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Neighborhood-scale air quality impacts of emissions from motor vehicles and aircraft

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HIGHLIGHTS

- Large inter-community variations in traffic-related pollutant levels were observed.
- Intra-community variations in pollutants were also observed.
- Disproportionate contributions of high-emitting vehicles to UFP levels were examined.
- UFP emissions appeared to have decreased over the past decade.
- On the closure day, particulate pollution was conspicuously reduced area-wide.

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ABSTRACT

A mobile monitoring platform (MMP) was used to measure real-time air pollutant concentrations in different built environments of Boyle Heights (BH, a lower-income community enclosed by several freeways); Downtown Los Angeles (DTLA, adjacent to BH with taller buildings and surrounded by several freeways); and West Los Angeles (WLA, an affluent community traversed by two freeways) in summer afternoons of 2008 and 2011 (only for WLA). Significant inter-community and less significant but observable intra-community differences in traffic-related pollutant concentrations were observed both in the residential neighborhoods studied and on their arterial roadways between BH, DTLA, and WLA, particularly for ultrafine particles (UFP). HEV, defined as vehicles creating plumes with concentrations more than three standard deviations from the adjusted local baseline, were encountered during 6–13% of sampling time, during which they accounted for 17–55% of total UFP concentrations both on arterial roadways and in residential neighborhoods. If instead a single threshold value is used to define HEVs in all areas, HEV's were calculated to make larger contributions to UFP concentrations in BH than other communities by factors of 2–10 or more. Santa Monica Airport located in WLA appears to be a significant source for elevated UFP concentrations in nearby residential neighborhoods 80–400 m downwind. In the WLA area, we also showed, on a neighborhood scale, striking and immediate reductions in particulate pollution (~70% reductions in both UFP and, somewhat surprisingly, $PM_{2.5}$), corresponding to dramatic decreases in traffic densities during an I-405 closure event ("Carmageddon") compared to non-closure Saturday levels. Although pollution reduction due to decreased traffic is not unexpected, this dramatic improvement in particulate pollution provides clear evidence air quality can be improved through strategies such as heavy-duty-diesel vehicle retrofits, earlier retirement of HEV, and transition to electric vehicles and alternative fuels, with corresponding benefits for public health.

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1. Introduction

Traffic-related pollutants cause significant adverse health impacts, including increased mortality, adverse birth outcomes, and respiratory and cardiovascular diseases (Beelen et al., 2008; Gehring et al., 2010; Wellenius et al., 2012; Wilhelm and Ritz, 2003). Of many pollutants from vehicular exhaust, ultrafine particles (UFP) have recently been the subject of increased focus because a growing number of toxicological and epidemiological studies suggest a causal relationship between UFP and adverse health effects (Hoek et al., 2010; Nel et al., 2006).

Vehicular emissions are known to be a dominant source of UFP in urban areas, commonly accounting for ~80% of total number concentrations (Kumar et al., 2010). Although UFP number concentrations tend to rapidly decline within 100–500 m from major roadways during daytime (Karner et al., 2010), dense networks of roadways in cities increase neighborhood UFP concentrations along with other pollutants (Hu et al., 2012). Under stable atmospheric conditions such as nocturnal inversions, traffic-related pollutants tend to be more elevated and have much wider impacts, up to 2 km, downwind of roadways (Choi et al., 2012; Hu et al., 2009b).

Air pollutants such as UFP, oxides of nitrogen (NO_x), and carbon monoxide (CO) are emitted mostly from mobile sources in urban areas (CARB, 2009). Mobile source direct emissions of carbonaceous aerosols combined with the smallest sized fraction of road dust, as well as secondary formation from mobile source NO_x, and VOCs, contribute roughly half of particulate mass less than 2.5 μm ($PM_{2.5}$) in the Los Angeles area (Pham et al., 2008). As a result, effective implementation of traffic interventions, stringent emission regulations, and/or improvements in engine efficiency and fuel composition can help mitigate air pollutant levels. Wahlin (2009) and Wang et al. (2011) reported significant decreases in nucleation mode particle concentrations after fuel regulations for lower sulfur content were adopted. Friedman et al. (2001) found 1-h peak ozone concentrations were 13% lower due to decreased traffic counts during the 1996 Summer Olympic Games in Atlanta, Georgia, accompanied by 16%, 18%, and 7% reductions in PM_{10}, CO, and NO_2 concentrations, respectively, although only weekday morning peak traffic flows near the downtown were noticeably decreased. Several studies reported significantly decreased air pollutant concentrations, including CO (−33%), NO_x (−42%), sulfur dioxide (SO_2) (−60%), black carbon (BC) (−26 to −74%), and surface area $PM_{1.0}$ (−37%) during the 2008 Summer Olympics in Beijing urban areas due to stringent traffic interventions and emission controls on industrial sources (Wang et al., 2009a, 2009b).

To date, only a handful of studies have investigated improvements in UFP-related air quality due to temporary suspension of traffic, based on curbside measurements at closed roadways. Whitlow et al. (2011) observed 58% lower UFP concentrations during the "Summer Streets" campaign in New York City, in which vehicular traffic was not allowed on Park Ave. in the morning of three consecutive Saturdays. Quiros et al. (2013) reported 83% and 60% decreases in UFP and $PM_{2.5}$ concentrations, respectively, at 50 m downwind of the I-405 freeway in West Los Angeles, California, during the closure of this freeway due to demolition of an overpass bridge in 2011, which lasted from July 15 at 20:00 to July 17 at noon. During this closure, area-wide traffic reductions occurred in response to long-term intense warnings of potential chaotic congestion ("Carmageddon"). This closure event provided a rare opportunity to evaluate the effects of substantially reduced traffic emissions on area-wide air quality improvement.

Recent studies showed aircraft emit significant UFP according to aircraft weights, fuel consumption rates, engine types, and operating cycles (Hu et al., 2009a; Mazaheri et al., 2009; Zhu et al., 2011) and hence airports can increase UFP levels in nearby neighborhoods (e.g., Westerdahl et al., 2008). Hu et al. (2009a) reported aircraft UFP plumes reach beyond 660 m downwind of Santa Monica Airport in WLA with UFP concentrations elevated by factors of 10 and 2.5 at 100 and 600 m downwind, respectively, based on 4 days of measurements. The WLA sampling route in the present study includes residential neighborhoods in the vicinity (80–400 m downwind) of Santa Monica Airport, and hence this study can evaluate Hu et al. (2009a) findings.

Consequently, this comprehensive study attempts to examine inter- and intra-community variations in traffic-related air pollutants both in residential neighborhoods and on arterial roadways; Santa Monica Airport impacts on UFP levels in nearby neighborhoods; variations in pollutants levels over a period of years in these same neighborhoods; as well as the area-wide effects of traffic emission reductions on air quality improvement.

2. Experimental methods

2.1. Study locations and routes

Measurements of traffic-related air pollutants were conducted using a mobile monitoring platform (MMP) in West Los Angeles (WLA), Downtown Los Angeles (DTLA) and Boyle Heights (BH) in Southern California (Fig. 1a). The BH measurements were reported in Hu et al. (2012) and are included for comparison here. Demographic and housing data for these communities are provided in the supplementary information (SI); briefly WLA has the by far the highest income, educational level and housing price; BH and DTLA are similar to each other, with BH having slightly higher median income, fraction of foreign born and much higher population density but lower median housing price and education level compared to DTLA.

Each route driven consisted of various urban environments: small quiet streets enclosed by dense residential neighborhoods in which light traffic volumes were encountered by the MMP during sampling periods; several major arterial roadways; and residential neighborhoods near Santa Monica Airport (SMA) in WLA. Each of these neighborhoods also contains light commercial activity, such as restaurants, small shops, and dry cleaners etc. Hu et al. (2009a,b) clearly established that the elevated concentrations downwind of Santa Monica airport above the levels in nearby neighborhoods arise from activities at SMA. In the West LA areas we have not been able to discern any appreciable point (non-vehicular) sources of ultrafine particles. While we did not observe any obvious large point sources in BH or DTLA (either in the data or as human observers), there may be some contribution from smaller sources, such as charbroiling activities.

