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Chemical and stable carbon isotopic composition of $PM_{2.5}$ from on-road vehicle emissions in the PRD region and implication for vehicle emission control policy

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Abstract

Vehicle emission is a major source of urban air pollution. In recent decade, the Chinese government has introduced a range of policies to reduce the vehicle emission. In order to understand the chemical characteristics of $PM_{2.5}$ from on-road vehicle emission in the Pearl River Delta (PRD) region and to evaluate the effectiveness of control policies on vehicles emission, the emission factors of $PM_{2.5}$ mass, elemental carbon (EC), organic carbon (OC), water-soluble organic carbon (WSOC), water-soluble inorganic ions (WSII), metal elements, organic compounds and stable carbon isotopic composition were measured in the Zhujiang Tunnel of Guangzhou, the PRD region of China in 2013. Emission factors of $PM_{2.5}$ mass, OC, EC, and WSOC were 92.4, 16.7, 16.4, and 1.31 mg vehicle^{-1} km^{-1} respectively. Emission factors of WSII were 0.016 (F^-) ~ 4.17 (Cl^-) mg vehicle^{-1} km^{-1}, totally contributing about 9.8 % to the $PM_{2.5}$ emissions. The sum of 27 measured metal elements accounted for 15.2 % of the $PM_{2.5}$ emissions. Fe was the most abundant metal element, with an emission factor of 3.91 mg vehicle^{-1} km^{-1}. Emission factors of organic compounds including *n*-alkanes, PAHs, hopanes, and steranes were 91.9, 5.02, 32.0 and 7.59 µg vehicle^{-1} km^{-1}, respectively. Stable carbon isotopic composition $\delta^{13}C$ value was measured and it was −25.0 ‰ on average. An isotopic fractionation of 3.2 ‰ was found during fuel combustion. Compared with a previous study in Zhujiang Tunnel in year 2004, emission factors of $PM_{2.5}$ mass, EC, OC, WSII except Cl^-, and organic compounds decreased by 16.0–93.4 %, which could be attributed to emission control policy from 2004 to 2013. However, emission factors of most of the metal elements increased significantly, which could be partially attributed to the changes in motor oil additives and vehicle condition. There are no mandatory national standards to limit metal content from vehicle emission, which should be a concern of the government. A snapshot of the 2013 characteristic emission of $PM_{2.5}$ and its constituents from on-road vehicular fleet in the PRD region retrieved from our study was found to be useful for the assessment of past and future implementation of vehicle emission control policy.

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1 Introduction

Vehicle emission is a major source of urban air pollution and it accounts for approximately 14 ~ 50 % of total fine particle mass in urban areas (Sheesley et al., 2007; Wang et al., 2008; Yu et al., 2013). The environmental and health effects of vehicle emissions are of our concern during the last decades. Numerous studies have been conducted to characterize vehicular particulate matter (PM) emission in many countries, with respect to emission factors, chemical composition, and size distribution (Chiang and Huang, 2009; Laschober et al., 2004; Pio et al., 2013). The characteristics of vehicle emission in China were studied by tunnel experiments, dynamometer tests or road monitoring (He et al., 2008; Jin et al., 2014; Song et al., 2012). Because of the differences in fuel qualities, engine conditions, and operation practices, the PM emission from vehicles varied from region to region and time to time.

The Pearl River Delta (PRD) region is located in the southern coast of China, noted for its rapid urbanization and industrialization in the last few decades, has experienced serious atmospheric pollution. Vehicle emission accounts for approximately 25–30 % of total fine PM in the PRD region (http://epaper.southcn.com/nfdaily/html/2014-01/03/content_7261687.htm). Peer reviewed papers had reported emission factors and chemical characteristic of $PM_{2.5}$ from vehicle emission in the PRD region, by means of tunnel studies in Zhujiang Tunnel (Guangzhou) and Wutong Tunnel (Shenzhen) (He et al., 2006, 2008; Huang et al., 2006). However, the sampling in these studies was conducted in 2004. During the past decade, the Environment Protect Agency of Guangdong Province revised the "Motor vehicle exhaust pollution prevention and control regulations of Guangdong Province" in 2008 and released the "PRD Regional Air Quality Management Plan" and "A Clean Air Plan" in 2010, to improve the relevant air quality through policies and measures. The emission standards for newly registered vehicles were tightened to China IV and the better quality of gasoline and diesel were supplied in 2013. Therefore, the characteristics of PM emission from vehicles in the PRD region might have changed throughout these years.

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Tunnel experiments and chassis dynamometer tests were widely used to measure various pollutants emitted from vehicles (He et al., 2006; Heeb et al., 2003). However, dynamometer test has the defect that it can not account for vehicle fleet composition and emissions characteristics related to break and tire wear and re-suspension of road dust (Thorpe and Harrison, 2008). Tunnel studies have been demonstrated to be a suitable setup to measure PM emissions from on-road mixed fleets (Chiang and Huang, 2009; Laschober et al., 2004; Pio et al., 2013).

This study was carried out in a roadway tunnel located in the PRD region. We report here the emission factors of $PM_{2.5}$ mass, organic carbon (OC), elemental carbon (EC), water-soluble inorganic ions (WSII), metal elements, water-soluble organic carbon (WSOC), organic compounds and stable carbon isotopic composition. WSOC has the potential to modify the hygroscopicity of particles, PM size and cloud condensation nuclei activities (Shulman et al., 1996), but it is often ignored in previous studies owing to the hydrophobic nature of the organic aerosol from primary vehicle emissions. Stable carbon isotope ($\delta^{13}C$) is very useful for tracing sources (Lopez-Veneroni, 2009; Widory, 2006), and it was also less reported for vehicular exhaust emissions (Ancelet et al., 2011; Widory, 2006). The objectives of this study are: (1) to obtain comprehensive information on the chemical and stable carbon isotopic composition of $PM_{2.5}$ emission from on-road vehicles in the PRD region, (2) to compare our results with the previous study conducted in the same tunnel in 2004, (3) to evaluate the effectiveness of the implementation of vehicle emission control policies from 2004 to 2013 in the PRD region.

