CHARACTERIZING ON-ROAD VEHICULAR EMISSIONS AND THEIR IMPACTS ON NEAR-ROADWAY AIR POLLUTION

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Traffic emission is a major source of urban air pollution. Vehicle is a significant contributor of carbon monoxide (CO), nitrogen oxides (NOx), black carbon (BC) and ultrafine particle (UFP) on-road and near-roadways. People living and working near roadways are facing health risks due to the exposure of elevated pollutant concentration. In order to develop cost-efficient strategies to mitigate near-road air pollution for protecting public health and promoting sustainable growth, it is imperative to characterize on-road traffic emission and its impacts on near-road air quality.

On-road chasing method was used in this study to investigate on-road vehicle emissions. This is the first application of this method in China. The method was developed and improved during a four-year field campaign and is an efficient approach to characterize emissions of a large number of on-road vehicles. Fleet average and individual emission factors (EF) are derived and reported. The large sample size significantly expands the database of real world Chinese vehicle emissions. Some of the results are reported for the first time in China: BC and UFP number EF of vehicles by type; EF of buses linked with emission standard; EF

distribution based on large sample size; contribution of “heavy emitter” to entire on-road emission; spatial variance of UFP number concentration at on-road, roadside and ambient environments.

My work suggests that diesel trucks are a major source of summertime BC in Beijing. Furthermore, “heavy emitter” accounts for a significant portion of BC emissions in Beijing and Chongqing. I also observed a clear downward trend of BC EF of diesel trucks in Beijing from 2008 to 2010, and of buses with more tighten emission standard. These observations indicate the effectiveness of traffic emission control measures (i.e. improvement of fuel quality in Beijing and enforcement of stringent emission standards) on BC EF reduction of diesel vehicles. However, comparison works between BC and NOx EF did not show NOx EF improvement as that of BC. The results and conclusions in this dissertation provide support for policy makers to evaluate and modify current traffic related air pollution control measures and propose future work.

BIOGRAPHICAL SKETCH

Xing Wang joined Sibley School of Mechanical and Aerospace Engineering of Cornell University in 2006. Prior to that he was at Tsinghua-BP Clean Energy Research and Education Center of Tsinghua University in Beijing, where he earned his Master of Science degree in 2006. His M.S. thesis focuses on the topic of the biomass energy utilization strategy in China. As a member of energy group, Mr. Wang participated the project of "Sustainable Urban Mobility" sponsored by BP in 2005. Mr. Wang earned his Bachelor of Science degree in Engineering in 2004, at the Department of Thermal Engineering of Tsinghua University in Beijing. His B.S thesis title is "Potential CO_2 Sequestration and Utilization Opportunities in China". Mr. Wang earned "Excellent Graduates Award of Tsinghua University" in 2004 to regard his excellent academic performance during his undergraduate life in the Department of Thermal Engineering.

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LIST OF ABBREVIATIONS

ABL: Atmospheric boundary layer

BC: Black carbon

BE: Badaling Expressway (equivalent to G6 BT in this study)

BJ: Beijing

China I~V: Stage I~V emission standards for Chinese vehicles

CNG: Compressed natural gas

CO: Carbon monoxide

CO_2: Carbon dioxide

CQ: Chongqing

EC: Elemental carbon

EF: Emission factor

Expy: Expressway

G1 BH: G1 Beijing-Harbin Expressway

G2 BS: G2 Beijing-Shanghai Expressway

G45 BK: G45 Beijing-Kaifeng Expressway

G6 BT: G6 Beijing-Tibet Expressway (equivalent to BE in this study)

HDV: Heavy duty vehicle

HDDT: Heavy duty diesel truck

HDDV: Heavy duty diesel vehicle

HB: Heibei

IM: Inner Mongolia

IR Expy: Inner Ring Expressway

LDV: Light duty vehicle

LDDV: Light duty diesel vehicle

LDGV: Light duty gasoline vehicle

NTC: Non traffic control days

PM: Particulate matter

$PM_{0.5}$: Particle with diameter less than 0.5 μm (equivalent to UFP in this study)

$PM_{2.5}$: Particle with diameter less than 2.5 μm

SD: Shandong

TC: Traffic control days

TC-I: Traffic control days period-I

TC-II: Traffic control days period-II

TJ: Tianjin

UFP: Ultrafine particle (equivalent to $PM_{0.5}$ in this study)

YLV: Yellow Environmental Protection Label Vehicles

6R Expy: The 6^{th} Ring Expressway

CHAPTER 1

INTRODUCTION

Urban traffic emissions are a major source of air pollution in many cities in developing countries, such as China (Fu et al., 2001; He et al., 2002) and India(Baidya and Borken-Kleefeld, 2009), and in developed countries(Shi et al., 2001; Twigg, 2007; Vestreng et al., 2009). It is estimated that over 35 million people estimated to live near roads and other major transportation facilities in the United States and countless others living, working or going to school in close proximity to traffic emissions throughout the world.

Researchers observed elevated concentrations of gaseous and particulate pollutants near roadways. Studies in the United States, Europe and Asia have reported that human exposure to traffic-related air pollutants is associated with a range of health effects (Garshick et al., 2003; Gauderman et al., 2007; Heinrich and Wichmann, 2004; Hoek et al., 2002; Maheswaran and Elliott, 2003; Peters et al., 2004; Wilhelm and Ritz, 2003, 2005). Human exposure to traffic-generated air pollutants near roadways has become a worldwide concern, especially in megacities encountering fast vehicle population growth associated with rapid economy growth and urbanization. Thus it is imperative to characterize on-road traffic emission and its impacts on near-road air quality, which is essential to develop cost-effective strategies to mitigate near-road air pollution for protecting public health and promoting sustainable growth.

Emission factor (EF), expressed as the weight of pollutant divided by a unit weight, volume, distance, or duration of the activity emitting the pollutant, is widely

employed to quantify vehicle emissions and generate emission inventories. Different approaches have been applied to evaluate vehicle EF, including chassis dynamometer test (Riddle et al., 2007; Schauer et al., 1999; Zielinska et al., 2004), remote sensing tests (Burgard et al., 2006; Chan and Ning, 2005), roadway tunnel tests (Ban-Weiss et al., 2009a, b; Geller et al., 2005), on-road chasing studies (Kittelson et al., 2004; Ronkko et al., 2006; Schneider et al., 2008; Westerdahl et al., 2005; Yli-Tuomi et al., 2005), and on-road testing with the portable emission measurement system (PEMS) (Johnson et al., 2009; Liu et al., 2009).

This dissertation presents a four-year field campaign conducted in Beijing and Chongqing, China. On-road chasing method was developed to investigate real-world on-road vehicle emissions. Near-road air quality monitoring was performed at various distances from roadways. Carbon monoxide (CO), nitrogen oxides (NOx), black carbon (BC) and ultrafine particle with diameter less than 0.5 μm (UFP or $PM_{0.5}$) are four major pollutants studied in this campaign. CO binds to haemoglobin modifying its conformation and reduces its capacity to transfer oxygen (Badman and JaffErnst, 1996). The reduced oxygen availability can affect the function of different organs especially high oxygen-consuming organs such as brain and heart(Kampa and Castanas, 2008). NOx has direct health implications such as the susceptibility to respiratory infections (Chauhan et al., 1998). It also act as precursors to photochemical formation of ozone (Marr and Harley, 2002; Sawyer et al., 2000), which can exacerbate chronic respiratory diseases and cause short-term reductions in lung function(Bernard et al., 2001); BC, a major component of diesel particulate matter, is reported as the second strongest contributor, after carbon dioxide (CO_2), to current

global warming (Ramanathan and Carmichael, 2008), and also affects the large-scale circulation and hydrologic cycle with significant regional climate effects (Menon et al., 2002); $PM_{0.5}$, usually possess tiny mass, but accounts for a major fraction of particle number concentration, is linked to respiratory and cardiovascular health effects by epidemiological and toxicological evidences (Chio and Liao, 2008; Fine et al., 2004).

Chapter 2 presents findings of the first step study. We characterized Beijing air quality under different micro-environments, i.e., on-road, roadside and ambient. The results demonstrate a strong traffic impact on CO, BC and $PM_{0.5}$ concentrations in Beijing. In this year, on-road chasing method was firstly used in China to investigate on-road vehicle emissions. Fleet average EFs of light duty vehicles (LDV) and heavy-duty vehicles (HDV) are reported. On-road BC EF and $PM_{0.5}$ number EF of Chinese vehicles are reported for the first time. Our results on CO EF agree with those derived from remote sensing and on-board vehicle emission testing systems, and hence validate this method as a feasible way to investigate real-world vehicle EFs.

Chapter 3 focuses on the second year campaign in the summer of 2008, when the 29^{th} Olympic Games were held in Beijing. A series of air pollution control measures were implemented prior to, during and post Olympics, providing a valuable opportunity to evaluate effectiveness of various traffic control measures in air quality improvement. Chapter 3 reports that significant reductions were observed in not only on-road vehicle EF but also ambient concentrations of traffic related air pollutants such as BC. Measurement of BC at different elevations also suggests that traffic activities contribute to a large portion of BC concentrations in the lower atmospheric

layer in Beijing during summertime. Our evaluation result was one of the earliest reports published focusing on air quality during Olympics.

Fast response instruments were added in the 2008 on-road chasing study, which allowed us to capture individual vehicle emissions. 19 diesel vehicles were sampled during a two-day pilot study.

The flexibility and cost-effectiveness of on-road chasing method drove us to expand the sample size in following studies, which was reported in Chapter 4. 230 individual diesel trucks and 57 individual buses were sampled on highways and city streets in Beijing in 2009. The sample size of our method is 10~25 times greater than chassis dynamometer test and PEMS, and it is more flexible to choose target sample vehicles than remote sensing and road tunnel test. Our results represent fleets of real trucks and buses during realistic on-road behaviors. The large sample size of individual diesel vehicles allowed us to reveal EF distributions of CO, BC, $PM_{0.5}$ number and mass, and quantify contribution of individual vehicles to total pollution emission. This unique work shows that a small number of "heavy emitters" are responsible for significant amount of total traffic generated air pollutants.

In Chapter 5, we present the work we had done in Beijing and Chongqing in 2010. Another important traffic generated pollutant, NOx, was measured in the campaign. NOx EF and BC EF are reported for 192 trucks in Beijing and 248 trucks in Chongqing. Comparison studies are performed between two cities, and between two pollutants (NOx and BC). The results indicate that stringent emission standard enforced in Beijing has effectively reduced BC emission, but it has limited impact on NOx emission reduction.

It would have been more valuable to connect the measured EF to vehicle identification information. As a proof-of-concept, we acquired registration information of 11 trucks in Chongqing and investigated how information of registration date and engine model might improve our understanding of emissions vs. control status of trucks. A downward trend of BC EFs is observed from China I to China III trucks, indicating the effectiveness of emission standard enforced in Beijing and nationwide.

Chapter 6 summarizes original contributions of my Ph.D. research. On-road chasing method was firstly adopted and developed to investigate on-road vehicles in China. Instructions and concerns of this method are presented based on my continuous experiments. This cost and time efficient method allowed us to sample large amount of individual on-road vehicles, which significantly enlarged the emission database for real-world diesel vehicles in different Chinese cities. A number of data in my work are reported for the first time in China: the on-road BC and $PM_{0.5}$ number EFs for Chinese vehicles by type, on-road EF of Chinese buses with various emission standards, spatial variance of $PM_{0.5}$ number concentration on different micro-environments including on-road, roadside and ambient. Evaluations of temporary and permanent traffic related air pollution control measures are presented. It could assist as data support for policy makers to evaluate and modify current work and propose future plans.

CHAPTER 2

CHARACTERIZATION OF ON-ROAD VEHICLE EMISSION FACTOR AND MICRO-ENVIRONMENTAL AIR QUALITY

2.1 Introduction

The transportation sector plays a vital role in every nation's economy. There are tremendous challenges in achieving sustainable transportation such as energy consumption and environmental pollution. The transportation sector consumes 21 percent of primary energy and account for 20 percent of greenhouse emissions worldwide (IEA, 2006). In recent years, studies in the United States, Europe and Asia have reported that human exposure to traffic-related air pollutants are associated with a range of health effects: adverse respiratory effects (Brauer et al., 2002; Garshick et al., 2003; Heinrich and Wichmann, 2004; Oosterlee et al., 1996; Peters et al., 1999); adverse effects on children's lung development(Gauderman et al., 2007); an increased risk of Cardiopulmonary and stroke mortality related to close proximity to traffic(Hoek et al., 2002; Maheswaran and Elliott, 2003); the onset of myocardial infarction within 1 hour after exposures to traffic during commuting activities (Peters et al., 2004); perinatal health effects (Ritz and Yu, 1999; Ritz et al., 2000; Wilhelm and Ritz, 2003, 2005).

The challenges facing the transportation sector are especially daunting for fast growing economies such as China and India, which have been undergoing motorization at a pace unprecedented in human development. The city of Beijing is

representative of rapid growing megacities of Asia. The vehicle population in Beijing has been increasing at an average annual rate of 14.4% since 1990 ((BSB), 2003), and is projected to maintain high levels of growth over the next two decades (Hao et al., 2006). Currently, traffic is one of the major sources of air pollution in the urban areas of Beijing (Hao et al., 2006; Hao et al., 2007). It is likely to dominate the urban anthropogenic emissions if, as planned industrial sources are relocated outside of Beijing, domestic usage of coal is banned and construction activities decrease (Guinot et al., 2007).

Beijing Municipal Government has been working on curtailing mobile emissions since early 1990s. The strategies and measures cover land use and traffic planning, emission control of in-use and new vehicles, fuel quality improvement, clean fuel vehicle technology and fiscal incentives (Hao et al., 2006). Beijing is the host city of the 29th Olympics Games in August 2008. The Olympics organizers will have to not only move athletes and spectators from various parts of the city to the Olympic venues in a reasonable amount of time, but also reduce the negative impact of transportation on air quality. Thus aggressive traffic intervention and emission control measures are expected, which will create a valuable case study to evaluate the effectiveness of the traffic and emission control measures on mitigating air pollution and protecting public health.

Evaluating the health benefits of traffic and emission control strategies requires accurate human exposure assessments, which involve tracking different individuals or targeted groups who encounter different levels of exposure resulting from differences in activity patterns. The levels of human exposure to traffic-related air pollutants vary

with different activities, e.g., driving on the road, living/working near or away from the road. For example, people are exposed to elevated air pollution within few hundred meters from roadways (Zhang et al., 2005; Zhang et al., 2004; Zhu et al., 2002). Wu et al. (2002) shows that the concentrations of PM_{10}, $PM_{2.5}$ and PM_1 decrease substantially on the top of a high-rise residential building compared to their ground-level values near roadways. Thus it is necessary to characterize the pollutant concentrations in different microenvironments.

In this chapter, we report the results and analysis from a recent field campaign we conducted in August 2007 in Beijing. The goal for this study is to characterize transportation-related air pollution in different types of microenvironments. We aim to answer the following questions: 1) what are the on-road emission factors of typical traffic-related pollutants? 2) What are the typical concentrations of traffic-related pollutants in different microenvironments? The results can be used to evaluate human exposure to traffic-related air pollutants and will also serve as a pre-Olympics baseline for comparisons with those during the Olympics (2008), and after the Olympics (2009). This chapter is organized as follows. First we describe the experimental methods, followed by discussions of the results and limitations. Finally we summarize our results and their implications in the conclusions.

2.2 Experimental Methods

2.2.1 Monitoring sites

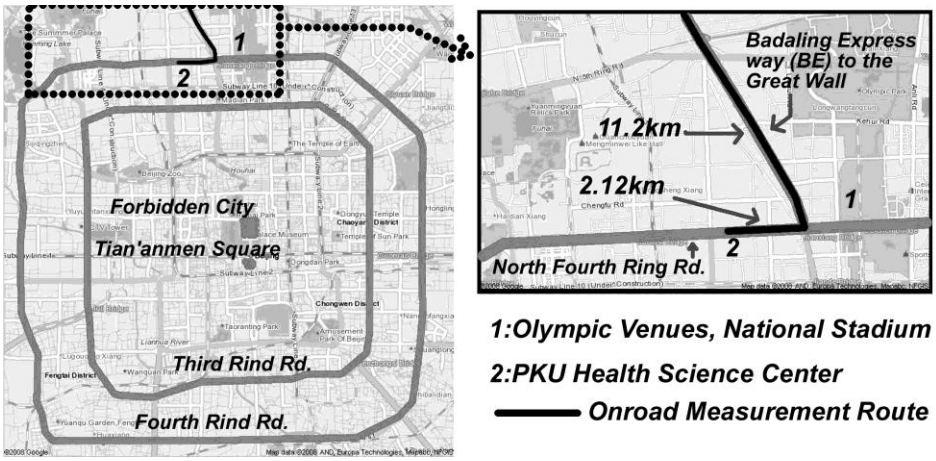


Figure 2.1 Sampling sites in Beijing
(left: Entire Beijing Map; Sampling sites)
Map source: http://maps.google.com

The field campaign was conducted in three different microenvironments: on-road, roadside and in community ambient air. The on-road measurements were conducted along the 4th Ring Road and the Badaling Expressway. The latter is a major truck corridor in the northern part of the city. The routes of the mobile laboratory monitoring activities are shown in Figure 2.1. The roadside and ambient sampling sites were located on the campus of Peking University Health Science Center (PKUHSC), which is approximately 3.2 km from the National Stadium and other main sports venues of the Olympic Games. The main roadside sampling site was on the sidewalk outside the north gate of the PKUHSC campus (39°59′07″.3N, 116°21′07″.6E), approximately 30 m from the center line of the 4th Ring Road and less than 1 m from the curb. The other roadside sampling site was on the second floor of a

student dormitory (39°59′05″.2N, 116°21′07″.8E), which is ~95 m away from the centerline of the 4th Ring Road. The ambient sampling site was at the athletic field of the campus (39°58′59″ N, 116°21′11″.3E), 280 m away from the centerline of the 4th Ring Road and at least this distance from other arterial or busy roads. Campus and near campus locations were approximately 50 m elevation above sea level.

2.2.2 Sampling period and traffic interventions

The measurements reported here were performed during two weeks in summer 2007, August 12 - August 24, 2007. During this period we conducted ambient air quality assessments and evaluated the impacts of two types of regulatory actions that were expected to impact air quality.

A ten-year-old regulation enforced by the Beijing Traffic Management Bureau forbids heavy-duty trucks from entering the city center (i.e., on or within the 4th Ring Road) before 11:00 pm and after 6:00 am. As a result, most traffic is light duty gasoline powered vehicles with very few diesel trucks on the highways during the daytime. In comparison, the number of diesel trucks on the Badaling Expressway and other major highways rise sharply after 11:00 pm. Diesel powered buses do use these highways and other roadways during the daytime. Accordingly, we conducted the on-road and ambient measurements during both the daytime and nighttime to capture the impact of this regulation on air quality.

From August 17 to 20, 2007, the Beijing Municipal Government conducted an experiment to test the effectiveness of traffic controls in improving air quality and easing the traffic congestion during the Olympics. It is estimated that around 1.3

million vehicles - nearly half of the total 3 million in the city were ordered off the roads each day based whether their licenses were even or odd. On August 17 and 19 (Friday and Sunday), only vehicles with the license plate ending with the odd numbers were allowed on the road. On August 18 and August 20 (Saturday and Monday), only those ending with the even numbers were allowed on the road.

Ambient and on-road measurements were conducted on both normal traffic days and traffic control days while the roadside measurements were only done on two normal days due to a limited number of instruments.

2.2.3 Instrumentation and methodology

The same sets of instruments were employed in the ambient, roadside and on-road measurements. The instruments included a Fast Mobility Particle Sizer (FMPS, TSI, Model 3091) for ultrafine particle (UFP) number concentrations and size distributions in range 5.6-560 nm; an Aethalometer (Magee Scientific, Model AE42) for the concentration of black carbon (BC); a DustTrak (TSI, Model 8520) fitted with a 2.5 micron cyclone and a diffusion dryer (Sioutas et al., 2000) for the mass concentration of $PM_{2.5}$; a Qtrak (TSI, Model 7655) for CO and CO_2 concentration; a Nanoparticle Surface Area Monitor (NSAM, TSI, Model 3550) for nanoparticle lung-deposited surface area. Data were recorded at 1 second interval by FMPS, 10 seconds interval by DustTrak, Qtrak, and NSAM, and two minutes interval by Aethalometer. The Q-Traks were recently calibrated by the manufacturer. A laptop was used to control and log data from the FMPS and NSAM. In addition, meteorological parameters including the temperature and relative humidity were recorded during the monitoring days. During

the on-road measurements, a portable GPS (Garmin Model 60 CSx) was used to record the locations, routes and vehicle speed with 10 second frequency. A digital camcorder was held by the staff in the front seat of the van used for on-road monitoring and recorded the traffic conditions continuously during the on-road measurements.

During the roadside measurement instruments were put on a cart and powered by a deep cycle battery connected to an inverter. Measurements were performed in the afternoons of August 12, 2007 and August 13 2007 from 4:00 pm to 6:00 pm, a period where few heavy duty diesels were present. During the measurements, the traffic volume on the south bound of the 4th Ring Road was counted by the staff approximately every half hour. The total traffic volume was roughly twice as much because the traffic volumes of both directions were similar during the sampling period.

At the ambient sampling site, all the instruments were placed in a room on the rostrum of the athletic field, around 5 m above the ground and approximately in the center of the campus. No pollutant sources were observed in the vicinity of this site. The window of the room faces to the athletic field and the inlets of the instruments were connected to short (<1m) conductive tubing and run out the window. Measurements were performed at this site from August 18th 2007 to August 24th 2007, during which there were three traffic control days (August 18th to August 20th, 2007).

The mobile platform was a gasoline-powered minivan, and the instruments were powered by the battery/inverter power supply. The on-road sample air was taken through a side window. The van was operated at highway speeds and self pollution appeared unlikely to have had a major impact. Measurements were conducted on two

normal days (August 14th and 15th 2007) and one traffic-control day (August 17th 2007), when the minivan ran three roundtrips during each sample run along the routes during both daytime and nighttime. After each trip, the van parked near the ambient site for ten minutes measuring the ambient concentrations. The route, shown in Figure 1 consists 22.4 km on the Badaling Expressway and 4.2 km on the 4th Ring Road.

2.3 Results and Discussion

2.3.1 On-road emission factors of gasoline and diesel vehicles in Beijing

The fleet average emission factors (EF) were calculated based on the data collected during the on-road measurements. By performing a carbon balance on the fuel combustion process, one can relate the emission of carbon-containing species from vehicle exhaust to fuel consumption (Geller et al., 2005; Kirchstetter et al., 1999; Stedman, 1989). We consider CO_2, CO and BC as the carbonaceous products during the combustion process. We did not have direct measurement of hydrocarbon (HC) gases. Neglecting HC should have very minor effects since carbonaceous products are usually dominated by CO_2 and CO. The fuel based emission factor can be calculated in the following equation:

$$EF_p = \frac{\Delta[P]}{\Delta[CO_2]\times\frac{MW_C}{MW_{CO_2}} + \Delta[CO]\times\frac{MW_C}{MW_{CO}} + \Delta[BC]\times\frac{MW_C}{MW_{BC}}} \times w_c , \quad (1)$$

where $\Delta[i] = [i] - [i]_0$; i = P, CO_2, CO and BC; subscription 0 denotes the baseline value; EF_p is the emission factor of pollutant P in grams of pollutant emitted per kilogram of fuel consumed (g kg^{-1}); $\Delta[P]$ is the on-road concentration of pollutant P

above the baseline, with the unit of grams per cubic meter of air (g cm^{-3}); $\Delta[CO_2]$, $\Delta[CO]$ and $\Delta[BC]$ represent the increase of CO_2, CO and BC, with the unit of grams per cubic meter of air (g cm^{-3}); MW_i is the molecular weight of the i^{th} species; w_c is the mass fraction of carbon in the fuel.

