Emission Measurement of Diesel Vehicles in Hong Kong through On-Road Remote Sensing: Performance Review and Identification of High-Emitters

Yuhan Huang[1], Bruce Organ[1,2], John L. Zhou[1,*], Nic C. Surawski[1], Guang Hong[3], Edward F. C. Chan[2], Yat Shing Yam[4]

[1]School of Civil and Environmental Engineering, University of Technology Sydney, NSW 2007, Australia

[2]Jockey Club Heavy Vehicle Emissions Testing and Research Centre, Vocational Training Council, Hong Kong

[3]School of Mechanical and Mechatronic Engineering, University of Technology Sydney, NSW 2007, Australia

[4]Environmental Protection Department, The Government of the Hong Kong Special Administrative Region, Hong Kong

Corresponding author:

Professor John L. Zhou

Email: junliang.zhou@uts.edu.au

Abstract

A two-year remote sensing measurement program was carried out in Hong Kong to obtain a large dataset of on-road diesel vehicle emissions. Analysis was performed to evaluate the effect of vehicle manufacture year (1949-2015) and engine size (0.4-20 L) on the emission rates and high-emitters. The results showed that CO emission rates of larger engine size vehicles were higher than those of small vehicles during the study period, while HC and NO were higher before manufacture year 2006 and then became similar levels between manufacture years 2006 and 2015. CO, HC and NO of all vehicles showed an unexpectedly increasing trend during 1998-2004, in particular ≥6001cc vehicles. However they all decreased steadily in the last decade (2005-2015), except for NO of ≥ 6001cc vehicles during 2013-2015. The distributions of CO and HC emission rates were highly skewed as the dirtiest 10% vehicles emitted much higher emissions than all the other vehicles. Moreover, this skewness became more significant for larger engine size or newer vehicles. The results indicated that remote sensing technology would be very effective to screen the CO and HC high-emitters and thus control the on-road vehicle emissions, but less effective for controlling NO emissions. No clear correlation was observed between the manufacture year and percentage of high-emitters for ≤ 3000cc vehicles. However, the percentage of high-emitters decreased with newer manufacture year for larger vehicles. In addition, high-emitters of different pollutants were relatively independent, in particular NO emissions, indicating that high-emitter screening criteria should be defined on a CO-or-HC-or-NO basis, rather than a CO-and-HC-and-NO basis.

Keywords: Real-world emissions; Diesel vehicles; Engine size; High-emitters; On-road remote sensing

1. Introduction

Vehicle emissions are often believed to be the single largest contributor of atmospheric pollutants (Franco et al., 2013; Liu et al., 2017; Ropkins et al., 2009). To reduce vehicle emissions and improve air quality, new model vehicles are required to comply with the ever tightening emission standards through laboratory testing for type approval, such as the New European Driving Cycle (NEDC). However, the expected reduction in the NO_2 concentration at European roadside monitoring sites was not observed with the more stringent standards (Carslaw et al., 2011). In recent years, increasing evidence has been reported on the significant gap in the emissions performance between laboratory testing and real-world driving, in particular diesel vehicles.

Portable emissions measurement system (PEMS) can be used to investigate vehicle emissions performance under real-world driving conditions. Weiss et al. (2011) investigated the on-road emission rates of twelve Euro 3-5 light-duty vehicles using PEMS. The results showed that CO, HC and NOx of gasoline vehicles and HC and CO of diesel vehicles were generally below the emission limits. However, NOx of diesel vehicles exceeded the limits by 320±90% and CO_2 surpassed the laboratory levels by 21±9%. Kousoulidou et al. (2013) tested the emission rates of six Euro 3-5 vehicles under real-world driving and NEDC type-approval conditions using PEMS. They found that gasoline vehicle emissions were well below the emission standards. However, NOx of diesel vehicles complied with the emission limits under NEDC conditions but constantly exceeded the limits under real-world driving conditions. Fu et al. (2013) assessed the NOx emissions of two Euro IV buses by PEMS. Their results indicated that NOx emission factors were 2.6-2.8, 2.3-2.7 and 2.2-2.3 times higher than the emission limits for urban, suburban and freeway driving, respectively. Degraeuwe and Weiss (2017) analysed the PEMS on-road driving emissions of seven Euro 4-6 diesel cars. It was found that the median NOx emissions of NEDC-matched conditions exceeded the limit by 206%, while NOx of all on-road conditions exceeded the limit by 266%. This implied that the narrow NEDC test conditions might be only responsible for part of the elevated on-road diesel NOx emissions.

The gap between real-world driving and type-approval emissions was commonly believed to be increasing with time and the factors responsible for this gap included driving behaviours, vehicle configurations, traffic conditions, road grade and weather which were not well considered in the laboratory testing (Fontaras et al., 2017). Therefore, understanding vehicle emissions under real-world driving conditions are critical to address this gap. Although PEMS can measure a long series of emissions data under various real-world driving conditions with acceptable accuracy, the long turnover time of PEMS testing limited its application for a large number of vehicles (Lau et al., 2015) (e.g. ranged from 2 to 12 in above reviewed studies) and thus might not be able to represent a full picture of the on-road vehicles emissions. Additionally, the extra weight of PEMS may bias the measurements, especially for light vehicles (Weiss et al., 2011).

On-road remote sensing provides a non-intrusive method to measure vehicle emissions in a large scale at a relatively low cost. It has been widely used to monitor and control the on-road vehicle emissions. An early long-term remote sensing study showed that CO, HC and NO emissions had decreased significantly over the period of 1997-2007 and the trend was not detailed for gasoline or diesel vehicles (Bishop and Stedman, 2008). However, recent remote sensing studies have identified different emission trends of gasoline and diesel vehicles. Chen and Borken-Kleefeld (2014) measured the emissions of light duty vehicles at one site in Zurich during 2000-2012. The results showed that diesel NOx emissions [g/kg fuel] had actually increased although emission limits had been progressively tightened. However, this discrepancy was not observed for other emissions or gasoline vehicles. Carslaw et al. (Carslaw et al., 2011; Carslaw and Rhys-Tyler, 2013) measured the NO and NO_2 emissions of on-road vehicles using remote sensing in London. They found that only gasoline vehicles showed reduction in NOx/CO_2 over the period of 1985-2012, while diesel vehicles, including those with after-treatment systems designed to reduce NOx, showed little evidence of NOx/CO_2 reduction. Pujadas et al. (2017) investigated the real-world driving emissions of passenger cars in Spain. Their results showed that CO/CO_2, HC/CO_2 and NO/CO_2 of gasoline vehicles and CO/CO_2 and HC/CO_2 of diesel vehicles were decreasing from pre-Euro to Euro 6 standards, while no NO/CO_2 reduction was observed for diesel vehicles during the same period.