2 Experimental

2.1 Tunnel sampling

$PM_{2.5}$ samples were collected during 10 to 14 August 2013 from the roadway tunnel (Zhujiang Tunnel) located in Guangzhou City, China. It has two bores, each of which

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has three lanes with traffic in the same direction, as shown in Fig. 1. Two high-volume $PM_{2.5}$ samplers (GUV-15HBL1, Thermo, USA) were placed at a distance of 75 m from the entrance and 75 m from the exit, respectively. The vehicle speed in the Zhujiang Tunnel was 18 to 45 km h^{-1}, with an average vehicle speed of 33.4 km h^{-1} during the sampling. The air samples were drawn at about 1.13 m^3 min^{-1} through the quartz fiber filters (QFFs, 20.3 cm × 25.4 cm, Whatman). Field blank samples were also collected by loading filters into the samplers but without pulling air through. The sampled filters were wrapped with annealed aluminum foil and stored in a refrigerator at −40 °C till analysis. The meteorological parameters were synchronously recorded. A video camera was placed at the exit to record the passing vehicles during the sampling periods. The videotapes were then used to determine the vehicle counts and to classify the vehicles into three categories, namely, diesel vehicles (DV) (heavy-duty trucks, light-duty trucks and large passenger cars), gasoline vehicles (GV) (small cars and motorcycles), and liquefied petroleum gas vehicles (LPGV) (bus and taxies). The average traffic density during sampling was 1797 per hour with DV, GV and LPGV proportion of 13.7, 59.8 and 26.5 % respectively. More details of the vehicle counts and meteorological conditions are summarized in Table S1 of the Supplement.

2.2 Chemical analysis

The $PM_{2.5}$ mass concentrations were determined gravimetrically by weighing the quartz filters before and after sampling in an electronic hygrothermost for 24 h at 25 °C and 50 % relative humidity (RH). Then, samples were analyzed for OC/EC, WSOC, WSII, metal elements, organic compounds and stable carbon isotopic composition. The experimental methods of the chemical analysis are available of the Supplement.

2.3 Calculation of emission factors

Average emission factors were calculated for each sampling period on the basis of the concentration differences between the exit and entrance of the tunnel by the following

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equation (Handler et al., 2008):

$$EF = (C_{out} - C_{in})V/NL$$

where EF is the emission factor of a species in unit of mg vehicle^{-1} km^{-1}, N is the number of vehicles passing through the tunnel, L is the distance between inlet and outlet sampling locations, C_{out} and C_{in} are the measured species concentrations at the tunnel outlet and inlet, respectively, and V is the corresponding air volume calculated from the cross-sectional area of the tunnel, the average wind speed, and the sampling duration of each filter. The average concentrations of all measured species at the inlet and outlet sampling locations and the corresponding emission factors in this study are presented in Table S2 of the Supplement and Tables 1–2.

3 Results and discussion

3.1 Characteristics of $PM_{2.5}$ emissions from vehicles in the PRD region

3.1.1 $PM_{2.5}$ mass, OC, EC, WSOC, WSII, metal elements

The $PM_{2.5}$ mass emission factors ranged from 79.8 to 107 mg vehicle^{-1} km^{-1}, with an average of 92.4 ± 8.9 mg vehicle^{-1} km^{-1}. Average OC and EC emission factors were 16.7 ± 1.9 and 16.4 ± 2.1 mg vehicle^{-1} km^{-1}, respectively, and they accounted for 19 ± 2 and 18 ± 2 % of $PM_{2.5}$ mass emission. The ratio of OC to EC in the Zhujiang Tunnel ranged from 0.77 to 1.35, with an average of 1.03. Previous studies have shown that the OC/EC ratio is useful to separate gasoline engine emissions from diesel emissions. Higher values (> 2) are associated with GV and LPGV exhaust and lower values (0.3 to ~ 0.9) associated with DV exhaust (Cadle et al., 1999; Cheng et al., 2010; Gillies and Gertler, 2000). Therefore, the low OC/EC ratios in this study, which are closer to that from DV exhaust, indicated that diesel vehicles played an important role in the $PM_{2.5}$

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emissions although the proportion of DV was only 13.7 % during the sampling. Additionally, it should be noted that emissions of EC from heavy duty trucks are expected to be relatively low under the low speed operating conditions in the tunnel (Kweon et al., 2002). Therefore, the ratio could be lower at the actual driving condition of vehicle fleet with a higher speed on the road. The concentration of WSOC in the inlet location was 6.21 $\mu g\,m^{-3}$ (Table S2 of the Supplement) with a percentage of 31.1 % of OC, which is close to that of ambient air (Ding et al., 2008; Ho et al., 2006). While in the outlet location, the concentration of WSOC was 8.00 $\mu g\,m^{-3}$ (Table S2 of the Supplement), with a percentage of 17.9 % of OC. The WSOC had been reported to contribute on average 20 % to OC in the exit of Marseille roadway tunnel (El Haddad et al., 2009), in which background influence was included. The calculated emission factor of WSOC in this study ranged from 0.5 to 2.8 mg vehicle^{-1} km^{-1}, with an average of 1.31 mg vehicle^{-1} km^{-1}, which consisted of 7.84 % of OC. Such a WSOC fraction is considerably lower than that previously measured for biomass burning particle (71 %) (Mayol-Bracero et al., 2002). However, it could influence the hygroscopicity of particles and the formation of secondary aerosols (Ho et al., 2006; Rogge et al., 1993b; Weber et al., 2007) and is worthy of more attention and in-depth research.

The sum of WSII comprised about 9.8 % of the $PM_{2.5}$ emission, with emission factors of 4.17, 0.104, 0.609, 2.88, 0.165, 0.177 and 0.953 mg vehicle^{-1} km^{-1} for Cl^-, NO_3^-, SO_4^{2-}, Na^+, NH_4^+, Mg^{2+} and Ca^{2+}, respectively. The other WSII had a minor contribution (< 0.1 mg vehicle^{-1} km^{-1}). Totally 27 measured metal elements contributed 15.2 % to the $PM_{2.5}$ emission. Fe was the most abundant element, with an emission factor of 3.91 mg vehicle^{-1} km^{-1}, followed by Na 3.53 mg vehicle^{-1} km^{-1}, Al 3.15 mg vehicle^{-1} km^{-1}, Ca 1.93 mg vehicle^{-1} km^{-1}, Mg 0.496 mg vehicle^{-1} km^{-1}, and K 0.338 mg vehicle^{-1} km^{-1}, which accounted for 4.2, 3.8, 3.4, 2.1, 0.5, and 0.4 % of $PM_{2.5}$ mass emission respectively. These six elements contributed 95.0 % to the total metal emission. Emission factors of other metals ranged from 0.0001 (Ag) to 0.25 (Ba) mg vehicle^{-1} km^{-1}, with a sum of 0.71 mg vehicle^{-1} km^{-1}. It is worth noting that emission factors of elements including Na, K, Mg and Ca were significantly higher

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than that of their corresponding water-soluble part (see Table 1). The differences can attribute to the water-insoluble matter with these metal elements, such as organic compound chelated metal elements.

