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Real-world gaseous emissions of high-mileage taxi fleets in China

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HIGHLIGHTS

- Real-world emissions from 44 gasoline and 24 bi-fuel taxis were measured using PEMS.
- A major part of high-mileage China 3 and 4 taxis far exceeded the emission limits.
- Switching from gasoline to CNG for bi-fuel taxis increased NO_X and THC emissions.
- Purposed tampering and natural deterioration of TWC converters caused high emissions.
- PEMS measurements show cold start has significant effects on THC and CO emissions.

GRAPHICAL ABSTRACT

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ABSTRACT

Mileage of taxi fleets is significantly higher than regular passenger vehicles in China, which might trigger greater tailpipe emissions of air pollutants. To investigate their real-world gaseous emissions, we applied portable emissions measurement systems (PEMSs) to test 44 gasoline and 24 bi-fuel taxis in seven cities. Our real-world measurement results indicated that a major part of the tested China 3 and China 4 gasoline taxis, especially the samples with high mileage (>300,000 km), far exceeded the corresponding emission limits of NO_X, THC and CO. Only the newest China 5 gasoline taxis with relatively lower mileage had effective emission controls and the gaseous emissions were below the limits. Illegal tampering, malfunction and deterioration of three-way catalytic converters (TWC) are major reasons for high emissions from high-mileage taxis. First, China 4 gasoline taxis without TWC (purposely removed by drivers) increased their gaseous emissions than TWC-equipped counterparts by more than one order of magnitude. Second, bi-fuel taxis when using compress natural gas (CNG) had much higher NO_X and THC emissions than those when using gasoline, which might be probably attributed to unsophisticated engine calibration and unfavorable TWC working conditions. Furthermore, TWC renewal could bring immediate and substantial emission reductions (up to 70%) for high-mileage taxis. However, such benefits from TWC renewal would become less significant as the mileage levels further increase. We also found a good correlation between CO and THC emissions for gasoline taxis, whose cold start effects were both significant. This study poses significant concerns regarding real-world emissions of high-mileage taxi fleets in China,

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which could consist of many gross emitters in the urban areas. Stringent in-use compliance programs and in particular frequent TWC renewals for high-mileage taxis should be implemented by policy makers in China.

1. Introduction

With the rapid growth of transportation demand, vehicle population in China has surged over the past two decades (Wu et al., 2017). Since 2009, China has been the largest vehicle market in the world for the nine straight years (CAAM, 2018). On-road vehicles have been identified as one of the major important sources of air pollution (Cui et al., 2015; Tong et al., 2016b; Franco et al., 2013; Anenberg et al., 2017), which have resulted in significant health impacts (Shindell et al., 2011; Du et al., 2012; Tong et al., 2016a). Among all the vehicle categories, light-duty vehicles (including taxis) are estimated to contribute 67% of total vehicular carbon monoxide (CO) emissions, 58% of total hydrocarbon (THC) and 20% of nitrogen oxides (NO_X) in 2017, reported by the Ministry of Ecology and Environment (MEE, 2018). THC and NO_X are important precursors of secondary particulate matters and ozone formations, both of great concerns by policy makers (Deng et al., 2017; Ke et al., 2017; Li et al., 2018). Beijing's source appointment results indicate that mobile sources, primarily vehicle emissions, were responsible for 45% of ambient $PM_{2.5}$ concentrations among all the local sectors. Particularly, nitrate has become one of the largest aerosol components, accounting for up to 40% of the total $PM_{2.5}$ mass (Beijing Daily, 2018). In addition, vehicular NO_X emissions would lead to urban NO_2 pollution, and the exceedance of ambient NO_2 limit is another challenge to improve air quality in megacities of China (e.g., Beijing) (Cheng et al., 2018).

China has implemented increasingly stringent emission regulations on light-duty passenger vehicles (LDPVs) since 1999. Thanks to the international experiences, China has released the stage 6 standard, i.e., China 6, in which the emission limits of gaseous pollutants (e.g., CO, THC, NO_X) have been lowered by 50%–80% compared with the stage 3 standards (MEP and AQSIQ, 2016). The China 6 standards have not only followed the recent regulation progresses in Europe (e.g., adopting a real driving emissions rule, i.e., the RDE rule) but also introduced a few additional advancements (e.g., fuel and technology neutral perspectives, stricter evaporative emission limits than Euro 6) (Wu et al., 2017). Although average emission factors from LDPVs have overall decreased as emission standards are tightened, serious concerns are attained to high-mileage vehicles (Zhang et al., 2014a; Wu et al., 2016). City taxis are a typical fleet with extremely high annual vehicle kilometers traveled (VKT), up to ten times of that for personal LDPVs, representing greater likelihood of deteriorated emissions than new vehicles (Bishop et al., 2016). To make matters worse, Zheng et al. (2018) have reported that some taxi drivers tempered three-way catalytic converters (TWC) of taxis to gain fuel economy benefits. The taxis that were claimed to meet strict emission standards (China 4 or 5) became high on-road emitters comparable or worse than the pre-China 1 level. Therefore, several megacities (e.g., Beijing, Shanghai) launched special programs that would subsidize in-use taxis to renew TWC periodically (e.g., every 2 years/after 160,000 km of service) (BMEPB, 2016; SMTC, 2015). Although taxis account for a small proportion of total LDPV population (1% in Beijing), high vehicle-use intensity and possible after-treatment tampering of city taxis in China motivated us to conduct a comprehensive investigation of their real-world emissions.

