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Review article

A review of traffic-related air pollution exposure assessment studies in the developing world

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Abstract

Exposure assessment studies in the developing world are important. Although recent years have seen an increasing number of traffic-related pollution exposure studies, exposure assessment data on this topic are still limited. Differences among measuring methods and a lack of strict quality control in carrying out exposure assessment make it difficult to generalize and compare findings between studies. In this article, exposure assessment studies carried out in the developing world on several traffic-related air pollutants are reviewed. These pollutants include particulate matter (PM), carbon monoxide (CO), nitrogen dioxide (NO_2), volatile organic compounds (VOCs), and polycyclic aromatic hydrocarbons (PAHs). In addition, it discusses advantages and disadvantages of various monitoring methods (ambient fixed-site monitoring, microenvironment monitoring, and personal exposure assessment using portable samplers) for these pollutants in exposure assessment studies. Also included in this paper is a brief introduction of standards for these pollutants in ambient air or in occupational settings established by the United States Environmental Protection Agency (USEPA), the United States Occupational Safety and Health Administration (OSHA) and the World Health Organization (WHO). The review ends with a summary of the limitations and gaps in recent studies and suggestions for future research in the developing world.

Keywords: Air pollution; Developing countries; Exposure assessment; Traffic-related

Contents

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1. Introduction

The United Nations estimated that over 600 million people in urban areas worldwide were exposed to dangerous levels of traffic-generated air pollutants (Cacciola et al., 2002). Air pollution and its public health impacts are drawing increasing concern from the environmental health research community, environmental regulatory agencies, industries, as well as the public. The quality of the air, both indoors and outdoors, is closely related to morbidity and mortality from respiratory and cardiovascular diseases. Common air pollutants that draw intense concerns include particulate matter (PM), ozone (O_3), carbon monoxide (CO), sulfur dioxide (SO_2), nitrogen dioxide (NO_2), lead (Pb), volatile organic compounds (VOCs), and polycyclic aromatic hydrocarbons (PAHs). We do not include ozone, a secondary pollutant, in this review. In addition, SO_2 and Pb are also excluded since the concerns they receive are not as intense as other pollutants.

Association between mortality rate and particulate air pollution has long been studied, but many studies may be limited by a lack of control for confounding factors. Dockery et al. (1993) related excess daily mortality from cancer and cardiopulmonary disease to several air pollutants, especially fine particulate matter ($PM_{2.5}$, particulate matter with aerodynamic diameter of equal to or less tan 2.5 μm) in their prospective cohort study. Since then, many other epidemiological studies on the adverse human effects of air pollutants have been carried out, ranging from variations in physiological functions and subclinical symptoms (heart rate variability, peak expiratory flow rate, etc.) to manifest clinical diseases (asthma, chronic obstructive pulmonary disease, stroke, lung cancer, leukemia, etc.), premature births and deaths (Delfino et al., 1998; Naeher et al., 1999; Laden et al., 2000; Suresh et al., 2000; Janssen et al., 2002; Calderón-Garcidueñas et al., 2003; Wilhelm and Ritz, 2003; O'Neill et al., 2004; Preutthipan et al., 2004).

Anthropogenic air pollution sources can be categorized based on different criteria. One criterion is whether the source is mobile or not. The former refers to traffic-related sources, including ground traffic (bus, private car, taxi, combi, motorcycle, etc.), underground traffic (metro or subway) and air traffic, and the latter is mainly industrial, commercial and personal emissions.

Table 1
1997 USEPA NAAQS standards

Pollutant (unit)	Time duration	Values
CO (ppm)	1 h	35
	8 h	9
NO_2 (ppb)	Annual	53
PM_{10} (μg/m^3)	24 h	150
	Annual	50
$PM_{2.5}$ (μg/m^3)	24 h	65
	Annual	15
O_3 (ppb)	1 h	120
	8 h	80
SO_2 (ppb)	24 h	140
	Annual	30
Lead (μg/m^3)	Quarter	1.5

Table 2
Regulatory values set by OSHA

Pollutants	Duration	Values
Benzene	8-h work day or	1 ppm
	40-h work week	(3.19 mg/m^3)
Toluene	8-h work day or	200 ppm
	40-h work week	(753.6 mg/m^3)
Ethylbenzene	8-h work day or	100 ppm
	40-h work week	(434 mg/m^3)
Xylene	8-h work day or	100 ppm
	40-h work week	(434 mg/m^3)

Traffic-related sources of air pollution are drawing increasing concerns from interested exposure assessors, epidemiologists, as well as toxicologists. Ground-level traffic vehicles in urban areas are typically natural gas-fueled, gasoline fueled or diesel-fueled. The physical characteristics and chemical compositions of natural gas, gasoline and diesel are not the same in different regions in the world, like benzene content (Verma and Tombe, 2002), making it complex for the findings in one location to be generalized to other locations. This complexity in generalization across studies is further complicated by different meteorological conditions, different percentage of heavy polluters (more motorcycles in the developing world), design of motor ways (graded or non-graded roads), driving habits, different maintenance as well as quality of and control measures for vehicles, and exposure profiles of people (Gwilliam, 2003).

Compared with the large volume and varieties of studies carried out in the developed world, exposure assessment studies in developing countries are relatively scarce. Despite the revised emission standards and technical improvement in pollution control measures, expanding industrialization and increasing traffic volumes in the developing countries will drastically increase total emissions of many air pollutants, as has been predicted by a study on air pollutant trends in East Asian countries (Klimont et al., 2001).

In this paper, recent investigations in exposure assessment studies on several main air pollutants conducted in the developing world are reviewed. Gaps and needs are suggested for future research.

2. Regulation

In 1997, the US Environmental Protection Agency (EPA) (USEPA, 2001) modified its National Ambient Air Quality Standards (NAAQS) (Table 1), in which the 24-h and annual average concentration limits for ambient $PM_{2.5}$ were 65 and 15 μg/m^3, respectively. Also included in the standards are CO, PM_{10}, SO_2, NO_2, ozone, and lead. Standards with different time durations are defined because for some pollutants, like ozone and CO, short-term effect is

a concern, while for lead and particulate matter, more attention is put on its long-term effect. In addition, which time duration is chosen depends on which durations are more associated with human health effects.

EPA has not proposed any standards for benzene, toluene and other ambient VOCs. But the US Occupational Safety and Health Administration (OSHA, 2004) set 1 ppm (3.19 mg/m^3) and 200 ppm (753.6 mg/m^3) as workplace time-weighted average regulation limits for a normal 8-h work day or 40-h work week for benzene and toluene, respectively. The 8-h and 15-min workplace time-weighted average limits for ethylbenzene set by OSHA were 100 and 125 ppm (434 and 543 mg/m^3), and those for xylene were 100 and 150 ppm (434 and 651 mg/m^3), respectively. These were summarized in Table 2. OSHA set permissible exposure limit for coal tar pitch volatiles at an 8-h time-weighted average (TWA) of 0.2 mg/m^3 for occupational workers. However, EPA has not set PAH limits in ambient air.

