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The challenge to NO_x emission control for heavy-duty diesel vehicles in China

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Abstract. China's new "Twelfth Five-Year Plan" set a target for total NO_x emission reduction of 10 % for the period of 2011–2015. Heavy-duty diesel vehicles (HDDVs) have been considered a major contributor to NO_x emissions in China. Beijing initiated a comprehensive vehicle test program in 2008. This program included a sub-task for measuring on-road emission profiles of hundreds of HDDVs using portable emission measurement systems (PEMS). The major finding is that neither the on-road distance-specific ($g\,km^{-1}$) nor brake-specific ($g\,kWh^{-1}$) NO_x emission factors for diesel buses and heavy-duty diesel trucks improved in most cases as emission standards became more stringent. For example, the average NO_x emission factors for Euro II, Euro III and Euro IV buses are $11.3 \pm 3.3\,g\,km^{-1}$, $12.5 \pm 1.3\,g\,km^{-1}$, and $11.8 \pm 2.0\,g\,km^{-1}$, respectively. No statistically significant difference in NO_x emission factors was observed between Euro II and III buses. Even for Euro IV buses equipped with SCR systems, the NO_x emission factors are similar to Euro III buses. The data regarding real-time engine performance of Euro IV buses suggest the engine certification cycles did not reflect their real-world operating conditions. These new on-road test results indicate that previous estimates of total NO_x emissions for HDDV fleet may be significantly underestimated. The new estimate in total NO_x emissions for the Beijing HDDV fleet in 2009 is 37.0 Gg, an increase of 45 % compared to the previous study. Further, we estimate that the total NO_x emissions for the national HDDV fleet in 2009 are approximately 4.0 Tg, higher by 1.0 Tg (equivalent to 18 % of total NO_x emissions for vehicle fleet in 2009) than that estimated in the official report. This would also result in 4 % increase in estimation of national anthropogenic NO_x emissions. More effective control measures (such as promotion of CNG buses and a new in-use compliance testing program) are urged to secure the goal of total NO_x mitigation for the HDDV fleet in the future.

1 Introduction

China's energy consumption is heavily dependent on coal-based energy (NBSC, 2011), resulting in severe air pollution during the 1990s and early 2000s. Consequently the Chinese government set a clear goal of a 10 % reduction of total SO_2 emissions (SO_2 is considered as a key indicator for coal-based pollution) between 2006 and 2010, the period of the "Eleventh Five-Year Plan" (11th FYP) (Zhao et al., 2009; Zhang et al., 2012). After an aggressive build up of flue gas desulfurization at coal-fired power plants, and phase-out of small, inefficient boilers in the coal-fired power sector, total SO_2 emissions decreased by 14 % during the 11th FYP (MEP, 2011a).

However, a surge of the vehicle population in China since the late 1990s, due in part to rapid economic growth and urbanization, triggers a concern about air pollution resulting

Table 1. Summary of emission standards for new HDDVs adopted in Beijing and China.

Vehicle type	Emission standards	Cut points, $g\,kW\,h^{-1}$				Date of standard implementation (type approval)	
		CO	HC	NO_x	PM	Beijing	National
HDDVs, >85 kW	Pre- Euro I	Only smoke tested at full load, filter smoke number < 4.0					
	Euro I (ECE R-49)	4.5	1.1	8.0	0.36	2000-1-1	2000-9-1
	Euro II (ECE R-49)	4.0	1.1	7.0	0.15	2003-1-1	2003-9-1
	Euro III (ESC)	2.1	0.66	5.0	0.10	2005-12-30	2007-7-1
	Euro IV (ESC)	1.5	0.46	3.5	0.02	2008-7-1[a]	2013[b]
	Euro V (ESC)					2012-10-1[c]	NA[d]

[a] Only for public fleets including public buses, postal and sanitation vehicles; [b] Euro IV emission standard for HDDVs was first planned to be implemented in 2010. However, it was postponed by the MEP due to several factors. For example, a sufficient supply of low sulfur fuel that complied with the Euro IV standard was unavailable. The most recent update is that the national Euro IV emission standard for HDDVs might be able to be implemented in 2013; [c] Euro V emission standard for public HDDV fleets in Beijing including public buses, postal and sanitation vehicles will probably be implemented, beginning from October 1^{st}, 2012. Prior to this, ultra low sulfur diesel that complied with Euro V standard will be fully delivered in the Beijing market from June 1^{st}, 2012; [d] not available.

from vehicle emissions. Vehicle sales jumped 32 % to 18.1 million in 2010 compared to that of 2009, topping all previous records and securing China's position as the world's largest vehicle market (CATARC and CAAM, 2011). On-road vehicles are considered one of the most important sources of air pollution in China's major cities, changing the pollution pattern from coal-based air pollution to a mix of coal- and vehicle-based air pollution in large cities (Du et al., 2012; Hao et al., 2001; Li et al., 2011a; Wang et al., 2010; Yang et al., 2011). Among the criteria air pollutants, NO_x is considered a key indicator for both vehicle and coal combustion sources. Over the past decade, observations from space and in situ indicated a worsening situation for NO_x pollution in many regions of China (Richter et al., 2005). Chan and Yao (2008) revealed that annual average NO_2 concentrations had remained almost constant from around 2000 to 2005 and still exceeded the Grade-II standard in China's mega cities such as Beijing, Shanghai and Guangzhou. They also emphasized the necessity of vehicle emission control in the future due to major contribution from on-road vehicles to ground NO_x emissions. In response to the NO_x trend, the new 12th FYP establishes a target for total NO_x emission reduction of 10 % for the period of 2011–2015 (NPC, 2011). However, average NO_2 concentration of China's 113 key environmental protection cities increased by 5.7 % during the first half of 2011 compared to same period of 2010 (MEP, 2011b), which indicated that NO_x control would be much harder than SO_2 control. Furthermore, on February 29th, 2012, China's State Council approved the National Ambient Air Quality Standards (NAAQs) Amendment, which will be fully in effect in 2016. It adds annual and 24-h average $PM_{2.5}$ concentration limits, and 8-h average O_3 concentration limit. Meanwhile, the limits of PM_{10} and NO_2 are tightened (MEP and AQSIQ, 2012). All the revisions of NAAQs call for more stringent controls of NO_x emissions, especially in high-emitting sectors that include on-road vehicles (Wang and Hao, 2012; Zhang et al., 2012).

Among the different vehicle categories, attention has been focused on heavy-duty diesel vehicles (HDDVs, including buses and trucks) due to their significantly higher NO_x emission factors and higher travel mileage relative to gasoline cars (Imhof et al., 2006; Liu et al., 2009; Thornhill et al., 2010). The Ministry of Environmental Protection (MEP) reported the first-ever total national NO_x emissions from on-road vehicles of 5.3 Tg in 2009, of which diesel vehicles (dominated by heavy-duty trucks and buses) contributed 60 % (MEP, 2010). Even in Beijing, with a large number of private gasoline cars, HDDVs emitted 25.0 Gg NO_x, accounting for 30 % of total NO_x emissions from vehicles based on our previous study (Wu et al., 2011). To mitigate emissions from HDDVs, MEP and local environmental protection bureaus (EPBs) have launched a series of control measures since 2000 (ICCT, 2010; Wu et al., 2011). For example, Table 1 illustrates new emission standards implemented by stage for HDDVs since 2000 in Beijing and at the national level. Accordingly, new advanced control technologies have penetrated into the HDDV fleet. For example, Selective Catalytic Reduction (SCR) technology has been required for new buses in Beijing since 2008 to comply with the Euro IV standard.

Previous studies of HDDV emissions in China relied heavily on results of engine tests on chassis dynamometers with typical driving cycles (e.g., US Environmental Protection Agency [EPA]'s Federal Test Procedure [FTP] cycle) (Wu et al., 2011; Zhou et al., 2010). However, such engine tests have often been criticized as an inaccurate representation of real-world on-road operation (Krishnamurthy et al., 2007; US DOJ and US EPA, 1998). In 2005, the US EPA released a final rule on a manufacturer-run, in-use testing program for HDDVs using Portable Emission Measurement Systems (PEMS) (US EPA, 2005). The in-use testing

Table 2. Summary of test HDDVs.

