic count or volume also affects on-road exposure. There were a limited number of studies that considered TC in PM modeling. The study by Hatzopoulou et al. (2013) found traffic counts as weakly associated with the exposure concentration. The study considered constant TC for all the study routes, which might be the primary reason for the weak association. Hankey and Marshall (2015) also reported a weak association between on-road $PM_{2.5}$ concentration and TC with the with a correlation coefficient of 0.14.

2.1.3 Ventilatory Factors

Inhalation Rate and Inhaled Dose (IR and ID)

The inhalation rate (IR) depends on the extent of physical activity. USEPA (1996) that vigorous activities have the highest IR (5.11×10^{-3} m^3 min^{-1}), while sedentary or resting activities have the lowest IR (3.9×10^{-3} m^3 min^{-1}). Driving motorcycles involves light activity with an IR of 13×10^{-3} m^3 min^{-1}.

Commuters' inhaled dose (ID) quantifies the total particulate pollution that enters a person's body through inhalation during their travel. It is influenced by inhalation rate, travel time, and location of the trip. The higher the ID, the riskier it would be for the commuter. The inhalation rate for motorcyclists was considered as 0.01 m^3 min^{-1} for calculating ID in most PM exposure studies. Traveling in Indian cities such as Varanasi, Delhi, and Vellore attracted $PM_{2.5}$ doses of 2.31 – 6.51, 2 – 3, and 5 μg km^{-1}, respectively (Behera et al., 2023; Goel et al., 2015; Manojkumar et al., 2021). In addition, motorcyclists in Ho Chi Minh (Vietnam) are exposed to 3.68 and 7.72 μg km^{-1} of $PM_{2.5}$ and PM_{10}, respectively (Huy et al., 2022). In contrast, Huy et al. (2022) found that motorcyclists in Ho Chi Minh (Vietnam) inhale lower PM dose ($PM_{2.5}$: 1.6 – 2.4 and PM_{10}: 6.0 – 8.2 μg km^{-1}) during the afternoon in comparison to morning ($PM_{2.5}$: 3.6 – 5.8 and PM_{10}: 11.7 –

16.3 µg km^{-1}). An adult commuter in Indonesia inhales 0.11 and 0.23 µg/kg/day of $PM_{2.5}$ and PM_{10}, respectively (Patel et al., 2016).

Facemask

Exposure to pollution can be reduced when using a facemask while commuting. Huy et al. (2022) found both surgical and cloth masks to be effective against exposure to PM. The study found the motorcyclists in Ho Chi Minh (Vietnam) were exposed to comparatively lower average $PM_{2.5}$ and PM_{10} exposure concentrations by using surgical masks (63.6 and 70.7 µg m^{-3}) and cloth masks (71.5 and 90.1 µg m^{-3}) in comparison to the motorcyclist who did not wear any masks (107.7 – 109.3 and 232.6 – 239.6 µg m^{-3}). Similarly, Patel et al. (2016) found that surgical masks are effective in reducing 30 and 71% of $PM_{2.5}$ and PM_{10} exposure concentrations, respectively.

2.1.4 Street Configurations

Land Use (LU)

The volume of traffic along a route depends on the land use (residential areas, commercial zones, educational institutions, and railway stations) in the area. Therefore, a commuter's exposure to pollutants would depend on the land use around the route. Routes with high traffic volumes resulting from commercial activities and government offices exhibited higher pollutant concentrations than routes with low traffic volumes associated with passenger and student movements around railway stations and educational institutions, respectively (Behera et al., 2023). Manojkumar et al. (2021) found that $PM_{2.5}$ levels were significantly lower in residential areas (71 – 73 µg m^{-3}) in comparison to traffic routes (119 – 127 µg m^{-3}).