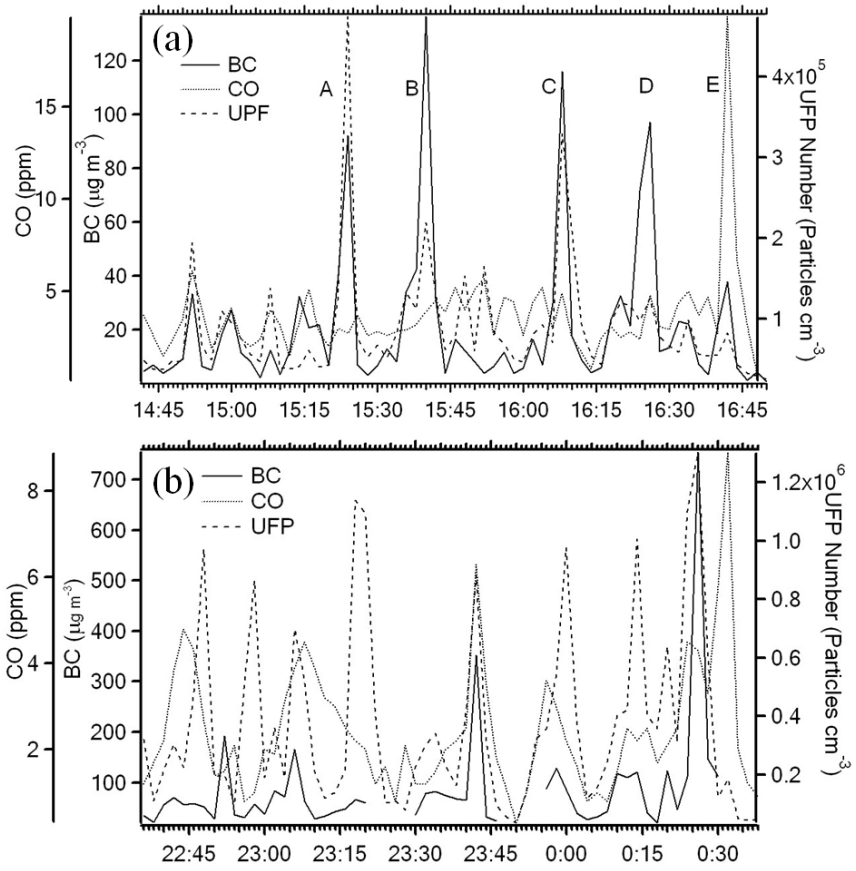


Figure 2.2 On-road measurements of CO, BC and UFP number concentrations (Panel a: afternoon; Panel b: night. Sampling date: August 14, 2007. Heavy-duty diesel vehicles are typically banned from entering the city from 6:00 am to 11:00 pm.)

Figure 2.2 depicts the results from a typical daytime and a nighttime trip during the on-road measurements. To calculate the emission factors, the baseline concentrations were taken to be the ambient concentrations recorded at the end of each

trip. The increase of the on-road concentrations above the baseline was considered as the traffic contribution. Typically, the traffic on Badaling Expressway was dominated by light-duty gasoline vehicles (LDGVs) during daytime and by heavy-duty diesel vehicles (HDDVs) at nighttime after 11:00 pm due to the regulation described in Section 2.2.2. This general pattern enabled us to differentiate the emission factors of LDGVs and HDDVs. We grouped the on-road concentrations in two-minute intervals and then calculated the emission factors for each interval during the whole trip. For LDGVs, the emission factors were based on the daytime data collected when the van was on the Badaling Expressway. For HDDVs, the median value was based on the nighttime (after 11:00 pm) data collected when the van was also on the Badaling Expressway but few LDGVs were present. The traffic information during the on-road measurements were provided by the camcorder in the van. To represent the fleet average, the emission factors were taken to be the median values of each dataset. Extreme concentrations such as those labeled A-E in Figure 2.2 (a) were excluded in calculating the fleet average.

The extreme concentrations indicate the existence of so called “heavy emitters” in the on-road fleet. Since there were multiple vehicles in a 2-minute interval, the actual heavy emitters were visually identified from the videos recorded during the on-road measurements in conjunction with the measured concentrations, where heavy emitters usually led to pronounced “peaks” (Figure 2.2). In the daytime measurements (Figure 2.2a), the BC concentration peaks (such as those labeled A to D) coincide with UFP number concentration peaks and were caused by the HDDVs in violation of the traffic control regulation. During the nighttime (Figure 2.2b), the BC and UFP peaks also

coincide but with much greater values than their daytime concentrations, due to the dominant HDDV traffic at the entrance of the 4th Ring Road.

Table 2.1 On-road emission factors of fleet-averaged light-duty vehicles (LDGVs) and heavy-duty vehicles (HDDVs) and heavy-emitting vehicles

Pollutant	Unit	LDGV	HDDV	Heavy Emitters
CO	g kg^{-1}-fuel	95	50	136
BC	g kg^{-1}-fuel	0.3	1.4	3.1
UFP	# kg^{-1}-fuel	1.8×10^{15}	1.1×10^{16}	2.1×10^{16}

Table 2.1 lists the CO, BC and particle number emission factors for fleet-averaged LDGVs and HDDVs, and heavy emitters. As expected, HDDVs emitted lower CO, higher BC and higher numbers of particles than LDGVs. In our literature search, we could only identify a limited number of on-road emission factors for Chinese vehicles and all the reported values were for gaseous species (Chen et al., 2007b; Wang et al., 2005b; Zhou et al., 2007). The BC and particle number emission factors derived in this study are therefore unique.

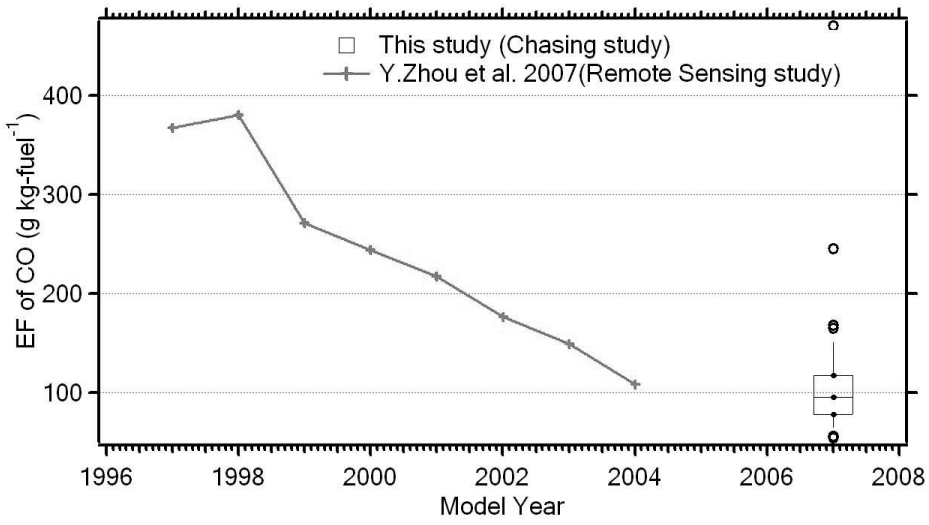


Figure 2.3 CO emission factors from this study and Zhou et al. (2007)

Figure 2.3 compares the CO emission factors derived from our study with those

measured by Zhou et al. (2007)using remote sensing techniques. We did not identify the model year of the on-road vehicles during our measurements. We plotted the emission factors in box and whisker showing a range of values with the median representing the fleet average. The comparison suggested 1) that CO emission factors show a clear downward trend since mid 1990s, which is likely a result of increasingly stringent emission standards and adoption of advanced emission control technologies by the Beijing government, and 2) that the fleet average derived in this study was equivalent to that of 2004 model year, which indicates that our results are in good agreement with those from the remote sensing measurements. All automobiles sold in China are equipped with catalysts that reduce CO emissions.

Similar downward trends of CO emissions were observed in other Chinese and US cities (Bishop and Stedman, 2008; Guo et al., 2007). Guo et al. (2007) conducted remote sensing studies in Hangzhou, China during 2004 and 2005, showing that the CO emission factors decrease sharply with increasing vehicle model years, especially for the model year later than 2001. Bishop et al. (2008)reported a similar trend of CO emission factors in several cities in the US during past decade.

Table 2.2 Comparisons of on-road CO emission factors from this study and previous studies

Vehicle Type	Location	Year Tested	CO g kg^{-1}-fuel	Source	Method
LDGV	Beijing, China	2007	95.3	This study	Mobile Platform
LDGV	Beijing, China	2004	109-271[a]	Zhou et al. (2007)	Remote Sensing
LDGV	7 Californian cities	1991	122	Singer, et al. 1996	Remote Sensing
LDGV	Denver, CO	2001	53±11	Pokharel, et al. 2002	Remote Sensing
HDDV	Beijing, China	2007	50	This study	Mobile Platform
HDDV	Shanghai, China	2006	47	Chen et al. (2007)	On-board testing system
HDDV	Dumont, CO	2005	37.9±1.6	Burgard, et al. (2006)	Remote Sensing

Mix	Mexico City	2003	190	Jiang et al. (2005)	Mobile Platform
Mix	Los Angles	1997	109	Singer and Harley (2000)	Remote Sensing
Mix	Chicago	2006	<20	Bishop, et al. (2008)	Remote Sensing

Table 2.1 compares the on-road CO emission factors from vehicles in China, Mexico, U.S., and European countries. It is worth noting that the CO emission factors for HDDVs from this study agree with that from Chen et al., (2007b) via on-board vehicle emissions testing system, which further demonstrated the validity of our method. The comparisons also showed that even with the tremendous progress in the past decade in reducing CO emission in China, the fleet-average CO emission factor for LDGVs in Beijing is still about twice as larger as that in the U.S, but substantially lower than that measured in Mexico City.

Table 2.3 Comparisons of on-road BC and particle number emissions factors from this study and previous studies in U.S.

Vehicle Type	Location	Year tested	BC (g kg^{-1}-fuel)	Particle Number (# kg^{-1}-fuel)	Source
LDGV	Beijing, China	2007	0.3	1.8×10^{15}	This study
LDGV	California, USA	2004	0.003	2.5×10^{15}	Geller et al.(2005)
LDGV	California, USA	1997	0.035	4.6×10^{14}	Kirchstetter et al. (1999)
LDGV	California, USA	1996	0.03	N/A	Miguel et al.(1998)
LDGV	Minnesota, USA	2000	N/A	$6.6x10^{15}$	Kittelson et al. (2004)
LDGV	Mexico City, Mexico	2003	0.27	N/A	
LDGV	Austria	2002	$\sim10^{-4}$		Laschober et al. (2004)
HDDV	Beijing, China	2007	1.3	1.1×10^{16}	This study
HDDV	California, USA	2004	0.78	8.2×10^{15}	Geller et al.(2005)
HDDV	California, USA	1997	1.3	6.3×10^{15}	Kirchstetter et al. (1999)
HDDV	California, USA	1996	1.44	N/A	Miguel et al.(1998)
DBus	Australia			1.5×10^{15}	Jamriska et al. (2004)

Table 2.3 compares the BC and particle number emission factors from this and previous studies conducted in other countries. For light-duty vehicles, the BC

emission factor is around 10 times higher in Beijing than those measured in U.S, and is similar to what has been measured in Mexico City. The particle number emission factor for light-duty vehicles in Beijing is actually slightly lower than the current values in U.S. For heavy-duty vehicles, both the BC and particle number emission factors are higher in Beijing than those in U.S, respectively. Since BC particles can provide surface areas for the condensation of volatile materials necessary to grow ultrafine particles as well as for coagulation of ultrafine particles, the higher BC emissions may actually contribute to the lower particle number emissions for light-duty vehicles in Beijing. This effect may not be significant for heavy-duty vehicles since the BC emission factors are only slightly higher for Beijing vehicles than those in U.S.

The high BC emissions factors of Chinese LDGVs may have important implications. BC particles affect the global and regional climate by strongly absorbing solar radiation (Liousse et al., 1996). The large fraction of BC in the aerosols in China and India were shown to have significant regional climate effects (Menon et al., 2002). The BC emission factors of LDGVs used to prepare the current emission inventory in China are in the range of 0.006-0.08 g kg^{-1}-fuel (Cao et al., 2006). Those values were taken from the measurements conducted in U.S. (Kirchstetter et al., 1999; Streets et al., 2001) and India (Reddy and Venkataraman, 2002). The emission factor from our study, 0.3 g kg^{-1}-fuel, was about five times higher than the upper limit, 0.08 g kg^{-1}-fuel. Thus the relative contribution of LDGVs to the total BC emissions in China may have been underestimated. It is worth noting that the BC emissions factor of HDDVs is in the range of 0.11 to 2 g kg^{-1}-fuel (Cao et al., 2006). The corresponding value from

our study, 1.3 kg^{-1}-fuel, falls within that range.

2.3.2 Micro-environmental Air Quality in Beijing

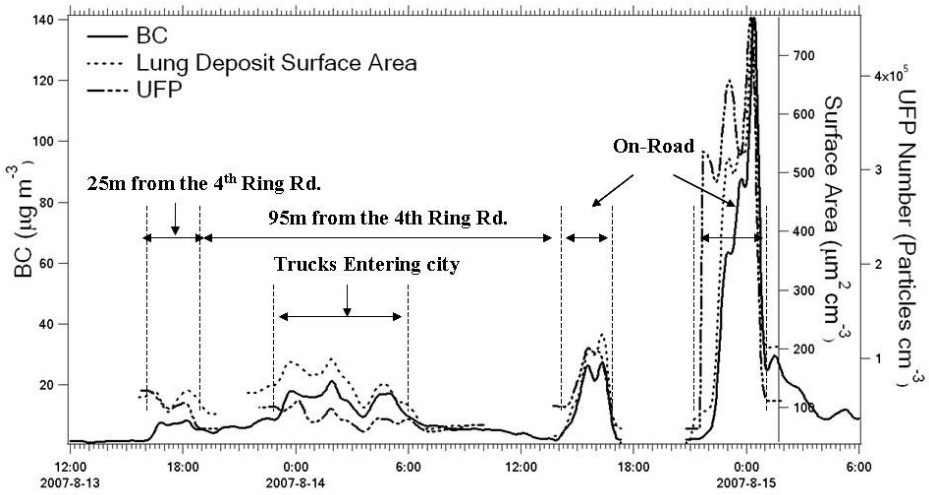


Figure 2.4 BC, Surface Area and UFP number concentrations in different microenvironments

Figure 2.4 illustrates the results from the continuous measurements of BC, Surface Area and UFP number concentrations under different microenvironments from August 13 to 15, 2007. The impact of HDDV activities was clearly demonstrated. First, data collected 100 m from the 4th Ring Road clearly suggested that pollutant concentrations started increasing at about 11:00 pm and fell back around 6:00 am in the morning, coinciding with the period when trucks were allowed to enter the city. Second, the on-road concentrations measured during daytime when LDGVs dominated the traffic were much lower than those measured during the nighttime with HDDVs dominating the traffic. Thus, drivers on the road and residents living near the roadways are likely to be exposed to elevated diesel particulate matter at night.

Table 2.4 Concentrations in different microenvironments and periods (TC: Traffic Control)

	BC ($\mu g\ m^{-3}$)	CO (ppm)	UFP (10^4 particles cm^{-3})
Ambient (no TC daytime)	3.4(2.8-5.4)	0.3(0.2-0.9)	2.72(2.47-3.23)
Ambient (TC daytime)	4.8(4.0-7.7)	1.1(0.8-1.3)	1.43(1.10-1.92)
Ambient (no TC nighttime)	10.5(6.0-14.9)	0.1(0.0-0.8)	1.91(1.65-2.38)
Ambient (TC nighttime)	7.3(6.2-9.0)	0.9(0.6-1.0)	1.46(1.27-1.82)
Roadside (no TC daytime)	4.6(2.6-7.0)	1.4(0.9-2.6)	3.84(2.26-4.96)
On-road (no TC daytime)	21.9(16.7-32.5)	4.0(3.0-5.2)	8.17(5.65-12.7)
On-road (TC daytime)	18.3(14.5-22.0)	3.7(3.3-4.0)	7.82(6.12-10.9)
On-road (no TC nighttime)	60.1(46.6-82.8)	2.6(2.0-3.1)	29.3(15.6-50.7)
On-road (TCD NT)	46.0(26.7-49.2)	2.8(2.4-3.2)	24.3(16.8-35.9)
Heavy emitter (daytime)	89.6	3.0	47.7
Truck jam (nighttime)	330.7	6.3	88.2

The median, lower quartile and upper quartile values of pollutant concentrations under different microenvironments were reported in Table 2.4, where micro-environments were categorized as ambient, roadside, on-road and some other specific conditions, and further divided into daytime and nighttime, normal days and traffic control days. As the distance from the road increases, the concentrations of each pollutant decrease significantly. For example, during daytime on the normal days, the ratios of on-road concentrations to roadside concentrations were 4.8, 2.9 and 2.1, for BC, CO and UFP number concentrations, respectively; the ratios of on-road concentrations to ambient concentrations were 6.5, 13.3 and 3.0. The nighttime on-road to ambient ratios were 5.7, 23.6 and 15.3, for BC, CO and UFP, respectively. For some specific situations, such as heavy emitting trucks passing by or the presence of truck jams, the exposure levels to BC and UFP could reach as high as 30-60 times the ambient level, a potential public health concern for truck drivers or drivers on the road.

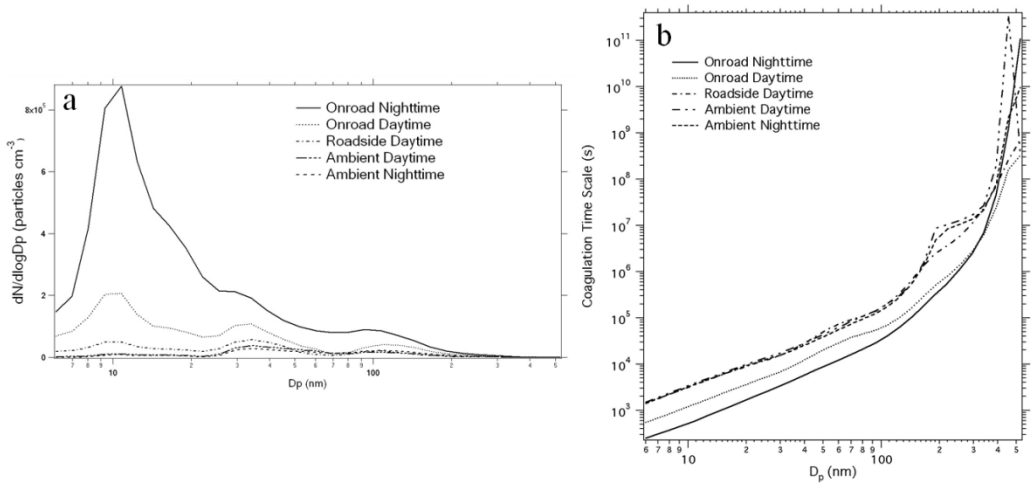


Figure 2.5 Particle size distributions in different environments (a) and coagulation timescale as a function of particle size in different types of micro-environments (b)

Figure 2.5a illustrates the average particle number size distributions in different environments and during different times. First, there are three major modes in the particle size distributions, centering around 10 nm, 30-40 nm, and 90-100 nm. The size distributions present a sharp gradient from on-road to roadside, to ambient conditions, mostly due to the dilution process, indicating that vehicle emissions make a great contribution to the UFP concentrations, especially the particles around 10 nm. The elevated UFP number concentrations during nighttime were largely due to the truck activities as mentioned earlier.

Furthermore, the concentrations of 10-nm mode particles decrease much faster than 100-nm mode particles, a trend similar to what we observed in Los Angeles (Zhang et al., 2004). At ambient level, the 10 nm-mode almost disappears. There are several atmospheric processes in addition to dilution that may contribute to this phenomenon. The 10 nm-mode particles coagulate with larger ones, and volatile and

semi-volatile materials may evaporate from 10 nm-mode particles, shrinking the particle size below the detection limit (Zhang and Wexler, 2004; Zhang et al., 2004).

We estimated the coagulation timescale, $\tau_{coag,i}$ for the smaller particles in size bin *i* coagulating with larger ones as:

$$\tau_{coag,i} = \frac{1}{\sum_{i<j} K_{i,j} N_j}, \quad (2)$$

where $K_{i,j}$ is the coagulation coefficient for particles in bin *i* coagulating with those in bin *j*, and N_j is the particle number concentrations in bin *j* (Zhang and Wexler, 2002). Figure 2.5b illustrates $\tau_{coag,i}$ as a function of particle size. The values of $\tau_{coag,i}$ range from few minutes to about half hour for the on-road size distributions, which indicates that coagulation may play a role in the aerosol dynamics. Evaporation of 10-nm size particles could be another possible mechanism in explaining the dynamics behaviors of exhaust particles in Beijing. However, we did not observe significant size shifts from on-road to ambient conditions in Beijing. If the particles were volatile, we should expect them to grow or shrink as exhaust moves away from roadways. Researchers in Southern California demonstrated that traffic-generated ultrafine particles are volatile in nature, and that the volatile composition increases as the particle size decreases (Kuhn et al., 2005a; Kuhn et al., 2005b). This type of experiment and chemical characterization of mobile emissions (Huang et al., 2006) need to be conducted in China to evaluate the volatility of traffic-generated particles, which has important health effects implications as particle volatility can potentially affect the toxicity (Kleinman et al., 2007).

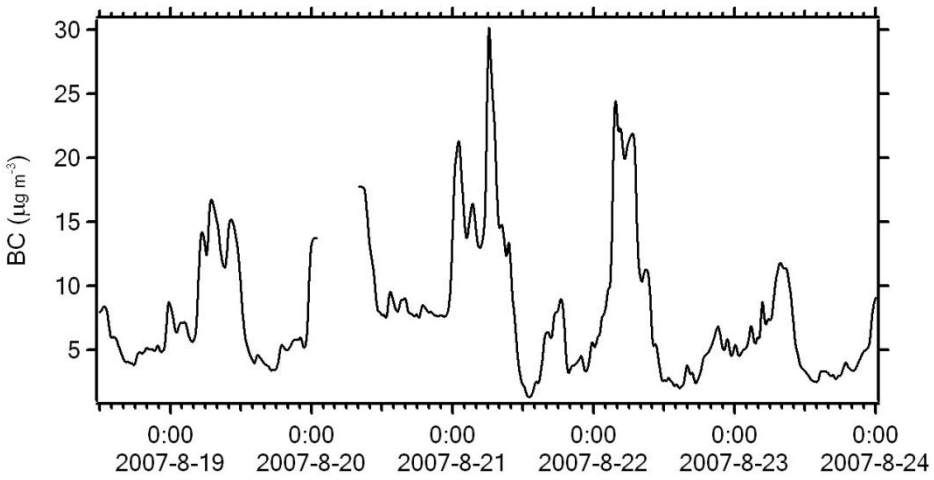


Figure 2.6 Diurnal pattern of BC concentration at ambient community

Figure 2.6 depicts the BC concentrations measured at the ambient site from August 18 to August 24. The diurnal patterns indicate that BC concentrations started building up from ~ 11:00 pm. Even though the presence of an inversion structure in the boundary layer would contribute to the elevated BC concentrations at nights (Wang et al., 2005a), the sharp gradient in the BC profile around 11:00 pm strongly suggests that the HDDV activities may play an important role in the observed BC diurnal patterns.

2.3.3 The impacts of traffic reduction experiment in summer 2007

As we mentioned in Section 2.2.2, a traffic control experiment was conducted from August 17 to August 20, 2007. The comparison of pollutant levels on normal days (i.e., without traffic control) and traffic control days are shown in Figure 2.7 as well as Table 2.4.

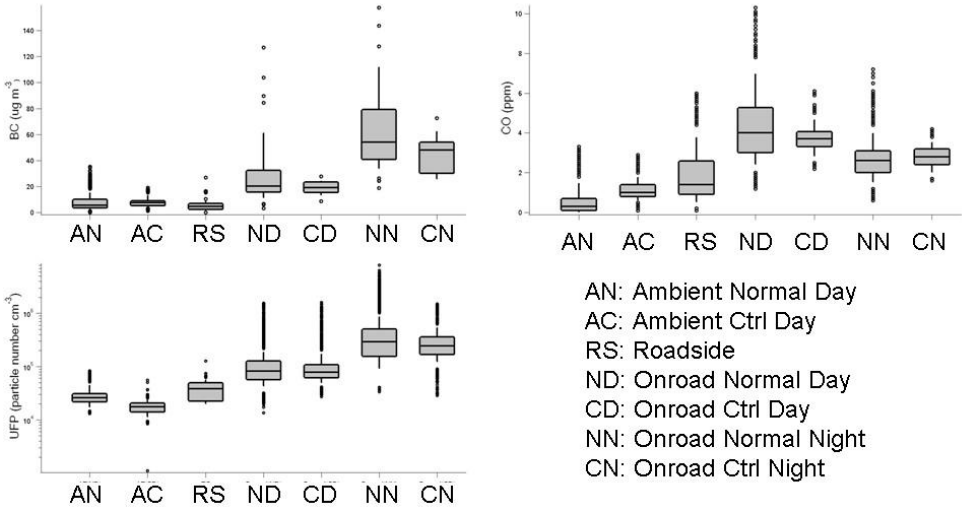


Figure 2.7 BC, CO and UFP concentrations in different microenvironments

We observed a decrease of median BC concentrations during the on-road measurements on the traffic control days at night (Figure 2.5a). More significantly, the extremes values of BC and CO concentrations from on-road (both daytime and nighttime) and ambient measurements were greatly reduced, probably by limiting the number of heavy emitters on the road. Thus, our results suggest that the traffic controls had a positive impact on 1) reducing potential human exposures to the pollutants from the heavy emitters in both the on-road environment and the ambient environment.

The removal of extreme values on traffic control days at the ambient community site may be caused by other reasons. Shown in Figure 2.6, the diurnal peak BC concentration value occurred around the early morning hours on each day. However, during the traffic control days, the morning peak data were collected only on Sunday morning due to logistical limitations imposed by campus security practices. We were unable to access to our ambient air monitoring site after we completed our on-road

monitoring. Thus we could not perform continuous air monitoring on those days. Owing to lack of peak data, it is difficult to provide ambient air quality data to fully document the impacts of controls on ambient air quality.

We did not identify clear trends in median concentrations of pollutants between normal days and traffic control days except for the on-road BC concentrations during nighttime. The explanations could be complex. The test period only lasted for four days including a Saturday and a Sunday. The differences in meteorology conditions may have affected the results.