The World Health Organization (WHO) (WHO, 2001) (Tables 3 and 4) also has guidelines for these air pollutants, but these are recommendations and not compulsory for governments to follow. $PM_{2.5}$ and PM_{10} guidelines are not available due to the insufficient research information. Because there is no threshold value for benzene in terms of its carcinogenicity, only unit risk was provided. For PAHs, BaP may serve as an indicator and the risk slope for it was set at 8.7×10^{-2} [μg/m^3]$^{-1}$ (Table 4).

It has to be noted that an exposure level lower than the recommended or regulatory standard does not mean that a life-long exposure at such a level is safe. Some standards are 15-min based, others are 1-h based, still others 8-h based and yet others 24-h or annually based. In addition, most of the standards were not established based on NOAELs (No observed adverse effect levels) or LOAELs (lowest observed adverse effect levels). Instead, many factors, including economic feasibility and technical feasibility, have been considered in establishing these standards. So many of the standards, especially those regulatory ones, are much higher than their respective NOAELs or LOAELs. In addition, for many pollutants, due to insufficient pool of evidences, the standards may need to be modified based on new discoveries or new economic and technical feasibilities.

Table 3
Guidelines of ambient air pollutants established by WHO in 2001

Pollutants	Averaging time	Values
CO	1 h	30 mg/m^3 (26 ppm)
	8 h	10 mg^3 (8.7 ppm)
Lead	1 year	0.5 μg/m^3 (0.059 ppb)
NO_2	1 h	200 μg/m^3 (106 ppb)
	1 year	40 μg/m^3 (21 ppb)
O_3	8 h	120 μg/m^3 (61 ppb)
SO_2	24 h	125 μg/m^3 (47.7 ppb)
	1 year	50 μg/m^3 (19.1 ppb)

Table 4
WHO guideline for BTEX in occupational settings (2001)

Pollutants	Averaging time	Guidelines
Benzene	NA	$(4.4–7.5)\times10^{-6}$ [μg/m^3]$^{-1}$
Toluene	1 week	260 mg/m^3
Ethylbenzene	1 year	22 000 mg/m^3
Xylene	24 h	4800 mg/m^3
	1 year	870 mg/m^3
PAH (BaP)	NA	$8.7\times10^{-}2$ [μg/m^3]$^{-1}$

For some carcinogens, such as benzene, there seems no NOAEL or LOEAL for its carcinogenicity. The standards for these carcinogens are usually set by considering feasibility and acceptability to the public.

Caution has to be paid even when the exposure level of an air pollutant is lower than LOAEL or NOAEL. For one thing, human exposure to a pollutant has multiple sources other than via air, like water and food. In addition, usually in ambient or occupational environment there are many coexisting air pollutants, several of which may have additive or even synergistic effects. For instance, both benzene and toluene can affect the central nervous system and they usually coexist with each other in the air due to several common resources. Therefore, it will be better if we consider this coexistence when setting standards for these pollutants.

3. Particulate matter (PM)

Among common ambient air pollutants, particulate matter is currently under intensive epidemiological and toxicological investigation. Airborne particulate matter refers to particles or droplets of various sizes, physical characteristics and chemical compositions present in the air. Previously, environmental epidemiologists had mainly focused on particulate matter with an aerodynamic diameter equal to or less than 10 μg (PM_{10}). As increasing evidence links $PM_{2.5}$ to various respiratory and cardiac effects, more and more attention is paid to the exposure assessment of $PM_{2.5}$ and its cardiopulmonary impacts (Goldberg et al., 2001; Janssen et al., 2002; Magari et al., 2002).

Anthropogenic airborne particulate matter comes from a variety of sources, which include, but are not restricted to traffic, industries, commerce and domestic heating and cooking. Among them, traffic-related particulates have been under intensive scrutiny for at least two reasons. One is due to the evidence that particulates generated from combustion processes, especially diesel exhaust particulates (DEP), are more potent in posing adverse health effects than those from non-combustion process (Laden et al., 2000; Janssen et al., 2002). Another reason is that traffic-generated emissions were estimated to account for more than 50% of the total emissions of particulate matter in the urban areas in highly industrialized countries (Briggs et al., 1997; Wróbel et al., 2000). In London, UK, more than 80% of particulate matter

is from road traffic (Department for Transport, 2002). In Athens, Greece, the contribution of road traffic to total $PM_{2.5}$ emission is estimated to be 66.5% (Economopoulou and Economopoulos, 2002). In addition, many cities in the developing world are facing serious problem from traffic-related particulate emissions (Kulkarni and Patil, 1999; Yang, 2002; Shendell and Naeher, 2002; Wang et al., 2003). In Malaysia, air pollutant emissions from traffic vehicles were estimated to account for 82% of the total emissions in 1996 (Afroz et al., 2003).

Airborne particulate pollution is more serious in the developing world than in the developed countries, especially in those developing countries currently under rapid industrialization and changes in land use. In the collection of articles for this review, preferences were given to studies on $PM_{2.5}$, since it is more related with traffic emissions than TSP or PM_{10}. In addition, studies on in-vehicle levels, personal exposures in traffic environments, and roadside (street) levels received more weight. Geographical considerations were also taken into account: we excluded some equally important studies when many such studies exist in the same regions and included some studies on ambient TSP when studies in a region are very limited. These rules also fit our review on other pollutants.

A particulate characterization study was carried out in five Asian sites: Manila in the Philippines, Hong Kong, Cheju Island in Korea, Sado Ishand, Japan, and Hanoi in Vietnam in 2001. It was found that, except for the site in Japan, annual average $PM_{2.5}$ and PM_{10} concentrations were well above the US EPA annual standard of 15 μg/m^3 for $PM_{2.5}$ in the four other sites (Cohen et al., 2002). In addition, the chemical composition of the pollutant is also different in different regions under study. For instance, the contents of organic matter (45%) and elemental carbon (28%) in $PM_{2.5}$ were the highest in Manila monitoring site compared to those in the other four sites, which ranged from 7% to 8.8% for elemental carbon and 8% to 25% for organic matter. Though the pollutants monitored at the five sites were sampled using the same method during roughly the same period, their comparison may still suffer from the different locations, since some locations (Cheju Island in Korea and Sado Island in Japan) were more remote than others. Further, this study did not characterize what portion of the concentrations was from traffic emissions. An investigation in Guangzhou, China (Chan et al., 2002) found that $PM_{2.5}$ and PM_{10} levels in several traffic micro-environments were generally much higher than those found in the United States and Europe (Table 5). A five-site survey

