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Research article

Heavy truck restrictions and air quality implications in São Paulo, Brazil

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ABSTRACT

This study quantified the effects of traffic restrictions on diesel fuel heavy vehicles (HVs) on the air quality of the Bandeirantes corridor using hourly data obtained by continuous monitoring of traffic and air quality at sites located on this avenue. The study addressed the air quality of a city impacted by vehicular emissions and that PM_{10} and NO_X concentrations are mainly due to diesel burning. Data collection was split into two time periods, a period of no traffic constraint on HVs (Nov 2008 and 2009) and a period of constraint (Nov 2010, 2011 and 2012). We found that pollutants on this corridor, mainly PM_{10} and NO_X, decreased significantly during the period from 2008 to 2012 (28 and 43%, 15.8 and 86.9 ppb) as a direct consequence of HV traffic restrictions (a 72% reduction). Rebound effects in the form of increased traffic of light vehicles (LVs) during this time had impacts on the concentration levels, explaining the differences between rates of reduction in HV traffic and pollutants. Reductions in the number of trucks resulted in longer travel times and increased traffic congestion as a consequence of the modal shift towards LVs. We found that a 51% decrease in PM_{10} (28.8 μg m^{-3}) was due to a reduction in HV traffic (vehicle emissions were estimated to be 71% of total sources, 40.1 μg m^{-3}). This percentage was partially offset by 10% more PM_{10} emissions related to an increase in LV traffic, while other causes, such as climatic conditions, contributed to a 13% increase in PM_{10} concentrations. The relationships analyzed in this research served to highlight the need to apply urban transport policies aimed at decreasing pollutant concentrations in São Paulo, especially in heavily congested urban corridors on working days.

1. Introduction

The Metropolitan Region of São Paulo (MRSP) is located near the southeastern Brazilian coast (23.30° S 46.37° O). The MRSP has a population of more than 20 million within an area of 8511 km^2 (IBGE, 2016). It is one of the largest megacities in the world and the biggest conurbation in the southern hemisphere. Most of the population is concentrated in an area of 1000 km^2 (Fig. 1). There are approximately 6.5 million vehicles (630 vehicles per 1000 inhabitants), of which 85% are light gasoline vehicles (LVs), 3% are heavy diesel vehicles (HVs) and 12% are motorcycles. The climate is subtropical with a mean annual air temperature of 19.3 °C. The MRSP has mild winters and summers with moderate to high temperatures, accentuated by the effect of the high concentration of buildings. Air pollutant concentrations have been an important concern during past years. Andrade et al. (2012) and Miranda et al. (2012) performed a source apportionment analysis in six Brazilian state capitals, including the MRSP, and found that high traffic volumes are responsible for the largest percentage of pollutant concentrations: vehicle emissions in the MRSP explained 40% of the particle matter (PM) mass. WHO (2015) reported that traffic is the main contributor to urban ambient PM and estimated a relative share of 34% at the Brazilian urban sites. Similarly, the State Environmental Protection Agency CETESB (2013a) reported a pollution source contribution by HVs to total PM of 36%. The MRSP is seriously impacted by emissions from gasoline and diesel motor vehicles. LV, HV and motorcycle fleets increased by 12.7%, 10% and 9.6% between 2009 and 2012 (CETESB, 2012). The growth of the vehicle fleet was coupled with a 58% increase in carbon dioxide (CO_2) emissions from 2007 to 2013, according to official inventories

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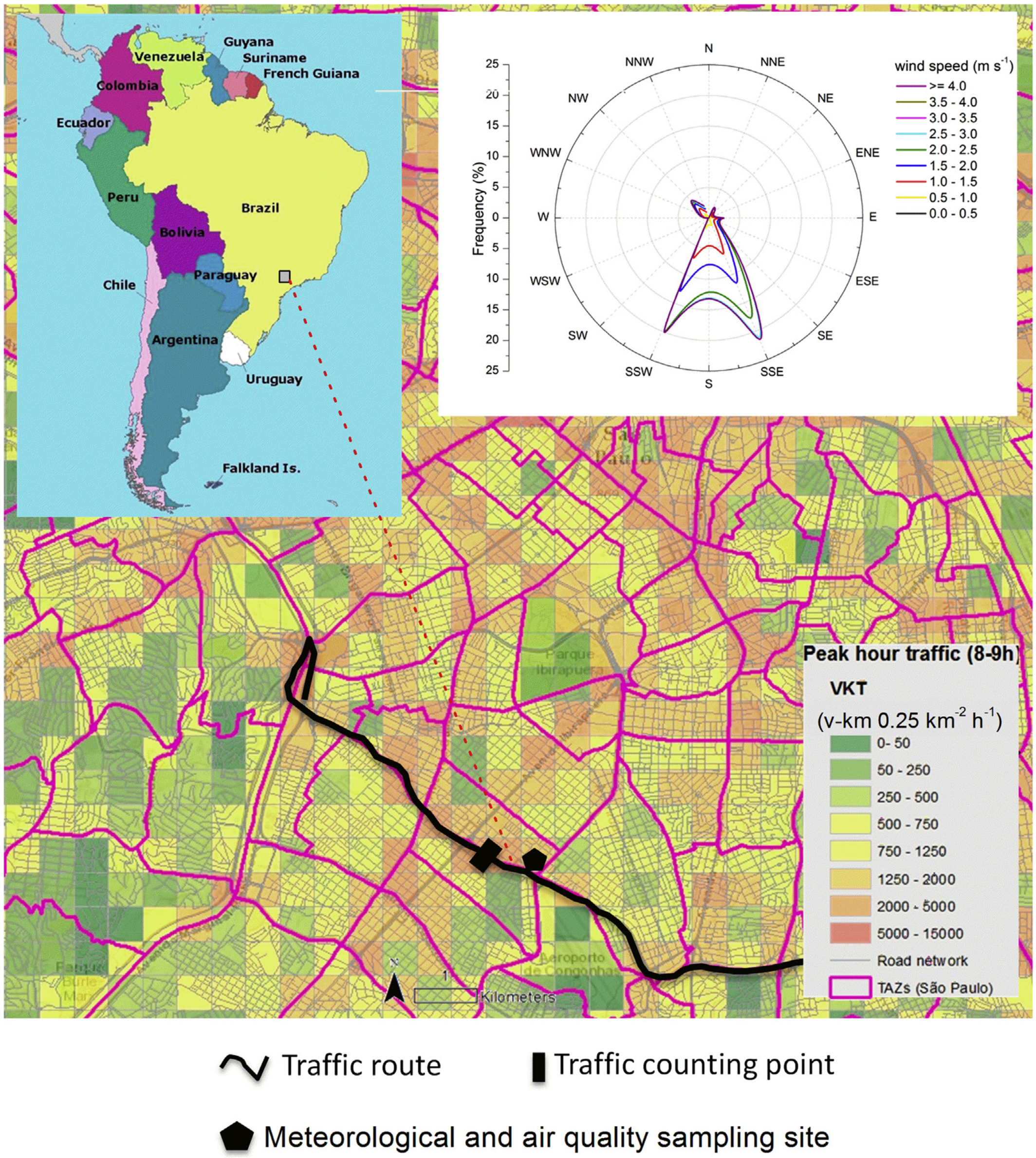

Fig. 1. Bandeirantes Avenue and location of the traffic, meteorology and air quality sampling site. Location of the site and map of the surroundings of CETESB's air quality and meteorological sampling site (black pentagon for the Congonhas station), traffic route (black line for CET route 19G) and traffic counting point (black square for CET point 3 on route 19G). Route 19G in Fig. 1 has 11 segments, including in both directions (Anchieta-Pinheiros Marginal and Pinheiros-Anchieta Marginal) the following avenues: Bandeirantes Ave, Affonso D'Escragnolle Taunay Ave, Maria Maluf Ave and Pres. Tancredo Neves Ave. Volume counting point 3 is located at Bandeirantes Avenue between dos Nhambiquaras Alley and dos Maracatins Alley (the segment starts at Boulevard João J. C. Aguilar and ends at Bandeirantes Ave and has a length of 1070 m). Fig. 1 unpublished using mobility data from the METRÔ-SP (2013).

conducted by the CETESB (2013a). This increase was associated with the increasing use of fossil fuels: 84% gasoline, 4% ethanol and 42% diesel. The small difference between the fleet and CO_2 emissions and increases in fossil fuel was associated with the shift in carbon intensity as the difference between the reductions in the vehicle carbon emission factor and the fuel efficiency (Kirchstetter et al., 1999; Kean et al., 2002; D'Angiola et al., 2010; Franco et al., 2013; Smit et al., 2013).

Recent work has also reported significant improvements in air quality concentrations related to the use of more efficient vehicle technologies (Millstein and Harley, 2010; Dalmann and Harley, 2010; McDonald et al., 2012). The challenge is to continue improving air quality and meet the requirements of the World Health Organization (WHO, 2016)[1] and São Paulo State Directives (CETESB, 2014).[2] The improvement in air quality is crucial since the deterioration of air quality standards is responsible for serious health problems, especially in urban site environments (OECD, 2015). Some studies have shown that the values for particle matter less than 10 μm (PM_{10}) and nitrogen dioxide (NO_2) standards are more restrictive or do not even exist in different urban

[1] Final Phase (FP): 20 μg m^{-3} for PM_{10} (anual mean), 40 μg m^{-3} for NO_2 (annual mean), 9 ppm for CO (8-h mean) and 100 μg m^{-3} for O_3 (8-h mean).