2.3.4 Limitations

Even though we have made several important findings, there were a number of limitations in our study. For the study design, we investigated two highways (the 4th Ring Road and Badaling Expressway) and one community (PKUHSC campus), covering only a small fraction of Beijing. Our study was conducted during a brief period of the summer so that our results are not representative of winter conditions. We did not explore the effects of meteorology on micro-environmental air quality. Since we only employed one set of equipment, we were not able to measure the air quality at different environments simultaneously, which may yield more information on the impact of traffic. We are hoping that what has been found in this study will improve our follow-up studies in the future.

2.4 Conclusions

We conducted a field campaign in August 2007 in the north part of Beijing, China. The goals of this study are to characterize 1) the on-road emission factors of Beijing vehicles, 2) the impact of transportation on air quality at on-road, roadside and ambient environments and 3) to provide data in advance of the 2008 Olympics to allow comparisons of on-road and ambient air when extensive control measures are anticipated.

The fleet-average carbon monoxide emission factors are 95 and 50 g kg^{-1}-fuel for light-duty and heavy-duty vehicles, respectively. These values agree with those derived from remote-sensing and via on-board vehicle emission testing systems. The on-road BC and particle number emissions factors for Chinese vehicles are reported for the first time. The fleet-average BC emission factor for light-duty vehicles was 0.3 g kg^{-1}-fuel, which is eight to fifty times higher than those used to construct the emission inventory in China (Cao et al., 2006). This discrepancy implies that the contribution of light-duty vehicles to the total BC emissions in China may have been underestimated. The fleet-average BC emission factor for heavy-duty vehicles was 1.3 g kg^{-1}-fuel, falling within the range of values used in the emission inventory (Cao et al., 2006). The fleet-average ultrafine particle number emission factors were 1.8×10^{15} and 1.1×10^{16} particles kg^{-1}-fuel for light-duty and heavy-duty vehicles, respectively.

We also characterized the Beijing air quality under different microenvironments, i.e., on-road, roadside and ambient. The results demonstrate a strong traffic impact on the carbon monoxide, BC, and ultrafine particle concentrations. Further experiments on particle volatility are needed to explain the causes of the aerosol dynamics near

roadways. Based on the data collected during the four-day traffic control experiment, we identify its positive impact of reducing extreme concentrations that occur at on-road and ambient environments. But the effectiveness of the traffic control was not clear for reducing the median values of pollutant levels, probably owing to the short test period and variable meteorological conditions.

CHAPTER 3

EVALUATING THE AIR QUALITY IMPACTS OF BEIJING 2008 OLYMPIC GAMES: ON-ROAD EMISSION FACTORS AND AMBIENT BLACK CARBON

3.1 Introduction

The 29th Olympic and Paralympics Games were held between August 8th and September 17th, 2008. To achieve good air quality and improve traffic, the Beijing Municipal Government enacted a series of the traffic and other emission control regulations before and during these events. These regulations drove nearly two million vehicles off the roads in Beijing. The aggressive traffic interventions and emission control measures created a valuable case study to evaluate the effectiveness of these control measures on mitigating air pollution and protecting public health. In this chapter, we report the results and analysis from a field campaign conducted from July to September 2008 in Beijing to evaluate the transportation-related emissions and air quality impacts before, during and after the Olympics.

We first describe the emission controls and traffic intervention measures put into place before and during the Olympics. Next, we present the on-road emission factors of carbon monoxide (CO), ultrafine particles and black carbon (BC) derived during the Olympics, and compare them to the values we reported from our field campaign conducted in 2007 (Westerdahl et al., 2009). Then, we compare the ambient BC concentrations, a primary pollutant, measured in 2007 and 2008, and link the changes

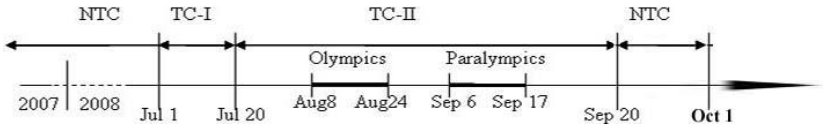
in the BC concentrations to the emissions and traffic intervention programs.

We devote more than half of the chapter to discussing the emissions and ambient concentrations of black carbon (BC) particles, which are released to the atmosphere from incomplete combustion of carbonaceous fuels. BC contributes to global and regional climate change, visibility degradation, lower crop yields, and human health effects. Liousse et al.(1996) reported the BC impacts on the global and regional climate by strongly absorbing solar radiation. Qiu and Yang(2000) have shown that BC contributes to the marked degradation of optical depths and visibility in northern China. Chameides et al.(1999) have demonstrated that crop yields in China are lowered because of reduced solar radiation reaching the earth. Menon et al.(2002) showed that high BC concentrations may contribute to flooding and drought in China as well as India.

BC in Beijing has been the focus of a number of previous investigations (Duan et al., 2005; He et al., 2001; Liu et al., 2008b; Wang et al., 2005a). We reported, for the first time, BC concentrations in different types of microenvironments in Beijing, namely, on-road, near-road and ambient from our field campaign in summer 2007 (Westerdahl et al., 2009). Elevated BC concentrations were observed on the road and near-road as compared to the ambient concentrations. We posited that during the summertime BC concentrations in Beijing are heavily impacted by truck traffic (Westerdahl et al., 2009). The aggressive truck traffic controls, as part of the overall traffic intervention program, implemented in 2008 provided a unique opportunity to re-examine the contributions of truck traffic to the ambient air quality.

3.2 Traffic control and other emission control measures

Table 3.1 Brief summary of traffic regulations in Beijing

Time Series			
TC-I [a]	LDGV	L	YLV are banned within the Beijing administrative area
		NL	Euro II emission standard
	HDDT	L	YLV are banned within the Beijing administrative area
		NL	Banned within the Beijing administrative area
TC-II	LDGV	L	YLV are banned; Odd-Even restriction;
		NL	Odd-Even restriction; Euro II emission standard
	HDDT	L	Banned between 6:00 to 24:00 everyday within the 6th Ring Road
		NL	Banned within the Beijing administrative area
NTC [c]	LDGV	L	No restriction
		NL	No restriction
	HDDT	L	Banned between 6:00 to 23:00 everyday within the 4th Ring Road; Vehicles over 8 tons in weight are banned between 6:00 to 22:00 everyday on the 5th Ring Road
		NL	Banned between 6:00 to 24:00 everyday within the 5th Ring Road

[a]: TC-I: Traffic control days period-I; TC-II: Traffic control days period-II; NTC: Non traffic control days. Additional traffic restrictions were implemented around Olympic venues during TC-II Olympic days and TC-II Paralympic days; HDDT: Heavy duty diesel truck; L: local vehicles; NL: Non local vehicles.
[b]: YLV: Yellow Environmental Protection Label Vehicles.
[c]: After October 1st, 2008, moderate restrictions (for example, the vehicles were banned on one business day every week upon the last digit of their license plates) were implemented.

A summary of the traffic regulations in different periods is shown in Table 3.1. The traffic control measures were effective from July 1st to September 20th, 2008 in the metropolitan area of Beijing, covering the periods of both Games. From July 1st to September 20th more than 300 thousand heavy-emitting vehicles, which account for about 10% of the total vehicles in Beijing, were barred from the roads within the city's administrative area. These vehicles, which are also identified as Yellow Environmental Protection Label Vehicles (YLV), are classified in two vehicle groups: 1) light duty vehicles, typically registered before 1999 and did not meet Euro I

emission standard and were not retrofitted with after-treatment devices; and 2) heavy duty trucks, typically registered before 2006 and did not meet Euro III emission standard. Secondly, local trucks were banned within the Sixth Ring Road (a roadway that circles the city at a distance of approximately 25 km from the city center) between 6:00 am and midnight, one hour longer than year 2007, and most non-local trucks were not allowed into the city. From July 20 to September 20, the Odd-Even rule was implemented in Beijing administrative area. Vehicles with license plates ending in odd numbers were banned from the roads on even-numbered calendar days, and those with plates ending in even numbers banned from the roads on odd-numbered days. Together, these traffic regulations resulted in reducing around 1.9 million vehicles, or close to 60% of the total fleet, from the roadways everyday in Beijing during the Olympic Games (http://www.beijing2008.cn/news/olympiccities/beijing/n214412139-.shtml, 2008-6-20: The temporary traffic control implementations for Beijing local vehicles during 2008 Beijing Olympic Games and Paralympic Games (in Chinese); http://www.beijing2008.cn/news/olympiccities/beijing/n214412173.shtml, 2008-6-20: The temporary traffic control implementations for non-Beijing local vehicles during 2008 Beijing Olympic Games and Paralympic Games (in Chinese); http://www.bjjtgl.-gov.cn/inquiries/roadcontrol.asp, Regional traffic regulations (in Chinese)). Figure 3.1 depicts the impact of the traffic control measures on the traffic conditions in Beijing.

In addition to traffic control measures, Beijing Municipal Government also implemented a series of air quality improvement measures in other sectors including enhancing the utilization of natural gas to replace coal for electricity generation and heating, reducing local power generation by importing electricity from the surrounding

areas, suspending construction as well as imposing strict dust control on construction sites, closing or relocating polluting industrial plants, constructing148.5 km new subway lines, nearly tripling the original length (54 km) (http://www.bjwmb.gov.cn-/zt/plan_aoyun/01.htm, 2002-7-13: Beijing Olympic Games Action Plan (in Chinese)).

Figure 3.1 Comparison of traffic conditions in 2007 and 2008. (a): August 15, 2007. Trucks were waiting near entrance of the 5th Ring Road to enter the city; (b): August 12, 2008. Fewer trucks near the same entrance due to ban during the Olympics; (c): typical afternoon traffic conditions in summer 2007; (d): Traffic volume was reduced during Olympics in 2008 (the left was reserved for Olympics-related vehicles).

3.3 Methodology

3.3.1 Ambient Monitoring

The ambient monitoring site operated in 2008 is located on the PKU Health Science Center campus, which is ~ 3.2 km from the main Olympics venues (39°58'52"N, 116°20'56"E). The sampling sites are about 100 meters (m) away from

the centerline of the nearest roadway. Two Aethalometers were installed at two different elevations. The first one (Magee Scientific, Model AE21), whose inlet was located approximately 6m above the ground and connected to a $PM_{2.5}$ GK2.05 (KTL) 4 LPM Cyclone followed by a 0.6m tubing, was being operated from July 25th to October 2nd, covering the periods of TC-II Olympic days, TC-II non Olympic days and NTC days. The second Aethalometer (Magee Scientific, Model AE42), stationed at 20m above the ground, was recording data from July 26th to September 5^{th}. The inlet of the second Aethalometer was connected to a BGI Model SCC-1.197 $PM_{2.5}$ Cyclone at 2 LPM sample flow rate followed by a 1 m conductive tubing. Models AE21 and AE 42 employed the same sensors, tapes and algorithm for data conversion, and thus are highly consistent with each other (http://www.mageesci.com/products/-aethalometer.htm)**.** The time intervals for data logging for both instruments were set to be 5 minutes for the ambient sampling.

The meteorological data for data analysis was acquired from the Weather Underground Website (http://www.wunderground.com). The meteorology station is located in the Beijing Capital International Airport (ZBAA), which is 23 km northeast from our ambient monitoring site. Temperature, relative humidity, wind speed, wind direction, precipitation and weather conditions at 30 minute time intervals were obtained from this meteorology station.

3.3.2 On-road Monitoring

The mobile platform was a gasoline-powered minivan, and the instruments were powered by a 12 volt storage battery/inverter (250 watts at 120 volts) power supply.

The on-road sample air was taken through a side window. The van was operated at highway speeds and self-pollution appeared unlikely to have had a major impact. Instruments used for on-road measurements include FMPS (TSI Model 3091), Aethalometer (Magee, Model AE42), and IAQ-CALC (TSI Model 7545), each of which was connected with 1 meter long conductive tubing. The time intervals for the three instruments were 1 second, 30 seconds and 10 seconds, respectively. A portable GPS (Garmin Model 60 CSx) was used to record the locations, routes and vehicle speed with a 10 s frequency. A digital camcorder was held by the staff in the front seat of the mobile platform and recorded the traffic conditions continuously during the on-road measurements.

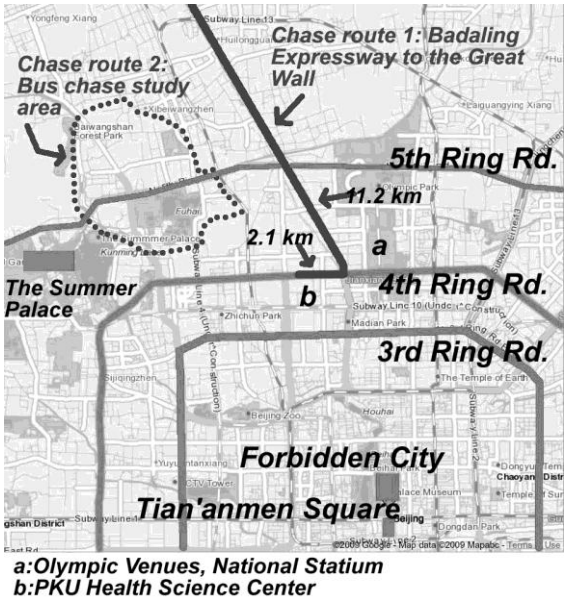


Figure 3.2 On-road chase study route in Beijing (Map source: Google Map)

The on-road measurements were conducted in two areas in the northern Beijing as illustrated in Figure 3.2. Route 1 consists of 22.4 km on the Badaling Expressway (BE) and 4.2 km on the 4th Ring Road. Most of our truck and passenger car chasing

studies were performed on this route since it is a major truck corridor in Beijing and we conducted on-road measurements on the same route during 2007. Route 2, bounded by dash line in Figure 3.2, is located in the northwest part of Beijing, close to the 5th Ring Road. Several famous tourist destinations are located in this area, resulting in heavy bus activity. The non-bus traffic volume in this area was not as high as in business districts, making it easier to follow the target buses for an extended period.

The on-road measurements were performed on two days during the TC-II Olympic days (August 12th and 13th 2008). On the first day, the mobile platform ran four roundtrips on Route 1, two were during the daytime (3:20pm~5:10pm) and two were during the nighttime (11:10pm~ 12:45am). On the second day, the mobile platform ran two roundtrips along Route 1 and one roundtrip along Route 2, only in the daytime (1:30pm~4:40pm). After each trip, the van parked near the ambient site for ten minutes measuring the ambient concentrations.

During our on-road sampling, we randomly chose the trucks and buses as the target vehicles to monitor for emission calculations, and followed each of them for 1 to 7 minutes, depending on the traffic conditions. The distance between our minivan and target vehicle is roughly in the range between 5m to 50m, varying with the speed of target vehicles. The identification of different types of trucks and buses was based on the visual information of the vehicles, for example, many vehicles (most of them were buses) had badges stating emission standard (Euro rating), the model, vehicle structure, etc. All on-road traffic information was also recorded by a camcorder in our mobile van, synchronized with each instrument before the trip. The number of

gasoline trucks, diesel trucks and diesel buses being chased in our study are 6, 19 and 30, respectively. Note that the traffic control measures imposed during the TC-II Olympic days greatly reduced the number of on-road trucks so our sample sizes for gasoline and diesel trucks were limited. The travel speeds of most vehicles on Badaling Expressway were normally between 55 to 75 km hr^{-1} similar to that recorded in 2007. The decrease in the traffic volume did not lead to significant increase in average vehicle speed on the Badaling Expressway, partially because its leftmost lane was dedicated to Olympic vehicles only. However, the traffic congestion near the expressway entrance that we encountered in 2007 disappeared due to the reduced traffic volume. Several high speed vehicles with heavy emissions were also observed during our 2008 on road monitoring periods. Different from the high speed traffic on the Badaling Expressway, buses traveled at a lower speed (normally around 40 km hr^{-1}) on the surface streets along Route 2.

3.3.3 Emission Factors (EF)

The fleet average emission factors (EF) were calculated based on the data collected during the on-road measurements. By performing a carbon balance on the fuel combustion process, one can relate the emission of carbon-containing species from vehicle exhaust to fuel consumption (Geller et al., 2005; Kirchstetter et al., 1999; Stedman, 1989). We consider CO_2, CO and BC as the carbonaceous products during the combustion process. We did not have direct measurement of hydrocarbon (HC) gases. Neglecting HC should have very minor effects since gas-phase carbonaceous products are usually dominated by CO_2 and CO. The fuel-based emission factor can be

calculated in the following equation:

$$EF_p = \frac{\Delta[P]}{\Delta[CO_2]\times\frac{MW_C}{MW_{CO_2}} + \Delta[CO]\times\frac{MW_C}{MW_{CO}} + \Delta[BC]\times\frac{MW_C}{MW_{BC}}} \times w_c, \qquad (1)$$

where $\Delta[i] = [i] - [i]_0$; i = P, CO_2, CO and BC; subscription 0 denotes the baseline value; EF_p is the emission factor of pollutant P in grams of pollutant emitted per kilogram of fuel consumed (g kg^{-1}); Δ[P] is the on-road concentration of pollutant P above the baseline, with the unit of grams per cubic meter of air (g cm^{-3}); Δ[CO_2], Δ[CO] and Δ[BC] represent the increase of CO_2, CO and BC, with the unit of grams per cubic meter of air (g cm^{-3}); MW_i is the molecular weight of the i^{th} species; w_c is the mass fraction of carbon in the fuel.

Following the procedure developed in (Westerdahl et al., 2009), we took the ambient concentrations recorded at a quiet central campus location at the end of each trip as the baseline concentrations. Other sources of urban carbonaceous compounds, such as meat cooking and power plants, were not observed in the close proximity of the Badaling Expressway. Therefore, the increment of the on-road concentrations above the baseline was considered as the traffic contribution. We grouped the on-road concentrations in thirty-second intervals and then calculated the emission factors for each interval during the whole trip. Since the daytime traffic on Badaling Expressway in Beijing was consistently dominated by light-duty gasoline vehicles (LDGVs), the fleet average emission factors of LDGVs were taken to be the median values of the emissions factors on the Badaling Expressway during daytime without chasing any individual vehicle, both in our 2007 and 2008 studies. In 2007, the emission factor of diesel trucks were taken to be the median values of the emission factors during

nighttime as HDDVs dominated the fleet after 11 pm. In contrast, fewer trucks were present in the on-road fleet in 2008. Therefore, we derived the emission factors for trucks based on the median values of emission factors during the periods when we were chasing individual target vehicles. A similar method was applied to the emission factors of the buses as buses did not dominate the fleet.

3.4 Results and Discussion

3.4.1 Emission factors

The derived emission factors for different types of individual vehicles, as well as the fleet average emission factors, were reported in Table 3.2. The box and whisker plots of emission factors for light-duty gasoline vehicles (LDGV) were also shown in Figure 3.3.

Table 3.2 On-road emission factors for different type of vehicles

Vehicle Types	Number Vehicles 2008	ofBC (g kg-fuel^{-1})	CO (g kg-fuel^{-1})	UFP(10^{15} particles kg-fuel^{-1})
LDV (fleet average)	N/A[a]	0.2(0.1~0.4)[b]	50(44~63)	0.4(0.2~0.7)
Gasoline trucks[c]	6	0.9(0.6~1.1)	53(43~69)	1.4(1.0~1.9)
Diesel trucks	19	1.4(0.6~4.3)	65(41~146)	3.6(0.7~15)
Diesel buses	30	0.5(0.3~0.9)	47(35~70)	0.9(0.3~1.1)
Airport diesel shuttle bus	1	0.3	43	0.2
Euro III Natural gas bus	1	0.2	68	0.1
High speed diesel van	1	6.8	75	3.7
New diesel trucks	1	0.3	39	0.4
2007				
LDV(fleet average)	N/A	0.3(0.2~0.5)	95(79~117)	1.8(1.1~2.8)
HDV	N/A	1.4(0.9~2.2)	50(41~67)	11(6.6~17)
Heavy emitters	N/A	3.1	136	21

[a]: No individual vehicles in 2007, no individual LDGVs in 2008.
[b]: The lower limit of the range represents the first quartile of the data set, and the upper limit of the range represents the third quartile of the data set.
[c]: If the number of vehicles chased was greater than one, the EFs represent the median emission factors of all vehicles; if the number of vehicles chased is one, EFs represent that vehicle. The chase approaches are described in section one.

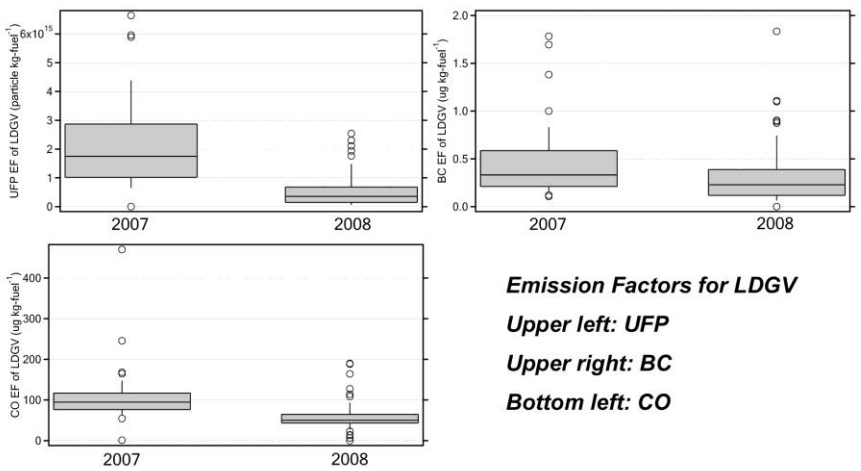


Figure 3.3 Emission factors of LDGV measured on the Badaling Expressway in 2007 and 2008.
(Line within box: Median value; Top line of box: third quartile; Bottom of box: first quartile; Upper whisker: ninety percentile; Lower whisker: ten percentile; Outlier: 1.25×whisker length.)

In order to test the statistical significance level of the difference between the sample sets in 2007 and 2008, the statistical hypothesis tests were made. "Two sample t-test" were used and the *p*-values for CO, BC and UFP are 6.12×10^{-6}, 0.03 and 5.9×10^{-9}, respectively. The small numbers (<0.05) of *p*-values indicate that all the differences between 2007 and 2008 EFs are statistically significant (Chiang, 2003).

For light-duty gasoline vehicles (LDGV), the BC, CO and UFP emission factors decreased 33%, 47% and 78%, respectively, during the Olympic year compared to the year before (Table 3.2). These sharp drops of emission factors were probably a result of several measures: firstly, the aggressive emission standard was implemented in 2008. For example, starting from March 2008, all new registered vehicles were required achieve the Euro IV standard. Starting from June 2008, over 20,000 buses and 66,000 taxies in Beijing were required achieve Euro III standard. Note that buses and taxis were exempt from Odd-Even traffic control regulation. Secondly, the Euro

IV standard for gasoline and diesel fuels was implemented in Beijing in early 2008 (http://www.bjepb.gov.cn/bjhb/tabid/426/Default.aspx, Beijing Municipal Environmental Protection Bureau. (in Chinese)). The improved fuel quality, corresponded with reduction of sulfur content in the fuel, enhanced the performance of both engines and exhaust catalyst converters (Beck and Sommers, 1995; Corro, 2002). For CO and UFP emissions factors of the LDGVs, we observed the reductions in not only their median values, but also the extreme values (or so called "heavy emitters") as illustrated in the box and whisker plots in Figure 3.3, which compare the emission factors we derived from two 2-hour on-road measurements on the same highway (Badaling Expressway) in 2007 and 2008, respectively. The vehicle inspection programs enforced before and during the Olympics, and the ban of "YLV" may be responsible for removing many heavy emitters from the roads. Furthermore, the restriction of non-local trucks and non-local passenger cars may eliminate the vehicles that did not meet the fuel and/or emission standards implemented in Beijing.

For heavy-duty diesel vehicles (HDDV), the UFP emission factor decreased by 67%. However, the fleet average BC and CO emission factors for HDDV in 2008 were comparable to the values we reported in 2007. Since the average vehicle speed did not change significantly, the limited number of target vehicles described in Section 3.3.3 may partially explain why these emission factors of HDDVs in 2008 did not decrease as expected (Fruin et al., 2008; Ntziachristos et al., 2007). In addition, the emission factor of each sub-type varied greatly. The average BC emission factor of diesel trucks that appeared to be new is only 25% of average diesel truck emission factors.

Westerdahl et al.(2009) compared the CO, BC and particle number emission factors in Beijing based on our pre-Olympics study (2007) with those reported in other countries. The CO and particle number emission factors of LDGV measured in 2008 further decreased to the level comparable to those obtained in US and European cities. But the BC emission factor of LDGV was still considerably higher than those measured in US. For HDDV, the particle number emission factor in 2008 was similar to those measured in US and Europe (Beddows and Harrison, 2008; Bishop and Stedman, 2008; Geller et al., 2005; Kirchstetter et al., 1999; Kittelson et al., 2004; Miguel et al., 1998; Pokharel et al., 2002; Singer and Harley, 2000).