Road Geometry (RG)

Road geometry plays an important factor in commuters' exposure to particulate pollution (Raj and Karthikeyan, 2020). Street configuration is responsible for the dispersion of pollutants, which directly influences the on-road pollutant concentration (Chan et al., 1994). Saksena et al. (2008) found that the PM concentration builds up due to the presence of narrower streets and multi-story buildings near the curb. PM pollution in street canyons was twice as large as that measured in wide streets in Bogota, Colombia (Betancourt et al., 2017). However, Huy et al. (2022) found that with an increase in the number of lanes, the exposure was found to be increased in Ho Chi Minh, Vietnam. The higher PM level might be due to the higher combined traffic volume on those lanes.

2.1.5 Ambient Concentration (AC)

The PM exposure concentrations may have a high number of outliers due to the strong influence of ambient air pollution (Swamy et al., 2015). Two different studies in Ho Chi Minh (Vietnam) found that the ratio of on-road to ambient PM concentrations ranged from 1.48 to 2.8 (Huy et al., 2022). Goel et al. (2015) and Tsai et al. (2008) found the ratio to be 1.3 and 1.2 in Delhi and Taipei, respectively. In contrast, Behera et al. (2023) found the ratio to vary from 1 – 2 during various hours of the day. In addition, the studies conducted by Behera et al. (2023) and Huy et al. (2022) observed a strong correlation between AC and on-road PM concentration, with Pearson correlation coefficient (r) of 0.83 – 0.96 and 0.81, respectively. This indicates a high chance that ambient air pollution will influence on-road PM concentration.

The commuters' exposure to PM depends on various factors. However, considering every known factor is not feasible and is constrained by the availability of instruments, human resources, and the cost.

2.2 Recent Advancements in Air Quality Monitoring

Air pollution monitoring is a diagnostic tool that helps determine the pollution level of a specific location. The monitoring data provide an indication or status of the air quality we breathe at a specific location and time. Various methodologies have been used to monitor global, regional, and local pollution based on their spatiotemporal resolutions. Several countries have established environmental protection agencies to monitor air pollutants and provide countermeasures to reduce their concentrations (Table 2.1). These organizations set National Ambient Air Quality Standards (NAAQS) for various pollutants in their respective countries.

Furthermore, the organization determines the number of monitoring locations with varying spatial resolutions based on the needs of a city. Various methodologies for air pollution monitoring and control techniques for pollution abatement have been used. The data gathered over time enables the pollution control board to develop plans and strategies to reduce pollutant concentrations, known as air pollution control policy. Because of technological advancement, pollution measurement methodologies and techniques are evolving. This review attempted to compile and present all these advancements to assist researchers in understanding the process of air pollution monitoring. The following sections describe different types of monitoring used by policymakers and researchers, such as satellite-based air quality monitoring, continuous ambient air quality monitoring, and on-road air quality monitoring.

2.2.1 Satellite-Based Air Quality Monitoring

Satellite-based air pollution measurements can assess global and regional pollution levels over time. Satellite monitoring can estimate exposure for the entire population at any time. Satellites equipped with pollutant monitoring instruments orbit the Earth to collect

Table 2.1 List of air pollution regulatory and control boards across different countries.

Air Quality Regulatory Boards	**Abbreviation**	**Country**
World Health Organization	WHO	Worldwide
Central Pollution Control Board	CPCB	India
Environmental Protection Agency	EPA	United States
European Environmental Agency	EEA	European Countries
Department of Climate Change, Energy, the Environment, and Water	DCCEEW	Australia