$PM_{2.5}$ mass was also obtained by summing chemical species including organic matter (OM), EC, inorganic ions (Cl^-, NO_3^-, SO_4^{2-}, and NH_4^+) and metal elements. OC was multiplied by 1.4 to estimate mass of OM (He et al., 2008). Metal elements were assumed to exist as oxides and the concentrations of these elements were multiplied by the corresponding factors, to account for the oxygen mass (Almeida et al., 2006). The gravimetric measurements exhibited excellent agreement with the reconstructed mass values. The average $PM_{2.5}$ reconstructed mass was 69.5 % of the gravimetric value. The uncertain component can be attributed to some important elements which have not been analyzed in this study, such as elements Si, S and their compounds.

3.1.2 Organic compounds

The average emission factors and abbreviated names of 67 individual organic compounds identified in the Zhujiang Tunnel, including *n*-alkanes, polycyclic aromatic hydrocarbons (PAHs), hopanes, and steranes are listed in Table 2. These organic compounds accounted for 0.59 % of the OM and 0.11 % of the $PM_{2.5}$ mass emissions, The distributions of organic molecular markers associated with $PM_{2.5}$ are known to be source indicative despite of their small mass fractions (Schauer et al., 1996; Simoneit, 1986). *n*-Alkanes are an important class of organic compounds in atmospheric aerosols, and their homologue distribution may indicate different pollution sources (Rogge et al., 1993a). In this study, the *n*-alkane traces were dominated by C11-C36 with no odd-even carbon number predominance and the maximum was at C24, consistent with the characteristics of vehicle emissions reported by Simoneit (1984, 1985). The emission factors of individual n-alkanes were in the range of 0.22 (C13) ~ 13.3 (C24) $\mu g\,vehicle^{-1}\,km^{-1}$ (Table 2).

There has been a worldwide concern to PAHs due to their known carcinogenic and mutagenic properties. PAHs are thought to be the result of incomplete combustion.

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Totally 15 priority PAHs (the results of naphthalene have not been discussed in this study due to low recovery) were identified and quantified. The emission factor of total PAHs varied from 4.56 to 5.54 μg vehicle^{-1} km^{-1} in this study. The emission factor of benzo[a]pyrene (BaP), which is often used as an indicator of PAHs and regarded by World Health Organization as a good index for whole PAHs carcinogenicity, was in the range of 0.37 to 0.46 μg vehicle^{-1} km^{-1}. The emission factors for other compounds ranged from 0.006 (acenaphthene) to 0.89 (pyrene) μg vehicle^{-1} km^{-1} (Table 2). Pyrene was the most abundant compound, followed by chrysene (CHR), benzo[ghi]perylene (BghiP) and benz[a]anthracene (BaA), which is different from biomass burning and coal combustion (Huang et al., 2014; Shen et al., 2012). PAHs diagnostic ratios have been used as a tool for identifying pollution emission sources including ANT/(ANT + PHE), FLA/(FLA + PYR), BaA/(BaA + CHR), BbF/(BbF + BkF), IcdP/(IcdP + BghiP) and BaP/(BaP + BghiP) (Tobiszewski and Namiesnik, 2012; Yunker et al., 2002; Zhang et al., 2005). We summarized PAHs ratios mentioned above in Fig. 2 from three combustion sources including vehicle emission (VE), biomass burning (BB) and coal combustion (CC). On the whole, the six ratios in this study are similar to the other tunnel experiments, though environmental conditions of tunnels are different to some extent. It is also suggested that the ratio of FLA/(FLA + PYR) and IcdP/(IcdP + BghiP) might be more suitable to distinguish VE PAHs emission with BB and CC.

Hopanes and steranes are known molecular markers of aerosol emissions from fossil fuel utilization (Simoneit, 1985). Rogge et al. (1993a) and Schauer et al. (1996) had shown that these petroleum biomarkers can be used to trace motor vehicle exhaust contributions to airborne PM in southern California atmosphere. Fourteen major hopanes homologues with emission factors ranging from 0.46 ∼ 9.14 μg vehicle^{-1} km^{-1} and twelve steranes homologues ranging from 0.31 ∼ 0.97 μg vehicle^{-1} km^{-1} were identified in this study. 17α(H),21β(H)-hopane (HP30) was the most abundant component with the emission factor of 9.14 μg vehicle^{-1} km^{-1}. The emission factor of total hopanes was 32.0 μg vehicle^{-1} km^{-1}. Emissions of the S hopanes for the extended 17α(H),21β(H)-hopane homologues > C31 were always higher than those of the cor-

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responding R pairs. All these characteristics of hopanes in the Zhujiang Tunnel are consistent with those in gasoline and diesel exhausts (Rogge et al., 1993a; Simoneit, 1985) and in other tunnel studies (see Fig. S1 of the Supplement). Emission factors of steranes ranged from 0.31 to 0.97 ng vehicle^{-1} km^{-1}, and the sum of their emission factors was 7.58 µg vehicle^{-1} km^{-1}. The most abundant homologue was C29$\alpha\beta\beta$ stigmastane (20R) (29$\alpha\beta\beta$R), followed by 29$\alpha\alpha\alpha$S and 29$\alpha\beta\beta$S.

3.1.3 Stable carbon isotopic composition

Stable carbon isotope analysis of vehicle emissions in Zhujiang Tunnel yielded δ^{13}C values ranging from −25.5 to −24.7 ‰ with an average value of −25.0 ± 0.2 ‰, and is comparable to previously reported ranges of −29 to −24.6 ‰ (Table 3) for vehicular fuel emission. Generally, the variation in $\delta^{13}C_{Fuel}$ could affect the δ^{13}C of hydrocarbons (Keppler et al., 2004; Yamada et al., 2009). In the PRD region, the δ^{13}C value of gasoline was on average −28.6 ± 0.6 ‰, and the δ^{13}C value of diesel was −27.8 ± 0.2 ‰, and small variation of fuel δ^{13}C was observed (Hu et al., 2014). We calculated the isotopic differences between $\delta^{13}C_{PM_{2.5}}$ and $\delta^{13}C_{Fuel}$, which represents the apparent isotopic fractionation occurring during fuel burning. It expressed as Δ^{13}C (‰), and is defined by the following equation (Yamada et al., 2009).

$$\Delta^{13}C_{PM_{2.5}\text{-Fuel}} = \left(\frac{\delta^{13}C_{PM_{2.5}} + 1000}{\delta^{13}C_{Fuel} + 1000} - 1\right) \times 1000$$

In this study, the value of $\Delta^{13}C_{PM_{2.5}\text{-Fuel}}$ was from 2.7 to 3.5 ‰, with an average of 3.2 ‰, indicating an isotopic fractionation occurred during fuel combustion. Comparing the stable isotopic carbon value of vehicular fuel emission with other particulate emission sources (see Table 3), it is found that different emission sources showed different stable carbon isotopic composition. Stable isotopic carbon values of coal combustion, street dust, and biomass burning of C4 plant were heavier than that of vehicle emission. The

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heaviest $\delta^{13}C$ value was −19 ∼ −13 ‰ for biomass burning of C4 plant, followed by −21 ∼ −16.4 ‰ for street dust and −23.9 ∼ −23.3 ‰ for coal combustion, while stable isotopic carbon values of charcoal combustion (−27.4 ‰) and biomass burning of C3 plant (−34.7 ∼ −27 ‰) are lighter than that of vehicle emissions from the tunnel studies. These significant differences between emission sources indicated that $\delta^{13}C$ value is an effective marker of motor vehicle emission, and could be very useful for tracing sources.