The on-road BC and UFP emissions factors for Chinese buses are reported for the first time here. For all three emission factors we measured, i.e., BC, CO and UFP, the emission factors for buses were substantially lower than those for the trucks (by 64%, 28%, and 75%, respectively). According to the report of Beijing Public Transport Holdings, Ltd. (BPT), 97.8% buses in Beijing achieve Euro III or Euro IV emission standards. Therefore, the 30 buses we chased in our study were mainly a mixture of Euro III and Euro IV. However, we could only clearly identify one Euro III bus, one Euro IV bus and one natural gas bus from their labels. The BC emission factor of the Euro III bus is almost three times higher than that of the Euro IV diesel buses and four times higher than that of the natural gas bus.

3.4.2 Community BC observations at 6m and 20m elevation

We analyzed the BC concentrations measured simultaneously at two different elevations to determine how the levels vary with height. In Recent studies, Guinot et

al.(2006) and Chan et al.(2005) showed vertical structure of the Atmospheric Boundary Layer (ABL) in Beijing and its effect on vertical distributions of major pollutants, including particulate matter and BC. In their works, the measurements were done on the meteorological tower with the height of 325m, representing multiple sub layers in ABL. Chan described the vertical profiles of Elemental Carbon (EC), showing a complex variation pattern below the 325m height. However, our studies only compared the BC concentrations at ground level (6m) and at 20m. These elevations, representing the height from the second floor to a seven floor building, encompassing spaces where people live and work.

Figure 3.4 depicts the comparison of BC concentrations from the two Aethalometers. Figure 3.4 (a) shows that the median value of 20m level is 3.0 (1.9-4.3) $\mu g\ m^{-3}$ while the value of 6m level is 4.4 (2.7-6.1) $\mu g\ m^{-3}$, suggesting that BC concentrations at 20m level were 32% lower than those at 6m. The values in the parenthesis represent the first and third quartiles of the data points, corresponding with the top and bottom lines of box in Figure 3.4 (b). This decreasing trend with height was consistent during the entire measurement period, evidenced by the high correlation coefficient between the concentrations at the two heights, shown in Figure 3.4 (c). We occasionally observed higher BC concentrations at 20 m than those at 6 m, which was likely due to the fluctuation of local wind direction and speed, condition of instrument, and nearby traffic conditions.

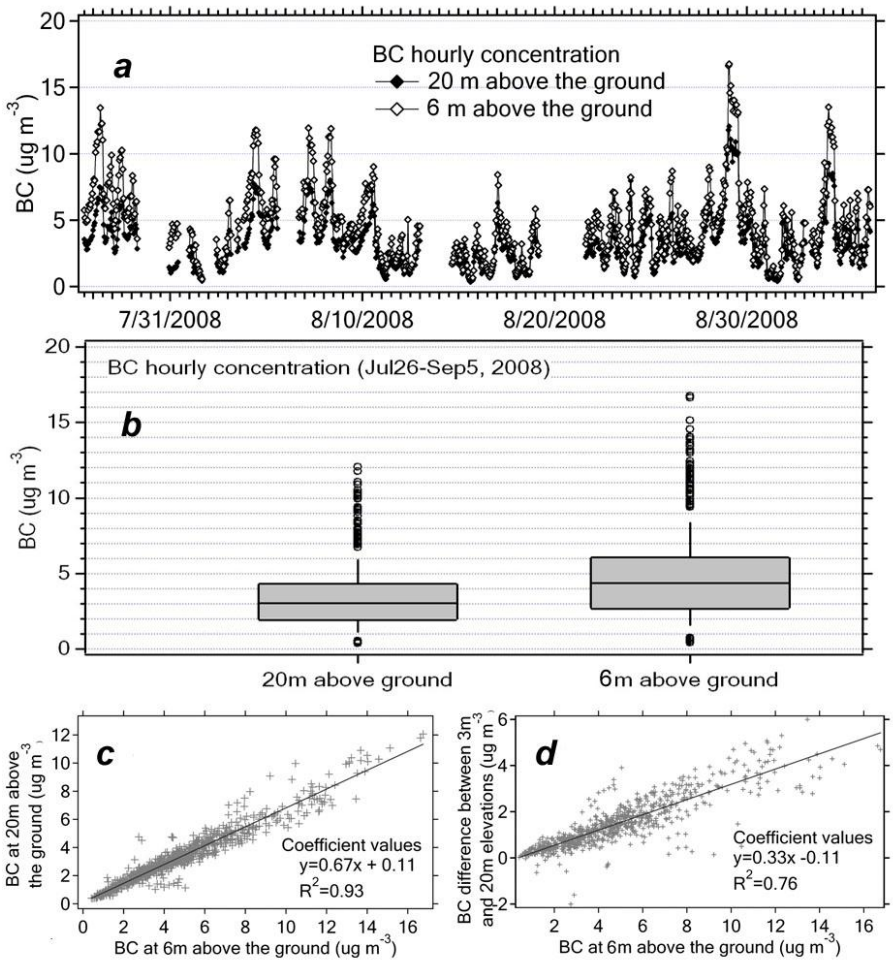


Figure 3.4 BC concentrations at 20 m and 6 m levels.
(a: BC concentration with respect to time; b: box and whisker plot; c: correlation of BC between 20m level and 6m level; d: correlation between BC difference and BC at 6m level.)

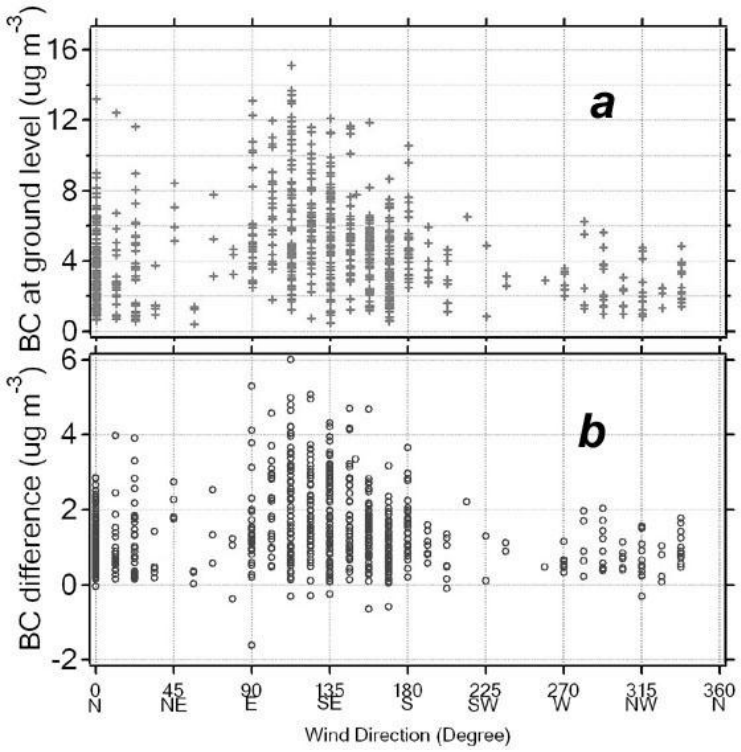


Figure 3.5 Correlation between BC concentration and wind direction. (a: ground level BC concentration vs. wind direction; b: BC difference at two elevations vs. wind direction.)

We also plotted the differences in BC concentrations between two. A moderate positive correlation was observed shown in Figure 3.4 (d), which suggests that the greater ground level BC concentrations are associated with larger BC vertical gradients and indicates that ground level sources may contribute to the major part of BC in the lower atmospheric layer in Beijing. In addition, the wind impacts on the BC concentration and its vertical gradient are revealed, respectively, in Figure 3.5 (a) and (b), in which data were presented from July 26th, 2008 through September 5th, 2008 (except few days in between when coincident BC data were missing). It is clear that the wind directions in this period are mostly located in two ranges: one is between east and south while the other is between west and northeast. Although the lower BC

concentration (less than 7 μg m^{-3}) and low vertical gradient (the BC difference is less than 2 μg m^{-3}) were observed to be randomly distributed in these two wind direction ranges, the high BC concentration and high vertical gradient were mostly observed during periods of southeast winds. This phenomenon suggests possible BC sources in the southeast direction of our monitoring site, but more work needs to be done to specify these sources. The preliminary results of $PM_{2.5}$ monitoring performed at this site suggest that the important contributors to urban regional PM pollution also originate in the southeast during this period. A major regional source of pollutants located in the south east quadrant is Tanjin and we postulate that our observations are consistent with transport of carbon and $PM_{2.5}$ from that region. This is also likely in 2008 because of the major controls placed on all local sources of these pollutants.

In summary, our measurement of BC at different elevations suggests a consistent decrease in BC concentrations as the height increases from the ground level. Therefore, the nearby ground level sources, possibly dominated by traffic, contributed to a large portion of BC concentrations in the lower atmospheric boundary layer in Beijing during summertime. This finding is consistent to the results in Chan's (2005) work that the layer between 8 and 47 m in Beijing was stabilized by strong temperature inversions, and to the observation made in Milan, Italy (Ferrero, 2009), in which vertical profiles of BC were measured using a tethered balloon. An important implication is that people living at higher elevation levels may have lower exposure to BC, and possibly traffic pollution, a factor to be considered for human exposure assessment.

3.4.3 Comparison of BC concentrations on traffic restriction and non restriction days

As described in section 3.2, a series of traffic restrictions were implemented during TC-I and TC-II period. Non-local and local heavy-duty truck activities were limited during the traffic restriction period. This could be expected to reduce ambient BC concentrations.

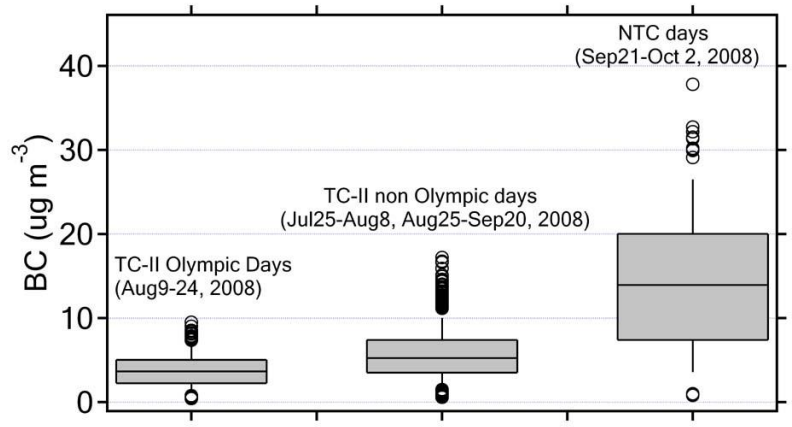


Figure 3.6 BC concentrations on TC-II Olympic days, TC-II non Olympic days and NTC days

Figure 3.6 compares the BC concentrations measured at the ambient site on three monitoring periods in summer 2008, i.e., TC-II non Olympic Days (July 25 to August 8 and August 25 to September 20, 2008), TC-II Olympic Days (August 9-24, 2008), Post Olympic NTC days (September 20 to October 2, 2008). The median BC concentrations were 5.25 (3.52~7.40) μg m^{-3}, 3.65 (2.25~5.04) μg m^{-3} and 13.90 (7.39~19.90) μg m^{-3} for the three periods, respectively. The ranges in parentheses represent the first and third quartiles, corresponded to the top and bottom line of the box in Figure 3.6. The significant increases of both BC median and maximum concentrations on NTC days demonstrate the positive impact of traffic control

regulations from July 20th to September 20th in reducing BC emissions. Moreover, BC concentrations during the TC-II Olympic days further decreased from those during TC-II non Olympic days, which is likely due to the additional traffic restrictions around Olympic venues enforced during the TC-II Olympic days (http://www.-bjepb.gov.cn/bjhb/tabid/426/Default.aspx, Beijing Municipal Environmental Protecti-on Bureau. (in Chinese)), because our sampling site is quite closed (0.5 km) to the weight lifting venues, and no significant wind condition or other meteorological differences were found between TC-II Olympic days and TC-II non Olympic days (Figure 3.7).

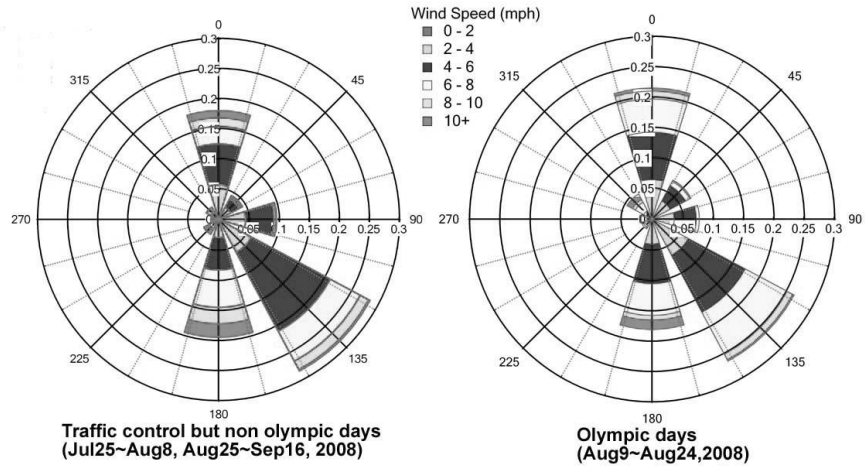


Figure 3.7 Wind conditions on TC-II non Olympic days (left) and TC-II Olympic days (right)

3.4.4 Comparison of BC concentrations in 2008 and previous years

BC concentrations in Beijing were also reported by other research groups in the past decade, shown in Table 3.3. It is clear that BC (and sometimes measured as Elemental Carbon or EC) concentrations in winter are higher than those in other seasons, due to the higher coal consumptions in the heating season coupled with

meteorological conditions (Dan et al., 2004; He et al., 2001; Wang et al., 2005a). The summertime BC concentrations obtained in 2007 and 2008 TC-II days were comparable with He's work.

Table 3.3 BC (EC) ambient concentrations in Beijing from this and previous studies

Method	Season	BC or EC μg m^{-3}	Reference
Aethalometer (BC)	Summer, 2008 (TC-II Olympic Days)	2.3(1.4~3.2) [a]	This study (20m)
		3.7 (2.3~5.0)	This study (6m)
	Summer, 2008 (TC-II Non Olympic Days)	3.5(2.4~4.9)	This study (20m)
		5.3(3.5~7.4)	This study (6m)
	Sep. 2008 (NTC Days)	13.9(7.6~19.9)	This study (6m)
Aethalometer (BC)	Aug. 2007	6.2(3.7~12.1)	Our study in 2007
Filter Sampler (BC)	Winter, 1996-99	24.9	(Wang et al., 2005a)
	Winter, 2000-04	15.9	
Filter Sampler (EC)	Summer, 1999-2000	6.27	(He et al., 2001)
	Winter, 1999-2000	11.1	
Filter Sampler (EC)	Feb. 2001	12	(Yang et al., 2005)
	Jul-Aug. 2001	6.9	
Filter Sampler (EC)	Summer, 2002	5.4 ±2.6	(He et al., 2004)
Filter Sampler (EC)	Jun – Jul. 2002	5.7 ±2.9	(Dan et al., 2004)
	Dec. 2002	15.2 ±11.1	
ACPM (EC)	Sep.-Nov., 2002	6.6 ±5.0	(Duan et al., 2005)
Filter Sampler (BC)	August, 2003, 2004	2.5	(Guinot et al., 2007)
	Jan. 2003-Aug. 2004	5.02	
Aethalometer (BC)	Feb. 23, 2006	8.4	(Liu et al., 2008b)

[a]: Data reported in this study are median values of hourly average concentration. The range represents the first quartile and third quartile values of the data set.

Due to the extensive traffic control measures in 2008 summer and even more strict regulation during TC-II Olympic days, the median BC concentrations on TC-II non Olympic days (5.25 μg m^{-3}) and TC-II Olympic days (3.66 μg m^{-3}) were reduced by 28.2% and 50.1%, respectively, compared to 2007 median BC concentrations (7.31 μg m^{-3}), shown in Table 3.3.

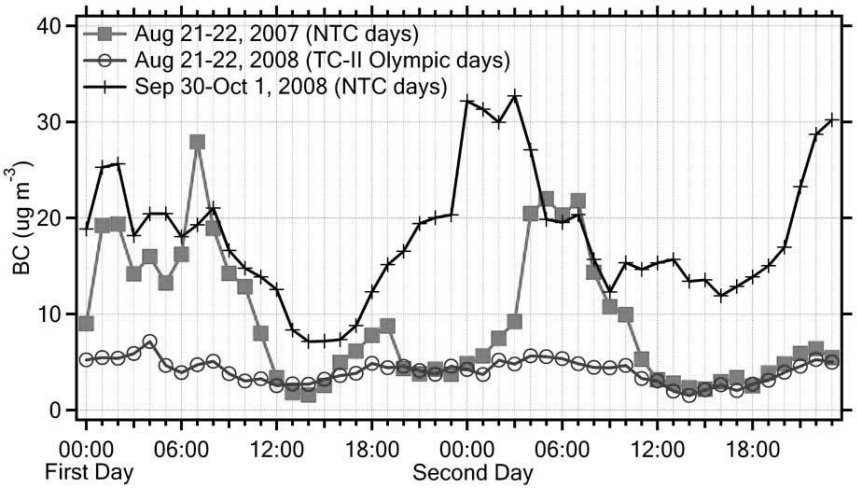


Figure 3.8 Diurnal patterns of BC in 2007 and 2008

BC diurnal profiles on six days in three representative periods, i.e., two days in summer 2007 without traffic controls, two days during the TC-II Olympic days with stringent traffic controls, and two days after the Olympics without traffic controls, are shown in Figure 3.8. The nighttime BC concentrations on the TC-II Olympic days (August 21-22, 2008) were appreciably lower than those on NTC days in both 2007 (August 21-22, 2007) and 2008 (September 30-October 1, 2008), suggesting the effectiveness of traffic control measures in reducing BC emissions. Once again, the sharp rise of BC concentrations after midnight, a phenomenon we observed on NTC days in both 2007 and 2008 corresponding to the traffic restriction (non-local trucks were only allowed to enter the city between 12 am and 6 am in 2007 and 2008), did not occur during the TC-II Olympic days due to even more stringent traffic restriction (i.e., non-local trucks were banned from entering the city) imposed by the Beijing Municipal government during the Olympics. Therefore, our findings in 2008 provide further evidence that diesel trucks are a major contributor to the ambient summertime BC levels.

The BC diurnal patterns for the NTC days of 2008 exhibit similar patterns, elevated at night and low in the afternoon. Similar to 2007, the peak concentrations that occurred after midnight were likely caused by the heavy trucks, which were allowed to enter the city after midnight. However, the BC concentrations on NTC days in 2008 were observed to be greater than those in 2007, throughout the day. The meteorological conditions may partially account for those greater values in 2008. The wind direction patterns on the four NTC days were similar, dominated by north wind in the morning and south wind in the afternoon. However, lower temperature, higher humidity, lower wind speed, and small visibility distance on September 30~October 1, 2008 all represent factors that would favor BC level buildup (Table 3.4).

Table 3.4 Meteorology conditions on four NTC days in 2007 and 2008

	2007-8-21	2007-8-22	2008-9-30	2008-10-1
Temperature (C)	31	29	18	18
Relative Humidity (%)	61	55	71	82
Wind Speed (km h^{-1})	8	8	5	3
Visibility (km)	8	17	4	3

3.5 Conclusion

Stringent emission control measures implemented before and during the 2008 Olympics, resulted in dramatic emission reductions from fossil fuel combustion, especially from the transportation sector. This chapter presents the results and analysis from our field campaign in summer 2008.

For LDGV, the fleet average emission factors of BC, CO and UFP decreased by 33%, 47% and 78%, respectively, in the Olympic year compared to the same period the year before. These reductions of emission factors were probably a result of several measures implemented during the Olympic year. The on-road CO, BC and UFP

emissions factors for Chinese buses were measured to be 47 g kg-fuel^{-1}, 0.5 g kg-fuel^{-1}, and 0.9x10^{15} particles kg-fuel^{-1}, respectively. The UFP emission factor of HDDV also decreased 67%. Whereas, there were no significant changes in the CO and BC emission factors between 2007 and 2008.

The median ambient BC concentration on the traffic-control days during the Olympic was 3.65 μg m^{-3}, only 26% of the value on non-traffic-control days in 2008. The sharp rise of BC concentrations after midnight, a phenomenon we observed on non-traffic-control days in both 2007 and 2008 corresponding to the traffic restriction (non-local trucks were only allowed to enter the city between 12 am and 6 am), did not occur during the traffic-control days during the Olympic due to stringent traffic restriction (i.e., non-local trucks were banned from entering the city) imposed by the Beijing Municipal Government during the Olympics. Therefore, our findings in 2008 further confirmed that diesel trucks are a major contributor to the ambient summertime BC levels.

Our measurement of BC at different elevations suggests a consistent decrease in BC concentrations as the height increases from 6 m to 20 m. Therefore, the nearby ground level sources, possibly dominated by traffic, contributed to a large portion of BC concentrations in the lower atmospheric layer in Beijing during summer time.

In summary, we have observed significant reductions in not only vehicle emissions factors that a likely to be a result of changes in the fleet composition, but also ambient concentrations of traffic-related air pollutants. Since some of the traffic intervention programs were temporary in nature (e.g., banning non-local trucks from entering the city and reducing all traffic by at least half). While we have reported

that the controls placed on sources were effective in improving measured pollutants it is important to perform further monitoring studies to to examine possible changes in emissions and traffic control measures after the Olympics and whether these result in long-term air quality impacts after the controls relaxed.

CHAPTER 4

ON-ROAD EMISSION FACTOR DISTRIBUTIONS OF INDIVIDUAL DIESEL VEHICLES IN AND AROUND BEIJING, CHINA

4.1 Introduction

Emissions from diesel-powered vehicles are a major source of urban, regional and global pollution. Diesel particulate matter is identified as one of the mobile source air toxics (Diaz-Robles et al., 2008; Maricq, 2007; Sawyer et al., 2000). Diesel exhaust also contains various gaseous pollutants such as nitrogen oxides (NOx), carbon monoxide (CO), and volatile and semivolatile organic compounds that have direct health implications and act as precursors to photochemical ozone formation (Marr and Harley, 2002; Sawyer et al., 2000). Black carbon (BC), a major component of diesel particulate matter, is reported as the second strongest contributor, after carbon dioxide (CO_2), to current global warming (Ramanathan and Carmichael, 2008), and also affects the large-scale circulation and hydrologic cycle with significant regional climate effects (Menon et al., 2002).

Emission factors (EFs), expressed as the weight of pollutant divided by a unit weight, volume, distance, or duration of the activity emitting the pollutant, are useful in calculating emission inventories. Different approaches have been applied to evaluating diesel EFs, including chassis dynamometer test (Riddle et al., 2007; Schauer et al., 1999; Zielinska et al., 2004), remote sensing tests (Burgard et al., 2006; Chan and Ning, 2005), roadway tunnel tests (Ban-Weiss et al., 2009a, b; Geller et al.,

2005), on-road chasing studies (Kittelson et al., 2004; Ronkko et al., 2006; Schneider et al., 2008; Westerdahl et al., 2005; Yli-Tuomi et al., 2005), and on-road testing with the portable emission measurement system (PEMS) (Johnson et al., 2009; Liu et al., 2009).

Recently, Westerdahl et al. (2009) and Wang et al. (2009) reported the fleet-average EFs of CO, BC and $PM_{0.5}$ number for Chinese light-duty gasoline and heavy duty diesel vehicles by conducting on-road chasing studies in and near Beijing. In our most recent field campaign, we improved our chasing methods, which allowed many more observations of individual diesel trucks and buses. This chapter describes the methodology and results of these improved methods. The uncertainty analysis of EF calculations and limitations of our method are discussed.

4.2 Methodology

In our 2009 field campaign, a mobile platform equipped with fast response instruments was used to conduct the on-road chasing studies, measuring the on-road concentration of CO_2, CO, BC and $PM_{0.5}$. Based on the carbon balance analysis (Kirchstetter et al., 1999; Stedman et al., 1991), EFs are calculated for trucks of different regions of origin and for different types of buses operated in Beijing.

4.2.1 On-road measurements

The mobile platform was a gasoline-powered minivan. Instruments were powered by a 12 volt deep-cycle storage battery/inverter (250 watts at 120 volts). Air sampling

was conducted through a side window. Particle-phase air pollutants were sampled using flexible conductive tubing that was situated with the inlets approximately perpendicular to the airflow outside the van. Self-pollution was not apparent upon inspection of the individual time series observations. The placement of the inlet on the side of the vehicle, which had it engine in the rear of the van, played an important role in reducing the likelihood of self pollution. The criteria for observation acceptance required stable baseline values. Self pollution would result in unstable baselines. Isokinetic sampling was not attempted because particles studied were in the submicron size range, and the Stokes number was in the range of 10^{-7}~10^{-3}, which was far less than unity. This indicated that the effects of particle inertia and isokinetic sampling were not significant (Baron and Willeke, 2001). The instruments used for on-road measurements were as follows: a Fast Mobility Particle Sizer (FMPS, Model 3091, TSI Inc.) for particle number concentration and size distribution, measuring particles ranging from 5.6 to 560 nm (mobility diameter and referred to in this chapter as $PM_{0.5}$). A micro Aethalometer (Model AE51, Magee Scientific.) was employed to measure BC mass concentration. A Q-Trak (Model 7565, TSI Inc.) measured CO using an electrochemical cell, and a Vaisala CARBOCAP (Model GM70, Vaisala) measured CO_2 using Non-Dispersive Infrared (NDIR) methods. The recording time intervals of all instruments were set to be 1 second. These observations are then averaged to 10 seconds. The actual response times of the FMPS and micro Aethalometer are stated by the manufacturers to be 1 second. Temporal response of CO_2 data from the Vaisala Carbocap, as operated, is approximately 3 seconds, while the response of the Q-Trak which provided CO data is more difficult to determine. Specifications for this

instrument state that it responds in less than 60 seconds. However, inspection of time-series data for this instrument appears to show a much faster response when aligned with other pollutants.