Research into remote sensing in Hong Kong began in 1993 and it has been used for various applications. Chan et al. used remote sensing to develop CO, HC and NO emission factors for gasoline (Chan et al., 2004), diesel (Chan and Ning, 2005) and liquefied petroleum gas (LPG) (Ning and Chan, 2007) vehicles. Lau et al. (2012) used remote sensing to monitor the vehicle emission trends in Hong Kong. It was found that CO, HC and NO emissions [g/km] of gasoline and LPG vehicles were continuously decreasing over the period of 1999-2008, while CO and NO emissions of diesel vehicles increased during 2004-2008. From 1 September 2014, the Hong Kong Environmental Protection Department (HKEPD) started using remote sensing as a legislative tool to detect high-emitting vehicles for enforcement purposes (HKEPD, accessed 18.05.2017). The high-emitters detected will be issued with an Emissions Test Notice (ETN) and are required to have the vehicles serviced/repaired and tested at an authorised emissions testing centre within 12 working days. If a vehicle failed the test, the licence would be cancelled and the vehicle would be removed from the road. However, this enforcement programme is currently only applied to gasoline and LPG vehicles, while further investigation is needed to extend this programme to diesel vehicles.

The above reviewed remote sensing studies had revealed an unexpected emission trend of diesel vehicles. However, a main limitation was that they were mostly for passenger cars and light commercial vehicles, and the number of diesel emission records in these studies were generally small. Since remote sensing only measures the snapshot emissions of a vehicle in a half second, a large sample size is needed to investigate the average emission trends accurately.

The focus of this paper is to investigate the recent emission trends of diesel vehicles and to identify the potential high-emitters based on on-road remote sensing measurement. The contribution of this study lies in the following three aspects. Firstly, we analysed a sample of 417714 records of on-road diesel vehicle emissions from two years of continuous measurement using remote sensing technology in Hong Kong. The large sample size in this study is believed to present the diesel emission trends in a more statistically accurate manner. Secondly, this study extends the time period and vehicle class covered. The manufacture year of vehicles ranged from 1949 to 2015 and the engine size varied from 0.4 to 20 litres. This large unique database allowed us to analyse the effect of manufacture year (emission standard) and vehicle category (engine size) on emissions. Finally, the emission characteristics of this study could be a good reference for setting up the high-emitting cutpoints of diesel vehicles in Hong Kong (Borken-Kleefeld, 2013) and thus help extend the enforcement programme to all the vehicles on road (HKEPD, accessed 18.05.2017).

2. Methods and Data

2.1. Remote sensing setup

In this study, 14 sets of remote sensing equipment were used to collect the data. The measurements were taken at 158 sites across Hong Kong by the HKEPD from 2 January 2014 to 26 January 2016, with 40 sites in Hong Kong Island, 36 sites in Kowloon, 81 sites in New Territories and 1 site in Lantau. Fig. 1 shows the setup of one remote sensing measurement site. A measurement site should be a 5m-width single lane with slight uphill gradient so vehicles are under constant gravity pull and away from traffic lights or intersections to avoid off cycle emissions from hard acceleration/deceleration. The site should also have sufficient traffic volume and vehicle speeds in the range of 7-90 km/h for repeatable measurements. Thousands of measurements were needed to develop and determine the site profile. This information was used to determine and validate the locations of the remote sensing units, cameras and support equipment. Two remote sensing units were placed in one measurement site with an approximately 1s separation distance. The second unit was to confirm the measurement consistency and repeatability. When the equipment was powered up and ready, a reference span gas was used to confirm that the units were operating according to the performance specifications. During the measurements, additional calibration checks were performed every two hours with the span gas to ensure results were not drifting outside of the measurement specification limits and the speed profile of vehicles was checked against the site reference information. The data from the previous two hours' measurements was considered valid if these checks were passed.

The ETC-S420 remote sensing system was used to collect the vehicle emissions data, with accuracy of ±15% of the readings. The system consisted of non-dispersive infrared (NDIR) and ultraviolet (NDUV) sources, detectors, a retroreflector, speed and acceleration sensors and a vehicle plate camera. The NDIR and NDUV beam sources and detectors were placed together on one side of the road and a retroreflector was placed on the other side of the road to reflect the beam from the sources to detectors. The measurement was triggered by the beam being blocked by a passing vehicle. CO, CO_2 and HC emissions were measured in the IR region and NO emissions were measured in the UV region. Meanwhile, the speed, acceleration and licence plate number of the passing vehicle were also measured and recorded. The registration information of the passing vehicle could be obtained from the vehicle plate number, including the make, manufacture year, engine size, fuel type and license class (as defined in Table 1).

2.2. *Data treatment and sampled fleet characteristics*

Since the effective plume path length and the amount of plume measured were influenced by a number of factors such as wind, turbulence, engine size and exhaust pipe height, remote sensing system could only determine the relative concentration ratios of pollutants over CO_2 (denoted by Q_P hereafter where P could be CO, HC or NO). For a given exhaust plume, these concentration ratios were constant (Bishop et al., 1989; Burgard et al., 2006). The emission concentrations in percentage (%) or part per million (ppm), which were the output of the remote sensing system, could be calculated based on a key assumption that the engine was running stoichiometric or rich with no excess oxygen in the exhaust. This was true for conventional gasoline vehicles but not for diesel vehicles. The original remote sensing data showed that most of the CO_2 concentrations were in the range of 14.5-15.0% which was the CO_2 concentration of stoichiometric combustion. This indicated that the calculations of absolute emission concentrations were not suitable because diesel engines were mostly operated under lean conditions, even at full load operation (Heywood, 1988). Therefore the emissions data were back calculated to the originally measured emission ratios (Q_P) and then converted to emission factors in [g/kg fuel] by Eqs. (1)-(3), which removed the assumption for calculating the absolute emission concentrations.

$$EF_{CO} = \frac{28}{0.014} * \frac{Q_{CO}}{1+Q_{CO}+6Q_{HC}} \text{ [g/kg fuel]} \qquad (1)$$

$$EF_{HC} = \frac{2*44}{0.014} * \frac{Q_{HC}}{1+Q_{CO}+6Q_{HC}} \text{ [g/kg fuel]} \qquad (2)$$

$$EF_{NO} = \frac{30}{0.014} * \frac{Q_{NO}}{1+Q_{CO}+6Q_{HC}} \text{ [g/kg fuel]} \qquad (3)$$

The two-year continuous measurements obtained 417714 records of diesel vehicle emissions with matched licence plate number information. Since remote sensing was not weather proof, the number of measurements per month ranged from 1542 (in 9 days of May 2014) to 7337 (in 21 days of March 2015). The number of measurements per day was low because diesel vehicles only accounted for a small percentage (18.5%) of the total vehicle fleet in Hong Kong (Transport Department of Hong Kong, 2017). This large dataset covered a wide range of on-road diesel vehicles, with manufacture year varying from 1949 to 2015 and engine size from 0.4 to 20 litres. A measurement was considered valid when the vehicle was running at steady speed or positive acceleration and the measured CO_2 exhaust plume size was sufficient to determine the emission ratios (Carslaw et al., 2011; Chen and Borken-Kleefeld, 2014; 2016). In addition, the vehicle speed was limited up to 75 km/h to avoid irregular off-cycle high emission events, which was the speed envelope of the Hong Kong transient emissions testing (HKTET) for emission certificates (Commissioner for Transport, 2012). Among the 417714 records obtained, 105627 records were invalid due to insufficient exhaust plume size (no CO_2 reading) and 212072 records were invalid due to speed or acceleration criteria. Finally, 161769 (39%) valid measurements were remained, which still represented a large sample of the on-road diesel fleet. As shown in Table 1, the total number of licensed diesel vehicles in Hong Kong was 138555 by April 2017 (Transport Department of Hong Kong, 2017). Private cars, medium and heavy goods vehicles, and special purpose vehicles were under-represented as the number of valid records were much less than the number of licensed vehicles in Hong Kong. On the other hand, light buses, buses and light goods vehicles were well captured in this study with 4.1, 2.1, 1.4 records per vehicle on average, respectively.