3.2 Comparison with previous study conducted in the same tunnel

To investigate the variation of chemical emission characteristics from vehicles in the PRD region over the past decade, we compared the chemical emission characteristics of this study with that of previous study (He et al., 2008) for the same tunnel in 2004 (see Figs. 3 and 4). Figure 3 shows that $PM_{2.5}$ mass, OC, EC decreased significantly from 2004 to 2013. The reason can be partly attributed to the implementation of pollution control measures for Chinese vehicle emission. During this 9 year period, vehicle emission standard have raised two levels (from China II in 2004 to China IV in 2013) (Table 4). Additionally, comparing the fleet composition of 2013 with 2004 in Zhujiang Tunnel, we found that the proportion of DV and GV decreased while that of LPGV increased. LPG is a type of clean energy, and LPGV is known to emit much less PM mass than GV and DV, while DV emits the most PM mass (Allen et al., 2001; Myung et al., 2014; Yang et al., 2007). LPG could be combusted more completely than gasoline and diesel. Changes mentioned above contributed greatly to the decrease of emission factors of OC and EC (31.3 and 66.9 %) and $PM_{2.5}$ mass (16.0 %) from 2004 to 2013. However, the emissions of $PM_{2.5}$ mass, OC and EC are still quite higher than those measured in other countries (see Table S3 of the Supplement). The implication of these high emission factors levels is that both the fuel quality and engine technologies in the PRD region need to be further improved.

It is also found from Fig. 3 that emission factors of NO_3^-, SO_4^{2-} and NH_4^+ decreased from 2004 to 2013. Improvement of fuel quality resulted in sulfate emission factor de-

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crease from 3.18 to 0.61 mg vehicle^{-1} km^{-1}, since the amount of sulfur in fuel is slashed by 81.5 ~ 95 % in China IV (2013) when compared that in China II (2004) (Table 5). The emission levels of nitrate and ammonium were about one-tenth of those observed in 2004, possibly because NO_x emission standard is tightened from 2004 to 2013 (Table 4) leading to the less production of ammonium nitrate. Emission factor of chloride is quite higher than that obtained from Zhujiang Tunnel in 2004 and other tunnels. Chloride was found up to 74 mg vehicle^{-1} km^{-1} in PM_{10} in the Howell Tunnel, due to the application of salt to melt ice on roadways in the winter (Lough et al., 2005). However, it is not applicable in Guangzhou City. The good correlation between Cl^- and Na^+ (r^2 = 0.992) indicates the re-suspension of sea salt particles combined with vehicle emission PM might be a major source (He et al., 2008).

Emission factors of most of the metal elements increased in Zhujiang Tunnel from 2004 to 2013 except Cd and Pb. Na emission increased 3.18 mg vehicle^{-1} km^{-1} in 2013 than in 2004. Na correlated weakly with Cl^- (r^2 = 0.374) and Na^+ (r^2 = 0.429), and Na emission was 23 % higher than Na^+ emission. This indicated that Na emission has other sources and it was not only from the re-suspension of sea salt particles. The other four most abundant elements including Fe, Ca, Mg, K increased by 1 to 3 times. Their weak correlations (except Fe) with EC (Table S4 of the Supplement), suggest that direct tailpipe emissions were not a major source. As they are major components of crustal material, and correlated well with each other (Table S4 of the Supplement), a significant source could be the resuspension of road dust caused by traffic. Therefore, their increase of emission factors can be attributed to higher wind speed of the traffic (3.8 m s^{-1} in this study, while 3.0 m s^{-1} in 2004) which resulted in more road dust distributed into PM. Fe correlated weakly with other crustal element, indicating other sources, such as oil additive and the wear of engines, brakes, and tires (Cadle et al., 1997; Garg et al., 2000; Gidney et al., 2010). Additionally, emissions of Zn, Cu, Mn, Cr, Ni, V, As, Co, U, and Tl increased by 0.5 to 4.5 times. Although the sum of these elements did not exceed 0.5 % of $PM_{2.5}$ mass, they are important for health effects.

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Lower emission factor of Pb (0.01 ± 0.0007 mg vehicle^{-1} km^{-1}) in 2013 than in 2004, could be a result of the phase out of leaded gasoline across China in the late 1990s.

Figure 4 shows comparison of organic compounds emission in Zhujiang Tunnel between 2004 and 2013. The *n*-alkane homologues exhibited a smooth hump-like distribution with the most abundance at C24, as shown in Fig. 4a. Such a distribution pattern was similar to patterns observed in Zhujiang Tunnel 2004. However, there are some differences. Firstly, the highest abundant *n*-alkane shifted from C23 in 2004 to C24 in 2013. It was reported that the *n*-alkane in the highest abundance was C20 for DV and C25 or C26 for GV in dynamometer tests (Rogge et al., 1993a; Schauer et al., 1999, 2002). As the emissions collected in tunnel studies present a composite result of emissions from a mixed vehicle fleet, the lower fraction of DV in 2013 was more likely the cause for the shift of the C_{max}. Secondly, emission factors of C16–C26 in 2013 were quite lower than that in 2004, while this trend reverses gradually after C27. Emission factors of the PAHs decreased significantly and the percentage of decrease reached up to 67.6 % ~ 93.4 %. BaP equivalents (BaPeq) emission factors decreased by 88.1 % from 2004 to 2013 (Table S5 of the Supplement). This could be attributed to the variation of fleet composition between 2004 and 2013. PAHs emitted from LPGV are about one-third of that from GV (Yang et al., 2007), while DV emit more PAHs than GV (Phuleria et al., 2006). Therefore, higher proportion of LPGV and lower proportion of DV resulted in the lower emission factor of PAHs in 2013 than that in 2004. Emission factors of hopanes also decreased from 2004 to 2013, the percentage of decrease ranged from 56.2 to 68.7 %. However, the percentage distributions of hopane series derived from different tunnel studies including Zhujiang Tunnel 2004 and Zhujiang Tunnel 2013 were very similar (see Fig. S2 of the Supplement). This seems to suggest that the hopane emission characteristic might be independent of the fleet composition. This is a reasonable result given that hopanes originate from the lubricating oil used in DV, GV and LPGV rather than from the fuel (He et al., 2008; Phuleria et al., 2006). Owing to more units in heavy-duty vehicle need lubrication, emission factors of hopanes attributable to heavy-duty vehicle were higher than that to light-duty vehicle (Phuleria

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et al., 2006). Reduction of the proportion of heavy-duty vehicle (bus, heavy-duty truck, large passenger cars) proportion in fleet composition in 2013 (11.3 %) compared that in 2004 (20 %) might be the reason that emission factors of hopanes decreased.