All sampling areas are, in general, influenced by consistent onshore sea-breezes (south-easterlies) during the day (10:00–18:00). Thus, air masses arriving at the WLA route from the ocean are relatively unpolluted (Westerdahl et al., 2008), whereas air masses at the DTLA route have incorporated more pollution during transport across the city. WLA is traversed by two major freeways (I-10 and I-405), and DTLA and BH are enclosed by numerous freeways (e.g., I-10, 101, I-110, 60, and I-5). Because the study areas are traversed and surrounded by many arterial roads and freeways, pollutant emissions depend strongly on nearby traffic volumes. More details about geographical and socioeconomic characteristics of sampling sites as well as pollutant sources are described in Supplementary information (SI) S1.

2.2. Sampling and instrumentation

Traffic-related air pollutants, including UFP number concentrations, $PM_{2.5}$, particle-bound polycyclic aromatic hydrocarbon (PB-

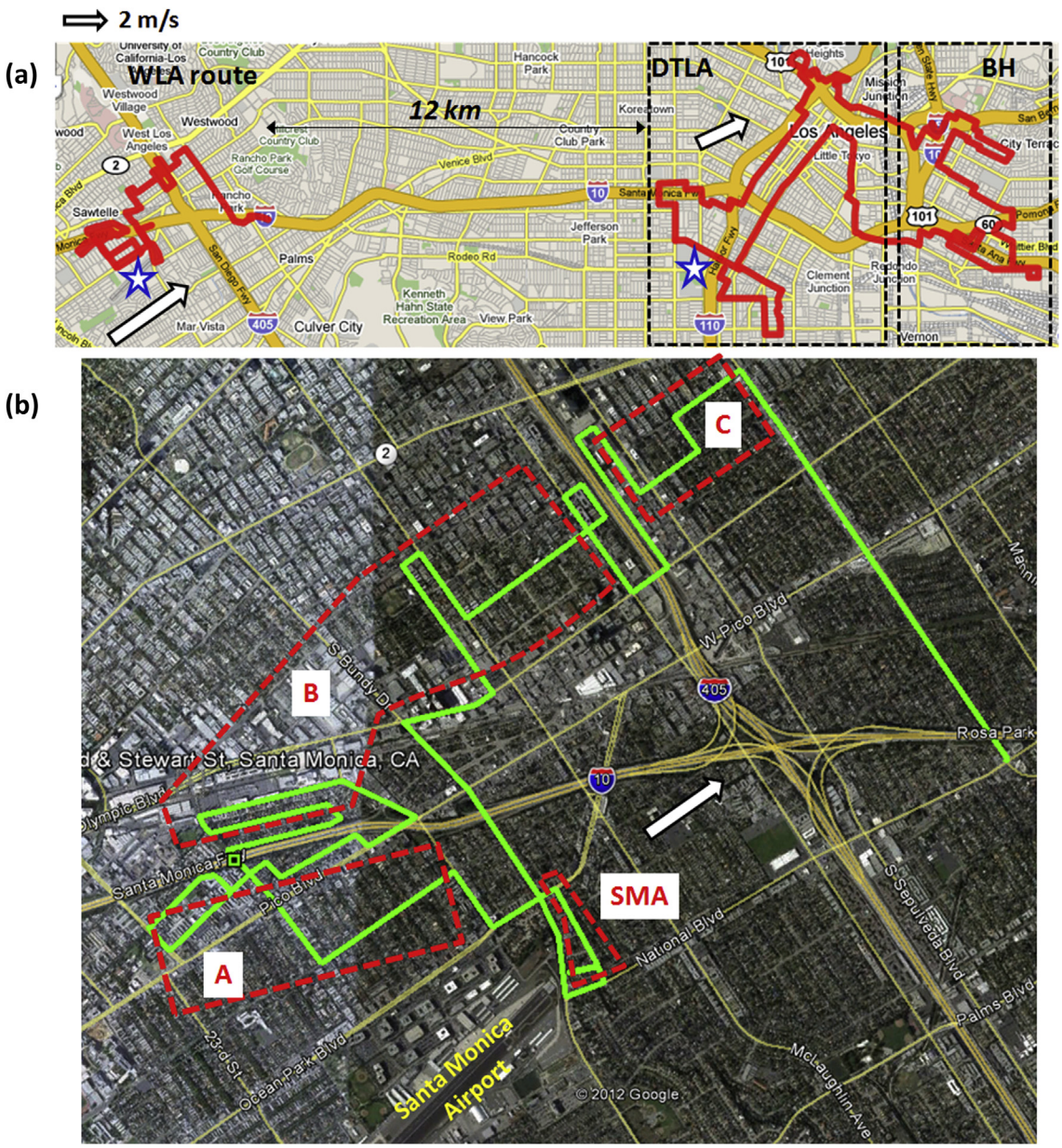

Fig. 1. (a) Map of West LA (WLA, red line in west of the map), Downtown LA (DTLA) and Boyle Heights (BH) routes (red line in east of the map; DTLA and BH are divided by black dotted squares). DTLA route is located 12 km east of WLA route. White arrows represent mean wind speeds and direction during the sampling periods, and blue stars are weather stations where meteorological data were obtained. (b) Closer map of WLA route (green line). Red dotted lines confine neighborhood sub-areas (*A*, *B*, *C*, and SMA: Santa Monica Airport). White arrow represents prevailing winds. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.) Map sources: (a) Google Map and (b) Google Earth.

PAH), NO, and CO, were measured using the mobile monitoring platform, a Toyota RAV4 electric sub-SUV free from self pollution (Table 1). The utility and applications of MMPs equipped with fast response instruments have been described extensively elsewhere (Choi et al., 2012; Kozawa et al., 2012; Westerdahl et al., 2005) and in SI S2. Sampling was conducted in the summer of 2008 (DTLA/BH and WLA) and in 2011 (WLA), twice per day in the afternoon in WLA and once per day in the afternoon in DTLA/BH (Table 2; SI S3). In 2008, measurements were conducted on three weekdays and one Saturday in WLA and three weekdays in DTLA. Boyle Heights (BH) data in 2008 were obtained from Hu et al. (2012) whose measurements were conducted during the same periods as DTLA measurements. In 2011, sampling was conducted for three consecutive weeks (pre-, post-, and during the I-405 closure) on three contiguous days (Friday–Sunday) from 8 to 23 July in WLA. Of

Table 1
Instruments on the mobile monitoring platform operational during the measurements.

Instrument	Measurement parameter	Response time[a] (inlet to record)
TSI Portable CPC, Model 3007	UFP count (10 nm–1 μm)	4 s
TSI FMPS, Model 3091	UFP size (5.6–560 nm)	9 s
TSI DustTrak, Model 8520	$PM_{2.5}$ mass	5 s
EcoChem PAS 2000	Particle-bound PAH	10 s
Teledyne API Model 300E	CO	21 s
Teledyne-API Model 200E	NO	22 s
Vaisala Sonic Anemometer and Temperature/RH sensor	Surface winds, temperature, and relative humidity (RH)	–
Garmin GPSMAP 76CS	Distance and relative speed	–

[a] Response time is an averaged value for smoke test results.

Table 2
Measurement dates, mean surface meteorological conditions, and the CART classification results for meteorological comparability.