In this study, we employed portable emissions measurement systems (PEMS) to measure on-road gaseous emissions from taxis in China. A total of 68 in-use taxi vehicles from seven cities were recruited, up to now the largest sample size for taxi PEMS studies. In addition to 44 gasoline taxis, 24 taxis were equipped with gasoline-compressed natural gas (CNG) bi-fuel engines. CNG is a welcomed automotive fuel by taxi drivers because of the lower fuel cost than gasoline. Many local governments in China also encouraged usage of natural gas among public fleets, primarily taxis and public buses (Wu et al.,), wishing to reduce air pollutant emissions than liquid fuel powered vehicles. This paper could provide first-hand data regarding emission characteristics of high-mileage taxis.

2. Methodology

2.1. Experimental section

Sixty-eight in-use taxis were recruited from local taxi companies in seven cities, including three provincial-level municipalities (Beijing, Tianjin and Chongqing) and four prefectural-level cities (Langfang, Hebei Province; Yangquan, Shanxi Province; Ji'nan, Shandong Province; Chengdu, Sichuan Province). On-road emission tests were accumulated during 2008 to 2016. The tested vehicles were declared to comply with China 3 to China 5 emission standards (equivalent to Euro 3 to Euro 5) at the new vehicle stage. The tested vehicles included 44 gasoline taxis and 24 bi-fuel (gasoline-CNG) taxis. Table 1 summarizes the vehicle samples in this study, while Table S1 lists the detailed vehicle specification of each sample. In China, emission regulations require manufacturers assure the efficacy of pollutant control devices during their useful life spans (i.e., durability of mileage; e.g., 80,000 km for China 3 LDPVs and 100,000 km for China 4 and China 5 LDPVs) (MEP and AQSIQ, 2007 and 2013). It is worth noting that the accumulated mileages of many tested taxis (see Table S1) were far greater than the durability requirements.

In this study, five taxis (#6, #8, #10, #12 and #13) were recruited for further evaluation of TWC renewal programs: multiple PEMS tests before and after changing new TWCs. Eight taxis (#26, #28, #31, #33, #38, #39, #40, #41) were identified that their TWCs had been purposely removed by the drivers. Generally, we conducted the PEMS test with hot start (i.e., 1-h soaked in ambient condition). In particular, five taxis (#15, #16, #18, #29 and #42) were selected to study their cold-start emissions with soak time >6 h.

Sensor Inc. PEMSs (SEMTECH-DS/ECOSTAR) were applied in this study, which were in compliant with regulation requirements (see Fig. S1 for a schematic diagram). The PEMSs employed the same technologies for gaseous emission measurements, i.e., non-dispersive infrared (NDIR) analyzers for CO and CO_2, heated flame ionization detector (HFID) for THC measurements and non-dispersive ultraviolet (NDUV) module for NO_X (NO and NO_2 separately). To ensure measurement accuracy, the analyzers were zeroed and span calibrated before and after each test (Hu et al., 2012; Zhang et al., 2014b; He et al., 2017). Second-by-second data regarding real-time exhaust flow rates were recorded via a 2-inch exhaust flow meter (SEMTECH-EFM, Sensors Inc.). Instantaneous vehicle speed and location information were collected by using Global Positioning System (GPS) data loggers, which were incorporated into the PEMSs.

The test routes were composed of arterial roads, sub-arterial roads and freeways, representing the typical driving routes of city taxis (see Fig. S2). The trip distance of each test for each taxi was over 20 km to assure adequate real-world driving data and emission profiles (see Table S2). All drivers were instructed to maintain their usual driving behaviors during the test trips. In accordance with the requirement of the regulations regarding real-world emissions measurement (MEP and AQSIQ, 2016), market fuel from the certified refueling stations was used. The gasoline fuels complied with China 2 to China 5 standards and CNG complied with GB 18047-2000 standard.

Table 1
Summary of tested taxis in this study.

Vehicle categories	Emission standard	Number of vehicles	Accumulated mileage (10^3 km)	Curb mass (kg)	Power rating (kW)	Engine displacement (L)
Gasoline taxi	China 5	3	194.8–246.9	1240	82	1.6
	China 4	27	14.1–832.6	1240	82	1.6
	China 3	14	75.6–560.7	1105–1265	70–82	1.6–2.0
Bi-fuel taxi (gasoline-CNG)	China 5	3	45.0–86.2	1150–1343	82–88	1.6
	China 4	4	67.4–548.0	1120–1150	81–88	1.6
	China 3	17	10.2–239.0	995–1475	64–94	1.5–1.8