3.3 Implication for vehicle emission control policy

Vehicle emission control strategies and policies adopted by Guangdong Province can be classified as follow: (1) Emission control on vehicles: the PM emission standards for newly registered vehicles were tightened from emission limits ranging from 0.08 ~ 0.20 g km^{-1} for light duty vehicle and 0.15 g kWh^{-1} for compression ignition and gas fuelled positive ignition engines of vehicles for China II in 2004 to 0.025 ~ 0.060 g km^{-1} for light duty vehicle and 0.02 ~ 0.03 g kWh^{-1} for compression ignition and gas fuelled positive ignition engines of vehicles for China IV in 2013 (Table 4). There was a large reduction in PM emission limit from China II through China III to China IV. Commercial "yellow-label" vehicles, which emits vehicle exhaust out of limit of emission standards, were phased out step by step through more rigorous inspection and maintenance programs after 2012. A considerable percentage of vehicles made in early 2000s which could not comply with limit values at the time of inspection test was deregistered. The reduction of on road high PM emiting vehicles and the phasing in of lower PM emiting vehicles with the implementation of China III and IV were effective for decreasing PM emission. Emission factors of PM decreased by 16 % from 2004 to 2013. Also for NO_x, the emission limit was reduced to half from China II to China IV for heavy duty vehicle, and about half from China III to China IV for light duty vehicle. This change in emission standard which limit NO_x emission was a major factor for emission factors of nitrate and ammonium decreased by about 90 %. (2) Fuel quality improvements: National Standard has been revised several times to improve fuel quality to adapt to stringent vehicle emission standards (Table 5). Vehicle fuel with 1000, 500 and 270 mg m^{-3} limit of sulfur content for gasoline, diesel and LPG was used in 2004, while for 2013 it was 50 mg kg^{-1} for all types of fuels. The significant result is that emission factors of sulfate decreased by 70 % owing to significant sulfur content

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decreasing in fuel. Also, to comply with the China III and China IV emission standard, new vehicles in the country will need to go through tougher tests. For example, the durable requirements were advanced from 80 000 km for China III to 100 000 km for China IV, similar to the European standard Euro III and Euro IV. The result was that there are more environmental friendly vehicles on road with better advanced engines following the implementation of these emission standards. (3) Alternative fuel and vehicles: LPG is gradually taking the place of diesel and gasoline as the fuel of taxi and bus after 2004, and now seldom taxi and bus use diesel and gasoline as fuel (http://www.southcn.com/news/gdnews/nanyuedadi/200707040173.htm). The other clean fuel, liquefied natural gas (LNG), was also becoming common. The application of clean fuel led to closer complete combustion, and resulted in much less emission from taxis and buses. As a result, the decrease in emission factors of OC and EC (31.3 and 66.9 %) was much higher than the decrease in $PM_{2.5}$ (16.0 %). In general, our results suggest that these strategies are effective to reduce emission factors of $PM_{2.5}$ mass, as well as OC, EC, WSII, and organic compounds in $PM_{2.5}$. However, the total vehicle population increased year by year. As shown in Fig. 5a, the total vehicle population increased by 49.1 % from 2004 to 2013. Total emission of vehicle exhaust of $PM_{2.5}$ mass (calculated as emission factors multiply by annual average driving distance per car and vehicle population (Wu et al., 2012)) increased by 25.2 % from 2004 to 2013 (Fig. 5b). Consequently, we have demonstrated that more stringent emission standards, higher quality of fuel, cleaner fuel used for more vehicles, quicker replacement of high emission vehicles to environment-friendly ones are effective and will be necessary to offset the impacts on the growth in vehicle population and to improve air quality in the PRD region. Additionally, owing to no mandatory national standards to limit metal content from vehicle emission, both emission factors and total emission of most metal increased from 2004 to 2013 (Fig. 3 and 5b). In China, heavy metals, including As, Cr, Cu, Ni and Tl, had been listed as key substances that should be preferentially monitored in the atmospheric environment (SEPA, 2003). These increases

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of metal elements should raise the awareness of the government due to their health concern.

4 Conclusions

$PM_{2.5}$ samples were collected between 10 and 14 August 2013 in Guangzhou Zhujiang Tunnel to acquire a comprehensive snapshot of the chemical characteristics of vehicle emission. The average emission factors of $PM_{2.5}$ mass, EC, OC, WSOC, WSII, metal elements, organic compounds and stable carbon isotopic composition were measured. Stable carbon isotopic composition $\delta^{13}C$ value indicated an isotopic fractionation occurred during fuel combustion. The significant difference between vehicle emission and other emission sources indicate that $\delta^{13}C$ value is an effective marker of motor vehicle emission. Compared with a previous study in Zhujiang Tunnel in year 2004, emission factors of $PM_{2.5}$ mass, EC, OC, and major WSII decreased due to control policy induced changes throughout the nine years from 2004 to 2013, that is, change of fleet composition, implementation of more stringent gasoline and diesel emission standards and improvement in fuel quality, clean fuel used for taxi and buses, and enforced replacement of high emission vehicles to environmental friendly ones that can attain to the more stringent standards. The shift in *n*-alkanes distribution and the overall lower homologous *n*-alkanes and PAHs in $PM_{2.5}$ emission are due to the lower proportion of DV in 2013 than in 2004, and the decrease in emission factor of hopanes were due to the reduction of proportion of heavy duty vehicles. Our analysis show that control polices for vehicles emission by the government were effective to decrease the emission factors of $PM_{2.5}$, EC, OC, and WSII from on-road vehicular fleets. However, the increase in emission of most metal elements should raise the awareness of the government, since metal elements, especially heavy metals could affect human health. Also, to offset the impacts in the growth of vehicle population and to improve air quality in the PRD region, more stringent emission and aggressive control policies are necessary.

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Table 1. Average emission factors (mg vehicle^{-1} km^{-1}) of species in $PM_{2.5}$ in the Zhujiang Tunnel.