Area	Measurement date (time)	Day of week	Temp. (°C)	Relative humidity (%)	Wind speeds (m s^{-1})	Wind direction (°)	CART final node[a]
DTLA/BH	07/14/2008 (14:00–17:00)	Mon.	27.6	41	2.6	240	2
	07/16/2008 (14:00–17:00)	Wed.	26.7	49	2.4	260	2
	07/18/2008 (14:00–17:00)	Fri.	24.6	61	2.9	250	2
	Mean (std.)		26.3 (1.5)	50 (9)	2.6 (0.7)	250 (10)	
WLA	06/30/2008 (14:00–16:30)	Mon.	21.9	60	4.1	243	2
	07/08/2008 (14:00–16:30)	Tue.	20.7	73	5.1	240	5
	07/10/2008 (14:00–16:30)	Thu.	23.4	63	4.4	227	2
	07/12/2008 (14:00–16:30)	Sat.	23.9	63	4.3	240	2
	Mean (std.)		22.5 (1.5)	65 (5)	4.5 (0.6)	238 (13)	
WLA	07/08/2011 (12:00–14:00)	Fri.	22.6	70	3.9	233	2
	07/09/2011 (12:00–13:30)	Sat.	21.5	72	3.8	228	2
	07/10/2011 (12:00–13:30)	Sun	21.8	68	4.1	235	2
	07/15/2011 (13:30–15:00)	Fri.	21.3	57	4.6	245	2
	07/16/2011[b](14:30–16:00)	***Sat.***	***20.3***	***67***	***5.1***	***240***	***1***
	07/17/2011 (13:15–14:45)	***Sun***	***20.9***	***68***	***4.3***	***240***	***2***
	07/22/2011 (14:20–16:00)	Fri.	20.9	66	4.8	233	2
	07/23/2011 (13:30–15:00)	Sat.	21.1	66	4.4	240	2
	Mean (std.)		21.3 (0.7)	67 (4)	4.4 (0.4)	237 (6)	

[a] CART classifications were made based on daily maximum CO data obtained at N. Main monitoring station operated by South Coast Air Quality Management District as described in detail in Choi et al. (2013).
[b] Bold–Italic indicates the I-405 Freeway closure period.

the measurement periods in 2011, the I-405 freeway was closed for construction for the entire day of July 16 and until 12:00 of July 17. While the sampling described here was not adjacent to the closure area, traffic in the general area was dramatically reduced during the closure period.

2.3. Traffic and meteorological data

Freeway traffic data were obtained from the Freeway Performance Measurement System (PeMS) operated by the Institute of Transportation at University of California, Berkeley. Data were collected from sensors located at the Pico station (VDS ID: 717794-5, 34.038°N/−118.439°W) for the I-405 freeway and the Cloverfield station (VDS ID: 737246, 34.025°N/−118.467°W) for I-10 freeway in WLA. For DTLA and BH sites, traffic data were collected from three sensors near the sampling route (VDS ID: 718335, 34.037°N/−118.289°W for the I-10 freeway; VDS ID: 764032, 34.026°N/−118.275°W for the I-110; VDS ID: 764853, 34.065°N/−118.251°W for 101 freeway). Unfortunately, however, no traffic data on surface roadways are available.

Meteorological data were obtained from a weather station located at Santa Monica Airport (<1 km from the route) in WLA and at the University of Southern California (<2 km from the route) for DTLA and BH routes (Fig. 1a). Data from both stations were collected through the MesoWest website operated by the Department of Atmospheric Sciences at University of Utah. To determine regional meteorological comparability among measurement days in 2011 and 2008, a classification and regression trees (CART) method for primary pollutants, developed to evaluate meteorological comparability in air quality studies in California's South Coast Air Basin (Choi et al., 2013), was applied. Our CART method yields statistically exclusive groups (*nodes*) of a target variable based on a number of meteorological variables such as pressure, temperature, wind speeds, relative humidity, and pressure gradients both in the upper air and at the surface. Thus, individual final nodes created by the CART model are associated with specific meteorological conditions for a specific level of traffic-related primary pollutants. More details about the CART analysis and regression trees developed for the SoCAB study areas are found elsewhere (Choi et al., 2013) and briefly described in SI S3.

3. Results and discussions

3.1. Meteorological comparability

The mean air temperature, relative humidity (RH), wind speeds and direction during measurement periods are shown in Table 2. In general, higher air temperature, lower RH, and lower wind speeds were observed in DTLA than in WLA. Prevailing winds were consistently from the southwest strongly influenced by sea-breezes in both DTLA and WLA (Fig. 1).

The CART analysis allowed us to investigate day-by-day meteorological comparability on a more regional scale. Summer season regression trees for daily maximum CO concentrations ($[CO]_{max}$) classify five specific meteorological conditions (*nodes*) to explain observed $[CO]_{max}$ (Choi et al., 2013). Of the total 15 measurement days treated here, 13 days were classified to be under meteorologically comparable conditions for primary pollutant dispersion (*node 2; most typical summer conditions*). Only two days (7/8/2008 and 7/16/2011 WLA) fell into meteorologically different nodes (*node 1* and *5*, respectively) (Table 2). In summary, our CART analysis provides support for comparing vehicle-related pollutant concentrations between the 2008 and 2011 sampling days as well as between the WLA and DTLA locations. For additional details see Choi et al. (2013) and SI S3.

3.2. Traffic on the freeways in WLA and DTLA

Average traffic flows (vehicle 5 min^{-1}) for the measurement periods are shown in Table 3 and Fig. 2. Although traffic flows (a sum of both directions) were consistent or slowly decreased beginning in early afternoon (Fig. 2a), we note that for many of our measurement periods traffic speeds decreased after 2 or 3 P.M. (particularly for the north-bound lanes on I-405) indicating traffic jams (Fig. 2b). Traffic jams reduce the number of vehicles passing by the sensor in a given time due to slower speeds, but the numbers of vehicles on a given length of road can be much larger. Thus vehicle density (vehicle km^{-1}), defined as traffic flow divided by vehicle speed, is more representative of traffic conditions, particularly for morning and late afternoon rush hours (Fig. 2c).

Table 3
Mean traffic flows (veh 5 min^{-1}) and densities (veh km^{-1}) on the freeways intersecting with the DTLA and WLA routes during sampling periods. Percent values are relative increase or decrease rates with respect to WLA 2008 values.

Freeway		WLA 2008		WLA 2011			DTLA 2008
		Weekdays	Sat.	Fri.	Sat.	Closure day (Sat.)	Weekdays
Traffic flow (Truck flow) (veh 5 min^{-1})	I-405	1231 (41)	1252 (16)	1214 (33) −1%	1454 (54) +16%	106 (8) −92%	
	I-10	848 (10)	735 (3)	827 (33) −2%	814 (22) +11%	502 (2) −32%	1171 (30) +38%
	I-110						1293 (58)
	101						927 (95)
Traffic density (std. dev.) (veh km^{-1})	I-405	319 (±44)	247 (±20)	269 (±26) −16%	268 (±24) +9%	10 (±2) −96%	
	I-10	123 (±10)	73 (±10)	171 (±21) +39%	148 (±21) +103%	52 (±3) −29%	195 (±29)
	I-110						293 (±72)
	101						156 (±18)