2.2. Data analysis

As Table S2 illustrates, driving conditions differed greatly among various on-road tests. We applied an operating mode binning method to eliminate the impact from distinctive driving conditions (Zhang et al., 2014b). In this study, we classified 28 bins representing instantaneous operating modes, including one deceleration/breaking bin (Bin 0), one idling bin (Bin 1), and 26 cruise/acceleration bins (Bins 11 to 3Y) that were defined by vehicle specific power (VSP) and speed (see Table S3). VSP is a proxy parameter of instantaneous power output per vehicle mass, considering integrated effects of gravity, rolling resistance and aerodynamic drag. In this study, a simple equation provided by Jiménez-Palacios (1999) was used to calculate VSP of taxis, as shown in Eq. (1).

$$VSP = v \times (1.1 \times a + 9.81 \times \sin\theta + 0.132) + 3.02 \times 10^{-4} \times v^3 \qquad (1)$$

where VSP is estimated vehicle specific power, kW/t; v is instantaneous vehicle speed, m/s; a is instantaneous vehicle acceleration, m/s^2; and θ is roadway grade, radians.

Average emission rates of CO, THC, NO_X and CO_2 for each operating mode were estimated based on their second-by-second emission profiles with Eq. (2).

$$\overline{ER_{i,j}} = \frac{1}{T_i}\sum_{t=1}^{T_i} ER_{i,j,t} \qquad (2)$$

where $\overline{ER_{i,j}}$ is the average emission rate of pollutant j for a tested vehicle in operating mode bin i, g/s; T_i is the number of seconds for a tested vehicle in operating mode bin i, s; and $ER_{i,\,j,\,t}$ is the real-time emission rate of pollutant j for a tested vehicle in operating mode bin i at second t, g/s.

Distance-based emission factors were further calculated based on the average emission rates and time allocation of operating mode bins within a driving cycle, illustrated Eq. (3).

$$EF_j = \frac{3600 \times \sum_i \left(\overline{ER_{i,j}} \times p_i \times T\right)}{\sum_t v_t} \qquad (3)$$

where EF_j is the distance-based emission factor for a tested vehicle of pollutant j, g/km; p_i is time allocation of operating mode bin i to the total real-world test for each tested vehicle; T is the total time of each individual on-road test, s; v_t is the instantaneous vehicle speed at t second of each test, km/h.

Eq. (4) illustrates the method to normalize emission factors of all tested vehicles to a baseline traffic pattern.

$$nEF_j = \frac{3600 \times \sum_i \left(\overline{ER_{i,j}} \times np_i\right)}{n\overline{v}} \qquad (4)$$

where nEF_j is the normalized distance-based emission factor of pollutant j under a baseline traffic pattern, g/km; np_i is the time allocation of operating mode bin i to the baseline typical driving pattern (see Fig. S3) based on previous on-road tests in China (Zhang et al., 2014b; Wu et al., 2015); $n\overline{v}$ is the average vehicle speed of the baseline driving pattern, 32 km/h. We acknowledge that various operating mode binning and VSP calculation methods have been used in existing studies. Zhang et al., 2014b has discussed that the inconsistency of defining operating modes between Eq. (2) vs. Eqs. (3) and/or (4) would lead to important errors, rather than the operating mode binning methodologies themselves.

We followed the RDE rule metric to calculate the conformity factors (CFs) of gaseous air pollutants for each tested taxi (see Eq. (5)). The regulatory limits of emission standards for spark-ignition engines (see Table S4) served as the baselines for both gasoline and bi-fuel taxis.

$$CF_j = \frac{nEF_j}{EL_j} \qquad (5)$$

where CF_j is the CF of pollutant j; EL_j is the applicable emission limit of pollutant j, g/km.

3. Results and discussion

3.1. Overview of gaseous emission factors

Fig. 1 shows the average CFs of gaseous pollutants from gasoline powered taxis (including bi-fuel taxis when using gasoline). The original and normalized emission factors of all the tested vehicles are listed in Table S2. The CF values of NO_X, THC, and CO emissions depict an ascending uptrend with the accumulated mileage; however, such trend is not identified from our CO_2 results (see Fig. S4). In our study, 64.7%, 33.8% and 52.9% of gasoline taxis ($N = 68$) are higher than the corresponding emission limits of NO_X, THC and CO, respectively. China 5 taxis could control their on-road emission factors below the regulatory limits, whose accumulated mileages were no higher than the three times of the regulated useful life (100,000 km). By contrast, 80.6% of China 3 and China 4 taxis could not control the emission factors below the emission standards, among which a few vehicle samples even with relatively low mileages (<300,000 km) significantly exceeded the emission limits. In this study, the highest CF values of NO_X, THC, and CO are 35.1, 15.3 and 16.6, respectively. We noticed that the durability of mileage has been extended to 160,000 km in the future China 6 standard (MEP and AQSIQ, 2016). Considering the extreme high mileage of city taxi fleets, we suggest that frequent in-use compliance inspections should be conducted for high-mileage vehicles, such as further RDE inspections during the extended useful life (e.g., beyond 160,000 km).