Vehicle categories	Emission standard	Sample numbers	Vehicle length (m)	GVW (ton)	Engine displacement (L)
Bus	Euro IV	24	11 ∼ 12	15 ∼ 18	6.5 ∼ 8.4
	Euro III	22			
	Euro II	9			
HDDT1 3.5≤GVW<4.5 tons	Euro III	4		3.8 ∼ 4.4	2.8 ∼ 3.2
	Euro II	9		3.5 ∼ 4.3	2.7 ∼ 3.2
	Euro I	8		3.5 ∼ 4.2	2.3 ∼ 3.2
HDDT2 4.5≤GVW<12.0 tons	Euro III	5		5.1 ∼ 11.0	3.2 ∼ 6.7
	Euro II	6		4.8 ∼ 10.9	3.8 ∼ 6.5
	Euro I	3		4.5 ∼ 11.4	3.3 ∼ 6.5
HDDT3 GVW ≥ 12.0 tons	Euro III	28		12.0 ∼ 35.8	4.7 ∼ 12.0
	Euro II	6		14.7 ∼ 20.2	4.7 ∼ 9.8
	Euro I	11		13.8 ∼ 31.6	4.3 ∼ 10.0

requires that engines meet the Not-to-Exceed limits by using PEMS during real-world operation to compose a complementary test suite with the FTP cycle and steady-state Supplemental Emission Test (Krishnamurthy et al., 2007; US EPA, 2005a). Europe has also begun to set its own in-use compliance regulations to supplement certification testing for HDDVs. The in-use compliance regulations would implement the Averaging Window Method by using PEMS on-road (JRC, 2007). In order to improve the accuracy of emission factors and inventories, several on-road emission test programs for HDDVs using PEMS have been conducted in China recently (Liu et al., 2009; Liu et al., 2011). The Vehicle Emission Control Center of MEP has been collecting on-road emission test results of the HDDV fleet in several Chinese cities, attempting to develop a national mobile sources emission inventory for China (Tang et al., 2011). Beijing, a pioneer in controlling vehicle emissions within China similar to California's role in the U.S., also launched a comprehensive vehicle emission test program beginning in 2008. This program included a sub-task for measuring second-by-second on-road emission profiles using PEMS and developing a new emission database for the HDDV fleet in Beijing.

This study represents the real-world on-road NO_x emission profiles of more than 130 HDDVs (including both buses and trucks) within the Beijing test program. We calculate and evaluate the NO_x emission factors of HDDVs in detail for various emission standards and vehicle weight classifications under typical on-road driving cycles. We also explore detailed on-board diagnostic (OBD) data regarding real-time operating conditions of engines, discuss on-road brake-specific NO_x emission factors, and evaluate the real-world SCR performance for the new Euro IV buses in Beijing. We then compare and discuss the new NO_x emission factors of HDDVs with other previous estimates and suggest an update of the emission inventory for the HDDV fleet in Beijing and China.

2 Methodology

2.1 Test vehicles and experimental procedures

The on-road emission tests were conducted in 2008–2010 in Beijing. A total of 135 HDDV test results were collected in this study, including 55 buses and 80 trucks. These tested vehicles are summarized in Table 2. They were mainly recruited from bus and freight transportation companies which represent the typical vehicle technology and maintenance conditions in Beijing.

According to the scrappage policy for the bus fleet in Beijing, public buses over 8 yr are to be phased out. Based on registration data from the bus company, Euro I and pre-Euro I diesel buses have been almost completely phased out. Therefore, our diesel bus samples are classified into 3 emission standard categories – Euro II, Euro III and Euro IV. Almost all of the tested buses have a vehicle length of 11–12 m and gross vehicle weight (GVW) of 15–18 tons. As for trucks, they are classified into 3 emission standard categories as Euro I, Euro II and Euro III. There have been delays in implementing Euro IV emission standards for heavy-duty diesel trucks (HDDTs) nationwide (see Table 1) and there is no formal scrappage policy for trucks. The weight of tested trucks varied considerably, which generally has significant influence on vehicle emissions. Thus, the tested trucks are further classified into 3 weight classifications (see Table 2).

Two Sensor Inc. SEMTECH-D PEMSs (Liu et al., 2009) and a Horiba OBS-2200 PEMS (Bougher et al., 2010) were used in the test program to measure the on-road emissions of gaseous pollutants. The SEMTECH-D employs a non-dispersive ultraviolet analyzer to measure NO and NO_2 separately, a flame ionization detector to measure total hydrocarbons (THC), and a non-dispersive infrared analyzer to measure CO and CO_2. The SEMTECH exhaust flow meter (SEMTECH-EFM) was used to measure the exhaust flow. The OBS-2200 PEMS employs a heated chemiluminescence detector to measure NO_x (NO and NO_2 together), a heated flame ionization detector to measure THC, and a heated non-dispersive infrared analyzer to measure CO and CO_2. The flow measurement is conducted with a factory calibrated pitot tube exhaust flow meter in the a tailpipe attachment unit. The SEMTECH-D PEMS and OBS-2200 PEMS have both passed validation testing with laboratory systems for in-use compliance testing and were demonstrated to be both accurate and precise, meeting the Section 40 Code of Federal Regulations standards, Part 1065 (BMSTC and Beijing EPB, 2010; Durbin et al., 2007; US EPA, 2005b). Besides, a Dekati Mass Monitor (DMM) (Liu et al., 2009) and two Electric Low Pressure Impactors (ELPI) (Liu et al., 2011) were used in the test to measure the on-road particle emissions.They are both manufactured by Dekati Co. Ltd. and based on ELPI^{TM} technology, which is combined the particle charging and electrical detection. They are able to provide real-time second-by-second PM mass and number emission profiles. Vehicle speed, altitude, latitude, and longitude were recorded second-by-second by GPS receivers.

The test cycles in Beijing included both urban and highway driving segments. Each test lasted more than 30 minutes so as to collect enough vehicle operation and emission profile data. A total of more than 323 000 second-by-second data for trucks and more than 163 000 second-by-second data for buses were collected. We have also recorded the driving condition data for more than 1 020 000 seconds with GPS receivers on 18 typical bus routes to develop a typical driving cycle for urban buses in Beijing. As heavy-duty trucks are not allowed within the Fourth-Ring Road (downtown in Beijing) during the daytime, the driving cycles for HDDTs are typically different than those for buses. For detailed descriptions of test routes and driving cycle development please refer to BMSTC and Beijing EPB (2010).

2.2 Operating mode binning methodology

Vehicle specific power (VSP), defined as the instantaneous power demand by the engine per unit vehicle mass, was first introduced by Jimenez (1999). VSP can represent driving conditions in an integrated way and show better correlation with pollutant emission rates than other operating parameters. As a result, VSP is now widely used in developing new emission factor models, such as the International Vehicle Emissions Model (IVE) developed by the University of California, Riverside (CE-CERT et al., 2008) and the Motor Vehicle Emission Simulator (MOVES) developed by US EPA (US EPA, 2009). In this study, we applied the equation from the MOVES model (Koupal et al., 2005) for VSP calculation of all HDDV samples, which is shown below.

Table 3. Summary of road-load coefficient values for calculating VSP of each major HDDV category.