Table 5
Traffic-related exposure studies on airborne particulate matter

Study	Year	Location	Exposure type (Sampling duration)	Size	Subject or place	Level (ìg/m^3)
Salma et al., 2004	April–May 2002	Budapest, Hungary	Ambient (12-h daytime)	$PM_{2.0}$ PM_{10}	Urban canyon	14–22
Leong et al., 2001	January–June 1999	Bangkok, Thailand	Ambient (24 h)	PM_{10}	Busy street	84.33
Shendell and Naeher, 2002	May and June 1997	Three cities, Guatemala	Ambient (248–370 min)	$PM_{2.5}$	Guatemala City	150[a]
					Quetzaltenango	120[a]
Bogo et al., 2003	1999	Buenos Aires, Argentina	Ambient (24 h)	PM_{10} ($PM_{2.5}$)	Fixed (January–March)	52 (41)
					Fixed (April –August)	44 (33)
Chan et al., 2000	Winter 1997	Hong Kong, China	Street (24 h)	PM_{10}	Roadside	25.56–337.4
Wang et al., 2003	2001	Nanjing, China	Ambient (8:30 am–4:30 pm)	PM_{10} ($PM_{2.5}$)	Suyuan Hotel (high traffic)	632 (423)
Chan et al., 2002	Summer 2000	Guangzhou, China	In-vehicle (Rush hours: 150 min)	PM_{10} ($PM_{2.5}$)	A/C Bus	128 (101)
					A/C Taxi	82 (73)
					Non-A/C Bus	203 (145)
					Non-A/C Taxi	150 (106)
Gómez-Perales et al., 2004	Spring 2002	Mexico City	In-vehicle (Rush hours: 180 min)	$PM_{2.5}$	Minibus	68
					Bus	71
					Metro	61
Kulkarni and Patil, 1999	1995–1996	Bombay, India	Personal (48 h)	PM_5	Outdoor worker	322
Pfeifer et al., 1999	Summer 1996	London, UK	Personal exposure (7 days)	$PM_{2.5}$	Taxi driver	33.36
Adams et al., 2001	1999–2000	London, UK	In-vehicle (27 min)	$PM_{2.5}$	Bus (summer)	39.0
					Bus (winter)	38.9
					Car (summer)	37.7
					Car (winter)	33.7
Riediker et al., 2003	Fall 2001	Raleigh, NC, US	In-car (3 pm–midnight)	$PM_{2.5}$	Patrol trooper	23.0
			Roadside (same)	$PM_{2.5}$	Near traffic road	31.7
			Ambient (same)	$PM_{2.5}$	Fixed site	29.9

[a] Value for the zone with the highest integrated average estimated over a workday of 8 h.

from February to December 2001 in Nanjing, China (Wang et al., 2003) found that the daytime (8:30 am to 4:30 pm) ambient levels of PM_{10} and $PM_{2.5}$ were much higher than those found in other studies, with the highest traffic site having an average level of 423 and 632 ìg/m^3 for $PM_{2.5}$ and PM_{10}, respectively. Due to different emission inventory, physical characteristics and chemical composition of motor vehicle fuels, design and technology of motor vehicles, geographical and meteorological conditions, and socio-economic background, the physical and chemical characteristics of the ambient particulate matter in urban areas in China and other Asian countries may be different from those in the United States and European countries (Wang et al., 2003; Cohen et al., 2002).

Some traffic-related particulate matter exposure studies and results conducted were summarized in Table 5. Two UK studies and one US study were included in the table for comparisons (Pfeifer et al., 1999; Adams et al., 2001; Riediker et al., 2003). From Table 5, it can be seen that studies on exposure assessment on traffic-related airborne particulate matter, especially $PM_{2.5}$, are scarce in the developing world, and that particulate levels in developing countries are clearly higher than those in the developed world.

In Jamaica, measurement campaign was carried out to examine ambient level of TSP in the city (Davis et al., 1997). Though the average level was within the WHO standard of 60 μg/m^3, some sites with high traffic density had levels of TSP that exceeded this standard. An investigation in Dar es Salaam, Tanzania also measured ambient TSP levels in the city (Jonsson et al., 2004). TSP levels were found to be 40 and 20 μg/m^3 in urban and suburban areas, respectively. These low levels of pollutant may be due to low density of road traffic in the city. Generally speaking, studies on particulate matter in Africa are extremely scarce.

A study in Mumbai, India (Kulkarni and Patil, 1999) found that 48-h integrated exposure to respiratory particulate matter (PM_5) among 24 outdoor workers (traffic constables and outdoor workers) during winter months was 322 ìg/m^3. This was 2.25 times that indicated by ambient air quality monitoring data. This showed that ambient air monitoring through a few fixed sites cannot give accurate exposure data of the population, especially those subpopulations that are highly exposed. This conclusion was further confirmed by an intensive Hong Kong investigation (Chan et al., 2000) in which PM_{10} concentrations near busy streets were found to range from 25.56 to 337.4 μg/m^3 and these levels were significantly higher than the nearby fixed station data.

A pilot exposure study (Shendell and Naeher, 2002) conducted in three cities in Guatemala in May and June 1997 obtained ground $PM_{2.5}$ and CO levels in busy streets in these urban areas during work hours in the daytime. Levels of $PM_{2.5}$ measured in different zones were drastically different from each other, with the two zones having the highest integrated $PM_{2.5}$ levels of 90 and 100 μg/m^3 in Guatemala City and Quetzaltenango, respectively, while the other zones had low levels ranging between 5 and 60 μg/m^3. In an air pollutant exposure study carried out in Bangkok, Thailand (Leong et al., 2001) the 24-h time-weighted-average (TWA) in ambient air along busy streets was 84.33 μg/m^3.

In May 2002, an exposure study in Mexico City gave exposures of $PM_{2.5}$ for three transport modes during morning and evening rush hours, the arithmetic means (geometric means) of which were 68 (62), 71 (65), 61 (57) μg/m^3 for minibus, bus and metro, respectively (Gómez-Perales et al., 2004). In the study, the chemical composition of $PM_{2.5}$ was also investigated, which indicated that only 11% of $PM_{2.5}$ was elemental carbon (EC) compared with higher EC content in $PM_{2.5}$ in other studies. Since EC is considered an indicator of diesel engine emission, these results imply that, compared to gasoline consumption, diesel consumption in Mexico City may be rather low. The large contribution of sulfate (21%) was an indication of high volume of SO_2 that released into the atmosphere in Mexico City, mainly from industries and high sulfur content of the gasoline and diesel fuels.

Systematic data on particulate matter is scarce in Buenos Aires, the capital of and the largest city in Argentina. Nine-month monitoring data through a fixed site (Bogo et al., 2003) indicated that the average concentrations of PM_{10} in the summer (January to March) and winter (April to August) were 52 and 44 μg/m^3, respectively. $PM_{2.5}$ levels in the two seasons were 41 and 33 μg/m^3, respectively. These levels are clearly higher than US EPA annual standards for PM_{10} and $PM_{2.5}$. Indeed, particulate matter data are not systematic or even not available in many cities in the developing world. A report from Beirut, Lebanon (Chaaban et al., 2001) found that most of the daily TSP levels obtained from a fixed monitoring site in the city varied between 70 and 200 μg/m^3. The particulate matter may have reached an alarming level considering many other streets with much higher traffic intensities than the street in which the fixed site was located.