Compared with the current official emission model in China, namely the National Emission Inventory (NEI) Guidebook (on-road transportation chapter), the actual emissions for these high-mileage taxis would be much higher than the official estimates (MEP, 2015; Wu et al., 2017). This is because the NEI Guidebook considers that the emission deterioration with mileage would be not significant after the accumulated mileage exceeds 200,000 km (Zhang et al., 2014a; MEP, 2015). However, the on-road measurements in this study suggest that revisits and corrections of the deterioration module in the NEI Guidebook are needed for high-mileage fleets (see Fig. 1).

Fig. 2 presents the correlations between the NO_X, CO and THC emissions for taxis organized by various fuel (i.e., gasoline and CNG). A good

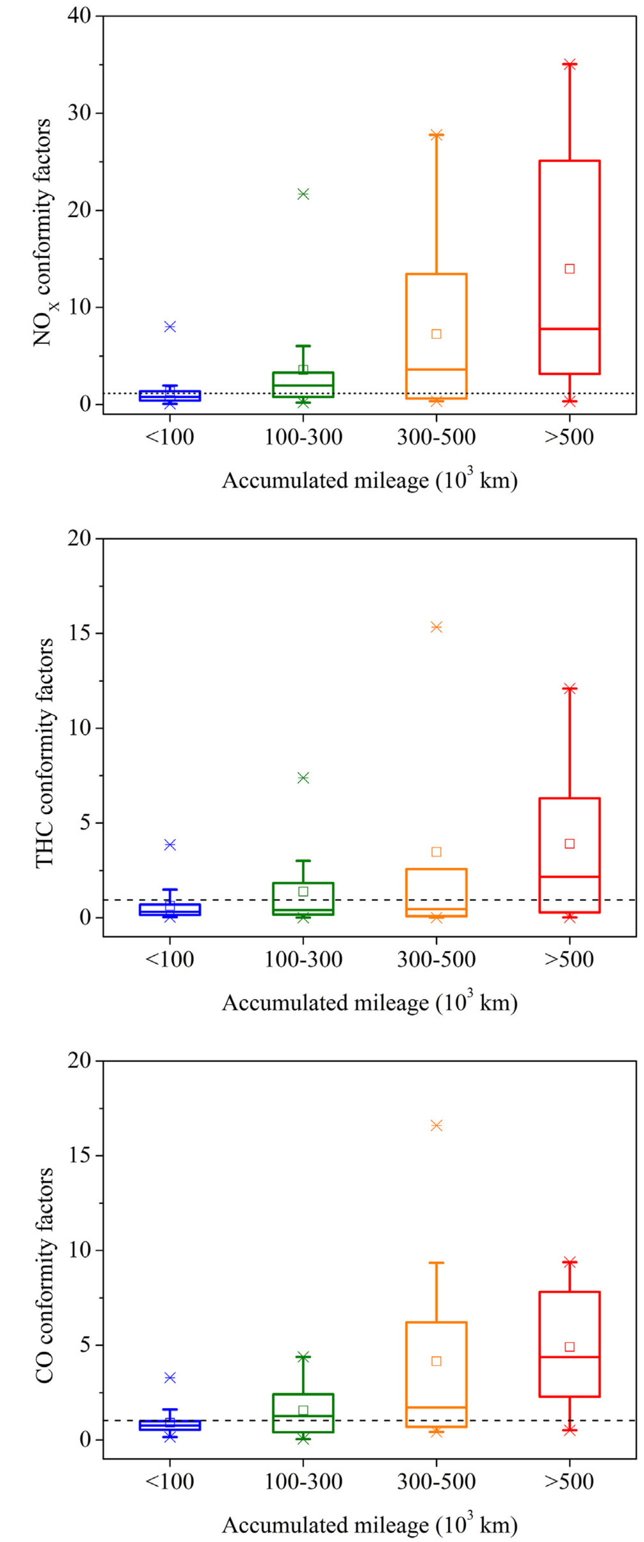

Fig. 1. CF values of NO_X, THC and CO emissions from gasoline powered taxis. The boxes show the 25th and 75th percentiles, and whiskers indicate 5th and 95th percentiles. Median is indicated by the line inside the box, and the mean is indicated by the square.