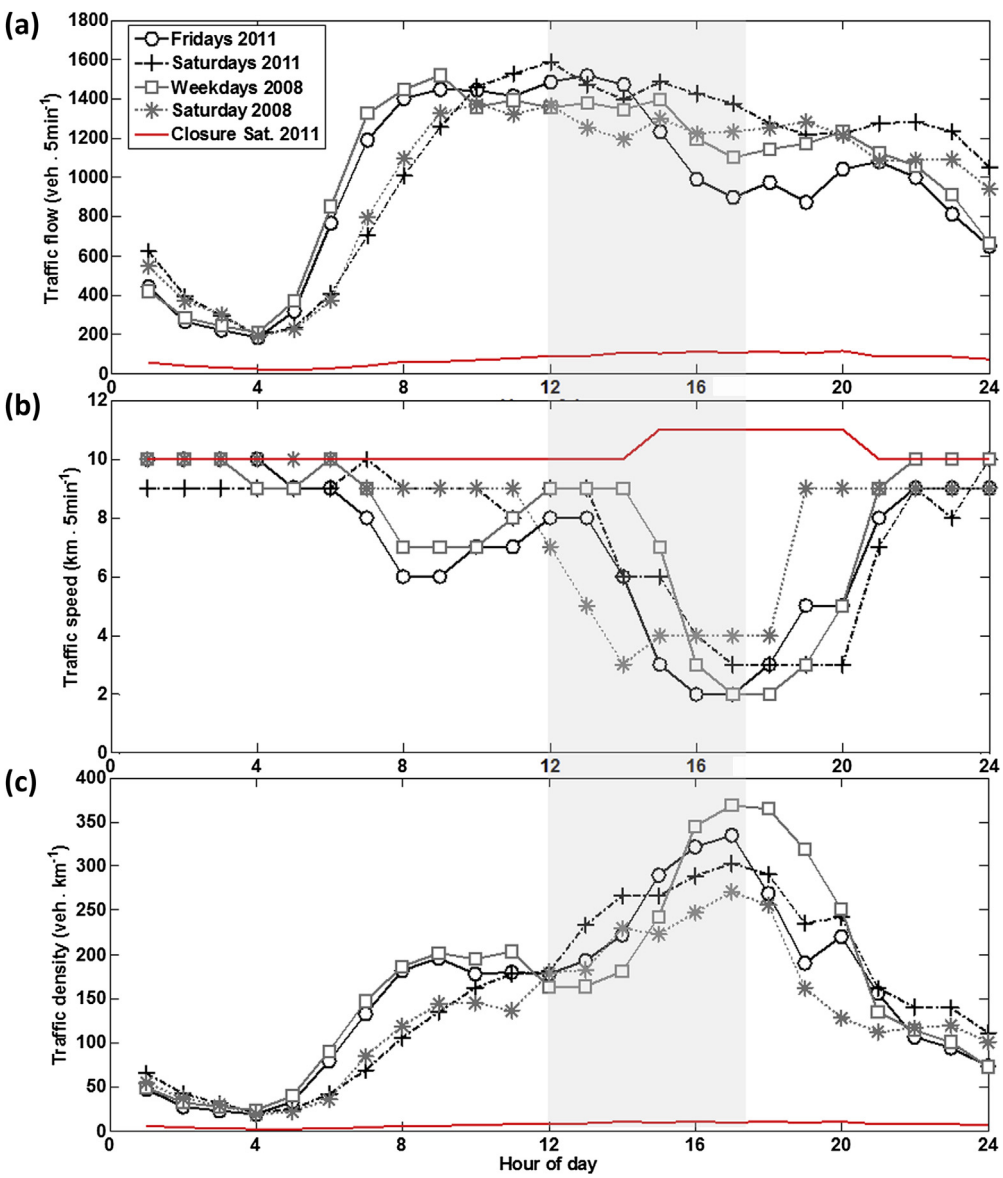

Fig. 2. Mean diurnal variations of traffic data obtained from Pico-station sensors on I-405 freeway: (a) traffic flows (vehicles 5 min^{-1}), (b) vehicle speeds (km 5 min^{-1}), and (c) traffic density (vehicles 5 km^{-1}). Black circles are for Fridays in 2011 data, black crosses for non-closure Saturdays in 2011, gray squares for weekdays in 2008, gray asterisks for Saturday in 2008, and red lines for I-405 closure Saturday. The gray-shaded area indicates the period during which measurements were made. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

In general, traffic emissions are likely to be enhanced in DTLA, which is intersected by five busy freeways and congested arterial streets (e.g., the I-10 freeway had 59% more vehicles per km and 38% greater traffic flows in the DTLA area than in WLA, Table 3). In WLA, Friday traffic flows in 2011 were comparable to those in 2008 for both the I-405 and I-10 freeways, but vehicle densities in 2011 were 16% lower on the I-405 and 39% higher on the I-10 freeway than those in 2008 due to differences in vehicle speeds. Traffic on Saturdays significantly increased in 2011 on both the I-405 and I-10 compared to 2008. Traffic flows and densities on the I-10 freeway in neighborhoods we studied in WLA on the I-405 closure day decreased by 38% and 65%, respectively, compared to normal 2011 Saturdays. In addition to freeways, significant traffic reductions on nearby arterial streets in WLA were observed during I-405 closure periods, although these reductions were not quantified.

3.3. Inter-community variations in pollutant concentrations in residential neighborhoods

Significant differences in traffic-related pollutant concentrations in residential neighborhoods were observed between BH, DTLA, and WLA (Fig. 3). The mean UFP concentrations in the neighborhoods of BH, DTLA, and WLA in 2008 were $3.3 \pm 2.2 \times 10^4$, $2.2 \pm 1.7 \times 10^4$, and $1.1 \pm 1.4 \times 10^4$ particles cm^{-3}, respectively. We note that standard deviations are large due to strong impacts from individual high-emitting vehicles (HEV). The Kolmogorov–Smirnov (KS) test after removing the local spikes from high emitting vehicles (HEV) encountered during the measurements, verified the inter-community variations in UFP concentrations are statistically significant ($p << 0.01$) (see SI S4 for the details of identification of HEV spikes).

Similarly, PB-PAH concentrations were highest in BH (16 ± 58 ng m^{-3}), followed by DTLA (8 ± 23 ng m^{-3}) and WLA (4 ± 10 ng m^{-3} in 2011) (Fig. 3b). Nitric oxide concentrations were comparable between DTLA (7.2 ± 10 ppb) and WLA (7.5 ± 6.8 ppb), but higher in BH (13.5 ± 12.7 ppb) (Fig. 3c). Although daytime NO is rapidly converted to NO_2 by reaction with ozone and peroxy radicals produced by the photooxidation of VOCs, O_3 concentrations during measurement periods varied little between sites (44 ppb in BH and DTLA, and 38 ppb and 41 ppb in WLA in 2008 and 2011, respectively). Thus, assuming O_3–NO–NO_2 photochemical processes are comparable in these areas (within spatial scales of ~20 km), higher NO concentrations in BH were likely to result from more emissions from denser traffic networks and a greater number of HEV in BH compared with WLA. Hu et al. (2012) attributed elevated concentrations of traffic-related pollutants in BH to relatively higher traffic density compared to other regions of the Southern California (Houston et al., 2004), combined with substantial numbers of HEV and a high density of stop signs and traffic lights with short block lengths. They also suggested the possibility of secondary formation of ultrafine particles through photochemical processes while an air mass travels from the coast through urbanized DTLA. We also attribute concentration differences between BH/DTLA and WLA to the difference in the magnitude of wind speeds (2.6 vs. 4.5 m s^{-1}) because stronger winds in near-coast WLA can enhance ventilation effects but wind speed difference cannot explain the differences between BH and DTLA. In the border between BH and DTLA, railroads are located along LA River, thus further study is needed to investigate if railroads are significant sources of particulate pollutants nearby neighborhoods and responsible to elevated pollution in BH than DTLA.

$PM_{2.5}$ did not show noticeable differences on an inter-community scale (Fig. 3d). Relatively homogeneous distributions of fine particles are likely due to a large fraction of $PM_{2.5}$ being formed secondarily through regional photochemical processes (Zheng et al., 2002). Thus, differences in direct emissions of fine particles from vehicular sources are relatively insignificant within these study areas (~20 km). Several near-roadway studies have shown small to insignificant elevations of $PM_{2.5}$ in the close vicinity of major roadways (Choi et al., 2012; Quiros et al., 2013; Zhu et al., 2002).

No significant differences in UFP and other pollutant concentrations were observed between weekdays and weekend days in WLA either in 2008 or 2011 (e.g., 1.1 vs. 1.2×10^4 in 2008 and 0.5 vs.

