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Evaluation of emission factors for light-duty gasoline vehicles based on chassis dynamometer and tunnel studies in Shanghai, China

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HIGHLIGHTS

- 51 in-use LDGVs were tested using a chassis dynamometer in Shanghai, China.
- Continuous monitoring in a gasoline vehicle dominated tunnel were conducted in Shanghai, China.
- Emission factors of LDGVs were determined based on dynamometer test and tunnel experiment.
- High-emitting vehicles contributed the majorities of emissions from older vehicles in Shanghai, China.
- Emission factors of LDGVs were underestimated due to the overlook of high-emitting vehicles in the previous studies in China.

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ABSTRACT

CO, THC, NO_x, and PM emission factors of 51 light-duty gasoline vehicles (LDGVs) spanning the emission standards from Euro 2 to Euro 5 were measured by a chassis dynamometer. High frequencies of high-emitting vehicles were observed in Euro 2 and Euro 3 LDGV fleet. 56% and 33% of high-emitting vehicles contributed 81%–92% and 82%–85% of the emissions in Euro 2 and Euro 3 test fleet, respectively. Malfunctions of catalytic convertors after high strength use are the main cause of the high emissions. Continuous monitoring of a gasoline vehicle dominated tunnel in Shanghai, China was conducted to evaluate the average emission factors of vehicles in real-world. The results indicated that the emission factors of LDGVs were considerably underestimated in EI guidebook in China. The overlook of high-emitting vehicles in older vehicle fleet is the main reason for this underestimation. Enhancing the supervision of high emission vehicles and strengthening the compliance tests of in-use vehicles are essential measures to control the emissions of in-use gasoline vehicles at the present stage in China.

1. Introduction

Vehicle emission has been an important source of air pollution. Their NO_x, VOCs and primary PM emissions have been recognized the key precursors of $PM_{2.5}$ and ozone pollution in the regions of China (Zhang et al., 2015a; Cheng et al., 2016; Sun et al., 2016; Li et al., 2016a; Shao et al., 2016). Light-duty gasoline vehicles (LDGVs) dominate the total motor vehicle fleet in China. The statistic data show that the population of motor vehicle in China reached 172 million in 2015, of which LDGVs were 140 million, accounting for 86.2% of the total. Correspondingly, their CO, HC and NO_x emissions occupied 83.7%, 71.5%, and 26.7% of the total vehicle emissions, respectively (MEP, 2016). Their primary PM emissions also cannot be ignored according to recent studies (Huang et al., 2013). Furthermore, the SOA productions of gasoline exhaust even exceeded their primary emissions according to the smog experiment studies (Platt et al., 2013; Gordon et al., 2014; Presto et al., 2014; Huang et al., 2015; Liu et al., 2015a). LDGVs are now experiencing rapid growth in China, which has increased by 1.2 times in the last 5 years. It will be very important to accurately

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Table 1
Specifications and emission factors of test vehicles in this study.

No.	Manufacturers	Model year	Odometer (km)	Emission standard	CO (g/km)	THC (g/km)	NO_x (g/km)	PM (mg/km)	Fuel Con. ($L \cdot 100\ km^{-1}$)
1	Volkswagen	2000	254,606	Euro 2	2.73	0.456	0.860	1.35	6.91
2	Volkswagen	2003	176,427	Euro 2	0.25	0.081	0.060	1.85	6.40
3	Buick	2003	139,898	Euro 2	7.38	1.212	1.619	2.30	9.15
4	Chery	2004	216,261	Euro 2	5.90	1.127	2.256	94.5	6.93
5	Volkswagen	2004	165,337	Euro 2	1.75	0.235	0.623	4.96	7.96
6	Buick	2004	130,963	Euro 2	32.7	2.345	2.096	516	14.3
7	Buick	2004	102,923	Euro 2	5.87	1.087	1.300	13.6	6.74
8	Zhonghua	2007	118,153	Euro 2	1.98	0.106	0.173	0.81	8.12
9	Volkswagen	2007	109,440	Euro 2	21.8	2.818	0.181	49.7	8.48
10	Nissan	2006	314,110	Euro 3	6.87	0.705	2.159	61.1	9.64
11	Buick	2006	229,187	Euro 3	0.02	0.006	0.373	3.55	10.4
12	Hyundai	2007	182,649	Euro 3	10.6	1.536	2.485	123	8.87
13	Peugeot	2007	161,637	Euro 3	1.87	0.221	0.101	20.6	6.57
14	Santana	2007	146,916	Euro 3	0.57	0.055	0.119	4.34	7.99
15	Peugeot	2007	98,389	Euro 3	1.23	0.038	0.048	0.11	8.19
16	Volkswagen	2008	354,988	Euro 3	1.74	0.092	0.078	0.65	8.88
17	Audi	2008	248,132	Euro 3	0.34	0.041	0.039	12.7	9.41
18	Peugeot	2008	117,422	Euro 3	0.26	0.022	0.447	2.28	7.34
19	Hongxing	2009	84,685	Euro 3	1.65	0.025	0.688	36.8	5.35
20	BYD	2009	54,201	Euro 3	0.44	0.055	0.054	4.82	6.40
21	BYD	2010	227,715	Euro 3	14.2	0.784	0.530	28.1	7.26
22	Chevrolet	2008	139,692	Euro 4	1.77	0.121	0.087	3.53	6.57
23	Volkswagen	2008	119,755	Euro 4	0.03	0.000	0.042	0.66	7.62
24	Chevrolet	2008	65,261	Euro 4	0.70	0.025	0.070	2.48	6.33
25	Mazda	2009	193,880	Euro 4	0.42	0.013	0.034	2.76	7.83
26	Nissan	2009	99,096	Euro 4	0.39	0.039	0.021	5.42	9.83
27	Skoda	2009	83,145	Euro 4	0.37	0.043	0.008	11.2	6.83
28	Buick	2009	67,382	Euro 4	0.74	0.001	0.086	0.26	7.47
29	Toyota	2010	209,169	Euro 4	1.34	0.097	0.181	3.14	9.27
30	Chevrolet	2010	147,359	Euro 4	1.30	0.081	0.083	2.81	8.22
31	Passat	2011	153,529	Euro 4	4.78	0.075	0.108	0.01	8.72
32	Chery	2011	126,838	Euro 4	1.23	0.028	0.088	1.20	7.20
33	Toyota	2011	62,323	Euro 4	0.07	0.002	0.004	9.97	5.94
34	Passat	2011	60,159	Euro 4	0.99	0.048	0.025	2.72	7.88
35	Chery	2011	31,774	Euro 4	0.04	0.008	0.018	1.30	6.45
36	Chevrolet	2012	48,741	Euro 4	0.41	0.040	0.024	8.87	7.15
37	Mazda	2012	26,526	Euro 4	0.74	0.001	0.086	3.84	7.83
38	Hyundai	2013	145,999	Euro 4	0.59	0.056	0.040	15.7	6.24
39	Chery	2013	81,460	Euro 4	0.93	0.045	0.143	1.87	6.82
40	Audi	2013	78,018	Euro 4	0.74	0.119	0.070	28.8	7.82
41	Volkswagen	2013	16,779	Euro 4	1.04	0.074	0.064	9.03	6.95
42	Buick	2012	36,902	Euro 5	0.17	0.036	0.012	3.35	8.12
43	Soueast	2014	57,840	Euro 5	0.08	0.012	0.016	0.65	5.57
44	Buick	2014	46201	Euro 5	1.38	0.038	0.036	27.3	10.2
45	Buick	2014	45,820	Euro 5	1.99	0.025	0.006	7.39	8.73
46	Audi	2016	75,021	Euro 5	0.26	0.049	0.039	31.3	8.32
47	Buick	2016	51,151	Euro 5	1.69	0.052	0.024	0.86	6.91
48	Hyundai	2016	13,596	Euro 5	0.85	0.054	0.029	8.19	6.72
49	Hyundai	2016	10,276	Euro 5	0.69	0.026	0.012	8.18	6.75
50	Honda	2016	7,857	Euro 5	0.90	0.044	0.006	11.9	7.89
51	Kia	2016	5,109	Euro 5	0.69	0.028	0.018	2.18	6.22

quantify their emissions.