data such as spectral bands and wavelength groups from retrieving desired pollutant concentrations. The pollutants such as PM, nitrogen oxide (NO_x), sulfur dioxide (SO_2), carbon monoxide (CO), and ozone (O_3) can be estimated using satellite measurements of aerosol optical thickness or depth (AOT/AOD). The AOT comprises the total column between the satellite sensor and the surface (Brauer et al., 2016). Alternatively, AOT can be defined as the degree of aerosols that hinder light transmission (AbdelSattar, 2019). As a result, a higher AOT value indicates that a location has a higher pollution level. Lee et al. (2011) used the concept to calculate the daily $PM_{2.5}$ (PM having diameters less than 2.5 µm) levels by calibrating the AOT data from a moderate resolution imaging spectroradiometer (MODIS) in the New England region, whereas Brauer et al. (2016) used it to calculate global annual average $PM_{2.5}$ at a spatial resolution of $0.1°\times0.1°$ (approximately 10 km × 10 km). Lee (2019) estimated California's annual average ambient $PM_{2.5}$ concentrations at a resolution of 1 km in 2016 utilizing satellite multi-angle implementation of atmospheric correction (MAIAC) AOT, land use characteristics, and meteorology (Lee, 2019). Besides the PM, satellite remote monitoring can estimate CO, CO_2, NO_x, O_3, SO_2, and NH_3 pollutants. Various satellites have been used for monitoring air pollutants (Zhang and Zhao, 2022). These satellite instruments have varied spatiotemporal resolutions and can capture various pollutants (Table 2.2). Zhang and Zhao (2022) used the tropospheric ozone monitoring instrument (TROPOMI) with a spatial resolution of 7 km × 3.5 km for daily retrieval of NO_2 using the GaoFen-5/EMI algorithm. Lalitaporn and Mekaumnuaychai

(2020) used the measurements of pollution in the troposphere (MOPITT) to retrieve CO columns. They compared the columns to ground concentrations from 2014 to 2017 and observed that the satellite and ground datasets agreed well.

Table 2.2 List of major pollutants' satellite monitoring instruments and their spatiotemporal resolutions (Adopted from: AbdelSattar, 2019).

Instrument	**Satellite**	**Spatial Resolution**	**Temporal Resolution**	**Pollutants**
GOME – 1	ERS – 2	320 × 40 km^2	3 days	NO_2, SO_2, PM and O_3
MODIS	Terra	0.25 – 1 km^2	1 – 2 days	PM
MISR	Terra	0.275 km^2	9 days	PM
MOPITT	Terra	22 × 22 km^2	3 days	CO and NH_4
SCIMACHY	ENVISAT	60 × 30 km^2	6 days	NO_2 and SO_2
OMI	Aura	24 × 13 km^2	1 day	O_3, SO_2, HCHO, and NO_2
TES	Aura	8 × 5 km^2	2 days	O_3, CO, CO_2, and NH_3
GOME – 2	MetOP	80 × 40 km^2	1.5 days	NO_2, SO_2, PM and O_3
IASI	MetOP	50 × 50 km^2	0.5 day	SO_2, CO, NH_3, NH_4 and CO_2
TROPOMI	Sentinel – 5P	7 × 3.5 km^2	1 day	O_3, NO_2, SO_2, CO, NH_4, and formaldehyde

2.2.2 Continuous (Stationary) Ambient Air Quality Monitoring

Satellite monitoring may provide high spatial resolution, but the data generally lags desirable temporal resolution. Also, the estimation of pollutant concentrations via satellite monitoring is based on several assumptions. Therefore, continuous monitoring of pollutants is required to improve temporal resolution and accuracy. In continuous monitoring the process of sampling, weighing, analyzing, recording, and dissemination of the information and insights are automated. Instruments are generally placed 3 – 10 m above the ground to avoid pollutant contributions from ground zero objects such as vehicle exhaust and soil dust resuspension. For monitoring at a city level, the selection of the measurement sites is critical for obtaining an unbiased measure of pollutant concentrations. For unbiased sampling, instruments may be placed far from industrial areas, traffic junctions, the middle of a park, or a site obstructed by surrounding trees. The ambient monitoring station has a spatial resolution of 4 × 4 km, corresponding to 16 sq. km. Therefore, for a 1000 sq. km area, at least 62 monitoring stations are required for reasonable spatial coverage. Apart from spatial resolution, the optimization of monitoring stations or networks depends on various factors such as population density, similarities between two monitoring locations, funds and resource availability, number of pollutants to be monitored, and identification of the location as high traffic emission zone (Modak and Lohani, 1985).