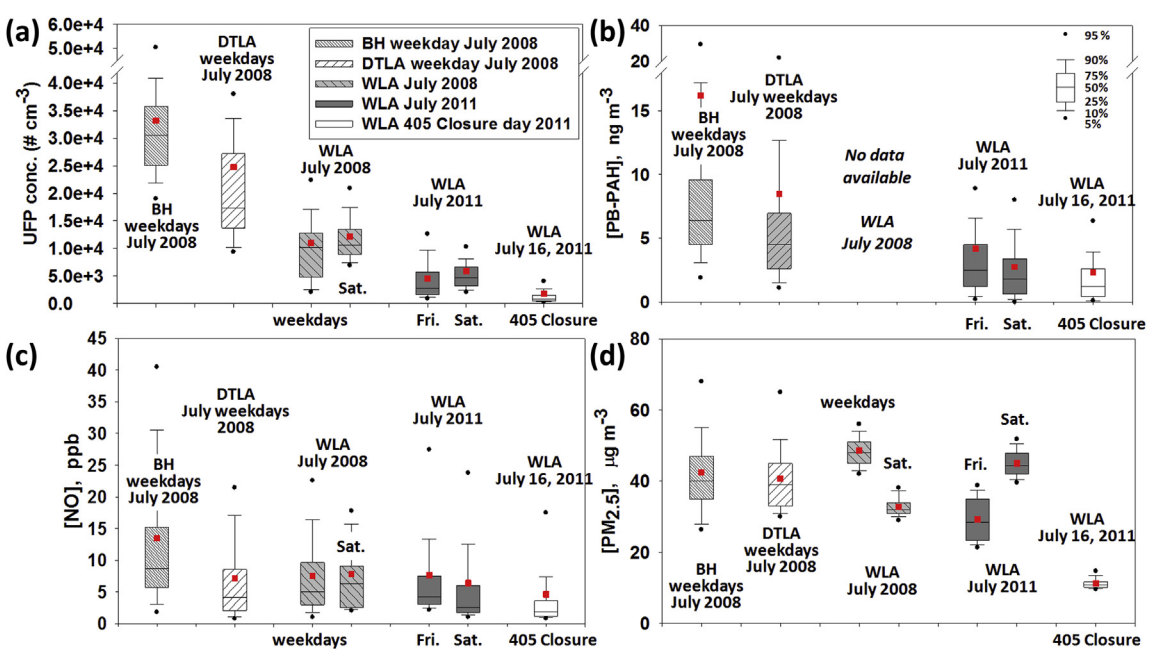

Fig. 3. Box plots of pollutant concentrations sampled in residential neighborhoods of Boyle Heights (BH; black fine slant lines), Downtown LA (DTLA; black coarse slant lines), West LA in 2008 (WLA; gray coarse slant lines), WLA in 2011 (simple gray boxes), and WLA on I-405 closure Saturday (white simple boxes): (a) Ultrafine particles (particles cm^{-3}), (b) PB-PAH (ng m^{-3}), (c) NO (ppb), and (d) particle mass less than 2.5 µm diameter ($PM_{2.5}$, µg m^{-3}). Red squares represent the mean values. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

0.6×10^4 particles cm^{-3} in 2011 for UFP, and 7.5 vs. 7.8 in 2008 and 7.6 vs. 6.4 ppb in 2011 for NO).

3.4. Intra-community variations in pollutant concentrations in residential neighborhoods

WLA residential neighborhoods (minor streets running through quiet residential areas $> \sim 150$ m from freeways) were divided into four sub-areas to investigate intra-community variations in traffic related pollutants: *A* (neighborhood southwest of the I-10/I-405 intersection), *B* (neighborhood north of I-10 and west of I-405 freeways), *C* (neighborhood downwind of both I-405 and I-10 freeways), and SMA (adjacent and downwind of Santa Monica Airport) (Fig. 1b).

Over all measurement periods, elevation of pollutant levels in area *C* was normally observed (Table 4 and Fig. 4) likely resulting from consistent southwesterlies during afternoons in WLA (mean wind direction $237 \pm 5^\circ$, Table 2). As an air mass travels from areas *A* and *B* to area *C*, it experiences emissions from the surface streets as well as freeways (e.g., areas *A* and *B* are influenced only by surface streets and partly by the I-10 freeway, whereas *C* has additional influence from the I-405 freeway). The increments of additional vehicle-related pollutants during the north-eastward air parcel transport were more readily observed in median concentrations than mean values because mean values are likely more strongly influenced by intermittent encounters with high-emitting vehicles.

The median UFP concentrations in area *C* were 39% and 262% higher, compared to *A* in 2008 (weekdays) and 2011 (Fridays), respectively. The mean UFP concentrations removing spikes due to HEV (SI S4) showed similar distributions; 40% and 158% higher in 2008 and 2011, respectively, compared to area *A*. The KS test showed the intra-community UFP variations were statistically significant at 99% confidence level ($p << 0.01$) in both 2008 and 2011. These trends in spatial distributions were consistently observed for other pollutants during weekdays (Table 4) although elevations in CO and $PM_{2.5}$ in area *C* were less significant compared to the other pollutants measured. CO and $PM_{2.5}$ are typically more regional pollutants, consistent with less noticeable intra-community variations.

Differences between areas *A* and *B* are less pronounced, although pollutant concentrations in area *B* appeared slightly higher than area *A* in general. Areas *A* and *B* are expected to experience similar pollutant contributions from the upwind areas except for the I-10 freeway that is likely a dominant contributor to pollutant level differences between areas *A* and *B*. Given that prevailing winds are somewhat parallel to the I-10 freeway (237° winds vs. 252° freeway orientation), pollutant plume transport could be limited compared to the case of perpendicular winds and the instantaneous variations in wind direction during sampling might dampen the intra-community differences between areas *A* and *B*.

3.5. Santa Monica Airport (SMA) impacts on locally elevated UFP concentrations

A striking feature in intra-community variations in pollutant levels was found in the neighborhood immediately downwind of the Santa Monica Airport (SMA), particularly for UFP concentrations. These results support Hu et al. (2009a)'s findings that UFP concentrations were about a factor of 10 higher than background levels 100 m downwind of SMA. The SMA residential area in this study covered 120–480 m downwind of the north end of the runway (Fig. 1b). Fig. 4a shows a remarkable increase in UFP levels in the SMA residential area with extremely wide variations. The mean UFP concentrations in the SMA residential area were 6.8 (weekdays in 2008), 1.5 (Fridays in 2011), 3.0 (Saturdays in 2011), and 1.3×10 particles cm^{-3} for the I-405 closure day in 2011. These values are factors of 7, 4, 5, and 37, respectively, higher compared to those in the nearby area *A* for the above sampling periods. In addition, the ratios of mean to median values of UFP concentrations ranged from 2 to 46 through the measurement periods, consistent with exceedingly high levels of UFP emitted intermittently during idling and takeoff of aircraft (Hu et al., 2009a). Each measurement period was coincident with 2–7 takeoff operations of jet and reciprocal engine aircraft (81% reciprocal engine and 19% jet, data from Santa Monica Airport Administration).

Even on the I-405 closure day, UFP concentrations around SMA were comparable to those of other sampling periods, whereas exceptionally low concentrations were observed in other residential areas due to significantly reduced traffic densities in WLA (Fig. 4a). Although measurements from a moving vehicle preclude quantification of UFP emissions from an individual aircraft,

Table 4
Median pollutant concentrations obtained in the sub-areas (*A*, *B*, and *C*) of residential neighborhoods in West LA (see text), and % increments of median values as an air mass travels *A* through *C*.

		Relative wind direction[a] (1σ) to		Median concentrations (% increase compared to *A*)		
		I-10	*I-405*	*A*	*B*	*C*
UFP	Weekdays in 2008	13° (14)	78° (13)	9165	10,600 (+16%)	12,700 (+39%)
	Fridays in 2011	14° (8)	78° (6)	1725	3040 (+76%)	6245 (+262%)
	Saturdays in 2011	13° (11)	78° (6)	4510	4410 (−2%)	4840 (+7%)
	Sunday (07/10/2011)	15° (13)	76° (8)	1440	3385 (+135%)	4350 (+202%)
NO	Weekdays in 2008			4.4	7.7 (+76%)	7.1 (+64%)
	Fridays in 2011			3.7	4.3 (+17%)	4.9 (+34%)
	Saturdays in 2011			2.5	2.3 (−9%)	3.0 (+20%)
	Sunday (07/10/2011)			1.4	2.5 (+82%)	2.0 (+48%)
PAH	Fridays in 2011			2.1	2.8 (+33%)	3.5 (+67%)
	Saturdays in 2011			1.4	1.4 (0%)	2.6 (+86%)
	Sunday (07/10/2011)			1.6	1.0 (−38%)	1.8 (+13%)
CO	Fridays in 2011			0.49	0.50 (+3%)	0.56 (+14%)
	Saturdays in 2011			0.53	0.52 (+-3%)	0.58 (+9%)
	Sunday (07/10/2011)			0.43	0.45 (+4%)	0.50 (+16%)
$PM_{2.5}$	Weekdays in 2008			45	52 (+16%)	50 (+11%)
	Fridays in 2011			27	30 (+11%)	33 (+22%)
	Saturdays in 2011			44	45 (+2%)	45 (+2%)
	Sunday (07/10/2011)			27	33 (+24%)	30 (+13%)

[a] 90° is normal to the freeway orientation and 0° is parallel to the freeway.