Accurate vehicle emission factors are essential to recognize their contribution to air pollution. Vehicle emission models, such as MOVES, COPERT, EMFAC, and IVE, have been widely used to develop vehicle emission inventory in previous studies (Wang et al., 2008; Wallace et al., 2012; Cai and Xie, 2013; Jing et al., 2016). In China, the MEP (Ministry of Environmental Protection of People's Republic of China) released an on-road vehicle emission inventory guidebook (shorten as EI guidebook) which recommended a series of vehicular emission factors based on the local studies (MEP, 2014). However, some measurements on LDGVs demonstrated that the modeled emission factors still have some differences with the measured ones (Fujita et al., 2012; Liu and Frey, 2015). Most of the results from real world measurements by remote sensing and chasing studies indicated that high emitters caused by deterioration of emission control devices tended to be underestimated in vehicle emission inventories (Park et al., 2011; Zhou et al., 2014; Wang et al., 2015; Pujadas et al., 2017). To understand the emission factors of LDGVs, some measurement studies have been conducted in recent studies in China (Huo et al., 2012; Shen et al., 2014; Qu et al., 2015; Li et al., 2016a). However, the test samples were limited and the results still show big differences compared with the

Table 2
Adjustment factors of EFs under different speed ranges.

Pollutants	Average speed ($km \cdot h^{-1}$)							
	10	20	30	40	50	60	70	80
CO	2.72	1.36	1.00	0.84	0.72	0.56	0.48	0.41
THC	2.93	1.47	1.07	0.89	0.75	0.60	0.52	0.45
NOx	2.46	1.23	0.97	0.88	0.77	0.65	0.56	0.52
PM	2.25	1.13	1.03	1.02	0.91	0.66	0.56	0.45

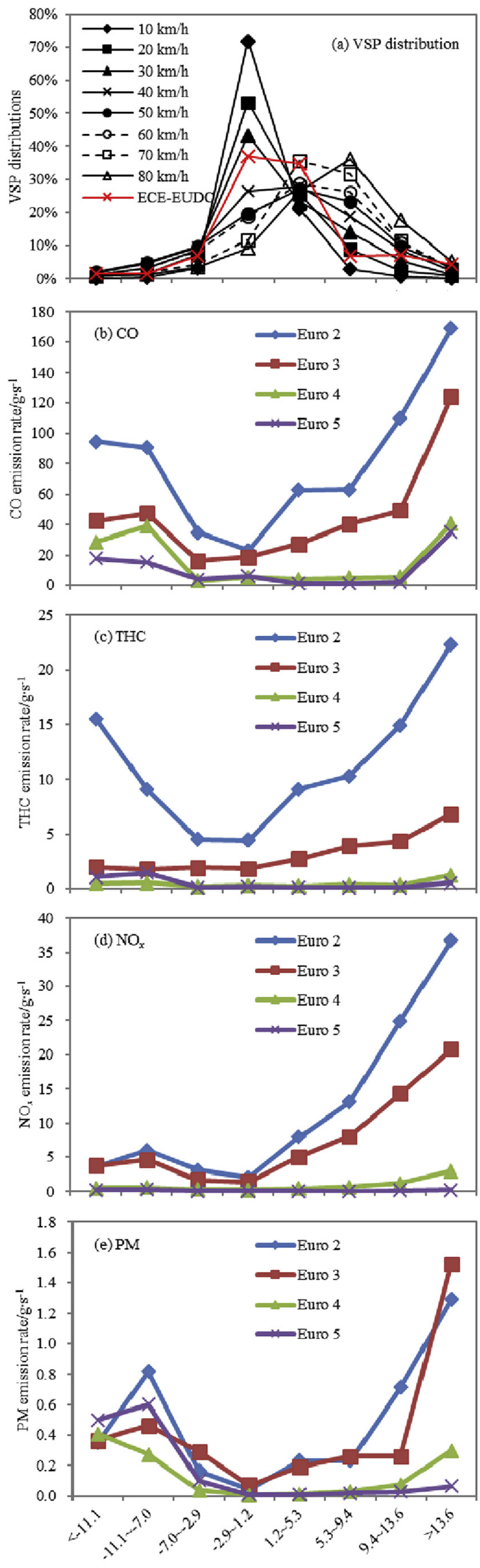

Fig. 1. Fractions of VSP bins under different average speeds in real-world and under ECE-EUDC driving cycle (a) and CO, THC, NO_x, and PM emission rates in each VSP bin for the vehicles with different emission standard (b)~(e).

emission factors in MEP EI guidebook and emission inventory studies (Cai and Xie, 2013; MEP, 2014; Lang et al., 2014).

China is experiencing a rapid improvement of vehicle emission control technologies. Emission standards of LDGVs upgraded from Euro 1 to Euro 5 over the past 16 years. Euro 0 and Euro 1 gasoline vehicles were almost eliminated in recent 4 years. Rapid change of vehicle emission control technologies makes it necessary to reconsider their emission factors. Tunnel study is considered to be an effective method to evaluate the real world emission factors of vehicle fleets, which has been widely used all over the world (Colberg et al., 2005; Grieshop et al., 2006; Ning et al., 2008; Dallmann et al., 2013; Zhang et al., 2015b). In this study, we recruited 51 LDGVs with different emission standards of Euro 2 to Euro 5 to measure their emission factors on a chassis dynamometer using a constant volume sampler, commercial fuels and standard ECE (Economic Commission of Europe) and EUDC (Extra Urban Driving Cycle) cycles. In addition, we installed 2 continuous monitoring devices in the inlet and outlet of a tunnel dominated by LDGVs to investigate the real world emission factors. Both the emission factors obtained by chassis dynamometer and tunnel experiments were compared with the recommended ones of emission inventories to evaluate the reliability of LDGVs emissions in China.

2. Materials and methods

2.1. Chassis dynamometer measurements

2.1.1. Tested vehicles and fuels

Tailpipe emissions from 51 LDGVs spanning model years from 2000 to 2016 were measured in this study. The tested vehicles were recruited from the residents and car rental agencies in Shanghai, China. The vehicles were randomly selected to span a wide range of model years, emission standards, accumulative mileages, engine displacements, and manufacturers. The test fleet comprised of 9 Euro 2, 12 Euro 3, 20 Euro 4, and 10 Euro 5 vehicles. Their accumulative mileages ranged from 0.5×10^4 km to 35.5×10^4 km. All of the vehicles were tested using Euro 5 commercial fuels in Shanghai, China. Table 1 lists the LDGV test fleet and their detail information.