4. Carbon monoxide (CO)

In traffic-related exposure studies and epidemiologic investigations, another important pollutant is carbon monoxide (CO), which results from incomplete combustion of natural gas, diesel or gasoline in traffic engines. High concentrations of CO generally occur in areas with heavy traffic intensity and congestion. Point sources of CO emissions also include industrial processes, non-transportation fuel combustion, and natural sources such as wild forest fires. Indoor sources include leaking gas stoves, heaters, generators, etc. CO is a colorless, odorless and tasteless gas. Unlike NO_2, CO is comparatively stable in the air. After inhalation by the lungs, CO is absorbed by the blood and

inhibits oxygen transport by competing with oxygen for combining with hemoglobin and thus leads to hypoxia.

A number of studies have shown that CO pollution is a serious problem in urban areas around the world (Table 6), especially in big cities where traffic intensity is routinely high. Like many other air pollutants, CO levels in urban regions are highly influenced by such factors as traffic density, traffic congestion, and meteorological conditions. Ambient CO concentrations have daily and seasonal variations, as well as complex spatial distributions. Ambient concentrations, in-vehicle levels, and personal exposures of CO are generally higher in developing countries than in the developed countries, as has been frequently demonstrated by various kinds of studies (Table 6). This may have been caused by several factors including poor vehicle maintenance and insufficient use of vehicle emission control systems. Particularly, the study in Mexico City (Gómez-Perales et al., 2004) found lower CO levels in the three transport modes (15, 12, and 7 ppm for minibus, bus, and metro during the 180 min rush hours, respectively) compared with the previous findings (Fernandez-Bremauntz and Ashmore, 1995) in the same area, though the decrease may be partly due to seasonal difference. A pilot study in urban areas in Guatemala (Shendell and Naeher, 2002) also found low levels of CO (Table 6). Several studies conducted in the European countries and the US were also listed in the table to assist comparisons (Ashmore et al., 2000; Zagury et al., 2000; Duci et al., 2003; Riediker et al., 2003).

Traffic tunnels may accumulate CO due to their enclosed structure. A study in Hong Kong investigated CO levels in 11 tunnels (Chow and Chan, 2003). During rush hours, CO levels varied between 6 and 21 ppm on average. These levels ranged from 8 to 28 ppm in rush hours. The design of ventilation system is crucial to prevent accumulation of CO in these tunnels.

It has to be noted that study outcomes from different regions and countries may not be completely comparable. Due to differences in sampling method, sampling date, time and duration, sampling technique, traffic profile, and meteorological conditions, we can only get a larger picture from these comparisons. In the several commuting micro-environments studied, private cars and taxi provide highest exposure of CO to commuters compared with other transport modes, mainly public traffic systems (Chan et al., 2002; Duci et al., 2003). An investigation in Bangkok (Leong et al., 2001) revealed that emissions from two-stroke motorcycles were on average 1.5 and 5 times those from four-stroke motorcycles in terms of carbon monoxide and

Table 6
Traffic-related exposure studies on CO

Study	Year	Location	Exposure type	Sampler (duration)	Subject or place	Level (ppm)
Chow and Chan, 2003	Summer 1999	Hong Kong, China	Ambient	Metrosonics pm-7700 CO-meter (2–6 min)	Traffic tunnels Rush hours Non-rush hours	 6–21 8–28
Abdollahi et al., 1998	NA	Tehran, Iran	Ambient	Automatic carboxymeter (180 rush hours)	Central of the city	6.7–12.3
Venegas and Mazzeo, 2000	1994–1996	Buenos Aires, Argentina	Ambient	Non-dispersive infrared monitor (8 h)	Downtown street canyon	10.2
Leong et al., 2001	January–June 1999	Bangkok, Thailand	Ambient	Non-dispersive infrared monitor (8 h)	Busy street	6.15
Shendell and Naeher, 2002	May and June 1997	Three cities in Guatemala	Ambient	Langan Databear V (248–370 min)	Busy street	7.2[a] 10.9[b]
Atimtay et al., 2000	Spring 1998	Ankara, Turkey	Personal	Electrochemical sensor (8 h)	Traffic policeman	6.26–23.89
Fernandez-Bremauntz and Ashmore, 1995	Winter 1991	Mexico City	In-vehicle	Electrochemical sensor (38–99 min)	Minibus Bus Metro	32–63 26–38 17–25
Chan et al., 2002	Summer 2000	Guangzhou, China	In-vehicle	Electrochemical sensor (2.5 h peak)	Bus Taxi	8.6 23.7
Gómez-Perales et al., 2004	Spring 2002	Mexico City	In-vehicle	Electrochemical sensor (180 min peak hours)	Minibus Bus Metro	15 12 7
Zagury et al., 2000	Spring 1997	Paris, France	Personal	Pac II sensor (8 h)	Non-smoking taxi driver	3.8
Ashmore et al., 2000	Spring 1998	Northampton, UK	Personal	Pac III sensor (6–8 min in car traffic)	School children	4–7
Duci et al., 2003	Winter 1998–1999	Athens, Greece	In-vehicle	Electrochemical sensor (25–45 min)	Private car Bus Trolley	21.4 10.4 9.6
Riediker et al., 2003	Fall 2001	Raleigh, NC, US	In-car Roadside Ambient	Electrochemical sensor (3 pm–midnight)	Patrol trooper Near traffic Fixed site	2.6 1.1 0.8

[a] Value for the zones with the highest average over the whole monitoring period.
[b] Value for the zone with the highest Maximum average over 30 min.

hydrocarbon, respectively. In the same study, ambient average level of CO over 8 h in Bangkok streets was found to be low (6.15 ppm). It has been well established that fixed-site monitoring data of ambient CO level are not accurate and thus not suitable for exposure assessment of commuters. In a study among 49 occupational bus drivers in Tehran, Iran (Abdollahi et al., 1998), highest ambient levels of CO were found in the central parts of the city where traffic intensity was the highest. An increase of 6 ppm in ambient concentration of CO was associated with 1% elevation of blood carboxyhemoglobin. The scale of this association may change if personal exposures had been measured and used. Various models based on street canyon CO concentrations and surrogate factors (traffic factors, street canyon parameters, and meteorological data) are currently being evaluated for their abilities to predict air quality (Venegas and Mazzeo, 2000; Ashmore et al., 2000; Kukkonen et al., 2000; Manning et al., 2000).