[2] Intermediate Phase I (PI): 40 μg m^{-3} for PM_{10} (annual mean), 60 μg m^{-3} for NO_2 (annual mean), 9 ppm for CO (8-h mean) and 140 μg m^{-3} for O_3 (8-h mean).

environments (Martins et al., 2010; Guaita et al., 2011; Jiménez et al., 2012; Hori et al., 2012). Several policy measures and environmental control programs were implemented in the MRSP to address the environmental problems caused by the growth of vehicle fleets. These measures intended to reduce emissions by LV and HV traffic and ensure compliance with PM_{10} and NO_2 air quality standards on a regional scale (Jacobi et al., 1999; da Silva et al., 2012; CETESB, 2013b; Silveira et al., 2015). These measures and programs included the limits of vehicle emission standards for new vehicles in Brazil (Program for the control of air pollution emissions by motor vehicles, PROCONVE), LV and HV traffic restrictions, investments in public transport infrastructure, reductions in the number of high emission vehicles and increase in vehicle purchase taxes. Traffic restrictions on the driving of LVs on the city's ring roads one non-holiday weekday per week from 7–10 a.m. to 5–8 p.m. and on the driving of all HVs from 5–9 a.m. to 5–10 p.m. on non-holiday weekdays were determined by the last number of the license plate. It is estimated that 20% (0.7 million cars) of registered LVs can be restricted each day as a direct result of the implementation of “rodízio” in São Paulo (PMSPa, 2015). There is no accurate estimate of the share of registered HVs that can be restricted each day, since trucks anticipate or delay their trips (and even change their routes) according to traffic constraints (PMSPb, 2015). The contribution from the industrial sector to total emissions has decreased by 80%, as the industries have moved away from the MRSP due to the high taxes applied to the industrial sector (CETESB, 2012). The annual mean nitrogen oxides (NO_X) and PM_{10} concentrations in the MRSP decreased from 67.3 ppb to 48.4 $\mu g\ m^{-3}$ in 2000 to 45.3 ppb and 33.2 $\mu g\ m^{-3}$ in 2013 (Pérez-Martínez et al., 2015), mainly as a result of transport emission policies such as the Diesel Vehicle Maintenance Improvement Program PMMVD (CETESB, 2013b; Silveira et al., 2015). Alternative measures to restrain the use of the private car and trucking (such as urban congestion tolls, implementation of rodízio, reduction in available traffic lanes and car sharing), also aid in improving air quality in the urban environment (Jacobi et al., 1999; da Silva et al., 2012). The actual pollution levels are above the WHO requirements (40 $\mu g\ NO_2\ m^{-3}$ and 20 $\mu g\ PM_{10}\ m^{-3}$ in the Final Phase-FP) and fulfill the São Paulo State Directives (60 $\mu g\ NO_2\ m^{-3}$ and 40 $\mu g\ PM_{10}\ m^{-3}$ in the Intermediate Phase I-PI) for the established air quality standards (final and intermediate phases).

These levels reflect the PROCONVE compliance of implementing relevant vehicle emission limits[3] (Pérez-Martínez et al., 2015). Regional pollutant reductions have also been reported by several research projects and studies (Martins et al., 2006; Gallardo et al., 2012; Vara-Vela et al., 2015). It is challenging to assess the effects of control measures on metropolitan regions. Meteorological conditions significantly affect the city's air quality, making reference to some relevant air quality and pollution research in Brazil and elsewhere (Gokhale, 2011; Guttikunda, and Gurjar, 2012; Kuznetsova, 2012; Pérez-Martínez and Miranda, 2015).

The Bandeirantes Avenue corridor connects the city to the coast and the country's main seaport, Porto de Santos. Most of the goods transported to the port before the building of the Rodoanel beltway had to go via the Bandeirantes Avenue. During the period of this study, the air quality standards were not fulfilled. In this paper, we studied the impacts of restricting HV traffic on the concentrations of ground-level pollutants during weekdays in the Bandeirantes Avenue corridor over the 2008–2012 period. For this purpose we used hourly peak data of air concentrations and traffic counts (CETESB, 2014; CET, 2013) from the CETESB air quality monitoring site of Congonhas and traffic count datasets from the São Paulo Municipality *Companhia de Engenharia de Trânsito* (CET, Traffic Engineering Company). To date, few studies have analyzed specifically the relationships between transport policies and pollutant concentrations in São Paulo, and we want to address this research gap here and contribute knowledge to this field. By analyzing these pollutant concentrations, we determined whether, despite increases in LV traffic and changing weather conditions, changes in air quality levels could be related to adopted limits on HV traffic in this urban corridor. In this statistical analysis, it was important to assess the importance of transport related air quality policies. We isolated the effects of other changes in air quality impacts. In our approach, we adopted a general linear model (GLM) in order to determine the relative effects of HV traffic on air quality separately from other effects such as LV traffic and driving and meteorological conditions. A constraint on HV traffic, in response to stricter transport policies, could be accompanied by a local decrease in on-road pollutant emissions and diesel consumption. This constraint could be coupled with a contemporaneous decrease in the concentrations of primary ground-level pollutants: NO_X and PM_{10}. Changing driving conditions and the increase in LV traffic flows could partially offset the effect of HV limit on air quality. Carbon monoxide (CO) and NO_2 and secondary pollutants such as ozone (O_3) were also considered in our analysis. The reduction in HV traffic and the lower concentrations of NO_X could be important in the increase in O_3 concentrations. This study presented only a case study on a corridor in the megacity and no universal law could be drawn from our main results. Another limitation of this study is related to fine particulate matter less than 2.5 μm ($PM_{2.5}$). Although $PM_{2.5}$ is a more important indicator of impact on health than PM_{10}, and tailpipe particle emissions are also dominated by fine particulate matters (Karagulian et al., 2015), $PM_{2.5}$ was only included in São Paulo State's legislation in 2013 and monitoring data is only available at the research site after that date (CETESB, 2014).

2. Methods and data

2.1. Monitoring site

The traffic, meteorological and air quality measurements were taken on Bandeirantes Avenue, an eight-lane urban expressway in downtown São Paulo. The meteorological and air quality monitoring site was located on the southwest side of the avenue (Congonhas station, 23.37° S 46.40° O) and the air quality sampling data was provided by CETESB (2013a,b). Gasoline and ethanol LVs (cars and motorcycles) and diesel HVs (urban buses, interurban buses, 2-axle, 3-axle and 4 or + axle trucks) use this avenue. A map of the MRSP sampling site and its surroundings is shown in Fig. 1. The figure shows peak-hour traffic (sum of LVs and HVs) in the MRSP (8–9 h), expressed in vehicle kilometer travelled (VKT) per 0.25 km^{-2} and h^{-1}, together with the city transport area zones (TAZs) and the road network. Most of the traffic leads to downtown and to the inner parts of the MRSP where the study corridor is located. Fig. 1 provides location details on the HV and LV traffic route (black line for CET route 19G) and traffic flow counting point on Bandeirantes Avenue (hollow black square for CET point 3 on route 19G). Fig. 1 also shows the local wind rose in November 2009 at the Congonhas CETESB station: distribution of wind speed ($m\ s^{-1}$) and frequency of wind direction (%). We divided wind direction into three categories: upwind (wind blowing to the road and in the direction of the coast, $0° \le \alpha \le 90°$ and $315° \le \alpha \le 360°$), downwind (wind blowing from the road and from the coast, $135° \le \alpha \le 270°$) and wind blowing parallel to the road's axis and calm wind ($90° < \alpha < 135°$ and $270° < \alpha < 315°$). Hourly mean analysis

[3] 0.08 g $NO_X\ km^{-1}$ and 0.03 g PM km^{-1} for LVI LVs and 2.00 g $NO_X\ kWh^{-1}$ and 0.02 g PM kWh^{-1} for P7 HVs.

was used to define whether there were differences between the three categories (statistical values were means ± standard deviations). The air quality station was mainly influenced by the corridor's road emissions and traffic flow changes. Finally, Fig. 2 shows vehicle peak flows at the counting point 3 during consecutive campaigns from Nov 2008 to Nov 2012 by HV and LV type (dark/light black lines and dots represent HVs and dark/light blue lines and dots represent LVs).

2.2. Data acquisition

In this paper, meteorological parameters such as relative humidity (RH), wind speed (w_s), direction (w_d), atmospheric pressure (P), solar radiation (R) and temperature (T) and pollutant concentrations such as PM_{10}, NO_X, nitrogen monoxide (NO), NO_2, CO and O_3 were considered. These parameters were linked to transport sources (cars, motorcycles, urban and interurban buses and 2 to 4 or more axle trucks) and related fossil fuel engine emissions and emission factors. We used meteorological and pollutant data, hourly average values from the Congonhas station provided by CETESB's air quality network (2013a,b). Meteorological and pollutant data were measured about 25 m to the east of the Bandeirantes Avenue traffic counting point on the CET route. The instruments were located on the northeast side of the avenue, which was the mostly downwind side based on the predominant wind direction in the MRSP (Fig. 1). CETESB used an automatic weather station to measure hourly average data for RH, w_s, w_d, P, R and T. Measurements of the mass concentration of PM_{10} were taken by beta-gauges (5014i-Beta). Measurements of NO_x, CO and O_3 were carried out with NO_x (Thermo electron 42i-HL), CO (Thermo electron 48B) and O_3 (Thermo 49i) analyzers. Meteorological and air quality measurements were shown for two different periods, the period of no constraint and that of restriction of HV traffic (Fig. 3). These periods will be used to analyze the effect of transport policies on air quality improvement. Measurements were obtained during five time periods from November 2008 to November 2012 (morning 7:00–9:00 and evening 17:00–19:00 peak traffic hours on work days): November 27th–28th and December 1st–2nd (2008, 4 days), November 11th–13th and November 16th–18th (2009, 6 days), November 9th–12th (2010, 4 days), November 22th–25th (2011, 4 days) and November 20th–23th (2012, 4 days). In total we considered 132 peak-hour traffic observations distributed over 22 days (Table 1). Data were split into two periods and years 2008–2009 and 2010–2012, corresponding to periods of no traffic constraint and traffic constraint on HVs, respectively. We use set observations of contemporaneous measurements of road traffic flows, meteorology and pollutant concentrations to demonstrate that ambient NOx concentrations fell as the trucks were mainly banned from the corridor. On the contrary, O_3 concentrations increase due to lower concentrations of fresh NO. We use November as a study period from 2008 to 2012 due to traffic data availability: CET (2013) controls for vehicle traffic congestion and speed and performs a monitoring survey of the main road system every year in November during the morning and evening peak hours. We combine the extensive traffic-meteorological-pollutant data during the peak hours to stress the influence of HV flow changes over other effects. Finally, we compare the traffic peak-hour estimates and near-road air quality trends with observations and annual trends from 2008 to 2012 to check for potential consistency.

We also used vehicle speed (VS), travel time (TT) and transportation delay (TD), obtained by CET (2013) on a road segment basis, as the measures of traffic driving conditions in the MRSP. TD is an aggregate index of road-segment delay that is estimated using VS and TT of floating cars on Bandeirantes Avenue, weighted according to the traffic flows on the road segment close to the meteorological and pollutant monitoring station (Fig. 2). TD is measured as the percentage ζ (%) of TT subject to congestion episodes and traffic lights and is scaled between 0% (no congestion and free flow) to 100% (complete jam and interrupted flow):

$$TD = \zeta \cdot TT \tag{1}$$

Transportation delays are daily averages for morning (MP) and evening peak hours (EP) during the traffic-sampling period in both segment directions. Fig. 2 includes panels for the route's segment length (1070 m), which show traffic flows during peak hours together with mean MP and EP traffic-related parameters (VS, TT and TD). Traffic counts by peak hour were split into the two HV traffic limit periods: from November 27, 2008 to Nov 18, 2009 (no limitation, dark black and blue colors) and from Nov 9, 2010 to November 23, 2012 (limitation, light black and blue colors).