Table 2 shows the sampled fleet characteristics by vehicle manufacture year and engine size. The mean manufacture year of the total fleet is 2006.9 and the mean engine size is 4576cc. As shown in Table 2, the majority of the vehicles (83.6%) were manufactured in or after 2001, with 34.4% in 2011 or newer, 28.7% in 2006-2010 and 20.6% in 2001-2005. In addition, small vehicles (≤3000cc) have become more and more popular in recent years and the mean engine size has been decreasing with the new manufacture year. This demonstrates the recent engine downsizing trend which is a key technology for reducing both fuel consumption and pollutant emissions (Huang et al., 2015a; Turner et al., 2014). Table 2 also shows that the ≤2000cc, 5001-6000cc and 8001-9000cc vehicles are relatively new, with mean manufacture years of 2012.1, 2010.8 and 2010.4, respectively. The oldest group is the 3001-4000cc vehicles with a mean manufacture year of 2000.6. Table 3 shows the emission standards and their corresponding effect years for different vehicle types in Hong Kong. Generally, the introduction of each emission standard in Hong Kong was about 2-3 years later than that in the European Union.

3. Results and Discussion

3.1. Effects of manufacture year and engine size on emission trends

Fig. 2 shows the mean emission factors of CO (a), HC (b) and NO (c) as a function of manufacture year. The vehicles are categorized into three groups by the engine size of ≤3000cc, 3001-6000cc and

≥6001cc because they can generate three samples in reasonable sizes and ensure the statistical validity (at least 100 records for each data point (Chen and Borken-Kleefeld, 2016)). In addition, these three ranges of engine size generally correspond to the passenger cars, light goods vehicles, and medium and heavy goods vehicles, respectively. The data point of 1990 includes all the vehicle manufactured in 1990 and before.

As shown in Fig. 2(a), vehicles equipped with larger size engines generally have higher CO emission rates than that with smaller engines. CO of ≤3000cc vehicles reduce slightly or remain stable, while CO of 3001-6000cc vehicles reduce moderately over the entire period. However, CO of ≥6001cc vehicles reduce significantly before 1998 and then increase noticeably during 1998-2006 before dropping again in 2006-2015.

Regarding HC emission rates, as shown in Fig. 2(b), larger vehicles show higher emission rates than those of ≤3000cc vehicles before 2006, but they become comparable after that. The ≤3000cc and 3001-6000cc vehicles show steady reduction of HC while large vehicles (≥6001cc) show the same trend as that observed in CO, which decrease before 1998, increase during 1998-2006 and then decrease after 2006.

Fig. 2(c) shows that larger vehicles emit significantly higher NO emission rates than those of ≤3000cc vehicles before 2004, but they have similar NO emission rates after 2005. For ≤3000cc vehicles, NO emission rates are stable before 1995, decreasing in 1995-1999, increasing in 1999-2006 and decreasing again after 2007. Particularly, a significant jump is observed from 2004 to 2005, making their NO emission rates comparable to those of 3001-6000cc and ≥6001cc vehicles. This jump is mainly caused by a significant higher percentage (6.5%) of NO high-emitters in 2005 vehicles than that of 2004 vehicles (3.2%). For 3001-6000cc and ≥6001cc vehicles, NO emission rates decrease in 1993-1998, increase in 1998-2002 and then decrease steadily after 2002 except for ≥6001cc vehicles between 2013 and 2015. The present remote sensing system ETC-S420 could only measure NO emission. The total NOx emission rates can be estimated by assuming NO_2/NOx ratios for different vehicle types. Carslaw and Rhys-Tyler (2013) investigated the total NOx emissions and NO_2/NOx ratio of various vehicles using remote sensing. The results showed that NO_2/NOx varied significantly between car manufacturers, after-treatment technologies and emission standards. Therefore, only the originally measured NO emissions were presented in this study.

An unexpected trend observed in Fig. 2 is that all the CO, HC and NO emission rates show a more or less increase during the period of 1998-2004, in particular ≥6001cc vehicles. This unexpected increasing trend against the automotive emission standards agrees with the results reported previously. Chen and Borken-Kleefeld (2014) studied the diesel NOx emissions [g/kg fuel] from 18000 valid records of diesel cars and 7900 valid records of light commercial vehicles in Zurich with model year from 1985 to 2012 (mostly Euro 1-4). It was found that NOx increased during 1992-2002 and decreased in 2003-2012. Lau et al. (2012) found that CO and NO emissions [g/km] of Hong Kong diesel vehicles increased during 2004-2008. The diesel fleet covered a full range from passenger cars to heavy goods vehicles and buses (74729 valid records), while the model year was only up to 2008 (Euro 4). Bishop et al. (2013) investigated the emissions [g/kg fuel] of heavy-duty diesel vehicles (4293 vehicles) in California with model year up to 2013. They found that NOx increased in 1990-1995 and decreased steadily from 1995 to 2013, and CO increased slightly in 1998-2004 and decreased in 2004-2013. Carslaw et al. (2011) studied an UK diesel fleet with 35705 records, 60% of which were cars, 34% were light goods vehicles (<3.5t), 2% were heavy goods vehicles (>3.5t) and 4% were buses. The manufacture years were ranged from 1985 to 2010. The results showed that NOx/CO_2 of heavy goods vehicles tended to decrease, buses trended to increase and passenger cars and light goods vehicles were stable during 2000-2010. Pujadas et al. (2017) analysed the emission trends of pre-Euro to Euro 6 vehicles based on 196985 total measurements, 92% of which were passenger cars, 7% were light duty vehicles, 1% were heavy duty vehicles and 78.5% were diesel vehicles. They found that while CO/CO_2 and HC/CO_2 of diesel vehicles were decreasing from pre-Euro to Euro 6 standards, NO/CO_2 showed complex trends. NO/CO_2 increased slightly from pre-Euro to Euro 2, was stable from Euro 2 to 3, decreased from Euro 3 to 4, increased from Euro 4 to 5 and then decreased from Euro 5 to 6. However, each Euro standard covered several years and thus the detailed trends would have been hidden by the averaged values.