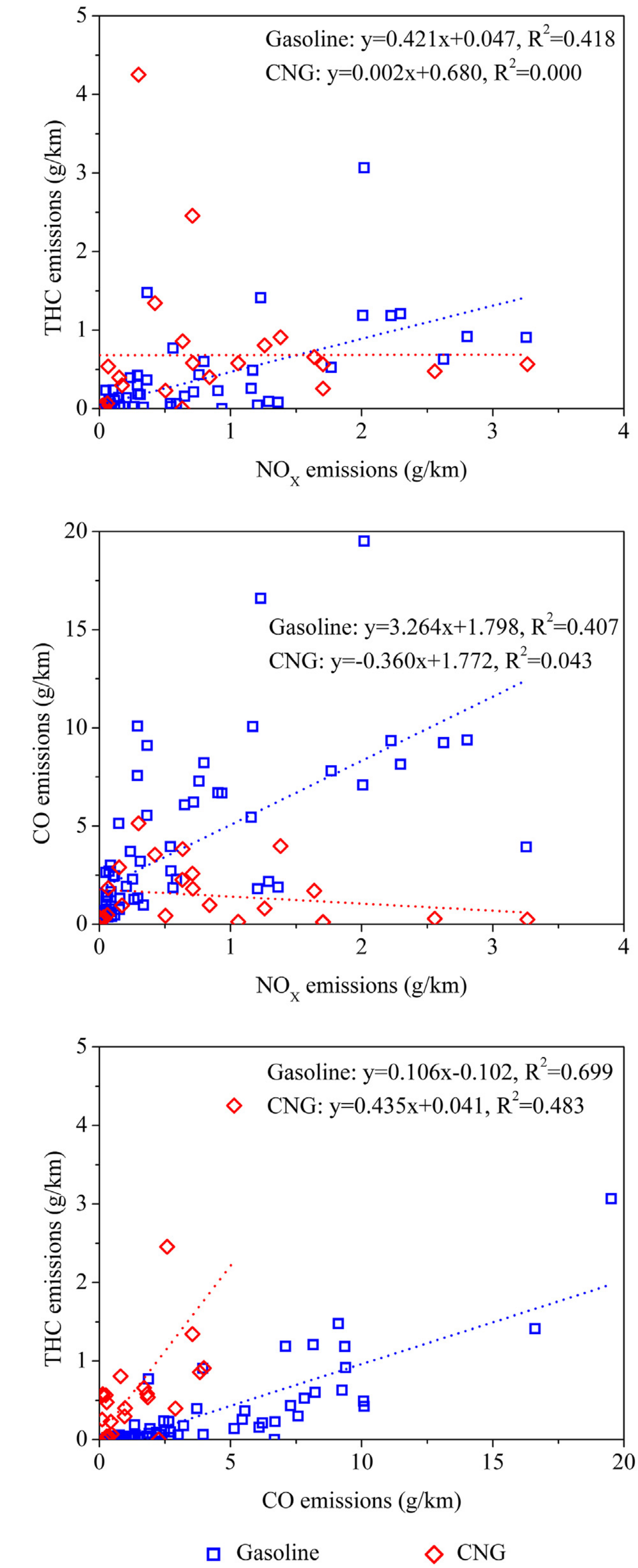

Fig. 2. Correlations between NO_X, CO and THC emissions for 44 gasoline taxis and 24 bi-fuel taxis.

correlation is identified between CO and THC ($R^2 = 0.7$) for the 68 gasoline taxis (including 24 bi-fuel taxis when using gasoline). This result is quite consistent with the another PEMS study (59 gasoline vehicles) reported by the Intentional Council on Clean Transportation (ICCT) (Yang, 2018), although the THC to CO ratios were slightly dissimilar. The reason might be that Yang (2018) involved not only public taxis but also private cars in the data samples. In contrast, less significant correlations between NO_X and CO (or THC) ($R^2 = \sim 0.4$) were observed. Compared with gasoline taxis, we found weaker correlations for CNG taxis ($R^2 = 0.48$ for THC-CO, $R^2 = 0.00$ for THC-NO_X and $R^2 = 0.04$ for CO-NO_X, $N = 24$). The results indicate that emission exceedances of NO_X and CO/

THC could be caused by various reasons for CNG-fueled vehicles, which will be detailed discussed later.

The China 6b RDE rule has specified mandatory emission limits for NO_X and particle number (PN), with CF values of 2.1. CF values of CO emissions shall be reported by manufactures in China 6. THC or non-methane hydrocarbon (NMHC) emissions, however, have not been regulated yet due to the safety in using HFID. We suggest that a mandatory limit of CO emissions be proposed to better control THC emissions for high-mileage gasoline vehicles, given the strong correlations between THC and CO emissions for gasoline cars.

3.2. Impact of after-treatment devices

TWC converters are highly efficient to eliminate engine-out gaseous pollutants for gasoline vehicles. Natural deterioration, malfunction and illegal tampering of TWC converters are all likely to cause high emissions. In China, some taxi drivers would purposely remove TWC converters from the taxi tailpipes in order to reduce fuel consumption. Fig. 3 compares the average CF values for China 4 taxis with and without TWC. The CF values of NO_X, THC and CO from TWC-equipped taxis ($N = 23$) were 2.14 ± 2.98, 0.30 ± 0.43 and 1.54 ± 1.71. Compared with the TWC-equipped taxis with TWC ($N = 23$), the CF values of taxis without TWC ($N = 8$) were much higher, i.e., 25.2 ± 7.52 for NO_X, 9.16 ± 4.05 for THC and 9.14 ± 3.30 for CO. Our results did show higher on-road CO_2 emissions for TWC-equipped taxis, confirming the motivation to remove after-treatment devices. Thus, taxis in association of TWC tampering could be important high emitters, and stringent in-use programs should be implemented for city taxis.