5. Volatile organic compounds (VOCs)

Volatile organic compounds are a class of air pollutants sharing the same characteristic of high volatility in the ambient environment. Currently there are more than 300 different kinds of VOCs that can be detected by chromatog-

Table 7
Traffic-related exposure studies on VOCs

Study	Location	Exposure type (duration)	Subject or place	Pollutants	Level (ig/m^3)
Wang et al., 2002	China	Urban roadside (30 min)	Guangzhou	BTEX	51.5/77.3/17.8/81.6
			Macau	BTEX	34.9/85.9/24.1/95.6
			Nanhai	BTEX	20.0/39.1/3.0/14.2
Bae et al., 2004	Seoul, Korea	Indoor (8 h)	Shoe stall salesperson	BTX	732/6777/5382
Mukherjee et al., 2003	Kolkata, India	Personal (3–4 h)	Bus driver	Benzene	527.3
				Toluene	472.8
				o-Xlyene	1265.5
				p-Xylene	402.8
Jo and Yu, 2001	Taegu, Korea	Personal (7–8 h)	ETS bus driver	BTEX	28.1/88.7/8.1/30.2
			Non-ETS bus driver	BTEX	14.5/49.5/7.0/21.4
			ETS taxi driver	BTEX	44/141/10.2/37.3
			ETS taxi driver	BTEX	24.8/80.8/8.8/23.6
Jo and Song, 2001	Taegu, Korea	Personal (6–11 h)	Traffic policeman		
			Smoker	BTEX	35.3/114/7.8/22.1
			Non-smoker	BTEX	24.2/125/7.7/27.2
			Gas station attendant		
			Smoker	BTEX	84.4/141/12.9/55.1
			Non-smoker	BTEX	72.1/126/12.1/50.7
Romieu et al., 1999	Mexico City	Personal (workshift)	Gas station attendant	BTEX	310/680/110/490
			Street vendor	BTEX	77/160/28/128
			Office worker	BTEX	44/470/17/81
Bravo et al., 2002	Mexico City	Ambient (2–24 h)	Gas station	Benzene	82.4
				Toluene	319.8
Barletta et al., 2002	Karachi, Pakistan	Ambient (4–6 h)	Traffic street	BTX	16.6/26.8/8.2
Lau and Chan, 2003	Hong Kong, China	In-vehicle (30–50 min)	Non-A/C bus	BTEX	4.8/54.3/3.1/6.2
			A/C bus	BTEX	6.1/72.9/6.9/15.5
			Taxi	BTEX	5.9/43.5/4.4/7.8
Gómez-Perales et al., 2004	Mexico City	In-vehicle (3 h)	Minibus	Benzene	22
			Bus	Benzene	19
			Metro	Benzene	13
Chan et al., 2003	Guangzhou, China	In-vehicle (2.5 h)	Bus	BTEX	12.4/56.4/8.3/17.5
			Taxi	BTEX	33.6/108.5/20.3/43.2
Bono et al., 2003	Biella and Torino, Italy	Personal (8 h)	Gas pump attendant	Summer BTX	503/712/379*
			Policeman	Winter BTX	161/568/285*
			Policeman	Summer BTX	31/215/73*
				Winter BTX	21/144/150*
Edwards et al., 2001	Helsinki, Finland	Personal (48 h)	Non-ETS group	BTEX	2.6/17.1/3.3/15.1
			ETS exposed group	BTEX	4.7/73.6/14.6/66.5
Gonzalez-Flesca et al., 2000	Rouen, France	Personal (5 days)	Non-smoker	Benzene	10.3
Duffy and Nelson, 1997	Sidney, Australia	In-vehicle (45–60 min)	Non-catalyst-equipped	Benzene	153.7
			Catalyst-equipped	Benzene	70.6
Batterman et al., 2002	Detroit, USA	Vehicle/roadway (2–3 h)	Bus	BTEX	4.5/10.2/9/2.1
Riediker et al., 2003	Raleigh, NC, US	In-car, road, and ambient (3 pm–midnight)	Patrol trooper	BTEX	4/10.4/0.9/4.5
			Near traffic	BTEX	0.2/1.5/0.2/1
			Fixed site	BTEX	0.1/1.7/0.2/1

* Geometric means (the remaining are averages).

raphy. The concentration of VOCs in the air is determined by such processes as emissions, evaporation, deposition, and photochemical reactions under the sunlight. Among traffic-related VOCs, aromatic compounds, including benzene, toluene, ethylbenzene, and isomers of xylene (BTEX), namely *m*-, *o*-, and *p*-xylene, have public health importance. In urban regions, these aromatic VOCs are mainly released from traffic vehicles. Currently, traffic is a predominant source of ambient VOCs in many urban areas in industrialized countries. Besides the penetration of outdoor VOCs, indoor VOCs may also come from indoor tobacco smoking, household cleaning or degreasing, air freshening, domestic heating and cooking, painting, disinfecting, and varnishing, etc.

Among the BTEX compounds, benzene has been widely recognized as a human carcinogen (IARC, 2002) and the others also possess high toxicity, especially to central nervous system in humans. Therefore, this group of VOCs has received much attention in exposure assessment studies. Control plans have been established to lower their levels in ambient air. Just like particulate matter, personal exposure of VOCs cannot be accurately estimated based on several microenvironmental exposure levels and exposure time durations (Gonzalez-Flesca et al., 2000). Exposure data from stationary monitoring sites cannot give the real exposure profile in urban areas, since the level of traffic VOCs decreases drastically as the distance from the main traffic roads increases, causing high spatial variations in the distribution of VOCs. Indeed, the influence of industrial sources on VOC levels along traffic road seemed negligible (Batterman et al., 2002), indicating that air VOCs were so much limited to the small area around the source that even curbside levels were frequently found to have lower concentration of VOCs than the middle lanes of the main roads. Unsurprisingly, in several studies comparing VOC exposures in various commuting modes, roughly the same conclusion was reached: private cars or taxis were exposed to higher levels of VOCs than buses or trains (Chan et al., 2003; Lau and Chan, 2003). Such factors as traffic density, wind, temperature, and city buildings make the spatial variation even greater (Upmanis et al., 2001). Due to the difficulties in directly measuring these small-scale spatial variations, it is a promising job to investigate whether it is feasible to find some traffic indicators as surrogates for traffic-related VOCs and NO_2 exposures. Some good results have been produced but their potential for generalization needs to be validated in further studies (Carr et al., 2002). In one study, traffic volume and the percentage of traffic jam were able to account for 0.76–0.80 of the variability in concentration changes of benzene, toluene, and ethylbenzene. In addition, ambient VOC levels have clear seasonal variation and are higher in the winter season, as was observed in an exposure study in Greece (Kourtidis et al., 2002; Pankow et al., 2003; Ho et al., 2004).

Traffic-related VOC pollution has frequently been demonstrated to be a more serious problem in the developing countries than in the United States and Europe, as indicated by the VOC data obtained in Karachi, Pakistan (Barletta et al., 2002), a VOC exposure study in India which gave very high levels of VOCs (Mukherjee et al., 2003), and data on BTEX ambient air levels in three cities in Southern China (Wang et al., 2002). A study in Mexico City gave different levels of benzene at different monitoring sites, some of which were rather high (Bravo et al., 2002). Ten-year fixed-site monitoring data of VOCs in Mexico City (Arriaga-Colina et al., 2004) showed that total VOC levels in the city may have a decreasing trend due to the effective emission control measures, though it was still higher than in many other cities in the world. A personal exposure measurement campaign carried out among service station attendants, street vendors, and office workers discovered that BTEX exposures among service station attendants were the highest (310/680/110/490 ìg/m^3) (Romieu et al., 1999). One study in Australia (Duffy and Nelson, 1997) also reported very high in-vehicle benzene exposures. A study conducted in the US on traffic-related exposures among highway patrol troopers found low levels of VOC exposures (Riediker et al., 2003).