The number of diesel emission records in the above reported studies were generally smaller and they were mostly for passenger cars and light commercial vehicles. In this study, the diesel fleet contained 161769 valid records with manufacture year from 1949 to 2015 and engine size from 0.4 to 20 litres. As shown in Table 1, this fleet consisted of 1% of private cars, 60% of light goods vehicles, 11% of medium goods vehicles, 1% of heavy goods vehicles, 9% of light buses and 17% of buses. The large sample size in this study is believed to present the diesel emission trends in a more statistically accurate manner and extend the time period and vehicle class covered. The results in this study show that all CO, HC and NO emission rates are decreasing steadily over the last decade (2005-2015) except for NO of ≥6001cc vehicles which increases again during 2013-2015.

Fig. 3 shows the development of emission rates per vehicle type. For remote sensing data analysis, a large sample size is needed to investigate the average emission trends accurately. Therefore, Fig. 3 only plots the emission rates of light goods vehicles, light buses, medium goods vehicles and buses because the numbers of valid measurements for other vehicle types are too small (as shown in Table 1). Some data points are omitted due to the small number of valid records (<100). The ranges/averages of engine size are 1248-5193/3158cc for light goods vehicles, 2148-4899/3950cc for light buses, 2998-19688/7273cc for medium goods vehicles and 2998-18024/7734cc for buses. As shown in Fig. 3(a), the CO emission rates of light goods vehicles, light buses and buses are very similar to each other. Medium goods vehicles show significantly higher CO emission rates than other vehicle types. Fig. 3(b) shows that medium goods vehicles have slightly higher HC emission rates and all the four vehicle types show steady decrease in HC from 2005 to 2015. Regarding NO emission rates, as shown in Fig. 3(c), light goods vehicles have the lowest emissions before 2005. However, all the vehicle types show similar NO emission rates and decrease steadily after 2005 except buses, whose emission rates increase from 2013 to 2015. Although buses and medium goods vehicles have similar engine size ranges/averages, they show very different CO and HC emission rates. This may be caused by the fact that few buses (91) have engines larger than 12000cc, while much more medium goods vehicles (2189) have >12000cc engines. These larger engine size vehicles tend to have higher emission rates, as discussed in Fig. 2.

The light goods vehicles are one of the most popular diesel vehicles in Hong Kong for commercial transport. The Toyota HiAce fleet alone contributes 30732 valid measurements, accounting for 19.0% of the total records. HiAce vehicles are equipped with a 2982cc diesel direct-injection 1KD-FTV engine launched in 2006 and the gross vehicle weight is 2800 kg. Fig. 4 shows the effect of manufacture year on the average emission factors of the HiAce fleet from 2006 to 2015. This period was compiled with the Euro 4 and 5 standards in which the CO, HC and NOx emission limits remain unchanged for 1760-3500 kg light commercial diesel vehicles. The European emission limits in [g/km] are converted to [g/kg fuel] by applying a combined fuel economy factor of 8.7 [L/100km] as given by the manufacturer (Totota, 2017). The NOx limits are converted to NO limits by assuming a NO_2/NOx ratio of 25% for

Euro 4 and 5 vans (Carslaw and Rhys-Tyler, 2013). As shown in Fig. 4, all the three emission rates reduce with the new manufacture year vehicles. Particularly, a significant reduction is observed between 2010 and 2012. This could be explained by the introduction of a new model HiAce in October 2010 when the old HiAce model MK.5-I was replaced by MK.5-II with an upgraded 1KD-FTV engine (AustraianCar.Reviews, accessed 15.05.2017), indicating that engine retrofitting could be effective in reducing vehicle emissions. The CO emission rates of 2011 or newer vehicles are well below the emission limit and the HC emission rates of 2015 vehicles generally meet the emission standard. However, the NO emission rates of even the newest manufacture year vehicles are higher than the emission limit. This agrees well with findings of previous PEMS study that CO and HC emission rates of diesel vehicles generally remained below the emission limits while NOx emission rates significantly exceeded the limits for Euro 4 and 5 diesel vehicles (Kousoulidou et al., 2013; Weiss et al., 2011). The gap between the real-world driving emission rates and the emission limits becomes larger with older manufacture year vehicles due to the deterioration of combustion and exhaust after-treatment systems.

3.2. Emission distributions and identification of potential high-emitters

The emission rates are sorted from the lowest to the highest and then divided into ten equal sized groups (deciles). Fig. 5 shows the mean emission factors of CO, HC and NO in each decile for different engine size and manufacture year vehicles. As shown in Fig. 5, the distributions of CO and HC emission factors are highly skewed so that the dirtiest 10% vehicles emit much higher emissions than the remaining 90% of vehicles do. However, the distribution of NO is much less skewed. Fig. 5 also shows that the mean emission rates of each decile are higher for larger engine size or older manufacture year vehicles. However, the 1st to 8th deciles of NO show little difference between ≤2004 and 2005-2009 vehicles. This could be explained by the fact that diesel NO emission rates showed little evidence of reduction (Carslaw et al., 2011; Carslaw and Rhys-Tyler, 2013; Pujadas et al., 2017) or even increased (Chen and Borken-Kleefeld, 2014; Lau et al., 2012) in recent years in spite of the tightened emission standards.

To quantify the skewness of the emissions distribution, a commonly used parameter is what percentage of the total fleet emissions are emitted by the dirtiest 10% vehicles. Fig. 6 shows the skewness of diesel emissions by manufacture year and engine size. As shown in Fig. 6, CO and HC emissions are highly dominated by a few dirty vehicles, where 77% and 54% of the total fleet CO and

HC emissions are emitted by the 10% highest emitting vehicles. However, NO emissions are less skewed where only 30% of total NO emissions are emitted by the dirtiest 10% vehicles. The skewness becomes more significant for the newer manufacture year vehicles for all the three emissions. Moreover, this skewness increases for CO of larger engine size vehicles, but not obvious for HC and NO. A previous remote sensing study conducted in 2004, 2006 and 2008 showed that 45% of total CO, 39% total HC and 24% of total NO were emitted by the dirtiest 10% Hong Kong diesel vehicles (Lau et al., 2012). Compared with the previous results, this study demonstrates that the skewed distribution of on-road emissions is getting much more significant for modern diesel vehicles, in particular CO and HC emissions. The emission distributions and trends shown in Figs. 5 and 6 indicate that remote sensing technology would be very effective to screen the CO and HC high-emitters and thus control the on-road vehicle emissions, in particular large engine size and new manufacture year vehicles. However, remote sensing might be less effective for identifying NO high-emitters since NO is more evenly distributed within the fleet.

Vehicles with high instantaneous emissions do not necessarily mean that they are permanent high-emitters, as clean vehicles may have high emissions occasionally. However, if a remote sensing reading is significantly higher than the normal emission level (e.g. the highest 5% emission concentrations), then the chance of this vehicle being a high-emitter is relatively high. Remote sensing only measures the snapshot emissions of a vehicle in a half second and the accuracy of remote sensing is relatively low comparing with laboratory emissions testing. As a result, remote sensing is unable to distinguish vehicles that nearly-fail or just-exceed the standard limits, and thus it is not aimed to identify all the vehicles that exceed the standard limits (or exceed the limits by certain times). Instead, remote sensing aims to screen out vehicles that significantly exceed the limits, such as the 5% highest emitting vehicles. This is an effective and efficient emission control measure, as these small percentage of high-emitting vehicles contribute to a significant share of the total fleet emissions. In addition, targeting at a small percentage of high-emitting vehicles could reduce the resistance for implementing such enforcement programs in practice and thus reducing on-road emissions gradually.