In government agencies and research institutes, various continuous ambient air quality monitoring systems (CAAQMS) have been used, as described in Table 2.3. All instruments are factory-calibrated before deployment and calibrated regularly to ensure consistency. The instruments work on various principles such as light scattering, betta-attenuation, electrochemical analysis, oscillating frequency and gravimetric analysis. The operating principle determines the precision of the instruments. The mass of particulate matter is weighed in gravimetric analysis. In tapered element oscillating microbalance (TEOM), real-time particulate mass is estimated by the principle of oscillating frequency. The synchronized hybrid ambient real-time particulate monitor (SHARP) works on betta attenuation and aerosol light scattering principle (Li et al., 2019). The continuous pollution monitoring process is presented in Fig. 2.2.

2.2.3 On-Road (Mobile) Air Quality Monitoring

The satellite and ambient continuous monitoring cannot capture on-road exposure because both instruments are stationed at least 700 km and 3 – 10 m above the ground, respectively. The commuters' breathing levels are typically at the height of an adult (1.6 – 1.7 m). Thus, on-road monitoring with portable monitors can provide greater spatial (on-road) and temporal (1-second) resolution of pollution exposure. On-road or mobile monitoring

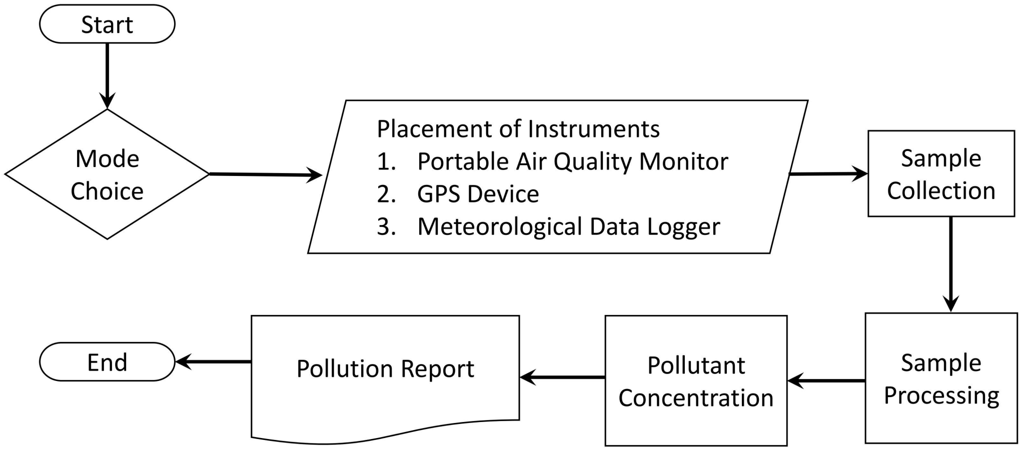

Fig. 2.2 Schematic diagram of continuous pollution monitoring process.

Table 2.3 Working principles and pollutants monitored by various CAAQMS.

Instruments	**Working Principle**	**Frequency (Seconds)**	**Pollutants**
OIZOM Polludrone Pro	Light scattering and electrochemical analysis	1	UFP, $PM_{2.5}$, PM_{10}, CO, CO_2, SO_2, NO, NO_2, O_3, and H_2S
Prana Air CAAQMS Ambient Pro	Light scattering	< 30	$PM_{2.5}$, PM_{10}, CO, SO_2, NO_2, O_3, and H_2S
ThermoFisher 1405-DF TEOM	Tapered element oscillating microbalance	3600	PM_1, $PM_{2.5}$, and PM_{10}
ThermoFisher 5030i SHARP	Beta attenuation and light scattering	1	PM_1, $PM_{2.5}$, and PM_{10}