Coefficient	HDDT1	HDDT2	HDDT3	Bus
A/m, $kW\,s\,m^{-1}\,t^{-1}$	0.0996	0.0875	0.0875	0.0643
B/m, $kW\,s^2\,m^{-2}\,t^{-1}$	0	0	0	0
C/m, $kW\,s^3\,m^{-3}\,t^{-1}$	0.000542	0.000356	0.000331	0.000279

Note: With reference to the MOVE model, those coefficients are estimated according to the typical vehicle weight (Koupal et al., 2005).

$$\mathrm{VSP} = \frac{A}{m}\cdot v + \frac{B}{m}\cdot v^2 + \frac{C}{m}\cdot v^3 + av + gv\sin\theta \tag{1}$$

where m is vehicle weight, tons; v is instantaneous vehicle speed, $m\,s^{-1}$; a is instantaneous vehicle acceleration, $m\,s^{-2}$; θ is road grade, radians; A is the rolling resistance coefficient, $kW\,s\,m^{-1}$; B is the rotational resistance coefficient, $kW\,s^2\,m^{-2}$; C is the aerodynamic drag coefficient, $kW\,s^3\,m^{-3}$. These road-load coefficients (i.e., A, B, and C) by each major HDDV category used in this study are summarized in Table 3.

We further developed operating mode bins defined by VSP and vehicle speed with reference to the operating mode binning methodology applied in MOVES (US EPA, 2009). These bins were slightly modified due to the availability of test data in each bin, specifically in the high-speed segments. Finally, we established 22 operating modes as described in Table 4, including a deceleration or braking bin, an idling bin, and 20 bins to represent cruise or acceleration driving modes.

2.3 Emission rates and emission factors

The second-by-second emission rates (in $g\,s^{-1}$) for each individual HDDV were collected. Equation 2 summaries the way we calculated the average emission rate for each group by emission standard, GVW and operating mode bin (US EPA, 2009).

$$\overline{\mathrm{ER}}_{i,j,k} = \frac{1}{N_i}\sum\nolimits_1^{N_i}\left(\frac{1}{T_k}\sum\nolimits_1^{T_k}\mathrm{ER}_j\right) \tag{2}$$

where T_k is the number of second-by-second data for each vehicle in operating mode bin k; N_i is the total number of vehicles in vehicle category i; ER_j is the instantaneous emission rate of pollutant j, $g\,s^{-1}$; and $\overline{\mathrm{ER}}_{i,j,k}$ is the average emission rate of pollutant j for vehicle category i and operating mode bin k, $g\,s^{-1}$.

The distance-specific emission factor in $g\,km^{-1}$ was further developed based on the average emission rates and time distribution of operating mode bins within a specific driving cycle, as shown in Eq. (3).

$$\mathrm{EF}_{i,j} = \frac{3600\sum_k\left(\overline{\mathrm{ER}}_{i,j,k}\cdot P_k\cdot T\right)}{L} \tag{3}$$

Table 4. Definition of operating mode bins by using VSP and vehicle speed (v).

VSP ($kW\,t^{-1}$)		Vehicle speed ($km\,h^{-1}$)			
		$v < 1.6$	$1.6 \le v < 40$	$40 \le v < 80$	$v > 80$
VSP< -4	Bin 0 Deceleration or braking	Bin 1 Idle	Bin 11	Bin 21	Bin 35*
$-4 \le$ VSP< -2			Bin 12	Bin 22	
$-2 \le$ VSP< 0			Bin 13	Bin 23	
$0 \le$ VSP< 2			Bin 14	Bin 24	
$2 \le$ VSP< 4			Bin 15	Bin 25	
$4 \le$ VSP< 6			Bin 16	Bin 26	Bin 36
$6 \le$ VSP< 8			Bin 17	Bin 27	Bin 37
VSP≥ 8			Bin 18	Bin 28	Bin 38

* Bin 35 indicates the operating modes with speed higher than 80 $km\,h^{-1}$ and VSP lower than 4 $kW\,t^{-1}$. Time percentages of high-speed mode bins are significantly lower than low-speed and middle-speed mode bins. According to the operating mode bins developed in MOVES 2004 (Koupal et al., 2005), those original five mode bins with VSP lower than 4 $kW\,t^{-1}$ were ultimately merged into the Bin 35.

where $EF_{i,j}$ is the emission factor for vehicle category i of pollutant j, $g\,km^{-1}$; $\overline{ER}_{i,j,k}$ is the average emission rate of pollutant j for vehicle category i and operating mode bin k, $g\,s^{-1}$; P_k is the time percentage of operating mode bin k to the total driving cycle; T is the total time of the driving cycle, s; and L is the total distance of the cycle, km.

For comparison with the emission limits (in $g\,kW\,h^{-1}$), we convert the distance-specific emission factors into brake-specific emission factors using on-road CO_2 and NO_x emission data. First, we estimated the fuel mass based NO_x emission factors ($g\,kg\,fuel^{-1}$) with Eq. (4) (Liu et al., 2009).

$$EF_{NO_x-\mathrm{fuel}} = \frac{EF_{NO_x} - \mathrm{distance} \cdot W_{C\mathrm{diesel}}}{0.273E_{CO_2-\mathrm{distance}} + 0.429EF_{CO-\mathrm{distance}} + 0.866EF_{THC-\mathrm{distance}}} \quad (4)$$

where $W_{C\mathrm{diesel}}$ is the carbon mass in diesel fuel, 870 g carbon per kg diesel.

We then applied the brake-specific fuel consumption (BSFC) value in converting fuel consumption-based emission factors ($g\,kg\,fuel^{-1}$) into brake-specific emission factors ($g\,kWh^{-1}$) with Eq. (5). The BSFC values for buses and trucks by weight are estimated based on the method applied by the MOBILE6 model (US EPA, 2002).

$$EF_{NO_x-\mathrm{brakespecific}} = \frac{0.454}{0.746} \cdot EF_{NO_x-\mathrm{fuel}} \cdot \mathrm{BSFC} \quad (5)$$

3 Results and discussion

3.1 NO_x emission profiles for buses

Figure 1 presents the average emission rates of diesel buses for each operating mode bin from Euro II to Euro IV. Generally, the emission rates increase with VSP, similar to the trend in other studies (Liu et al., 2009; US EPA, 2009). However, we did not observe a clear decrease in emission rates from Euro II to Euro IV, although NO_x emission limits were

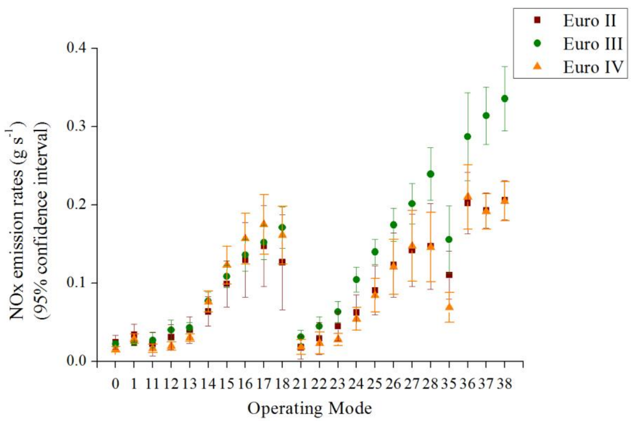

Fig. 1. Summary of average NO_x emission rates for Euro II, Euro III and Euro IV buses by operating mode.

tightened. Using one-way ANOVA method, there are no statistically significant differences in NO_x emission rates between Euro II, Euro III and Euro IV buses for operating mode bins 0 to 18. For buses in urban driving conditions, operating mode bins 0 to 18 usually cover most of the driving time (see more detailed discussion later). Furthermore, the average NO_x emission rates for Euro III buses for those operating modes in the median-speed segment with VSP higher than 0 $kW\,ton^{-1}$ (e.g. operating mode bins 24–28) are all statistically significantly higher than those for Euro II buses for the corresponding operating modes. On average, such increments in NO_x emission rates for operating mode bins 24 to 28 are 42–67 %. It indicates Euro III buses would have higher on-road NO_x emission levels on-average than Euro II buses under relatively high speed and high power demand conditions.