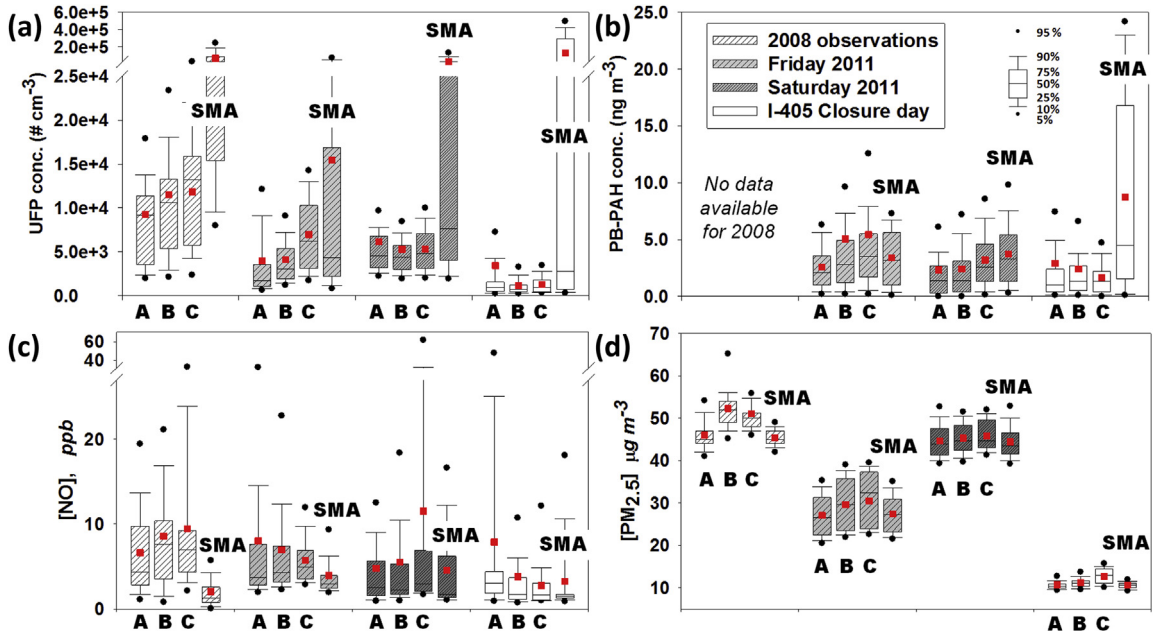

Fig. 4. Box plots of pollutant concentrations observed in residential sub-areas (*A*, *B*, *C*, and *SMA*) of WLA for weekdays in 2008 (coarse slant lines in white boxes), Fridays in 2011 (fine slant lines in light gray boxes), non-closure Saturdays in 2011 (fine slant lines in dark gray boxes), and I-405 closure Saturday in 2011 (simple white boxes): (a) UFP, (b) PB-PAH, (c) NO, and (d) $PM_{2.5}$. Red squares represent the mean values. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

qualitatively, the highest UFP peak concentrations were associated with mid-size jet takeoffs, followed by small jets and smaller reciprocal-engine aircraft (not shown). These results are consistent with observations by Hu et al. (2009a) and their calculation of fuel consumption rates for aircraft at SMA.

PB-PAH were found by Hu et al. (2009a) to be associated with jet takeoffs but not with other aircraft operations such as idling, descents or takeoffs by reciprocal-engine aircrafts. As a result, PB-PAH are expected to be elevated around SMA more sporadically than UFP, especially given that measurements in the SMA neighborhood were short (5–10 min twice per day). PB-PAH were elevated little on the Fridays and Saturdays in 2011 (no PB-PAH data are available for 2008) and highly elevated on the closure day (Fig. 4b). This is likely explained by the jet takeoffs in 2011 that coincided with our sampling on the closure day; such events were not coincident on the other days. Similar but less intense trends were observed for CO concentrations (not shown). Concentrations of other pollutants in the SMA neighborhood, such as NO and the more regional pollutant, $PM_{2.5}$, were comparable to or lower than those in sub-area *A*.

3.6. Comparisons of pollutant concentrations on arterial roadways

Pollutant concentrations on arterial roadways were compared between BH, DTLA, and WLA in 2008 and for WLA in 2011 (Fig. 5). With the exception of $PM_{2.5}$, which as discussed is a more regional pollutant, on-road pollutant concentrations were highest in BH, followed by DTLA and then WLA, as expected. Median UFP concentrations were 4.0, 3.0, 1.7, and 0.8 $\times$ 10^4 particles cm^{-3} in BH, DTLA, WLA in 2008, and WLA in 2011, respectively. The KS test for data with transient spikes due to HEV removed also showed the differences in UFP distributions by location were statistically significant ($p << 0.01$). UFP concentrations on the arterial roadway adjacent to SMA (S. Bundy Dr.) showed the highest extreme and mean values with exceptionally wide variations, consistent with aircraft activities (discussed above, Fig. 5a) and with Hu et al. (2009a).

Relative PB-PAH concentrations between neighborhood arterial roadways were similar to UFP, with median concentrations of 15, 11, 6, 4, and 3 ng m^{-3} in BH and DTLA in 2008, and Fridays, Saturdays, and the closure Saturday in WLA 2011, respectively. PB-PAH appeared to be more strongly influenced by HEV than any of the other pollutants on arterial roadways as can be seen from the upper end of the interquartile bars in Fig. 5b. The ratios of 95% (90%) quantile value of PB-PAH concentration distributions to median are 11 (6.0), 12 (6.5), and 15 (8.9) in BH, DTLA, and WLA 2011, respectively, which are larger than those of UFP (i.e., 3 (1.9), 4 (2.4), and 5 (3.3) in BH, DTLA, and WLA 2011, respectively). NO concentrations were higher in BH and DTLA (28–30 ppb median) compared to WLA (14–18 ppb median) (Fig. 5c). Consistent with our residential neighborhood measurements, no significant spatial and temporal differences in $PM_{2.5}$ were found during the measurement periods.

3.7. Impact of high emitting vehicles on observed UFP concentrations

Percent of time HEV were encountered and total UFP from HEV were calculated according to the approach of Hu et al. (2012) (SI S5), but in the present study the threshold values were determined statistically instead of using arbitrary threshold values as in Hu et al. (2012). Threshold values were defined as the baseline-subtracted concentrations equal to 3σ for HEV-spike-removed concentration variations (SI S4). As summarized in Table 5, the percent of time HEV were encountered (AN_i) was slightly higher on arterial roadways (9–13%) than in residential areas (6–11%). However, we note that threshold values for residential areas were just 37–57% of the threshold values for arterial roadways, thus a wider range of vehicles were classified as HEV in neighborhoods. Similarly, threshold values in BH and DTLA were more than double WLA, both on arterial roadways and in residential areas. These differences result from relatively lower baseline concentrations and smaller UFP concentration variations in WLA; characterizing spikes in UFP concentrations due to HEV is specifically influenced by the background concentrations at a given site.