2.1.2. Emission measurements

All LDGVs were tested on a VULCAN EMSCD48 1.22-m single roll electric chassis dynamometer produced by Horiba at Shanghai Motor Vehicle Inspection Certification & Tech Innovation Center. Tailpipe emissions were sampled using a constant volume sampling (CVS) system (Horiba CVS-7000). Raw exhaust was diluted with clean air filtered by high-efficiency particulate air (HEPA) filters in the CVS. The CVS was operated at a constant flow rate. The flow rates of the measurements in this study were 6.2, 8.2, and 10.2 $m^3\ min^{-1}$. For high-emitting vehicles, the CVS was operated at a higher flow rate. The dilution ratios were about 18–36 to 1. Exhaust gas analysis was performed using a Horiba MEXA-7000 measurement system. CO and CO_2 were measured using non-dispersive infrared (NDIR) detectors, THC by Flame Ionization Detection (FID), and NO_x by chemiluminescence (CLD). To measure PM emissions, a small portion of exhaust was extracted by a fine particle sampler with two stage dilution system (Dekati FPS 4000) before the exhaust was accessed to the CVS. The exhaust sample was heated up to 200–250 °C and then diluted by 10–12 times before the measurement. Diluted exhaust from Dekati FPS 4000 was sampled into a fine particle measurement instrument named Electrical Low Pressure Impactor (Dekati ELPI) to measure the instantaneous PM concentrations. All of vehicles were tested using the "hot-start" ECE + EUDC Cycle. Each vehicle was fully preheated prior to testing. The time length was 1180 s for each cycle. The average and maximum speed was 33.5 and 120 km h^{-1}, respectively.

2.1.3. Adjustments of emission factors under different speeds

Since ECE-EUDC driving cycle is different with real-world driving conditions, we adjusted the EFs under ECE-EUDC driving cycle to those under different average speeds in real world. Vehicle Specific Power (VSP) of ECE-EUDC driving cycle was calculated second by second according to the equation introduced by Jimenez (1999) and then classified into 8 bins. Emission rates of each VSP bin were averaged for the LDGVs with same emission standard. Then we calculated the EFs under different speeds based on the VSP distribution under each speed range from the previous surveys in real-world. Then the EFs under different speeds were calculated by equation (1) as follows. According to equation (1), adjustment factors were calculated by dividing the EFs under different speeds to those under ECE-EUDC driving cycle, as shown in Table 2.

$$EF_i = \sum \frac{ER_j \times R_{i,j}}{v_i} \times 3600 \quad (1)$$

Where, EF_i ($g \cdot km^{-1}$) is adjusted EFs under average speed i. $R_{i,j}$ is the proportion of VSP bin j for average speed i. v_i ($km \cdot h^{-1}$) is the average speed as shown in Fig. 1(a). ER_j ($g \cdot s^{-1}$) is the average emission rates in VSP bin j. The ER_j of the tested vehicle with each emission standard was shown in Fig. 1(b)~(e).

2.2. Tunnel measurements

2.2.1. Air pollutant monitoring

A tunnel in the city center of Shanghai (31.24 °N, 121.50 °E) was selected to conduct air pollution monitoring. The tunnel is called East Yan'an Road tunnel, is an underwater tunnel crossing the Huangpu River in Shanghai. It has two bores with two lanes for each direction. The total length is 2,107 m; cross-sectional area is 33.75 m^2. To continuously monitor major air pollutants from vehicle exhaust, we installed two integrated air pollution monitoring units at 150 m away from the inlet and outlet of the tunnel, respectively. The height of the unit was about 1.2 m. Fig. 2 shows the location of the tunnel and the monitoring sites. There were 4 sensors which could separately monitor NO, NO_2, CO, and $PM_{2.5}$ concentrations in each unit. NO, NO_2, and CO were detected using electrochemical method. Their detection limits were 0.01 mg m^{-3}, 0.01 mg m^{-3}, and 0.15 mg m^{-3}, respectively. $PM_{2.5}$ was detected by light scattering method. Its resolution was 0.1 μg m^{-3}. The monitoring data was continuously transmitted to a remote database through wireless network. The system was built in July 2016 and has been operated by now. To ensure the reliability of monitoring

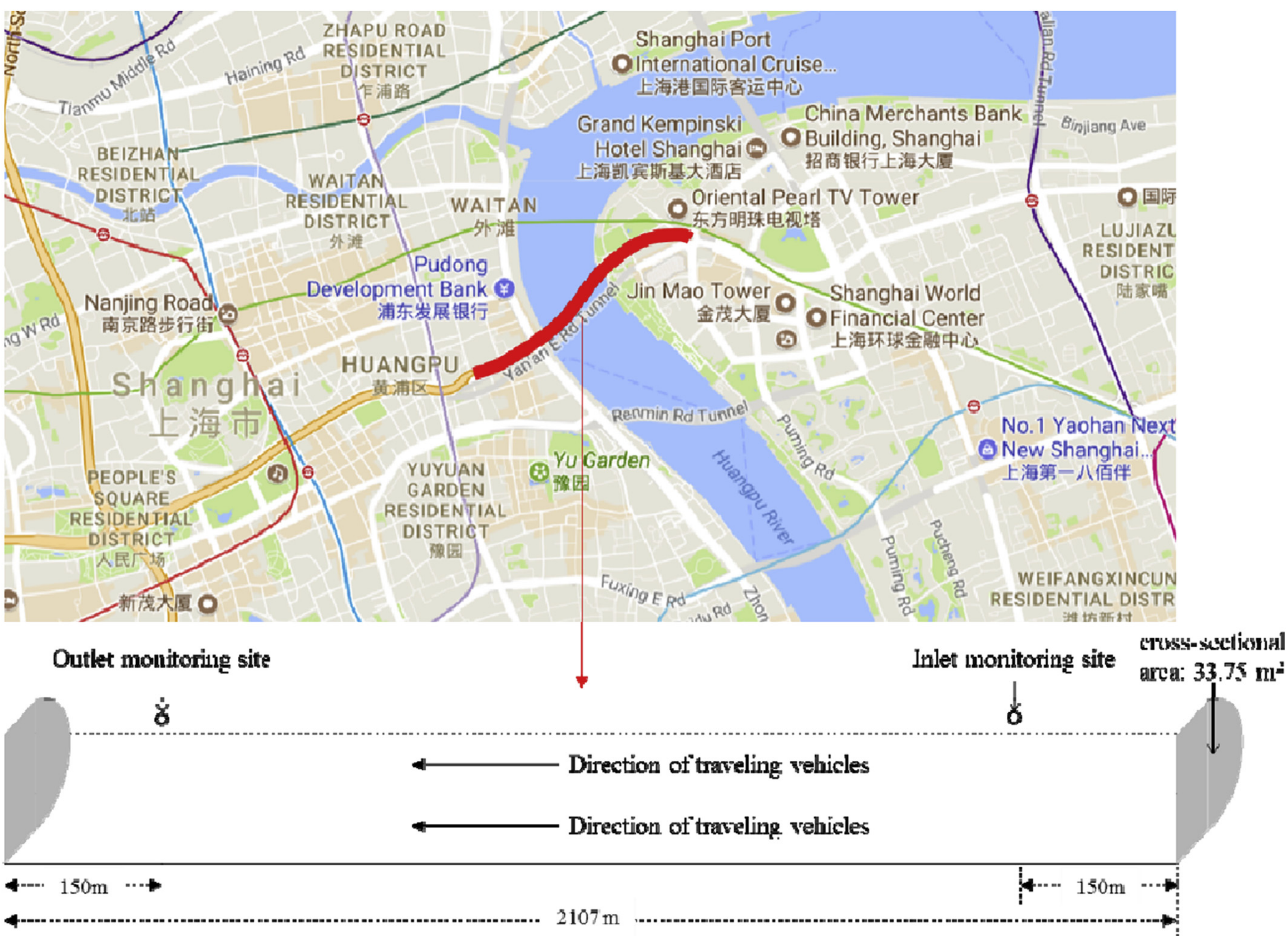

Fig. 2. Schematic diagram of the monitoring site in the East Yan'an Road Tunnel, Shanghai.

data, we regularly carried out maintenance of the monitoring units, calibrated the sensors, and cleaned the sampling pipes. Fig. 3 shows a fragment of NO, NO_2, CO, and $PM_{2.5}$ concentrations at inlet and outlet of the tunnel.