The adoption of instruments with fast response compared to those in our previous on-road work (Wang et al., 2009; Westerdahl et al., 2009) enabled us to identify the corresponding pollutant data of the target vehicles more precisely. It was flexible allowing the selection of many different types of target vehicles in desired locations, and selection of proper time when only one target vehicle was in front of the mobile platform with minimum impact of other on-road vehicles. The recorded video and written notes served to describe the on-road conditions and identified the target vehicle for linkage with the monitoring data. A portable GPS (Garmin Model 60 CSx) was used to record the locations, routes and vehicle speed with 10 second time resolution. A digital camcorder was operated in the front seat of the van and recorded the traffic conditions continuously during the on-road measurements. Times of all instruments were all synchronized to a portable computer, which also functioned as a data logger for the CARBOCAP, Micro Aethalometer and the FMPS. The plate numbers of chased vehicles and the speed of our mobile platform were written into a notebook during the on-road measurements.

Figure 4.1 Map of sampling areas.
(A: bus chasing area; dash line: 4th Ring Rd.; The expressway labels are interpreted in the text)

Table 4.1 Sampling routes, dates, and numbers of sampled vehicles

Sampling Area	Sampling Date	Sampling Time	Vehicles Sampled
Truck Chasing Study			
G6 BT	2009.11.22	3:30pm-5:35pm	18
	2009.11.24	11:35pm-1:30am(2^{nd} day)	32
	2009.11.26	10:30pm-12:25am(2^{nd} day)	30
	2009.12.1	10:05pm-12:20am(2^{nd} day)	43
G2 BS	2009.11.24	1:40pm-5:20pm	40
G1 BH	2009.11.26	2:55pm-4:35pm	30
G45 BK	2009.11.25	1:25pm-4:35pm	28
	2009.11.25	11:05pm-1:10am(2^{nd} day)	25
Bus Chasing Study			
Northwest Suburban	2009.11.27	12:10pm-3:09pm	26
	2009.12.1	1:22pm-3:40pm	33

The regions where on-road measurements were made are illustrated in Figure 4.1. The date, time and the number of sampled vehicles for each on-road measurement are listed in Table 4.1. Our truck chasing studies were mostly conducted on four major expressways outside the city where traffic was not congested. The G6 Beijing-Tibet Expressway (G6 BT, former Badaling Expressway) is located in the northwest; the G2 Beijing-Shanghai Expressway (G2 BS, former Beijing-Tianjin-Tanggu Expressway) is

located in the southeast connecting Beijing and Tianjin, an important port city in Northern China; the G1 Beijing-Harbin Expressway (G1 BH, former Beijing-Shenyang Expressway) is located in the east and leading towards the northeast part of China; and the G45 Beijing-Kaifeng Expressway (G45 BK) is located in the south and leading towards to the southern provinces. These expressways are the major corridors where trucks bring goods to and from Beijing or to ports for shipment. We selected trucks that were separated from others for data analysis. In some cases we failed to collect data that were clearly from individual trucks. We did not use this data. The bus chasing measurements were conducted in area A in Figure 4.1, located in northwest of Beijing and close to the 5th Ring Road. Several famous tourist attractions, e.g. the Summer Palace, the Fragrant Hills Park, and the Yuanmingyuan Park, are located in this area, resulting in heavy bus activities. The non-bus traffic volume in this area is low compared to the business districts, making it feasible to follow the target buses for an extended period.

During our on-road sampling, we selected trucks and buses that we were separated from others and that could be followed for periods of time necessary to assure valid pollutants data recording. Typically we followed the target vehicles for 1 to 7 minutes. We sampled a variety of trucks from various origins as identified by license numbers. The distance between our minivan and target vehicle were estimated to be between 5m to 50m. Ronkko (2007; 2006) showed that the nucleation mode particles are already formed after 0.7s residence time in the atmosphere (corresponding to ~ 5 m behind the target vehicle) and no significant changes are observed for longer residence time, which indicates that when the distance between

our platform and target vehicle was greater than 5m, it did not significantly impact the particle concentration and size distribution.

The on-road measurement study period was dominated by stagnant weather conditions, during which no strong winds were encountered. The wind speed during the on-road chasing period was in the range of 1 to 5 m/s according to the data shown in the Weather Underground Website (http://www.wunderground.com).

4.2.2 Data analysis

4.2.2.1 Vehicles classification

We sort data from trucks by their region of origin and buses by their emission compliance standards. The identification of different types of trucks and buses was based on the information such as the labels of fuel types and emission standard (Euro equivalent rating), the model, vehicle structure, license plate numbers, etc. Information regarding engine types was considered to be important and was readily observable on most buses, but was rarely apparent on trucks. License plate numbers were used to identify the regions where the trucks were registered. The number of the sampled trucks and buses is presented in Table 4.1. More details of sampled vehicles are discussed in Section 4.3.1.

4.2.2.2 Adjustments to Aethalometer BC data

In our monitoring campaign, the BC mass concentration was measured by a micro Aethalometer. The operating principle of the Aethalometer is based on the

measurement of the optical attenuation of a beam of light transmitted through a sample collected on a filter, and the mass of BC on the filter is calculated as:

$$BC = \frac{ATN}{\sigma} = \frac{100\ln\left(\frac{I_0}{I}\right)}{\sigma}$$

where ATN is a dimensionless parameter representing the light attenuation; I_0 and I refers to the transmitted light intensity at the initial condition and after particle loading, respectively(Hansen et al., 1984); the parameter σ, also known as the attenuation coefficient, is assumed constant and it implies a linearly proportional relationship between BC and ATN. It is recently reported that as the filter becomes loaded with particles, freshly-collected ones shadow those that are previously collected resulting an underestimation of BC readings (Arnott et al., 2005; Jimenez et al., 2007; Kirchstetter and Novakov, 2007; Park et al., ; Virkkula et al., 2007; Weingartner et al., 2003). This effect causes non-linearity of the BC-ATN relation. The non-linear BC-ATN relationship is pronounced when the Aethalometer filter is heavily loaded. This non-linearity can be corrected using an empirical correction algorithm. In our study, the algorithm presented by Kirchstetter et al.(2007) is adopted. We assume that the Aethalometer filter was clean initially, when it reported the correct BC data. The following equation is used to compensate the filter loading effect of BC concentration:

$$BC = \frac{BC_0}{0.88T_r + 0.12}$$

where BC and BC_0 refer to the corrected and original BC mass concentration, respectively; T_r is the measured filter transmission, which can be computed directly

from the ATN measured by the Aethalometer (Kirchstetter and Novakov, 2007):

$$T_r = \exp\left(-\frac{ATN}{100}\right)$$

4.2.2.3 Emission factor calculations

The overall average and individual EFs are calculated based on the data collected during the on-road measurements. Carbon balance calculations are based on the fuel combustion process, relating the emission of carbon-containing species from vehicle exhaust to fuel consumption (Geller et al., 2005; Kirchstetter et al., 1999; Stedman, 1989; Stedman et al., 1991). We consider CO_2, CO and BC as the carbonaceous products during the combustion process. We do not have measurements of hydrocarbon (HC) gases. Neglecting HC should have a minor effect in our calculations since gas-phase carbonaceous products are usually dominated by CO_2 (Ning et al., 2008; Yli-Tuomi et al., 2005). The fuel-based emission factors are calculated in the following equation:

$$EF_p = \frac{\Delta[P]}{\Delta[CO_2] \times \frac{MW_C}{MW_{CO_2}} + \Delta[CO] \times \frac{MW_C}{MW_{CO}} + \Delta[BC] \times \frac{MW_C}{MW_{BC}}} \times w_c$$

where $\Delta[i]=[i]-[i]_0$, i = P, CO_2, CO and BC; subscript 0 denotes the baseline value; EF_p is the emission factor of pollutant P in grams of pollutant emitted per kilogram of fuel consumed (g kg^{-1}); $\Delta[P]$ is the on-road concentration of pollutant P above the baseline, with the unit of grams per cubic meter of air (g m^{-3}); $\Delta[CO_2]$, $\Delta[CO]$ and $\Delta[BC]$ represent the increase of CO_2, CO and BC, with the unit of grams per cubic meter of air (g m^{-3}); MW_i is the molecular weight of the i^{th} species; w_c is the

mass fraction of carbon in the fuel. For this protocol we assume homogeneous dilution of the species of interest; an assumption which is reinforced by the averaging of multi minute observations.

In calculations, we consider BC to be elemental carbon. The properties of diesel fuel are collected from a number of reports on Chinese fuel standards, according to which fuel density is reported to be approximately 0.85 and carbon content is 85.5% with an estimated 5% variability (BQTSB, 2007; GAQSIQPRC, 2003)

In the calculation of overall average EFs, Westerdahl et al.(2009) took the ambient concentrations recorded at a quiet campus location at the end of each trip as the baseline concentrations. The elevation of the pollutant level on road was regarded as the contribution of all vehicles. EFs presented in Westerdahl et al. (2009) could best be considered as 'fleet" averages collected on a roadway dominated by heavy duty diesel traffic. Improved techniques were employed by Wang et al.(2009) which allowed data on a limited number of individual vehicles to be generated. In the present study, our objective is to derive the EFs for many individual vehicles in real traffic conditions. Specifically, when we met the target vehicle, the elevated concentrations above the proximal baselines recorded by the instruments for each chase event were considered as the contribution of the target vehicle in front of our mobile platform. This real-time baseline is chosen to calculate the EFs of individual vehicles. This is similar to the approach taken by Ban-Weiss et al. (2009a) in their road tunnel tests. The recorded video and written notes are used to confirm the identity of the target vehicle and to identify the corresponding CO_2 peak in the data files. The real-time baseline for an individual vehicle is chosen in the period before the sudden increase or

after the sudden drop of the corresponding CO_2 concentration before or after chasing that vehicle, depending on the traffic condition around it. We selected events where a 30ppm of CO_2 increase was observed as successful capture of the plume of the target vehicles in our study as described by Ban-Weiss (2009a). In a few cases, this approach was not applicable. For example, insufficient CO_2 elevation (<30ppm) during the vehicle presence indicating that we had not actually captured the plume from the vehicle or when multiple trucks or buses were present in front of our mobile van. Valid exhaust plumes were identified for 230 out of 246 trucks and 57 out of 59 buses. CO data were not available for G2 BS roadway because of sensor failure on that day. Therefore we only have 195 CO observations for all truck measurements.

In EF calculations, we group the on-road concentrations in 10-second intervals, and then calculate the EFs for each interval during the whole trip. The EFs for each vehicle are derived based on the mean values of emission factors during the periods when we were chasing individual target vehicles.

4.2.2.4 Calculation of size resolved $PM_{0.5}$ mass concentration

Size resolved $PM_{0,5}$ number concentration was recorded by FMPS for all vehicles. In order to calculate the mass concentration of each size bin of $PM_{0.5}$ particles, a particle density value is required. However, particle density data are not readily available for diesel exhaust. Park et al.(2003) reported that the effective density of diesel vehicle emission particles decreases with increasing particle size, which was between 0.3 and 1.2 g/cm^3. Maricq et al.(2000) made the similar observation reporting the particle density of diesel vehicle emissions to range from 0.2 to 1.5 g/cm^3. Sakurai

et al.(2003a; 2003b) reported that the nuclei mode of diesel engine was mainly composed of volatile material from lubrication oil whose density is 0.9 g/cm^3. In research conducted by Kittelson et al. (2004), the density of 1 g/cm^3 was selected for diesel vehicles. Symonds (2007) investigated the density of diesel exhausted particles, reporting a nearly flat profile with densities varying from 0.6 g/cm^3 to 0.9 g/cm^3 for all particle sizes smaller than 200nm. To simplify the calculation of mass concentration of different size particles, we assume an effective density, ρ, of 0.9 g/cm^3 in this chapter, and generated the size resolved mass concentration in the following equation (Seinfeld et al., 2006):

$$\frac{dM}{dlog(Dp)} = \frac{dN}{dlog(Dp)} \times \rho \frac{\pi}{6} {D_p}^3$$

where dN/dlog(D_p) and dM/dlog(D_p) refers to the normalized number and mass concentration, respectively; D_p is the mean diameter of the selected size range. In the calculation of size resolved EFs, dN in above equation is replaced by d(ΔN) and correspondingly dM is replaced by d(ΔM), where Δ indicates that the background concentration is subtracted from the plume concentration and the number represents the contribution of the target vehicle. The size resolved EFs of $PM_{0.5}$ number and mass concentration are presented in Section 0.

4.3 Results and discussion

4.3.1 Sampled vehicle statistics

As shown in Section 4.2.1, our on-road measurements were conducted on six days, during which we chased 246 trucks and 59 buses. Eliminating the invalid chase

observations or instruments failures, valid data were collected for the following number of trucks: 195, 230 and 229 for CO, BC and $PM_{0.5}$, respectively, and the number of valid observations for buses were: 32, 57 and 52 for these pollutants. The characteristics of the sampled truck and bus population and speed distribution are presented in Figure 4.2.

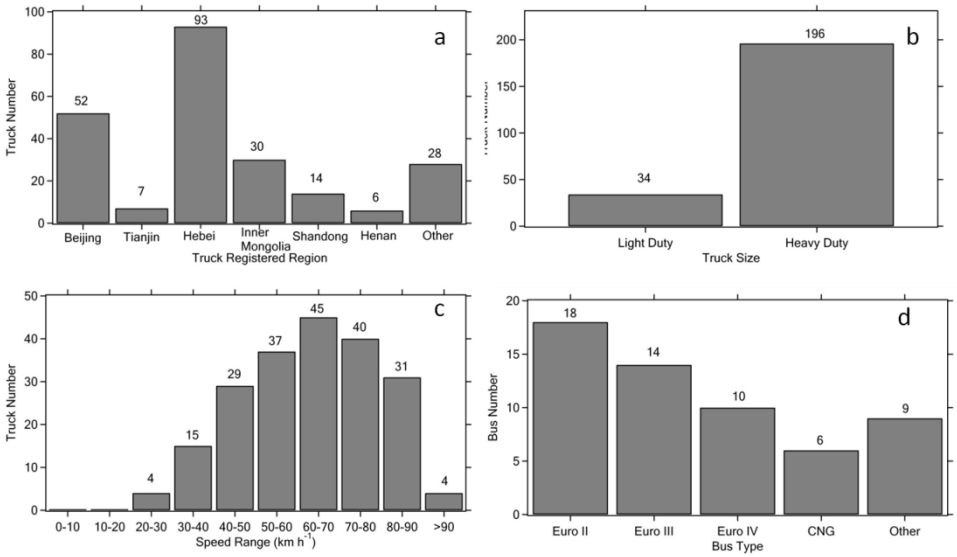


Figure 4.2 Vehicle statistics chart
(a: Number of trucks by region; b: number of trucks by type; c: number of trucks by speed; d: number of buses by emission standard)

The trucks sampled during the 6 day campaign came from 17 provinces, but a majority of them (88%) were registered in 6 provinces, shown in Figure 4.2a. Trucks from Beijing, Hebei and Inner Mongolia accounted for the largest portion of the truck traffic around Beijing. Heavy-duty trucks (with gross weight greater than 6 metric tons) dominated our samples (Figure 4.2b). According to our notes, all sampled trucks were heavy-duty on the G2 BS Expressway, and the outbound truck volume was greater than the inbound volume on the same Expressway, while on other expressways, both outbound and inbound truck traffic were balanced.

The speed distribution chart (Figure 4.2c) shows that nearly 70% sampled trucks were in the range of 50~80 km/h. The low speed vehicles (<30km/h) were diesel tricycles driving on service roads that ran parallel to G6 BT and G45 BK. The speed of urban buses ranged from 10 to 40 km/h. Figure 4.2d presents the numbers of buses with different emission standards.

4.3.2 Observations from on-road measurements

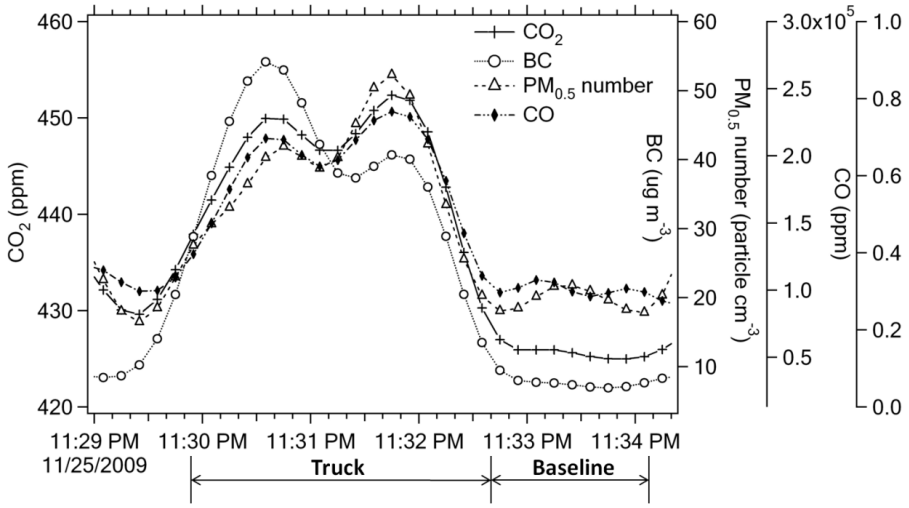


Figure 4.3 An example of time series data collected during and after chasing a truck on G6 BT Expressway

Figure 4.3 presents an example of data observed during on-road chase studies. The figure demonstrates several important factors regarding the nature of event sampling that provided the data for emission factor calculations. The pollutant time series made up of 10 second data averages align well for the pollutants under study during the chase event which is in excess of 2 minutes long. There is a brief baseline before and a much longer stable baseline period following the event; in this case the

baseline used for calculations was from the later data. The nature of data from all monitors provide adequate time resolution to represent the event; even the CO data from the Q-trak, which has the slowest response times, follows the event data shown by monitors with known very fast response.

Mean baseline data observed during truck and bus event sampling for BC was 24.8 and 6.9 ug m^{-3}, $PM_{0.5}$ number counts was 5.45×10^5 and 6.95×10^4 particle cm^{-3}, CO_2 was 495 and 463 ppm, and CO was 2.1 and 1.2 ppm, respectively. It should be noted that we use individual baselines, rather than the mean baselines, for the EF calculations.

4.3.3 Emission Factors

4.3.3.1 EF distributions and heavy emitters

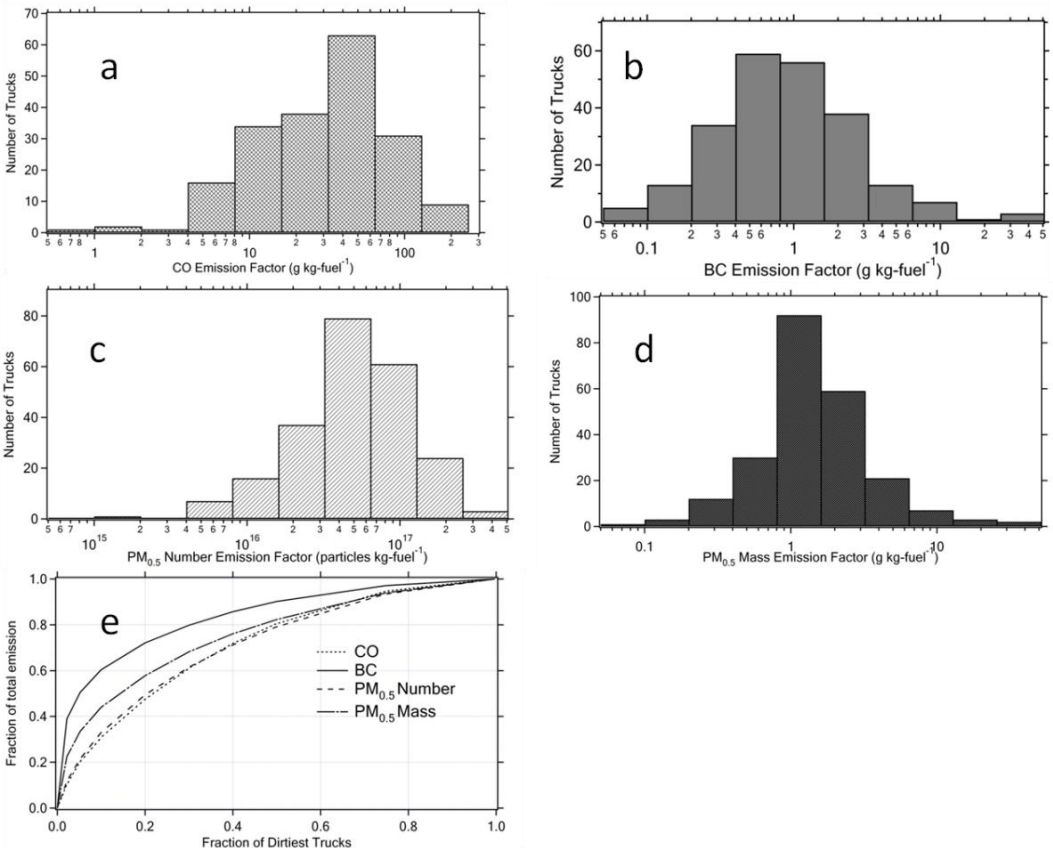


Figure 4.4 EF distributions and cumulative distribution of truck emission (a: CO; b: BC; c: $PM_{0.5}$ number; d: $PM_{0.5}$ mass; e: cumulative distribution)

Figure 4.4 presents the EF distributions for CO, BC, $PM_{0.5}$ number and mass (Figure 4.4a-d) for trucks and the cumulative distribution of truck emissions (Figure 4.4e). The BC, $PM_{0.5}$ number and mass EF distributions all show a log-normal shape, with the median value of 0.84 g kg-fuel^{-1},5.2×10^{16} particles kg-fuel^{-1}, and1.34 g kg-fuel^{-1} respectively, and the mean value of 2.21 g kg-fuel^{-1} and 7.1×10^{16} particles kg-fuel^{-1}, 2.35 g kg-fuel^{-1} respectively. The CO EFs do not show a clear log-normal shape. The median value of CO EFs for all trucks is 33 g kg-fuel^{-1} and mean value is 42 g kg-fuel^{-1}. Zielinska et al.(2004) reported PM EFs, ranging from 0.3-1.0 g kg-fuel^{-1} from chassis dynamometer test of 4 diesel vehicles; Schneider et al. (2008) reported

BC and particle (10-300nm) number EFs of 0.22 g kg-fuel^{-1} and 8×10^{15} part kg-fuel^{-1}, respectively, by conducting on-road chasing study of 46 trucks in Aachen in Germany; Ban-Weiss et al.(2008) also reported the fleet average BC EFs ranging from 0.85 to 0.99 g kg-fuel^{-1} from the road tunnel study. The mean BC EFs in this study is comparable with that in Ban-Weiss (2009a), who did the similar evaluation of emission distribution and reported the mean BC EF of 1.7 g kg-fuel^{-1}. Moreover, we find that the median BC EF is 63% of median $PM_{0.5}$ mass EF, suggesting the BC domination of $PM_{0.5}$. Kleeman et al (2000) and Robert et al (2007)reported that elemental carbon (EC) accounted for large portion (greater than 50%) in PM mass emissions during driving cycle. The further analysis of correlation between BC and size resolved $PM_{0.5}$ mass is discussed in section 0 of this chapter.

The $PM_{0.5}$ number EF reported in this chapter is significantly greater than our previous studies (Wang et al., 2009; Westerdahl et al., 2009). One factor that may contribute to this was the low ambient temperatures encountered in the 2009 winter campaign. The mean temperature of our on-road test days in 2009 was 2 ℃ while the mean temperature was 26 ℃ in 2008 and 27 ℃ in 2007 (Weather underground website: http://www.wunderground.com/). Virtanen et al. (2006) reported that the total aerosol particle concentrations in wintertime were 2-3 times higher than in the summertime. Wei et al. (2001) and Kittelson et al.(2004) also presented that the colder ambient temperature contributed to increased nano-particle formation by conducting the winter on-road measurement and laboratory dilution tunnel test. It is also possible that fuel and fleet engine changes contributed to this observation.