Therefore, a fixed threshold of 95th percentile was used in this study to investigate the characteristics of the absolute high-emitting vehicles. Fixed thresholds were also used in previous studies, such as 95th percentile (Pujadas et al., 2017), CO>3% and HC>500ppm (Lau et al., 2012), and

CO>4.5% (top 3.4% high-emitters) (Bishop et al., 2000). Under this criterion, the cutpoints are 65.49, 9.95 and 29.11 [g/kg fuel] for CO, HC and NO, respectively. A vehicle is considered as a high-emitter when it exceeds any one of the three defined cutpoints. By applying the above criteria, 20447 (12.7%) vehicles are identified as potential high-emitters. These 12.7% high-emitters contributed 70%, 45% and 23% of the fleet total CO, HC and NO emissions, respectively. Fig. 7 shows the percentages of high-emitters of different engine size vehicles as a function of manufacture year. Generally, ≤3000cc vehicles have lower chance (8.7%, overall percentage for the whole period covered) being identified as high-emitters than that of larger vehicles (13.9% for 3001-6000cc and 21.0% for ≥6001cc). As shown in Fig. 7, for ≤3000cc vehicles, the percentage of high-emitters has no clear relationship with the vehicle's manufacture year. No statistically significant difference is observed in the percentages between pre-Euro vehicles (before 1992) and the newest vehicles. This result agrees well with previous study by Pujadas et al. (2017), which reported that the proportion of high-emitting diesel passenger cars (<2.5 litres) had no correlation with the Euro standards. However, the same trend does not apply for larger engine size vehicles. As shown in Fig. 7, the percentages of high-emitters within 3001-6000cc and ≥ 6001cc vehicles decrease with the newer manufacture year, except for vehicles ≥ 6001cc from 1998-2004.

Table 4 shows the number of high-emitting vehicles for one pollutant or at least two pollutants simultaneously. Of the 20447 high-emitters identified, about 40% of them are high for either CO, HC or NO emissions. However, a high-emitter of one pollutant does not necessarily mean that it is also high of another pollutant. As shown in Table 4, the overlapping of the three emissions is relatively small, especially when it involves NO (<2.5%). This is mainly due to their different/conflicting emission formation mechanisms. HC and CO are results of incomplete combustion (mainly rich fuel combustion) while NO is formed in high-temperature rich-oxygen condition (slightly lean fuel combustion) (Huang et al., 2015b). The low overlapping percentage of high-emitters of each pollutant has also been reported in previous remote sensing studies for gasoline vehicles (Bishop et al., 2012; Mazzoleni et al., 2004). The little correlation of diesel high-emitters between CO, HC and NO emissions suggests that the screening criteria should be based on a CO-or-HC-or-NO basis (a vehicle will be considered as a high-emitter if it exceeds any one of the three cutpoints), rather than a CO-and-HC-and-NO basis (a vehicle will be considered as a high-emitter only when it exceeds all of the three cutpoints). This is because the high-emitters of each pollutant are relatively independent and using the later criteria would miss a large number of high-emitters.

4. Conclusions

This study aimed to investigate the real-world driving emission characteristics of diesel vehicles in Hong Kong. Remote sensing technology was used to measure the on-road diesel vehicle emissions in a two-year continuous measurement program. The program obtained 417714 measurements with matched vehicle plate number information and 161769 valid records were selected from this large dataset. Analysis was performed to evaluate the effects of vehicle manufacture year (1949-2015) and engine size (0.4-20 litres) on the emission factors and high-emitters. The major conclusions of this study are:

1) CO emission rates of larger engine size vehicles were higher than that of small vehicles for all the years covered, while HC and NO emission rates were higher for the vehicles manufactured before 2006 and then became similar levels between 2006 and 2015. CO, HC and NO showed an unexpected increasing trend during 1998-2004, in particular for vehicles equipped with engines ≥6001cc. They all decreased steadily in the past decade (2005-2015), except for NO of ≥6001cc vehicles during 2013-2015. Analysis on the HiAce fleet (the most popular light commercial vehicles in Hong Kong) showed that CO and HC of new vehicles were compliant with the emission standards while NO was not. A remarkable reduction of all emissions was observed between 2010 and 2012, indicating that engine retrofit could be an effective strategy for reducing exhaust emissions.
2) The distributions of CO and HC emissions were highly skewed so that the dirtiest 10% vehicles emitted much higher emissions than the rest 90% vehicles, while NO emissions were much less skewed. 77%, 54% and 30% of the fleet total CO, HC and NO emissions were emitted by the dirtiest 10% vehicles. Moreover, this skewed distribution became more significant for larger engine size or newer vehicles. These trends indicated that remote sensing technology would be very effective to screen the CO and HC high-emitters and thus control the on-road vehicle emissions, in particular large engine size and new manufacture year vehicles, while it might be less effective for controlling NO emissions.
3) 12.7% of the vehicles were identified as potential high-emitters under the criteria of the 5% most polluting of the respective emission factors. No clear correlation was observed between the

manufacture year and percentage of high-emitters for ≤3000cc vehicles. However, the percentage of high-emitters decreased with newer manufacture year for larger vehicles. Due to different/conflicting emission formation mechanisms, high-emitters showed little overlapping between CO, HC and NO, particularly when it involved NO.

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Table 1. Licensed diesel vehicles in Hong Kong by April 2017 (Transport Department of Hong Kong, 2017).

Type of vehicle	Number of licensed vehicles	Number of valid records
Private cars	8217	2411
Light buses	3402	14057
Buses	13562	27955
Light goods vehicles	69836	97238
Medium goods vehicles	36238	18426
Heavy goods vehicles	5953	1457
Special purpose vehicles	1347	225
Total	138555	161769

Table 2. Fleet sampling characteristics by vehicle manufacture year and engine size.

Manu. year / Engine size	1990 or before	1991-1995	1996-2000	2001-2005	2006-2010	2011 or newer	Mean MY	Total	%
≤ 2000cc	30	59	0	35	131	2077	2012.1	2332	1.4%
2001-3000cc	177	2057	8004	13221	24075	32614	2007.7	80148	49.5%
3001-4000cc	134	865	2539	2404	1175	209	2000.6	7326	4.5%
4001-5000cc	1	935	4823	11783	6932	4376	2004.6	28850	17.8%
5001-6000cc	20	87	67	41	4289	6772	2010.8	11276	7.0%
6001-7000cc	30	155	120	441	748	502	2006.1	1996	1.2%
7001-8000cc	3	161	773	1828	3609	2624	2007.3	8998	5.6%
8001-9000cc	11	93	157	94	2994	4393	2010.4	7742	4.8%
9001-10000cc	8	131	1475	1362	720	314	2002.2	4010	2.5%
≥ 10001cc	76	330	3132	2058	1698	1797	2003.9	9091	5.6%
Mean engine size	5350	4650	5283	4622	4590	4256	4576/2006.9		
Total	490	4873	21090	33267	46371	55678		161769	
%	0.3%	3.0%	13.0%	20.6%	28.7%	34.4%			100.0%

Table 3. Emission standard and the effect year for diesel vehicles in Hong Kong (HKEPD, 2015). "-" indicates that no vehicles under this emission standard were registered in Hong Kong.