To eliminate the distinction of operating conditions between each individual test, all the emission factors were corrected to a representative driving cycle. This typical driving

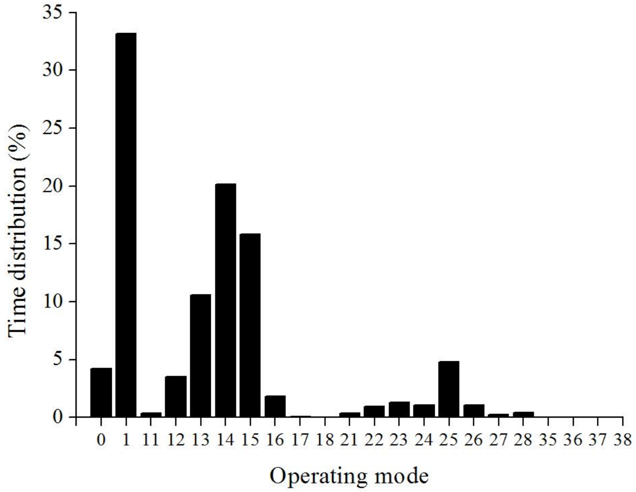

Fig. 2. Allocation of time of each operating mode bin to total time of the typical driving cycle for urban buses in Beijing, with an average speed of 18 km h^{-1}.

Table 5. Summary of information of two tested Euro IV buses with data regarding with real-time engine conditions recorded by OBD system.

Test number		Bus 1	Bus 2
Odometer (km)		116 031	97 937
Vehicle length (m)		12	12
Vehicle weight (t)		17.8	17.8
Load mass (t)		1.1*	0.9*
Engine power rating (kW)		165	165
Average speed (km h^{-1})		15.0	12.0
Average exhaust temperature (°C)		174	153
Brake-specific emission factor (g kWh^{-1})	CO	2.50	1.47
	THC	0.11	0.14
	NO_x	9.51	6.63
Distance-specific emission factor (g km^{-1})	CO	3.94	2.65
	THC	0.18	0.26
	NO_x	15.0	12.0

* The load mass was estimated based on the mass of the PEMS and the number of people on board during the test. Such load mass data are equivalent to 13 to 16 passengers. It could represent a typical passenger load during non-rush hours in Beijing.

cycle was developed based on more than 1 million second-by-second driving condition data on 18 typical bus routes in Beijing. Figure 2 illustrates the time allocation of each operating mode bin to total time of this typical driving cycle. Due to frequent stops and congestion for the bus fleet in urban areas, the idling time share is as high as 33 % and low-speed zones with operating mode bins 11 to 18 contribute more than 50 % of total time in such a driving cycle. There are no data in the high speed segment, because the maximum speed is lower than 80 km h^{-1}. Not surprisingly, the average speed of this typical bus driving cycle is as low as 18 km h^{-1}.

Figure 3 presents the distance-specific emission factors of NO_x as well as other three air pollutants (i.e., CO, THC and $PM_{2.5}$) under this typical bus driving cycle for each emission standard category. For comparison, the diesel bus emission factors from MOBILE-China model (Wu et al., 2011) and COPERT4 v9.0 model (Gkatzoflias et al., 2011) are also provided in the figure. The NO_x emission limits of certification testing for heavy-duty diesel engines have been significantly tightened. As shown in Table 1, the NO_x emission limits for Euro III and Euro IV HDDVs are reduced by 29 % and 50 %, respectively, relative to Euro II HDDVs. However, the on-road NO_x emission factors for diesel buses in this study did not improve as expected. The average NO_x emission factors (with 95 % confidence interval, CI) for Euro II, Euro III and Euro IV urban buses are 11.3 ± 3.3 g km^{-1}, 12.5 ± 1.3 g km^{-1}, and 11.8 ± 2.0 g km^{-1}, respectively. With one-way ANOVA method, no statistically significant difference in NO_x emission factors has been observed between Euro II and III buses. Even Euro IV buses equipped with an SCR system specifically to control NO_x did not have significantly lower NO_x emission factors than Euro III buses. The results are clearly different from the estimates calculated by the MOBILE-China model that were used in our previous study (see Fig. 3c) (Wu et al., 2011). For example, with MOBILE-China we estimate a significant decrease in NO_x emission factors (54 %) from Euro II to Euro III buses, and a significant decrease of 50 % from Euro III to Euro IV buses. Such estimates using a continuous decrease in NO_x emission factors as emission standards tightened for HDDV have been used widely in previous policy evaluations (Wu et al., 2011; Zhou et al., 2010). Thus, NO_x emissions for China's diesel bus fleet may be significantly underestimated based on these new NO_x results.

The trend for other gaseous pollutants, such as CO and THC, is different. As shown in Fig. 3a and b, both THC and CO emission factors of diesel buses decrease as emission standards tightened, especially for Euro IV buses. On-road emission factors of CO and THC for Euro IV buses decreased by 54 % and 70 %, respectively, compared to Euro III buses. In addition, the results of $PM_{2.5}$ measurements in this test program indicate that $PM_{2.5}$ emissions of buses improved as well, as shown in Fig. 3d. For buses, average emission factors of $PM_{2.5}$ for Euro IV are reduced by 59 %, compared to Euro III buses. These trends are likely a result of engine modifications. More discussion on this issue is in the next sub-section.

To further evaluate the SCR performance, two more Euro IV diesel buses with an OBD system installed were selected. Using the OBD systems, we obtained detailed data regarding real-time engine profiles, shown in Table 5. Their exhaust temperature was measured as the SCR system would not activate when exhaust temperature is too low (Liu et al., 2011). The detailed information and test results of these two buses are listed in Table 5. For Bus 1, its exhaust temperature was lower than 170 °C during 29 % of its operating time, and Bus 2 operated at low temperatures as much as 68 % of operating time. This is a major reason why the SCR of these two

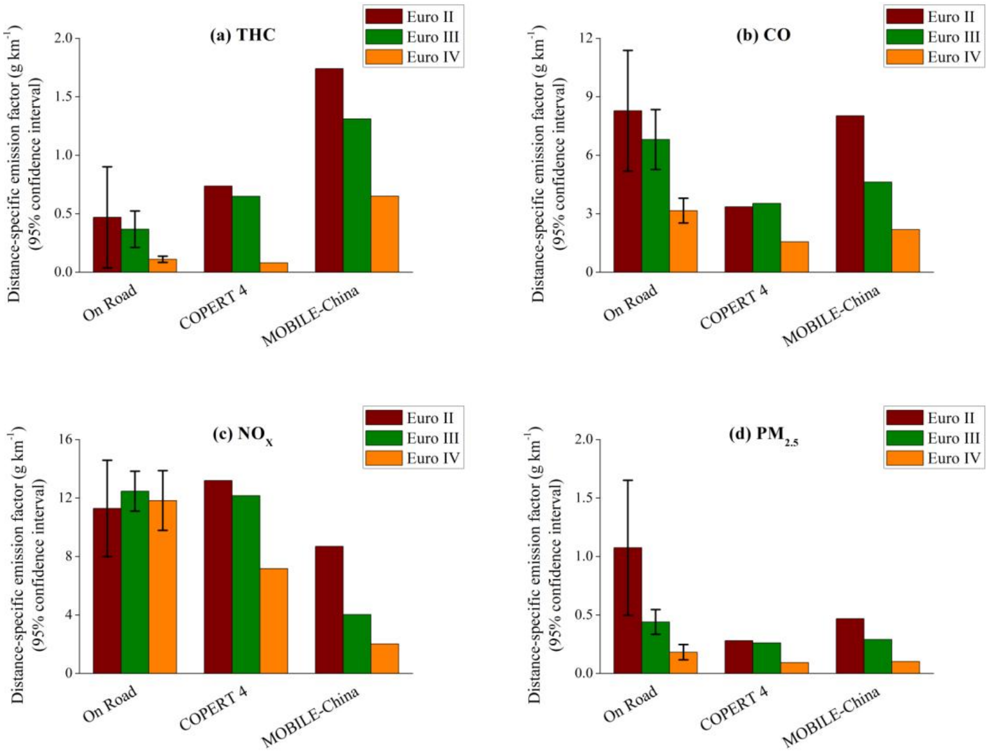

Fig. 3. Emission factors of **(a)** THC, **(b)** CO, **(c)** NO_x and **(d)** $PM_{2.5}$ for Euro II, Euro III and Euro IV diesel buses under a typical urban driving cycle with an average speed of $18\,km\,h^{-1}$.