2.3. Statistical model

We used hourly values for traffic, average meteorological and traffic parameters and mean variables of the pollutants (PM_{10}, NO_X, NO, NO_2, CO and O_3) to study the relationships between the mean variables of interest and air quality. To study the effects of climatic conditions, driving conditions and vehicle composition on air quality, we used a statistical model (Table 1). We applied a multivariate general linear regression model (GLM) of the Bandeirantes Avenue data using exogenous time variation in transport policies, driving and weather conditions. The GLM was constructed following the recommendations found in Yang et al. (2011), Sun et al. (2014) and Salvo and Geiger (2014), using the version 21.0 SPSS software (IBM, USA). For instance, the situation was to compare air quality across subsamples, which differed only by driving conditions but were similar in regards to other air quality factors. The GLM predicted air quality at time of day *t* using samples obtained during periods when HV traffic policies and/or driving conditions varied. We used the following linear regression model:

$$\begin{aligned} Air - quality_{i,t} = &\theta_0 + \theta_1 \cdot P_t + \theta_2 \cdot D_t + W_t' \cdot \theta_3^W \\ &+ T_t' \cdot \theta_4^T + \theta_5 \cdot PD_t + \varepsilon_t \end{aligned} \tag{2}$$

where P_t and D_t are dummy variables which change from 0 to 1, reflecting the transport policy *P* related to the HV traffic constraint and the driving condition *D* associated with travel times less than 120 s and vehicle speeds more than 25 km h^{-1}; θ_0, θ_1 and θ_2 are regression coefficients. As transport policies and driving conditions may have certain characteristics based on year, day of the week and hour of the day, we included these two dummy variables to control the effects on air quality. We also included meteorological variables – relative humidity (%), wind speed (m s^{-1}) and air temperature (°C) – and transport continuous variables – cars h^{-1}, 4-axle trucks h^{-1} and urban buses h^{-1} – to control for weather and traffic events. Therefore W_t' and T_t' are vectors of meteorological and traffic records, which may affect air quality, and θ_3^W and θ_4^T are regression coefficients. To account for hourly variation and seasonal trends, we included a final term in the model to represent the period of the day *PD*, evening vs. morning peak hours, as a fixed time-varying effect.

2.4. Emissions of PM_{10}

To determine the emission factors (EFs) of particle matter (PM_{10}) from the air quality concentration measurements of traffic emissions on Bandeirantes Avenue, we used an estimation based on hourly dilution rates (D_{PM10}), expressed in m^2 h^{-1}. D_{PM10} was estimated using NO_X as a tracer gas based on the fact that EFs of NO_X

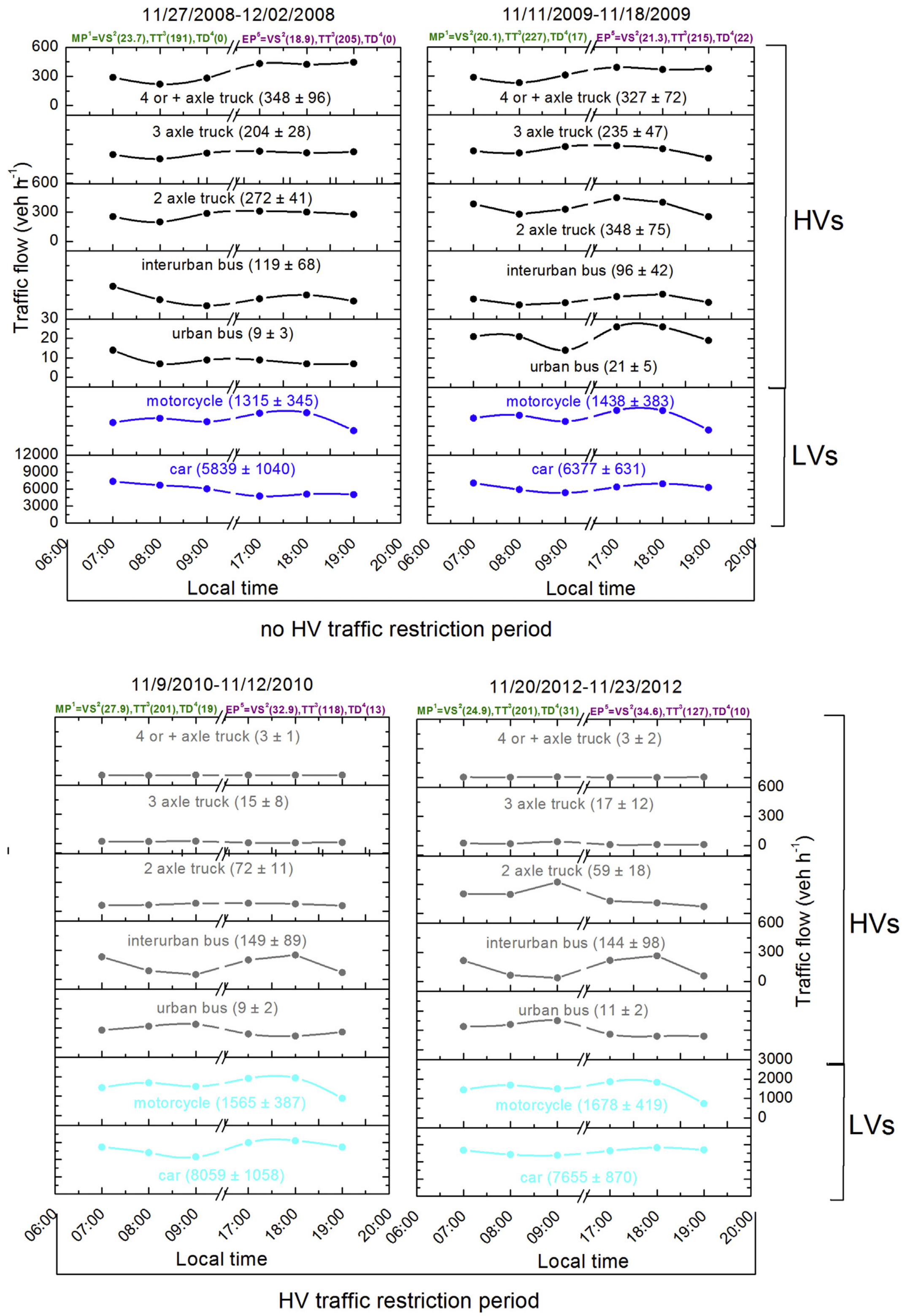

Fig. 2. Hourly analysis of peak-hour traffic flows. Notes: The Congonhas station is located at the address dos Tupiniquins Alley, 1571 (Prof. J.C. da S. Borges Municipal School - Pto. Paulista). Panels include traffic flow means ± standard deviations during all the peak hours together with mean morning peak (MP, green letters and numbers)[1] and evening peak (EP, purple letters and numbers)[5] traffic-related parameters: vehicle speed (VS, in km h^{-1})[2], travel time (TT, in s)[3] and transportation delay (TD, in %)[4]. Source: CET (2013) and CETESB (2014). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

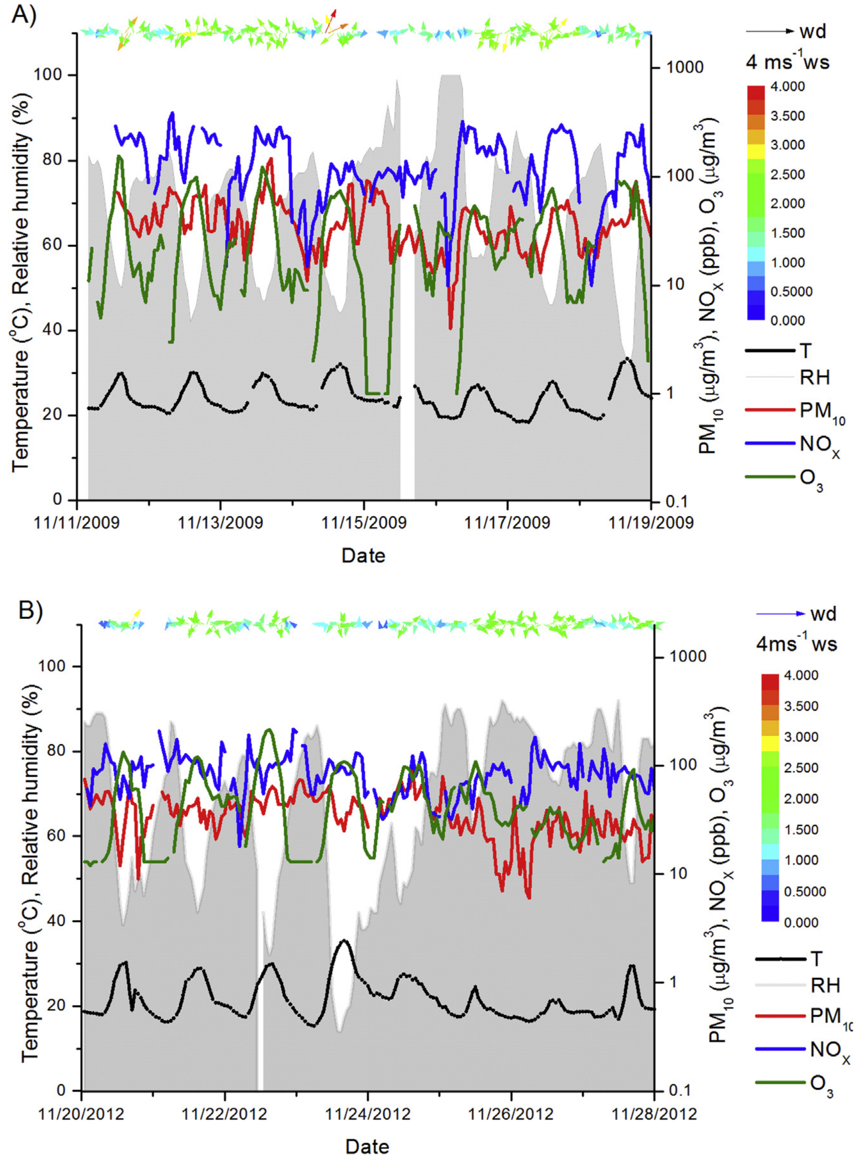

Fig. 3. Hourly average meteorological and air quality data during the periods of no HV traffic restriction (2009) and traffic restriction (2012). a) No traffic constraint (November 11–19, 2009). b) Traffic constraint (November 20–28, 2012). T: air temperature; RH: relative humidity; wd: wind direction; ws: wind speed.