2.2.2. Traffic flow counting

Vehicle flow and average speed in the tunnel were detected by a buried induction coil and transmitted to a data platform in real time. The fleet compositions were recognized by automatic acquisition of vehicle license plate information through a high-definition video in the tunnel. Fig. 4 shows the average traffic flow and fleet composition in each hour of weekday and weekend. The data between August and November of 2016 were used in this study. Daily average vehicle flow and speed were 34,818 ± 2,214 veh·h^{-1} and 33.47 ± 7.95 km h^{-1} on weekday and were 30,818 ± 1,848 veh·h^{-1} and 33.86 ± 8.83 km h^{-1} on weekend. Vehicle flows in peak hour were 2,173 ± 161 and 1,952 ± 68 veh·h^{-1} on weekday and weekend, respectively. East Yan'an Road tunnel was located in the city center of Shanghai, where heavy-duty diesel vehicles (HDDVs) were restricted to access except for some medium-duty diesel vehicles (MDDVs). Therefore, fleet in the tunnel was mainly comprised of light-duty gasoline vehicles, which accounted for 93.7 ± 2.2% and 94.6 ± 1.9% of the total on weekday and weekend, respectively. The rest of the vehicles were mainly MDDVs (5.3 ± 1.7%) and Buses (0.5 ± 0.5%). According to the license plate information, we recognized that LDGVs were mainly from Euro 2 to Euro 5, which accounted for 10.3%, 9.9%, 62.1%, and 17.7%, respectively; MDDVs were from Euro 3 to Euro 5, accounted for 57.5%, 42.0%, and 0.5%; Buses were also from Euro 3 to Euro 5, accounted for 61.3%, 32.4%, and 6.3%, as shown in Fig. 5.

2.2.3. Emission factor calculation

Emission factors for vehicles traveling through the tunnel were calculated using the following equation which has been widely used in previous studies (Chang et al., 2009; Liu et al., 2015b; Zhang et al., 2015b; Cui et al., 2016).

$$EF = \frac{(C_{outlet} - C_{inlet}) \times A \times v \times t \times 10^{-3}}{N \times L} \quad (2)$$

Where, EF (g·km^{-1}) is the average emission factor per vehicle per kilometer of fleet in the tunnel. C_{outlet} and C_{inlet} (mg·m^{-3}) are air pollutant concentrations at outlet and inlet of the tunnel. A (m^2) is the cross-sectional area of the tunnel ($A = 33.75$ m^2). v (m·s^{-1}) is air velocity parallel to the tunnel monitored by an operation management company of the tunnel. Daily average air velocity was

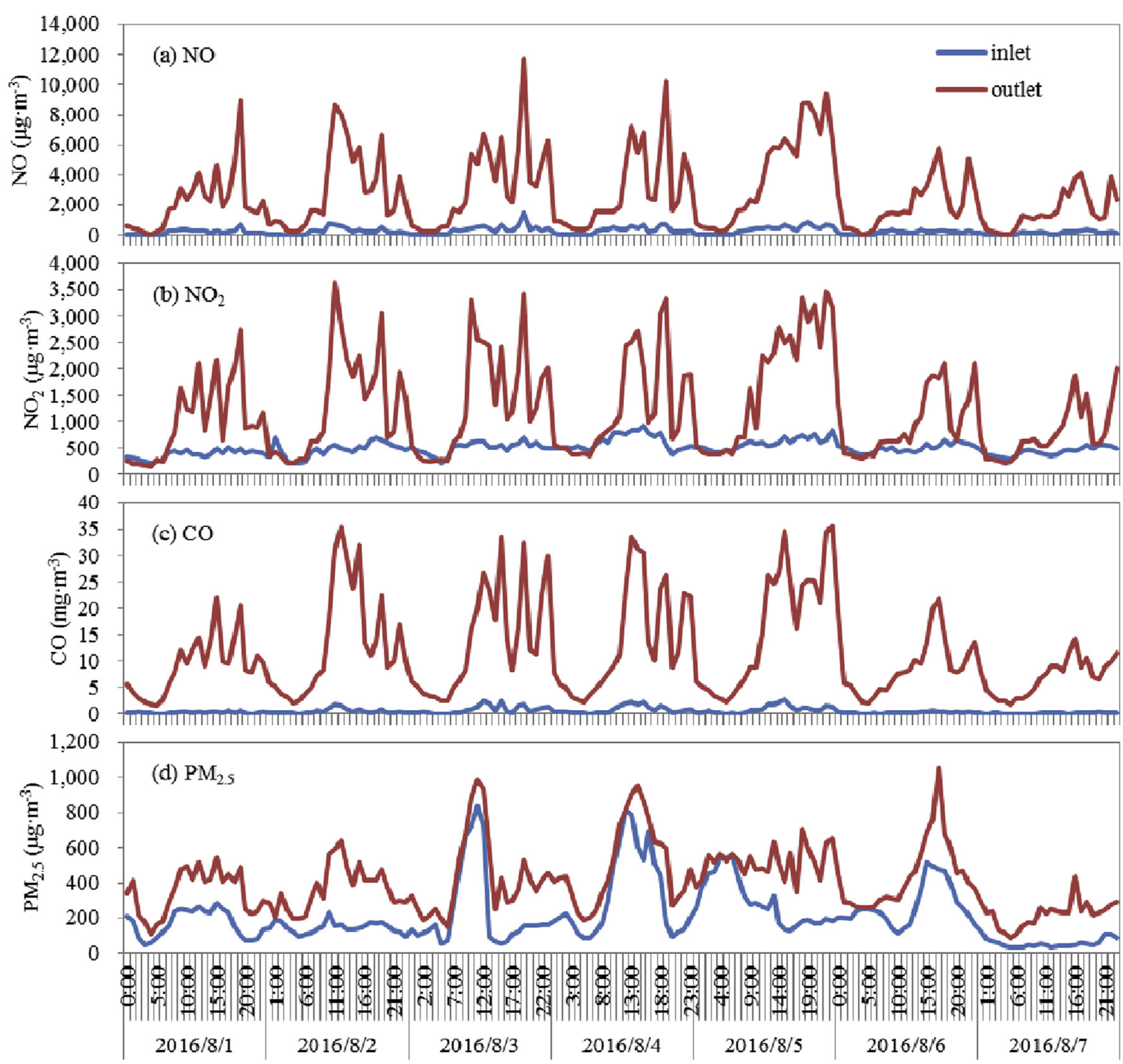

Fig. 3. Air pollutant concentrations at inlet and outlet of the tunnel. (a) NO; (b) NO_2; (c) CO; (d) $PM_{2.5}$.

about 3.3 m s^{-1}. t (s) is time interval (1 h in this study). N is the vehicle number passing the tunnel during the time interval. L (km) is the length between the two monitoring sites ($L = 1.807$ km).