refers to measurements taken while traveling in various modes of transportation. Study reveals that office commuters in India spend around 7% of the day traveling (Bureau, 2019). The commuting gets worse as the on-road exposure concentrations are higher than off-road (Wang et al., 2017). Commuters use different modes of travel, such as cars, buses, motorcycles, auto-rickshaws, walking, and cycling, to reach their destinations (Kumar et al., 2018). Personal exposure to air pollution depends upon in-vehicle air quality, traffic flow conditions, the extent of emissions, background concentrations, the position breathing zone, personal behavior, choice, and meteorological conditions (Kumar et al., 2018). The in-vehicle air quality varies with the type of the modes (Transport Microenvironments: TMEs). The in-vehicle air quality may be a combination of Particulate Matter (PM), Black Carbon (BC), and Ultrafine Particles (UFP). Again, in-vehicle air quality in different TMEs depends on the characteristics of the transportation system (e.g., ventilation type, vehicle model (year and design), ventilation settings (windows open/closed, AC on/off), and fuel type (De Nazelle and Orjuela, 2017; Goel and Kumar, 2014; Knibbs et al., 2011). In-vehicle air quality can be detected by placing the measuring instrument in different positions in the vehicle. For example, the placement of the portable instrument in the car or auto-rickshaw may be front or back; in the bus, it may be front/back/middle/cabin, upper/lower deck; in a cycle or motorcycle, it may be carried by the users. The air quality detection may be influenced by where the instrument is placed in the vehicles (Kaur et al., 2007). Therefore, simultaneous air quality monitoring is needed to get the various in-vehicle locations. The process of mobile monitoring is shown in Fig. 2.3.

Several types of equipment are used for monitoring pollutants like PM, BC, and UFP. Table 2.4 lists the equipment details as well as their characteristics. These instruments have different working principles and the costs associated with them. The choice of instruments is based on the measurement frequency, concentration range, resolution, battery life, operational conditions (temperature and humidity), and portability. Other considerations

Table 2.4 Specification of instruments used for in-vehicle pollution monitoring (Adopted from: Kumar et al., 2011).

PM Monitors	
GRIMM, model 1.108 • Measurement frequency: 1-6 s • Particle size range: 0.23-20 µm • Concentration range: 0.1-1 x 10^5 µg m^{-3} • Measurement principle: light scattering and filter-sampling	**TSI DustTrak, model 8533** • Measurement frequency: 1 s-1 hr • Particle size range: 0.1-15 µm • Concentration range: 1-1.5 x 10^5 µg m^{-3} • Measurement principle: light scattering
TSI DustTrak, model 8534 Handheld model • Measurement frequency: 1 s-1 hr • Particle size range: 0.1-15 µm • Concentration range: 1-1.5 x 10^5 µg m^{-3} • Measurement principle: light scattering	**TSI OPS, model 3330** • Measurement frequency: 1 s • Concentration range: 0.001-2.75 x 10^5 µg m^{-3} • Measurement principle: light scattering
Thermo pDR 1500 (Active) • Measurement frequency: 1 s-1 hr • Particle size range: 0.1-10 µm • Concentration range: 1-4.0 x 10^5 µg m^{-3}	**Thermo pDR 1000N (Passive)** • Measurement frequency: 1 s-1 hr • Particle size range: 0.1-10 µm • Concentration range: 1-4.0 x 10^5 µg m^{-3}
UFP Monitors	
TSI CPC, model 3007 • Measurement frequency: 1 s-30 min • Particle size range: 0.01 - >1.0 µm • Concentration range: 0-1 x 10^5 particles/cm^3 • Measurement principle: light scattering	**TSI P-Trak, model 8525** • Measurement frequency: 1 s-30 min • Particle size range: 0.02 - 1.0 µm • Concentration range: 0-5 x 10^5 particles/cm^3 • Measurement principle: light scattering
BC Monitors	
AethLabs Micro-aethalometer, model AE51 • Measurement frequency: 1-300 s • Measurement range: 0-1000 µg m^{-3} • Measurement principle: light absorption	