Species	Emission factor	SD*
$PM_{2.5}$ mass	92.4	8.9
OC	16.7	1.9
EC	16.4	2.1
WSOC	1.31	0.3
WSII		
F^-	0.02	0.008
Cl^-	4.17	0.9
NO_3^-	0.1	0.03
PO_4^{3-}	0.02	0.02
SO_4^{2-}	0.61	0.1
Na^+	2.88	0.5
NH_4^+	0.17	0.06
K^+	0.06	0.02
Mg^{2+}	0.18	0.02
Ca^{2+}	0.95	0.3
Metal elements		
Li	0.002	0.001
Be	0.0002	0.0001
Na	3.53	0.4
Mg	0.5	0.08
Al	3.15	0.3
K	0.34	0.04
Ca	1.93	0.3
V	0.007	0.0004
Cr	0.01	0.0008
Mn	0.08	0.02
Fe	3.91	0.2
Co	0.002	0.003
Ni	0.008	0.002
Cu	0.09	0.01
Zn	0.16	0.02
Ga	0.04	0.003
As	0.005	0.001
Se	0.0009	0.0003
Rb	0.01	0.002
Sr	0.03	0.003
Ag	0.0001	0.00002
Cd	0.0003	0.00004
Cs	0.0008	0.0002
Ba	0.25	0.03
Tl	0.0001	0.00001
Pb	0.01	0.0007
U	0.0003	0.0002

* SD: standard deviation.

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Table 2. Average emission factors (EF, μg vehicle^{-1} km^{-1}) of organic compounds in $PM_{2.5}$ in the Zhujiang Tunnel.

Species	Abbr.[a]	EF	SD[b]
n-alkanes			
n-Hendecane	C11	0.27	0.12
n-Dodecane	C12	0.33	0.15
n-Tridecane	C13	0.22	0.04
n-Tetradecane	C14	0.50	0.15
n-Pentadecane	C15	0.66	0.17
n-Hexadance	C16	1.10	0.33
n-Heptandecane	C17	1.12	0.23
n-Octadecane	C18	1.35	0.20
n-Nonadecane	C19	1.85	0.18
n-Eicosane	C20	2.84	0.24
n-Heneicosane	C21	4.46	0.40
n-Docosane	C22	7.22	0.65
n-Tricosane	C23	10.6	0.91
n-Tetracosane	C24	13.3	1.11
n-Pentacosane	C25	10.9	0.87
n-Hexacosane	C26	7.98	0.54
n-Heptacosane	C27	7.94	0.62
n-Octacosane	C28	5.76	0.51
n-Nonacosane	C29	3.62	0.59
n-Triacontane	C30	3.36	0.45
n-Hentriacontane	C31	1.35	0.88
n-Dotriacontane	C32	2.10	0.36
n-Tritriacontane	C33	0.50	0.38
n-Tetratriacontane	C34	1.63	0.24
n-pentatriacontane	C35	0.68	0.14
n-Hexatriacontane	C36	0.27	0.08
PAHs			
acenaphthylene	ACY	0.028	0.01
acenaphthene	ACE	0.006	0.001
fluorene	FLO	0.047	0.004
phenanthrene	PHE	0.374	0.04
anthracene	ANT	0.068	0.01
fluoranthene	FLA	0.523	0.06
pyrene	PYR	0.890	0.10
benz[a]anthracene	BaA	0.568	0.06
chrysene	CHR	0.761	0.08
benzo[b]fluoranthene	BbF	0.351	0.05
benzo[k]fluoranthene	BkF	0.133	0.02
benzo[a]pyrene	BaP	0.416	0.05
indeno[cd]pyrene	INcdP	0.209	0.04
dibenzo[ah]anthrathene	DBahA	0.023	0.003
benzo[ghi]perylene	BghiP	0.622	0.09
Hopanes			
22,29,30-trisnorhopane	Ts	1.25	0.09
17αH-22,29,30-trisnorhopane	Tm	1.49	0.11
Norhopane	HP29	5.75	0.42
Hopane	HP30	9.14	0.68
22S-Homohopane	HP31S	3.13	0.22
22R-Homohopane	HP31R	2.17	0.16
22S-Bishomohopane	HP32S	2.06	0.14
22R-Bishomohopane	HP32R	1.61	0.10
22S-Trishomohopane	HP33S	1.54	0.09
22R-Trishomohopane	HP33R	1.02	0.06
22S-Tetrahomohopane	HP34S	1.02	0.07
22R-Tetrahomohopane	HP34R	0.65	0.04
22S-Pentahomohopane	HP35S	0.73	0.04
22R-Pentahomohopane	HP35R	0.46	0.03
Steranes			
C27$\alpha\alpha\alpha$-cholestane(20S)	27$\alpha\alpha\alpha$S	0.46	0.04
C27$\alpha\beta\beta$-cholestane(20R)	27$\alpha\beta\beta$R	0.69	0.05
C27$\alpha\beta\beta$-cholestane(20S)	27$\alpha\beta\beta$S	0.55	0.04
C27$\alpha\alpha\alpha$-cholestane(20R)	27$\alpha\alpha\alpha$R	0.56	0.05
C28$\alpha\alpha\alpha$-ergostane (20S)	28$\alpha\alpha\alpha$S	0.31	0.03
C28$\alpha\beta\beta$-ergostane (20R)	28$\alpha\beta\beta$R	0.60	0.05
C28$\alpha\beta\beta$-ergostane (20S)	28$\alpha\beta\beta$S	0.49	0.05
C28$\alpha\alpha\alpha$-ergostane (20R)	28$\alpha\alpha\alpha$R	0.55	0.05
C29$\alpha\alpha\alpha$-stigmastane(20S)	29$\alpha\alpha\alpha$S	0.85	0.07
C29$\alpha\beta\beta$-stigmastane(20R)	29$\alpha\beta\beta$R	0.97	0.07
C29$\alpha\beta\beta$-stigmastane(20S)	29$\alpha\beta\beta$S	0.84	0.06
C29$\alpha\alpha\alpha$-stigmastane(20R)	29$\alpha\alpha\alpha$R	0.72	0.06

[a] Abbr.: abbreviation; [b] SD: standard deviation.

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Table 3. $\delta^{13}C$ values (‰) of PM from vehicle emission in this study and other emission sources.