Table 6. Summary of on-road PEMS test results for CNG EEV buses.

	Engine power (kW)	Odometer (1000 km)	Length (m)	GVW (t)	Emission factor ($g\,km^{-1}$)[a]		
					THC	CO	NO_x
Bus 1	147	49	12	18.0	0.47	3.97	3.18
Bus 2	147	41	12	18.0	1.43	30.27	3.06
Bus 3	147	46	12	18.0	1.02	10.72	5.49
Bus 4	147	58	12	18.0	2.18	12.63	10.72
Bus 5	147	102	12	18.0	NA[b]	1.93	6.20
Bus 6	147	125	12	18.0	NA	7.53	5.98

[a] Those distance-specific emission factors in this table are normalized to the typical driving cycle (see Fig. 2);
[b] Not available due to malfunction of the flame ionization detector for SEMTECH-D PEMS.

buses did not perform well. As a result, their NO_x emission factors are both high (15.0 and $12.0\,g\,km^{-1}$). The low exhaust temperature might be a frequent symptom, especially when the average speed and load mass of a bus both are low. Therefore, a challenge remains for manufacturers to ensure satisfactory performance of SCR systems for bus fleets under real-world conditions, especially when speed and load mass are low. Similar results were found in Europe. Several on-road PEMS and remote sensing studies for the European fleet also indicated unsatisfactory performance of SCR systems for Euro IV and even Euro V HDDVs in low-speed urban driving circumstances (Carslaw et al., 2011; Verbeek et al., 2010; Velders et al., 2011). As the Euro IV emission standard has not been implemented in the HDDT fleet in China, on-road emission data of Euro IV trucks equipped with SCR systems are unavailable. It might be reasonable to believe SCR systems for trucks could perform better than those for buses because the average load and driving speed of trucks are usually significantly higher than buses (Carslaw et al., 2011; Verbeek et al., 2010; Velders et al., 2011).

In addition, directly emitted NO_2 (i.e., primary NO_2) is of more concern due to its greater impacts on local air quality and public health. In particular, several studies have suggested that oxidation catalysts such as the diesel oxidation catalyst (DOC) and the diesel particulate filter (DPF) might increase primary NO_2 emissions significantly (Grice et al., 2009; Hu et al., 2012; Kousoulidous et al., 2008). In China, most heavy-duty diesel trucks and buses that will comply with the Euro IV or V standard in the near future will apply SCR but not DPF since DPF is more sensitive to sulfur content in diesel fuel. In this study, we collected on-road NO and NO_2 emission test profiles by using SEMTECH-D PEMS for four Euro IV diesel buses with SCR and eight Euro III diesel buses without SCR for comparison. The average fraction of primary NO_2 to total NO_x (by volume, with 95 % CI) for those eight Euro III buses is 3.2 ± 1.5 %. For comparison, such a ratio for those four tested Euro IV buses equipped with SCR system is as low as 1.0 ± 1.3 %. The results suggest that those HDDVs equipped with SCR might have very low primary NO_2 emission factors under the real-world conditions, which is consistent with the results from Europe (Kousoulidous et al., 2008; Ntziachristos et al., 2009; Velders et al., 2011). They reported that HDDV equipped with SCR would reduce the fraction of NO_2 to total NO_x effectively compared to other technologies (such as HDDV with DPF).

Beijing EPB planned to release more stringent Euro V emission standards for the public-bus fleet in late 2012 (Beijing EPB, 2012). However, the technology option would still heavily rely on SCR. We envision that those SCR devices installed to comply with Euro V would not operate well for diesel buses during the low-speed stop-and-go driving cycles. Other solution options are necessary for the bus fleet in Beijing as well as other cities in China. Alternative-fuels and advanced vehicle technologies have been introduced in the Beijing bus fleet since 2000. Compressed natural gas (CNG) buses represent such a success story. By 2009, the CNG buses reached over 4,200, accounting for nearly 20 % of bus stock in Beijing. Since 2008, new CNG or liquefied natural gas (LNG) buses in Beijing could even meet the Euro EEV (Enhanced Environmentally-friendly Vehicle) emission standards (Wu et al., 2011). We employed SEMTECH-D PEMS to measure emission factors of gaseous pollutants for 6 CNG EEV buses for comparison. The detailed information of each CNG bus and their emission factors are summarized in Table 6. The average NO_x emission factor (with 95 % CI) is $5.8 \pm 2.6\,\mathrm{g\,km^{-1}}$. A significant decrease of ∼50 % is achieved as compared to Euro IV diesel buses. This indicates that the CNG bus could be a competitive alternative to NO_x control for the bus fleet especially under the low-speed urban driving conditions. Since 2009, hybrid diesel buses and pure electric buses have also been introduced in Beijing. Many other cities in China also plan to substantially expand the bus stock for these so-called "Energy-Saving and New-Energy Vehicles" during the next decade (Wu et al., 2012). However, for NO_x control the results from different studies are quite mixed. NO_x emission factors for tested hybrid diesel buses are approximately within ± 20 % of those for diesel buses (León, 2011; Li et al., 2009, 2011b). At this stage, we cannot confirm a considerable improvement

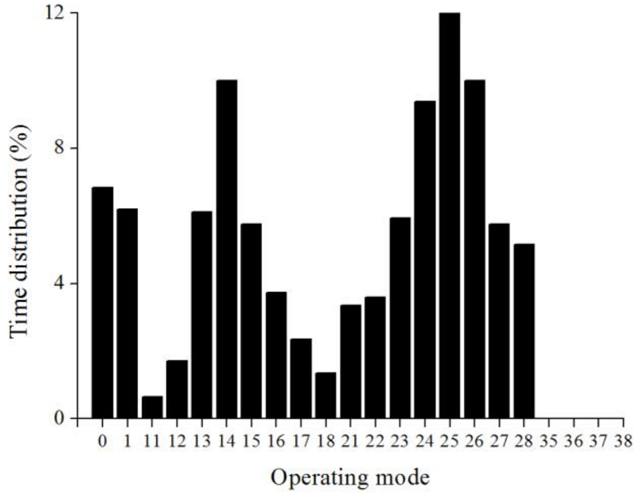

Fig. 4. Allocation of time of each operating mode bin to total time of the typical driving cycle for trucks in Beijing, with an average speed of $40\,\mathrm{km\,h^{-1}}$.

in NO_x emissions for hybrid buses. For pure electric buses, no doubt they have zero NO_x emissions at vehicle operation stage; however, Huo et al. (2010) pointed out a significant increase in that high upstream NO_x emissions might offset the NO_x reduction benefit from the tailpipe using a life-cycle perspective due to the fact that coal power plants dominate in China. It should also be noted that zero emissions at vehicle operation stage have larger positive impact on local air quality than increased same amount of emissions at power plants since the majority of vehicle emissions are concentrated in the urban area. Therefore, the environmental impact of electric vehicle to local/regional air quality needs to be carefully evaluated in the future.

3.2 NO_x emission profiles for trucks

Similar to buses, the emission factors for trucks were normalized to a typical driving cycle to eliminate the distinction of operating conditions between each individual test. Not surprisingly, the driving cycle for trucks is far different than that for buses. Figure 4 presents the allocation of time of each operating mode bin to total time of this typical driving cycle for trucks. The idling time share for trucks is 6 %, much lower than the 33 % for buses. Consequently, the time share of medium-speed zones with operating mode bins 21 to 28 increases significantly to ∼55 %, versus only ∼10 % for buses in the same zones. The average speed of the truck driving cycle is $40\,\mathrm{km\,h^{-1}}$. It should be noted that we were unable to collect data for high-speed zones with speed more than $80\,\mathrm{km\,h^{-1}}$ due to the Beijing city speed limit.