It has to be pointed out that most of these monitoring data are from fixed-site monitoring, while traffic-related exposure in traffic environment may be higher and pose a more serious threat to commuters and traffic-exposed workers. The comparisons between different studies and between developing and developed countries were summarized in Table 7. As indicated in the table, exposures to VOCs were very high for some occupations (petrol pump attendant and shoe stall salesperson).

6. Nitrogen dioxide (NO_2)

Nitrogen dioxide, one of the main traffic-related air pollutants and precursors forming photochemical smog (together with VOCs) and ground-level ozone, is currently under intensive investigations. The gas is reddish brown and highly reactive in ambient air. As one member of nitrogen oxides (NO_x), it undergoes a complex chain of chemical and photochemical reactions with nitric oxide (NO), ozone, and other gases. Usually the NO_2 in the atmosphere comes from two sources, either directly from emission sources (primary pollutant) or from chemical reactions in the atmosphere (secondary pollutant). Short-term exposures to the irritant gas may cause airway responsiveness and lung function injury. Long-term exposures may reduce immunity and lead to respiratory infections.

Sharing the same seasonal pattern with several other air pollutants, NO_2 level is usually higher in the winter than in the summer. Many studies showed that NO_2 concentration decreased drastically with increasing distance downwind from traffic (Gilbert et al., 2003; Singer et al., 2004). In a Canadian study (Gilbert et al., 2003), the authors found that wind and the logarithm of distance from a major highway

under study may serve as surrogates for traffic NO_2 exposure, which needed further validation. In this Canadian study, the NO_2 levels ranged from 11.9 to 29.3 ppb. Investigation conducted at schools near Northern California freeways (Singer et al., 2004) found highest NO_2 levels (24–30 ppb) in schools downwind and close to freeways.

In developing countries, exposure studies on NO_2 usually indicate higher exposure levels than in the developed world. In Tartu, Estonia, ambient level of NO_2 increased by 50% to 100% in several monitoring stations, according to the yearly monitoring data from 1994 to 1999. This increase may have been mainly caused by increasing number of vehicles, poor maintenance of many of these vehicles and narrow streets in the city (Kimmel and Kaasik, 2003). In 1998, transport emission accounted for 52% of the total NO_2 emission in the city.

In addition to conditions of vehicles, fuel type plays an important role in traffic emissions. The increasing use of compressed natural gas (CNG) attributed to decreases of ambient air pollutants in Delhi, India. From 1995 to 2001, the annual average concentrations of suspended particulate matter, CO, and NO_x decreased to 347 (from 405), 4197 (from 4681), and 34 (from 36) $\mu g/m^3$, respectively (Goyal and Sidhartha, 2003).

In a Korean study carried out in 32 shoe stalls beside busy streets in Seoul (Bae et al., 2004), NO_2 levels in shoe stalls and outside were found to be nearly equivalent (57.4 and 58.1 ppb, respectively). In this study, outdoor traffic-generated NO_2 may be the main source of indoor exposures. In another Korean study carried out in two cities (Son et al., 2004), 31 taxi drivers were monitored for their in-vehicle personal exposures to NO_2. At the same time the ambient level and their indoor residential level of NO_2 were also measured. It was found that personal exposure (30.3 ppb on average) among these drivers was about 1.6 times the indoor or ambient level. The high exposure mainly came from traffic emissions. One-year monitoring data from several intersections in Calcutta, India (Mondal et al., 2000) showed that NO_x levels were highest in the winter (222 $\mu g/m^3$) and lowest in the summer monsoon season (55 $\mu g/m^3$). In October 1998, an exposure study in Buenos Aires, Argentina (Fagundez et al., 2001) did not find elevated levels of ambient NO_2 at several monitoring sites in urban areas. No site had an average level of NO_2 higher than 40 ppb (Table 8). Low ambient levels of NO_2 (30.16 for 1 h on average) were also found in busy traffic streets in Bangkok, Thailand (Leong et al., 2001) monitored by a few fixed sites that were close to main roads in the city. Low levels of NO_2 exposures were found among patrol troopers during their work shift while driving the car patrolling highways in the United States (Riediker et al., 2003). These studies were summarized in Table 8. Also included in Table 8 was a study conducted in Australia for comparisons (Farrar et al., 2001).

Table 8
Traffic-related exposure studies on NO_x

Study	Location	Exposure type (Sampling duration)	Subject/Location	Pollutant	Level (ppb)
Son et al., 2004	Asan and Seoul, Korea	Personal and microenvironments (varies)	Taxi drivers	NO_2	30.3
			In-vehicle	NO_2	27.4
			Indoor home	NO_2	24.7
			Outdoor home	NO_2	23.3
Leong et al., 2001	Bangkok,Thaiand	Ambient (1 h)	Busy street	NO_2	30.2
Bae et al., 2004	Seoul, Korea	Indoor and traffic (8 h)	Shoe stalls	NO_2	57.4
			Busy street	NO_2	58.1
Mondal et al., 2000	Calcutta, India	Ambient (14-day passive sampling)	Intersection (Winter)	NO_x	222
			Intersection (Summer)		55
Fagundez et al., 2001	Buenos Aires, Argentina	Ambient (18–26 days)	Street level	NO_2	5–36
Chow and Chan, 2003	Hong Kong, China	Ambient (2–6 min)	Tunnels (Rush hours)	NO_2	366–538
			(Non-rush hours)	NO_2	349–478
Lin et al., 2004	São Paulo, Brazil	Ambient (24 h)	Fixed stations	NO_2	50.3
Ta et al., 2004	Lanzhou, China	Ambient (continuous)	Fixed sites (winter)	NO_2	36–47
			Fixed sites (spring)	NO_2	19–28
			Fixed sites (summer)	NO_2	14–28
			Fixed sites (fall)	NO_2	27–40
Kimmel and Kaasik, 2003	1999	Ambient (yearly average)	Fixed site	NO_2	15.8–46.8
Farrar et al., 2001	Perth, Australia	In-vehicle (1 week to 15 days sampling, reduced to 24-h averages)	Cityclippers	NO_2	37
			Suburban buses	NO_2	31
			Taxis	NO_2	15
			Bycycles	NO_2	22
			Couriers	NO_2	14
Riediker et al., 2003	Raleigh, NC, US	In-car, roadside, and ambient (3 pm–midnight)	Patrol trooper	NO_2	41.7
			Near traffic	NO_2	49.9
			Fixed site	NO_2	30.4
Singer et al., 2004	Northern California, US	Ambient (1-week passive sampling)	Schools (downwind) near a highway	NO_2	24–30

A study in Hong Kong (Chow and Chan, 2003) studied NO_2 levels in 11 traffic tunnels in the city. Very high concentrations of NO_2 were found (349–478 ppb during non-rush hours and 366–538 ppb in rush hours). It is not clear how these levels of NO_2 affect human health since vehicles usually pass these tunnels in minutes.