As illustrated in Figure 4.4a-d, the number of trucks with large emission factors is

small. These “heavy emitters” accounted for a large fraction of the total emissions from the sampled vehicles. Assuming that the fuel consumption rates and vehicle mileage traveled (VMT) are the same for all the sampled vehicles, the cumulative distribution of truck emissions presented in Figure 4.4e indicates that approximately 5% of the trucks are responsible for 50% BC emission, and 20% of the trucks are responsible for nearly 50% CO and $PM_{0.5}$ number emission, nearly 60% $PM_{0.5}$ mass and over 70% BC emission. Considering that the heavy emitters typically have higher fuel consumption rates than regular vehicles, the contributions of heavy emitters to the overall emissions could even be higher than estimated here. These results are consistent with findings from similar studies in the U.S.(Ban-Weiss et al., 2009a; Cadle and Stephens, 1994; Guenther et al., 1994; Sadler et al., 1996). Ban-Weiss et al (2009a) found that the highest-emitting 10% heavy duty trucks were responsible for 42% of total BC emissions based on 226 diesel trucks in California, while in our study, the contributions of the heavy emitters to the total fleet average emission in Beijing were even more significant, especially for BC. During our study, the highest BC emitting truck was encountered on G2 BS expressway and its BC EF was almost 70 times larger than the median value of other trucks. Another type of diesel vehicle, a three-wheeled goods truck, was sampled on the G45 BK expressway. Its CO emission was 10 times, and its BC emission was 60 times of the median values of other trucks. These vehicles are not commonly driven on expressways but are very common in rural areas and small cities. They are powered by one to three cylinder engines and use diesel fuel. This single observation must not be viewed as representative, but it is similar to estimates made by Sperling(2005) that Chinese rural vehicle emitted as

much total air pollution as all other motor vehicles and they were likely having a large impact on local air quality. Our findings suggest that a small number of heavy emitters account for a large fraction of the total emission, and, thus removing the heavy emitters, for example, rural vehicles, from the road, or improving identified heavy emitters could be an effective strategy in reducing the overall emission.

It is noted that vehicle speed also affects emission factors. However, a majority of the trucks (70%) we sampled were operated in a relatively small speed range (50-80 km h^{-1}), and we did not identify a clear speed dependency in the emission factors from our dataset.

4.3.3.2 EFs of trucks from different regions

Figure 4.5 shows the EFs of trucks from different regions. Trucks from Beijing had the lowest BC mass EFs among all the regions, and the statistical analysis (t-test) demonstrates that the BC EFs of Beijing trucks are 60% ($p=0.0015$) lower than the overall mean value of all trucks. However, $PM_{0.5}$ number EFs of Beijing trucks are lower than all other regions, but this trend is not statistically significance ($p = 0.35$). The relatively low emissions rates from Beijing vehicles are possibly due to the more stringent emission and fuel quality standards implemented for trucks registered and fuels sold in Beijing. In 2008, Euro IV fuel standard were implemented in Beijing, which constrained the fuel sulfur content to 50ppm, while in most parts of China, the sulfur standard for diesel was 500ppm. Similar observations were made by Liu (2009), who tested 75 diesel vehicles using PEMS. Liu's work showed that PM EFs of diesel trucks in Beijing was 20-50% lower than in Xi'an, NOx was 15-30% lower in Beijing

than in Xi'an, while CO was 50-70% lower. The regional variation of trucks' EFs also suggests that improving engine and fuel standards in other areas would be an effective approach to improve the traffic-related air pollution in Beijing.

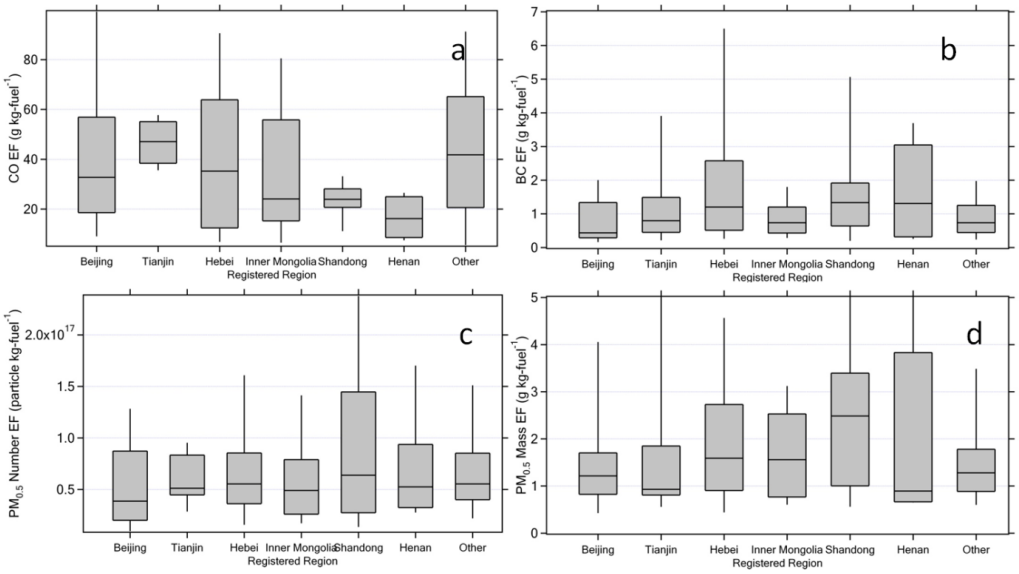


Figure 4.5 EFs of diesel trucks by region

4.3.3.3 Size resolved EFs of $PM_{0.5}$ number and mass

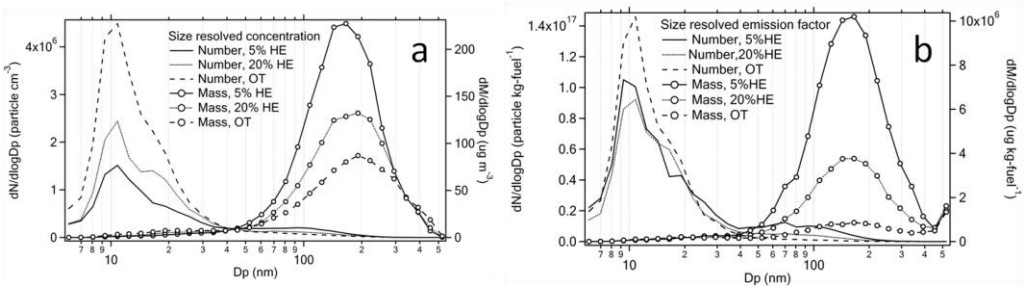


Figure 4.6 Size resolved concentrations (a) and EFs (b) by truck type (5%HE: top 5% BC Heavy Emitters; 20%HE: top 20% BC Heavy Emitters; OT: Other Trucks)

Based on the methods described in section 4.2.2.4, the size resolved number and mass concentrations of $PM_{0.5}$ are shown in Figure 4.6a, and the corresponding size resolved number and mass EFs are shown in Figure 4.6b. Both concentration and EFs

are sorted into three groups: the top 5% BC heavy emitters, top 20% BC heavy emitters and other trucks. In the EF calculation, with the background concentrations subtracted, the number and mass EF size distributions (Figure 4.6b) show the same shape and modes as the absolute number and mass concentration distributions (Figure 4.6a), indicating that vehicle emissions dominate the on-road particle concentrations. The $PM_{0.5}$ number EF size distributions are dominated by a mode of particles at approximately 11 nm (standard deviation 1.4 nm). This mode is observed for all the trucks and buses. Since the background concentrations are removed, the results reported here are attributed to be the contribution of the vehicles chased. The second mode of number EF size distribution of trucks is approximately 80nm (standard deviation 38 nm). Differing from the number EF distributions, the $PM_{0.5}$ mass EF size distributions are dominated by one mode, at approximately 110nm. Our results of EF number and mass size distributions are consistent with the findings of other groups. Kittelson et al. (2006) evaluated four heavy-duty diesel engines by conducting the on-road and laboratory measurement and reported that all four engines produced bimodal size distributions with the nuclei mode ranging in size from 6 to 11nm while the second mode (accumulation mode) from 52 to 62nm for all on-road test. Maricq et al. (1998) also did a chassis dynamometer and on-road chasing tests for a diesel passenger car engine and found uni-modal number size distribution with peaks around 50-70nm in laboratory test while an additional nucleation mode around 10-20nm in the on-road test by using high sulfur fuels. Another roadside test was performed in Helsinki, Finland, by Virtanen et al.(2006), who reported two modes of particle number distributions: nucleation mode at 19nm and the second mode at 75nm. One

note was that all these measurements were completed with Scan Mobility Particle Sizer (SMPS) while our size distribution data were reported by an FMPS.

The solid line and short dash lines in Figure 4.6b represent the $PM_{0,5}$ EF size distribution for both number and mass of the top 5% (12 trucks) and top 20% (46 trucks) BC heavy emitters. The long dash line represents the remaining 184 trucks, which are considered as "other trucks". Both BC heavy emitters and other trucks show the same shape of $PM_{0.5}$ size distribution in number and mass EFs with different peak values for each mode. The mass EFs of the 110nm mode of top 5% and top 20% BC heavy emitters are 11 and 4 times higher than other trucks, respectively. Furthermore, as shown in Figure 4.7, the correlation coefficients (R^2) between BC EFs and size-resolved $PM_{0.5}$ mass EFs indicate that BC is a significant part of the $PM_{0.5}$ mass between 120nm and 250nm. Similarly, Robert et al.(2007) conducted the chassis dynamometer experiments for 4 heavy duty diesel trucks, finding that the peak in PM mass EF distribution between 100-180nm was dominated by elemental carbon.

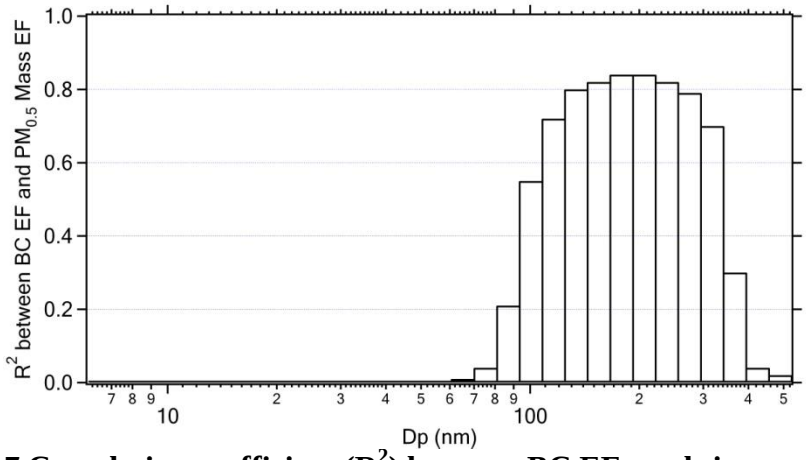


Figure 4.7 Correlation coefficient (R^2) between BC EFs and size resolved $PM_{0.5}$ mass EFs

However, the peak number EFs for the 10nm mode particles shown in Figure 4.6b did not show high values for BC heavy emitters. Ban-Weiss et al. (2009a) observed a similar trend when investigating the diesel EF of BC and ultrafine particle number. Kittelson et al.(2006) also reported similar findings and explained that the accumulation mode particles provided the surface for adsorption of volatile hydrocarbons that would otherwise be incorporated into the nuclei mode. Furthermore, higher BC concentration could potentially lead to increased coagulation of small particles and there by reduced nucleation mode particles

4.3.3.4 EFs of buses

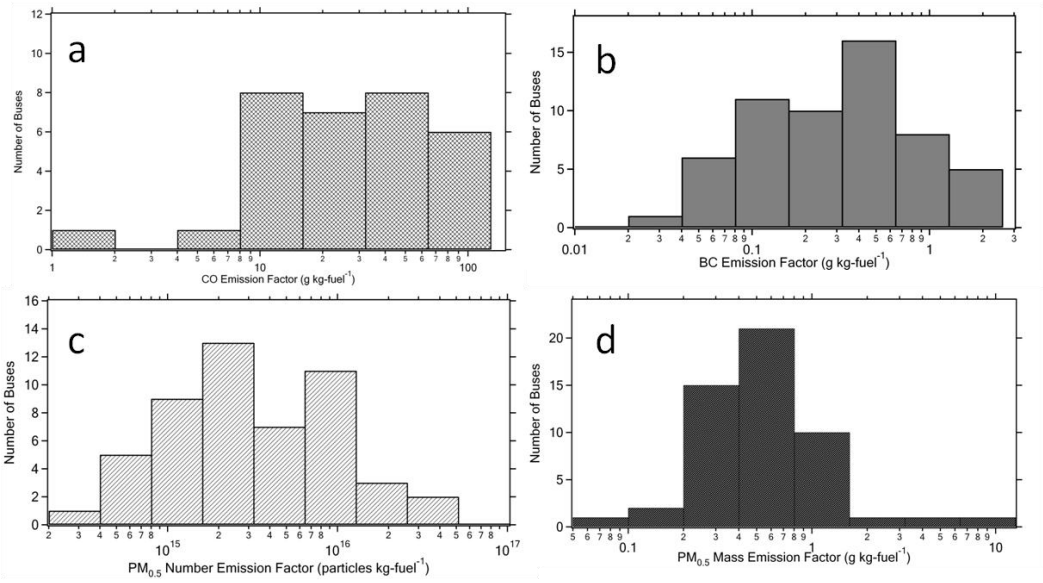


Figure 4.8 EF distributions of buses

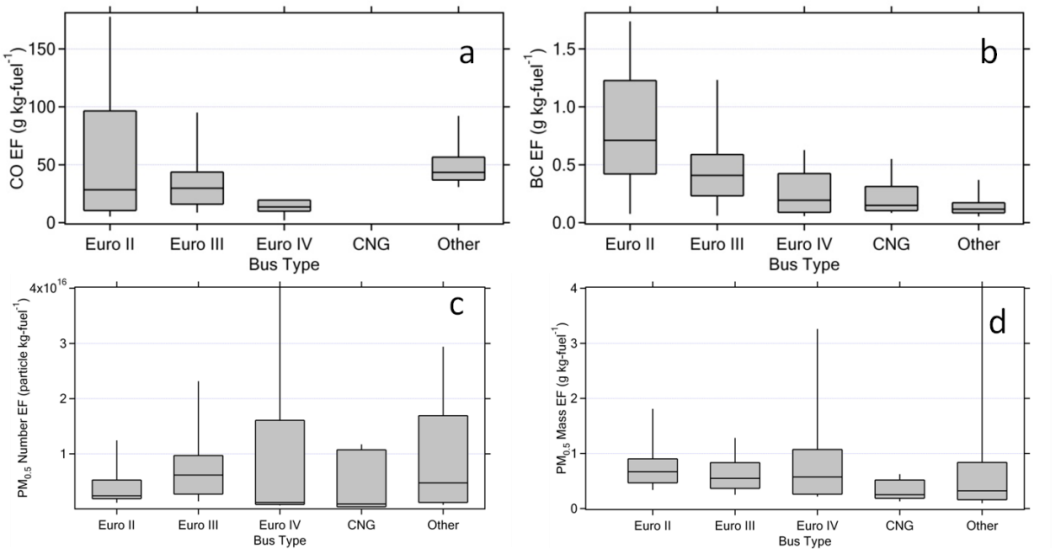


Figure 4.9 EF of buses with different engine emission certification levels

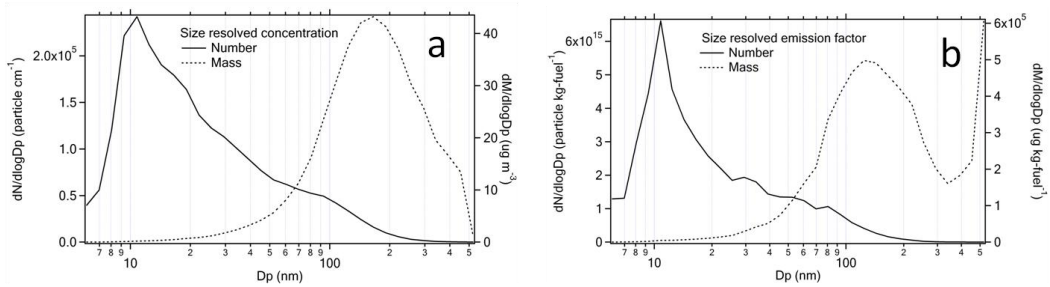


Figure 4.10 Size-resolved particle mass and number concentrations and the corresponding EFs for buses

The EFs for 57 buses are calculated based on two bus chasing trips during our winter campaign, shown in Figure 4.8 through Figure 4.10. For most of the buses the engine certification level was available, allowing the comparison between Euro II, Euro III, Euro IV and Compressed Natural Gas (CNG) buses. Those not showing their engine certification levels are termed "other" in Figure 4.9. The EF distributions of buses for three pollutants do not show strong log-normal patterns shown in Figure 4.8a-d, possibly due to the smaller sample size compared to the truck data. However, the EFs exhibit clear decreased BC emissions with increased engine certification level

(Figure 4.9b), The mean BC EFs are 0.71, 0.41, 0.20 and 0.15 g kg-fuel^{-1} for Euro II, Euro III, Euro IV and CNG buses, respectively. The statistical analysis also reveals that the mean EFs of Euro IV buses are 76% (p=0.03) and 69% (p=0.0004) lower than those of Euro II buses, for CO and BC, respectively. The mean BC EFs of CNG buses are 75% (p=0.0003) lower than the mean BC EFs of Euro II buses. However, the $PM_{0.5}$ number EFs of Euro IV and CNG buses show reducing trends compared to those of Euro II buses but are not statistically significant (p=0.3-0.4). Due to the temporarily instrument failure during our test period, we did not have the CO EFs for all CNG buses. Our data suggests that the $PM_{0.5}$ number EFs of Euro III buses are 93% (p=0.04) greater than those of Euro II buses. There is no ready explanation for this, but the number of observations included for comparison is small. However, other studies have reported that as controls have been implemented to reduce particle mass emissions of diesel engines that particle number emissions have increased (Bagley, 1996). The lower emissions from Euro IV and CNG buses suggest that introducing clean-burning buses is an effective way to reduce the overall vehicle emissions.

Size resolved $PM_{0.5}$ number and mass concentration as well as the corresponding EFs are illustrated in Figure 4.10a-b. Similar to the trucks, the $PM_{0.5}$ number EF of buses shows the first mode around 11nm. Among all 51 buses investigated, 40 buses show a second mode of $PM_{0.5}$ number EFs, around 57 nm, and 11 buses show the third mode around 96 nm.

4.3.4 Factors affecting the EFs calculation and method limitations

As described in Section 4.2.2.3, EFs are calculated by performing the carbon balance equation between the emission pollutant and fuel consumption. Since CO_2 accounts for the major part (over 98% by mass) of the carbon emission in the plume, the CO_2 concentration contributed by the individual vehicle becomes the most sensitive parameter which affects the EF results. For example, the 2% variance of CO_2 baseline concentration causes 26%-40% variance of pollutant EF results, while the 2% variance of CO and BC baselines causes 0.1%-5%, and 0.0%-1.4%, variance of pollutant EF results, respectively. This indicates that imprecision in the CO_2 baseline period selection would be the major source of uncertainty in calculating the EFs. As discussed in Section 4.2.2.3, the baseline periods were chosen before or after the vehicle chasing where the CO_2 concentration was the lowest, as reconfirmed by examining the on-road videos. In order to estimate the magnitude of the error, the EFs are re-estimated by choosing a different baseline period, e.g. selecting the baseline period after the vehicle chasing if the original was before the chasing, and vice versa. The results show that when choosing the different baseline period, the median value of EFs of all trucks increased -21.1%, 6.5%, 21.0%, for CO, BC and $PM_{0.5}$ number respectively. The changes reflect that baseline period selection is important for EF calculation, especially for CO and $PM_{0.5}$ number. Practically, we chose the baseline by examining the on-road traffic conditions obtained by video recording.

In order to check the impact of different time interval of original data outputs, we evaluate the impact of using 10 second interval averaging to describe data during events. The medians of EFs of all trucks are not significantly changed, increasing

2.1%, 0.3%, -0.6% for CO, BC and $PM_{0.5}$ number EFs, respectively when 10 second averaging is employ vs. 5 second averaging. Considering the calculation efficiency and the nature of instrumental responses, a 10-second time interval is an acceptable number for averaging the instrument output data during the events.

4.4 Conclusion

This study demonstrates a time- and cost-effective approach to obtain the emission characteristics of individual on-road trucks and buses, and to obtain the distribution of the emission factors (EFs) from this on-road population. All on-board instruments provided rapid response, which captured emission data during real-world vehicle driving conditions. The protocol was flexible allowing the selection of different types of target vehicles, and periods when only one target vehicle was in front of the mobile platform to reduce the impact of other on-road vehicles. This approach allows us to sample a large number of vehicles: 230 individual trucks on 4 major expressways around Beijing as well as 57 individual buses on city streets, representing fleets of real trucks and buses during real on-road behaviors. Not all events were suitable for quantification, but the large sample size enabled to identify the heavy emitters in the on-road fleet.

EFs for CO, BC and $PM_{0.5}$ are derived from the measurements and for trucks frequency distributions of the BC, $PM_{0.5}$ number and mass, EFs follow a clear log-normal pattern with the median EFs of BC at 0.84 g kg-fuel^{-1}, EF of $PM_{0.5}$ number at 5.04×10^{16} particle kg-fuel^{-1} and that of $PM_{0.5}$ mass at 1.34 g kg-fuel^{-1}, respectively.

The size resolved $PM_{0.5}$ number EFs demonstrate bimodal maximums at approximately 10 nm and 80nm, while the mass EFs distributions demonstrate a unimodal maximum at approximately 110nm. In addition, the BC mass emissions are shown to be well correlated with the mass emission of particles with 100~250nm diameters. All of these results are consistent with those from laboratory dynamometer tests. We also observe that while black carbon and carbon monoxide levels are reduced in urban buses with increasingly stringent controls and that PM number does not show the same degree of reductions.

The results suggest and/or provide support for the following air pollution control measures: removing heavy emitters from roadways; implementing emissions reduction programs targeted at heavy emitters; improving fuel quality in the entire region (not limited to particular cities); and introducing clean-burning buses into cities. Our results can also be used to quantify the benefits of implementing those measures.

There are limitations to our refined chasing methods. For example, a careful data analysis procedure must be followed to ensure the validity of the collected data in representing the plumes of sampled vehicles. This is very time consuming. In addition, we have little control over the driving conditions of the sampled vehicles. Thus the derived EFs only reflect the conditions when we were conducting the measurements. Similarly, we do not have the detailed vehicle information as compared to the laboratory or PEMS tests. In collaboration with local agencies, we plan to conduct a pilot program to acquire more vehicle information through the recorded license numbers in an upcoming study. Further, we do not measure all the pollutants of concern in this study, most notably oxides of nitrogen. This study represents a step

forward in characterizing the nature of emissions of trucks and buses in the Beijing area. However, application of the findings to fully represent the many kinds of vehicles and the conditions that they operate will require much more research.

CHAPTER 5

NITROGEN OXIDES AND BLACK CARBON EMISSION FACTOR DISTRIBUTIONS OF INDIVIDUAL ON-ROAD DIESEL VEHICLES IN TWO CHINESE CITIES

5.1 Introduction

The transportation sector has gradually become a dominant contributor to the already high air pollution levels in China due to the rapid growth in vehicle population (Fu et al., 2001; Hao et al., 2000; Hao J, 2001; Walsh, 2007). In large Chinese cities such as Beijing, Shanghai and Guangzhou, emissions of nitrogen oxides (NOx) from motor vehicles accounted for 41%~ 70% of total urban NOx pollution in 2002 (Yi et al., 2007). Diesel and gasoline exhaust was shown to account for 8% of fine particulate matter ($PM_{2.5}$) in Beijing on annual basis in 2000 (Zheng et al., 2005) and 15% during the summertime in 2004 (Song et al., 2007).

Environmental impacts associated with vehicular emissions are tremendous. NOx act as precursors to photochemical formation of ozone (O_3) (Marr et al., 2002; Marr and Harley, 2002; Sawyer et al., 2000), which can exacerbate chronic respiratory diseases and cause short-term reductions in lung function (Bernard et al., 2001). NOx also contributes to the secondary formation of $PM_{2.5}$ (Harrison et al., 1997; He et al., 2001; Tan et al., 2009). $PM_{2.5}$ exposure is associated with serious human health impacts, including increased rates of hospitalization and death (Mar et al., 2005; Pope Iii and Dockery, 2006; Schwartz and Neas, 2000). Black carbon, a major PM

component, is reported to be the second strongest contributor, after carbon dioxide (CO_2), to current global warming (Ramanathan and Carmichael, 2008), and also affects the large-scale circulation and hydrologic cycle with significant regional climate effects (Menon et al., 2002).