are well known from previous studies (Pérez-Martínez et al., 2015; Martins et al., 2006). We assumed that the particle emissions and NO_X are transported simultaneously to the measurement station at the corridor's kerbside (Ferm and Sjöberg, 2014). According to Imhof et al. (2005) dilutions of NO_X and PM_{10} are coincident and it is possible to determine D_{PM10} using mean average *EFs* of NO_X (EF_{NOX} in g km^{-1}), separated by vehicle category (LVs and HVs), vehicle distribution (η_{LV} and η_{HV} in vehicles h^{-1}) and concentration difference (ΔNO_x in µg m^{-3}) – hourly mean NO_X differences between the monitoring station and a background concentration of 10 µg m^{-3} in the MRSP (CETESB, 2014). Consequently, EF_{PM10} can be estimated using the following equation:

$$EF_{PM10} = \frac{\Delta PM_{10} \cdot D_{PM10}}{\eta_{total}} \tag{3}$$

where ΔPM_{10} is the concentration difference for PM_{10}, considering a background concentration of 4-µg m^{-3} in the MRSP (CETESB, 2014), and η_{TOTAL} is the traffic of vehicles at the sampling site (point 3) corresponding to a segment length of 1.07 km (Figs. 1 and 2) in vehicle kilometers per hour (v-km h^{-1}, $\eta_{TOTAL} = \eta_{LV} + \eta_{HV}$). To separate the contributions of each vehicle category, we estimate the coefficients f_1 ($EF_{PM10(LV)}$) and f_2 ($EF_{PM10(HV)}$) of the linear regression model:

$$EF_{PM10} = f_1 \cdot \eta_{LV} + f_2 \cdot \eta_{HV} \tag{4}$$

In equation (3), D_{PM10} (m^2 h^{-1}) is estimated using the mean average *EFs* of LVs (0.5 g NO_X km^{-1}) and HVs (11.9 g NO_X km^{-1}) from the study by Pérez-Martínez et al. (2014) and the NO_X concentration differences simultaneous to the traffic counts (η_{LVs} and η_{HVs}):

$$D_{PM10} = \frac{0.5 \cdot \eta_{LV} + 11.9 \cdot \eta_{HV}}{\Delta NO_x} \tag{5}$$

In equations (3) and (5), concentrations are estimated as the differences between the corridor's kerbside and urban background concentrations, assuming that measuring times during peak hours can be studied independently of wind direction (Imhof et al., 2005).

Table 1
Summary statistics of characteristic meteorological data, pollutant concentrations and vehicle flows for the periods of no HV traffic restriction and traffic restriction.

Sampling times[1,2] (units)	**RH**[3] (%)	$\mathbf{w_s}$[4] ($m\ s^{-1}$)	$\mathbf{w_d}$[5] (°)	**T**[6] (°C)	$\mathbf{PM_{10}}$ ($\mu g\ m^{-3}$)	$\mathbf{NO_X}$ (ppb)	**NO** ($\mu g\ m^{-3}$)	$\mathbf{NO_2}$ ($\mu g\ m^{-3}$)	**CO** (ppm)	$\mathbf{O_3}$ ($\mu g\ m^{-3}$)
No traffic restriction period (peak hours: morning 7–9 h and evening 17–19 h)										
11/27 to 28 and 12/1 to 2/2008	65.01± 9.99[7]	2.09 ± 0.54	164.17 ± 65.70	22.06 ± 3.08	49.7 ± 18.1	185.9 ± 82.1	184.9 ± 85.8	84.0 ± 17.9	1.7 ± 0.5	23.9 ± 12.0
11/11 to 13 and 11/16 to 18/2009	66.92 ± 10.33	1.88 ± 0.52	167.69 ± 48.01	23.36 ± 1.25	46.0 ± 28.3	191.9 ± 81.1	203.9 ± 98.4	84.7 ± 22.9	1.9 ± 0.5	21.3 ± 9.0
Traffic restriction period (peak hours: morning 7–9 h and evening 17–19 h)										
11/9 to 12/2010	88.54 ± 18.77	1.95 ± 0.46	182.54 ± 90.06	19.95 ± 5.44	26.4 ± 10.3	106.5 ± 30.1	95.4 ± 40.6	54.1 ± 16.1	1.8 ± 0.4	42.9 ± 17.8
11/22 to 25/2011	89.54 ± 10.06	1.91 ± 0.46	195.76 ± 91.15	21.59 ± 2.60	28.0 ± 9.9	95.0 ± 32.2	80.5 ± 33.0	57.4 ± 15.6	1.0 ± 0.5	41.0 ± 14.1
11/20 to 23/2012	62.04 ± 10.70	1.39 ± 0.41	204.44 ± 64.16	23.51 ± 2.12	41.8 ± 12.4	104.6 ± 22.1	82.5 ± 24.7	71.9 ± 13.5	1.2 ± 0.4	52.0 ± 15.4
Sampling times (units)	**Car** (veh/h)	**Motorcycle** (veh/h)	**LVs** (veh/h)	**Urban bus** (veh/h)	**Int. bus** (veh/h)	**2-axle** (veh/h)	**3-axle** (veh/h)	**4 or + axle** (veh/h)	**HVs** (veh/h)	**Total** (veh/h)
No traffic restriction period (peak hours: morning 7–9 h and evening 17–19 h)										
11/27 to 28 and 12/1 to 2/2008	5839 ± 1040	1315 ± 345	8159 ± 916	9 ± 3	119 ± 68	272 ± 41	204 ± 28	348 ± 96	952 ± 167	9108 ± 864
11/11 to 13 and 11/16 to 18/2009	6377 ± 631	1438 ± 383	8946 ± 1072	21 ± 5	96 ± 42	348 ± 75	235 ± 47	327 ± 72	1028 ± 187	9974 ± 1228
Traffic restriction period (peak hours: morning 7–9 h and evening 17–19 h)										
11/9 to 12/2010	8059 ± 1058	1565 ± 387	9964 ± 1423	9 ± 2	149 ± 89	72 ± 11	15 ± 8	3 ± 1	347 ± 86	9155 ± 2103
11/22 to 25/2011	7841 ± 514	1508 ± 405	9817 ± 829	10 ± 3	144 ± 100	173 ± 90	21 ± 12	5 ± 2	353 ± 111	10 170 ± 885
11/20 to 23/2012	7655 ± 870	1678 ± 419	9648 ± 1180	11 ± 2	144 ± 98	59 ± 18	17 ± 12	3 ± 2	232 ± 96	9880 ± 1262

Notes: Monitoring for periods of no traffic restriction and traffic restriction lasted 60 and 72 h, respectively;[1] the sampling times of the field (meteorology and concentrations) and route (traffic) measurements were related to the site located on Bandeirantes Avenue (near route 19G and counting point 3 in Fig. 1);[2] relative humidity;[3] wind speed;[4] wind direction;[5] air temperature;[6] mean values ± standard deviations.[7] Source: CETESB air quality monitoring network, 2008–2012 [http://www.cetesb.sp.gov.br/ar/qualidade-do-ar/32-qualar] and CET traffic monitoring, 2008–2012 [http://www.cetsp.com.br/media/334435/relatorio_dsvp2013b.pdf].

In this paper, we used linear regression simplifications of the analogous quadratic three-panel speed flow relationships used in traffic engineering (Gokhale, 2011; Williams, 2001; TRB, 2001). The HV traffic emissions of PM_{10} (E_{HV}, kg PM_{10} h^{-1} km^{-1}) are a product of the emission factors (EF_{HV}, g PM_{10} km^{-1}) and the total number of vehicles traveling in the corridor per hour and per kilometer (η_{HV}, veh h^{-1} km^{-1}):

$$E_{HV} = EF_{HV} \cdot \eta_{HV} \tag{6}$$

Finally, the HV flow and PM_{10} concentration (Y) elasticity (ε) with respect to the average PM_{10} emissions (X) was calculated by the following equation:

$$\varepsilon_{x,y} = \frac{\partial Y}{\partial X} \cdot \frac{X}{Y} \tag{7}$$

The vehicle-specific EFs of PM_{10} in unit of g km^{-1} $vehicle^{-1}$, estimated using the reversion method of equations (3)–(5), were compared to those different vehicle emission models and previous studies available for the MRSP. The emission studies reviewed in the MRSP include the measurement techniques most frequently used to produce transportation emission inventories (Smit et al., 2013), such as engine dynamometer measurements EDMS (IEMA, 2013; CETESB, 2013a), portable emission measurements PEMS (ISSRC, 2004, 2007) and road tunnel studies (Sánchez-Ccoyllo et al., 2009; Pérez-Martínez et al., 2014). The aim of calculating EFs of PM_{10} using roadway ambient measurements to estimate vehicle emissions (equations (3)–(5)) was to replicate the real-world operation of vehicles using the researched corridor. EDMS data are not the best for vehicle EF modeling as they do not fully reproduce real-world emissions (Franco et al., 2013; Anenber et al., 2017). The presented measurements, which use NO_x as vehicle tracer, complement the estimates in controlled locations such as road tunnels and can be combined with source apportionment methods (Pant and Harrison, 2013). In the EF model of equation (3), we consider a road traffic contribution of 35% (CETESB, 2013a), similar to the source apportionment of 40% considered by Andrade et al. (2012). In our model, the standard deviations (sd) of the EF mean estimates summarize the relative uncertainties in traffic parameter measurements. Finally, the corridor's total PM_{10} road emissions were estimated from the CET traffic monitoring data and compared to the CETESB pollutant concentrations.

3. Results

3.1. Meteorological and traffic effects on air quality

According to Fig. 1, the corridor researched was oriented southeast to northwest and the CETESB monitoring station was positioned on the northeastern side of the avenue, which was the mostly downwind side according to the main wind direction in the MRSP (the wind direction is from the sea, usually in the afternoon). The wind rose at the sampling site also showed that the wind was coming mostly from the south sea during the period of calculation (Fig. 1).