7. Polycyclic aromatic hydrocarbons (PAHs)

PAHs are a group of environmental contaminants that are formed during the incomplete combustion of fossil fuels, or other organic substances like tobacco or food. There are more than 100 different PAHs. PAHs that draw health concerns include acenaphthene, acenaphthylene, anthracene, benz[*a*]anthracene, benzo[*a*]pyrene (BaP), etc. Several PAHs are human carcinogens, the most famous being benzo[*a*]pyrene, a highly carcinogenic compound. Some others are mutagenic. A large amount of PAHs are from incomplete combustion in motor vehicle engines, especially heavy-duty and diesel engines (Barakat, 2002). Another main source is tobacco smoke. In the air, they usually coexist as a mixture and most of them are adsorbed on particulate matter, with only a small portion being in a gaseous state, depending on the ambient temperature. PAHs are mainly bounded onto fine respiratory particles (pPAHs), especially ultra-fine particles, though PAH accumulation may occur on larger particle sizes later in the atmosphere (Miguel et al., 2004).

Using a real-time photoelectric aerosol sensor which measured ambient air pPAH levels at an interval of 2 min, a study carried out in Tokyo and Bangkok during the summertime (Chetwittayachan et al., 2002) found that the roadside average concentration of particle-bound PAHs (pPAHs) in Bangkok (52 ng/m^3) was significantly higher than that in Tokyo (29 ng/m^3), but the pPAH level in the general areas in Bangkok (12 ng/m^3) was lower than that of Tokyo (19 ng/m^3). This may be due to the fact that road traffic in Bangkok had more trucks and heavy-duty vehicles, even though Tokyo had a higher percent area of traffic roads and a higher volume of vehicles. In addition, the two sites showed similar diurnal variations as the traffic volume changed. An average of 32 ng/m^3 of total pPAHs in the ambient air was found in a study in heavy traffic areas in Alexandria City, Egypt (Barakat, 2002). It also suggested that, compared with gasoline vehicles, diesel vehicles contributed more to the total emission of urban air PAHs. In the study, the sampling duration was 1 month, which may have influence on the determined concentration of PAHs because of the degradation of the samples. In Macao, China, data obtained from four fixed sites (excluding the fifth one which is a non-traffic site) in November 1998 indicated that the total concentrations of 13 PAHs were between the range of 13–51 ng/m^3 at night and 16–80 ng/m^3 day in the daytime (Qi et al., 2001). In Hong Kong (Guo et al., 2003), winter levels of 16 PAHs in total in $PM_{2.5}$ ranged between 3.1 and 330.3 ng/m^3 with an average of 41.75 ng/m^3. While these levels were much lower in the summer with a range of 3.21–6.93 ng/m^3 and an average of 4.87 ng/m^3. The PAHs in the study were mainly bounded to $PM_{2.5}$ in PM_{10}. These PAHs were mainly from vehicular exhaust.

An investigation conducted in Zagreb, Croatia (Šišovic et al., 2002) found that the mean summer level (0.05 ng/m^3 for BaP) of PAHs measured by a high volume sampler was much lower than that of the winter (5.12 ng/m^3 for BaP). The possible explanations may be that high summer temperature increased the degradation of PAHs and that winter PAHs also came from sources other than traffic, such as heating. Another explanation for low level of PAHs in summer may be that summer rainfall may clear pPAHs from the ambient air. The annual average of 1.87 ng/m^3 for BaP in this study was lower than annual standards set in many countries.

A PAH personal exposure investigation among 28 traffic policemen and 10 road builders during their working hours (Szaniszló and Ungváry, 2001) conducted in downtown Budapest, Hungary indicated that PAH exposure levels among these subjects (60.7 and 79.2 ng/m^3 for traffic policemen and road builders respectively) were comparable to those in most of the other European cities. But BaP exposure among traffic policemen (8.16 ng/m^3) was much higher than that among road builders (0.58 ng/m^3). This may be due to the fact that, unlike tar or tar pitch, the asphalt used in road paving is devoid of PAHs. In comparing results obtained from different studies, caution must be paid to seasonal difference and whether the levels measured are personal exposures or ambient levels. Another personal exposure investigation performed among 44 traffic policemen and 45 office policemen in Bangkok, Thailand found that PAH exposures among traffic policemen (72.79 ng/m^3) were much higher than those for office staff (6.88 ng/m^3) (Ruchirawat et al., 2002).

A measurement campaign in Salvador, Brazil found that average BaP level in ambient air near a bus station was 3.06 ng/m^3 in April, while the level in a traffic tunnel was 12.60 ng/m^3 in August (de P. Pereira et al., 2002). However, the two levels are not directly comparable, since they were measured in different seasons (Table 9). Another Brazilian study in São Paulo (Vasconcellos et al., 2003) found low levels of PAHs (Table 9) which may have been caused by rainfall during the sampling period and the limitations of the sampling method. A very high level in traffic roads came from a study in Mexico City (Marr et al., 2004), where median level of total particulate-bound PAHs ranged from 60 to 910 ng/m^3. In another study also conducted in Mexico City (Velasco et al., 2004), indoor, outdoor, and roadside PAHs in the ambient air were measured. Except for one indoor site (more than 1200 ng/m^3 in an underground parking lot), the average PAH level in outdoor environment beside busy traffic was the highest (582 ng/m^3). A very high level of in-vehicle exposure (with an average of 929 ng/m^3) to PAHs was found among five subjects in Taiwan

Table 9
Traffic-related exposure studies on PAHs

Study	Location	Exposure type (Sampling duration)	Subject or place	Pollutant	Time	Level (ng/m^3)
Ruchirawat et al., 2002	Bangkok, Thailand	Personal (3 h)	Traffic police	PAHs	Dry season	72.79
			Office police	PAHs	Dry season	6.88
Kuo et al., 2003	Taichung, Taiwan, China	Personal (6 h)	Incense smoke	PAHs	Not available	147
		Personal (1 h)	In-vehicle	PAHs		929
Szaniszló and Ungváry, 2001	Budapest, Hungary	Personal (work shift)	Traffic polices	PAHs	Jan.	60.7
			Road builders	PAHs	Jan.	79.2
			Traffic police	BaP	Jan.	8.2
			Road builder	BaP	Jan.	0.6
Qi et al., 2001	Macao, China	Ambient	Four traffic sites	13 PAHs	Winter night	13–51
					Winter day	16–80
Šišovic et al., 2002	Zagreb, Croatia	Ambient (24 h)	One fixed site	BaP	Summer	0.05
				BaP	Winter	5.12
Guo et al., 2003	Hong Kong, China	Ambient (24 h)	Two fixed sites	PAHs (in $PM_{2.5}$)	Winter	4.87
					Summer	41.75
Ho and Lee, 2002	Hong Kong, China	Ambient (24 h)	Fixed site with heavy traffic	BaP	Whole year	490
Vasconcellos et al., 2003	São Paulo, Brazil	Ambient (24 h)	Urban site	PAHs (in TSP)	Winter	3.1
			Vegetation area			2.73
			Forest area			1.92
de P. Pereira et al., 2002	Salvador, Brazil	Ambient (2–3 h)	Bus station	BaP	April	3.06
			Traffic tunnel	BaP	August	12.60
Velasco et al., 2004	Mexico City	Ambient (10–30 min)	Outdoor	pPAHs	December	17–582
Marr et al., 2004	Mexico City	Ambient (≥1 h)	Roadside	pPAHs	Winter and summer	60–910
Chetwittayachan et al., 2002	Bangkok	Ambient (7 days)	Roadside	PAHs	March	52
	Tokyo		Roadside	PAHs	August	29
Barakat, 2002	Alexandria City, Egypt	Ambient (30 days)	Roadside	pPAHs	July–August	32
Levy et al., 2001	Roxbury, Massachusetts	Fixed (7–11 am)	Bus terminal	PAHs	Summer	16
		Personal (7–11 am)	Bus terminal	PAHs	Summer	29
Lodovici et al., 2003	Florence, Italy	Ambient (24 h)	Traffic site	BaP	Winter	2.1