The GVWs for test trucks vary widely from each other, from ∼3.5 tons to as heavy as more than 30 tons. As shown in Fig. 5, generally the NO_x emission factors increase with vehicle weight because the trucks with high vehicle weight are

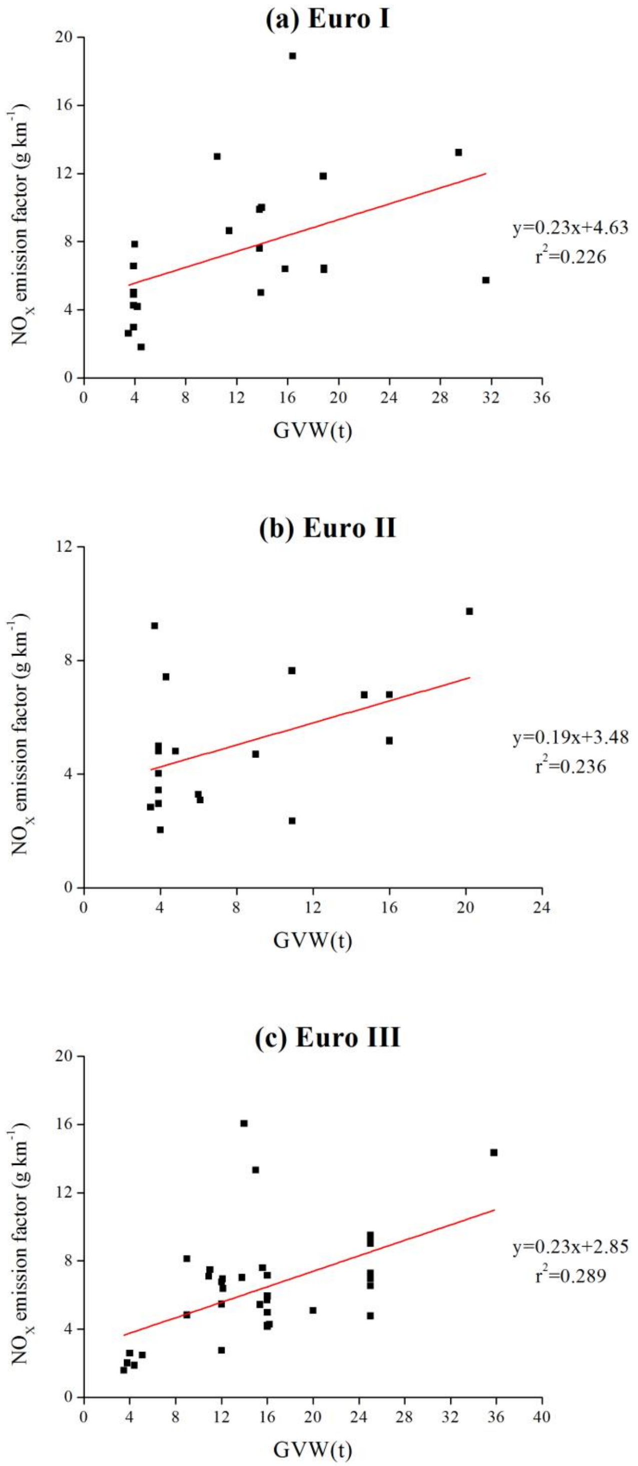

Fig. 5. The scatter graph of the relationship between vehicle weight and emission factor under the typical driving cycle for trucks with an average speed of 40 km h^{-1}, from Euro I to Euro III.

usually equipped with higher powered engines. To minimize the impact of vehicle weight when comparing different emission standards, the test trucks were grouped into three weight classifications as HDDT1, HDDT2 and HDDT3. Within each weight group, the average GVW results of trucks complied with each of the three different emission standards.

Figure 6 shows the on-road distance-specific emission factors of THC, CO, NO_x and $PM_{2.5}$ for the three emission standards and three vehicle weight classifications, together with emission factors estimated by MOBILE-China and COPERT4 v9.0 models. The NO_x emission factors (with 95 % CI) for Euro I, Euro II and Euro III HDDT1 are 4.8 ± 1.5 g km^{-1}, 4.6 ± 1.8 g km^{-1}, 2.0 ± 0.7 g km^{-1}, respectively. For HDDT2 the NO_x emission factors for Euro I, Euro II and Euro III are 7.8 ± 14.0 g km^{-1}, 4.3 ± 2.0 g km^{-1} and 6.0 ± 2.9 g km^{-1}, respectively. For HDDT3, the NO_x emission factors for Euro I, Euro II and Euro III are 9.2 ± 2.8 g km^{-1}, 6.7 ± 1.7 g km^{-1}, 7.2 ± 1.2 g km^{-1}, respectively.

For HDDT2 and HDDT3, the NO_x emission factor trend from Euro I to Euro III is similar. The average NO_x emission factors decrease 45 % (HDDT2) and 27 % (HDDT3) from Euro I to Euro II, then rebound from Euro II to Euro III at an increment of either 39 % for HDDT2 or 6 % for HDDT3. With one-way ANOVA method, there is no statistically significant difference between the Euro II and Euro III HDDT2 and HDDT3 fleet. Again, we did not observe any improvement in NO_x emissions from Euro II to Euro III, similar to the findings for the bus fleet. A similar trend has also been identified by a recent on-road chasing study in Beijing and Chongqing (Wang et al., 2012). However, we did see a clear decrease in THC, CO, and $PM_{2.5}$ for most transitions from Euro II to Euro III (see Fig. 6a, b and d). These test results for HDDT2/HDDT3 fleets indicate that on-road engine performance has been modified significantly. Under real-world driving conditions, the internal combustion performance of these newer engines might be significantly improved. These improvements help reduce $PM_{2.5}$, CO, and THC, but increase NO_x emissions. To comply with the Euro III emission standards, the HDDV fleet in China is usually either equipped with High-pressure Common Rail (HPCR) systems or Exhaust Gas Recirculation (EGR) systems. However, it seems the new control technologies cannot offset the increment of NO_x emissions due to the engine improvement, at least in real-world driving conditions.

It should be noted that HDDT1 is an exception. The difference in NO_x emission factors for Euro II and Euro III is statistically significant. The average NO_x emission factor for Euro III HDDT1 is 57 % lower than for Euro II HDDT1 in this study. A probable reason is the engine improvement in combustion performance for this relatively lighter truck category is not as significant as that for the heavier truck category (e.g., HDDT2 or HDDT3). However, due to limited test data available for Euro III HDDT1 in this study (only four