Restricting HVs caused rebound in LV traffic (Fig. 2). Traffic counts taken by CET (2013) during morning (MP, 7:00 to 9:00 a.m.) and evening (EP, 17:00 to 19:00 p.m.) peak hours were 9108 ± 864 in November 2008 (8159 ± 916 LVs h^{-1} and 972 ± 167 HVs h^{-1}). Permanent traffic restriction measures were implemented in the MRSP from 2010 onwards, affecting the circulation of commercial trucks and resulting in a significant change in the distribution of vehicles and a modal shift from HVs to LVs. The hourly traffic counts of HVs taken in November 2010 were 347 ± 86 vehicles h^{-1} in peak hours with a decrease of more than 60% from those in November 2008. However, the counts of LVs increased by 22% to 9964 ± 1423 vehicles h^{-1}, resulting in an increase in emissions from gasoline and ethanol engines along with the reduction in emissions from diesel engines. Vehicle speeds increased, especially during EP hours, due to the modal shift from HVs to LVs, as did MP transport delays. On the contrary, EP travel times decreased slightly.

Table 2 shows that during the period of no restriction of HV traffic in November 2008, the mean values for all the pollutants except O_3 were statistically significantly higher for the downwind than for the other two wind direction categories, based on the overlapping confidence intervals of the standard deviations, indicating that the transport of pollutants directly from the HV traffic of Bandeirantes Avenue affected air quality considerably. During the

Table 2
Mean concentration levels of NO, NO_2, NO_X, CO, PM_{10} and O_3 related to the location of the corridor and under different wind conditions; November 2008 and November 2010, no holiday weekdays, 7:00–8:00–9:00 and 17:00–18:00–19:00 peak-hour records only, no traffic restriction on trucks (year 2008).

Wind direction	NO ($\mu g\ m^{-3}$)	NO_2 ($\mu g\ m^{-3}$)	NO_X (ppb)	CO (ppm)	PM_{10} ($\mu g\ m^{-3}$)	O_3 ($\mu g\ m^{-3}$)	Frequency (%)
No traffic restriction (2008)							
Upwind[1]	105.9±21.7[4]	73.2 ± 8.6	123.9 ± 21.6	1.5 ± 0.2	33.6 ± 3.8	21.2 ± 1.9	14.0 (16)[5]
Downwind[2]	239.8 ± 9.1	94.7 ± 2.8	243.8 ± 7.5	2.0 ± 0.0	55.4 ± 2.2	19.8 ± 4.0	58.8 (67)
Parallel and calm[3]	110.7 ± 14.2	61.7 ± 3.0	121.0 ± 12.3	1.2 ± 0.1	33.4 ± 2.2	29.1 ± 4.0	27.2 (31)
All directions	185.9 ± 12.3	82.7 ± 3.7	193.6 ± 10.8	1.7 ± 0.1	46.3 ± 2.4	23.2 ± 2.7	100 (114)
Traffic restriction (2010)							
Upwind	47.3 ± 3.8	58.0 ± 2.5	69.0 ± 3.8	1.4 ± 0.1	28.6 ± 1.6	44.0 ± 3.1	40.4 (46)
Downwind	104.8 ± 5.4	61.8 ± 2.3	117.5 ± 4.1	1.9 ± 0.0	34.2 ± 2.3	53.4 ± 8.3	47.4 (54)
Parallel and calm	68.4 ± 14.4	47.7 ± 2.1	80.2 ± 11.7	1.3 ± 0.1	30.9 ± 2.9	41.0 ± 3.5	12.3 (14)
All directions	77.1 ± 5.8	58.5 ± 2.4	93.3 ± 4.9	1.6 ± 0.1	31.5 ± 2.1	44.0 ± 4.0	100 (114)
Mean difference 2010–2008 (%)							
Upwind	−55.4	−20.8	−44.3	−11.0	−14.8	107.4	
Downwind	−56.3	−34.8	−51.8	−5.0	−38.3	169.6	
Parallel and calm	−38.2	−22.8	−33.7	11.2	−7.6	40.9	
All directions	−58.5	−29.3	−51.8	−5.6	−32.0	89.6	

Notes: blowing to the avenue;[1] blowing from the avenue;[2] blowing parallel to the avenue and calm wind;[3] mean values of air quality concentrations under different wind conditions ± standard deviations;[4] number of samples.[5] Source: CETESB air quality monitoring network, 2008–2012 [http://www.cetesb.sp.gov.br/ar/qualidade-do-ar/32-qualar].

period of restriction of HV traffic in November 2010, the effect of meteorology on air quality decreased and only the mean values for NO_x (NO and NO_2) were significantly higher in the downwind category, which indicated that HV direct emissions were strongly reduced. PM_{10} concentrations also decreased in the period of HV constraint, but at lower rates than NO_x concentrations, and no significant differences between wind categories were observed. Contrary to the other pollutants, O_3 concentrations strongly increased in the period of limitation of HVs regardless of wind category. Increased O_3 concentrations were related to lower NO emissions from trucks and less O_3 depletion.

In Fig. 3, low concentrations of PM_{10} and NO_X at the CETESB traffic side station from November 2009 to 2012 were related to transport policy measures such as traffic constraint of HVs and the consequent reduction in emissions. Hourly air concentrations of PM_{10} and NO_X during peak hours in the period of HV traffic restriction in November 2012 were 28% (15.8 $\mu g\ m^{-3}$) and 43% (86.9 ppb) lower, respectively, than those in the period of no HV traffic constraint in November 2008. However, pollutant levels decreased at lower rates than the HV traffic reduction of 72% (712 vehicles h^{-1} corridor-km^{-1}), showing that additional factors prevented a greater reduction in the concentration levels (Fig. 4). For instance, meteorological dispersion conditions changed during the HV restriction period (Fig. 3) together with increased traffic of LVs in the corridor (Fig. 2). Fig. 4 also shows mean diurnal variations in different parameters (pollutant concentrations and HVs), and it was found that PM_{10} and NO_X concentrations had been substantially reduced by 46% (27.7 $\mu g\ m^{-3}$) and 57% (126.0 ppb), respectively, during the evening peak hours. During the morning peak hours, it was estimated that concentration levels were reduced only 11% (3.9 $\mu g\ m^{-3}$ PM_{10}) and 30% (47.7 ppb NO_X), lower than the evening peak hours, showing that HVs traffic restrictions improved air quality substantially, specially during evening peak hours (68/599 vs. 76%/825 vehicles h^{-1} corridor-km^{-1}).

3.2. Statistical model

The negative effect of evening peak hour was shown in the regression estimates in Table 3 for all the nitrogen pollutants and CO, using hourly peak concentrations of pollutants as the dependent variables in the GLM in equation (2) (we used both absolute and log-transformed levels of pollutants). Lower concentrations (−13.4 μg PM_{10} m^{-3} and -40.5 ppb NO_X) are shown for stops shorter than 20% of total driving time (related to better traffic) than for stops longer than 20%, and the scale of the difference was statistically significant for all the pollutants – for instance about 30% for PM_{10} (0.61 times the SD) and NO_X (0.17 times the SD) concentrations – showing the importance of the effect of traffic conditions in a congested corridor during peak hours. In the GLM, traffic conditions and congestion reductions tended to be most relevant for decreasing air pollution than transport policies restricting HV traffic and statistically significant differences between pollutants and policies were seen only for O_3: 70% more O_3 production (0.96 times the SD, 26.0 $\mu g\ m^{-3}$). The significant effect of an increase in w_s on pollution reduction was detected for NO_X (21.7 ± 11.3 ppb), NO_2 (9.6 ± 3.3 $\mu g\ m^{-3}$) and CO (386.5 ± 92.9 ppb). The effects of meteorological parameters on the increase in pollution were detected for O_3 (a unit increase in w_s and T increased O_3 production by 8.48 ± 2.51 and 3.54 ± 0.76 $\mu g\ m^{-3}$, respectively) and NO_2 (a unit increase in T increased NO_2 by 2.86 ± 0.98 $\mu g\ m^{-3}$). Borges et al. (2012) used a neural network model to predict O_3 concentration in the MRSP and morning wind field components had a high influence, since this variable can change condition of dispersion of O_3 precursors in the morning. Temperature in the afternoon was also presented highly significant. Freitas et al. (2005) showed that meteorological parameters have a significant influence on O_3 formation. The reduction in the traffic of 4-axle trucks contributed to decreasing PM_{10}, NO_X and CO concentration levels significantly and the mean reduction of about 300 trucks per peak hour achieved during the period of HV traffic restriction (Fig. 2) contributed to reducing concentration levels, all other things being equal, by 8.4 $\mu g\ m^{-3}$ (21%), 20.7 ppb (14%) and 489.3 ppb (32%), respectively. The substitution of 4-axle trucks by cars could have improved air quality, and the mean use of free road capacity of about 1750 cars per peak hour achieved in 2012 could have contributed significantly to reducing NO_X, NO_2 and CO concentrations from the 2008 levels by 15.8 ppb (11%), 12.3 $\mu g\ m^{-3}$ (17%) and 194.3 ppb (13%), respectively (on the contrary, it could have contributed to increasing O_3 production by 5.3 $\mu g\ m^{-3}$).