Emission sources and Sampling site	Particle types	$\delta^{13}C$ values	Sampling time	Reference
Vehicular fuel emission				
Vehicle emissions (Zhujiang Tunnel, China)	$PM_{2.5}$/TC	−25.0 ± 0.3	Aug 2013	This study
Vehicle emissions (Tunnel of Rio de Janeiro, Brazil)	PM/OC	−25.4	Apr 1985	Tanner and Miguel (1989)
Vehicle emissions (Tunnel of Rio de Janeiro, Brazil)	PM/EC	−24.8	Apr 1985	Tanner and Miguel (1989)
Complete combustion of diesel	PM/TC	−29	N/A	Widory (2006)
Complete combustion of gasoline	PM/TC	−27	N/A	Widory (2006)
Vehicle emissions (Cassier Tunnel, Canada)	$PM_{2.5}$/OC	−27.1	N/A	L. Huang et al. (2006)
Vehicle emissions (Cassier Tunnel, Canada)	$PM_{2.5}$/EC	−26.9	N/A	L. Huang et al. (2006)
Diesel vehicle emissions (Central Camionera del Norte, Mexico)	$PM_{2.5}$/TC	−24.6 ± 0.3	Mar 2002	Lopez-Veneroni (2009)
Gasoline vehicle emissions (Tunnel of Avenida Chapultepec, Mexico)	$PM_{2.5}$/TC	−25.5 ± 0.1	Mar 2002	Lopez-Veneroni (2009)
Vehicle emissions (Mount Victoria Tunnel, New Zealand)	$PM_{2.5}$/TC	−25.9 ± 0.8	Dec 2008 to Mar 2009	Ancelet et al. (2011)
Non-vehicular fuel sources				
Coal combustion (Paris, Franch)	$PM_{2.5}$/TC	−23.9 ± 0.5	May to Sep 2002	Widory et al. (2004)
Coal combustion (Yurihonjo City, Japan)	$PM_{2.5}$/EC	−23.3	N/A	Kawashima and Haneishi (2012)
Charcoal combustion (Yurihonjo City, Japan)	$PM_{2.5}$/EC	−27.4 ± 1.7	N/A	Kawashima and Haneishi (2012)
Fireplace soot (Yurihonjo City, Japan)	PM/EC	−26.5 ± 0.1	N/A	Kawashima and Haneishi (2012)
Fuel oil combustion (Paris, Franch)	$PM_{2.5}$/TC	−26.0 ± 0.5	May to Sep 2002	Widory et al. (2004)
Dust particles				
Street dust (Mexico City, Mexico)	$PM_{2.5}$/TC	−21 ± 0.2	Mar 2002	Lopez-Veneroni (2009)
Street dust (Yurihonjo City, Japan)	$PM_{2.5}$/EC	−18.4 ~ −16.4	Nov 2009	Kawashima and Haneishi (2012)
Biomass burning				
C4 Plant	PM/TC	−13 ± 4	N/A	Boutton (1991)
C4 Plant (Yurihonjo City, Japan)	$PM_{2.5}$/EC	−19.3 ~ −16.1	Apr to Nov 2009	Kawashima and Haneishi (2012)
C3 Plant	PM/TC	−27 ± 6	N/A	Boutton (1991)
C3 Plant (Yurihonjo City, Japan)	$PM_{2.5}$/EC	−34.7 ~ −28.	Apr to Nov 2009	Kawashima and Haneishi (2012)

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Table 4. Vehicle emission standard and limit for PM and NO_x implemented in Guangzhou after 2000.

Emission standard	Year[a]	Limit for PM		Limit for NO_x	
		g km^{-1}[b]	g kWh^{-1}[c]	g km^{-1}[b]	g kWh^{-1}[c]
China I	2001	0.14 ~ 0.40	0.40 ~ 0.68	–	8.0 ~ 9.0
China II	2004	0.08 ~ 0.20	0.15	–	7.0
China III	2007	0.05 ~ 0.10	0.10 ~ 0.21	0.15 ~ 0.78	5.0
China IV	2010	0.025 ~ 0.060	0.02 ~ 0.03	0.08 ~ 0.39	3.5

[a] Year of implementation; [b] for light duty vehicle; [c] for compression ignition and gas fuelled positive ignition engines of vehicles.

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Table 5. Vehicle fuel standard and limit for sulfur content (mg kg^{-1}) implemented in Guangzhou after 2000.

Standard	China I		China II		China III		China IV	
	Limit	Year[a]	Limit	Year	Limit	Year	Limit	Year
Gasoline	1000	2001	500	2005	150	2006	50	2010
Diesel	2000	2002	500	2003	350	2010	50	2013
LPG	–	–	270[b]	2003	–		50	2013

[a] Year of implementation; [b] unit: mg m^{-3}.

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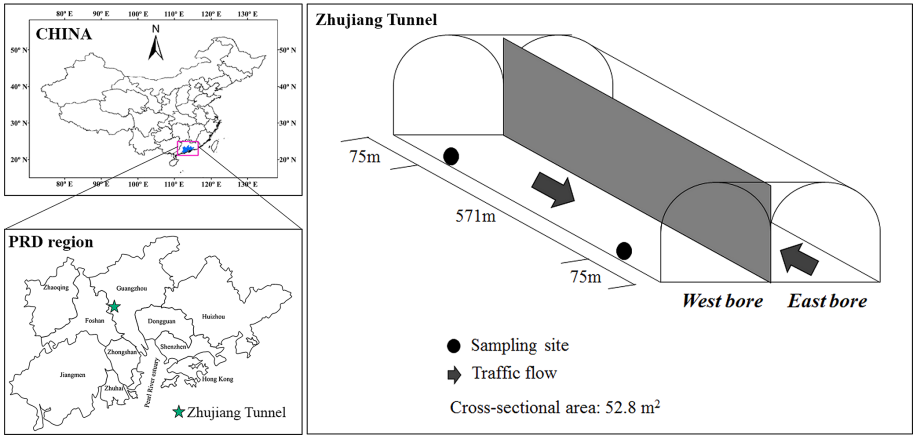

Figure 1. Sampling schematic diagram of the Zhujiang Tunnel.

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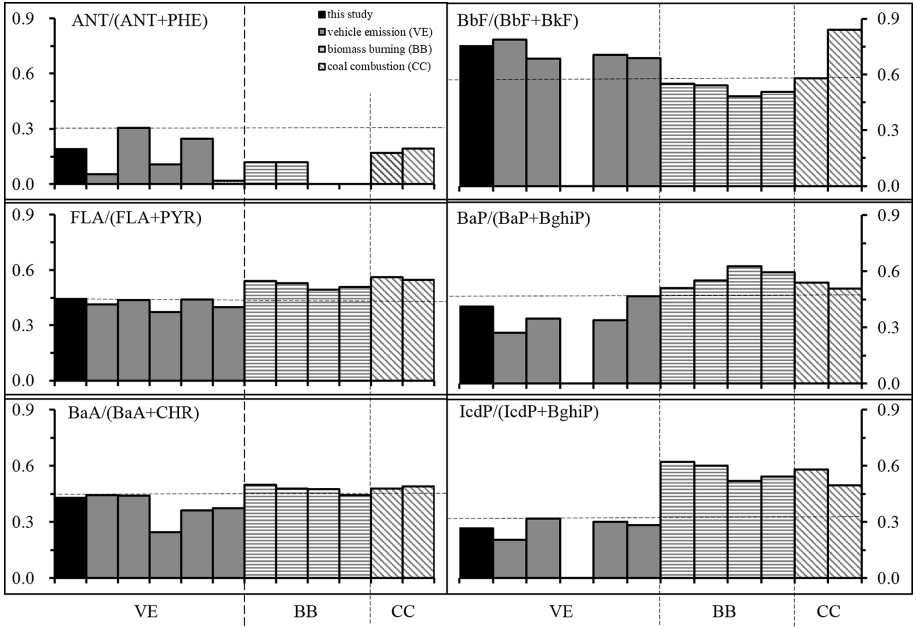