Vehicle type / Emission standard	Euro 1/I	Euro 2/II	Euro 3/III	Euro 4/IV	Euro 5/V	Euro 6/VI
Private cars	1995	-	-	2008	2012	-
Light goods vehicles (≤2.5t)	1996	1999	2003	2007	2012	-
Light goods vehicles (2.5-3.5t)	1995	1998	2002	2007	2012	-
Light goods vehicles (3.5-5.5t)	1995	1997	2001	2006	2012	-
Medium and heavy goods vehicles (>5.5t)	1995	1997	2001	2006	2012	-
Public light buses	1995	1998	2003	2006	2012	-
Private light buses (≤3.5t)	1995	1998	2002	2007	2013	-
Private light buses (>3.5t)	1995	1998	2003	2006	2012	-
Non-franchised Buses	1995	1997	2001	2006	2012	-
Single Deck Franchised Buses	1994	1996	2003	2007	2010	-
Double Deck Franchised Buses	1993	1997	2001	2006	2010	2014

Table 4. Correlations of diesel high-emitters between different pollutants.

Criteria	No. of high-emitters	% of all high-emitters
High CO	8089	39.6%
High HC	8078	39.5%
High NO	8093	39.6%
High CO and HC	3164	15.5%
High CO and NO	285	1.4%
High HC and NO	476	2.3%
High CO and HC and NO	112	0.5%

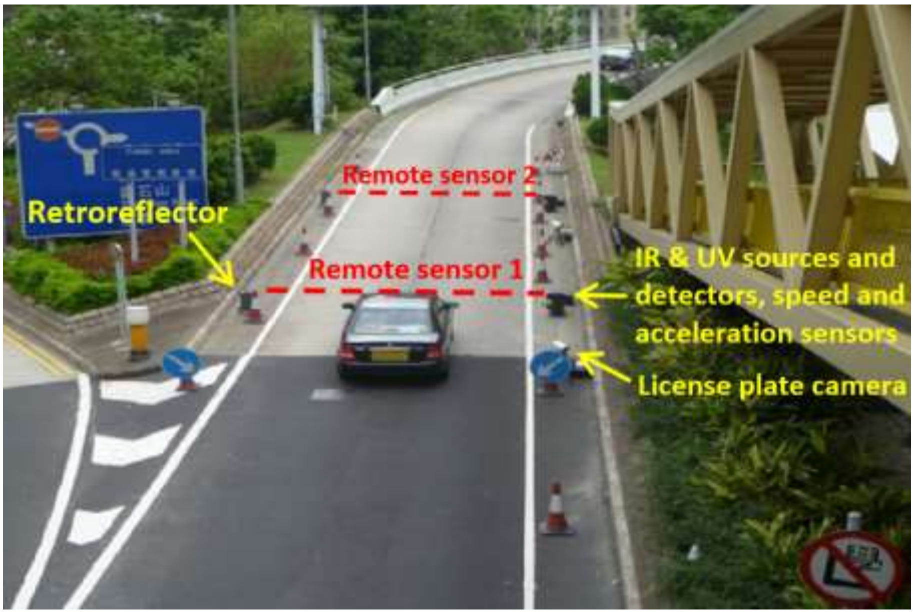
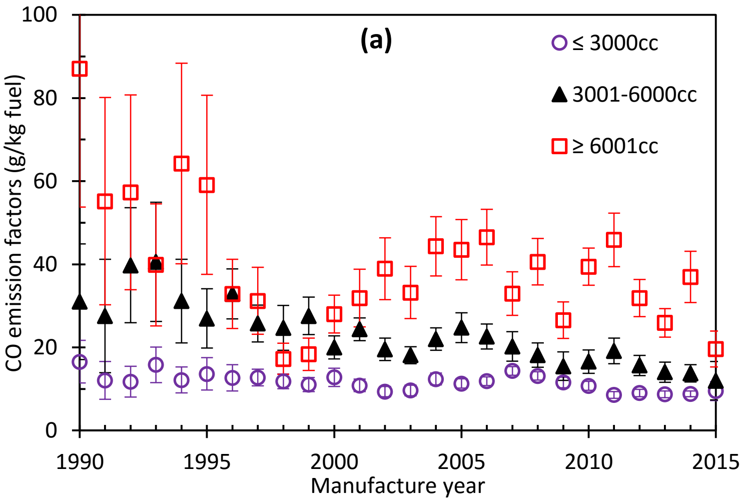
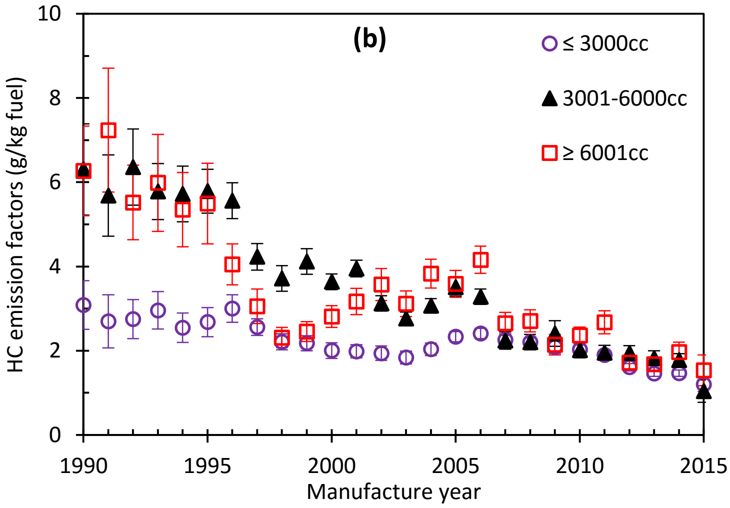


Fig. 1. Typical setup of on-road remote sensing measurement site.