The parameter estimates in Table 3 for the overall model and individual effects support the claim that driving conditions affected air quality more than transport policies focused on the restriction of HV traffic and that there was no statistically significant difference in pollutant concentration levels between peak hours on working

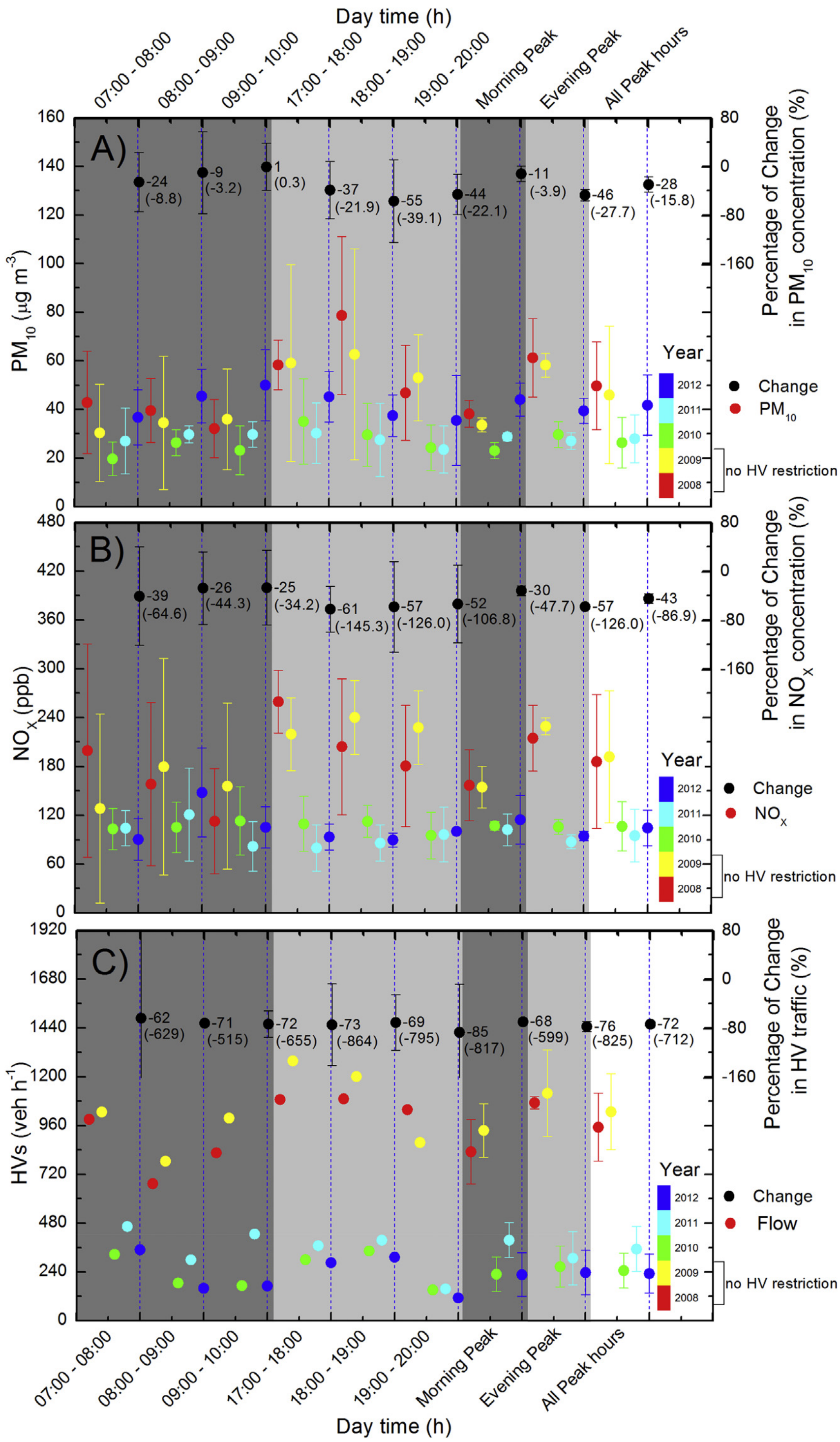

Fig. 4. Effects of applying the transport policy constraints on air quality and HV traffic. a) b) Time variation in PM_{10} ($\mu g\ m^{-3}$) and NO_X concentrations (ppb) and percentage of change in pollutant concentrations during daytime peak hours (h) – morning peak (07:00–10:00, dark grey shaded graph areas) and evening peak (17:00–20:00, light grey shaded graph areas) – on no holiday weekdays (sampling times taken from Table 1). c) HV traffic flows (veh h^{-1}) and change. Notes: Y error bars summarize the relative uncertainties in pollutant concentration measurements and traffic flows (the 95% confidence intervals as a proportion of mean recorded concentrations and flows). Estimated changes in PM_{10}/NO_X concentrations (28/43%) as the transport policy change during daytime (h) peak hours on nonholiday weekdays and as HVs decreased by 72%. Percentage of change (%) with absolute changes in parentheses ($\mu g\ m^{-3}$, ppb and veh h^{-1}). No HV traffic constraint (Nov 2009 and 2010) and traffic constraint (Nov 2010, 2011 and 2012). Data are from CETESB (2008–2012) and CET (2008–2012).

Table 3
Effects of traffic restrictions, meteorology, driving conditions and vehicle composition on air quality concentrations; predicting air quality on no holiday weekdays; 7:00 to 9:00 and 17:00 to 19:00 peak-hour readings only.

Dep. (unit)	PM_{10} (μg/m³)	Ln PM_{10} (μg/m³)	NO_X (ppb)	Ln NO_X (ppb)	NO (μg/m³)	Ln NO (μg/m³)	NO_2 (μg/m³)	Ln NO_2 (μg/m³)	CO (ppb)	Ln CO (ppb)	O_3 (μg/m³)	Ln O_3 (μg/m³)
Mean variables of interest												
tr[1]	−0.050	−0.023	−42.050	−0.073	−53.983	−0.126	−0.571	0.069	109.438	0.158	26.001	0.659
(0)	(12.651)[2]	(0.092)	(42.817)	(0.181)	(45.375)	(0.195)	(12.310)	(0.053)	(541.875)	(0.226)	(9.009)***	(0.144)***
ph[3]	4.102	0.009	36.311	0.190	40.580	0.254	14.397	0.137	333.97	0.209	−0.913	−0.016
(ep)	(5.805)	(0.115)	(18.478)*	(0.044)***	(19.89)**	(0.113)**	(5.662)**	(0.022)***	(154.3)**	(0.159)	(4.137)	(0.108)
Selected meteorology and traffic parameters												
RH	−0.176	−0.006	−0.405	−0.005	−0.112	−0.004	−0.176	−0.002	0.113	−0.001	−0.233	−0.007
(%)	(0.217)	(0.005)	(0.782)	(0.009)	(0.836)	(0.010)	(0.217)	(0.002)	(6.828)	(0.005)	(0.169)	(0.005)
w_s	−2.210	−0.044	−21.703	−0.200	−14.745	−0.169	−9.579	−0.121	−386.530	−0.273	8.477	0.428
(m/s)	(3.399)	(0.061)	(11.259)*	(0.054)***	(12.117)	(0.059)***	(3.305)***	(0.041)***	(92.95)***	(0.085)***	(2.51)***	(0.150)***
T	0.481	0.013	1.866	0.025	0.067	0.010	2.866	0.040	25.135	0.019	3.544	0.069
(°C)	(0.995)	(0.015)	(3.445)	(0.035)	(3.698)	(0.046)	(0.977)***	(0.016)**	(29.155)	(0.028)	(0.76)***	(0.026)***
car	−0.002	−0.000	−0.009	−0.000	−0.010	−0.000	−0.007	−0.000	−0.111	−0.000	0.003	0.000
(veh/h)	(0.002)	(0.000)	(0.007)	(0.000)**	(0.008)	(0.000)**	(0.002)***	(0.000)***	(0.061)*	(0.000)**	(0.002)*	(0.000)
bus	−0.009	−0.006	1.731	0.004	2.294	0.010	0.441	0.004	−17.713	−0.013	0.291	0.003
(veh/h)	(0.500)	(0.003)**	(1.751)	(0.008)	(1.865)	(0.011)	(0.513)	(0.002)**	(16.213)	(0.009)	(0.373)	(0.006)
4 axle	0.028	0.001	0.069	0.001	0.059	0.001	0.026	0.001	1.631	0.001	0.005	−0.001
(veh/h)	(0.038)	(0.000)***	(0.118)	(0.000)**	(0.127)	(0.001)	(0.037)	(0.000)*	(0.976)*	(0.000)***	(0.027)	(0.000)
stops[4]	−13.413	−0.348	−40.532	−0.364	−37.472	−0.405	−12.219	−0.134	−400.879	−0.331	8.078	0.085
(0)	(4.522)***	(0.050)***	(14.17)***	(0.061)***	(15.24)**	(0.111)***	(4.430)***	(0.040)***	(119.199)***	(0.051)***	(3.173)**	(0.075)
Specific regressions												
Model	OLS[5]	ROLS[6]	OLS	ROLS	OLS	ROLS	OLS	ROLS	OLS	ROLS	OLS	ROLS
R^2	0.513	0.577	0.635	0.637	0.667	0.660	0.689	0.770	0.572	0.610	0.824	0.834
dep.	39.264	1.594	143.969	2.158	132.492	2.122	73.371	1.866	1521.212	3.182	37.146	1.570

Notes: transport restrictions on HV traffic, dummy variable (0 HV restriction period);[1] the table report coefficients and standard errors (SE in parentheses) for the Congonhas station regressions; the number of regressors in the GLM was 9;[2] SE are calculated by bootstrapping (200 samples each); the pollutant, meteorology and traffic data, clustering by date and hour; SE of station-level estimates are the standard deviations of coefficients over the 200 replications; an observation is an hour-date pair falling within the specific time of day and type of day (working day and peak hour); in total 132 observations were sampled and the sample periods were 27 November 2008–23 November 2012, including the peak hours (Table 1); evening peak hours, dummy variable (evening peak);[3] traffic conditions, dummy variable (0 stops less than 20% of total driving time);[4] Ordinary Least Squares Estimates (OLS);[5] Robust Ordinary Least Squares Estimates (ROLS).[6] *$p < 0.10$ **$p < 0.05$***$p < 0.01$. Source: CETESB and CET.

days in the month of November for periods of different HV traffic with the exception of the O_3 concentration levels (26.0 μg m^{-3}). Reductions in NO_X and CO and the opposite differences in O_3 suggest that the improvement in air quality (in terms of NO_X and CO) and O_3 production might occur simultaneously with the improvement in traffic conditions (differences are statistically significant in Table 3). Traffic restrictions on trucks implied reductions in NO_X and increases in CO, due to the fact that gasoline LVs may have higher CO emission factors than their diesel HV counterparts (though differences were not statistically significant in Table 3). From the results in Table 2, and performing paired-samples statistical T tests, we can conclude that the average reduction in NO_X of 113.9 ppb during the 2008–2012 period is not due to chance variation and can be attributed to the HV traffic restriction (with a significance *p-value* for change in NOx of less than 0.1). Similarly, the average increment in O_3 of 26.6 μg m^{-3} can be related to lower NO emissions from HVs (*p-value* < 0.05). Increased LV could offset the CO mitigation from HVs and the small reductions in CO (66.7 ppb) may be not as significant as those in NOx: the *p-value* greater than 0.10 for reduction in CO concentration level shows that the HV traffic restriction did not significantly reduce traffic related CO levels (same as PM_{10} levels).