Moreover, it is equally essential to maintain good working performances of TWC converters for city taxis. Given the high vehicle-use intensity, a regular TWC replacement program for in-use taxis has been launched in Beijing (BMEPB, 2016). As Fig. 4 illustrates, the average CF values of NO_X, THC and CO were reduced by 71% ± 27%, 68% ± 21% and 74% ± 20% respectively, after new TWC converters were just installed. Except for NO_X emissions from #8 taxi, the new TWC converters were operating effectively in reducing gaseous emission factors below regulatory limits for all the 5 taxis. Nevertheless, significant emission deteriorations were observed after they traveled >100,000 km since the TWC renewals. Then NO_X emissions for #6 taxi and CO and THC emissions for #8 taxi even exceeded their pre-TWC levels. In China, annual VKT of city taxis would be higher than 100,000 km (Zhang et al., 2014a). Therefore, our results suggest TWC renewal should be conducted at least once per year when taxis exceed their durability of mileage.

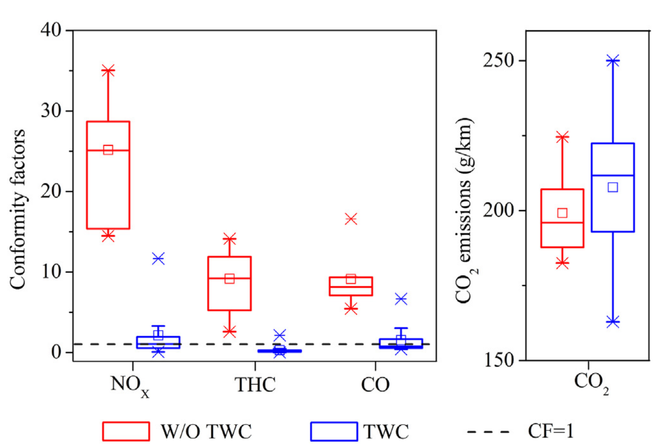

Fig. 3. CF values of NO_X, THC and CO, and emission factors of CO_2 from 31 China 4 taxis with TWC ($N = 23$) and without TWC ($N = 8$). The boxes show the 25th and 75th percentiles, and whiskers indicate 5th and 95th percentiles. Median is indicated by the line inside the box, and the mean is indicated by the square.

3.3. Impact of fuel type for bi-fuel engine taxis

Table S2 presents detailed distance-specific gaseous emissions of bi-fuel taxis (i.e., #45 to #68) when using different fuels (gasoline vs. CNG). Average emissions of NO_X, THC and CO are 0.94 ± 0.86 g/km, 0.77 ± 0.96 g/km and 1.63 ± 1.52 g/km for China 3 and China 4 CNG powered taxis ($N = 21$). They are significantly higher than those of the China 5 CNG powered taxis ($N = 3$) by 21.8, 19.5, and 4.40 times, respectively. Similar to bi-fuel taxis using CNG, gasoline powered China 3 and China 4 taxis ($N = 21$) emitted higher than China 5 taxis ($N = 3$) by 15.3 times for NO_X, 9.86 times for THC, 7.60 times for CO, respectively.

Based on the comparative experimental results between CNG and gasoline for the bi-fuel taxis, we found using CNG fuels significantly increased NO_X and THC emissions compared with the levels of using gasoline. Fig. 5 presents NO_X and THC emissions when using CNG increased by 325% ± 330% and 314% ± 395% relative to gasoline powered emissions, which far exceeded the regulatory limits. On the other hand, CNG-fueled tests could reduce CO and CO_2 emission factors by 40.9% ± 56.1% and 22.8% ± 9.1% than gasoline-fueled tests. Some of these bi-fuel taxis were retrofitted from conventional gasoline vehicles, and engine calibrations were not carefully conducted according to varying fuel properties (e.g., octane number, heat value). Consequently, engine and after-treatment devices (i.e., TWC) that were originally designed for gasoline vehicles could not work in optimum conditions when using alternative fuels (Nylund and Lawson, 2000; Wu et al., 2004). We calculated the average lambda values (λ) representing air-fuel ratios of all the bi-fuel taxis based on the ISO 16183 equations (ISO, 2002). The average λ of CNG-fueled (1.25 ± 0.16) and gasoline-fueled ($\lambda = 1.23 \pm 0.22$) tests both indicated that the engines might work on lean-burn rather than stoichiometric conditions probably due to switch in real service, leading to poor TWC efficiency. On the other hand, CH_4, one of the most stable hydrocarbons, could be a major component of increased THC emissions from CNG-powered vehicles. And higher combustion temperature of CNG fuels favors the thermal NO_X formation (i.e., through Zeldovich mechanism) than gasoline vehicles. Therefore, switching gasoline to CNG could not deliver reductions of NO_X and THC emissions but significant increases. Such fuel switching programs should be of great cautious, and careful calibrations of engines and after-treatment devices should be required to achieve joint benefits in mitigating greenhouse gases (CO_2 and CH_4) and air pollutants (NO_X).