3. Results and discussion

3.1. Emission factors of test vehicles

The EFs of the tested vehicles were listed in Table 1. Fig. 6 shows the EFs of the tested vehicles grouped by their usages and emission standards. High, medium, and low use represent for the accumulative mileages of "$>16 \times 10^4$ km", "$8–16 \times 10^4$ km", and "$<8 \times 10^4$ km", respectively. There are a considerable number of LDGVs are high-emitting vehicles in Euro 2 and Euro 3 test fleets. The high-emitting (H-emitting) vehicles in this study refer to the vehicles whose tailpipe emissions exceed 3 times of their limits (See the red lines in Fig. 6). The others were normal-emitting (N-emitting) vehicles. The H-emitting vehicles had 5–10 times higher emission rates than the others and mostly existed in the older vehicles with high and medium mileage uses. In contrast, emissions from Euro 4 and Euro 5 LDGVs had been better controlled. Fewer vehicles exceeded the limits of H-emitting vehicles.

There are many factors contributing to the high emissions of gasoline vehicles, such as fuel quality, driving condition, and deterioration, etc. In this study, the vehicles were all tested under ECE-EUDC cycle by the same driver based on the same dynamometer and measurement system. The impact of driving condition on high emission can be neglected. Since the fuel specifications of the tested vehicles were not detected in this study, their influences on emissions were hard to distinguish. However, the fuel quality in the cities of Eastern China has been greatly improved. Shanghai began to supply ultra-low sulfur (China 5) fuels since 2013 (Wu et al., 2017). Fuel quality is not very responsible for the high emission. Fig. 7 shows the comparisons of second by second emission rates of H-emitting and N-emitting Euro 2 and Euro 3 vehicles in this study. During the hot-starts, the emission rates of H-emitting and N-emitting vehicles were relatively close. After about 200 s, the emission rates of N-emitting vehicles decreased

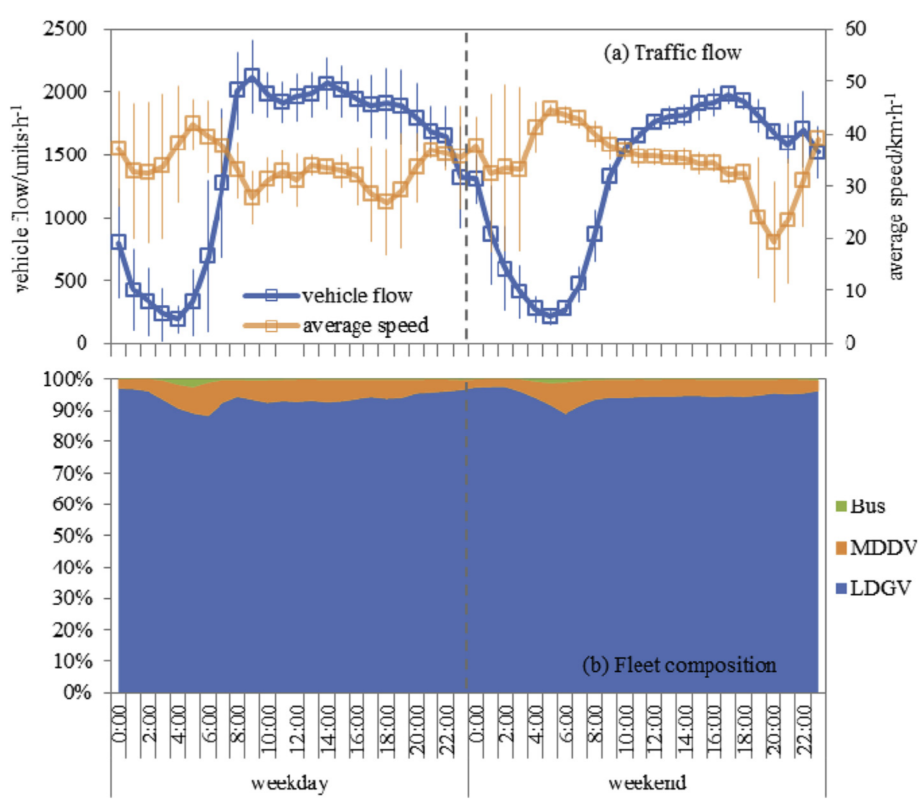

Fig. 4. Diurnal variations of traffic flow and fleet composition on weekday and weekend.

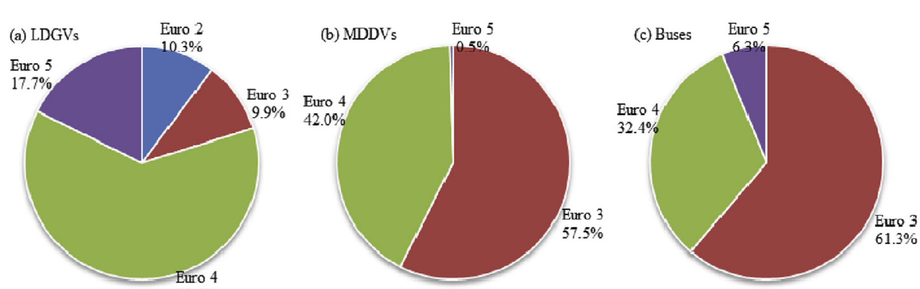

Fig. 5. Compositions of emission standards for LDGVs (a) and MDDVs (b).