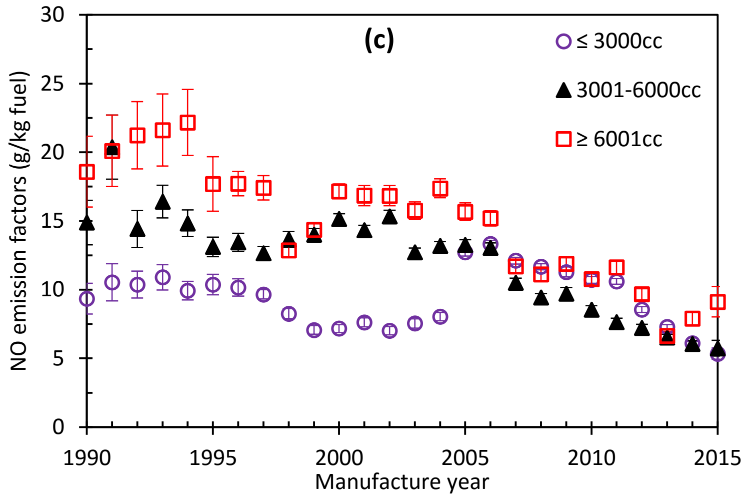
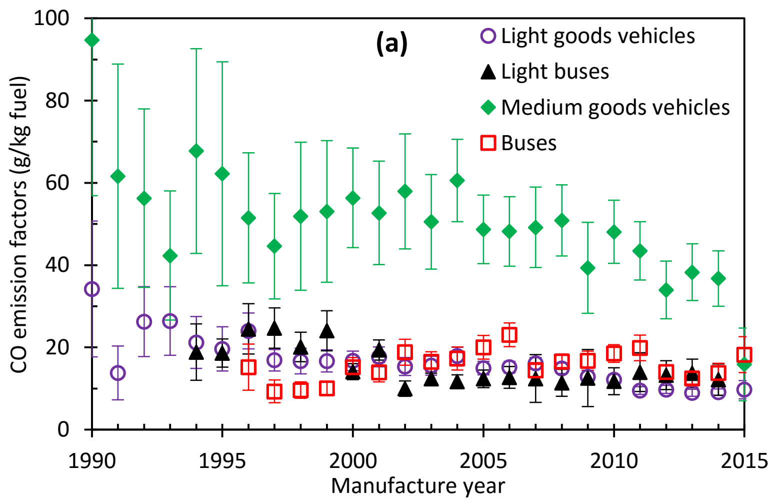
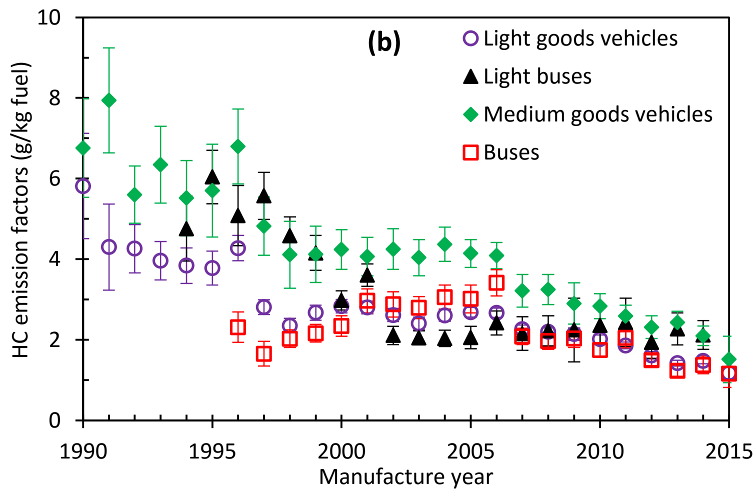


Fig. 2. Mean emission factors of CO (a), HC (b) and NO (c) of different engine size vehicles by manufacture year. Error bars indicate 95% confidence interval over the mean.





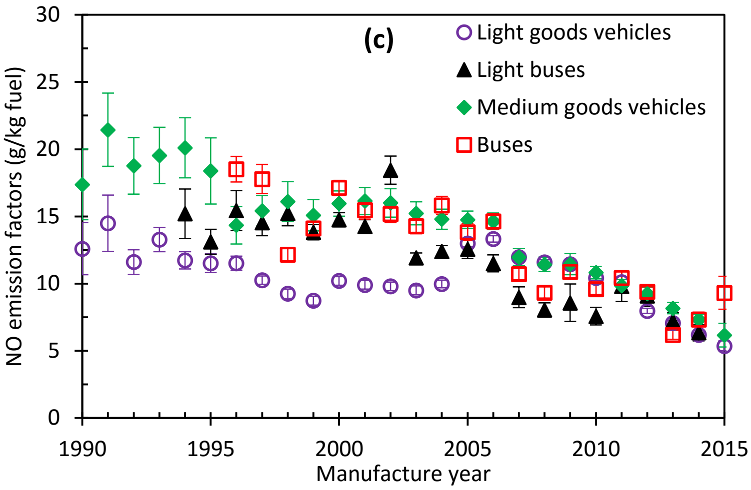


Fig. 3. Mean emission factors of CO (a), HC (b) and NO (c) of different vehicle types by manufacture year. Error bars indicate 95% confidence interval over the mean.

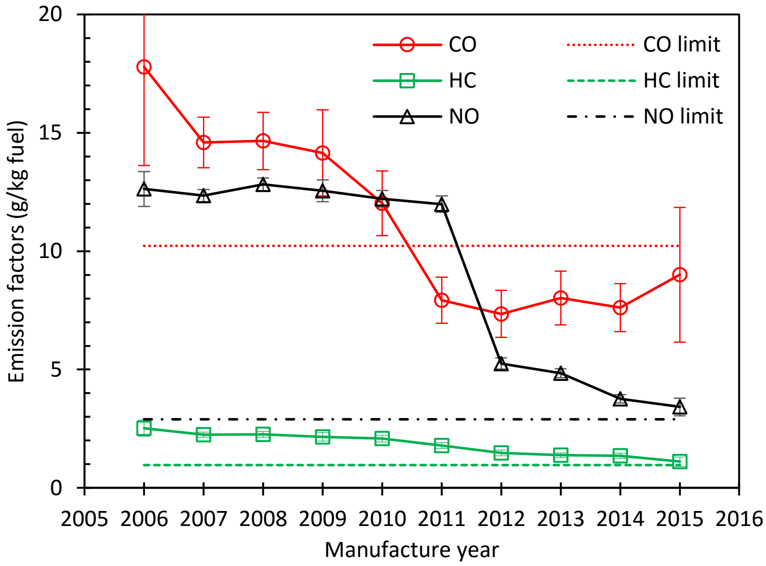
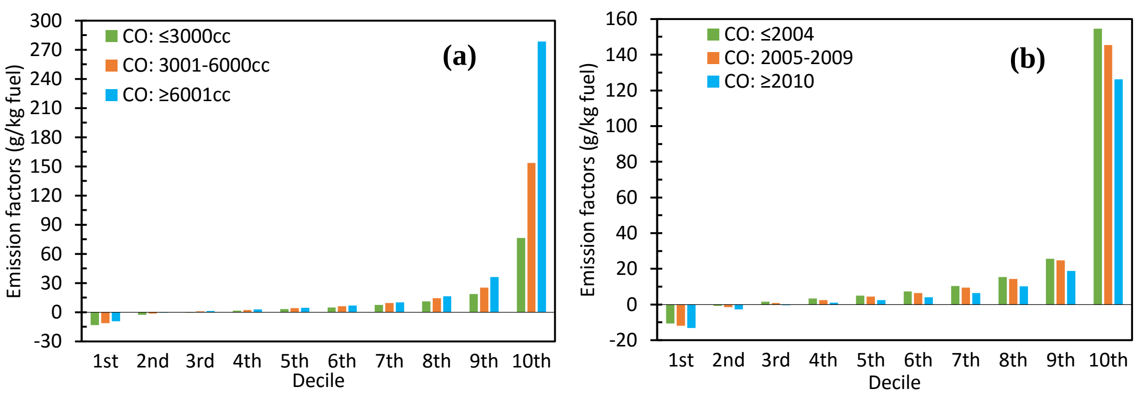


Fig. 4. Mean emission factors of Toyota HiAce fleet by manufacture year. Error bars indicate 95% confidence interval over the mean.