3.3. Emissions and emission factors

In the urban corridor researched, the reductions in HV traffic were related to air quality improvements and pollutant concentration levels were lower during years with lower HV traffic, especially from 3 or more axle trucks. Our results provide strong evidence that traffic restrictions significantly improved air quality during peak hours (Fig. 4). However, the expected reductions in pollutant levels were lower than reductions in HV traffic (28–43 vs. 70%) and additional potential causes could have affected these levels. Two different causes are presented and discussed below.

In the MRSP, road traffic is the main source of NO_X and also contributes to the production of coarse and fine particulate matter (direct tailpipe emissions and local dust of PM_{10} and $PM_{2.5}$, respectively). Of all emissions in the MRSP, those from road traffic account for 82% of the NO_X and 36% of the PM_{10} (CETESB, 2012). A literature review suggests a similar distribution of road transportation emissions in other cities (Sun et al., 2014; Karagulian et al., 2015). This special road contribution of pollutant emissions explains the difference between pollutant reduction rates. During traffic peak hours, vehicles could be an important source of pollutants in the corridor researched. Pérez-Martínez et al. (2015) estimated that in the MRSP vehicle emissions of NO_X were predominantly produced by HVs, accounting for approximately 83% of the total, followed by LVs, collectively accounting for 17%. Older HVs (with average vehicle age>30 years), which represent 20% of the fleet in the MRSP, are important sources of PM emissions, whereas newer LVs (average vehicle age of 10 years) contribute only slightly (less than 5%) to the total vehicle emissions of PM (CETESB, 2013b). Taking these proportions into consideration, we made a rough estimate of the 70% reduction in HVs in terms of pollutant reductions: ≈24% (70*0.36*0.95) and ≈47% (70*0.82*0.83) for PM_{10} and NO_X, respectively. These differences were similar to the most likely differences obtained in the five-year period of HV constraint comparison in Fig. 4.

The other cause of the difference between pollutant concentration levels and traffic reduction rates could be based on traffic and emission parameters (i.e. traffic rebound and the effects of background pollution) and their relationships (Fig. 5; Kiesewetter et al., 2014).

EFs were negatively correlated with HV traffic flow at this urban site until the corridor's capacity was reached (Fig. 5a, the shaded triangle represents the theoretical projection of the traffic

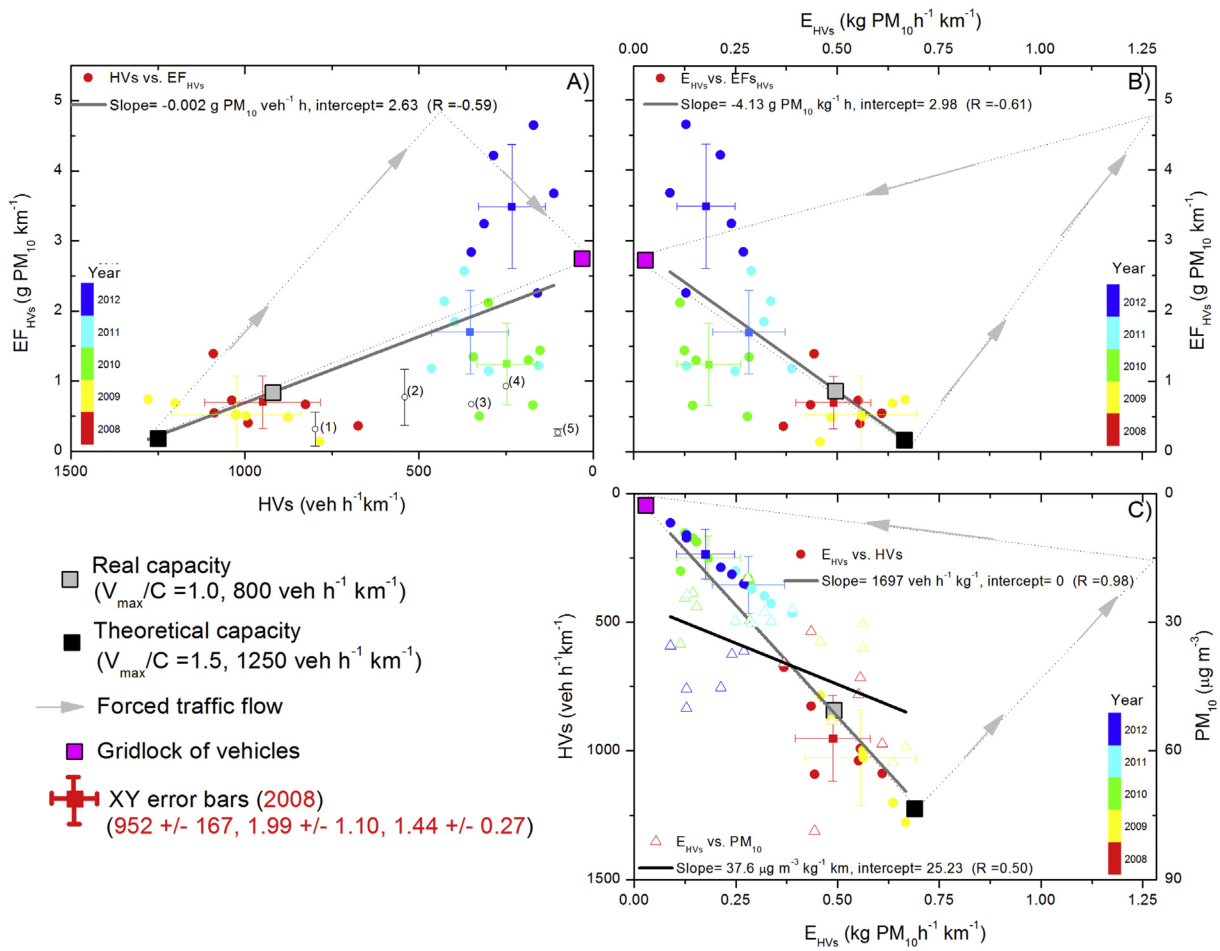

Fig. 5. Relationships of HV traffic (veh h^{-1} km^{-1}) with emissions (E, kg PM_{10} h^{-1} km^{-1}) and emission factors (EF, g PM_{10} km^{-1}). a) EFs and HV traffic. b) EFs and E. c) HV traffic and E and PM_{10} concentrations and E. Notes: the four solid linear fits reported the Pearson correlation (R), the slopes and the intercepts. The arrows represent theoretical traffic congestion conditions for the corridor's maximum volume capacity (Vmax/C, grey and black squares for real and theoretical capacities, respectively), showing forced traffic flow (arrows indicate the time when the corridor's capacity was reached up to the point of the total stop and gridlock of vehicles, magenta squares). XY error bars summarize the relative uncertainties in traffic parameter measurements. Data on emissions were estimates made by the São Paulo Environmental Company CETESB, 2008–2012 (http://www.cetesb.sp.gov.br/ar/qualidade-do-ar/32-qualar), and the São Paulo Traffic Engineering Company CET, 2008–2012 (http://www.cetsp.com.br/media/334435/relatorio_dsvp2013b.pdf). References used in Fig. 5a represent previous results of vehicle-specific EFs: Pérez-Martínez et al. (2014) [1], Sánchez-Ccoyllo et al. (2009) [2], CETESB (2013a) [3,4] and ISSRC (2007) [5].

congestion conditions). Analogously, EFs for HVs were negatively correlated with absolute emissions as a direct consequence of the reduction in HV traffic and substitution of the corridor's free capacity with LVs (Fig. 5b). EFs in 2010–2012 were higher than those in 2008–2009 due to changes in relative shares of HVs: 67% interurban and urban buses in 2012 compared to 14% in 2008 (Fig. 2). Deterioration of driving conditions, due to vehicle fleet substitution of HVs by LVs and reaching the corridor's capacity (Fig. 5), also contributed to raising the EFs especially during the years 2011 and 2012. Diesel transit bus emissions are 1.5–3 times higher than average at major intersections, bus stops and hot spots related to congestion episodes than average (Gouge et al., 2010). The vehicle-specific emission factors in our study were compared with previous results in the MRSP. Using a reversion method with tunnel measurements (similar to that adopted in this paper), Sánchez-Ccoyllo et al. (2009) and Pérez-Martínez et al. (2014) estimated EFs of 755 ± 401 and 277 ± 108 g v-km^{-1} for HV uninterrupted-flow facilities of 543 (14%) and 800 vehicles h^{-1} (30% of total traffic) in 2004 and 2011 respectively. During these experiments there were no evident congestion episodes. Similarly, CETESB (2013a) estimated an EF of 0.653 g v-km^{-1} for a 10-year fleet representing the HVs using the corridor in 2008: 1% urban buses, 13% interurban buses, 29% 2-axle trucks, 21% 3-axle trucks and 37% of 4- and more axle trucks (Fig. 2). An EF of 0.912 g v-km^{-1} was simulated again by means of laboratory dynamometer studies for the certificated 10-year fleet in 2012: 5% urban buses, 62% interurban buses, 25% 2-axle trucks, 7% 3-axle trucks in 2012 and 1% 4- and more axle trucks. During the simulations, higher EFs were detected in 2012 than in 2008 due to the weight of the interurban buses. A recent study published by Anenber et al. (2017) reported that around one-third of on-road HV diesel emissions in 11 major vehicle markets (including Brazil) are in excess of the certification limits for new vehicles, revealing that dynamometer data are not always accurate for EF modeling (as they often do not represent the real-world emissions of vehicles). Analogously, Gouge et al. (2010) identified a 19–30% increase in bus transit emissions from underestimates from macro-scale models. A study of the emissions from diesel vehicles operating in São Paulo in uninterrupted flows, using portable emission measurements for new HVs in 2006, estimated an EF of 300 ± 50 g v-km^{-1} (ISSRC, 2007).

Finally, Fig. 5c contains the relationship between HV traffic, PM_{10} concentration and emission. Total HV emissions of PM_{10} on Bandeirantes Avenue decreased with decreasing traffic flow, but at a higher rate than that of related pollutant concentrations (represented by the slopes of 1697 veh h^{-1} kg^{-1} and 37.6 μg m^{-3} kg^{-1} km h, respectively). Note that the ratio of concentrations and emissions represents the inverse of the dilution rate characterized in equation (5) (7.38 m^2 s^{-1}). Estimating the elasticity of the two regression lines at the crossing point in Fig. 5c (0.393 kg PM_{10} h^{-1} km^{-1}), it was shown that the elasticity of HV flow with respect to PM_{10} emissions was 2.7 times higher than the elasticity of PM_{10} concentrations. The average elasticities of 667 veh h^{-1} and 40.05 μg m^{-3} were equal to 1.00 and 0.37, indicating that for a 1% reduction in HV traffic and PM_{10} concentration, the emissions could be decreased by 1 and 0.37%, respectively.