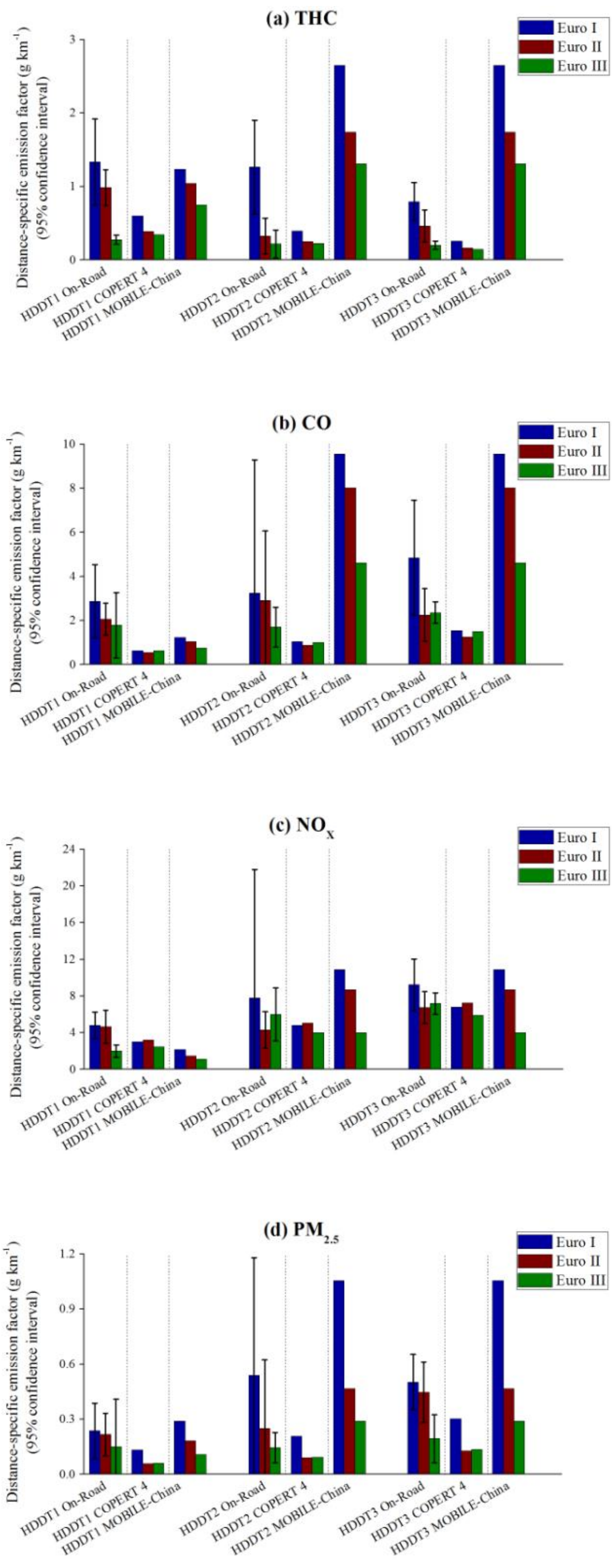

Fig. 6. Emission factors of **(a)** THC, **(b)** CO, **(c)** NO_x and **(d)** $PM_{2.5}$ for Euro I, Euro II and Euro III HDDTs under a typical driving cycle with an average speed of $40\,km\,h^{-1}$.

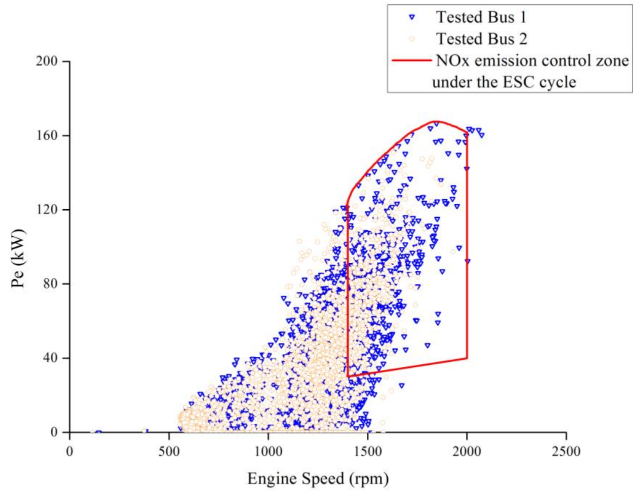

Fig. 7. The map of real-time engine speed and derived engine power (Pe) of two tested Euro IV buses. Inside the red frame box means the NO_x emission control zone under the certificated ESC cycle.

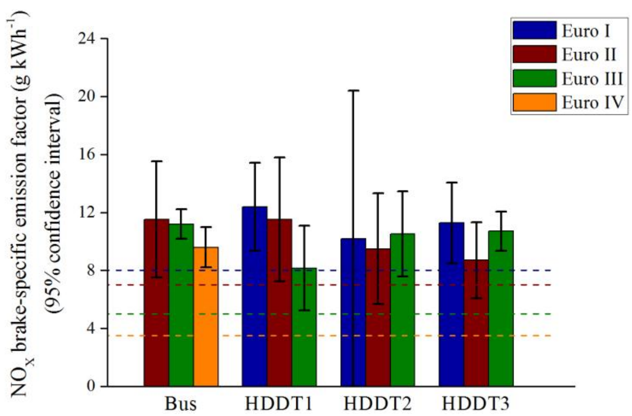

Fig. 8. Estimated brake-specific NO_x emission factors for buses under a typical driving cycle with an average speed of $18\,km\,h^{-1}$ and trucks under a typical driving cycle with an average speed of $40\,km\,h^{-1}$. The dotted lines in the figure, which vary in color, indicate the emission limits of certification testing for HDDVs from Euro I to Euro IV.

samples), more on-road test data are necessary to support this judgment.

As Euro III trucks have penetrated the market of mega cities (e.g., Beijing) since 2006 and nationwide since mid 2007 and Euro IV standards for HDDT were postponed to no sooner than 2013, Euro III HDDT has become the dominant category in the HDDT fleet, especially in big cities. Compared to previous estimates for Euro III HDDT by the MOBILE-China model (Wang et al., 2010; Wu et al., 2011), the new on-road NO_x emission factors for HDDT2 and HDDT3 are usually ∼50–80 % higher. Therefore, similar to the bus fleet, an update in total NO_x emissions for the HDDT fleet would be expected.

3.3 Comparison between emission factors under real-world condition and standard limits

As discussed above, the real-world operating conditions of HDDVs vary significantly. However, our certification standard test cycle is fixed. With OBD data from the two Euro IV diesel buses, we constructed a graph of real-time engine speed and engine power, as shown in Fig. 7. The average speeds of these two Euro IV buses were as low as 15 and 12 $km\,h^{-1}$, respectively. The engine speed was below 1400 rpm 85 % of the time based on real-time their OBD data regarding engine speed and engine power, which indicates for a majority of the real-world operation the engines worked beyond the emission control zone under the European Steady Cycle (ESC) (i.e., the area within 13 steady modes) (Krahl et al., 2009). The NO_x brake-specific emission factors for the two tested buses are 9.4 $g\,kWh^{-1}$ and 6.6 $g\,kWh^{-1}$, respectively. Not surprisingly, their on-road NO_x brake-specific emission factors are much higher than the emission standard limits of 3.5 $g\,kW\,h^{-1}$ for Euro IV engines. Some other studies have similarly found that the emission test results of heavy-duty engines under the certification standard test cycle do not reflect real-world emissions (Liu et al., 2011; Ligterink et al., 2009). To develop a specific compliance in-use testing program for buses is clearly necessary in the future since their operating conditions are quite unique with a typical low-speed profile. Otherwise, newer engines with advanced controls that make it possible to pass the certificate test might not approach standards in real-world conditions when their controls fail or operate suboptimally.

Figure 8 presents the estimated brake-specific emission factors for test buses from Euro II to Euro IV and for test trucks from Euro I to Euro III. With one-way ANOVA method, there is no statistically significant difference in brake-specific emission factors of NO_x between the different vehicle categories and weight classifications. Clearly, the estimated brake-specific NO_x emission factors do not improve under real-world operating conditions with more stringent emission standards, although the emission limits for certification tests are tighter. As a result, the gap between the on-road brake-specific emission factors and emission limits for newer engines becomes larger and larger. For example, NO_x brake-specific emission factors for Euro II HDDVs are typically 20–70 % higher than the Euro II limit. However, for Euro III HDDVs they are 60–130 % higher and Euro IV diesel buses average 170 % higher. Even for modern Euro V trucks in Europe, recent on-road PEMS test results also indicate significantly higher emission factors relatively to emission limits and estimates based on laboratory tests (Ligterink et al., 2009). Such a situation demonstrates the clear need for China's policy-makers to develop in-use compliance testing programs and relevant regulations for HDDVs to complement the current engine certificate test program, and to secure a real on-road emission reduction benefit of NO_x for the HDDV fleet in China in the future.