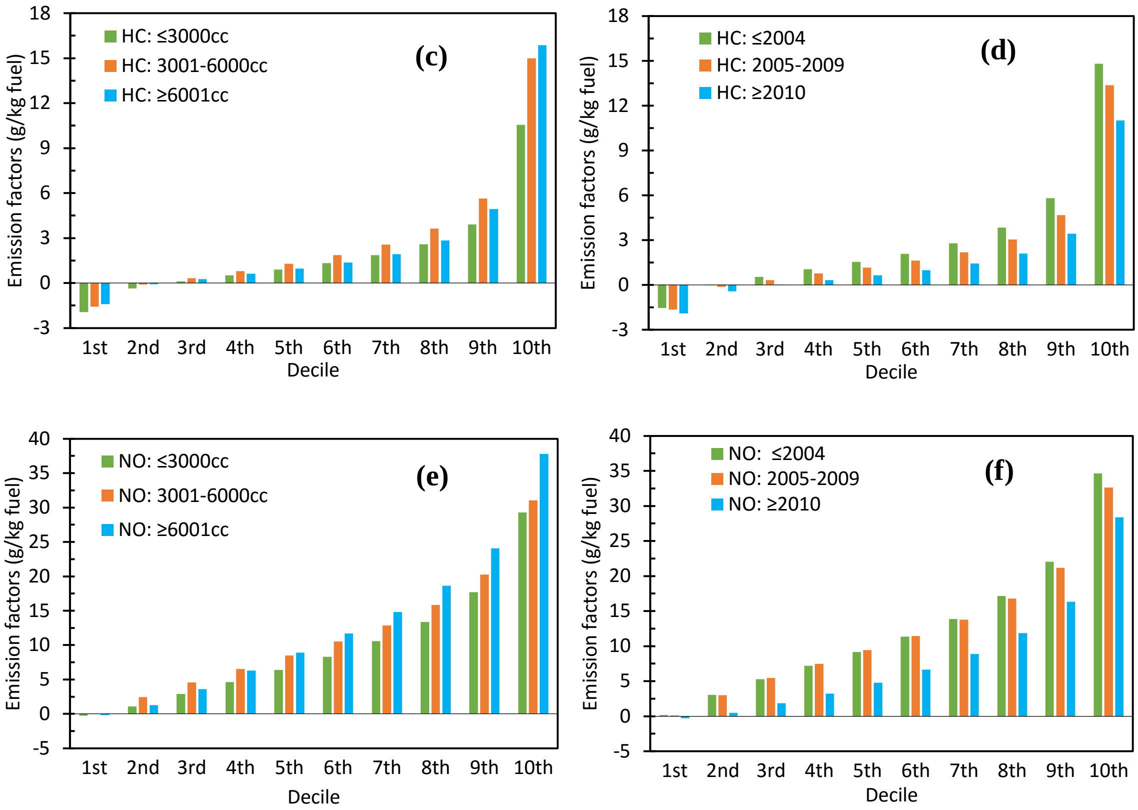


Fig. 5. Mean emission factors of each decile for different engine size and manufacture year vehicles: (a) CO by engine size, (b) CO by manufacture year, (c) HC by engine size, (d) HC by manufacture year, (e) NO by engine size and (f) NO by manufacture year.

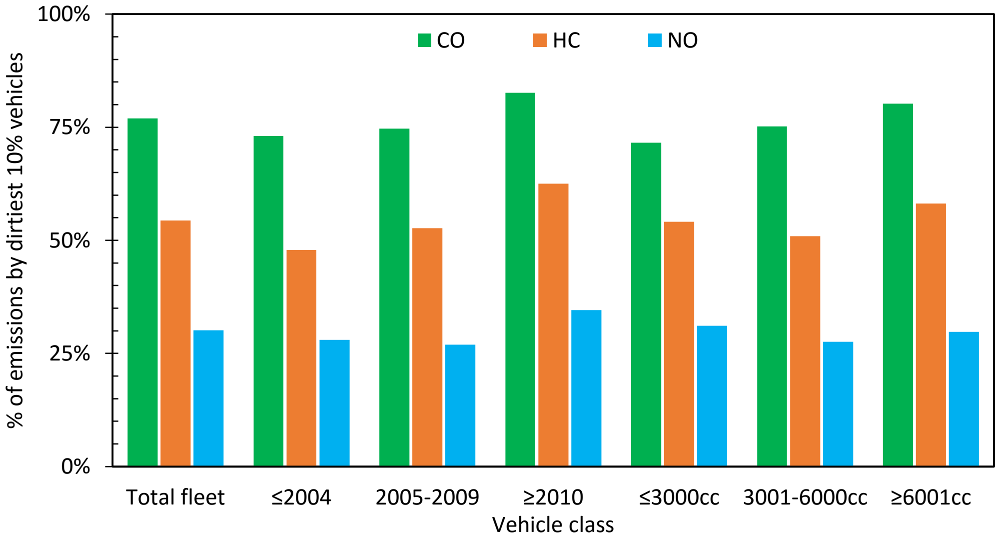


Fig. 6. Percentages of emissions contributed by the dirtiest 10% vehicles.

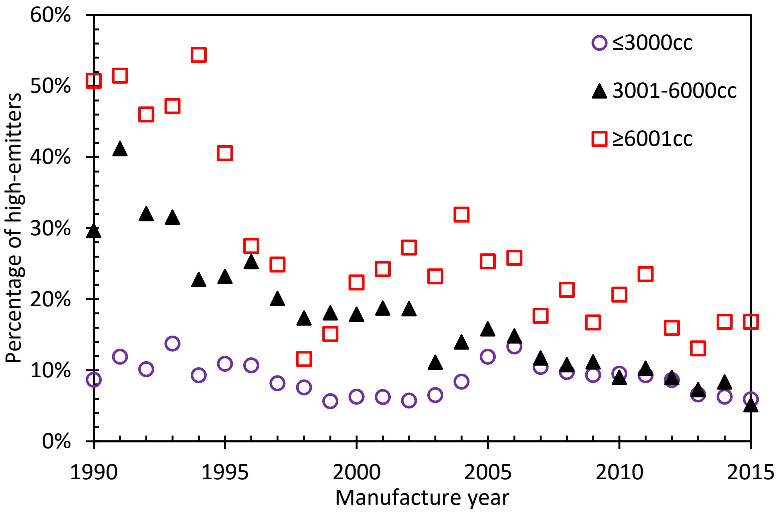


Fig. 7. Percentage of high-emitters in each manufacture year.