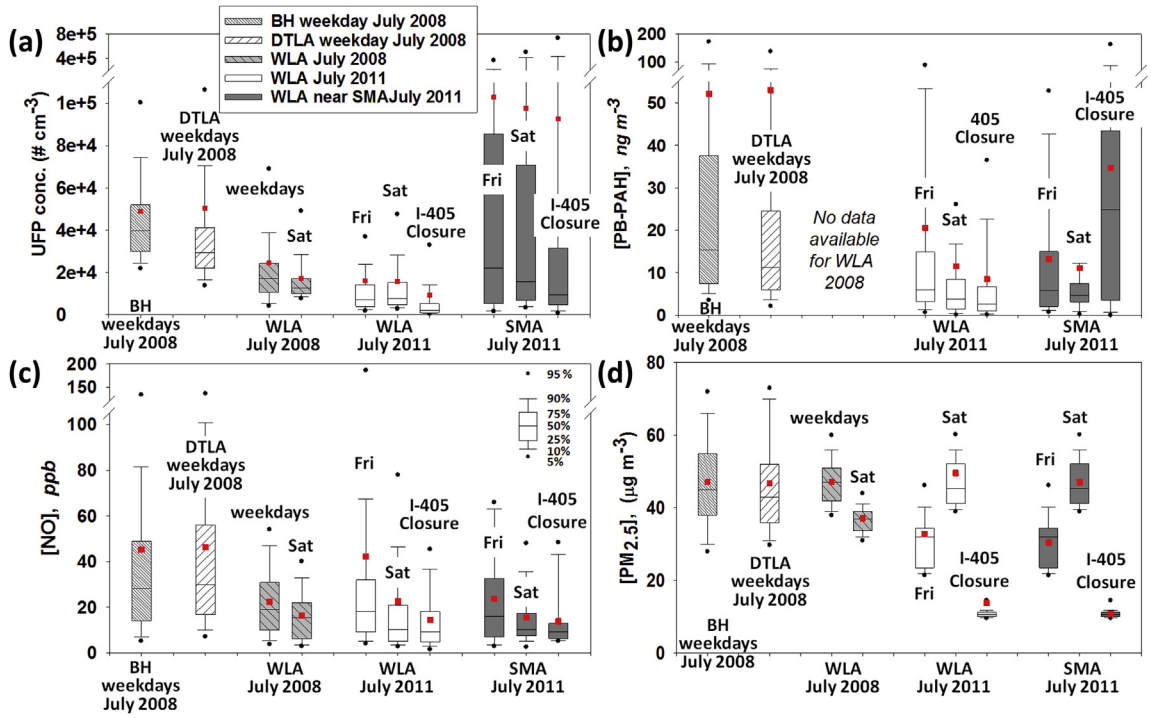

Fig. 5. Box plots of pollutant concentrations measured on arterial roadways in BH in 2008 (fine slant lines in white boxes), DTLA in 2008 (coarse slant lines in white boxes), WLA in 2008 (coarse slant lines in gray boxes), WLA in 2011 (simple white boxes), and WLA adjacent to SMA in 2011 (simple dark gray boxes): (a) UFP, (b) PB-PAH, (c) NO, and (d) $PM_{2.5}$. Red squares represent the mean values. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

If we were to apply the single threshold value chosen for BH by Hu et al. (2012) to DTLA and WLA, AN_i would be smaller than the values above. Importantly, a single value makes clearer the fact that there are substantially more HEV emitting above a given level in BH than in WLA (by about a factor of two in 2008). Specifically, for residential neighborhoods: AN_i decreases from 6% to 4%, 11% to 2% and 8% to 0.5% for DTLA 2008, WLA 2008 and 2011, respectively (8% for BH). For arterial roadways, AN_i decreases from 13% to 10%, 12% to 5%, and 11% to 2% for DTLA 2008, WLA 2008 and 2011, respectively (9% for BH). Nevertheless, using a single common value for all sites can over- or under-estimate HEV effects significantly depending on differences in baseline concentrations and fluctuations at the various sites.

We note that the magnitudes of transient UFP concentration spikes from HEV vary from site to site. For example, the mean baseline-subtracted UFP concentrations for HEV spikes were 4.2, 3.9, 2.3, and 1.0×10^4 particles cm^{-3} in the residential areas of BH, DTLA, WLA 2008, and WLA 2011, respectively, implying HEV in BH and DTLA tend to emit more UFP compared to HEV in WLA (Table 5). This is an expected result, given differences in composition and age of the vehicle fleets between the low-income communities of DTLA and BH vs. the affluent community of WLA (SI S1). Similar to the findings of Hu et al. (2012) in BH in 2008, a relatively large fraction of total UFP resulted from the small fraction of HEV for all neighborhoods. For example, the 9–13% of AN_i accounted for 29–55% of total UFP observed on arterial roadways, and the 6–11% of AN_i in residential areas contributed to 17–29% of total UFP concentrations observed in those areas. These large HEV impacts on observed total UFP concentrations were more prominent in the relatively cleaner WLA area both on arterials and in residential neighborhoods due to lower baseline concentrations of UFP.

Bishop et al. (2012) reported in a recent tunnel study that less than 1% HEV of the fleet contributed to more than a third of total CO and HC emissions. Our study shows UFP air quality can also be significantly improved if emissions of HEV can be reduced through retrofit and maintenance program and/or early-retirement of HEV, although HEV contributions to total UFP emissions are less dominant compared to the cases of CO and HC. The most appropriate

Table 5
HEV contributions to UFP concentrations on arterial roadways and residential areas in BH (2008), DTLA (2008), and WLA (2008 and 2011).

Sites (data#)	Mean (all data)	STD (all data)	Mean (HEV removed)	STD (HEV removed)	Mean threshold for HEV	% of time HEV encountered	% of total UFP from HEV[a]	Mean ΔUFP of spikes
On-arterial roads								
BH (1192)	49,100	55,900	39,800	13,700	69,800	9%	26%	1.0×10^5
DTLA (3051)	50,500	181,700	29,400	11,900	53,300	13%	49%	1.6×10^5
WLA '08 (1158)	24,500	38,800	16,100	8300	29,700	12%	42%	6.9×10^4
WLA '11 (2116)	16,200	58,300	8300	6400	17,600	11%	55%	6.3×10^4
Residential area								
BH (766)	33,200	22,200	29,600	6300	40,100	8%	18%	4.2×10^4
DTLA (1152)	21,600	17,100	19,100	7900	30,500	6%	17%	3.9×10^4
WLA '08 (1451)	12,300	14,400	9800	6400	14,400	11%	29%	2.3×10^4
WLA '11 (3592)	4500	5200	3600	2900	6500	8%	26%	1.0×10^4

[a] This value is based on a statistical definition of HEV such that lower emission vehicles qualify as HEV in cleaner areas. See text and Supplementary information.

way to identify UFP HEV is to measure UFP emissions directly from individual vehicles under standardized conditions, such as on a laboratory dynamometer or in dedicated chase car studies. Clearly, further research is needed to better characterize the impacts of HEV in various environments.

3.8. UFP emission reductions over time

Significant decreases in UFP concentrations both in residential neighborhoods and on arterial roadways in WLA were observed between 2008 and 2011 (Figs. 3–5). The median and mean concentrations of UFP in 2011 were reduced approximately 70% and 60%, respectively, in residential neighborhoods on weekdays compared to observations in 2008 (from 1.1 to 0.3 $\times$ 10^4 particles cm^{-3} for median and from 1.2 to 0.5 $\times$ 10^4 particles cm^{-3} for mean values). Median and mean UFP Saturday concentrations decreased ~55% and ~50%, respectively, from 2008 to 2011 in WLA neighborhoods. Similar reductions of median UFP concentrations between 2008 and 2011 were also observed on arterial roadways (~60% and ~40% on weekdays and Saturdays, respectively), although the declines in the mean UFP concentrations were less pronounced (~35% for weekdays and ~10% for Saturdays). This relatively smaller reduction in the mean concentrations resulted from more frequent encounters with HEV during sampling periods in 2011 (the standard deviation was about a factor of two larger in 2011). Given that both local and regional meteorological conditions were similar between the sampling periods in 2008 and 2011, and traffic densities on the I-10 and I-405 freeways were generally increased in 2011 (Table 3), we believe that reduced UFP concentrations in WLA resulted from reductions in emissions of UFP from on-road vehicles. We note that with the exception of the I-405 closure day, lower UFP have not resulted in noticeably lower $PM_{2.5}$ concentrations on either arterials or in neighborhoods. As noted earlier, $PM_{2.5}$ is a more regional pollutant that is typically minimally affected by emissions, at least in the immediate vicinity of roadways (e.g., Zhu et al., 2002). A recent $PM_{2.5}$ inventory for the West Side of Los Angeles is not available, however, inventories for other locations in the South Coast Air Basin from 2004 to 2006 indicate that direct emissions from roadway sources (including diesel, gasoline and roadway particulate) are less than roughly a quarter of the sources of $PM_{2.5}$, and roughly another quarter is formed from mobile sources via secondary pathways (Pham et al., 2008). Thus, while lower UFP throughout the area should produce somewhat lower $PM_{2.5}$, the linkage is fairly weak, and we may not have sufficient days of data to elucidate this relationship. A slight decline in median NO concentrations was found in residential neighborhoods and on arterial roadways between 2008 and 2011 but mean NO did not follow this declining trend.