3.4 Revisiting NO_x emissions of the HDDV fleet in Beijing and China

Using the on-road test data in this study, we updated total NO_x emissions for Beijing's HDDV fleet. In our previous study, the emission inventory for the vehicle fleet in Beijing in 2009 was based on emission factors simulated by the most recent version of MOBILE-China (Wu et al., 2011). The NO_x emission factors for HDDVs from this model are also presented in Figs. 3 and 6. We assume the vehicle registration data and activity data for buses and trucks remain the same as in the previous study. Emission factors for pre-Euro II non-public buses and pre-Euro I trucks are provided by the MOBILE-China model (Wu et al., 2011). The new results show that NO_x emissions (with 95 % CI) for the public diesel bus fleet, other diesel bus fleets, and the HDDT fleet in Beijing in 2009 are 9.1 ± 1.4 Gg, 4.8 ± 0.5 Gg, and 23.0 ± 2.7 Gg, respectively. These three values increase by 119 ± 15 %, 71 ± 11 % and 24 ± 12 % compared to our previous estimates. The most significant increase in NO_x emissions was found for the public bus fleet, which is attributed to Euro III and Euro IV diesel buses becoming the majority of the public bus fleet in Beijing. Such poor on-road performance in NO_x control for Euro III and Euro IV diesel buses causes a significant underestimate in NO_x emissions in previous studies. Overall, total NO_x emissions of the Beijing HDDV fleet in 2009 reached 37.0 ± 3.1 Gg, approximately an increase of 45 % compared to the previous study (25.0 Gg).

It should be noted that these new test data could not be directly used for national NO_x emission inventory development because there are other factors that need to be considered (e.g., the impact of the relatively higher sulfur content in diesel fuel on NO_x emissions in regions other than Beijing). However, we made a rough estimate of total NO_x emissions for the national HDDV fleet using these new on-road test data. Using statistical data and published reports and papers (CACART and CAAM, 2011, MEP et al., 2010, NBSC, 2010), we gathered HDDV fleet registration data by major vehicle category and vehicle activity data. We estimate that the total NO_x emissions for the national HDDV fleet in 2009 are approximately 3.9 Tg. This is higher than previous estimates. For example, MEP reported that national total NO_x emissions from vehicles were 5.3 Tg in 2009, of which diesel vehicles contributed 60 %, close to 3.2 Tg (MEP, 2010). With the data split between the HDDV and light-duty diesel vehicle (LDDV) fleets, we estimate that HDDV contribute roughly 95 % of total diesel vehicle NO_x emissions. Thus, the reported NO_x emission data for the HDDV fleet should be around 3.0 Tg. Our new estimate indicates a total increment of $\sim$1.0 Tg of NO_x for the national HDDV fleet. This is 18 % of total NO_x emissions from vehicles in China in 2009. It would also result in a 4 % increase in estimation of national anthropogenic NO_x emissions (State Council, 2010). Such a significant underestimate in vehicular

NO_x emissions triggers an urgent need to re-evaluate and update the national emission inventory as well as other regional/international emission inventories (e.g., GAINS-Asia, INTEX-B, etc.) (GAINS-Asia, 2008; Saikawa, et al., 2011; Xing et al., 2011; Zhang et al., 2009). This effort will also provide a better understanding of the future modeling for ambient NO_x concentration profiles as well as other relevant secondary pollutant profiles in the air (e.g., O_3 and nitrate particles).

Furthermore, the revised estimate of HDDV NO_x would also challenge the national control efforts for NO_x. As mentioned earlier, the new 12th FYP establishes a control target for NO_x emission reductions of 10 % for the period of 2011–2015. Previously, we expected the decrease in NO_x emissions from Euro II to Euro III HDDV could contribute to a considerable reduction in NO_x emissions for the vehicle fleet. However, the new on-road test results do not support this judgment. Therefore, to achieve the goal of NO_x mitigation from the HDDV fleet, policy-makers may have to rely on other effective control measures such as promotion of CNG/LNG buses, the development of an effective in-use compliance testing program and relevant regulations for HDDVs, and the implementation of Euro IV for heavy-duty diesel trucks nationwide.

4 Conclusions

More than 130 HDDVs in Beijing are tested using PEMS, and the real-world on-road emission profiles for NO_x as well as other pollutants (i.e., THC, CO and $PM_{2.5}$) are explored. An operating mode binning methodology is applied to obtain instantaneous emission characteristics and relate them to real-time operating conditions. We evaluate the NO_x emission factors for both buses and trucks, in detail by various emission standards and vehicle weight classifications under typical on-road driving cycles.

The average NO_x emission factors for Euro II, Euro III and Euro IV urban buses are $11.3 \pm 3.3\,\mathrm{g\,km^{-1}}$, $12.5 \pm 1.3\,\mathrm{g\,km^{-1}}$, and $11.8 \pm 2.0\,\mathrm{g\,km^{-1}}$, respectively, under a typical urban driving cycle. Different from a clearly decreasing trends in emission factors of THC, CO and $PM_{2.5}$, no statistically significant improvement in NO_x emission factors has been observed as emission standards get tightened from Euro II to Euro IV. With the detailed OBD data regarding real-time operating conditions of engines, the unsatisfactory SCR performance for Euro IV buses under low-speed driving cycles is identified. Therefore, to develop a specific compliance in-use testing program for buses is clearly necessary since their operating conditions are quite unique with a typical low-speed profile. Furthermore, the CNG bus could be a competitive alternative to NO_x control for the urban bus fleet.

Similar to diesel buses, we did not observe statistically significant improvement in NO_x emissions from Euro I to Euro III trucks for most cases. Under a typical urban driving cycle, the NO_x emission factors for Euro I, Euro II and Euro III HDDT1 are $4.8 \pm 1.5\,\mathrm{g\,km^{-1}}$, $4.6 \pm 1.8\,\mathrm{g\,km^{-1}}$, $2.0 \pm 0.7\,\mathrm{g\,km^{-1}}$, respectively. For HDDT2, the NO_x emission factors for Euro I, Euro II and Euro III are $7.8 \pm 14.0\,\mathrm{g\,km^{-1}}$, $4.3 \pm 2.0\,\mathrm{g\,km^{-1}}$ and $6.0 \pm 2.9\,\mathrm{g\,km^{-1}}$, respectively. For HDDT3, the NO_x emission factors for Euro I, Euro II and Euro III are $9.2 \pm 2.8\,\mathrm{g\,km^{-1}}$, $6.7 \pm 1.7\,\mathrm{g\,km^{-1}}$, $7.2 \pm 1.2\,\mathrm{g\,km^{-1}}$, respectively. They are lower than the NO_x emission factors for diesel buses because the average speed of the typical driving cycle for trucks is much higher.

The brake-specific NO_x emission factors did not improve either under real-world operating conditions with more stringent emission standards, although the emission limits for certification tests are tighter. Consequently, the gap between the on-road brake-specific emission factors and emission limits for newer engines becomes larger and larger. For example, NO_x brake-specific emission factors for Euro II HDDVs are typically 20–70 % higher than the Euro II limit. However, for Euro III HDDVs they are 60–130 % higher and Euro IV diesel buses average 170 % higher. Such a situation demonstrates again the clear need for China's policy-makers to develop in-use compliance testing programs and relevant regulations for HDDVs to complement the current engine certificate test program, and to secure a real on-road emission reduction benefit of NO_x for the HDDV fleet in China in the future.

Updated on-road NO_x emission factors for both buses and trucks would result in a significant underestimate in NO_x emissions for HDDV fleet from other previous studies. Our new estimate in total NO_x emissions for the Beijing HDDV fleet in 2009 is 37.0 ± 3.1 Gg, an increase of 45 % compared to the previous study. Furthermore, we estimate that the total NO_x emissions for the national HDDV fleet in 2009 are approximately 4.0 Tg, ~ 1.0 Tg higher than that estimated in the official report. The increment is equivalent to 18 % of total NO_x emissions from vehicles in China in 2009. This would also result in 4 % increase in estimation of national anthropogenic NO_x emissions. Such an underestimate in vehicular NO_x emissions triggers an urgent need to re-evaluate and update the national emission inventory as well as other regional/international emission inventories.

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