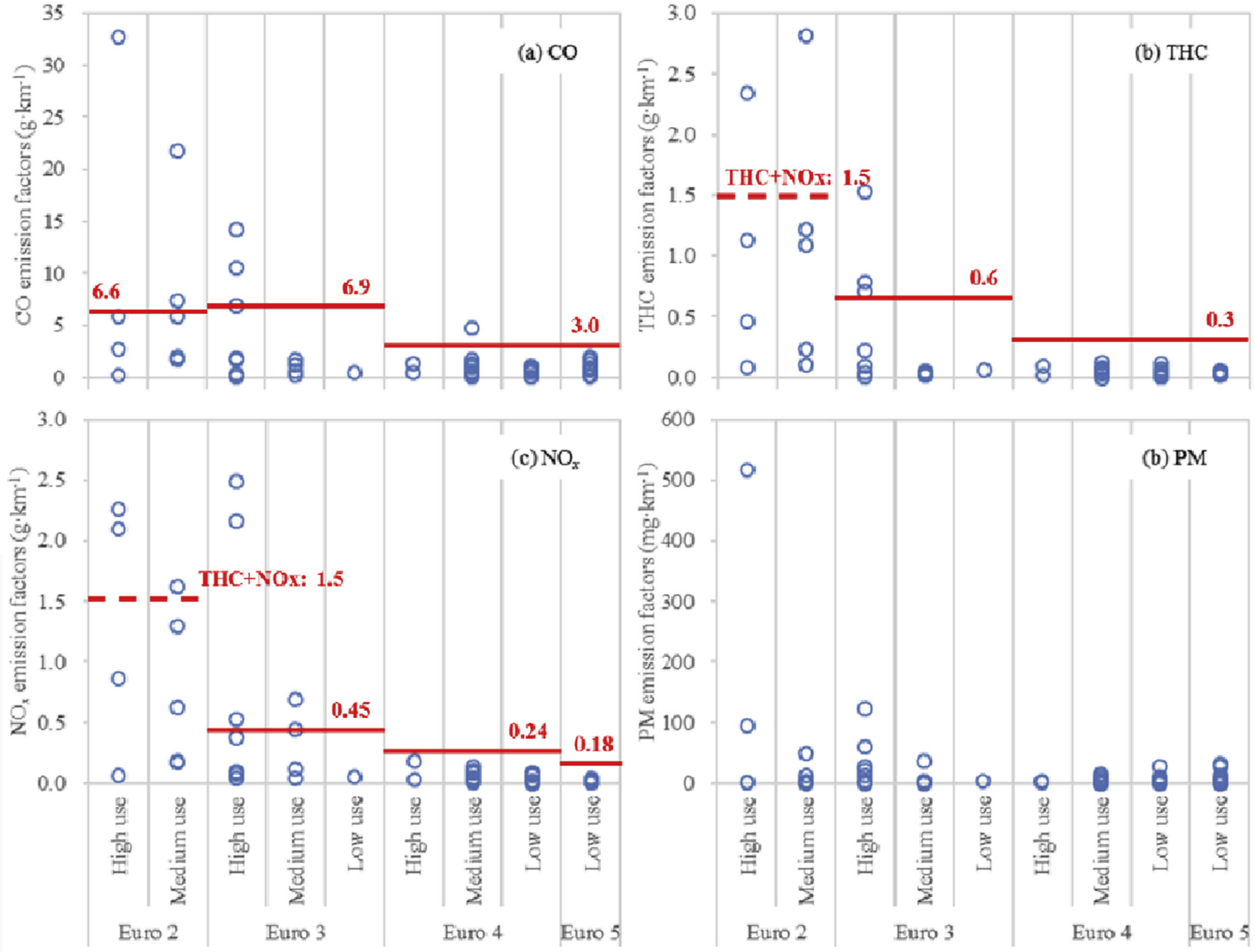

Fig. 6. Emission factors of the tested vehicles grouped according to the vehicle mileage uses and emission standards. (a) CO; (b) THC; (c) NO_x; (d) $PM_{2.5}$. The red lines represent the limits for high-emitting vehicles. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

significantly since the catalytic converters of the tested vehicles began to work with the rise of exhaust temperature. However, the emissions of H-emitting vehicles kept on high levels, indicating that the catalytic converters of these H-emitting vehicles were basically ineffective and failed to reduce the exhaust emissions. Deteriorations of catalytic convertors of older vehicles should be the main cause of their higher emission rates.

3.2. Contributions of high emission vehicles

Previous studies in the US have revealed that the emissions from a few of H-emitting vehicles often contribute the majorities of fleet emissions (Bishop and Stedman, 2008; Pang et al., 2014; Bishop et al., 2016). However, the contribution of H-emitting vehicle in China still remains unclear. Table 3 summaries the average EFs of the tested LDGVs in different emission standards in this and previous studies in China and the emission contributions of H-emitting vehicles. Only Huo et al. (2012)'s measurements were conducted by PEMS in real-world. Other studies were all based on the dynamometer tests under the ECE-EUDC cycle (Gao et al., 2010; Wang et al., 2014; Li et al., 2016b; Fu et al., 2017). The average EFs of Euro 2 and Euro 3 LDGVs in this study were significantly higher than those in the previous studies. The existence of H-emitting vehicles in this study is the main reason for its higher EFs. The proportions of H-emitting vehicles were 56% and 33% in Euro 2 and Euro 3 test fleet in our study. Their emission contributions of H-emitting vehicles came up to 81%–92% and 82%–85%, respectively. However, the contributions of H-emitting vehicles were underestimated in the previous studies. Huo et al. (2012) recognized 27% and 15% of H-emitting vehicles in Euro 2 and Euro 3 test fleet. Other studies haven't found any H-emitting vehicles. Sample selections should be important to result in the differences of H-emitting vehicle contributions. On the one hand, the numbers of test samples in most of the previous studies were too limited to obtain the H-emitting results. On the other hand, the mileages of test vehicles in other studies were mostly less than medium or high use. The tested vehicles haven't been deteriorated yet. The results imply that the EFs of LDGVs in previous studies may be underestimated due to the overlook of H-emitting vehicles.

3.3. Average emission factors by tunnel experiment

The EFs of CO, NO_x, and $PM_{2.5}$ in East Yan'an Road tunnel were calculated based on Eq. (2) as shown in Table 4. CO and NO_x EFs were 1.84 ± 0.90, 0.40 ± 0.25 g $km^{-1}\cdot veh^{-1}$, and $PM_{2.5}$ EF was 34.0 ± 23.5 mg $km^{-1}\cdot veh^{-1}$. The EFs in this study were compared with those from other gasoline vehicle dominated tunnel studies in China. Chang et al. (2009) tested vehicle emissions of Hsuehshan

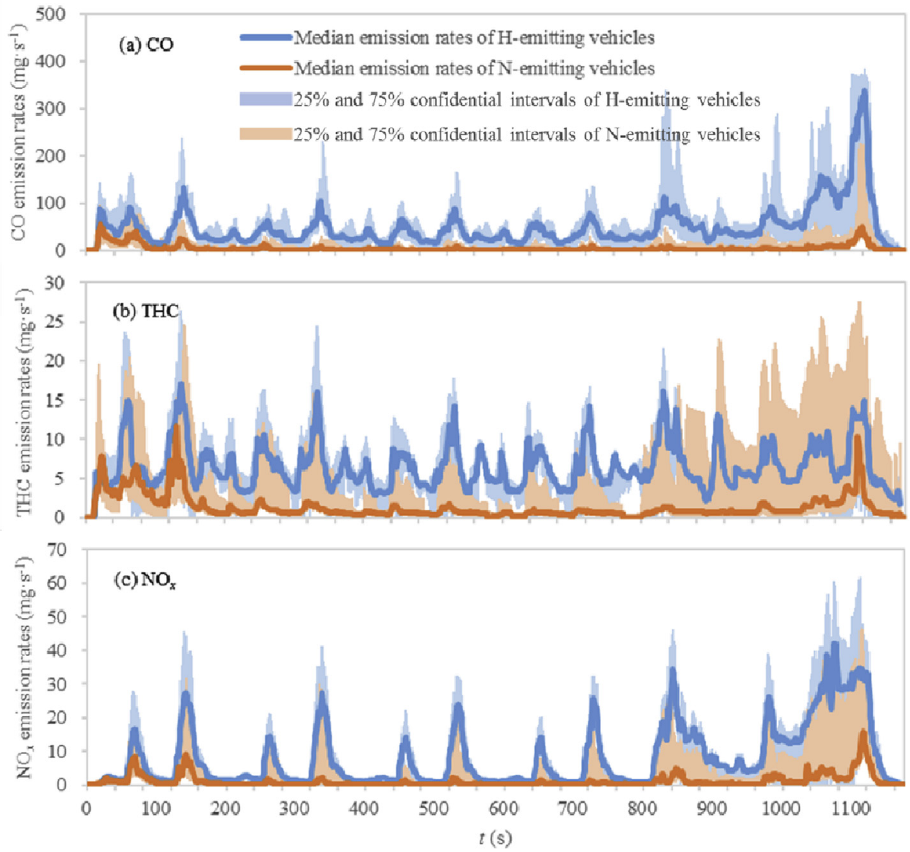

Fig. 7. Comparisons of second by second emission rates of H-emitting and N-emitting Euro 2 and Euro 3 vehicles. (a) CO; (b) THC; (c) NO_x.