3.4. Impact of cold start emissions

Previous PEMS studies in China often excluded cold start emissions, as vehicles entered the tests with the engines warmed up. However, Wu et al. (2016) noted total vehicle emissions of CO and THC could be underestimated if on-road measurement results are directly applied to develop emission inventories. In-lab dynamometer measurements reported dramatically higher emissions of gaseous pollutants and black carbon due to cold start compared with hot running emissions (Clairotte et al., 2013; Zhu et al., 2016; Zheng et al., 2017; Khan and Frey, 2018; He et al., 2018). As Fig. S5 indicates, our PEMS measurements including cold-start emissions for 5 selected taxis (#15, #16, #18, #29 and #42) confirm the significant contributions of cold-start CO and THC emissions from a real-world perspective.

According to Fig. S5, we consider cold-start emissions only occurred during the initial 300 s. Average emission factors of THC and CO with cold starts (i.e., the initial 300 s) were 15.8 ± 13.2 times and 15.9 ± 11.5 times higher than hot-start emission factors for the 5 taxis (see Table S5). In contrast, the cold-start effect on NO_X emissions is not significant, similar to previous findings Weilenmann et al. (2009). To further quantify the start effect, the ratios of extra start emissions (see Eq. (6)) to the hot running emissions per km, γ (see Eq. (7)), were estimated, representing the distance would need to travel between starts

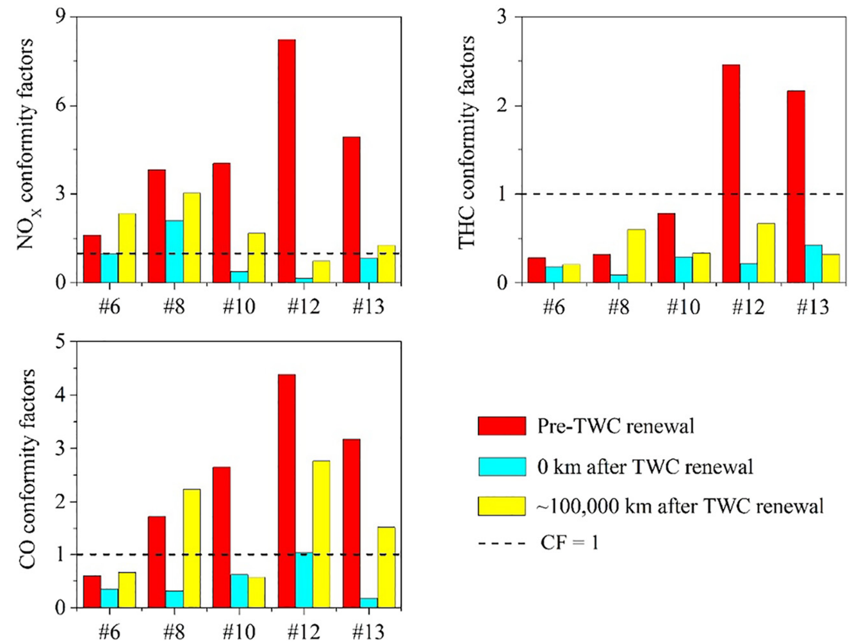

Fig. 4. CF values of NO_X, THC and CO emissions from the five in-use taxis before and after the TWC renewal.

before hot-running emissions would exceed start emissions (Drozd et al., 2016; Zheng et al., 2017).

$$E_{start} = E_{300s}\text{-}EF_{hot} \cdot D_{300s} \tag{6}$$

$$\gamma = \frac{E_{start}}{EF_{hot}} \tag{7}$$

where E_{start} is extra emissions due to vehicle start, g; E_{300s} is the total emissions during the initial 300 s, g; EF_{hot} is the hot-running emission factor after the initial 300 s, g/km; and D_{300s} is the driving distance during the initial 300 s. For each vehicle, EF_{hot} is normalized according to the operating mode binning method.

The results indicate that the γ values can vary by start conditions, vehicles and pollutants (see Fig. 6). For example, estimated γ values under cold start are 173.5–390.7 km for THC and 26.8–116.5 km for CO, much higher than those under hot start in this study. Average γ values for THC (289.6 km) and CO (55.9 km) under cold start are above the average total daily mileage among China's passenger car drivers (Zhang et al., 2014a; Wu et al., 2016). It turns out that the majority of the THC and CO total emissions in real-world driving come from cold-start extra emissions. Unlike THC and CO, average γ value of NO_X is much lower for both cold starts and hot starts.

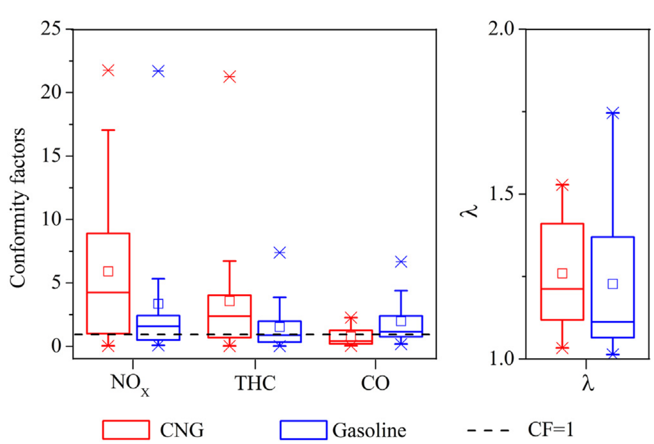

Fig. 5. CF values of NO_X, THC and CO emissions from 24 bi-fuel taxis. The boxes show the 25th and 75th percentiles, and whiskers indicate 5th and 95th percentiles. Median is indicated by the line inside the box, and the mean is indicated by the square.