Figure 2. ANT/(ANT + PHE), FLA/(FLA + PYR), BaA/(BaA + CHR), BbF/(BbF + BkF), BaP/(BaP + BghiP) and IcdP/(IcdP + BghiP) ratios for three source emissions. The vehicle emission (VE) composition is from data collected in roadway tunnel (this study), (He et al., 2008), (Ancelet et al., 2011), (He et al., 2006), (Ho et al., 2009), (Oda et al., 2001). The biomass burning profiles are obtained from 9 straws (Shen et al., 2011), 26 firewood (Shen et al., 2012), 3 plant leaves and branches (Sheesley et al., 2003) and 2 biomass briquettes burning (Sheesley et al., 2003). The coal combustion profiles are selected from 5 coals imitate combustion (Shen et al., 2011) and main coal-mining regions in China (Zhang et al., 2008).

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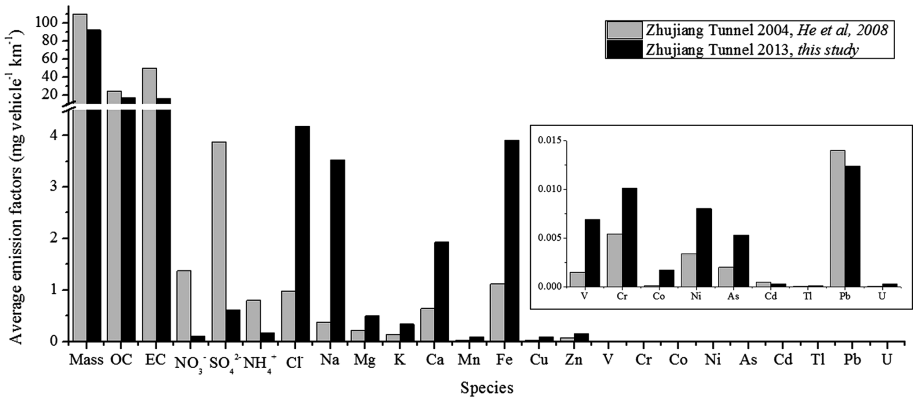

Figure 3. Comparison of $PM_{2.5}$, OC, EC, WSII and metal emissions in the Zhujiang Tunnel sampling in 2004 and 2013.

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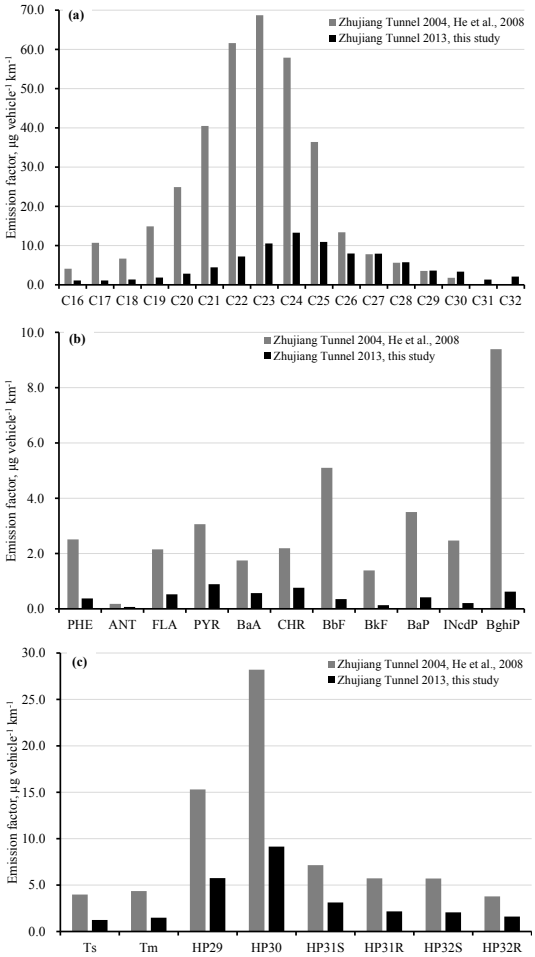

Figure 4. Comparison of organic compounds emissions in the Zhujiang Tunnel sampling in 2004 and 2013.

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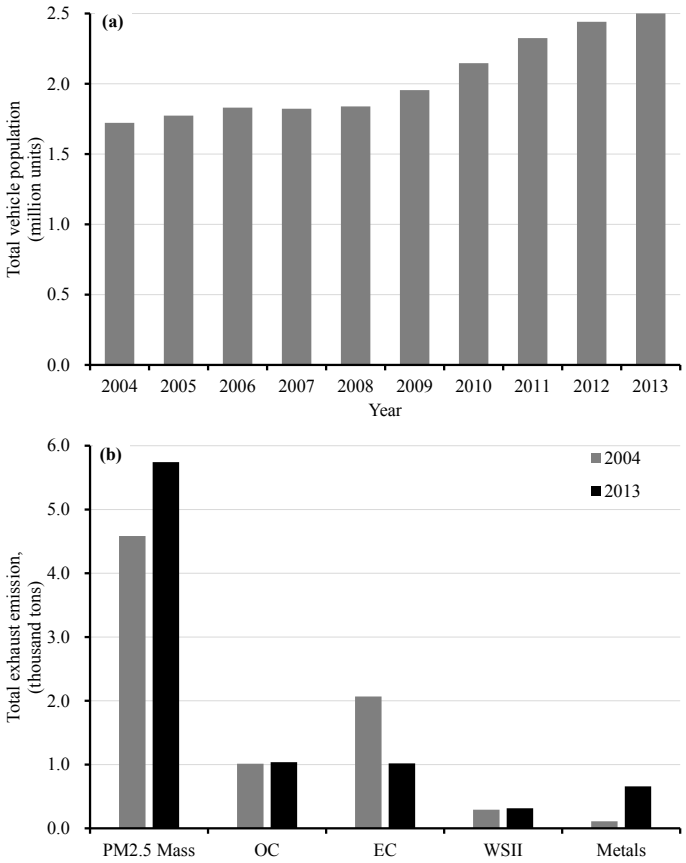

Figure 5. (a) Growth in total vehicle population in Guangzhou during 2004–2013. **(b)** Total exhaust emission of $PM_{2.5}$ mass, OC, EC, WSII and metal in 2004 and 2013.

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