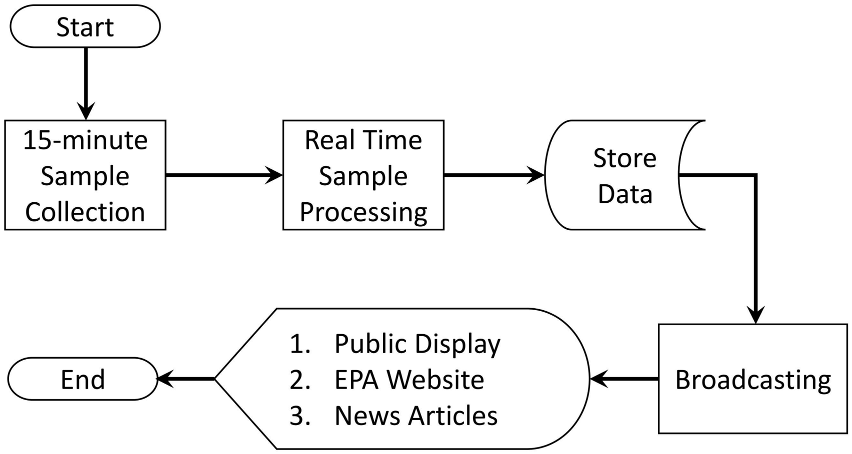

Fig. 2.3 Schematic diagram of on-road pollution monitoring process.

like ease of operation, calibration, and measurement artifacts are also important in selecting the instruments.

Mobile air quality monitoring systems provide higher spatial and temporal resolution as compared to satellite and stationary-based systems. However, the mobile monitors fail to provide the concentration of pollutants at various locations at the same instance of time. Thus, multiple mobile monitoring instruments are needed to eliminate such a limitation.

2.3 Research Gaps

On-road pollution monitoring has been widely used for measuring and modeling commuters' exposure to PM. However, various factors were explored to a lesser extent due to resource constraints such as the unavailability of advanced monitoring, human resources or funds to carry out the project for a longer duration. These factors may have a significant effect on commuters' exposure to on-road PM levels. The review of the literature reveals the research gaps in the literature. The key research gaps are identified below.

1. Firstly, traffic patterns and human behavior are significantly affected by the day of the week. Typically, higher traffic activities are observed during weekdays compared to

weekends. However, recreational activities during certain periods on weekends may add to the pollution levels. Also, the day of the week (Monday – Sunday) also has a significant effect on traffic patterns. The pollution on Monday may surge due to higher traffic after the weekend leisure period. The opposite trend may be observed on the weekends due to lower traffic movements. Therefore, the effect of day of week and day type on commuters' exposure to PM should be explored.

2. A limited number of studies have been conducted in Tier-II cities to investigate the effect of season on commuters' exposure to PM. Most of the exposure analyses were based on either meteorological factors (AT, RH, Season, WS and WD) or traffic factors (PHT, OPHT, and TT). As discussed in Section 2.1.1, the pollutant dispersion and dilution process is greatly affected by meteorology factors. Traffic congestion may not be the only factor that affects on-road PM levels since the PM would disperse more rapidly during summer than winter. Therefore, the combined effect of meteorology parameters and traffic factors should be considered in the PM exposure study.

3. On-road exposure can be modeled using various factors (Section 2.1). The most appropriate factors or combination of factors should be explored to understand and model PM better.

4. While traffic parameters can be controlled to some extent through traffic management, controlling meteorology, ventilatory conditions and ambient concentrations is not possible. The effects of traffic count and composition were less explored. Therefore, there is a need to study the effects of traffic count on commuters' exposure to on-road PM levels.