tunnel in Taiwan in 2006. The CO, NO_x, and $PM_{2.5}$ EFs were $0.91 \pm 0.47–1.47 \pm 0.63$, $0.15 \pm 0.07–0.33 \pm 0.17$ g km^{-1}·veh^{-1}, and $2 \pm 2–4 \pm 3$ mg km^{-1}·veh^{-1}, respectively. Their results were the lowest compared with other tunnel studies in China mainly because Taiwan implemented Tier 1 and Tier 2 emission standards in 1999 and 2008, earlier than the mainland of China. Liu et al. (2015a,b) and Deng et al. (2015) measured CO and PM_{10} EFs in East Yan'an Road tunnel in 2012. CO EF decreased by about 30% compared with our result in 2016. $PM_{2.5}$ EF also declined if we considered PM_{10} was similar to $PM_{2.5}$ for vehicle exhaust. The enforcement of vehicle emission control strategies should be the main reason for the decrease. Between 2012 and 2016, Shanghai implemented the emission standards of LDGVs from Euro 4 to Euro 5, eliminated Euro 0 gasoline vehicles from the whole city, and restricted Euro 1 gasoline vehicles in urban area. CO and NO_x EFs in Yingpan Road tunnel were $0.75 \pm 0.56–6.05 \pm 5.94$ and $0.12 \pm 0.02–0.82 \pm 0.76$ g km^{-1}·veh^{-1} (Deng et al., 2015). Our results were relatively lower than the average value of Yingpan Road tunnel. The fractions of gasoline and diesel vehicles have influences on EFs in the tunnels. CO, NO_x, and $PM_{2.5}$ EFs in Zhujiang tunnel were much higher than those in our study since there was higher proportion of diesel vehicles (Liu et al., 2014; Zhang et al., 2015b). Correspondingly, KXL tunnel had lower $PM_{2.5}$ EF due to its high proportion of gasoline vehicles (Cui et al., 2016).

3.4. Comparisons of emission factors based on tunnel, measurements, and EI guidebook

According to the compositions of vehicles driving through the tunnel (see Figs. 3 and 4), we calculated the weighted average of EFs for each vehicle type and emission standard. Measured EFs in this study and recommended ones in EI guidebook were both considered to make comparisons. The EFs were adjusted by the factors in Table 2 according to the speeds in the tunnel. EFs in EI guidebook have their own adjustment factors (MEP, 2014). The weighted average of EFs of MDDVs and Buses were simultaneously calculated based on the EFs of individual emission standard recommended in EI guidebook (MEP, 2014). Fig. 8 shows the calculated average EFs based on dynamometer tests in this study and EI guidebook, and their comparisons with measured EFs in the tunnel. The calculated EFs of CO and NO_x based on this study were very close to the measured EFs in the tunnel. The calculated $PM_{2.5}$ EF based on this study was 20.8 mg km^{-1}·veh^{-1}, about 40% lower than the measured one in the tunnel. A possible reason was non-exhaust $PM_{2.5}$ emissions, such as tyre wear, brake wear, and road wear, were not consider in this study. Timmers and Achten, 2016 reported $PM_{2.5}$ EFs from tyre, brake, and road wears of LDGVs were 2.9, 2.2, 3.1 mg km^{-1}·veh^{-1}, respectively. The average EF of $PM_{2.5}$ would increase to 29.0 mg km^{-1}·veh^{-1}, close to the measurement result in the tunnel. The calculated EFs of CO, THC, NO_x, and $PM_{2.5}$ based on EI guidebook were 55%, 25%, 32%, and 46% lower than those calculated by our study, and also lower than the measured EFs in the tunnel. Although there was no THC monitoring data in the tunnel to evaluate the THC EFs, we believe that THC EFs in EI guidebook also have large underestimates according to the analysis in section 3.2. The comparisons indicated that the EFs of LDGVs were generally underestimated in EI guidebook.

4. Conclusions

High frequencies of high-emitting vehicles were observed in Euro 2 and Euro 3 LDGV fleet according to the measurements by

Table 3
The average emission factors of the tested LDGVs in different emission standards in this and previous studies in China and the emission contributions of H-emitting vehicles.

Emission standards	Test year	Test methods	Numbers	Mileages ($\times 10^4$ km)	Share of H-emitting	Emission factors (proportions of H-emitting)				Sources
						CO ($g \cdot km^{-1}$)	THC ($g \cdot km^{-1}$)	NO_x ($g \cdot km^{-1}$)	PM ($mg \cdot km^{-1}$)	
Euro 2	2015–2017	Dynamometer	9	10.3–25.5	56%	8.9 ± 11.0 (92%)	1.05 ± 0.98 (91%)	1.02 ± 0.84 (81%)	76.1 ± 167.9 (99%)	This study
	2008–2010	PEMS	22	N/A	27%	4.1 ± 2.8 (54%)	0.31 ± 0.34 (69%)	0.47 ± 0.39 (53%)	/	Huo et al., 2012
	2015	Dynamometer	2	16.4–23.1	0%	2.1 ± 1.9	0.20 ± 0.17	0.47 ± 0.04	34.7 ± 18.4	Li et al., 2016b
Euro 3	2015–2017	Dynamometer	12	5.4–35.5	33%	3.3 ± 4.7 (84%)	0.30 ± 0.47 (85%)	0.59 ± 0.84 (82%)	24.8 ± 36.1 (84%)	This study
	2008–2010	PEMS	13	N/A	15%	2.2 ± 2.3 (49%)	0.09 ± 0.07 (15%)	0.23 ± 0.29 (41%)	/	Huo et al., 2012
	2009	Dynamometer	1	0.3	0%	3.4	0.07	0.01	7.7	Gao et al., 2010
	2011	Dynamometer	60	1.5–8.0	N/A	1.3	0.10	0.08	/	Wang et al., 2014
	2015	Dynamometer	2	2.2–13.5	0%	1.5 ± 0.2	0.06 ± 0.04	0.18 ± 0.18	22.6 ± 10.3	Li et al., 2016b
Euro 4	2015–2017	Dynamometer	20	1.7–20.9	5%	0.93 ± 1.02 (26%)	0.05 ± 0.04 (8%)	0.06 ± 0.05 (8%)	5.78 ± 6.87 (0%)	This study
	2008–2010	PEMS	7	N/A	0%	0.40 ± 0.21	0.02 ± 0.01	0.05 ± 0.03	/	Huo et al., 2012
	2009	Dynamometer	3	0.3–0.4	0%	0.59 ± 0.84	0.02 ± 0.01	0.02 ± 0.01	3.81 ± 0.80	Gao et al., 2010
	2011	Dynamometer	118	1.5–9.0	N/A	0.90	0.08	0.08	/	Wang et al., 2014
	2015	Dynamometer	2	0.2–0.9	0%	0.60 ± 0.03	0.01 ± 0.00	0.01 ± 0.00	0.95 ± 0.21	Li et al., 2016b
Euro 5	2015–2017	Dynamometer	10	0.5–7.5	0%	0.87 ± 0.64	0.04 ± 0.01	0.02 ± 0.01	10.1 ± 10.8	This study
	2015	Dynamometer	4	0.5–5.5	0%	0.19 ± 0.16	0.03 ± 0.02	0.01 ± 0.02	/	Fu et al., 2017

Table 4
Average emission factors of East Yan'an Road tunnel in this study and their comparisons with other tunnel studies.