Table 5.1 Summary of emission standards for new vehicles and fuel quality[a]

New registered vehicle					
		China I	China II	China III	China IV
HDDV	Beijing	2000-01-01	2003-01-01	2005-12-30	2008-07-01[b]
	Nationwide[c]	2001	2004-07-01	2008-01-01	
LDGV	Beijing	1999-01-01	2003-01-01	2005-12-30	2008-03-01
	Nationwide	2000	2004-07-01	2008-07-01	2011-07-01
Fuel quality (sulfur content in ppm)					
Diesel	Beijing	2002-01-01(2000)	2003-10-01(500)	2005 (350)	2008-01-01(50)
	Nationwide	2002-01-01(2000)	2003-10-01(500)	2011-07-01(350)	
Gasoline	Beijing		2003(500)	2005 (150)	2008-01-01(50)
	Nationwide	2000 (1000)	2006-12-06(500)	2010-01-01(150)	

a: source:(Fung et al., 2010; Wu et al., 2011; Zhang et al., 2010)
b: only for public fleets including buses, postal and sanitation vehicles.
c: Beijing, Shanghai, Guangzhou comply advanced standards

To reduce the impact of vehicle emissions on urban air quality, China has launched a series of vehicle emission control measures and policies, including adoption of European Union-like emission standards for new vehicles, improving fuel quality, and banning heavy emitters from urban Beijing. Stage I and II emission standards for Chinese vehicles (China I, China II) were promulgated on April 16, 2001, and State III, IV, and V emission standards for Chinese vehicles (China III, China IV and China V) on May 30, 2005. The actual implementation dates of new vehicle emission standards and fuel quality standards are listed in Table 5.1. China II, III, and IV emission standards are essentially equivalent to Euro II, III, and IV

emission standards, respectively, in regulating emission rates of vehicle engines for gaseous pollutants including carbon monoxide (CO), hydrocarbons (HC), nitrogen oxides (NOx), and particulate matter (PM) (GAQSIQPRC, 2001, 2005). According to Ministry of Environmental Protection (MEP) of China, total NOx emission control (i.e., from various sectors such as the power industry, the cement industry and transportation) was introduced in the "The Twelfth Five-year Plan" (http://www.zhb.gov.cn/zhxx/hjyw/201103/t20110314_206764.htm). Effective implementation of total NOx emission control requires accurate NOx emission inventories.

Emission factors (EFs), expressed as the weight of pollutant divided by a unit weight, volume, distance, or duration of the activity emitting the pollutant, are useful in assessing vehicular emissions and form the basis of emission inventories. Recently, Portable Emission Measurement Systems (PEMS) were employed by researchers to quantify emission factors of on-road diesel vehicles in several major Chinese cities (Chen et al., 2007a; He et al., 2010; Liu et al., 2009). He et al. (2010) and Liu et al. (2009) reported that no significant reduction in NOx EFs was observed between China III and China I diesel trucks. Wang et al. (2011) developed a refined chasing method and conducted on-road measurements in and around Beijing, from which on-road EFs of carbon monoxide (CO), black carbon (BC) and ultrafine particles (UFP) for 230 individual diesel trucks and 57 diesel buses were reported. The chasing method compliments PEMS method. Differing from PEMS, where individual vehicles are instrumented and operated on-road, the chasing method employs a mobile platform equipped with on-board fast response instruments to monitor real-time emissions of

the vehicle in front of it. It is more flexible than PEMS, in part because it is possible to identify specific vehicles for study over periods as brief as several minutes and then to switch target vehicles. This approach is capable of collecting emission data for a large number of vehicles within a few hours. By generating EF distributions for a sample of Beijing area trucks and buses, Wang et al. (2011) made a number of important findings, including identification of heavy emitters and their disproportionate contribution to total on-road emissions, observation of lower BC EFs from trucks registered in Beijing, and a demonstration of the effectiveness of introducing lower emission diesel buses. This work significantly expanded the sample size compared to previous studies, performed with PEMS or more traditional dynamometer methods and the results reflected real-world emissions of diesel vehicles.

This chapter describes the findings from the field campaign on charactering on-road EFs of Chinese diesel trucks conducted in December 2010, focusing on NOx and BC emission factors of Chinese diesel trucks. NOx emission standards were issued as part of China I ~ China V standards but the effectiveness in NOx reduction is still uncertain (He et al., 2010). BC, whose emission limit is not regulated directly in current Chinese emission standards, is an important component of diesel PM. In this chapter, we are not directly addressing effectiveness of China emission control for PM, but instead, we report BC as a PM indicator.

Primary objectives of this chapter were to 1) characterize on-road NOx emissions of Chinese diesel vehicles, 2) compare NOx and BC EFs of diesel trucks from different regions in China using the refined on-road chasing method developed in Wang et al. (2011). This is the first time that NOx emission factor was investigated by

chasing study in China. We conducted on-road measurements in two cities, Beijing and Chongqing, and sampled 440 on-road diesel trucks in total. As the capital of China and the host city of the 29th Olympic Games, the city of Beijing is a pioneer in implementation of stringent vehicle emission control measures during the past decade. Chongqing, locating approximately 1,500 km southwest of Beijing and in the east edge of Sichuan Basin, is the largest municipality in terms of population (~ 32 million) in China. On-road emissions studies are rare in Chongqing. The results from our study will provide scientific basis for implementing total NOx emission control as well as multi-pollutant emission control regulations in China.

5.2 Methodology

5.2.1 Instrumentation

The mobile platform was a gasoline powered minivan. (Wang et al., 2009; Wang et al., 2011; Westerdahl et al., 2009). Instruments were powered by two 12 V deep-cycle storage battery/inverters (500W at 120V). Air sampling was conducted through a side window. Particle-phase air pollutants were sampled using flexible conductive tubing while NOx was sampled using high density Teflon tubing. All sampling lines were situated with the inlets approximately perpendicular to the airflow outside the van. Self-pollution was not apparent upon inspection of the individual time series observations (Wang et al., 2011). Isokinetic sampling was not attempted because particles studied in this chapter were in the submicron size range, and the Stokes number was in the range of 10^{-7}~10^{-3}, indicating that the effects of particle inertia and

isokinetic sampling were not significant (Baron and Willeke, 2001; Wang et al., 2011). Instruments used for on-road measurements were as follows: a Vaisala CARBOCAP (Model GM70, Vaisala) measured CO_2 concentration using Non-Dispersive Infrared (NDIR) methods; a Micro-Aethalometer (Model AE51, Magee Scientific) was employed to measure BC mass concentration by optical analysis; a NO_2 converter (Model 401, 2B Technologies) connected with a NO monitor (Model 410, 2B Technologies) measured NOx concentration using Ultraviolet (UV) absorbance method. In addition, a Fast Mobility Particle Sizer (FMPS, Model 3091, TSI Inc.) reporting particle number concentration and size distribution (5.6-560 nm) and a Q-Trak (Model 7565, TSI Inc.) reporting carbon monoxide, humidity and temperature were operated in the mobile platform but their monitoring results are not reported in this chapter. These instruments were selected because of their low power requirements, rapid response, proven ability to produce high quality data, and small size.

The recording time intervals of Micro Aethalometer (BC), Vaisala CARBOCAP (CO_2) and NO monitor (NOx) were set to be 1 second, 5 seconds and 10 seconds, respectively. These were the lowest practical reporting rates for these devices. BC and CO_2 observations were then averaged to 10 seconds. The adoption of instruments with fast response enabled us to identify the corresponding pollutant data of the target vehicles more precisely. It was also flexible allowing the selection of many different types of target vehicles in desired locations as well as the selection of proper time when only one target vehicle was in front of the mobile platform with minimum impact of other on-road vehicles. A digital camcorder was operated in the front seat of

the van and recorded the traffic conditions continuously during the on-road measurements. The recorded video and written notes served to describe the on-road conditions and to identify the target vehicle for linkage with the monitoring data. Internal clocks of all instruments were synchronized to a portable computer, which also functioned as a data logger for the CARBOCAP, Micro Aethalometer and NOx monitor. The license plate numbers of target vehicles and the speed of our mobile platform were written into a notebook during the on-road measurements. The license plates include identification of the province of registration for each truck. It is possible, in principal, to obtain further information, such as registration date, engine model, emission standard and etc. from these license plate numbers. However, such data are not easily available in China. As a proof of concept, registration and engine information of 11 Chongqing trucks were obtained by investigating their license plate numbers after the campaign. Road gradients were qualitatively determined during each chase event in Chongqing and were recorded on written notes and camcorder for further analysis.

In our study, BC concentrations reported by Micro-Aethalometer were adjusted after experiments to account for high filter loading levels. Theories and methods of adjustment are presented in Wang et al.(2011).

5.2.2 Sampling routes and period

The on-road chasing studies were conducted on the 6th Ring Expressway in Beijing and the Inner Ring Expressway in Chongqing. Sampling routes in the two cities are illustrated in Figure 5.1. Dates, time, weather conditions and the number of

sampled diesel trucks for each field trip are listed in Table 5.2.

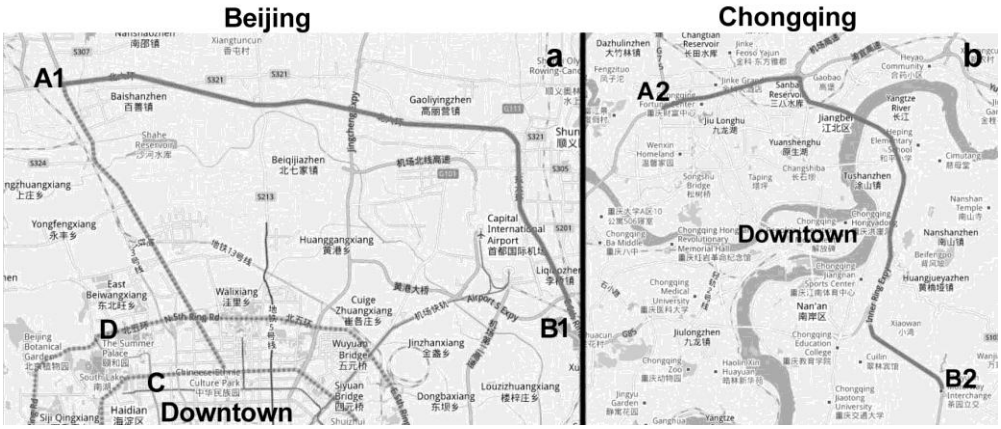


Figure 5.1 Sampling maps of Beijing (panel a) and Chongqing (panel b). (A1-B1: Sampling route in Beijing: the 6th Ring Expressway (48.1km per trip); A2-B2: Sampling route in Chongqing: the Inner Ring Expressway (25.1km per trip); C: the 4th Ring Road of Beijing; D: the 5th Ring Road of Beijing.)

Table 5.2 Sampling dates, time and routes of sampled trucks

Sampling date	Sampling time	Sampling area	Temp	RH	Vehicle sampled			
			C	%	Trucks	Up[a]	Le	Dn
2010-12-09	2:00pm-5:00pm	BJ 6R Expy[b]	3~5	39~42	24		24	
2010-12-11	1:30pm-5:30pm	BJ 6R Expy	0~2	24~32	56		56	
2010-12-13	1:30pm-5:30pm	BJ 6R Expy	-5~-3	19~24	54		54	
2010-12-14	1:30pm-5:30pm	BJ 6R Expy	-4	22~24	45		45	
2010-12-17	12:30pm-2:30pm	BJ 6R Expy	7~8	13	13		13	
		Total(Beijing):			192		192	
2010-12-21	11:00am-3:30pm	CQ IR Expy[c]	8~14	44~71	37	21	18	4
2010-12-22	11:00am-3:00pm	CQ IR Expy	8~9	66~76	58	24	22	16
2010-12-23	11:00am-3:00pm	CQ IR Expy	9	76~82	59	32	26	7
2010-12-26	10:45am-2:30pm	CQ IR Expy	5~6	81	53	28	31	7
2010-12-27	9:45am-1:00pm	CQ IR Expy	2~8	66~87	41	22	27	2
		Total(Chongqing):			248	127	124	36

a: Up: uphill segment; Le: level segment; Dn: downhill segment; b: Beijing 6th Ring Expressway; c: Chongqing Inner Ring Expressway

We sampled 192 diesel trucks in Beijing on five days from December 9th to December 17th 2010; 248 diesel trucks in Chongqing on five days from December 21st to December 27th 2010. Trucks in Beijing were all sampled on level gradient roads. However, Chongqing is a mountainous city. Measurements in Chongqing included

level, uphill and downhill segments. 127 trucks were sampled on uphill gradient roads, 124 trucks were sampled on level gradient roads and 36 trucks were sampled on downhill gradient roads. Suitable data from downhill samples were less available than for uphill and level samples because plumes emitted from trucks on downhill segments were usually small. It was difficult to capture acceptable data from these plumes and observe significant signals on instruments in our protocol. The basic operational requirement for chase methods employed to date is that a clear CO_2 plume must be identified when following a single truck.

5.2.3 Emission factor calculations

The individual EFs are calculated based on the data collected during the on-road measurements. Carbon balance calculations are based on the fuel combustion process, relating the emission of carbon-containing species from vehicle exhaust to fuel consumption (Geller et al., 2005; Kirchstetter et al., 1999; Stedman, 1989; Stedman et al., 1991). We consider CO_2, CO and BC as the primary carbonaceous products of the combustion process. We do not have measurements of hydrocarbon (HC) gases. Neglecting HC should have a minor effect in our calculations since gas-phase carbonaceous products are usually dominated by CO_2 (Ning et al., 2008; Yli-Tuomi et al., 2005). The fuel-based emission factors are calculated using the following equation:

$$EF_p = \frac{\Delta[P]}{\Delta[CO_2] \times \frac{MW_C}{MW_{CO_2}} + \Delta[CO] \times \frac{MW_C}{MW_{CO}} + \Delta[BC] \times \frac{MW_C}{MW_{BC}}} \times w_c$$

where $\Delta[i]=[i]-[i]_0$, $i = P, CO_2, CO$ and BC; subscript 0 denotes the baseline

value; EF_p is the fuel based emission factor of pollutant P in grams of pollutant emitted per kilogram of fuel consumed (g kg^{-1}); $\Delta[P]$ is the on-road concentration of pollutant P above the baseline, with the unit of grams per cubic meter of air (g m^{-3}); $\Delta[CO_2]$, $\Delta[CO]$ and $\Delta[BC]$ represent increases of CO_2, CO and BC, with the unit of grams per cubic meter of air (g m^{-3}); MW_i is the molecular weight of the i^{th} species; w_c is the mass fraction of carbon in the fuel. For this protocol we assume homogeneous dilution of the species of interest; an assumption which is reinforced by the averaging of multi minute observations.

In our calculations, MW_{BC}, MW_{CO}, MW_{CO2} are taken to be 12, 28 and 44 g mol^{-1}. NOx is treated as NO_2 equivalent with MW_{NO2} as 46 g/mol (EPA, 2010; GAQSIQPRC, 2005). When calculating NOx EFs, the impact of intake air humidity and temperature is considered (Yanowitz et al., 2000). The following formula is adopted to calculate the correction factor of NOx concentration (Dodge et al., 2003; lindhjem et al., 2004):

$$kNOx = 1 + 0.00446(T - 25) - 0.018708(H - 10.71)$$

$$NOx_{corrected} = \frac{NOx_{original}}{kNOx}$$

Where T is the ambient temperature in degree C, H is the ambient water content in g-H_2O kg-dry-air^{-1}. Ambient temperature and humidity data are downloaded from weather underground website (http://www.wunderground.com/) and shown in Table 5.2. The average kNOx in Chongqing and Beijing are 1.03 and 1.07, respectively.

The properties of diesel fuel were taken from a number of reports on Chinese fuel standards, according to which fuel density is reported to be approximately 0.85 and

carbon content is 85.5% with an estimated 5% variability (BQTSB, 2007; GAQSIQPRC, 2003)

In this study, EF calculations are performed for individual diesel trucks in real traffic conditions on expressways in each area. This approach is described in detail by Wang et al. (2011). On-road concentrations were reported or averaged to 10-second time intervals, and then EFs were calculated for each interval. Emitted pollutant concentrations were obtained by subtracting baseline concentration from real time concentration for each pollutant. EFs for each vehicle were derived based on the mean value of EFs during the periods when we were chasing individual target vehicles. For vehicles operating on multiple road gradients, EFs were derived based on mean value of corresponding segments (uphill, level, downhill) of chasing periods.

5.3 Results and discussion

5.3.1 Descriptions of sampled vehicles

Properties of sampled vehicles in two cities are illustrated in Figure 5.2, categorized by regions, vehicle classes and on-road speed. In Beijing, a majority (56%) of sampling trucks were registered in Beijing and Hebei provinces, whereas in Chongqing, 80% of sampling trucks are locally registered. The truck fleet was dominated by heavy duty diesel trucks (gross weight greater than 6 metric tons) in both cities as determined by visual characteristics. Speed distribution chart shows that most of sampling trucks were in the speed range of 60~70 km h^{-1}, and the average speed of trucks sampled in Beijing is 8% greater than that in Chongqing. In this

chapter, "*trucks sampled in Beijing*" came from more than 10 provinces, while "*Beijing-registered trucks*" only account for 17% of the trucks sampled in Beijing.

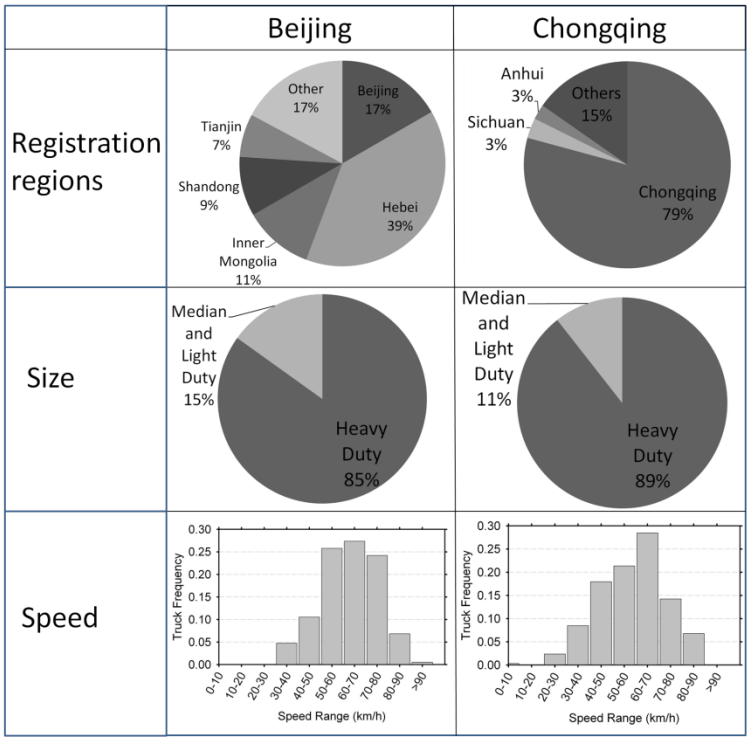


Figure 5.2 Characteristics of trucks of distribution of speed

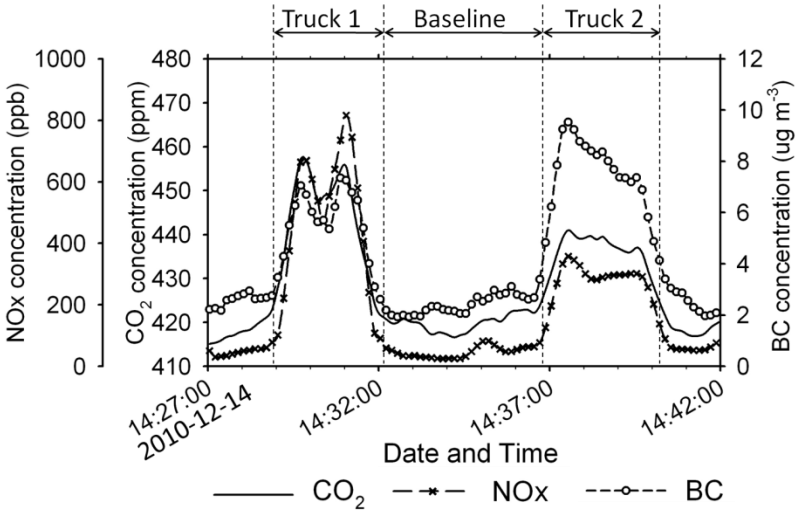


Figure 5.3 An example of time series data collected during chasing study on the 6th Ring Expressway in Beijing

Figure 5.3 presents an example of data observed during on-road chase studies.

The figure demonstrates several important factors regarding the nature of event sampling that provided the data for emission factor calculations. The pollutant time series made up of 10 second data averages align well for the pollutants under study during the two chase events which are in excess of 2 minutes long for each. There is a stable baseline period between the two events; in this case this baseline was used for calculations. The nature of data from all monitors provides adequate time resolution to represent the two events.

5.3.2 EFs of trucks on roads with different road gradients in Chongqing

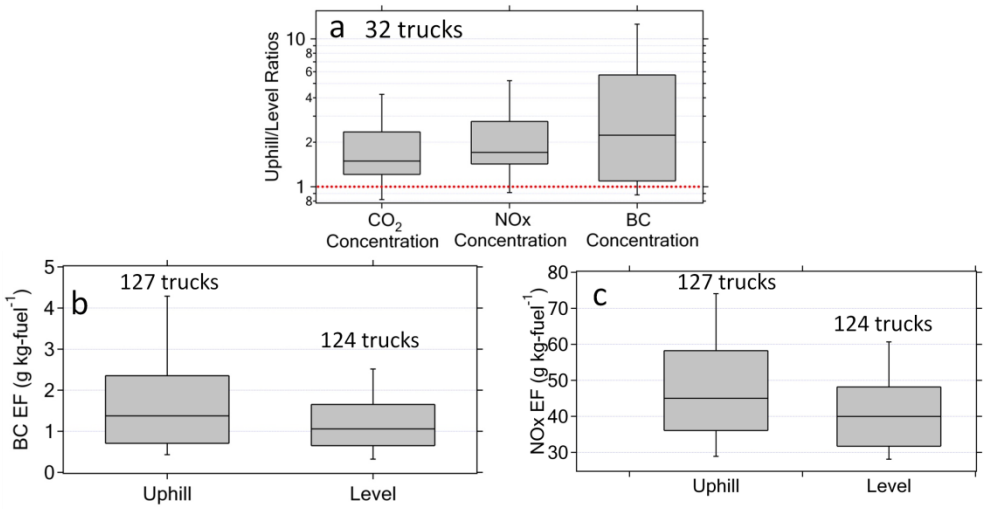


Figure 5.4 EFs of trucks on uphill and level gradient roads in Chongqing (panel a: Uphill/Level ratios of pollutant concentration of 32 trucks; panel b: BC EFs; panel c: NOx EFs)

Road gradient effect on NOx and BC emissions from heavy-duty diesels were investigated in our Chongqing field campaign because of the mountainous terrain in this city. We sampled 127 trucks on uphill roads and 124 trucks on level roads. 32 of them were on both uphill and level roads. The uphill/level ratios of emitted pollutant

concentration for those 32 trucks are shown in Figure 5.4a. NOx and BC EFs of 127 trucks on uphill roads and 124 trucks on level roads are shown in Figure 5.4b-c. In Figure 5.4, the centerline, top line and bottom line of each box refer to median, 75% percentile and 25% percentile value of samples, respectively. The whisker top and bottom refer to 90% and 10% percentile of samples, respectively. In Figure 5.4a, median values of uphill/level ratios of NOx and BC concentrations are 1.7 and 2.2, respectively, indicating higher pollutant emissions on uphill roads. The median value of the uphill/level ratio of CO2 concentration is 1.5, which is indicative of the greater fuel consumption of trucks on uphill roads because CO2 dominates carbon emission and is proportion to fuel consumption (Ning et al., 2008; Yli-Tuomi et al., 2005). In Figure 5.4b-c, the median BC EF of trucks on uphill and level roads are 1.4 and 1.1 g/kg-fuel, respectively; the median NOx EF of trucks on uphill and level roads are 45.0 and 40.0 g/kg-fuel, respectively. In our study, the EFs increase caused by road gradient increase is not as large as the results obtained by Rexeis et al. (2005) and Boulter et al. (2007). One reason is that we report fuel-based EFs whereas Rexeis and Boulter report distance-based EFs. Both fuel consumption and pollutant emission of trucks increase as road gradient increases, leading to moderate increase of fuel-based EFs,

EFs of trucks on downhill roads are not listed in Figure 5.4 because most of trucks on downhill roads emitted smaller plumes that were hard to clearly detect as required by our protocol. The sample size of trucks on downhill roads (36 trucks) is much smaller than that on uphill (127 trucks) and level roads (124 trucks).

5.3.3 BC and NOx EFs of trucks sampled in Beijing and Chongqing

As shown in Figure 5.2, trucks sampled in Beijing were primarily registered in five regions: Beijing, Hebei, Inner Mongolia, Shandong and Tianjin; trucks sampled in Chongqing were primarily registered in Chongqing. BC and NOx EFs of trucks registered in different regions are shown by box and whisker plots in Figure 5.5. The centerline in each box refers to median value; bottom and top lines refer to the 1st quartile and 3rd quartile value, respectively; whisker bottom and top lines refer to the 10 percentile and 90 percentile of the sample. In Figure 5.5a and Figure 5.5b, vertical dash lines divide samples by their sampling regions: the left part of the dash line refers to trucks sampled in Beijing whereas the right part of the dash line refers to trucks sampled in Chongqing.