Several recent studies have also reported significant decreases in vehicular UFP emissions. Quiros et al. (2013) reported a 70% reduction in UFP concentrations from measurements in the vicinity of the I-405 freeway during the same period of the present study in 2011, compared to 2001 measurements (Zhu et al., 2002). Choi et al. (2012) qualitatively reported reduced UPF peak concentrations in freeway plumes under pre-sunrise stable atmospheric conditions in the South Coast Air Basin in 2011 compared to observations for pre-sunrise hours in 2008 (Hu et al., 2009b) and nighttime measurements (22:30–05:00) in 2005 (Zhu et al., 2006) in the WLA area. Near-roadway studies in other geographical areas have reported similar findings, including a distinct declining trend in nucleation mode particles in Rochester, New York from 2002–2005 to 2005–2007 (Wang et al., 2011); a 21% decrease in UFP number concentrations (<50 nm) over five years (2006–2010) in Toronto, Canada (Sabaliauskas et al., 2012); and a 27% reduction in UFP (particularly for particles <30 nm) between 2002–2004 and 2005–2007 in Copenhagen, Denmark (Wahlin, 2009). One of the major contributors to these reductions was stringent regulation of sulfur content in gasoline and diesel fuels (Wahlin, 2009; Wang et al., 2011).

In California, Quiros et al. (2013) attributed the conspicuous reductions in UFP emissions over time to a combination of several factors, including retirement of older vehicles, adoption of more stringent regulations of particle emissions for heavy duty diesel vehicles and fuel composition (CARB, 2004, 2008; Ristovski et al., 2006), and increased use of smaller and more fuel-efficient engines. We also note that as of 2011 in California, statewide net taxable gasoline and diesel sales have declined ~8% (~1.5%/yr) and ~15% (−4%/yr) since 2006 and 2007, respectively (BOE, 2012). In addition, fleet fuel economy has significantly improved in the United States (e.g., from 19.9 MPG in 2004 to 23.2 MPG in 2010 (Schoenberger, 2011)).

3.9. Air quality benefits of traffic emission reductions

A valuable feature of the present study is that the 36-h I-405 closure event provided an excellent opportunity to investigate the air-quality benefits of traffic emission reductions on a larger neighborhood scale (several kilometers) not just at near-roadway scales (several hundred meters). During the I-405 closure Saturday, more than 95% and 65% reductions in traffic densities were observed on the I-405 and I-10 freeways, respectively, compared to the preceding and following non-closure Saturdays. Anecdotally, substantial drops in vehicle numbers on nearby arterial roads throughout the area were also observed during the closure Saturday. Quiros et al. (2013) reported a 20% decrease in traffic flows on the closure day on Sepulveda Blvd., a surface street running parallel to, and near, the I-405 freeway, concluding there was no spillover of freeway traffic onto alternative surface streets. Evidence indicates voluntary restraints on vehicle-use were larger than 20% and occurred extensively throughout WLA in response to the intensive warnings of potential chaotic traffic congestion, i.e. "Carmageddon".

Dramatic decreases in both particle number and mass concentrations were observed on the closure day accompanied by relatively smaller reductions in gaseous pollutants and PB-PAH (Table 6 and Figs. 3–5). The median UFP number and $PM_{2.5}$ concentrations were 800 particles cm^{-3} and 11 $\mu g\ m^{-3}$, respectively, in residential

Table 6
Median concentrations of pollutants measured in residential neighborhoods and on arterial roadways of WLA in 2011 for I-405 closure Saturday and non-closure Saturdays, and concentration reductions (%) on closure Saturday compared to non-closure Saturdays.

Median conc. and reduction rates		UFP (# cm^{-3})	$PM_{2.5}$ ($\mu g\ m^{-3}$)	PB-PAH (ng m^{-3})	NO (ppb)	CO (ppm)
Residential neighborhoods	Non-closure Saturdays	4720	44	1.8	2.5	0.53
	Closure Saturday	800	11	1.2	1.9	0.39
	% Reduction	−70%	−75%	−33%	−25%	−26%
Arterial roadways	Non-closure Saturdays	7660	48	3.8	10.3	0.64
	Closure Saturday	2200	12	2.6	9.2	0.49
	% Reduction	−71%	−74%	−32%	−10%	−25%

neighborhoods of WLA on the I-405 closure Saturday, only 30% and 25% of non-closure Saturday UFP and $PM_{2.5}$ levels, respectively. Even on arterial roadways, similar reductions in UPF number and $PM_{2.5}$ concentrations were observed throughout the WLA area. We note that $PM_{2.5}$ on non-closure Saturdays in 2011 was higher than that on Fridays, the inverse of 2008 observations. To validate the data quality of $PM_{2.5}$, we have compared $PM_{2.5}$ with $PM_{0.5}$ obtained from FMPS size distribution data with a density of 1.2 g cm^{-3}. Mean $PM_{2.5}$ and $PM_{0.5}$ in the residential areas showed excellent agreement (SI S6), and hence we conclude the relative variations in $PM_{2.5}$ during the measurement campaign are reliable, supporting a significant reduction in $PM_{2.5}$ during the I-405 closure period. Simultaneous measurements of UFP and $PM_{2.5}$ at a fixed site on Constitution Ave. (located 2 km north of our WLA route) also found 84% and 60% reductions in daily median UFP and $PM_{2.5}$ concentrations, respectively (Quiros et al., 2013). Given the emissions inventories for $PM_{2.5}$ (Pham et al., 2008) discussed earlier, such large reductions in $PM_{2.5}$ are surprising. Part of the explanation may lie in the larger contribution of roadway sources due to smaller contributions of other sources such as ammonium salts in this area of Los Angeles, but a complete explanation is unclear.

Gaseous pollutants and PB-PAH also showed modest drops during the closure event both in residential neighborhoods and on arterial roadways (~25%–~33%) with the exception of NO on arterial roadways (only ~10% reduction).

Our findings from the closure of the I-405 freeway, and the trends we have observed in pollutant concentrations over several years, provide evidence that reductions of vehicle emissions through practical and achievable strategies can improve local and regional air quality, particularly for particulate matter in urban areas. Clearly, the atmospheric responses of traffic-related particulate pollutants to the dramatic traffic reductions resulting from the I-405 closure were immediate and conspicuous. With the assumption that observed traffic on the I-10 freeway and Sepulveda Blvd. on the I-405 closure Saturday represented overall traffic patterns throughout the WLA areas, a 30–70% reduction in traffic flows (not quantitatively measured) resulted in approximately a 70% decrease in UFP and $PM_{2.5}$ concentrations both in the neighborhoods and on major arterial roads. We note, again, the 20% traffic reduction on Sepulveda during the closure was likely less than the general traffic reduction in WLA because Sepulveda is the primary alternate route to the closed section of freeway. Although the elevation of $PM_{2.5}$ directly from major roadways is insignificant compared to UFP, PB-PAH, and NO (Choi et al., 2012; Quiros et al., 2013), area-wide reductions in traffic densities can decrease direct $PM_{2.5}$ emissions as well as its precursors for secondary production, achieving improvements in $PM_{2.5}$ levels.

We also note that heavy-duty diesel trucks (HDDT) on the I-10 freeway virtually disappeared during the I-405 closure Saturday (2 trucks/5 min^{-1}, down to 10% of non-closure Saturday truck flows). Despite large variations in particle number emission factor (PNEF) estimated by a number of previous studies, HDDT appear to emit about 10–20 times more UFP than passenger cars (Kumar et al., 2011), which implies a 50–100% increase in PNEF with 5% of the fleet being HDDT compared to a non-HDDT fleet.

Consequently, the present case study makes clear the potential benefits for public health of achieving significant vehicle emission reductions through strategies such as HDDT retrofits, and transition to electric vehicles and alternative fuels such as natural gas. This study also showed the significant impact of HEV on total UFP concentrations, and hence, retrofits or earlier retirement of high-emitting vehicles can help improve urban air quality. The findings of this study should provide a useful data-set for cost-benefit analyses of such strategies.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.atmosenv.2013.07.043.

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