According to equation (6) and Fig. 5c, it was expected that the worst air quality conditions would occur after traffic exceeded the corridor's capacity and before emissions were reduced as a consequence of a total gridlock (all vehicles stopped at the same time). This theoretical situation never occurred as a direct outcome of the periods of HV traffic constraint (2010–2012). These results confirmed the GLM in Table 3, which examined the relationship between PM_{10} concentrations, traffic flow and driving conditions (columns 1 and 2 with log transformation and absolute concentration values, respectively). PM_{10} concentrations marginally decreased with decreases in 4 or more axle trucks and increases in car flow by 0.028 and 0.002 μg m^{-3}, respectively, and increased by 13.41 μg m^{-3}, with deterioration in driving time (stopping times longer than 20% of total driving time) as the conditioning dummy variable. According to the GLM results, the effects of decreasing HV flows were 14 times bigger than the effects of increasing LV flows (51% less vs. 4% more PM_{10} concentrations). This value was similar to the difference obtained by Pérez-Martínez et al. (2014) by comparing PM emission factors of HVs and LVs in tunnels in the MRSP (277 ± 108 vs. 20 ± 8 mg $PM_{2.5}$ km^{-1}). Apparently, substitution of HVs by LVs (712 HVs were substituted by more than 2300 LVs) was partially offset by the negative effect of the increase in travel time and related traffic conditions: the corridor's capacity was approached by the increase in LV traffic flows by more than 1000 equivalent vehicles per hour (Fig. 2). The results of this paper suggest that the restriction on HV traffic improved air quality significantly in this corridor (around 28%) while worsening traffic conditions: a 51% decrease in PM_{10} due to a reduction in HV traffic, 10% more PM_{10} emissions related to worse traffic conditions and an increase in LV flows and 13% more PM_{10} concentrations due to other causes such as an increase in negative weather conditions and more emissions from nonvehicular sources (Table 3).

The traffic restriction of HVs from 2010 onwards was followed by other environmental and transport-policy changes, such as the regulation of circulation of irregular interurban buses and the establishment of more restrictive emission factors (NO_X and PM_{10}) in new HVs, according to phase P7 of the Brazilian PROCONVE (Pérez-Martínez et al., 2015). In the MRSP, diesel-powered HVs meeting the standards for PROCONVE phases P5–P7 (and representing 50% of the total fleet in 2013) accounted for 45% and 24% of the total emissions of NOx and PM_{10} (CETESB, 2013b), respectively (newer vehicles used after-treatment technologies incorporated from 2007 onwards). As these changes occurred simultaneously with implementation of the limitation on HV traffic in city corridors, it was challenging to estimate an accurate empirical model of air quality in advance.

4. Conclusions and recommendations

Based on the results of this paper, it can be acknowledged that the policy measures to restrict HV traffic employed by the Municipality of São Paulo in the Bandeirantes urban corridor to reduce pollutant concentration levels were successful in improving air quality in the period 2008–2012. Air quality improved by 11–46% (3.9–27.7 μg m^{-3}) and 30–57% (47.7–126.0 ppb) in terms of PM_{10} and NO_X concentrations, respectively, while HV traffic emissions were reduced by 68–76% (599–825 vehicles h^{-1}). These traffic peak-hour estimates and near-road air quality trends for the November months (15.8 μg m^{-3}/0.26 μg m^{-3} $month^{-1}$ and 86.9 ppb/1.45 ppb $month^{-1}$) were compared with observations and annual trends from 2008 to 2012 to check for potential consistency: 3.6 μg PM_{10} m^{-3} (0.06 μg m^{-3} $month^{-1}$) and 117.6 ppb NO_X (1.96 ppb $month^{-1}$). Air quality deteriorated by 28.1 μg m^{-3} (0.46 μg m^{-3} $month^{-1}$) and improved by 0.50 ppm (0.008 ppm $month^{-1}$), in terms of O_3 and CO concentrations, compare to 22.2 μg O_3 m^{-3} (0.37 μg m^{-3} $month^{-1}$) and 0.23 ppm CO (0.004 ppm $month^{-1}$), respectively. NO_X and O_3 peak-hour concentrations and trends were especially consistent with annual trends from 2008 to 2012, confirming the reduction of NO truck emissions and subsequent O_3 formation. Using air quality data from 2000 to 2013 for the whole MRSP, Pérez-Martínez et al. (2015) found rates of −0.09 μg m^{-3} $month^{-1}$ (PM_{10}), −0.25 ppb $month^{-1}$ (NO_X), 0.013 ppb $month^{-1}$ (O_3) and −0.05 ppm $month^{-1}$ (CO). Significant positive impacts of decreases in the volume of 4-axle trucks on PM_{10}, NO_X and CO were also detected in our general linear statistical model (GLM). Evening peak hours showed higher significant pollutant nitrogen and CO concentration reductions than morning peak hours. The findings of Degraeuwe et al. (2016) are quite in line with the observations of the Bandeirantes corridor, suggesting that reducing the NOx emissions of diesel cars (equivalent to banning HV vehicles) can decrease urban NO_2 pollution.

Contrary to the reduction certified by the HV emission control measures, the GLM model showed a potential increase in PM_{10}, NO_X, NO concentrations of 20–30% (10–20% for NO_2 and CO), attributed to the deterioration in driving conditions from travel times with more than to travel times with less than 20% of total time stopped (rebound effect in the form of increase traffic of LVs). This deterioration partially explains the difference between HV traffic and pollutant reduction rates. Urban background concentrations may also explain part of this difference. The statistical model shows a potential increase in O_3 concentrations of 40–70%, attributed to the restriction on HV traffic in going from the period of no HV constraint to the period of HV constraint, partially offset by a potential reduction of 0–20% caused by the deterioration in driving conditions. It is evident that the space previously used by HV (trucks) has been used after by other LVs (cars, vans and motorcycles). Traffic of urban buses hardly changed during the researched period but not the traffic of interurban buses (Fig. 2). From our study, it is not possible to conclude that the banned heavy trucks were substituted by light vans as the cargo of the HVs could have been transferred to LVs.

In the GLM model, stronger relationships with pollutant concentrations were found for traffic parameters (stops and HV traffic) than for climate (wind speed and temperature), especially with the main vehicle pollutants (PM_{10}, NO_X and CO). The favorable impact of an increase in wind speed on NO_X, NO_2 and CO concentrations was observed (marginal reductions of 21.7 ± 11.3 ppb, 9.6 ± 3.3 μg m^{-3} and 386.5 ± 92.9 ppb, respectively, per 1 m s^{-1} increase in wind speed). On the contrary, a significant negative impact of wind speed and air temperature on O_3 was detected (8.5 ± 2.5 and 3.5 ± 0.8 μg m^{-3}). The statistical significance of wind direction was also observed according to the mean analysis shown in Table 2. The downwind orientation of the sampling site defined the predominant direction from which air pollutant concentrations were highest, with the exception of O_3. The relationship between wind direction and pollutant concentrations was maintained in the period of HV traffic constraint and indicates the main source of contributions of the corridor's vehicle emissions to site concentration levels (no statistically significant differences between reductions in concentration by wind direction category were detected). NO_X concentration levels were more dependent on wind direction than the other pollutants.

The GLM represented in equation (2), although explaining the relationship between traffic, meteorological and air quality data, has limited capability for explaining chemistry reaction dynamics. In this case, other complementary models and approaches are needed to reflect the dynamics between pollutants (Borges et al., 2012; Freitas et al., 2005; Madronich, 2014). Consequently, the modeling estimates of the pollutants NO, NO_2 and O_3 are not as accurately determined as the estimates of pollutants such as PM, CO and NO_X. In this study, the results of the statistical model reflect basically pollutant changes due to truck restrictions and temporal modification of traffic conditions. The atmospheric oxidation of NO, and subsequent O_3 depletion ($NO + O_3 \rightarrow NO_2 + O_3$), were decreased in the traffic environment of the researched corridor due to lower concentrations of fresh NO (as a consequence of the truck ban). Similarly, NO_2 can be reduced to NO by solar energy ($NO_2 + h\upsilon \rightarrow NO + O$ chemistry).

Although this study was carried out in only one point, it serves as a model for evaluating one of the public policies implemented in large cities. This paper shows a case study that could be used as a basis for new mitigation techniques. More measurement points at other locations in the city can accurately show the efficiency of making traffic changes in a region of the city. Although concluding causality from a unique before-after comparison is very limited, current trends in the MRSP outside the presented model confirmed our results (Pérez-Martínez et al., 2015). To link traffic restriction to HV volume reduction and then to the reduction in the roadside concentrations of certain air pollutants, we need some kind of control of the trend without the traffic restriction, such as one or multiple control sites where traffic patterns are similar to the corridor in question but the restriction is not implemented. Without such a control, we can only claim our work as an exploratory analysis describing what happened and a suggestive causal claim can be safely drawn.

For future research, we think it is important to collect more evidence of the impact of traffic and climatic variations on air quality over a bigger city area to study air quality accurately, especially since the permitted daily mean pollutant concentration levels, established by Brazilian Directives governing environmental air quality, are becoming more restrictive. It is also important to combine this evidence to a high-resolution vehicle traffic and emission model. The importance of the relationships analyzed in this air quality research, with pollutant concentrations together with levels of explanatory variables measured during five consecutive years (2008–2012), serves to underline the need to implement additional urban transport policy measures aimed at decreasing pollutant concentrations in São Paulo (especially in congested urban corridors on working days). The interactions of street-level concentrations and urban backgrounds are also important and it is clear that under high background concentrations, street-level measures alone may not be sufficient to address all air quality problems. This highlights the need of complementary measures and policy options to street-level performances to improve air quality: modal shift, public transport development and introduction of novel transport and energy technologies. Given the overall background pollution, the banning of trucks in congested urban corridors must be followed by the control of vehicle emission standards and the correct transport modal shift to environmentally efficient vehicles. There are potentials for improving air quality in urban agglomerations due to shifting part of the LV traffic to either public transport or electric vehicles, including the replacement of part of the LV fleet by electric two-wheelers (Weiss et al., 2015).

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