compared to personal exposure to PAHs from smoke of incense burning (average level of 147 ng/m^3) (Kuo et al., 2003). But the comparison of these findings with other studies is limited by a lack of description of the season during which the study was carried out.

Exposure studies on PAHs and the main findings were summarized in Table 9. Findings from a US study (Levy et al., 2001) and an Italian study (Lodovici et al., 2003) were also included for comparison purpose.

8. Monitoring methods

Despite the effort that has been made by epidemiologists from all over the world, health effects studies of particulate matter and gaseous air pollutants have been compromised by the difficulties in the exposure assessment. Traditionally, exposure assessment is focused on ambient air pollution levels, which can be easily obtained by establishing several fixed monitoring sites in the region of interest. Due to spatial variations of the pollutant levels in the study areas, usually exposure monitoring data obtained from these limited number of fixed-sites are not accurate enough for epidemiologic studies. Indeed, many studies have shown that ambient levels of air pollutants were poor predictors of personal exposures (Liard et al., 1999; Kousa et al., 2002; Adgate et al., 2002), a conclusion that was not surprising at all, considering the many microenvironments people occupy in daily life and work. In the Paris study (Liard et al., 1999), on average only 41% of the variance of personal NO_2 exposure among adult subjects could be explained by fixed-site monitoring data, while this percentage is 17% among children. The difference between adults and children may be due to different daily activity patterns. And for ozone, stationary measuring data were also a poor predictor of personal ozone exposure. A US study (Adgate et al., 2002) indicated that only 4% and 26% of personal $PM_{2.5}$ exposure variance were determined by outdoor and indoor levels respectively.

An alternative for this is the adoption of traffic indicators (population density and traffic intensity) and geographic information system (GIS), especially when traffic-related exposure is the main focus. By collecting traffic indicator information, personal exposure can be estimated. Several studies have used this technique in long-term exposure assessment of particulate matter and other gaseous pollutants (Hoek et al., 2002; Brauer et al., 2003). This method is suitable for long-term exposure assessment for large populations in urban areas. It is an effective method when the main goal is to estimate the exposure profiles of a certain area, and it is better in addressing spatial variations of air pollution levels in a certain area than fixed-site monitoring.

But it is not a method to estimate exposure levels for individuals.

The introduction of portable measuring equipment made personal exposure assessment feasible. This increasingly popular method, however, has received much criticism since it is expensive and not suitable for large population-based study and long-term monitoring. But it has its own advantages. One is that it measures personal exposure directly, which is the very need in rigorous epidemiologic studies. The second strength is that it can be used to address exposure issues for high risk populations. For instance, traffic policemen and drivers are much more exposed to traffic-related air pollutants. It is impossible to measure their exposure using large-scale cost-saving methods, since stationary monitoring is not feasible for these highly mobile groups.

In one investigation (Saarela et al., 2003), researchers had tried to measure VOC exposure levels of subjects in several microenvironments: home indoor, home outdoor and work place outdoor. At the same time, time activity diaries were maintained by each subject. To make comparisons, portable personal samplers were carried by the subjects continuously for 48 h for measurement of real personal exposure. They found that personal exposure of VOCs was on average 30% higher than the total exposure value calculated with microenvironment exposures and time durations in each of these microenvironments. Other VOCs exposure may come from commuting and other indoor environments, such as sports or recreation places. Therefore, even for people who are typically less mobile than drivers or traffic policemen, portable samplers are also an important tool in assessing their personal exposure levels.

The objective of health risk assessment for many pollutants is not only to protect the health of the general public, but to set limits for these pollutants at levels under which high risk populations and susceptible subpopulations are also acceptably safe. Therefore, personal exposure assessment has its unique advantages and therefore is irreplaceable in exposure assessment studies. However, compared with fixed-site studies, personal exposure assessment investigations are even more limited in the developing world. Actually, it has unique advantages to carry out epidemiologic studies utilizing personal exposure data in the developing world. With higher exposure levels in these underdeveloped countries, there is likely to be more chance to quantify the effect of individual pollutants. This necessity is further added by the need for scientific community to complete the dose–response profile through various epidemiologic investigations.

9. Conclusions

Exposure assessment and epidemiologic studies in the developing world are important and have advantages. On the one hand, increasing exposure data of traffic-related air pollution will provide scientific basis for pollution control in local areas. On the other hand, in-depth human health studies in these countries are necessary for assessing the degree of health outcomes of the public and for setting priorities in taking environmental control measures. In addition, epidemiologic investigations in regions with different metrological and socioeconomic backgrounds are helpful in strengthening scientific evidence about the association or causative relationships between these traffic-generated pollutants and various health endpoints.

In recent years, an increasing number of traffic-related pollution exposure studies and epidemiologic investigations have been reported, many of which are under the collaboration of researchers from developed countries and developing countries. Though the volume of scientific investigation on traffic-related air pollutants is increasing, exposure assessment and epidemiologic data are still not abundant. The differences among measuring methods and a lack of strict quality control in carrying out exposure assessment make it difficult for the findings to be generalized and the comparisons to be made between studies, which is especially true in exposure assessment research on particulate matter. Many of the existing epidemiologic investigations conducted in these underdeveloped regions suffer from inaccurate exposure assessment and insufficient control for potential confounders.

Therefore, future research in the developing world should emphasize the sharing of technical resources and communications between different countries and the use of standard measuring methods. Source-specific exposure assessment studies and studies using source-specific exposure data need to be more widely carried out since this serves directly as the basis for exposure regulations and public health measures. Source-specific studies are particularly crucial for particulate matter for an additional reason: different sources of particulate matter have different physiochemical compositions and thus different biological potentials (Laden et al., 2000; Janssen et al., 2002).

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