Tunnel	City/Region	Vehicle type	Test year	Emission factors			References
				CO ($g \cdot km^{-1} \cdot veh^{-1}$)	NO_x ($g \cdot km^{-1} \cdot veh^{-1}$)	$PM_{2.5}$ ($mg \cdot km^{-1} \cdot veh^{-1}$)	
Hsuehshan tunnel (Downslope)	Taiwan	LDV	2006	0.91 ± 0.47	0.15 ± 0.07	2 ± 2	Chang et al., 2009
Hsuehshan tunnel (Upslope)	Taiwan	LDV	2006	1.47 ± 0.63	0.33 ± 0.17	4 ± 3	Chang et al., 2009
East Yan'an Road tunnel	Shanghai	94.5% Gasoline	2012			56.7 ± 69.4[a]	Liu et al., 2015b
East Yan'an Road tunnel	Shanghai	91.5%–96% Gasoline	2012	1.27 ± 0.89–3.97 ± 2.19			Deng et al., 2015
Yingpan Road tunnel	Changsha	96.3%–98.4% Gasoline	2013	0.75 ± 0.56–6.05 ± 5.94	0.12 ± 0.02–0.82 ± 0.76		Deng et al., 2015
Zhujiang tunnel	Guangzhou	88.2% Gasoline	2013		0.56 ± 0.05		Liu et al., 2014
Zhujiang tunnel	Guangzhou	61% LDV; 27% LPG; 12% HDV	2014	3.10 ± 0.68	1.29 ± 0.20	82.7 ± 28.3	Zhang et al., 2015a,b
KXL tunnel	Yantai	96.6% Gasoline	2014			22.2 ± 6.5	Cui et al., 2016
East Yan'an Road tunnel	Shanghai	94.1% Gasoline	2016	1.84 ± 0.90	0.40 ± 0.25	34.0 ± 23.5	This study

[a] PM_{10} emission factor.

chassis dynamometer in this study. Their EFs were 5–10 times higher than those of normal-emitting LDGVs. Malfunctions of catalytic convertors of LDGVs after high strength use are the main cause of their high emissions. The vehicle ages and mileage travels of high-emitting LDGVs mostly exceeded 8 years and 100 thousand km. Due to the existence of high-emitting vehicles, the average EFs of Euro 2 and Euro 3 LDGVs in this study were generally higher than those of the previous studies in China. In comparison, emissions from newer LDGVs, like Euro 4 and Euro 5 vehicles, have been well controlled. No high-emitting vehicle has been found.

The EFs of vehicles passing through a gasoline vehicle dominated tunnel were obtained by continuous monitoring in this study. The EFs of vehicles in the tunnel were consistent with those calculated by the measurement results of chassis dynamometer. This result indicates that the EFs of LDGVs measured in this study generally represent the actual emission level of vehicles in real-world. On the contrary, the EFs of LDGVs in EI guidebook were considerably underestimated. This underestimation would result in the misjudgment of the emission contributions of LDGVs in emission inventories. The main reason for this underestimation could be the overlook of high-emitting vehicles in older vehicle fleet.

In order to alleviate the impact of older vehicles on air quality, some cities in China, like Shanghai and Beijing, have adopted the policies to restrict the driving area of Euro 2 LDGVs and encourage the elimination of Euro 2 LDGVs through fiscal incentives. There measures will be definitely helpful to reduce the high-emitting vehicles. However, some normal-emitting vehicles will be unfairly treated at the same time. Inspection and maintenance (I/M) program for in-use vehicles is another important measure to supervise high-emitting vehicles. In recent years, most of the cities in China have established the ability on emission inspection by simple loaded mode methods for LDGVs. However, the results of this study implies that the existing I/M program has failed to effectively supervise the high emissions from old vehicles. LDGV is the most popular and rapid growing vehicle type in China. Enhancing the supervision of high pollution vehicles and strengthening the compliance tests of in-use vehicles should be the focus of vehicle

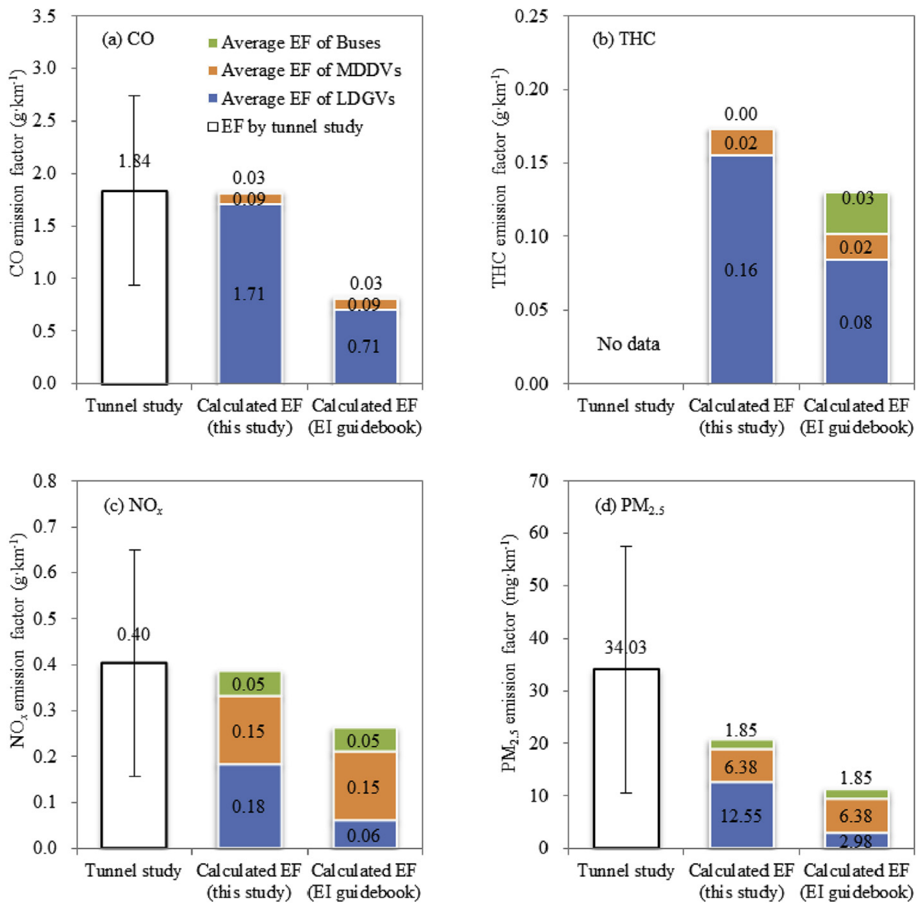

Fig. 8. Comparisons of measured emission factors in East Yan'an Road tunnel with calculated emission factors based on dynamometer tests and EI guidebook. (a) CO; (b) THC; (c) NO_x; (d) $PM_{2.5}$.

pollution control at the present stage.

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