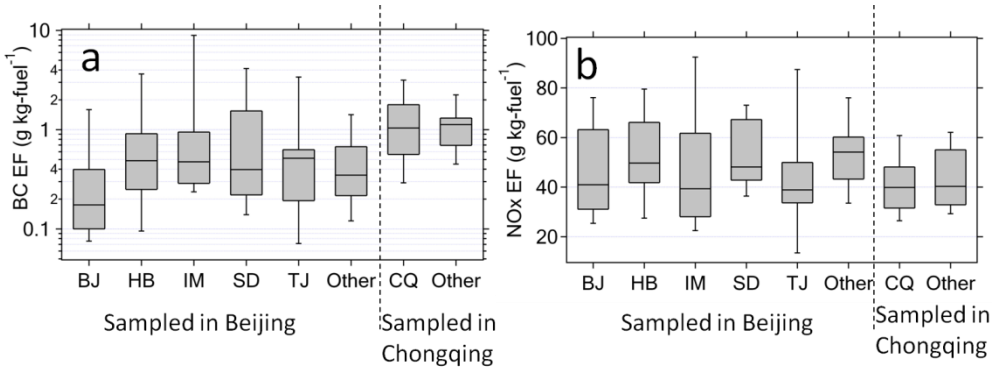


Figure 5.5 EFs of trucks registered in different regions (a:BC EF;b: NOx EF; BJ: Beijing; HB: Heibei; IM: Inner Mongolia; SD: Shandong; TJ: Tianjin; CQ: Chongqing)

5.3.3.1 EF of diesel trucks registered in different regions

Shown in Figure 5.5a, the median BC EF of Beijing-registered trucks is 0.2 g kg-fuel^{-1}, which is 63% lower than that of trucks sampled in Beijing but registered in other regions; it is also 84% lower than that of trucks registered in Chongqing. This is

possibly due to implementation of the more stringent emission standard implemented in Beijing than in other regions of China. China III standard for new registered vehicles were enforced from December 30, 2005 in Beijing but only from January 1, 2008 in other Chinese cities (Fung et al., 2010; Wu et al., 2011). If we assume the age distributions of on-road trucks are the same in every city, the earlier implementation date in Beijing suggests that China III diesel trucks account for a greater portion of the on-road fleet in Beijing than in other cities. Moreover, improved fuel quality in the region may also contribute to the findings.

In contrast, as shown in Figure 5.5b, the median NOx EF of Beijing-registered trucks is not the lowest among entire samples. Median NOx EF values for Beijing trucks are 18% lower than those from Hebei and 15% lower than those from Shandong, but it is 4% higher than those from Inner Mongolia, 5% higher than those from Tianjin and 3% higher than those from Chongqing. This observation indicates that emission control measures, such as advanced emission standards and improved fuels implemented in Beijing, have limited impacts on NOx emission improvement. This agrees with that presented by He et al. (2010), which characterized EFs of diesel vehicles in Beijing, Shenzhen and Xi'an, finding that NOx EFs of China III heavy duty diesel vehicles (HDDV) is greater than that of China I and II.

Furthermore, the median value of BC EFs of trucks sampled on level roads in Chongqing (right side of the vertical dash line in Figure 5.5a) is greater than that of trucks sampled in Beijing (left side of the vertical dash line in Figure 5.5a). As listed in Table 5.3, the median value of NOx EFs of trucks sampled in Beijing and Chongqing are 47.4 and 40.0 g kg-fuel-1, respectively; the median value of BC EFs of

trucks sampled in Beijing and Chongqing are 0.4 and 1.1 g kg-fuel-1, respectively. The NOx EFs for trucks sampled in Beijing are comparable to the value reported in other studies conducted in China (Chen et al., 2007a; He et al., 2010; Liu et al., 2009; Wang et al., 2001) and in USA (Ban-Weiss et al., 2008; Jimenez et al., 2000; Kirchstetter et al., 1999; Yanowitz et al., 1998). These results are also listed in Table 5.3.

Table 5.3 NOx and BC emission factor from this study and other sources

Sample categories	Mean of samples (g kg-fuel^{-1})	Median of samples (1st quartile~3rd quartile) (g kg-fuel^{-1})
NOx, Chongqing Uphill	48.8	45.0(36.1~58.0)
NOx, Chongqing, Level	42.1	40.0(31.7~48.1)
NOx, Beijing, Level (2010)	50.7	47.4(38.1~63.0)
BC, Chongqing Uphill	2.2	1.4(0.7~2.3)
BC, Chongqing, Level	1.6	1.1(0.7~1.6)
BC, Beijing, Level (2010)	1.2	0.4(0.2~0.8)
BC, Beijing, Level (2009)	2.2	0.8(0.4~1.7)
	(Source) Method	
NOx, HDDV in Beijing	(He et al., 2010), PEMS	40.9±11.4 (E1)[a] 40.1(30~50) (E2) 46.1(35~58) (E3)
NOx, DV in Xi'an	(Liu et al., 2009) PEMS	48(37~60)(E2) 50(44~58)(E3)
NOx, HDDV in Shanghai	(Chen et al., 2007a) PEMS	43.1 (E0 E1)
NOx, HDV in Guangzhou	(Wang et al., 2001), Tunnel	34 (E0)
NOx, HDDT in USA	(Jimenez et al., 2000), Remote Sensing	45±2
NOx, HDDT in CO, USA	(Yanowitz et al., 1998), Chase dynamometer	39±2
NOx, HDDT in CA, USA	(Kirchstetter et al., 1999) Tunnel	42±5
NOx, HD in CA, USA	(Ban-Weiss et al., 2008) Tunnel	40±3

a: E refers to Euro. In this study, Euro I, II, III standard are equivalent to China I, II, III standard.

Lower BC EFs of trucks sampled in Beijing may be attributed to better fuel quality with much lower sulfur level in the Beijing area than in other areas in China as well as a possibly newer fleet composition (Fu et al., 2001; Hao et al., 2006; Katherine et al., 2006; Zhang et al., 2010). Hao et al. (2006) and Liu et al. (2008a) show that

sulfur content of fuels significantly affects emissions of light duty gasoline vehicles (LDGV) and HDDV. Zhang et al. (2010) analyzed 107 diesel samples from gas stations along highways in China. He found that majority of their diesel fuel samples contained sulfur ranging from 500 ppm to 2000 ppm, which was equivalent to pre-China II diesel vehicle fuel standard. Because no advanced fuel quality standards have been implemented on a nationwide basis in China since 2007, the sulfur contents of diesel fuel reported by Zhang et al. (2010) are likely representative of diesel fuel in Chongqing area. In Beijing, on the other hand, sulfur content of diesel fuel must be less than 50 ppm to comply with the new China IV emission standard beginning in 2008 (Wu et al., 2011).

5.3.3.2 BC EFs of diesel trucks sampled in Beijing from 2008 to 2010

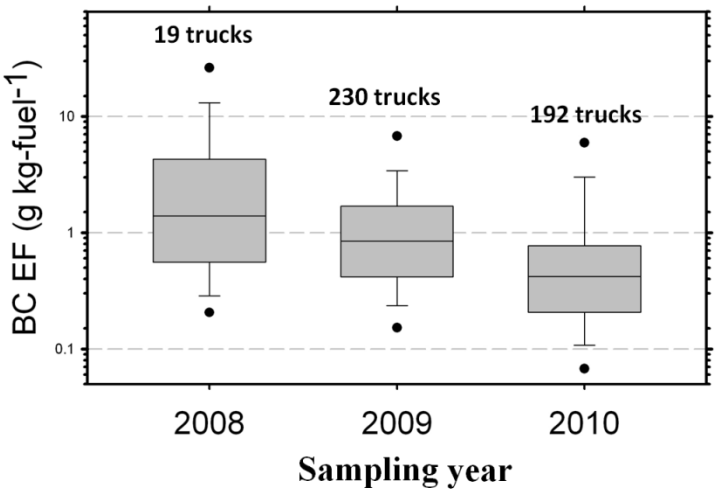


Figure 5.6 BC EFs of HDDV sampled in Beijing from 2008 to 2010

BC EFs of diesel trucks sampled in Beijing in 2008, 2009 and 2010 are shown by box and whisker plots in Figure 5.6. The median BC EF of diesel trucks sampled in

Beijing in 2010 is 0.4 g kg-fuel-1, which is 50% and 70% lower than that we collected in 2009 and 2008 studies, respectively. In 2008, trucks sampled in Beijing were dominated by Beijing-registered trucks due to the restriction of non-Beijing trucks during our sampling period in that year; in 2009 and 2010, trucks sampled in Beijing are from more than 10 provinces. Since PM emission standard in Beijing are more stringent than nationwide standards, this downward trend in BC EF may reflect the effectiveness of advanced vehicle emission standard (China III) implemented nationwide since 2008 (Wu et al., 2011).

5.3.3.3 Linking on-road EF with registration data

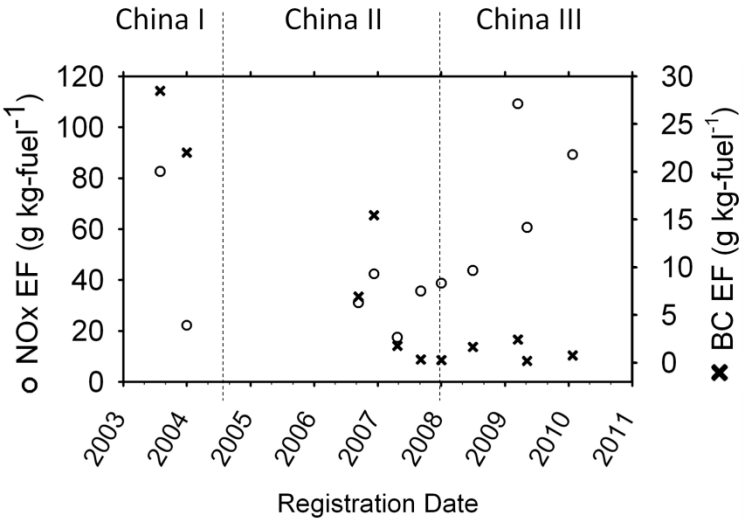


Figure 5.7 EFs, registration dates and required emission standards of 11 diesel trucks

Truck plate numbers provides a way to obtain further information, such as truck registration date, engine model and emission standard for each truck on the road. However, this data is not readily available. As a proof-of-concept study, the

registration information of 11 trucks sampled in Chongqing was acquired from governmental sources based on their license plate numbers for further analysis. EFs, registration dates and complied emission standards of the 11 trucks are illustrated in Figure 5.7.

A decreasing trend of BC EFs is observed from China I to China III emission standard among the 11 diesel trucks. However, a similar trend is not observed for NOx EF. This observation further indicates that advanced emission standards (i.e. China III) implemented might have limited impact on NOx emissions. The reader is cautioned that this small sample does not represent the overall group of vehicles tested or the larger fleet. Further evaluations of the larger data set would prove valuable to reinforce the tentative observations.

New emission standards will be enforced in Beijing (China V) and other Chinese cities such as Chongqing (China IV) in 2012. As fuel economy and BC (and likely PM) emissions improve, the implementation of effective control technologies is critical for NOx emission reduction.

5.3.4 EF distributions and high emitting diesel vehicles

Figure 5.8 presents the EF distributions and the cumulative distributions (by assuming that the fuel consumption rates and vehicle mileage traveled are the same for all the sampled vehicles) of trucks sampled in Chongqing and Beijing. NOx and BC EFs are grouped by different road gradient in Figure 5.8. Median and mean values of EFs are represented by solid and dash lines in each subplot in Figure 5.8, corresponding to the values listed in Figure 5.8. Both NOx and BC EFs of trucks show

log-normal shapes except that a second mode is observed for BC EFs of trucks sampled in Beijing (Figure 5.8h). Further analysis shows that Hebei-registered trucks dominate this mode, with BC EFs ranging between 2.9 and 9.9 g kg-fuel-1.

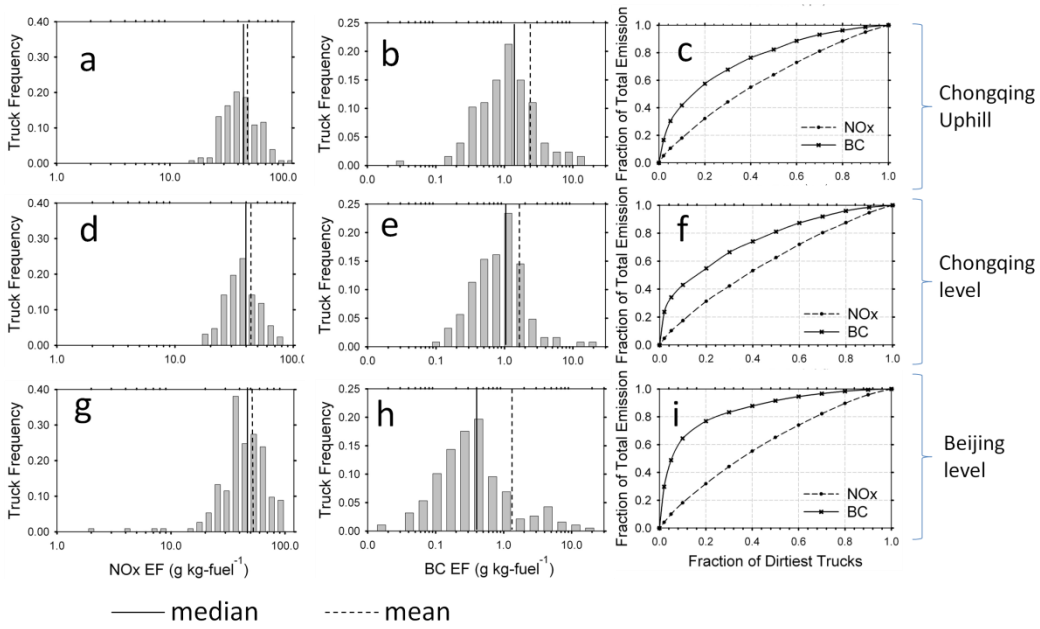


Figure 5.8 EF distributions of trucks and cumulative distribution of truck emission
(first row (a-c): on Chongqing uphill roads; second row (d-f): on Chongqing level roads; third row (g-i): on Beijing level roads; first column (a,d,g): NOx EF distribution; second column (b,e,h): BC EF distribution; third column (c,f,i): cumulative distributions)

Cumulative distributions of truck emissions are presented in Figure 5.8 (c,f,i). On level roads, the top 10% highest emitting trucks are responsible for 17.5% NOx emission and 43.0% BC emission in Chongqing, and 18.1% NOx emission and 64.4% BC emission in Beijing. The 20% highest emitting trucks are responsible for 40.7% NOx emission and 54.8% BC emission in Chongqing, and 31.9% NOx emission and 76.9% BC emission in Beijing. The proportion of total BC emission attributed by high

emitting trucks sampled in Beijing is similar as that we observed in 2009 study (Wang et al., 2011).

Our results suggest that, while a small number of high BC emission trucks contribute disproportionally to the total BC emissions, high NOx emission trucks do not dominate the total NOx emissions. Furthermore, we observed that high BC emission trucks are usually not high NOx emission sources and vice versa (Figure 5.9**)**. For example, the dash lines in Figure 5.9 refer to the 90 percentile value of NOx EF and BC EF of the entire sample. The median NOx EF value of the 10% highest BC emission trucks (top-left area in Figure 5.9) is 41.8 g kg-fuel^{-1}, while the median NOx EF value of entire sample is 43.8 g kg-fuel^{-1}; The median BC EF value of the top 10% highest NOx emission trucks (bottom-right area in Figure 5.9) is 0.4 g kg-fuel^{-1}, while the median BC EF value of entire sample is 0.6 g kg-fuel^{-1}.

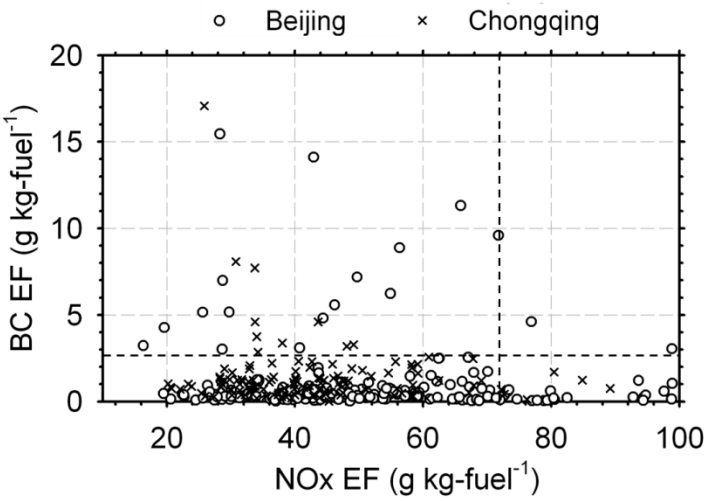


Figure 5.9 Relationship between NOx EF and BC EF of individual trucks in Chongqing and Beijing

These observations, as well as findings in Section 5.3.3.1 and Section 5.3.3.3 of this chapter, suggest the challenges in multi-pollutant control strategies for diesel

trucks in China. Though NOx emission limits of diesel vehicles came into effect in China II and China III emission standard, effectiveness of control technologies in NOx reduction was uncertain. Control technologies complied with each Chinese emission standard is presented by Wu et al. (2011). For China II HDDVs, a combination of electronic fuel injection (EFI), turbo charger with intercooler and combustion improvement is suggested. For China III HDDVs, the control technology is the combination of electronic high pressure common rail/unit pump and combustion improvement. However, these control technologies are primarily put in place to limit PM emission, with some possible improvements in NOx emissions. The upcoming China V and VI emission standards for HDDV,will focus on both NOx and PM controls, which requires selective catalytic reduction (SCR), fuel quality improvements and diesel particulate filter (DPF). The main challenge is to improve fuel efficiency without increasing NOx emission because high NOx concentration is typically formed in diesel engine chambers with relatively high oxygen concentration and temperature, which is the favorable condition to fuel efficiency improvement.

5.4 Conclusion

Emissions of 440 diesel trucks were sampled in Beijing and Chongqing, China, by employing the on-road chasing method developed in Wang et al.(2011). It's the first time that chasing study was used to investigate on-road NOx emissions in China. EFs for BC and NOx were calculated, and their EF distributions were reported. Median values of NOx EFs for trucks in Chongqing and Beijing are 40.0 and 47.4 g kg-fuel^{-1}, respectively, on level roads. These numbers are comparable to data reported by other

studies in China. Median values of BC EFs for trucks in Chongqing and Beijing are 1.1 and 0.4 g kg-fuel^{-1} on level roads. The lower BC EFs in Beijing may reflect the better fuel quality used in this region. Moreover, BC EFs of trucks in Beijing are compared with the data reported in our previous work. A clear downward trend is observed from 2008 to 2010, reflecting cumulative effects of improvements in fleet composition that has taken place since 2008. Reduced BC emission from diesel trucks is expected because of better fuel quality and replacement of old trucks in the future.

As a proof-of-concept, we acquired registration information of 11 trucks in Chongqing and investigated how information of registration date and engine model might improve our understanding of emissions vs. control status of trucks. A downward trend of BC EFs is observed from China I to China III trucks, indicating the effectiveness of emission standard enforced in Beijing and nationwide.

However, NOx EFs for trucks from Beijing does not show lower values than trucks from other regions, despite improved fuel and a large proportion of trucks with newer emission standard. NOx EFs of China III trucks did not show lower values than China I and II trucks for the 11 trucks with further engine information. Scatter plot between NOx EFs and BC EFs illustrates that high BC emission trucks are usually not high NOx emitters. These observations suggest that current emission standard implemented in Beijing and nationwide may have limited impacts on NOx emission reduction, likely due to the ineffectiveness of current control technologies in NOx reduction. It is challenge to reduce NOx emission without reducing fuel efficiency by any means other than catalytic controls. These controls require considerable improvements in fuel quality in some regions of China. Therefore, multi-pollutant

control strategies are imperative when new emission standard and regulations are launched in China. Clearly, the inclusion of data regarding truck engine characteristics provides data that is very valuable to the interpretation of emissions data collected in chase studies.

CHAPTER 6

MAJOR CONTRIBUTIONS

Table 6.1 Major contributions and significance of the dissertation

Contributions	Significance
Methodology: On-road chasing method • First application in China. • Developed and modified during campaign: fleet average→ individual trucks. • Presented instructions of EF calculation based on the method.	➢ Time- and cost- efficient. ➢ Expanded database of vehicle emissions in China.
Data: First time reported in China • BC and $PM_{0.5}$ number EF of Chinese vehicles by type. • $PM_{0.5}$ number EF of Chinese buses with various emission standards. • Spatial variance of $PM_{0.5}$ number concentration at on-road, roadside and ambient environments. • CO, BC, $PM_{0.5}$, NOx EF distributions based on large sample size of on-road diesel trucks	➢ Complemented BC and $PM_{0.5}$ number EFs of Chinese vehicles by type
Evaluations: Temporary and permanent air pollution control measures • Diesel truck restriction at daytime. • Traffic activity restriction during Olympics. • Fuel quality improvement and emission standard enforcement in Beijing and China • Introduction of clean energy buses • Heavy emitter removal • NOx reduction program	➢ Provides data support for policy makers to evaluate and modify current work and propose future plans

The primary contributions of this dissertation are summarized in Table 6.1. As the largest developing country with rapid urbanization and vehicle population growth rate, China is now facing significant challenges in environmental sustainable development. Traffic-generated air pollution control measures and their effectiveness are getting more attention during the past decades. My contributions focus on three aspects:

Firstly, it is the first time that on-road chasing method is applied in China to investigate real world vehicle emissions. This method was developed and modified during the four year field campaign. As fast response instruments employed, it is

capable to characterize individual truck emission as well as fleet average emission. Major advantages of on-road chasing method include the flexibility of target vehicle selection and time and cost efficient sampling, which complement the conventional emission test methods such as remote sensing test, chassis dynamometer test and PEMS. Sample size of my studies was usually 10~25 times greater than chassis dynamometer test and PEMS. It provides a way to picture and quantify real world vehicle emissions at a relatively small cost. EF results reported in this work significantly expand the database of diesel vehicle emissions in China.

In field experiments and data manipulating, several key factors are essential to EF calculation: a) fast response instruments are needed to capture signal variance of all pollutants; b) baseline period prior to or post chasing period is recommended for each individual vehicle; c) distance between target vehicle and mobile platform will not affect EF calculation, however, proper distance is necessary to capture significant signal but avoid potential dangers; d) vehicles around target vehicle will affect signal identification and need to be avoided; e) on-road video records synchronized with instruments are strongly recommended for post-experiment data analysis.

The major limitation of this method is the connection between vehicle identification information, driving condition and emissions. As a proof-of-concept study, the registration information of 11 trucks sampled in Chongqing was acquired from governmental sources based on their license plate numbers. However, the reader is cautioned that this small sample does not represent the overall group of vehicles tested or the larger fleet. Further evaluations of the larger data set would prove valuable to reinforce the tentative observations. Part of our EF results, i.e. EF of buses,

is linked with emission standard and shows good agreement, because those information are labeled on each bus and recorded during the chasing experiment. Therefore, I suggest that the next stage research would focus on obtaining vehicle identification information and link it with emission data. Vehicle emission standard labels are recommended to be stuck on each individual vehicle when they are conducting annual inspections.

Secondly, a number of the data in this work are reported for the first time in China. For example, on-road BC and $PM_{0.5}$ number EFs for Chinese vehicles by type, EF of Chinese buses with various emission standards, and the spatial variance of $PM_{0.5}$ number concentration at on-road, roadside and ambient environments. Moreover, CO, BC, $PM_{0.5}$ and NOx EF distributions generated from large sample size of on-road diesel trucks are presented for the first time in China.

The third contribution focuses on the evaluation of traffic related air pollution control measures. Data we collected covered various periods when different air pollution control measures were implemented in Beijing and China, which allowed us to evaluate those temporary and permanent measures. For example, my work demonstrates the effectiveness of traffic activity restrictions by observing a strong traffic impact on CO, BC and $PM_{0.5}$ concentrations. The BC diurnal pattern associated with truck activities helped us determine that diesel trucks are a major source of summertime BC in Beijing. It also provided evidence showing the effectiveness of truck activity restrictions in BC reduction during Olympic days and the daytime period of non-Olympic days. Significant reduction in EFs of LDV and ambient BC concentrations in 2008 summertime indicates the effectiveness of stringent traffic

control measures implemented during the 29th Olympic Games. A clear downward trend of BC EF of diesel trucks in Beijing area from 2008 to 2010, and of buses with more tighten emission standard indicates positive long-term impacts of air quality improvement measures maintained after Olympics, such as the improvement of fuel quality in Beijing and enforcement of tighten emission standard nationwide. I identified "Heavy emitter" and quantified its significant contribution to entire fleet emission. This finding indicates the effectiveness of heavy emitter removal measures such as the ban of YLV in Beijing.

However, the comparison work between cities reveals that NOx EFs of trucks from Beijing does not show lower values than trucks from other regions, despite improved fuel and a large proportion of trucks with newer emission standard. Moreover, scatter plot between NOx EFs and BC EFs illustrates that high BC emission trucks are usually not high NOx emitters. These observations suggest that current emission standard implemented in Beijing and nationwide may have limited impacts on NOx emission reduction, likely due to the ineffectiveness of current control technologies in NOx reduction.

Evaluation results provide support for following traffic related air pollution control measures: a) Continuing emission reduction programs targeting at "heavy emitter" removal; b) improving fuel quality and enforcing advanced emission standards in the entire regions rather than limited to particular cities; c) Continuing introducing clean-burning buses; d) ensuring the implementation of multi-pollutant control strategies when new emission standard and regulations are launched in China. The results and conclusions in this dissertation could provide data support for policy

makers to evaluate and modify current emission control strategies and propose future ones.

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