4. Conclusion

Taxi fleets have much higher mileage traveled than regular vehicles in China, posing substantial concerns regarding their real-world emissions of air pollutants. This study applied PEMS to measure on-road emissions of gaseous pollutants for 44 gasoline taxis and 24 gasoline-CNG bi-fuel taxis from seven cities in China. Our measurement results indicate that on-road emissions of NO_X, THC and CO for the major part of high-mileage gasoline taxis (>300,000 km) far exceeded the emission limits. Notably, the conformity factors of NO_X emissions, defined as the ratios of real-world emission factor to regulatory limit, could be up to 35 for China 4 taxis with extremely high mileage (>500,000 km). In terms of the bi-fuel taxis, we found that using CNG fuels significantly increased NO_X and THC emissions by up to 325% ± 330% and 314% ± 395% than using gasoline, although the CO_2 and CO emissions decreased.

TWC converters play an essential role in controlling gaseous emissions from gasoline taxis. This study found that taxis without TWC converters, which had been purposely removed by drivers to improve fuel economy, emitted more NO_X, THC and CO than TWC-equipped taxis by up to one order of magnitude. Thus, we recruited five China 3 taxis in a TWC renewal program. Using new TWC converters helped reduce gaseous emissions of these taxis by approximately 70% compared with the pre-TWC renewal levels. However, the emission control benefits became less substantial as emission deteriorations with mileage still appeared. This study also found considerable CO and THC emissions due cold starts. To sum up, high-mileage taxis have substantial risks of exceeding emission limits. Switching gasoline engines to bi-fuel engines would not receive emission control benefits as expected, and for CNG-fueled taxis careful engine calibration is needed. Periodical TWC renewal (e.g., at least annually) would be an effect option to control emissions for high-mileage taxis.

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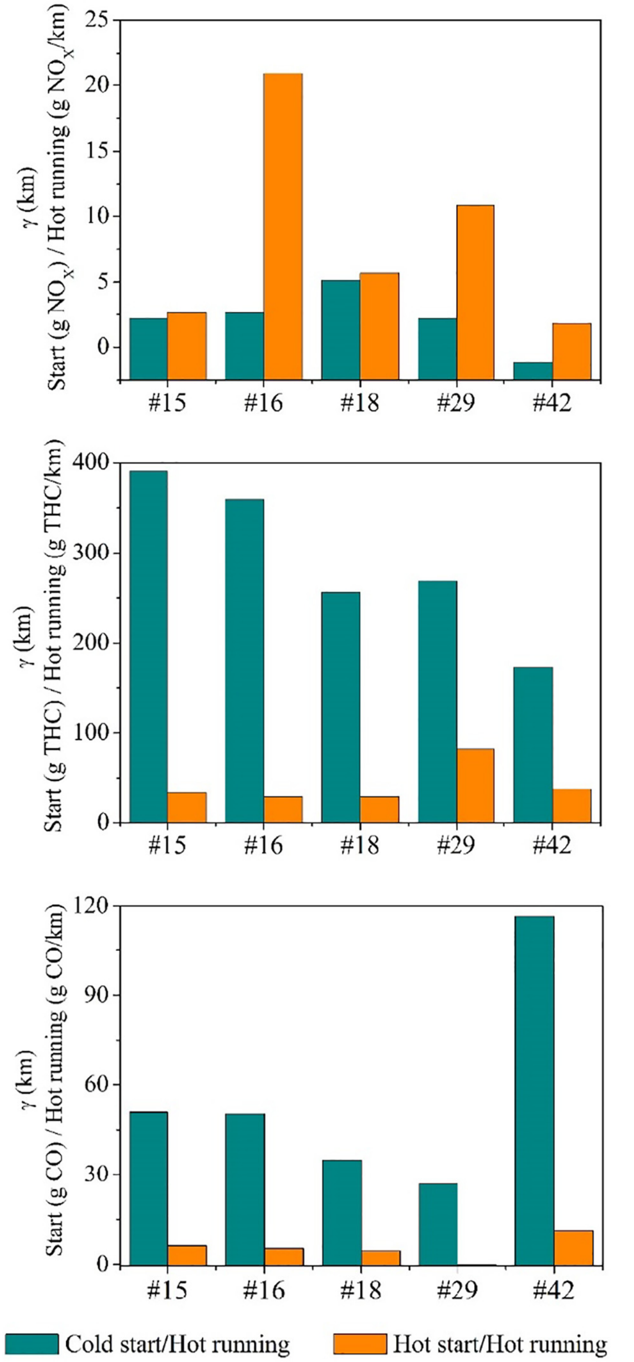

Fig. 6. The ratios of extra start emissions (initial 300 s) to the hot running emission factors.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2018.12.336.

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